Northeast Fisheries Science Center Reference Document 18-11

65th Northeast Regional Stock Assessment Workshop (65th SAW) Assessment Report

by Northeast Fisheries Science Center

October 2018

Northeast Fisheries Science Center Reference Document 18-11

65th Northeast Regional Stock Assessment Workshop (65th SAW) Assessment Report

by Northeast Fisheries Science Center

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

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts

October 2018

Northeast Fisheries Science Center Reference Documents

This series is a secondary scientific series designed to assure the long-term documentation and to enable the timely transmission of research results by Center and/or non-Center researchers, where such results bear upon the research mission of the Center (see the outside back cover for the mission statement). These documents receive internal scientific review, and most receive copy editing. The National Marine Fisheries Service does not endorse any proprietary material, process, or product mentioned in these documents.

Editorial Treatment: To distribute this report quickly, it has not undergone the normal technical and copy editing by the Northeast Fisheries Science Center's (NEFSC's) Editorial Office as have most other issues in the NOAA Center Reference Document NMFS-NE series. Other than the four covers and first two preliminary pages, all writing and editing have been performed by – and all credit for such writing and editing rightfully belongs to – those so listed on the title page.

If you do not have Internet access, you may obtain a paper copy of a document by contacting the senior Center author of the desired document. Refer to the title page of the document for the senior Center author's name and mailing address. If there is no Center author, or if there is corporate (*i.e.*, non-individualized) authorship, then contact the Center's Woods Hole Laboratory Library (166 Water St., Woods Hole, MA 02543-1026).

Information Quality Act Compliance: In accordance with section 515 of Public Law 106-554, the Northeast Fisheries Science Center completed both technical and policy reviews for this report. These predissemination reviews are on file at the NEFSC Editorial Office.

This document may be cited as:

Northeast Fisheries Science Center (NEFSC). 2018. 65th Northeast Regional Stock Assessment Workshop (65th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 18-11; 659 p. Available from: http://www.nefsc.noaa.gov/publications/

TABLE OF CONTENTS

B. STOCK ASSESSMENT OF ATLANTIC HERRING FOR 2018

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 65th SARC was convened in Woods Hole at the Northeast Fisheries Science Center, June 26-29, 2018 to review benchmark stock assessments of Sea scallop and Atlantic herring. CIE reviews for SARC65 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-65 Review Panel reports (available at http://www.nefsc.noaa.gov/nefsc/saw/ under the heading "SARC-65 Panelist Reports").

SARC 65 concluded that the sea scallop stock is neither overfished nor did it experience overfishing in 2017. The Panel also concluded that the SAW WG had reasonably and satisfactorily completed all tasks specified in the ToRs. A gonad-based

SSB and related reference points were developed and presented. But the panel recommended that in the interim meat weight-based reference points continue to be used. The method of using gonad weight to calculate spawning stock size looks promising, but additional work is needed to fully develop the approach. The panel also recommended determining whether to continue with multiple survey approaches, as is currently done, or to select a single survey methodology for the future.

SARC 65 concluded that the Atlantic herring stock is neither overfished nor did it experience overfishing in 2017. The Panel also concluded that the SAW WG had reasonably and satisfactorily completed all tasks specified in the ToRs. The key changes in the ASAP model used from the last assessment were in assumptions about M and selectivity, in the introduction of new index time series (including an acoustic survey series for the first time). The sensitivity analyses successfully explained the observed assessment scale difference from 2015. The recruitment estimates from the most recent five years were among the lowest in the time series. This suggests that the short-to-medium term prognosis for the stock is likely to be relatively poor.

Table 1. 65th Stock Assessment Review Committee Panel.

SARC Chairman (NEFMC SSC):

Dr. Patrick Sullivan
Cornell University
Email: pjs31@cornell.edu

SARC Panelists (CIE):

Dr. Cathy Dichmont
Cathy Dichmont Consulting
69 Headsail Drive
Banksia Beach, Qld, 4507
Australia
Email: Cathydichmont@gmail.com

Dr. Geoff Tingley
Gingerfish Ltd (Fisheries Science and Management Consultancy)
New Zealand
Email: fishinnz@hotmail.co.uk

Dr. Coby Needle
MSS Marine Laboratory
Aberdeen, UK
Email: needlec@marlab.ac.uk

Table 2. 65th Stock Assessment Workshop/Stock Assessment Review Committee (SAW/SARC) Benchmark Stock Assessment for A. Sea scallop and B. Herring

June 26-29, 2018

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

AGENDA* (version: 6/22/2018)

TOPIC	PRESENTER(S)	RAPPORTEUR

Tuesday, June 26

10 – 10:45 AM Welcome/Description of Review Process Introductions/Agenda Conduct of Meeting	**James Weinberg**, SAW Chair **Patrick Sullivan,** SARC Chair	
10:45 – 12:45 PM	Assessment Presentation (A. Scallops) **Dvora Hart, Jui-Han Chang, Jonathon Peros**	**Alicia Miller**
12:45 – 1:45 PM	Lunch	
1:45 – 3:45 PM	Assesssment Presentation (A. Scallops) **Dvora Hart, Jui-Han Chang**	**Toni Chute**
3:45 – 4 PM	Break	
4 – 5:45 PM	SARC Discussion w/ Presenters (A. Scallops) **Patrick Sullivan,** SARC Chair	**Toni Chute**
5:45 – 6 PM	Public Comments	

Wednesday, June 27

8:30 – 10:30 AM	Assessment Presentation (B. Herring) **Jon Deroba**	**Dan Hennen**
10:30 – 10:45 AM	Break	
10:45 – 12:30 PM	Assessment Presentation (B. Herring) **Jon Deroba**	**Dan Hennen**
12:30 – 1:30 PM	Lunch	
1:30 – 3:30 PM	SARC Discussion w/presenters (B. Herring) **Patrick Sullivan,** SARC Chair	**Brian Linton**
3:30 – 3:45 PM	Public Comments	

3:45 -4 PM Break

4 – 6 PM Revisit with Presenters (A. Scallops)
Patrick Sullivan, SARC Chair **Brian Linton**

7 PM **(**Social Gathering**)**

TOPIC	PRESENTER(S)	RAPPORTEUR

Thursday, June 28

8:30 – 10:30 Revisit with Presenters (B. Herring)
Patrick Sullivan, SARC Chair **Tony Wood**

10:30 – 10:45 Break

10:45 – 12:15 Review/Edit Assessment Summary Report (A. Scallops)
Patrick Sullivan, SARC Chair **Tony Wood**

12:15 – 1:15 PM Lunch

1:15 – 2:45 PM (cont.) Edit Assessment Summary Report (A. Scallops)
Patrick Sullivan, SARC Chair **Tony Wood**

2:45 – 3 PM Break

3 – 6 PM Review/edit Assessment Summary Report (B. Herring)
Patrick Sullivan, SARC Chair **Tony Wood**

Friday, June 29

9:00 AM – 5:00 PM SARC Report writing

*All times are approximate, and may be changed at the discretion of the SARC chair. The meeting is open to the public; however, during the SARC Report Writing sessions we ask that the public refrain from engaging in discussion with the SARC.

Table 3. 65th SAW/SARC, List of Attendees, June 26-29, 2018

NAME	AFFILIATION	EMAIL
Patrick Sullivan	Cornell University	pjs31@cornell.edu
Cathy Dichmont	Cathy Dichmont Consulting	cathy.dichmont@gmail.com
Coby Needle	Marine Laboratory, Aberdeen, Scotland	needlec@marlab.ac.uk
Geoff Tingley	Gingerfish Ltd	fishinnz@hotmail.co.uk
Russell Brown	NEFSC	russell.brown@noaa.gov
Jim Weinberg	NEFSC	james.weinberg@noaa.gov
Jonathon Peros	NEFMC	jperos@nefmc.org
Deirdre Boelke	NEFMC	dboelke@nefmc.org
Dvora Hart	NEFSC	deborah.hart@noaa.gov
Jui-Han Chang	NEFSC	jui-han.chang@noaa.gov
Jon Deroba	NEFSC	Jonathan Deroba@noaa.gov
Burton Shank	NEFSC	burton.shank@noaa.gov
Gary Shepherd	NEFSC	gary.shepherd@noaa.gov
Bill DuPaul	VIMS/College of William and Mary	dupaul@vims.edu
Kevin Stokesbury	SMAST	kstokesbury@umassd.edu
Liese Siemann	Coonamessett Farm Foundation	lsiemann@cfarm.org
Sarah Gaichas	NEFSC	sarah.gaichas@noaa.gov
Chris Legault	NEFSC	chris.legault@noaa.gov
Alicia Miller	NEFSC	alicia.miller@noaa.gov
Toni Chute	NEFSC	toni.chute@noaa.gov
Dan Hennen	NEFSC	daniel.hennen@noaa.gov
Brian Linton	NEFSC	brian.linton@noaa.gov
Tony Wood	NEFSC	anthony.wood@noaa.gov
Charles Adams	NEFSC	charles.adams@noaa.gov
Sam Asci	NEFMC	sasci@nefmc.org
Steve Cadrin	SMAST	scadrin@umassd.edu
Matt Cieri	Maine DMR	matthew.cieri@maine.gov
Kiersten Curti	NEFSC	kiersten.curti@noaa.gov
Emmanuel Dovlo	URI/CRC	ekdovlo@gmail.com
Liz Duskey	Cornell University	epd48@cornell.edu
Rachel Feeney	NEFMC	rfeeney@nefmc.org
Travis Ford	GARFO	travis.ford@noaa.gov
Aris-Aja Horsey	Partnership Ed. Program	arisajahorsey@gmail.com
Mike Jech	NEFSC	michael.jech@noaa.gov
Peter Kendall	NEFMC	peter.kendall@comcast.net
George Lapointe	Fisheries Survival Fund	georgelapointe@gmail.com
George Manning	Partnership Ed. Program	gmanning@eagles.ncu.edu
Drew Minkiewicz	Fisheries Survival Fund	aminkiewicz@kelleydrye.com
Jenna Munden	Herring Science Council	herringsciencecoordinator@gmail.com
Tom Nies	NEFMC	tnies@nefmc.org
Cate O'Keefe	Mass DMF	Catherine.O'Keefe@state.ma.us
Mike Palmer	NEFSC	michael.palmer@noaa.gov
David Pierce	Mass DMF	David.Pierce@state.ma.us
Rabindra Singh	DFO Canada	rabindra.singh@dfo-mpo.gc.ca
Mike Sissenwine	NEFMC	m.sissenwine@gmail.com
Mary Beth Tooley	O'Hara Corp.	mbtooley@oharcorporation.com
Brooke Wright	SMAST	brooke.wright@umassd.edu

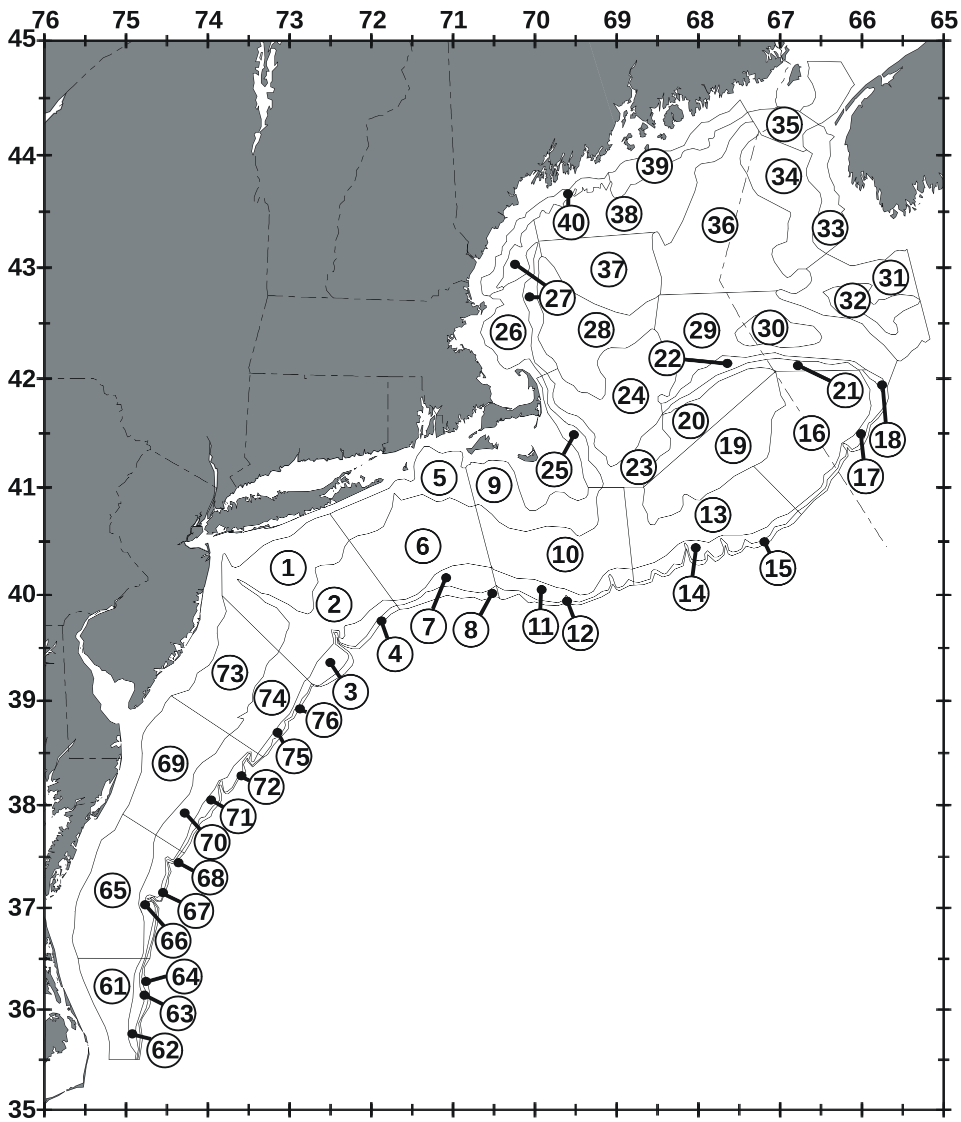


Figure 1. Offshore depth strata that have been sampled during Northeast Fisheries Science Center bottom trawl research surveys. Some of these may not be sampled presently.

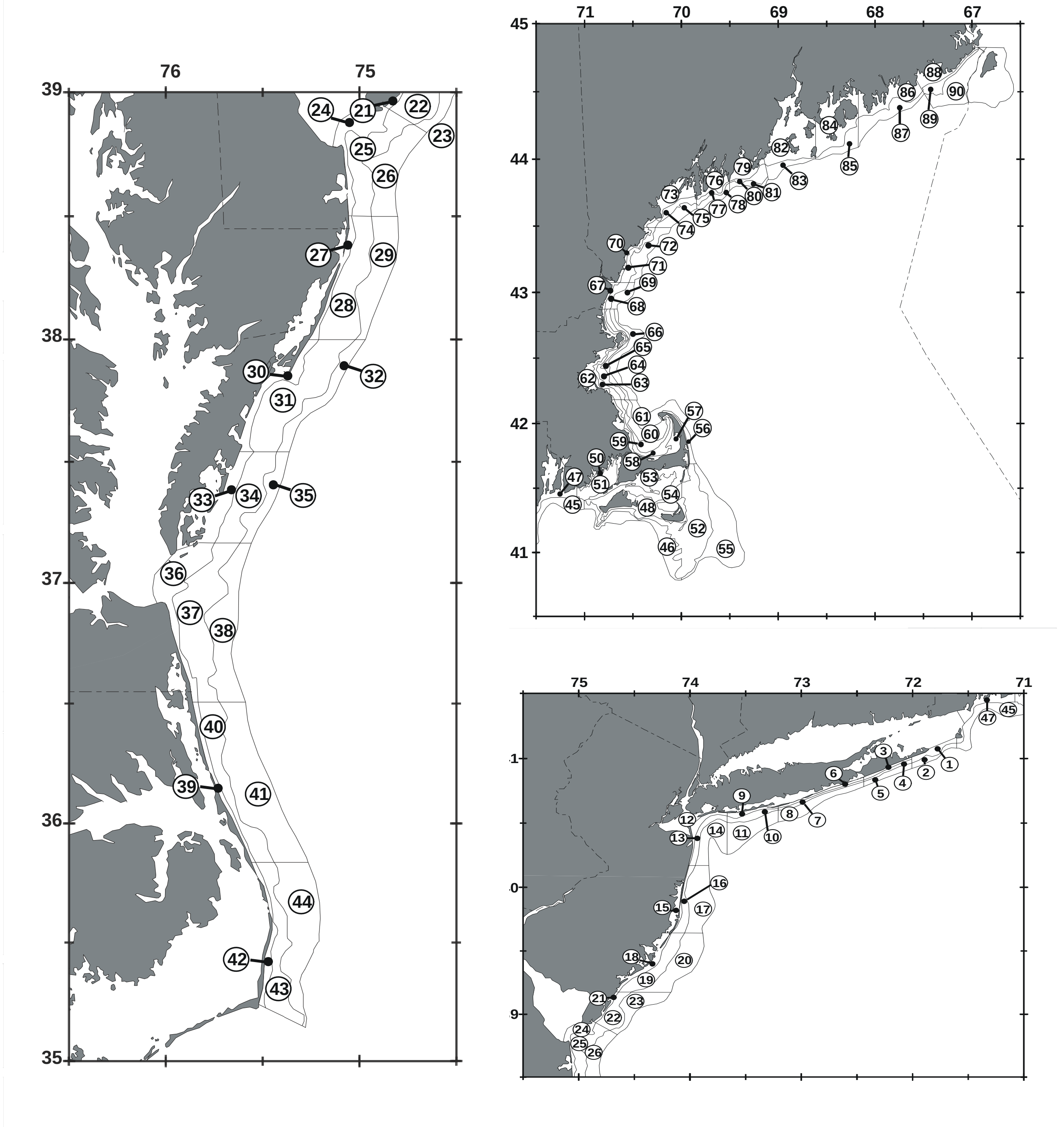


Figure 2. Inshore depth strata that have been sampled during Northeast Fisheries Science Center bottom trawl research surveys. Some of these may not be sampled presently.

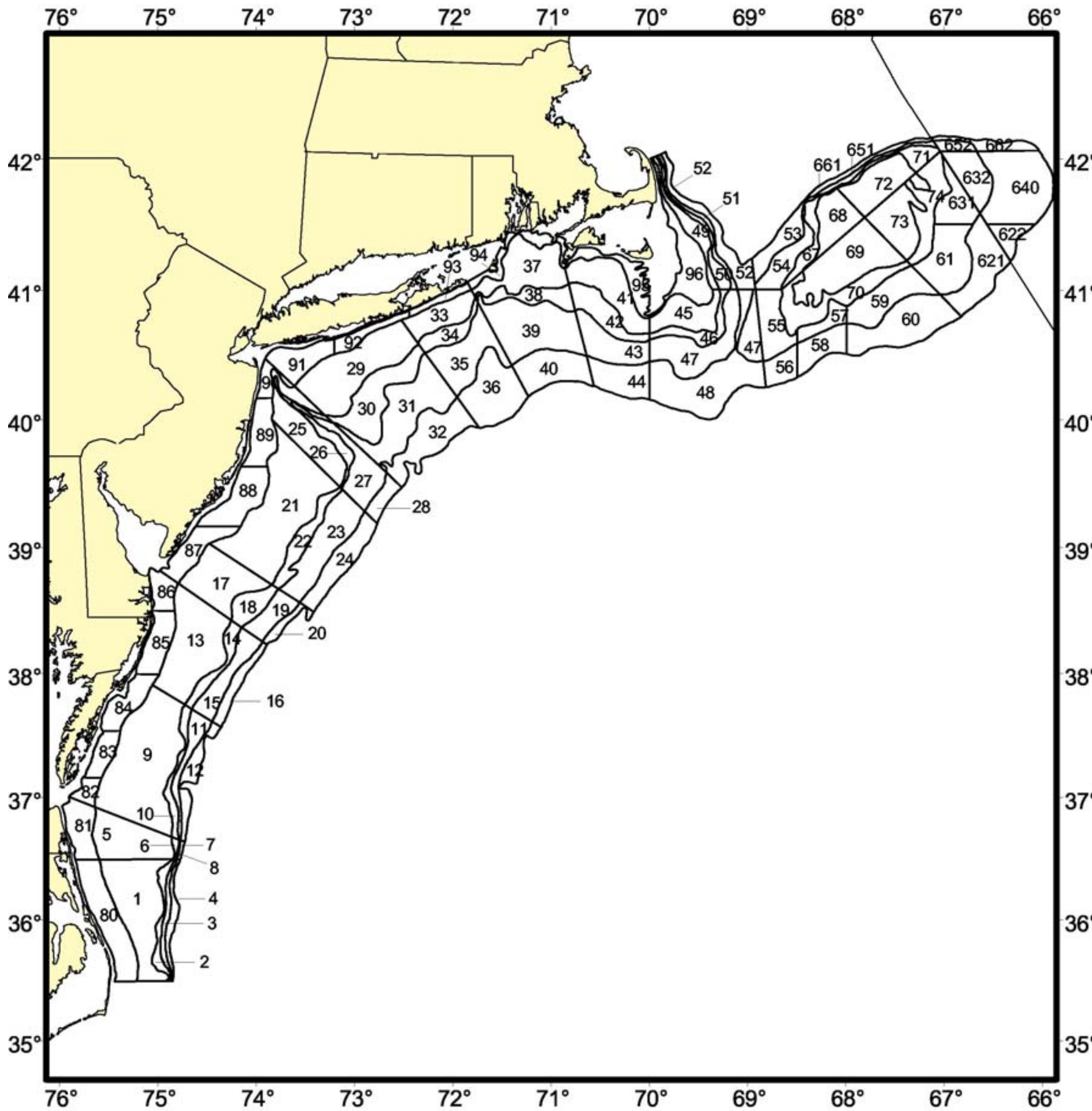


Figure 3. Depth strata sampled during Northeast Fisheries Science Center shellfish (scallop, surfclam and ocean quahog) research surveys. Different subsets of the strata are surveyed for each of the species. For scallop survey details, see: A. Scallop Assessment / Introduction / Figure A4.1.

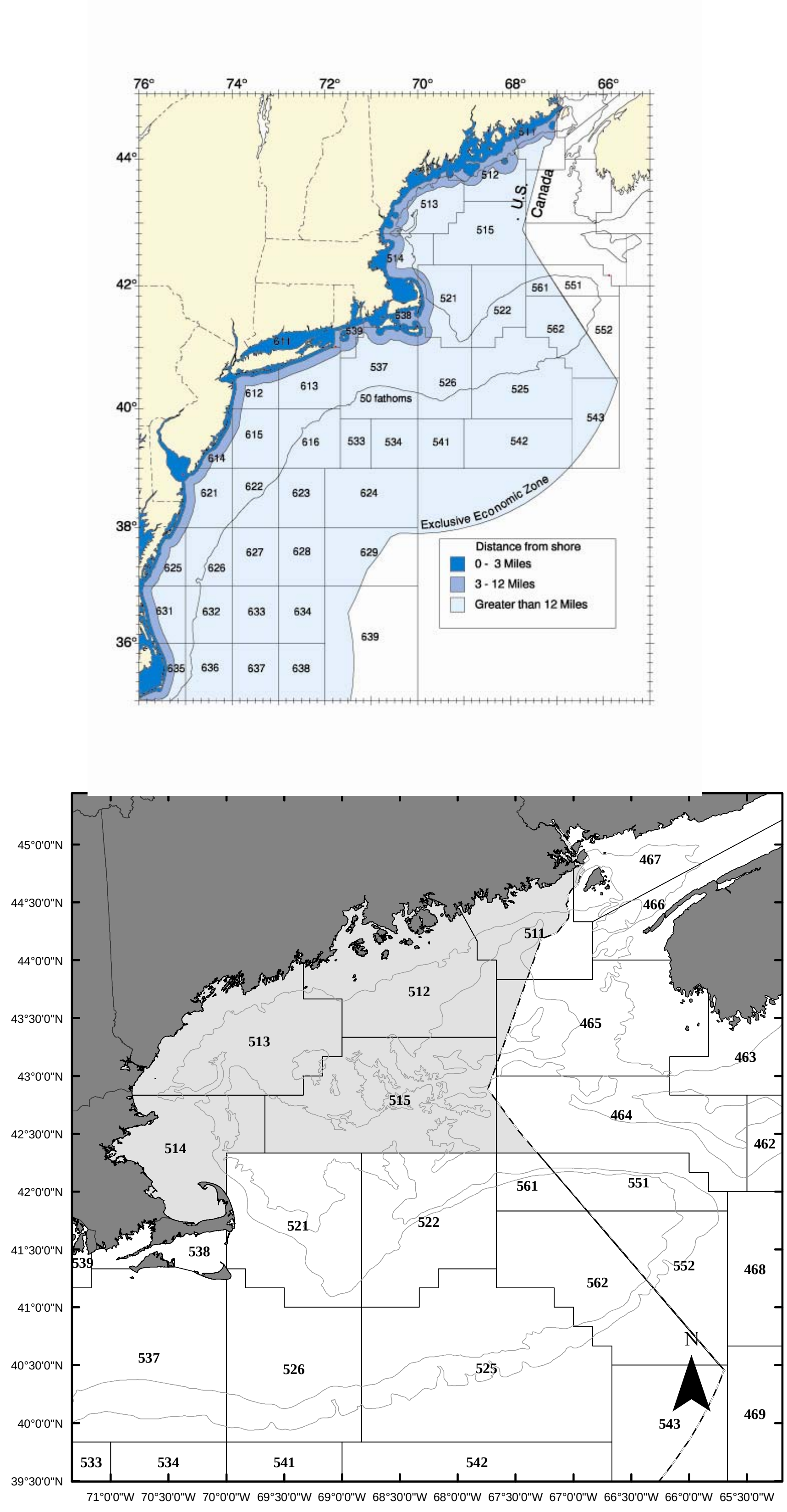


Figure 4. Statistical areas used for reporting commercial catches.

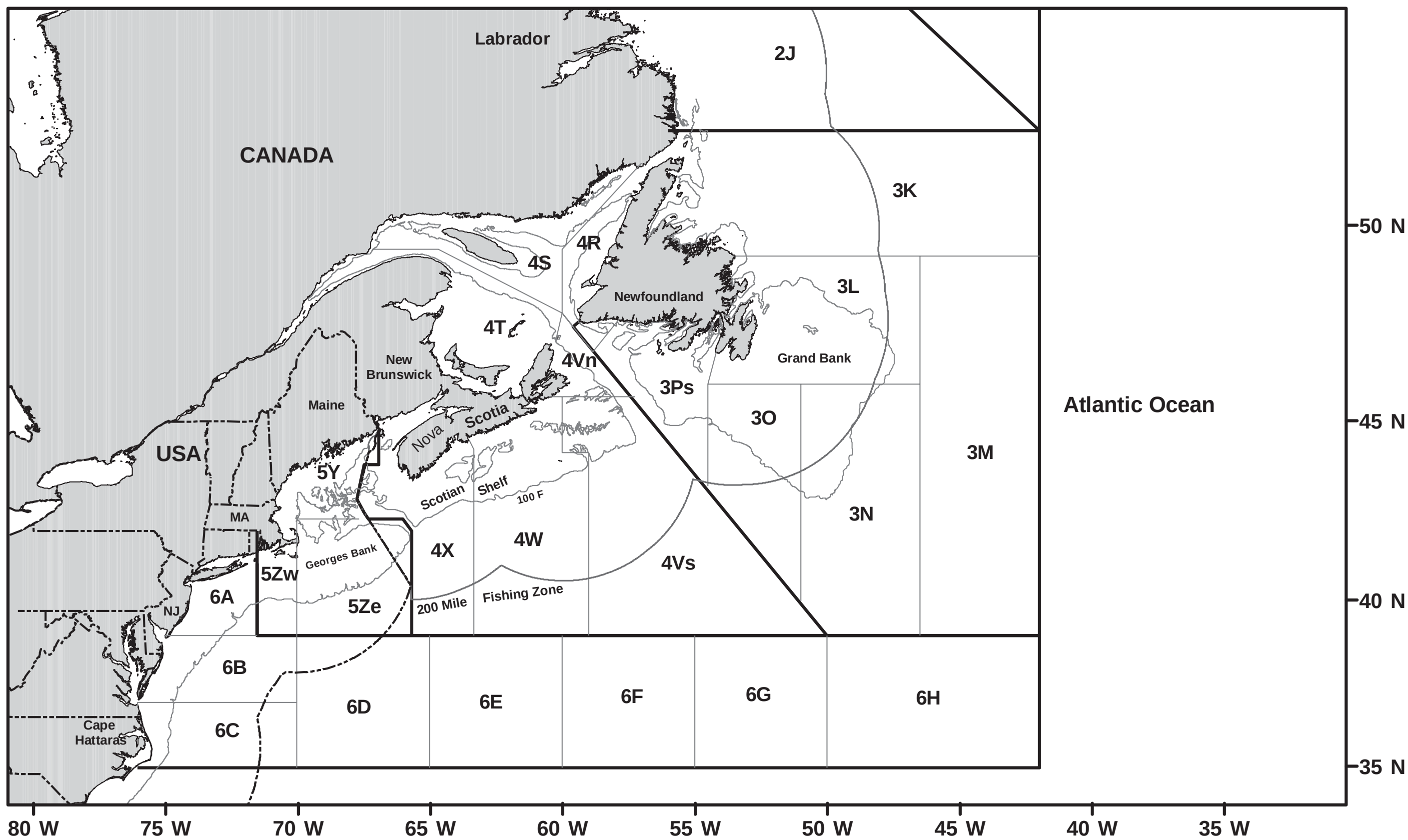


Figure 5. Catch reporting areas of the Northwest Atlantic Fisheries Organization (NAFO) for Subareas 3-6.

A. STOCK ASSESSMENT FOR ATLANTIC SEA SCALLOPS FOR 2018

1 Terms of Reference

1. Estimate catch from all sources including landings, discards, and incidental mortality. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize the uncertainty in these sources of data.

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. Summarize existing data, and characterize trends if possible, and define what data should be collected from the Gulf of Maine area to describe the condition and status of that resource. If possible provide a basis for developing catch advice for this area.

4. Investigate the role of environmental and ecological factors in determining stock distribution and recruitment success. If possible, integrate the results into the stock assessment.

5. 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 retrospective analyses (historical, and within-model) to allow a comparison with previous assessment results and previous projections.

6. 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} , $B_{\mathrm{THRESHOLD}}$, 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.

7. Make a recommendation about what stock status appears to be based on the existing model (from previous peer reviewed accepted assessment) and based on 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).
   c. Include descriptions of stock status based on simple indicators/metrics.

8. Develop approaches and apply them to conduct stock projections.
   a. Provide numerical annual projections (through 2020) and the statistical distribution (i.e., probability density function) of the catch at F_{MSY} or an F_{MSY} proxy (i.e. the overfishing level, OFL) (see Appendix to the SAW TORs). 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 which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions. Identify reasonable projection parameters (recruitment, weight-at-age, retrospective adjustments, etc.) to use when setting specifications.
c. Describe this stocks vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

9. 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.

2 Executive Summary

TOR 1 (Landings, fishing effort, etc.) U.S. sea scallop landings were high and stable during 2003-2012, averaging about 25,000 mt meats, over twice the long-term 1950-1999 mean. Landings declined during 2013-16, averaging 17,163 mt meats, but rebounded in 2017 to 23,458 mt meats. A majority of landings (59%) during 2013-17 came from the Mid-Atlantic, with most of the rest from Georges Bank (36%). Landings in the Gulf of Maine have been increasing in recent years, although still small relative to the Mid-Atlantic and Georges Bank. The Gulf of Maine landings in 2017, 976 mt meats, were the highest landings in this region since 1981.

Fishing effort increased from the mid-1970s to the early 1990s. Effort then declined rapidly, but increased somewhat during the early to mid-2000s, and then declined again. Recent fishing effort levels are about half that of the peak in the early 1990s.

Discards were highly variable with year and region. They have generally been lower since 102 mm dredge rings were required by regulation in December 2004, but increased in 2016 and 2017 to 2190 and 1413 mt meats, respectively, mainly in the Mid-Atlantic. The increase is likely related to the large 2013 year class in the Mid-Atlantic that was generally below marketable size in 2016. Incidental fishing mortality (mortality of scallops that interact with the gear but are not caught) is highly uncertain. However, recent studies evaluated during this assessment indicate that it is likely lower than previously estimated. Estimates were reduced from 0.2 and 0.1 of fully recruited fishing mortality on Georges Bank and the Mid-Atlantic, respectively, from the last benchmark assessment, to 0.1 and 0.05 in this assessment. The CASA model uses a single term to account for both discard and incidental mortality; for this assessment, these estimates were 0.11 and 0.06 of fully recruited fishing mortality on Georges Bank and the Mid-Atlantic, respectively.

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. They have declined slightly since, although the indices are still well above levels observed prior to 1995.

A drop camera survey was conducted between 2003 and 2017 on Georges Bank and the Mid-Atlantic, using a systematic grid design. This survey showed trends and scale similar to the dredge survey from 2003-2014, but was above the dredge survey in 2015 and 2017. A towed camera (Habcam) survey was conducted during 2011-2017 on Georges Bank and 2012-2017 in the Mid- Atlantic. Biomass estimates from Habcam were similar to those from the dredge and the drop camera during 2011-2014, but it has shown increases since 2014 similar to that from the drop camera. It is likely that the disparity between the dredge and the two optical surveys is due to decreased dredge efficiency in the very high density areas that have been observed during 2015-2017.

TOR 3 (Gulf of Maine). During most of the last three decades, the scallop population and fishery in the Gulf of Maine has been relatively small ($< 1\%$ of overall landings in most years), and mostly concentrated near shore in state waters. However, substantial increases in both the population and the fishery have been observed in recent years. These include some offshore areas such as Stellwagen and Platt's Bank, Jeffery's Ledge, and Ipswich Bay, as has been observed in recent surveys and commercial activity.

TOR 4 (Environmental and ecological effects). Food supply (phytoplankton), temperature, pH, predators, invasive species, and parasites and diseases can all affect scallop populations. Growth and weight at shell height tend to be greater at shallower depths, probably due to increase food supply. Warming temperature may increase scallop productivity in the short term by increasing growth, but ocean acidification may have negative impacts on sea scallops long-term. The sea star *Astropecten americanus* consumes small invertebrates, including early post-settlement sea scallops, and is likely the main factor excluding sea scallops from the deep (> 80 m) portion of the Mid-Atlantic Bight. Evidence is presented that shows that *A. americanus* is spreading northward and to shallower waters, perhaps due to warming temperatures. *Cancer* crabs are predators of juvenile sea scallops (< 90 mm), and may be an factor in inducing density-dependent mortality in these juveniles. The invasive tunicate *Didemnum vexillum* has been observed in portions of Georges Bank and the Gulf of Maine, and especially on the northern edge of Georges Bank. Scallop larvae cannot settle on *Didemnum vexillum*, so this tunicate may be reducing available habitat for sea scallops. Scallop gray meat disease has been observed in recent years, mainly on Georges Bank, and in particular in Closed Area I. Larval nematodes have become common in scallop meats in the southern Mid-Atlantic, perhaps related to increases in abundance of sea scallops and loggerhead sea turtles.

TOR 5 (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. Size- and year-specific natural mortalities were estimated in the model. Three models were used, one each for the open and closed portions of Georges Bank, and a model for the Mid-Atlantic. Adult natural mortality was revised upward from 0.16 and 0.2 on Georges Bank and the Mid-Atlantic, respectively, to 0.2 and 0.25 in this assessment. For the first time in this assessment, natural mortality was directly estimated by year. For the

Georges Bank closed areas, natural mortality was estimated by year, independent of size for shell heights 40 mm and greater. In the Georges Bank open areas and Mid-Atlantic, adult natural mortality was fixed at the above values, but juvenile natural mortality (less than $65 - 70$ mm) was estimated by year. In all three regions, natural mortality showed little variation in most years, but showed high spikes in a few years. The spikes in juvenile natural mortality appear to associated with large juvenile year classes.

The models appeared to give good estimation of abundance and biomass for most years, except for the 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. Whole stock abundance and biomass have increased dramatically starting in 2015, due to very large year classes in both Georges Bank and the Mid-Atlantic; biomass in 2017 was the highest in the 1975-2017 modelled time series. Combined biomass, abundance and fishing mortality in 2017 for the three models were 317,335 mt meats (153,182 mt gonads), 14,171 million scallops and 0.12, respectively. Adding the "DSENLS" scallops from the southeast corner of the Nantucket Lightship Closed Area to these estimates, the totals are: 380,389 mt meats (223,268 mt gonads), and 22,422 million scallops.

TOR 6 (Reference points). Reference points were estimated using the Stochastic Yield Model (SYM), which outputs probabilistic reference points that take into account parameter uncertainty. The new model (point) estimates are: $F_{\mathrm{MSY}} = 0.64$, $B_{\mathrm{MSY}} = 116{,}766$ mt meats, and MSY= 46,531 mt meats. This compares to $F_{\mathrm{MSY}} = 0.48$, $B_{\mathrm{MSY}} = 96{,}480$ mt meats, and MSY= 23,798 mt meats in the last assessment (NEFSC 2014). The fishing mortality associated with annual catch limits (ACLs) was calculated at $F_{\mathrm{ACL}} = 0.51$, compared to $F_{\mathrm{ACL}} = 0.38$ in the last assessment. Reference points were also calculated using spawning stock biomass in terms of gonad weight instead of meat weight. While this approach is promising, it was not used as the primary method to calculate reference points in this assessment.

TOR 7 (Status determination). Estimated overall fishing mortality in 2017 was 0.12, which was well below the estimate of $F_{\mathrm{MSY}} = 0.64$ from this assessment as well as the estimate of $F_{\mathrm{MSY}} = 0.48$ from the previous one (NEFSC 2014). Therefore overfishing was not occurring in 2017. Spawning stock biomass in 2017 was estimated as 380,389 mt meats (317,334 mt meats excluding scallops in the deep water southeast portion of the Nantucket Lightship area), which is over three times the estimated $B_{\mathrm{MSY}} = 116{,}766$ meats, and over six times the overfishing threshold of $\frac{1}{2}B_{\mathrm{MSY}} = 58{,}383$ mt meats. Therefore, the stock is not overfished. According to the previous assessment, $B_{\mathrm{MSY}} = 96{,}480$ mt meats, so the stock is not overfishing under the previous definition either. There is almost no chance that the stock is either overfished or overfishing is occurring.

TOR 8 (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 projections indicate that biomass is expected to decline gradually from their current very high levels, but it is highly likely that biomass will remain well above B_{MSY} through 2020. Landings are also expected to be well above average for the next several years.

3 Sea Scallop Working Group

The Sea Scallop Working Group met four times (February 5-9, March 26-29, April 30-May 4 and May 30, 2018) in the development of this assessment. The SAW/SARC Sea Scallop Working group members are:

Burton Shank, NEFSC Population Dynamics, WG Chair
Deborah Hart, NEFSC Population Dynamics, Assessment Lead
Jui-Han Chang, NEFSC Population Dynamics, Co-Assessment Lead
William DuPaul, VIMS, College of William and Mary
Benjamin Galuardi, NMFS GARFO
Jonathan Peros, NEFMC
David Rudders, VIMS, College of William and Mary
Liese Siemann, Coonamesset Farm Foundation
Kevin Stokesbury, SMAST, U. Mass. Dartmouth

Other participants in one or more of the meetings include: John Ceccolini, Jason Clermont, Ron Smolowitz, Coonamesset Farm Foundation; George Lapointe, Fisheries Survival Fund; Josh Brighton, Martin Johnson, Fleet Fisheries; Kevin Kelly, Maine DMR; Cate O'Keefe, Massachussetts DMR; Michael Bergman, Russel Brown, Nicole Charriere, Peter Chase, Toni Chute, Jonathan Duquette, Kevin Friedland, Dan Hennen, Larry Jacobson, Charles Keith, Min-Yang Lee, Chris Legault, Brian Linton, Alicia Miller, Michael Simpkins, James Weinberg, NEFSC; Sam Asci, Vincent Balzano, NEFMC; Michael Torre, U. Maine; David Bethoney, Steve Cadrin, Susan Inglils, Brooke Wright, SMAST, U. Mass. Dartmouth; Danielle Ferraro, Arthur Trembanis, U. Delaware; Roger Mann, Sally Roman, VIMS, College of William and Mary; Scott Gallager, WHOI.

4 Introduction

Distribution

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, typically on firm sand and gravel bottoms. Major aggregations in US waters occur in the Mid-Atlantic from Virginia to Long Island at depths of 35 to 80 m, on Georges Bank, including the Great South Channel and Nantucket Shoals, at depths from 40 to 120 m, and, to a lesser extent, in the Gulf of Maine, mainly in relatively shallow waters (Hart and Chute 2004). 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 A4.1). Results for Georges Bank and the Mid-Atlantic are combined to evaluate the stock as a whole.

According to the Atlantic Sea Scallop Fishery Management Plan, all sea scallops in the US EEZ belong to a single stock. However, the US sea scallop stock can be divided into Georges Bank (survey strata ≥ 45, Figure A4.1), Mid-Atlantic (survey strata ≤ 36), Southern New England (survey strata between 37 and 44), and Gulf of Maine (north of Georges Bank and Cape Cod) regional components based on survey data, fishery patterns, and other information. For assessment modeling purposes, Southern New England is considered to be part of the Georges Bank region. During most years, a large majority of the scallops in the Gulf of Maine lie in state waters, which is managed by the state of Maine.

Recent population and assessment history

Historically strong year classes (2012 year class on Georges Bank, and 2013 in the Mid-Atlantic) are currently a major component of the populations in both Georges Bank and the Mid-Atlantic. The 2012 year class on Georges Bank occurred primarily on the southern portions of Georges Bank, especially in the Nantucket Lightship Closed Area. A substantial portion of these are in the deep water southeast corner (hensforth referred to as DSENLS) of this area, where there has been few scallops and no fishing activity prior to 2012. Scallops in this small area number in the many billions, with densities as high as hundreds per square meter, but are growing very slowly. Their modal shell height in 2017, when they were five years old, was just under 80 mm, which is more typical of a somewhat slow-growing three year old, and still below fishable size. Patches of comparable densities in other portions of the Nantucket Lightship Closed Area are growing faster, although still slightly slower than would be expected. Thus, the slow growth is probably more related to food supply than density, although density may be a contributing factor. Because of their unusual growth and location, and the fact that they have not to date been fished, the DSENLS scallops (defined as in the area bounded by 40.55° and 40.333° N, 69.5° and 69° W, and at depths greater than 70 m) were excluded from the Georges Bank population models. Their population was instead estimated empirically using survey data.

There is a somewhat similar situation in the Mid-Atlantic, where the large 2013 year class there is widespread, but most concentrated in the shallower portions of the Elephant Trunk region, off of Delaware Bay. Scallops at the highest densities there appear to be growing slowly, although still somewhat faster than the DSENLS scallops. These scallops were still included in the Mid-Atlantic area, but were accounted for by using slower growth estimates for the Mid-Atlantic during $2015 - 2017$.

These extraordinary densities have also caused difficulties for dredge surveys. Estimated dredge surveys abundances in the high density areas have been consistently about one third of that estimated by optical surveys, whereas dredge and optical surveys give similar estimates in other areas.

The sea scallop fishery is one of the most valuable fisheries in the U.S., with annual ex-vessel value ranging from \$400 to \$600 million during recent years. US landings during 2003-2012 exceeded 24,000 metric tons meats (mt) each year, roughly twice the long-term mean, but declined below 20,000 mt during 2013-2016. Landings rebounded to 22,946 mt meats in 2017, and is expected to increase further in 2018.

Area closures and openings have had a strong influence on sea scallop population dynamics and the fishery (Figure A4.1). Roughly 40% of the productive scallop grounds on Georges Bank were closed to both groundfish and scallop gear during most of the time since December 1994. Portions of these areas have been reopened as part of a rotational fishing program between June 1999 and January 2001 and since November, 2004. In the Mid-Atlantic, there have been four rotational scallop closure areas since 1998, although one (Virginia Beach) was short-lived. These areas are generally closed for two to three years, and then reopened to allow harvesting. The areas are closed again after strong recruitment is observed.

Sea scallops in U.S. waters have been assessed used forward projecting size-structured models since 2007. Past fishing mortality, biomass and recruitment are estimated using a version of the CASA (Catch-At-Size Analysis) model, loosely based 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 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 Stochastic Yield Model (SYM, Hart 2013), which takes into account parameter uncertainties in these calculations.

Life History

Sea scallops feed by filtering phytoplankton, microzooplankton, and detritus particles. Sexes are separate and fertilization is external. Larvae spend $5-7$ weeks in the water column before settling. Sea scallops typically become mature at age 2 ($35-75$ mm shell height (SH)), but gamete production is limited until age 4. Major predators include sea stars (e.g., *Asterias* spp. and *Astropecten americanus*) and *Cancer* spp. crabs (Hart and Chute 2004, Hart 2006, Shank et al. 2012). Scallops fully recruit to the NEFSC survey at 40 mm SH, and to the current commercial fishery at around 95-110 mm SH, although scallops as small as 70 mm were landed during the 1980s and 1990s.

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 have 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 since early rings may be 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 (individual) random effects on the paramaters L_∞ and K. This allows not only estimation of mean von Bertalanffy coefficients, but also their variability among individuals in the population. Note that 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 estimates in Hart and Chute (2009b) were based on scallops collected between 2001 and 2007; these estimates have been updated in the past two benchmark assessments (NEFSC 2010, 2014). For this assessment, these estimates were updated using analysis of archived shells collected between 1982 and 2000, as well as shells collected from 2008−2017. However, no new data were available for the Mid-Atlantic after 2013.

These data give evidence of temporal changes in growth, with faster growth in the most recent periods, and slowest growth coinciding with peak fishing effort in the early 1990s. For this reason, separate estimates of growth were made for different periods (see Table A4.1). More discussion on growth, and in particular growth estimates on a finer spatial scale or using covariates, can be found on Appendix A1.

Maturity and fecundity

Sexual maturity commences at age 2; sea scallops > 40 mm that are reliably detected in the surveys used in this assessment are all considered mature individuals. However, individuals younger than 4 years may contribute little to total egg production, but fecundity increases rapidly with age (MacDonald and Thompson 1985; Hart and Chute 2004, Hennen and Hart 2012).

Spawning generally occurs in late summer or early autumn throughout the sea scallops' range. Spring spawns or minor dribble spawns at other times can also occur. The spring spawn is often strong in the Mid-Atlantic Bight (DuPaul et al. 1989) and spring spawns on Georges Bank have also been observed, and may be becoming stronger with increases in winter 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). The timing of spawning has no direct effect on a size-based stock assessment.

Shell height to weight relationships

Shell height-meat weight relationships allow conversion from numbers of scallops at a given size to meat weights. They are expressed in the form $W = \exp(\alpha + \beta \ln(H))$, where W is meat weight in grams and H is shell height in mm.

Shell height/meat weight data have been collected during annual NEFSC sea scallop surveys since 2001, and in recent years, also by VIMS dredge surveys. These data have been used in scallop assessments since 2007, and were updated for this assessment. Parameters were estimated using mixed-effect generalized linear models, similar to Hennen and Hart (2012). Details can be found in Appendix A2.

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. Depth and subarea have a significant effect on the shell height/meat weight relationship (Hennen and Hart 2012, Appendix A2). In this assessment, covariate-adjusted shell height/meat weight relationships were used to calculate survey biomass, and the simple relationships given in Table A4.2 were used in the models (CASA and SAMS), where depth is not explicit.

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 dredge surveys during the late spring and early summer may differ from those in the rest of year. Additionally, commercial fishers concentrate on speed, and often leave some meat on the shell during shucking (Naidu 1987, DuPaul et al. 1990). On the other hand, meats in fishery catches may gain weight due to water uptake during storage on ice. Finally, fishers may target areas with relatively large meat weight at shell height, which may increase commercial meat weights compared to those collected on research vessels.

Observer data was used to adjust meat weights for seasonal variation and for commercial practices (Appendix A2). 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. These indicate a temporal trend towards greater meat weight at size since 1994, similar to the trends observed for growth.

For purposes of estimating spawning stock biomass (SSB), gonad weight may be a better surrogate for egg production than meat weight. Although gonad weight was not the primary metric used to for spawning stock biomass this assessment, reference point calculations using gonad weights as well as meat weights are presented here. For this purpose, we use the shell height to gonad weight relationships of Hennen and Hart (2012) to calculate gonad weight (Table A4.3).

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 (on Georges Bank, where this study occurred) was approximately to 0.12 (NEFSC 2010). Hart et al. (2013) estimated M within the CASA stock assessment model as 0.16 in the Georges Bank closed areas, which was then used in the previous benchmark assessment (NEFSC 2014). Based on CASA model estimates for this assessment, it appears that M is somewhat higher; an estimate of $M = 0.2$ for Georges Bank sea scallops was used in this assessment for reference points and projections.

No direct estimate of M is available for Mid-Atlantic sea scallops. The ratio of the Brody 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). Because K is higher for Mid-Atlantic scallops, M should also be higher there; an estimate of $M = 0.25$ was used in this assessment.

In two of the CASA models, we directly estimate juvenile natural mortality in the model, and show that it is in some years higher than the adult natural mortality given above, which may be due to density-dependent factors.

Another way to estimate M uses the oldest age observed t_{MAX}. For sea scallops, there is no year 1 ring, and year two and three rings are sometimes missing, and rings at the edge of very old scallops may be hard to discern. Thus, using 1 + maximum number of observed rings represents a lower bound for t_{MAX}. We will consider three such longevity-based estimators for natural mortality (Then et al. 2015):

$$M = a/t_{\text{MAX}}, \quad \text{where } a = 5.109 \tag{A4.1}$$

$$M = \exp(a + b\ln(t_{\text{MAX}})), \quad \text{where } a = 1.717 \text{ and } b = 1.01 \tag{A4.2}$$

$$M = at_{\text{MAX}}^{b}, \quad \text{where } a = 4.118 \text{ and } b = -0.916. \tag{A4.3}$$

For both regions, the maximum number of rings observed is 19, making these scallops at least 20 years old. Using $t_{\text{MAX}} = 20$ gives maximum estimates of natural mortality of 0.256, 0.270, and 0.315, respectively. Given that these scallops may in fact have been older than 20, and that there may be life history differences between finfish and mollusks, natural mortality obtained from these estimators is consistent with those estimated above.

Tables

Table A4.1. Estimated regional Von Bertalanffy parameters, by temporal periods.

Years	L_∞	SDL_∞	K	SDK	SEL_∞	SEK
GB All						
75-93; 97-99	141.5	9.9	0.429	0.109	0.5	0.003
93-96	136.6	9.3	0.429	0.109	0.5	0.003
00-06	140.2	9.0	0.478	0.114	0.4	0.003
07-11	144.6	9.8	0.464	0.113	0.4	0.003
12-16	151.5	11.4	0.429	0.109	0.5	0.003
GB Open						
75-93; 97-99	136.8	9.1	0.442	0.110	0.6	0.003
93-96	132.1	8.5	0.442	0.110	0.6	0.003
00-06	135.8	8.2	0.494	0.116	0.6	0.004
07-11	140.1	8.9	0.479	0.114	0.6	0.004
12-16	146.7	10.4	0.442	0.110	0.7	0.003
GB Closed						
75-93; 97-99	145.7	10.7	0.420	0.108	0.7	0.003
93-96	140.8	10.0	0.420	0.108	0.7	0.003
00-06	143.8	9.5	0.470	0.113	0.6	0.004
07-11	148.5	10.4	0.455	0.111	0.7	0.004
12-16	156.0	12.3	0.420	0.108	0.7	0.003
Mid-Atlantic						
75-77; 87-03; 06	131.4	8.2	0.534	0.117	0.4	0.003
78; 83-86; 04-05; 07	133.4	8.1	0.564	0.121	0.4	0.003
79-82; 08-12	137.4	8.6	0.564	0.121	0.4	0.003

Table A4.2. Parameter estimates for the basic shell height H to meat weight W relationship $W = \exp(\alpha + \beta \ln(H))$

Region	α	SE	β	SE
Georges Bank	-9.67	0.09	2.732	0.019
GB Open	-10.47	0.13	2.889	0.026
GB Closed	-9.011	0.12	2.604	0.025
DSENLS	-11.84	0.69	3.167	0.15
Mid-Atlantic	-9.192	0.11	2.602	0.028

Table A4.3. Parameter estimates for shell height H to gonad G relationships, where the relationship takes the form $G = \exp(\alpha + \beta \ln(H) + \gamma \ln(D) + \delta \ln(L)$, where D is depth (in meters), and L is latitude (in degrees).

Region	α	β	γ	δ
Georges Bank basic	-10.55	2.8		
GB with depth covariate	-6.38	2.79	-0.97	
Mid-Atlantic basic	-14.11	3.41		
MA with depth and latitude covariates	-12.4	3.34	-0.9	0.06

Figure

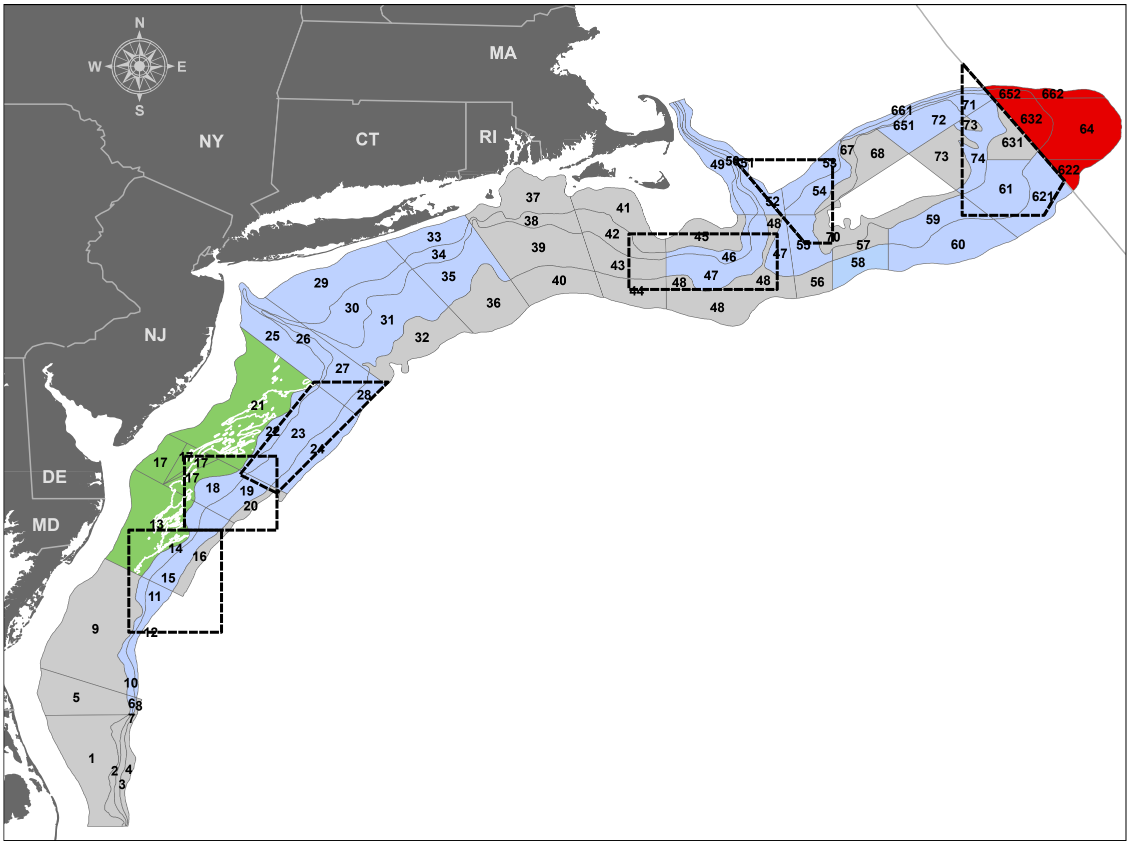


Figure A4.1. NEFSC shellfish survey stratra. The regularly surveyed strata are in blue and the strata that have not been regularly surveyed since 1989 are in gray. The deeper portions of the green strata in the Mid-Atlantic were regularly surveyed prior to 1990, and since 2000. Strata in Canadian waters, shown in red, were often surveyed prior to 2011, but these data are not included in this assessment. The polygons outlined with black dashed lines are the rotational closed areas in the Mid-Atlantic and the combined groundfish/scallop closed areas on Georges Bank.

5 Fishery and Catch (TOR 1)

Overview and Management History

The sea scallop fishery in the US Exclusive Economic Zone (EEZ) is managed under the Atlantic Sea Scallop Fishery Management Plan (FMP), first implemented on May 15, 1982. An overview of important management measures is given in Table A5.1. From 1982 to 1993, the primary management control was a minimum average meat weight requirement for landings, commonly known as the "meat count" requirement. 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.

Amendment 4 of the Sea Scallop Plan (NEFMC 1993), implemented in 1994, changed the management strategy from meat count regulation to limited access, effort control and gear regulations for the entire EEZ. Limited access permits were issued to vessels with a history in the fishery; no new permits have been issued since. Restrictions were gradually made on days-at-sea (DAS), minimum ring size, and crew limits; DAS has been reduced from over 200 in fishing year 1994 to 120 during 1999-2003, to 24 in open areas in 2018 ("fishing years" used by management have typically been March-February, but was recently changed to April-March; however, unless otherwise stated, this assessment uses calendar years).

The minimum size of the rings in the dredge bag was gradually increased from 76 mm (3") in 1994 to 83 mm (3.25") in 1995, 89 mm (3.5") during 1996-2004 and 102 mm (4") since December 2004. The minimum size of the twine top mesh has also been increased from 15 to 25 cm (6" to 10") since December 2004; while this measure is mainly to reduce bycatch of finfish, it also likely allows some small swimming scallops to escape. Finally, the crew size on a vessel has been restricted, usually to seven persons, in order to incentivize fishing for larger scallops (since most scallops must be shucked at sea by regulation, a limited crew can process a greater weight of large scallops than smaller ones per unit time). 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 A4.1). Scallop biomass increased about 20-fold 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 conducted in 1999, using an early version of the SAMS model to take into account the rebuilding effects of closed areas, indicated that the stocks would rebuild with less severe effort reductions than called for in Amendment 7, so this days at sea schedule was thus modified. A combination of the closures, effort reduction, gear and crew restrictions led to a rapid increase in biomass (Murowski et al. 2000, Hart and Rago 2006, Hart et al. 2013), and sea scallops were declared to be rebuilt in 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 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 (in days-at-sea or TACs) for reopened closed areas and general open areas. The three Georges Bank closed areas were divided into access areas, where fishing is periodically permitted, and long-term closures, where no scallop fishing is permitted. Some of the long-term closures were reopened to scallop fishing in 2018.

Unlike the Georges Bank closures, which applied to all scallop and groundfish fishing, the Mid-Atlantic rotational areas are specific to the scallop fishery. 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, and reopened in 2015. 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 portion of this area was reopened in 2015, and the remainder was reopened in 2017. A fourth closed area, Delmarva, directly south of the Elephant Trunk area, was initially closed in 2007, reopened in 2009, closed in 2012 and reopened in 2014.

Since the implementation of Amendment 4 in 1994, most landings have come from about 350 vessels with limited access permits. Two types of allocation are given to each of these vessels. The first are trips (with a trip limit, typically 12,000-18,000 lbs or 5443-8165 kg meats) to specified rotational access areas that had been closed to scallop fishing in the past. The second are days at sea, which can be used in areas outside the closed and access areas (Table A5.1). Vessels fishing under days at sea 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% 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 otter trawls are used to some extent 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 (termed a "small dredge permit"). Most general category vessels also use a single 3.2 m or smaller dredge, but some use otter trawls in the Mid-Atlantic. Most bycatch of sea scallops in other fisheries occurs in otter trawls, where the target species are squid, flounders, and other groundfish, although this is relatively small compared to the directed fishery (see Appendix A3). Recreational catch is negligible.

Landings

Prior to 1994, landings and effort data were collected during port interviews by port agents and based on dealer data. Since 1994, commercial data are available as 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. A standardized method (Wigley et al. 2008) for matching DR to VTRs and assigning areas to landings was used to allocate landings to region for 1994-2017.

Most landings of sea scallops only retain the adductor muscle, or "meat", although there is a small market for roe-on scallops. If not otherwise specified, landings in this assessment will be in terms of meat weight.

Sea scallop landings in the US increased substantially after the mid-1940s (Figure A4-1), 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 2017 were $23,458$ mt meats, the highest since 2012.

Landings from the Georges Bank and the Mid-Atlantic regions have dominated the fishery since 1964 (Table A4-1; Figure A4-2). US Georges Bank landings had peaks during the early 1960s, around 1980 and 1990, but declined precipitously during 1993 and remained low through 1998 (Figure A4-2). 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 again increased in 2012-2013, this time mainly due to shift of open effort from the Mid-Atlantic to Georges Bank, to take advantage of large year classes in the latter region, but have since declined.

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. 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 during 2012-2014, reflecting the poor 2007-2009 year classes there and concomitant effort shift onto Georges Bank, but have been increasing since then.

Landings from other areas (Gulf of Maine and Southern New England) are minor in comparison. Much of the Gulf of Maine scallops are in state waters, which is assessed and managed by the State of Maine (see Appendix A4). Gulf of Maine landings have been less than 3% of the total US sea scallop landings in most recent years. Maximum landings in the Gulf of Maine were $1,614$ mt during 1980 but trended downward afterwards through 2009, when landings were just 84 mt. Landings have been increasing in the most recent years; landings in 2017 were the highest since 1981.

Fishing effort and LPUE

Landings per unit effort (LPUE) (Figure A4-3) was computed as landings per day fished (days fished represent the time in days that the gear is fishing). This was obtained from the port interview records from larger vessels prior to 1994 and from at-sea observers on limited access vessels since 1994. 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 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 during 1994 need to be made cautiously, there is no clear break in the LPUE trend in 1994.

Fishing effort (days fished) was computed as the quotient Landings/LPUE. Effort is thus in units of days fished on limited access vessels; general category vessels, which usually only fish with one small dredge, would likely fish for several days to account for a single "day fished" under this metric. 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 (Figure A4-4). Effort increased in the Mid-Atlantic during 2000-2005, initially due to reactivation of latent effort among limited access vessels, and then due to increases in general category effort. Total effort since 2005 has remained fairly stable, although there have been shifts between regions.

Data from vessel monitoring systems (VMS) that are required on all scallop vessels fishing on Georges Bank and the Mid-Atlantic can give detailed spatial information on fishing activity. The VMS gives the positions of the vessels every half hour, from which average speed can be calculated. These were then filtered to eliminate times where the vessels were not fishing (either steaming or simply shucking scallops without gear in the water, Palmer and Wigley 2009), and the resulting spatial distributions were plotted (Figure A5.8).

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. Ratios of discard to total catch (by weight) were recorded by sea samplers aboard commercial vessels since 1992, though sampling intensity on non-access area trips was low until 2003; see Appendix A3 for detailed estimates.

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 (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 crew 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 but not caught, primarily due to damage (e.g., crushing) caused 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). A recent study estimated somewhat lower incidental mortality rates of 2.5% in the Mid-Atlantic and 8% on Georges Bank (Ferraro et al. 2017). Two other unpublished studies presented during the working group meetings suggests similar rates (Bochenek et al., Smolowitz et al.).

In order to use the above estimates 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 ratio R of scallops in the path of the dredge that were caught, to those killed but not caught is:

$$R = \frac{e}{c(1-e)} \tag{A5.1}$$

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):

$$F_I = F_L/R = \frac{1}{e} F_L c(1-e). \tag{A5.2}$$

Assuming $c = 0.025$ and $e = 0.6$ for the Mid-Atlantic and $c = 0.08$ and $e = 0.5$ for Georges Bank gives estimates of F_I of about 0.02 for the Mid-Atlantic and 0.08 for Georges Bank. Using an estimate of $c = 0.04$ from Murawski and Serchuk (1989) for the Mid-Atlantic and $c = 0.12$ from Caddy (1973) for Georges Bank gives estimates of incidental mortality of 0.03 and 0.12, respectively. The CASA model does not explicitly model discarding or discard mortality of small scallops, although it is included in the SYM and SAMS model. Because of this, the working group agreed to set $c = 0.05$ in the Mid-Atlantic and $c = 0.1$ on Georges Bank in the SYM and SAMS models, and $c = 0.06$ and 0.11, respectively, in the CASA model.

The above calculations are based on the assumption that the scallop is fully selected to the gear. If that is not the case, equation A5.2 becomes:

$$F_I(h) = \frac{1}{e} F_L c(1 - eq(h)). \tag{A5.3}$$

where $q(h)$ is the catchability of commercial gear on a scallop of shell height h. We took $q(h)$ to be of the form:

$$q(h) = q_0 s(h) \tag{A5.4}$$

where q_0 is taken as 0.5 on Georges Bank and 0.6 in the Mid-Atlantic, and $s(h)$ is commercial selectivity as estimated by the CASA model.

Commercial shell height data

Since most sea scallops are shucked at sea, it has often 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 (except for the access area fisheries) due to limited observer coverage. Except for 2006, observer coverage rates have been over 5% since 2003, and have been over 10% in some years.

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 A5.6).

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 A5.3). These data also indicate a trend towards larger sea scallops in landings. 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.

Tables

Table A5.1. Key allocations and management measures for the sea scallop fishery, including open area days at sea allocations (DAS) for full time limited access vessels and access area allocations (in kg meats) to Georges Bank and Mid-Atlantic access areas. Prior to Amendment 10 (2004), access area allocations were optionally obtained in exchange for a certain number of days at sea.

Year	DAS	GBAcc	MAAcc	Other key measures
1982-93	NA	NA	NA	Open access, meat count regulation
1994	204	0	0	Limited access, GB closures
1995	182	0	0	83 mm rings
1996	182	0	0	89 mm rings
1997	164	0	0	Overfishing/overfished declaration
1998	142	0	0	First MA rotational closures
1999	120	13608	0	
2000	120	27216	0	
2001	120	0	23133	Fishery declared rebuilt
2002	120	0	24494	
2003	120	0	28577	
2004	42	8165	32659	102 mm rings, rotational management
2005	40	16329	24494	
2006	52	40824	0	
2007	51	16329	24494	
2008	35	8165	32659	
2009	42	8165	32659	
2010	38	8165	24494	General category ITQs established
2011	32	16329	16329	ACL management implemented
2012	34	20412	12247	
2013	33	7502	3787	
2014	31	5210	5210	
2015	30.86	0	23133	
2016	34.55	0	23133	
2017	30.41	16329	16329	
2018	24	32659	16329	

Table A5.2. (a) Sea scallop landings (mt meats) in the Mid-Atlantic, Georges Bank, and in all areas. Dredge landings were reported as “other” before 1978.

	Mid Atl				Geo Bank				Total			
Year	Dredge	Trawl	Other	Sum	Dredge	Trawl	Other	Sum	Dredge	Trawl	Other	Sum
1964		0	137	137		0	6241	6241		52	6590	6642
1965		0	3974	3974		3	1478	1481		5	5592	5598
1966		0	4061	4061		0	883	884		1	5055	5056
1967		0	1873	1873		4	1217	1221		4	3178	3182
1968		0	2437	2437		0	993	994		0	3599	3599
1969		5	846	851		8	1316	1324		14	2302	2317
1970		14	459	473		5	1410	1415		19	2006	2026
1971		0	274	274		18	1311	1329		22	1949	1971
1972		5	653	658		5	816	821		11	1995	2006
1973		4	245	249		15	1065	1080		19	1773	1792
1974		0	937	938		15	911	926		16	2076	2091
1975		52	1506	1558		13	844	857		80	3132	3212
1976		819	2972	3791		38	1723	1761		864	5061	5925
1977		255	2564	2819		27	4709	4736		286	7536	7823
1978	4435	207	0	4642	5532	37	0	5569	10234	247	0	10481
1979	2857	29	1	2888	6253	25	7	6285	9572	64	9	9645
1980	2202	85	79	2366	5382	34	2	5419	9204	245	83	9532
1981	772	14	2	788	7787	56	0	7843	9852	144	9	10005
1982	1602	6	2	1610	6204	119	0	6322	8562	153	7	8723
1983	3092	19	10	3121	4247	32	4	4284	8398	124	21	8542
1984	3695	53	2	3750	3011	29	3	3043	7518	103	14	7635
1985	3230	49	2	3281	2860	34	0	2894	6575	90	12	6677
1986	3407	386	6	3799	4428	10	0	4438	8218	400	12	8631
1987	7639	1168	1	8808	4821	30	0	4851	12900	1199	10	14109
1988	6071	938	8	7017	6036	18	0	6054	12678	966	21	13666
1989	7894	534	5	8433	5637	25	0	5661	14258	570	49	14876
1990	6364	541	10	6915	9972	10	0	9982	16991	558	38	17587
1991	6408	878	14	7300	9235	77	0	9311	16225	973	89	17288
1992	4562	570	5	5137	8230	7	0	8238	13587	584	50	14221
1993	2412	393	3	2808	3637	18	0	3655	6878	413	36	7327
1994	5201	754	0	5955	1182	7	0	1189	6822	768	9	7599
1995	5786	798	7	6591	992	4	1	997	7161	810	21	7992
1996	4467	653	4	5124	2126	7	4	2137	7196	670	20	7886
1997	2703	378	1	3082	2347	9	1	2357	5804	403	26	6233
1998	2411	564	6	2981	2045	19	1	2065	4951	591	22	5564
1999	3629	959	1	4589	5172	6	1	5179	9146	968	4	10118
2000	8139	1210	2	9351	4910	40	5	4955	13296	1274	51	14621
2001	14144	1543	16	15703	4879	58	6	4943	19386	1645	23	21054
2002	15981	1426	36	17443	5967	33	11	6011	22354	1489	48	23891
2003	19040	1226	10	20276	4859	22	2	4883	24149	1255	13	25417
2004	22202	1194	26	23422	4249	146	11	4406	27446	1503	48	28997
2005	14361	1096	109	15566	8958	69	15	9042	24327	1190	176	25693
2006	7944	780	46	8770	15688	51	21	15760	25808	844	80	26732
2007	16234	345	55	16634	9419	45	18	9482	26085	416	92	26593
2008	16819	556	13	17388	6405	24	11	6440	23630	598	45	24273
2009	17487	12	1851	19350	6451	8	16	6475	24235	21	1873	26129
2010	19172	281	97	19550	5855	18	47	5920	25429	321	177	25927
2011	17224	318	205	17747	8159	14	135	8309	25914	373	366	26653
2012	11140	272	176	11620	13614	37	16	13724	25265	334	228	25915
2013	5683	229	54	5966	11823	27	25	11875	18263	311	89	18664
2014	9184	218	40	9442	5270	12	4	5286	14962	325	56	15343
2015	10301	171	29	10501	5096	5	0	5101	15886	280	41	16207
2016	13001	327	142	13470	3846	3	151	4000	17725	414	300	18439
2017	15067	174	83	15324	6873	12	76	6961	22946	341	171	23458

Table A5.2. (b) Sea scallop landings (mt meats) in Southern New England (SNE) and the Gulf of Maine (GOM). Dredge landings were reported as "other" before 1978.

	SNE				GOM			
Year	Dredge	Trawl	Other	Sum	Dredge	Trawl	Other	Sum
1964		52	3	55		0	208	208
1965		2	24	26		0	117	117
1966		0	8	8		0	102	102
1967		0	8	8		0	80	80
1968		0	56	56		0	113	113
1969		0	18	19		1	122	123
1970		0	6	6		0	132	132
1971		0	7	7		4	358	362
1972		0	2	2		1	524	525
1973		0	3	3		0	460	460
1974		0	4	5		0	223	223
1975		8	42	50		6	741	746
1976		4	3	7		3	364	366
1977		1	10	11		4	254	258
1978	25	2	0	27	242	1	0	243
1979	61	5	0	66	401	5	1	407
1980	130	3	0	133	1489	122	3	1614
1981	68	1	0	69	1225	73	7	1305
1982	126	0	0	126	631	28	5	664
1983	243	1	0	243	815	72	7	895
1984	161	3	0	164	651	18	10	678
1985	77	4	0	82	408	3	10	421
1986	76	2	0	78	308	2	6	316
1987	67	1	0	68	373	0	9	382
1988	65	4	0	68	506	7	13	526
1989	127	11	0	138	600	0	44	644
1990	110	6	0	116	545	0	28	574
1991	55	16	0	71	527	3	75	605
1992	119	5	0	124	676	2	45	722
1993	65	1	0	66	763	2	32	797
1994	29	1	0	30	410	6	9	425
1995	41	2	0	43	342	6	13	361
1996	59	5	0	64	544	5	12	561
1997	81	11	3	95	673	5	21	699
1998	103	3	0	106	392	5	15	412
1999	78	1	0	79	267	2	2	271
2000	85	3	1	89	162	21	43	226
2001	28	37	0	65	335	7	1	343
2002	20	12	0	32	386	18	1	405
2003	53	4	0	57	197	3	1	201
2004	830	151	11	992	165	12	0	177
2005	845	13	40	898	163	12	12	187
2006	2029	10	8	2047	147	3	5	155
2007	335	18	7	360	97	8	12	117
2008	303	6	16	325	103	12	5	120
2009	216	1	3	220	81	0	3	84
2010	254	9	26	290	148	13	6	168
2011	338	24	24	386	193	17	2	212
2012	118	4	32	154	392	22	3	417
2013	308	13	5	326	449	43	6	498
2014	220	6	1	227	288	89	11	388
2015	207	6	1	214	282	98	11	391
2016	322	5	2	329	556	79	5	640
2017	193	3	1	197	813	152	11	976

Table A5.3. Estimated discards (mt meats) by region. Southern New England discards are included in the Georges Bank region.

Year	GB	MA	GOM	Total
1989	4	213		217
1990	1	8		9
1991	5	11		16
1992	465	131		596
1993	346	22		368
1994	10	703		713
1995	23	495		518
1996	116	38		154
1997	46	11		57
1998	4	53		57
1999	142	114		256
2000	991	813		1804
2001	531	1969		2500
2002	107	1970		2077
2003	332	2244		2576
2004	102	2559		2661
2005	238	424		662
2006	378	244		622
2007	236	294		530
2008	341	457		798
2009	389	1013		1402
2010	717	730	3	1447
2011	555	536	2	1091
2012	890	278	0	1168
2013	362	157	6	519
2014	240	70	9	310
2015	202	562	2	764
2016	116	2074	6	2190
2017	482	931	34	1413

Figures

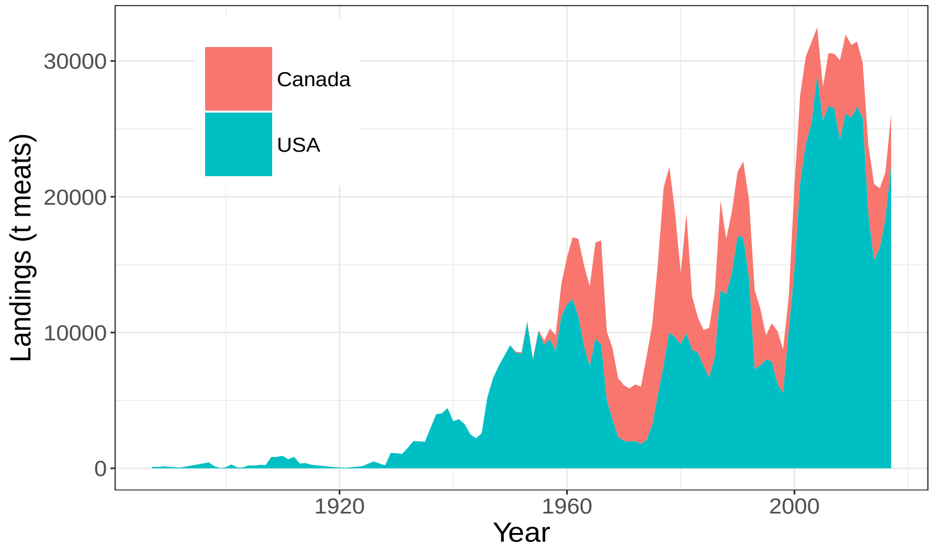


Figure A5.1. Long-term sea scallops landings in NAFO areas 5 and 6, comprising all U.S. scallop grounds as well as the Canadian portion of Georges Bank. The split between U.S. and Canadian landings prior to 1985 is in respect to port landed, rather than fishing location.

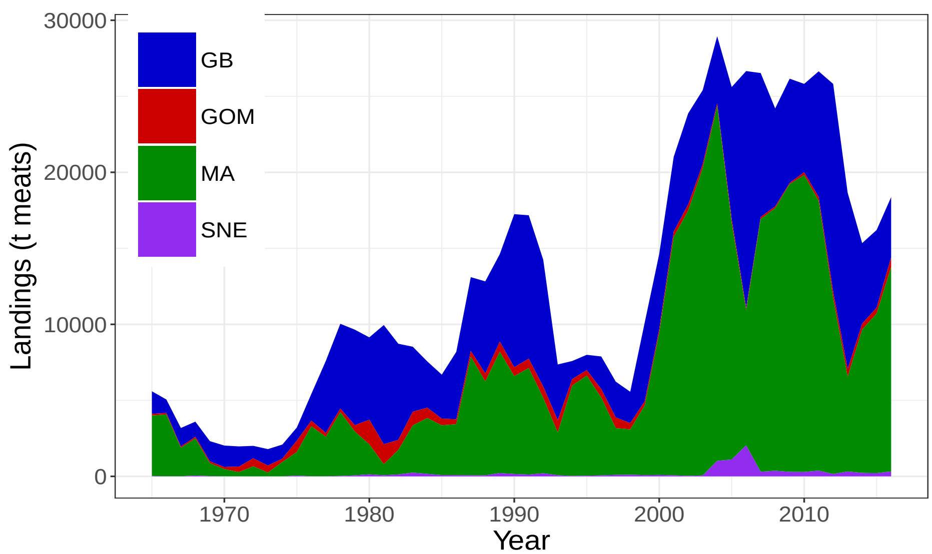


Figure A5.2. U.S. sea scallop landings by region.

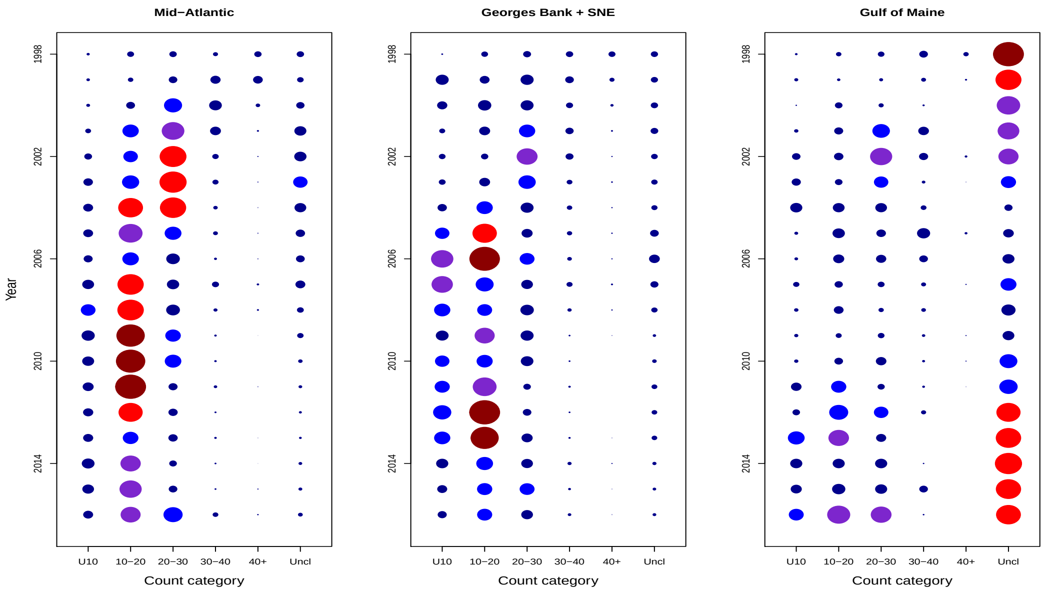


Figure A5.3. U.S. sea scallop landings by region and meat count. Market categories are U10 (> 45 g), 10-20 (23 to 45 g), 20-30 (15 to 23 g), 30-40 (11 to 15 g), 40+ (less than 11 g), and unclassified

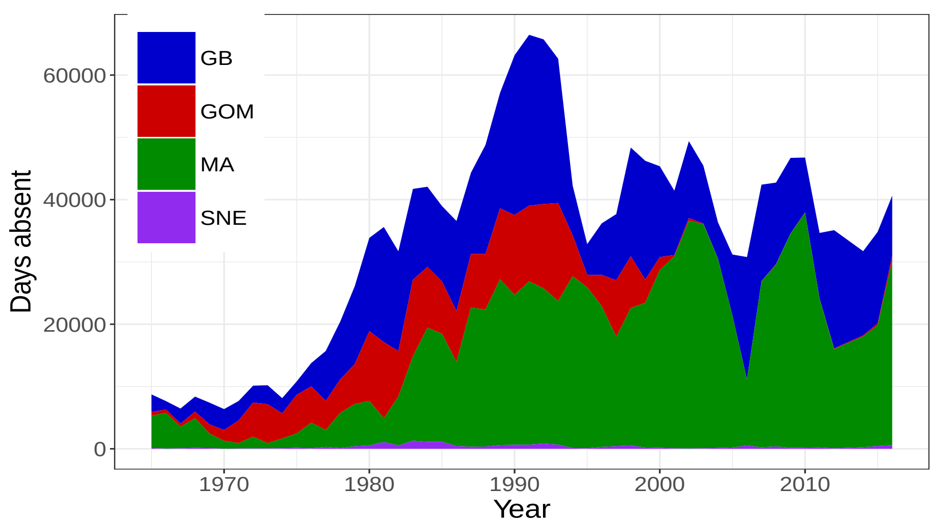


Figure A5.4. Limited access sea scallop effort, in days absent, by region. Much of the effort in the Gulf of Maine prior to 2000 is due to vessels primarily targeting sea urchins with dredge gear, but also landed sea scallops.

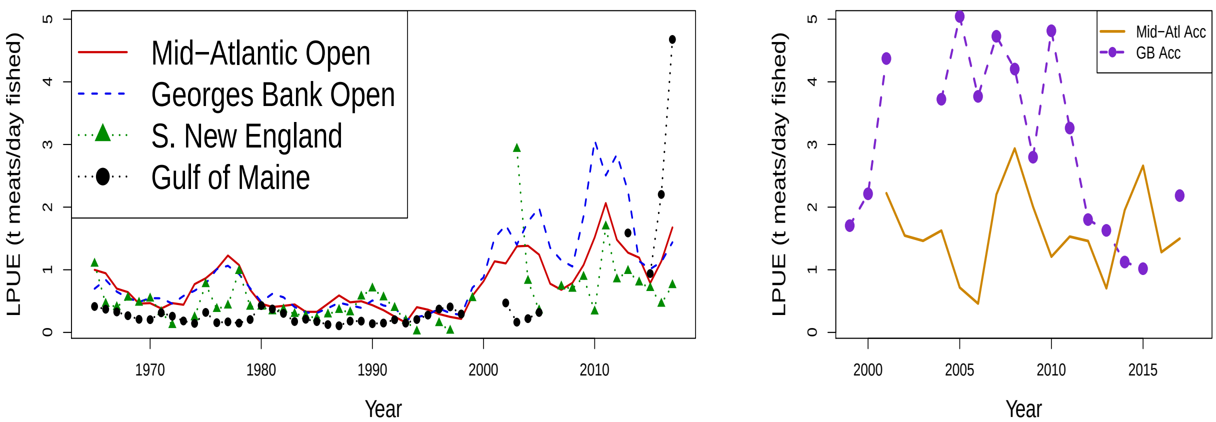


Figure A5.5. LPUE of limited access sea scallop vessels (mt per day fished), by region, not including access areas after 1998 (left), and in access areas (right).

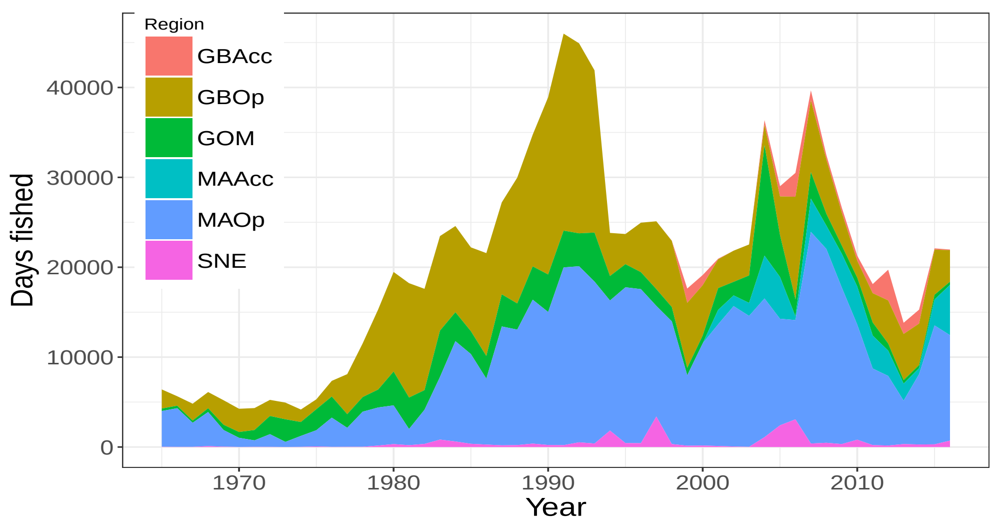


Figure A5.6. Sea scallop fishing effort, in days fished (equivalent). All areas are considered open prior to 1995.

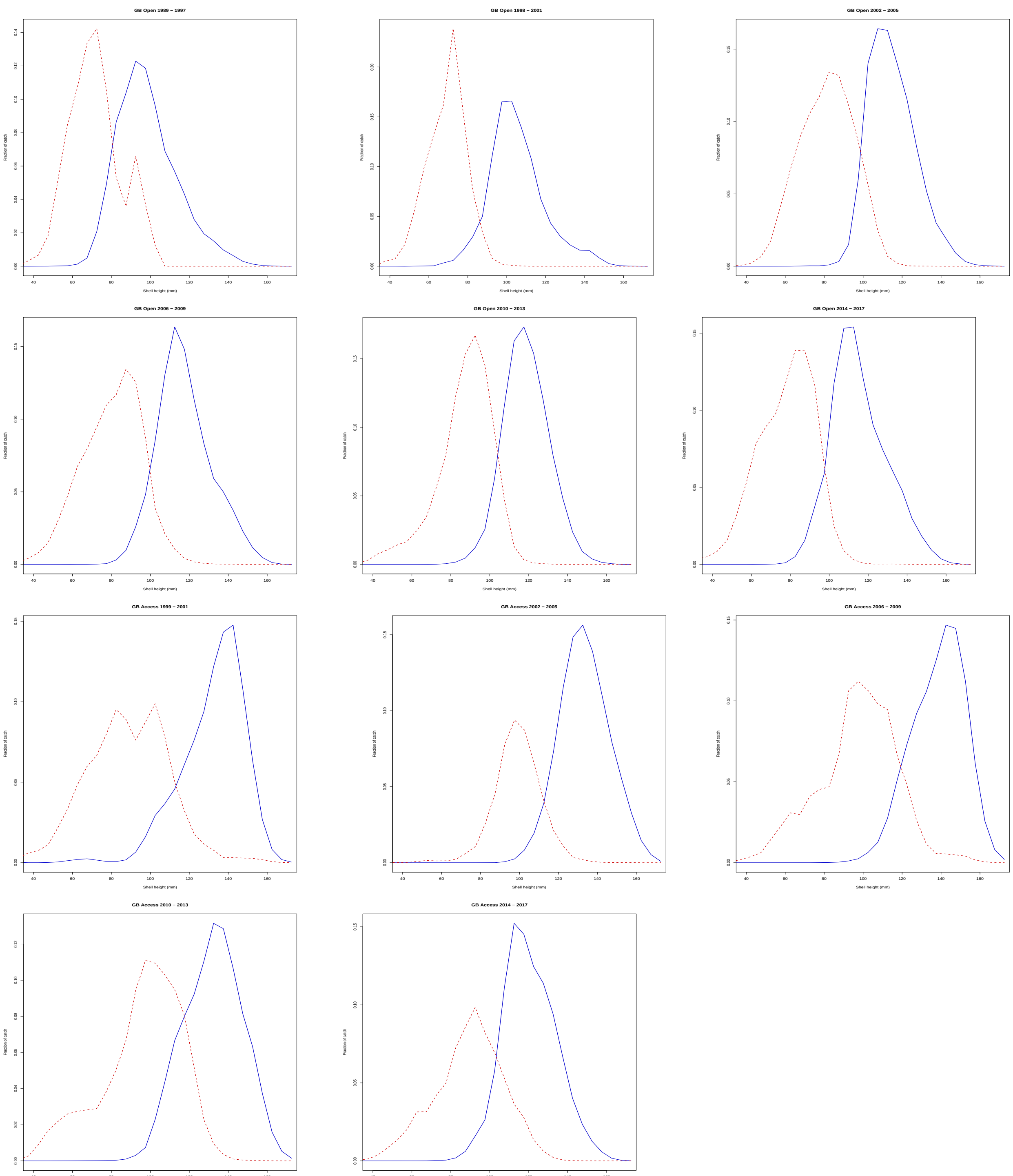


Figure A5.6(a) Size-frequencies of commercial catch for Georges Bank open areas (top two rows), and access areas (bottom two rows) during the specified periods. The solid blue line is retained scallops, and the dotted line is discarded ones.

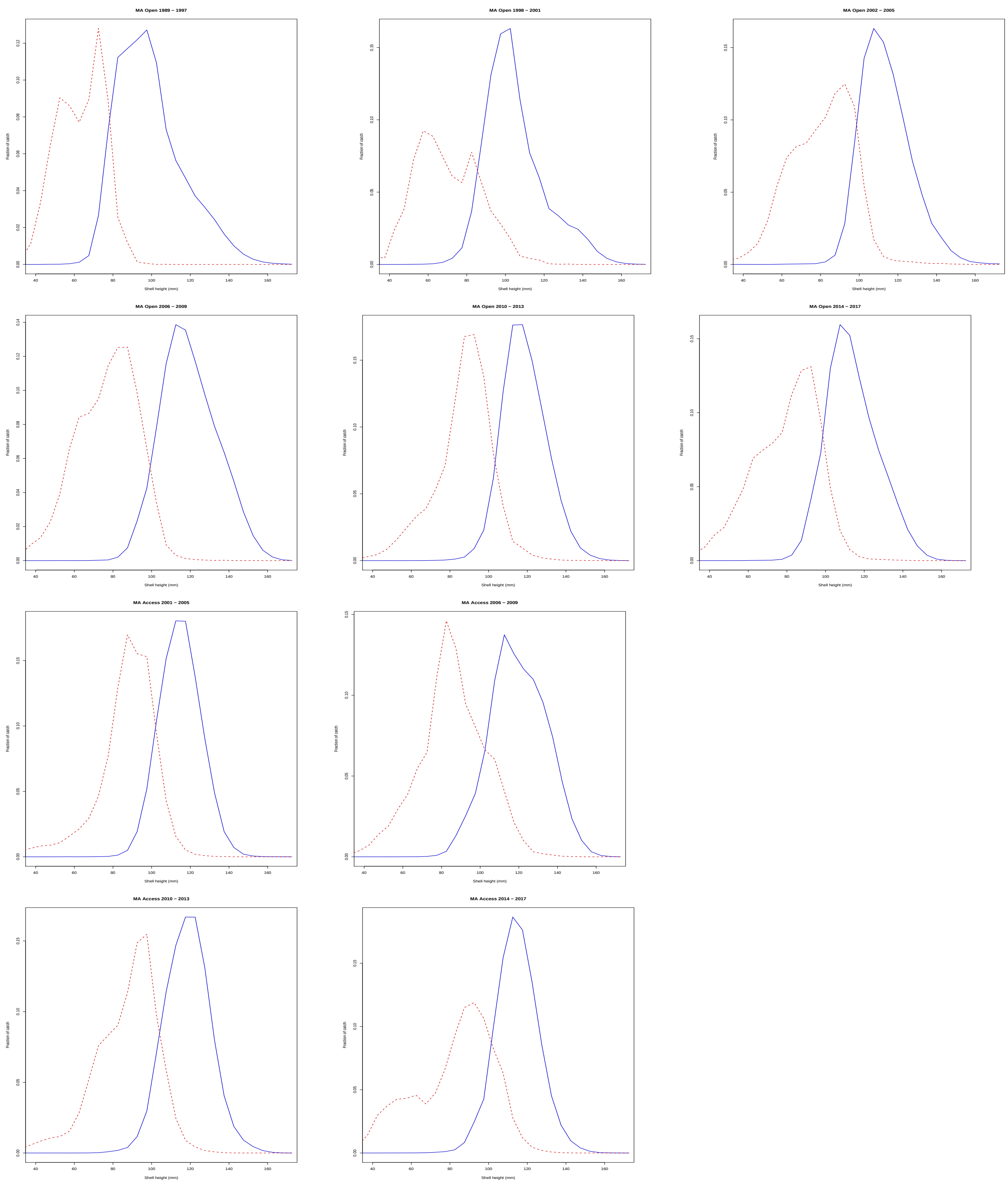


Figure A5.6(b) Size-frequencies of commercial catch for Mid-Atlantic open areas (top two rows), and access areas (bottom two rows) during the specified periods. The solid blue line is retained scallops, and the dotted line is discarded ones.

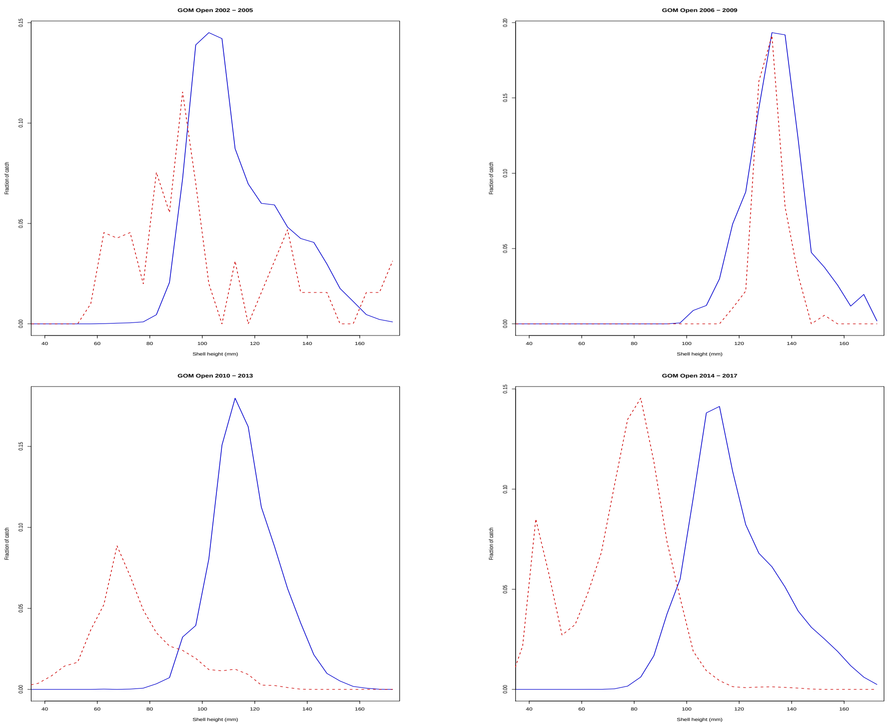


Figure A5.7(c) Size-frequencies of commercial catch for the Gulf of Maine during the specified periods. The solid blue line is retained scallops, and the dotted line is discarded ones.

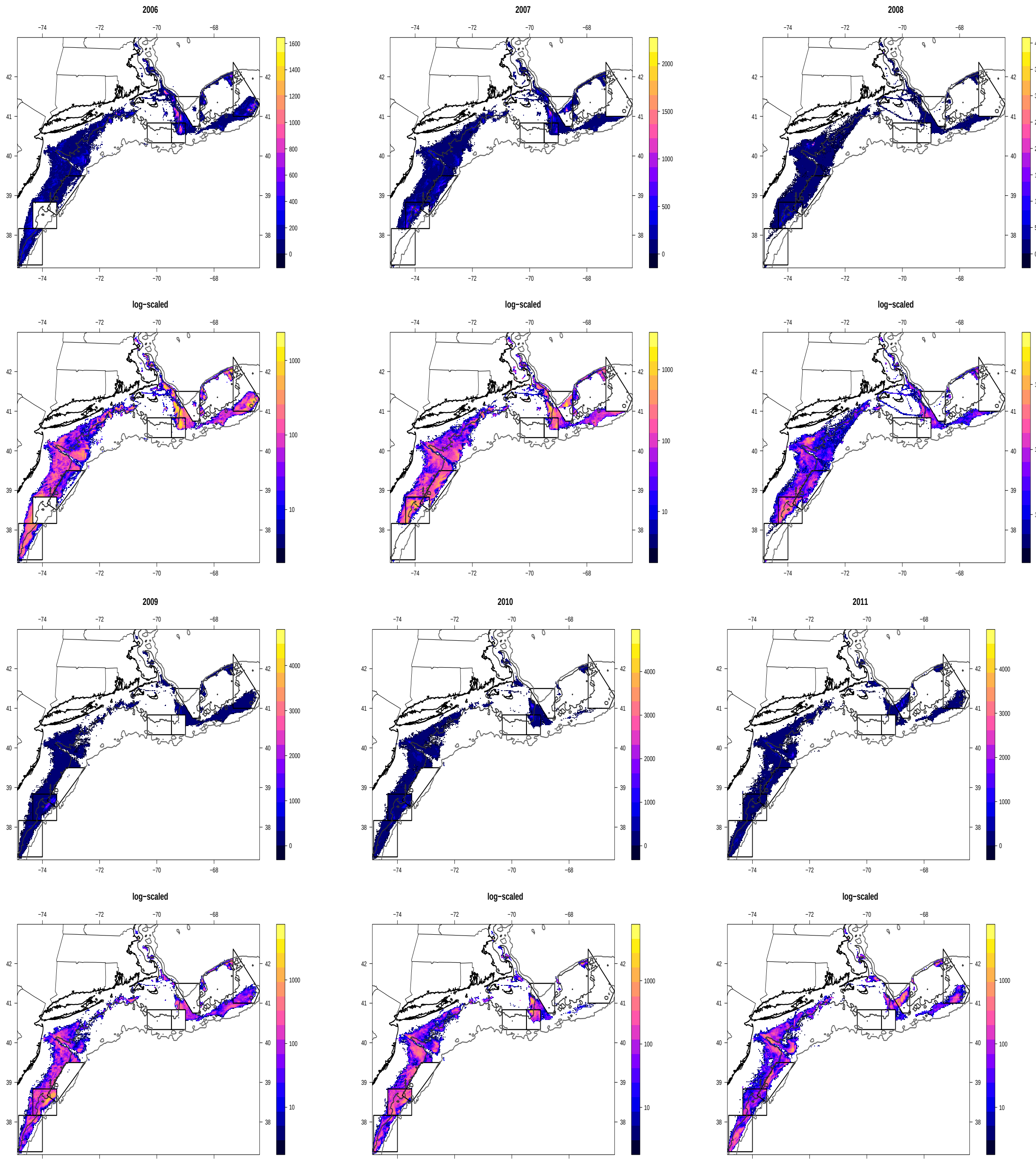


Figure A5.8. VMS plots of fishing effort on both an arithmetic (above) and logarithmic scale (below).

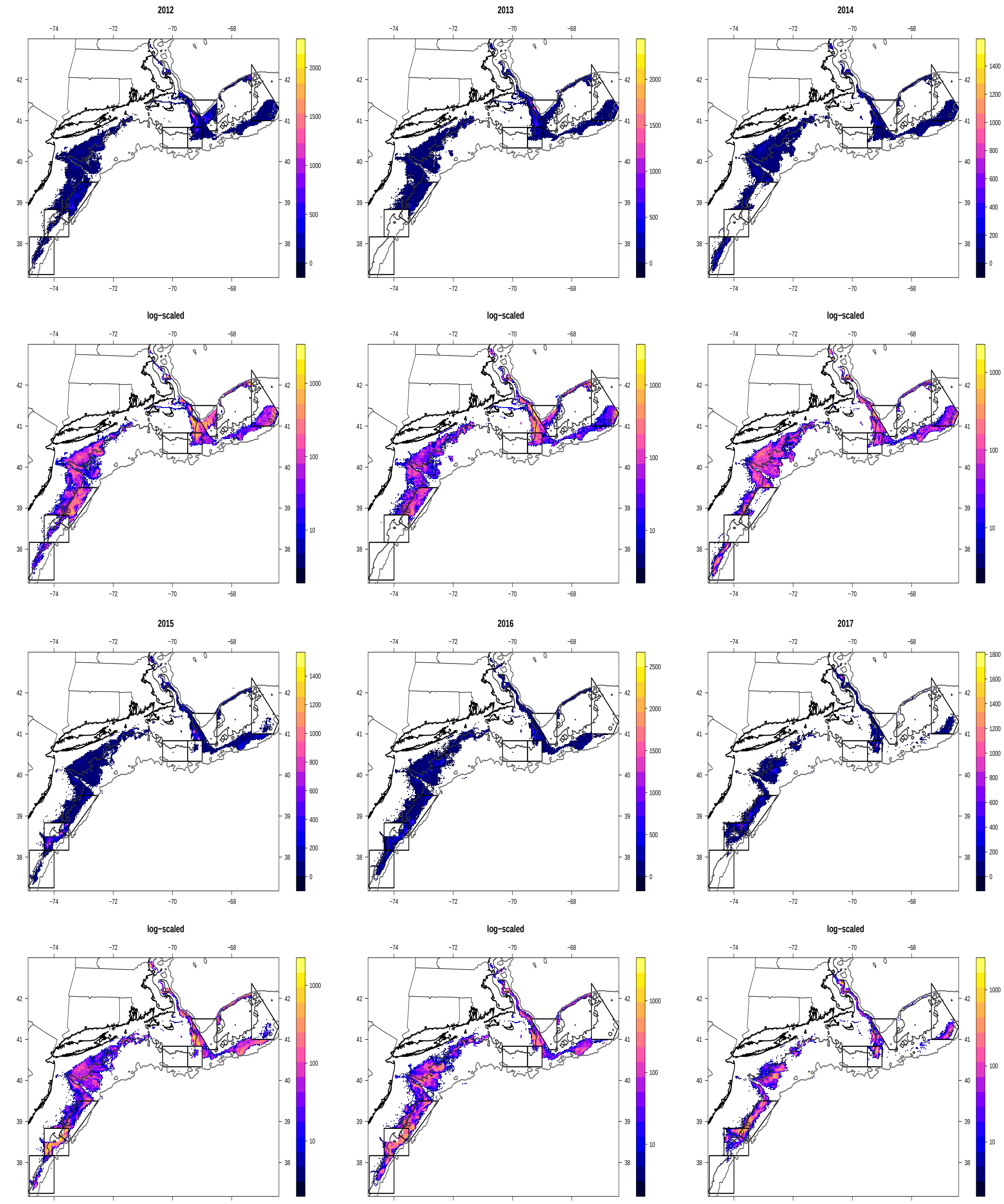


Figure A5.8 continued.

6 Surveys (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 (Figs B3-1 and B5-1). Means and standard errors were calculated using the standard methods for stratified random surveys (Cochran 1977, Serchuk and Wigley 1989; 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 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).

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.

During 2008-2017, 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 (Fig B5-2) 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, suggesting that the slight increase in catch on the Sharp is due to its longer tow distance.

As in NEFSC (2014), 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*:

$$L = -0.0388 + 0.001484 * D + 1.061 * N \quad (R/V\ Hugh\ Sharp) \tag{A6.1}$$

$$L = 0.08640.000444 * D + 0.972 * N \quad (R/V\ Albatross\ IV) \tag{A6.2}$$

where tow length L, and nominal tow length N is in nautical miles and depth D 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 were multiplied by 1.31 when calculating stratified means. Variance calculations in these strata include a term to account for the uncertainty in the adjustment factor (NEFSC 2007).

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 (NEFSC 1999). After re-stratification, the new strata set were combined into open, closed or other areas as required for assessment and management purposes.

The Virginia Institute of Marine Science (VIMS) conducted intensive dredge surveys of selected regions on commercial vessels since 2005, and region-wide dredge surveys of the Mid-Atlantic Bight since 2014. Between 2005 and 2014, they used partially randomized grid designs, but since then, they have used a random stratified design as in the NEFSC dredge surveys. The VIMS surveys are conducted on commercial scallop vessels using two dredges, one fished on each side of the vessel; 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* suggested that the survey dredge has the same fishing power on both vessels (NEFSC 2010).

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.

In some years, a few strata, typically those of relatively low density, were unsurveyed. In these cases, densities in these strata were imputed using GAM models. The models were fit to estimate trends in average catch rates over time for individual survey strata within subregions. Length composition data for such strata was estimated as the stratified mean length composition for the surveyed strata in the same subregion.

Capture efficiency of the survey dredge has been estimated by comparing dredge catches to densities observed by the Habcam system towed at the same location (Miller et al., 2018). The best estimates of dredge efficiency were 0.4 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, and other areas, including all of the Mid-Atlantic are assumed to be predominately sand and are expanded based on a survey dredge efficiency of 0.4.

Dredge survey results

Biomass and abundance trends for the dredge survey are presented in Table A6.1 and Figure A6.4. 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, and have remained high since.

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, and has been fairly stable in recent years. Survey shell height frequencies show a trend to larger shell heights in both regions since 1995 (Table A6.1).

Drop Camera Survey

Region-wide drop camera surveys were conducted by the School for Marine Sciences and Technology (SMAST), University of Massachusetts, Dartmouth since 2003 (Table A6.2, Stokesbury et al. 2004, Bethoney et al. 2016, Bethoney and Stokesbury 2018). Each survey station consists of clusters of four nearby drops, and stations are placed on a grid generally 3 nm (5.56 km) apart, although in some areas they are closer together. There were several cameras on the camera pyramid, but the survey index through 2015 is based on the “large” camera, which was a standard definition (SD) video camera that was mounted 1.575 m above the bottom in the center of the sampling frame. Each drop quadrat covered about 2.8 m^2.

The 2017 survey used a much higher resolution digital still camera as the primary camera. Although there were no direct calibrations between this camera and the SD video camera, comparisons could be made between the video camera and different digital still cameras that were deployed along side the video camera in previous years’ surveys (Appendix A5). These comparisons indicate that the video camera did not fully detect smaller scallops, and there may have been some variability in the selectivity among years. In the CASA models, the video camera was assumed to have a fixed logistic selectivity to account for this.

The precision of measurements must be considered in interpreting shell height data from video. Based on tank experiments, Jacobson et al. (2010) estimated measurement error for the video survey with a standard deviation of 6.1 mm. Field measurements are likely less precise than in a tank. For this reason, measurement error was estimated 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 (NEFSC 2014). The standard deviation that best fit the SMAST shell heights over all years and regions was 11 mm, which was the value used in modeling for this assessment.

Variances for estimated densities are approximated using the estimator for a simple random survey applied to station means. The CVs for biomass were assumed to be the same as for numbers. There was some variability in the areas covered during each year (Table A6.2).

Habcam Towed Camera Survey

Habcam is an underwater towed digital camera system. The camera(s) take rapid-fire photos of the sea floor (typically 6/sec) as it is towed at 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.

Four Habcam vehicles have been used. The first, known as"v2", carried a single camera, and was in operation from 2005 to 2013. The other three, known as "v3", "v4", and "v5", have two cameras to allow 3D viewing and more precise measurements. The v4 unit also carries a side-scan sonar and a full array of oceanographic sensors (e.g., CTD, chlorophyll, dissolved oxygen, pH, water spectrometer, etc.), and has been used for broad-scale sea scallop surveys since 2012.

Region-scale Habcam surveys were conducted on Georges Bank in 2011 using the v2 system, and on both Georges Bank and the Mid-Atlantic since 2012 using the v4 system. All broadscale Habcam surveys were conducted on the *R/V Hugh Sharp*. The broadscale survey was supplemented by intensive surveys of selected areas using the v2, v3, v4 and/or v5 systems deployed on commercial vessels.

Because of the large number of images collected, only subsets were examined for sea scallop measurements and counts; typically between 1 in 50 to 1 in 100 photographs were analyzed, corresponding to about one photograph annotated every 25 to 50 meters. These were expanded to large scales using zero-inflated generalized additive models followed by ordinary kriging of the residuals (Chang et al. 2017, Table A6.3, Figure A6.3). An alternative method, taking stratified means of the main transects, gave similar results. Measurement error was estimated for Habcam by comparing the shell heights to dredge data, as was done for the drop camera survey. Best fit occurred at a standard deviation of 13 mm, which was the estimate used in the CASA models.

Expanded biomass and abundance for the three surveys are shown in Figure A6.5. They give similar estimates, except in the most recent years, where the dredge survey is below the optical surveys. Even in these years, the dredge survey is comparable to the optical surveys except in the high density regions.

Scallops in the Deep Southeast Portion of Nantucket Lightship Closed Area

Table A6.4 gives survey estimates for the DSENLS scallops. As discussed earlier, these are scallops from the large 2012 year class that settled in deep water in the southeast portion of the Nantucket Lightship Closed Area, and are growing very slowly (Figure A4.1). In 2014, many of these scallops were below the 40 mm threshold for inclusion in the Habcam abundance estimate, and were only partially selected to the large camera drop camera survey. Habcam estimates for 2015-16 were well above the drop camera and dredge estimates, but they were internally consistent, declining slightly each year from 2015-17, likely due to natural mortality. The drop camera estimate from 2017 is also not significantly different than the Habcam estimate from that year, but drop camera estimates were much less than those from Habcam prior to 2017, likely due to selectivity of the large camera in 2014-15, and coverage and water clarity issues in 2016. Estimates from the 2016-17 dredge survey are much less than those from Habcam, suggesting reduced dredge efficiency in this area. For the purposes of this assessment, we will use the Habcam estimates because they appear the most reliable.

Examination of dredge efficiency using paired tows

In recent years, estimates from dredge surveys have been well below that of optical surveys in high density areas. For example, in both 2016 and 2017, Habcam estimates were about three times that of the dredge in the two areas containing most of the high densities, Elephant Trunk and Nantucket Lightship Closed Area. However, if these areas are excluded, dredge and Habcam estimated abundances were within 5% of each other.

To further examine this issue, it is useful to examine paired Habcam/dredge tows. In 2008-9, there were 137 paired tows between the *R/V Hugh Sharp*, towing a survey dredge, and Habcam v2 (Figure A6.6). These were all designed pairs that were towed within a few hundred meters of each other, and were the basis for the estimation of dredge efficiency used in this assessment (Miller et al. 2018). The y axis represents the ratio of the dredge catch divided by its swept area, to the mean density of the paired Habcam photos. There is little evidence in these data for declines in dredge efficiency, except possibly at densities exceeding 10 m^{-2}, and even that is unclear. The other set was from 2016 and 2017, and the dredge tows were a mix of ones from commercial vessels (conducted by VIMS), and from the *R/V Hugh Sharp*. Although some of the pairs in this set were designed experiments, a majority were opportunistic, where the Habcam tracks happened to come close to a dredge tow. For the 2016-17 pairs, it appears that dredge efficiency is clearly reduced past densities of 2 m^{-2}, at least for the commercial vessels.

The Working Group felt that this was a serious issue, and included as a research recommendation to further study this problem. In the meantime, for this assessment, in the expanded dredge survey estimates used in the CASA model (but not in tables in this section), the assumed dredge efficiency in the high density portions of the Elephant Trunk and Nantucket Lightship in 2015-17 was reduced by a factor of three, to compensate for the apparent reduction in efficiency in these areas.

Tables

Table A6.1. Dredge survey indices. Rel.Ab and Rel.Bms are in per tow units, whereas Abs.Ab and Abs.Bms are expanded using assumed dredge efficiency. Prop.Pos is the proportion of tows which caught at least one scallop.
(a) Georges Bank without DSENLS scallops

Year	Rel.Ab num/tow	CV	Rel.Bms kg/tow	CV	Ntows	Prop.Pos	MeanWt g	Abs.Ab millions	Ab.Bms mt
1979	87.4	0.41	1.758	0.34	108	0.89	20.1	1269	25522
1980	75.8	0.24	0.844	0.16	118	0.81	11.1	1031	11469
1981	61.2	0.13	1.065	0.13	82	0.83	17.4	753	13094
1982	132.9	0.46	1.050	0.32	118	0.83	7.9	2076	16407
1983	61.2	0.22	0.783	0.21	126	0.88	12.8	890	11392
1984	39.3	0.11	0.550	0.10	128	0.85	14.0	536	7513
1985	61.8	0.15	0.713	0.16	154	0.90	11.5	830	9573
1986	116.8	0.13	0.795	0.10	153	0.90	6.8	1445	9832
1987	120.1	0.17	1.055	0.16	170	0.86	8.8	1619	14223
1988	98.7	0.16	0.881	0.14	175	0.80	8.9	1289	11502
1989	63.6	0.11	0.525	0.08	120	0.78	8.3	806	6655
1990	184.1	0.24	1.222	0.22	175	0.81	6.6	2415	16028
1991	257.9	0.37	1.384	0.25	176	0.89	5.4	3678	19737
1992	232.0	0.44	1.933	0.43	171	0.89	8.3	3300	27492
1993	61.8	0.24	0.484	0.16	164	0.87	7.8	753	5892
1994	46.7	0.20	0.472	0.16	177	0.84	10.1	561	5672
1995	111.8	0.20	0.788	0.16	176	0.88	7.0	1637	11533
1996	133.6	0.20	1.624	0.19	171	0.90	12.2	1855	22555
1997	89.4	0.15	1.721	0.17	190	0.88	19.3	1292	24884
1998	283.0	0.26	3.828	0.32	195	0.87	13.5	3646	49304
1999	193.5	0.15	3.361	0.16	173	0.98	17.4	2663	46251
2000	766.7	0.29	7.364	0.22	164	0.91	9.6	9996	96008
2001	408.9	0.13	6.532	0.13	208	0.95	16.0	5560	88826
2002	334.5	0.14	7.043	0.14	214	0.93	21.1	4498	94727
2003	277.9	0.12	7.051	0.13	207	0.94	25.4	3839	97407
2004	291.5	0.11	8.498	0.12	218	0.94	29.2	3959	115432
2005	265.6	0.12	6.900	0.09	343	0.95	26.0	3888	101024
2006	221.3	0.13	6.439	0.13	236	0.94	29.1	3258	94792
2007	224.8	0.10	4.953	0.07	363	0.97	22.0	3453	76092
2008	321.8	0.10	6.542	0.08	239	0.97	20.3	4805	97676
2009	362.7	0.15	5.831	0.11	214	0.97	16.1	5497	88380
2010	413.1	0.21	7.468	0.09	268	0.97	18.1	6407	115838
2011	279.4	0.12	6.781	0.08	225	0.96	24.3	3946	95787
2012	225.3	0.13	4.970	0.08	224	0.97	22.1	3488	76943
2013	336.5	0.23	4.599	0.14	213	0.94	13.7	4416	60352
2014	519.8	0.47	3.859	0.18	124	0.90	7.4	8439	62644
2015	416.5	0.28	4.397	0.24	178	0.94	10.6	8975	94766
2016	164.2	0.14	2.846	0.16	310	0.90	17.3	3356	58174
2017	244.0	0.17	4.370	0.16	280	0.88	17.9	5558	99539

Table A6.1(b) Mid-Atlantic

Year	Rel.Ab num/tow	CV	Rel.Bms kg/tow	CV	Ntows	Prop.Pos	MeanWt g	Abs.Ab millions	Ab.Bms mt
1979	34.7	0.10	0.609	0.10	166	0.92	17.6	550	9661
1980	42.8	0.12	0.518	0.08	167	0.94	12.1	679	8223
1981	32.1	0.16	0.403	0.13	167	0.91	12.6	509	6391
1982	33.5	0.11	0.435	0.08	185	0.91	13.0	532	6898
1983	32.3	0.10	0.404	0.08	193	0.89	12.5	512	6416
1984	32.2	0.11	0.383	0.09	204	0.91	11.9	510	6083
1985	74.1	0.12	0.595	0.09	201	0.94	8.0	1177	9438
1986	129.6	0.09	1.028	0.08	226	0.93	7.9	2056	16310
1987	131.9	0.08	0.917	0.07	226	0.93	6.9	2093	14548
1988	147.8	0.10	1.423	0.08	227	0.91	9.6	2345	22590
1989	172.8	0.09	1.207	0.07	244	0.93	7.0	2742	19159
1990	215.2	0.22	1.335	0.18	216	0.89	6.2	3415	21193
1991	81.0	0.10	0.775	0.10	228	0.92	9.6	1285	12292
1992	43.5	0.11	0.441	0.07	229	0.87	10.1	690	6991
1993	135.6	0.10	0.615	0.08	214	0.96	4.5	2152	9768
1994	145.1	0.13	0.866	0.09	227	0.94	6.0	2302	13737
1995	173.4	0.13	1.218	0.11	227	0.96	7.0	2751	19325
1996	58.8	0.08	0.606	0.07	211	0.89	10.3	933	9612
1997	43.2	0.13	0.430	0.06	225	0.93	9.9	686	6824
1998	168.4	0.15	0.828	0.12	215	0.92	4.9	2672	13137
1999	238.3	0.24	1.734	0.20	226	0.92	7.3	3782	27524
2000	292.1	0.14	3.028	0.13	229	0.88	10.4	4636	48058
2001	308.4	0.11	3.419	0.12	227	0.90	11.1	4894	54257
2002	284.0	0.10	3.578	0.11	206	0.89	12.6	4508	56791
2003	654.5	0.16	5.766	0.10	201	0.90	8.8	10387	91512
2004	471.0	0.12	5.061	0.08	248	0.89	10.7	7475	80322
2005	344.6	0.08	5.406	0.07	278	0.94	15.7	5469	85792
2006	386.6	0.09	6.138	0.07	302	0.95	15.9	6136	97420
2007	314.6	0.06	5.487	0.06	304	0.94	17.4	4994	87078
2008	373.7	0.09	5.549	0.08	259	0.97	14.8	5932	88061
2009	370.5	0.12	6.134	0.10	196	0.92	16.6	5880	97351
2010	250.3	0.08	4.625	0.07	281	0.94	18.5	3973	73407
2011	172.7	0.10	3.482	0.10	298	0.96	20.2	2740	55264
2012	260.2	0.12	2.728	0.06	269	0.94	10.5	4130	43295
2013	256.1	0.10	3.201	0.08	309	0.98	12.5	4065	50801
2014	255.9	0.17	3.664	0.12	445	0.90	14.3	4061	58154
2015	624.1	0.12	3.923	0.09	495	0.95	6.3	9904	62255
2016	452.9	0.09	5.299	0.08	388	0.94	11.7	7187	84088
2017	261.4	0.12	3.725	0.11	379	0.93	14.3	4149	59120

Table A6.1(c) Georges Bank and Mid-Atlantic combined, without DSENLS scallops.

Year	Rel.Ab num/tow	CV	Rel.Bms kg/tow	CV	Ntows	Prop.Pos	MeanWt g	Abs.Ab millions	Ab.Bms mt
1979	57.6	0.26	1.108	0.23	274	0.91	19.3	1819	35183
1980	57.2	0.14	0.660	0.09	285	0.89	12.6	1710	19692
1981	44.7	0.10	0.690	0.09	249	0.88	16.4	1261	19486
1982	76.7	0.33	0.702	0.19	303	0.88	9.6	2608	23305
1983	44.8	0.13	0.569	0.12	319	0.88	13.6	1402	17808
1984	35.3	0.08	0.456	0.07	332	0.89	14.3	1047	13597
1985	68.8	0.09	0.646	0.09	355	0.92	10.8	2007	19011
1986	124.0	0.07	0.927	0.06	379	0.92	9.6	3501	26143
1987	126.8	0.09	0.977	0.08	396	0.90	9.3	3712	28771
1988	126.5	0.08	1.188	0.07	402	0.86	11.1	3634	34092
1989	125.3	0.08	0.911	0.06	364	0.88	9.2	3548	25814
1990	201.7	0.16	1.286	0.14	391	0.85	8.3	5830	37221
1991	157.8	0.27	1.039	0.15	404	0.91	7.7	4963	32029
1992	125.4	0.35	1.089	0.33	400	0.88	9.3	3990	34483
1993	103.6	0.10	0.558	0.08	378	0.92	7.2	2905	15661
1994	102.4	0.11	0.695	0.08	404	0.90	8.6	2864	19408
1995	146.6	0.11	1.031	0.09	403	0.92	8.7	4388	30858
1996	91.3	0.13	1.048	0.13	382	0.90	12.3	2788	32167
1997	63.3	0.11	0.991	0.13	415	0.91	15.8	1978	31708
1998	218.2	0.17	2.130	0.26	410	0.90	11.7	6318	62441
1999	218.8	0.16	2.441	0.13	399	0.95	12.8	6445	73775
2000	498.2	0.20	4.911	0.15	393	0.89	11.3	14633	144066
2001	352.0	0.09	4.771	0.10	435	0.93	15.1	10454	143083
2002	305.9	0.09	5.083	0.10	420	0.91	18.3	9006	151518
2003	490.9	0.12	6.324	0.09	408	0.92	14.5	14226	188919
2004	393.0	0.09	6.554	0.08	466	0.91	18.5	11434	195754
2005	310.3	0.07	6.055	0.06	621	0.95	20.8	9357	186816
2006	314.8	0.08	6.269	0.07	538	0.95	21.2	9394	192211
2007	275.6	0.06	5.255	0.05	667	0.95	20.0	8447	163170
2008	351.2	0.07	5.980	0.06	498	0.97	18.2	10737	185737
2009	367.1	0.10	6.002	0.07	410	0.95	18.0	11377	185731
2010	321.0	0.13	5.860	0.06	549	0.95	19.3	10380	189244
2011	219.0	0.08	4.915	0.06	523	0.96	23.8	6687	151051
2012	245.0	0.09	3.702	0.06	493	0.96	16.7	7618	120238
2013	291.0	0.12	3.808	0.10	659	0.96	14.5	8481	111154
2014	370.5	0.30	3.749	0.10	569	0.90	10.1	12500	120798
2015	533.9	0.12	4.129	0.12	673	0.95	7.7	18880	157020
2016	327.5	0.09	4.234	0.07	698	0.92	12.9	10543	142262
2017	253.8	0.10	4.005	0.10	659	0.91	15.8	9706	158659

Table A6.2. SMAST drop camera survey estimates. Note that the 2017 survey used high resolution digital cameras whereas 2003-15 used standard definition video cameras.

	Year	Density m^{-2}	CV	Nsta	Area.Surv km^2	Number millions	MeanWt g	Biomass mt
(a) Georges Bank								
	2003	0.147	0.08	929	28677	4213	27.8	117140
	2004	0.122	0.12	935	28863	3513	30.6	107320
	2005	0.116	0.11	902	27844	3235	30.4	98331
	2006	0.110	0.11	939	28986	3177	35.1	111401
	2007	0.142	0.11	912	28153	3989	27.8	110799
	2008	0.098	0.09	910	28091	2744	23.4	64238
	2009	0.157	0.11	899	27751	4351	22.7	98710
	2010	0.116	0.10	939	27937	3241	28.3	91583
	2011	0.147	0.12	918	28338	4169	24.4	101579
	2012	0.129	0.14	892	27535	3555	20.9	74375
	2014	0.643	0.23	1001	30900	19867	4.4	86696
	2015	0.661	0.25	1444	33022	21844	4.3	94497
	2017	0.910	0.21	1304	20852	18968	11.7	222427
(b) Mid-Atlantic								
	2003	0.483	0.17	804	24819	11995	8.7	103889
	2004	0.224	0.10	840	25930	5801	12.9	75032
	2005	0.210	0.12	864	26671	5598	14.0	78141
	2006	0.191	0.10	897	27690	5292	13.7	72312
	2007	0.179	0.09	941	29048	5202	14.5	75227
	2008	0.184	0.10	931	28739	5288	14.3	75356
	2009	0.134	0.06	928	28647	3844	15.1	57904
	2010	0.109	0.08	988	30499	3324	20.6	68363
	2011	0.066	0.06	1359	41951	2756	23.3	64305
	2012	0.111	0.08	1168	35999	3996	9.3	37187
	2014	0.152	0.07	1165	35963	5477	10.9	59606
	2015	0.580	0.22	940	28879	16856	6.1	102127
	2017	0.265	0.09	1500	32783	8691	12.6	109269
(c) Combined								
	2003	0.303	0.12	1733	53496	16208	13.6	221029
	2004	0.170	0.08	1775	54793	9313	19.6	182352
	2005	0.162	0.08	1766	54515	8834	20.0	176472
	2006	0.149	0.07	1836	56676	8468	21.7	183713
	2007	0.161	0.07	1853	57201	9192	20.2	186026
	2008	0.141	0.07	1841	56830	8032	17.4	139593
	2009	0.145	0.06	1827	56398	8196	19.1	156615
	2010	0.112	0.07	1927	58436	6565	24.4	159946
	2011	0.099	0.08	2277	70289	6925	24.0	165884
	2012	0.119	0.08	2060	63534	7551	14.8	111562
	2014	0.379	0.18	2166	66863	25344	5.8	146302
	2015	0.625	0.17	2384	61901	38700	5.1	196624
	2017	0.516	0.15	2804	53635	27659	12.0	331696

Table A6.3. Habcam survey estimates, excluding DSENLS scallops. The 2011 survey used the single camera "v2" system, and the 2012-2017 ones used the stereo camera "v4" system.

	Year	Number millions	CV	Biomass mt	MeanWt g
(a) Georges Bank					
	2011	7661	0.18	102720	13.41
	2012	9181	0.08	93440	10.18
	2013	7933	0.06	49483	6.24
	2014	11796	0.22	76819	6.51
	2015	18056	0.05	107281	5.94
	2016	21120	0.03	156315	7.40
	2017	13039	0.02	137117	10.52
(b) Mid-Atlantic					
	2012	4892	0.16	49076	10.03
	2013	4571	0.15	61248	13.40
	2014	6072	0.13	91540	15.08
	2015	15745	0.17	114287	7.26
	2016	10910	0.31	121781	11.16
	2017	6761	0.14	116170	17.18
(c) Georges Bank and Mid-Atlantic combined					
	2012	12553	0.12	142516	11.35
	2013	13752	0.08	110731	8.05
	2014	14006	0.07	168359	12.02
	2015	27541	0.13	221568	8.05
	2016	28966	0.12	278096	9.60
	2017	27881	0.04	253287	9.08

Table A6.4. Survey estimates of DSENLS scallops. "MeanGW", "MeanMW", "BmsGW", "BmsMw", are mean gonad weight, mean meat weight, biomass in gonad weight, and meat weight, respectively, all from Habcam data. Habcam and dredge numbers are restricted to those greater than 40 mm shell height; 2014, many of the scallops were smaller than this size. Gonad weights may be overestimated.

Year	Dredge mill.	CV	DropCam mill.	CV	Habcam mill.	CV	MeanGW g	MeanMW g	BmsGW mt	BmsMW mt
2014	N/S	N/S	850	0.85	201	0.02	2.2	1.8	438	353
2015	N/S	N/S	2261	1.00	11083	0.11	5.1	4.4	55926	48567
2016	5598	0.24	284	0.36	10664	0.04	5.9	4.2	63186	44803
2017	3152	0.26	11676	0.30	8251	0.03	8.5	7.6	70087	63054

Figures

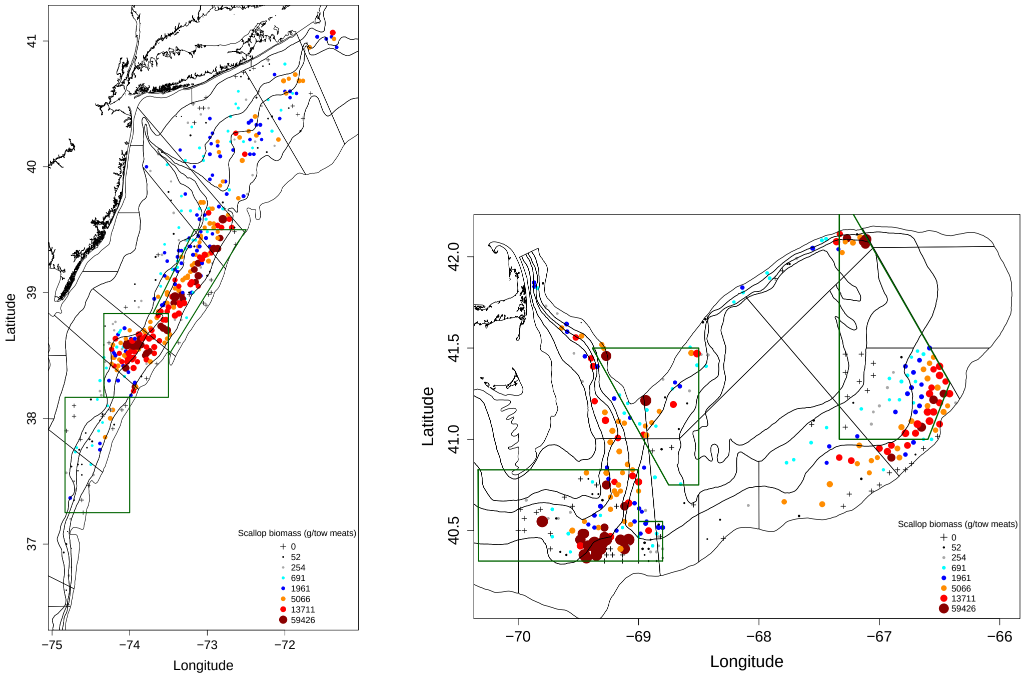


Figure A6.1. Dredge survey charts, showing 2017 catches, for Mid-Atlantic (left), and Georges Bank (right).

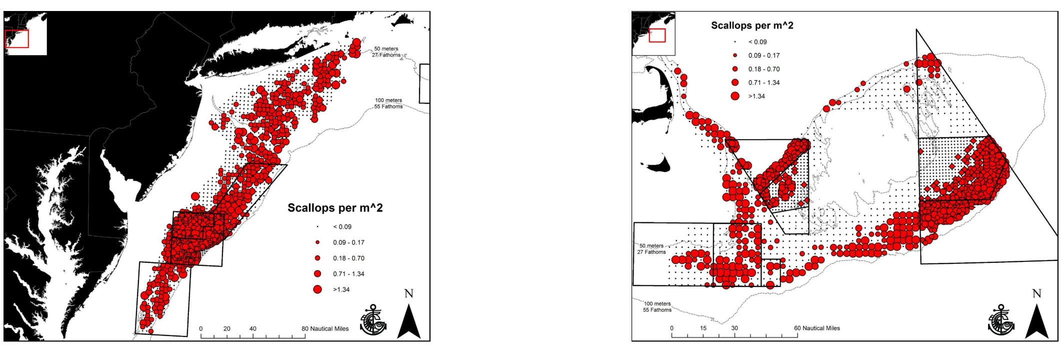


Figure A6.2. Drop camera survey charts, showing 2017 observed densities, for Mid-Atlantic (left), and Georges Bank (right).

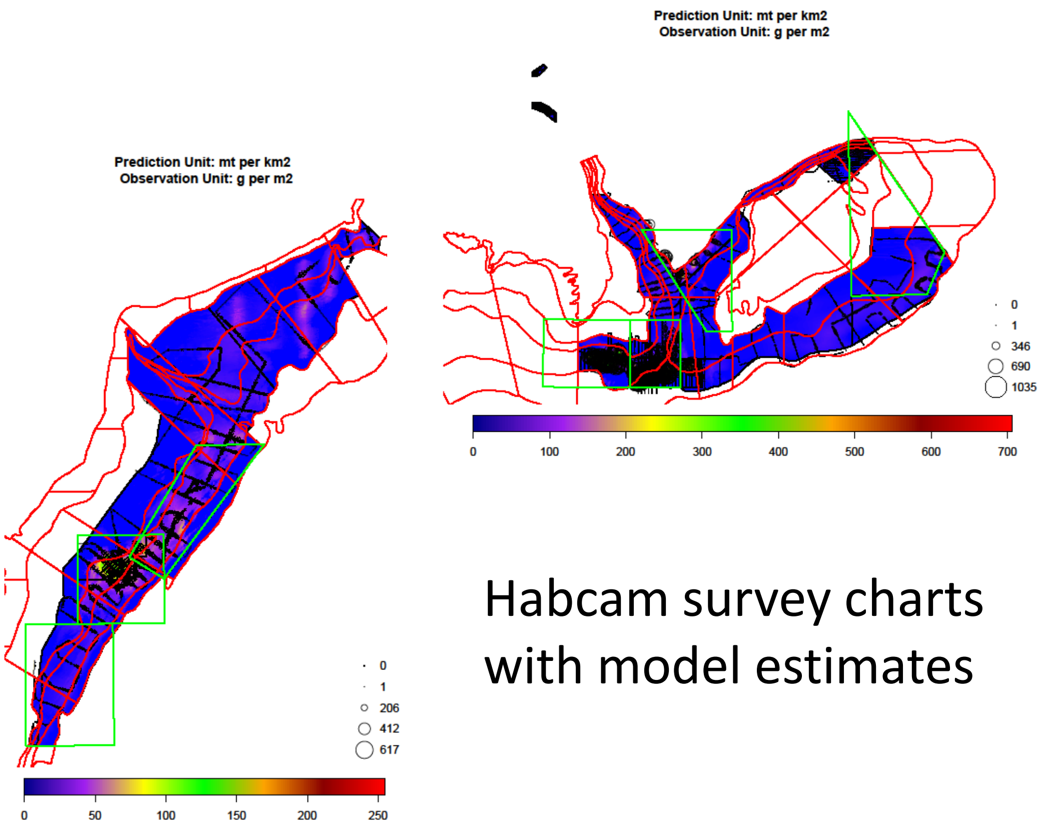


Figure A6.3. Habcam survey charts, showing survey track with observations of scallops (black dots), and geostatistical model estimated densities.

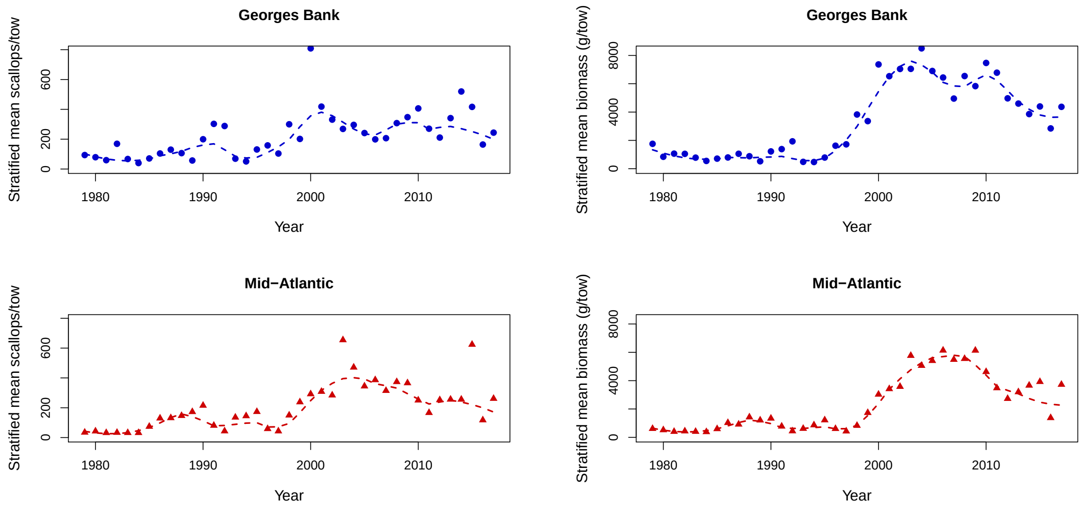


Figure A6.4. Dredge time series for numbers (left) and biomass (right) (dots), including lowess smoothers (dashed lines).

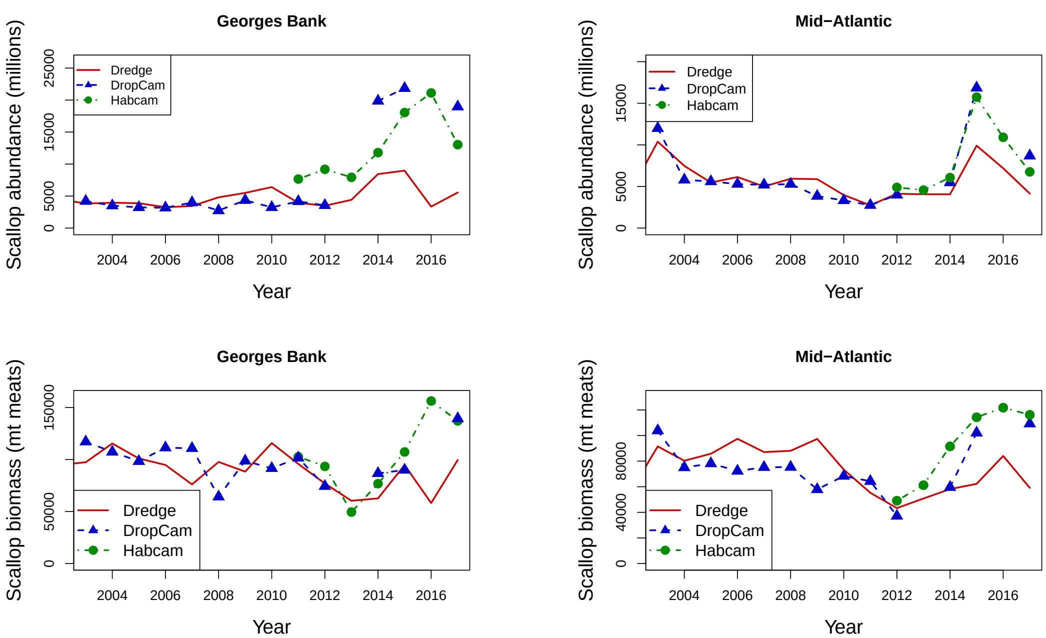


Figure A6.5. Comparison of survey estimates, 2003-2017, for Georges Bank (left), and Mid-Atlantic (right). Numbers are on the top row, and biomass on the bottom row.

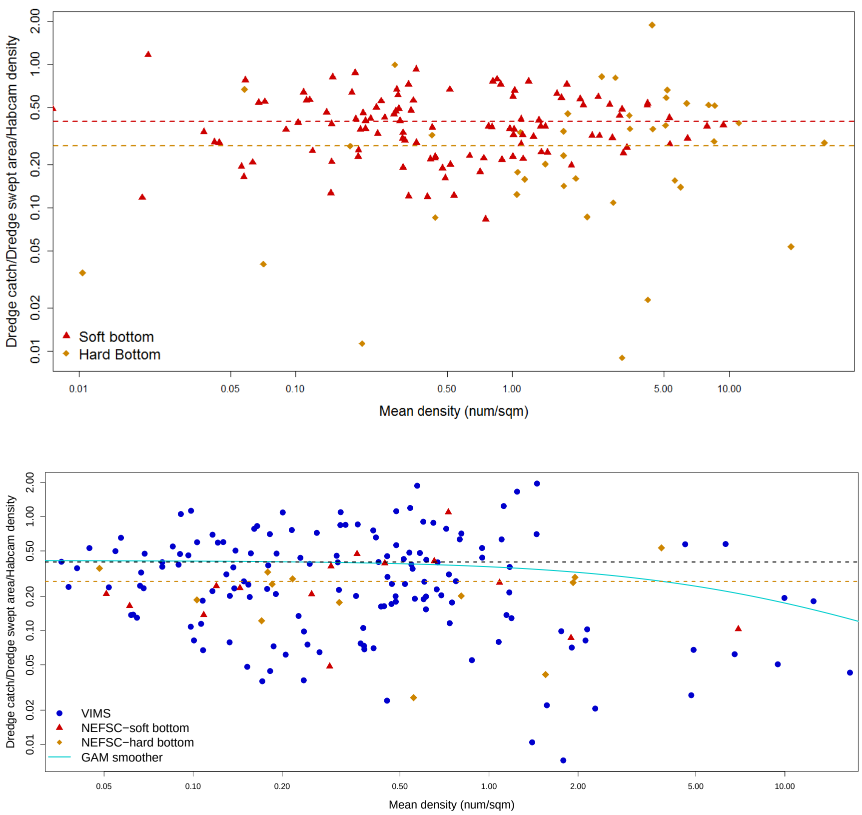


Figure A6.6. Dredge to Habcam density ratios. *Above:* Dredge tows conducted on the *R/V Hugh Sharp* in 2008-9, and *Below:* Dredge tows conducted on both commercial vessels and the *R/V Hugh Sharp* in 2016-17 as a function of density. The solid blue line is a GAM smoother; the upper dashed line is the predicted mean efficiency on soft bottom and the lower dashed line is the mean efficiency on hard bottom (Miller et al. 2018).

7 Gulf of Maine (TOR 3)

The sea scallop population and fishery in the Gulf of Maine are discussed in Appendix A3.

8 Environmental and Ecological Factors (TOR 4)

Environmental and ecological factors that may affect scallops include food supply (phytoplankton), temperature, pH, predators such as sea stars and decapods, invasive species, and parasites and diseases.

There is strong evidence that food supply affects growth and reproductive output of sea scallops (MacDonald and Thompson 1985,1986ab, MacDonald et al. 1987). Phytoplankton supply declines with increasing depth (Shumway et al. 1987), which explains observed declines in growth, and weight at size (Langton et al. 1987, Barber et al. 1988, Hart and Chute 2009b, Hennen and Hart 2012). For survey data, these effects are taken into account by using shell height to meat or gonad weight relationships that depend on depth (Appendices A1 and A2). The SAMS model uses area-specific growth and allometric relationships, based on data from surveys and shell growth analysis. For example, the relatively deep water Hudson Canyon area is slower growing with smaller meats at size than average, and the shallow inshore Mid-Atlantic area is faster growing with larger meats at size. The CASA and SYM models, which operate on a region-wide level, cannot explicitly take geographic variations in growth and weights into account. However, the different growth periods built into the CASA model can reflect changes in growth and weights at size over time, possibly due to environmental conditions (Appendices A1 and A2), and also due to shifts of the distributions of scallops to deeper or shallower areas.

Climate change and ocean acidification (OA) may affect scallop populations. In particular, there have been several studies demonstrating that OA can reduce the growth and survival of larval scallops (Talmage and Gobler 2009, White et al. 2013, Andersen et al. 2013), and swimming ability (Schalkhausser et al. 2013). Cooley et al. (2015) used a forward projection model similar to a non-spatial verion of SAMS together with forecasts of future ocean warming and acidification. They predicted that short term, warming may increase scallop growth and yields, consistent with observations presented in this assessment, but long-term (past 2050), OA will reduce yields, due to reduced larval survival and adult growth.

Predators can affect scallop distribution and mortality. In particular, the sea star *Astropecten americanus* appears to be reducing or excluding sea scallops from the deeper water of the Mid-Atlantic (Figure A8.1, Hart 2006, Shank et al. 2012). This sea star consumes a wide variety of small invertebrates, including early post-settlement sea scallops (Franz and Worley 1982). *A. americanus* appear to be limited by cold water temperatures, and for that reason, are only common in areas where winter minimum bottom temperatures remain above 5° C (Franz et al. 1981). *A. americanus* densities have been monitored in dredge surveys since 2000, and there is evidence that their distribution is expanded northward and inshore, likely due to warming temperatures (Figure A8.2).

Cancer spp. crabs (*Cancer irroratus* and *C. borealis*) are important predators on juvenile sea scallops (shell heights $<$ 90 mm, Elner and Jamison 1979, Nadeau et al. 2009), and may be agents of density dependence. 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, primarily 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 were greater when scallops are more common than alternative prey species, and increase with increasing scallop density (Barbeau et al. 1998, Wong and Barbeau 2005). This is accounted for in this assessment by directly estimating juvenile natural mortality by year.

The invasive colonial tunicate *Didemnum vexillum* has been observed in portions of Georges Bank and Gulf of Maine. This tunicate can rapidly spread over gravel/cobble substrate, often completely covering it. Scallop larvae cannot settle on *D. vexillum* (Morris et al. 2009), so this tunicate turns preferred settlement substrate into unavailable habitat for juvenile scallops. Consistent with this, Habcam survey data indicate that sea scallops are less common in areas dominated by *D. vexillum* (Figure A8.3, Kaplan et al. 2017). Additionally, this tunicate is more common in fished areas than in an adjacent closed area. This may be due to fishing activity dispersing the tunicates by both direct transport and fragmentation (Morris and Carman 2012). *D. vexillum* appears to be restructuring the benthic invertebrate community in areas that it dominates (Kaplan et al. 2018). Fortunately, *D. vexillum* has been observed to date only in relatively small portions of Georges Bank, mainly on the northern edge near the Hague line border with Canada.

There were several presentations during the Working Group meetings regarding parasites, most particularly gray meat disease and nematode infections. Gray meats have been most commonly observed on Georges Bank, and in particular in Closed Area I (Levesque et al. 2016). They have been associated with infections with Apicomplexa protists (Inglis et al. 2016), and may have contributed to the increased natural mortality observed in the Georges Bank closed areas during 2011-13. The variable natural mortality included in the Georges Bank Closed Areas model is one method to account for this.

Since 2015, a substantial portion of the scallops in the southern Mid-Atlantic have been infected with larvae of the nematode *Sulcascaris sulcata* (Figures A8.4, A8.5). They create brown or orange lesions in the scallop meat, caused by the immune response in the scallops. The adult nematode lives in the gastrointestinal track of sea turtles, primarily loggerheads, who acquire the nematodes by consuming infected scallops. These nematodes are associated with scallop and sea turtle populations around the world (Lichtenfels et al. 1978, 1980; Lester et al. 1980). It is likely that this outbreak of nematodes was due, at least in part, to the large 2013 scallop year class in the Mid-Atlantic, which may have increased the amount of scallops consumed by the turtles. Increases in the loggerhead turtle population off the northeast US may also be a contributing factor. It is unclear whether infections by the nematodes increases the natural mortality of the scallops. Lightly infected scallops do not appear to be seriously negatively affected by the nematodes, but heavy infections may increase the scallops' mortality. However, the nematodes have had a clear effect on the fishery, as heavily infected scallops are difficult to market (even though there is no public health risk). For example, there was almost no fishing in the Delmarva area near the southern boundary of the fishery in 2017 and also little in 2015, primarily due to the high nematode infection rate there (Figure A5.8)

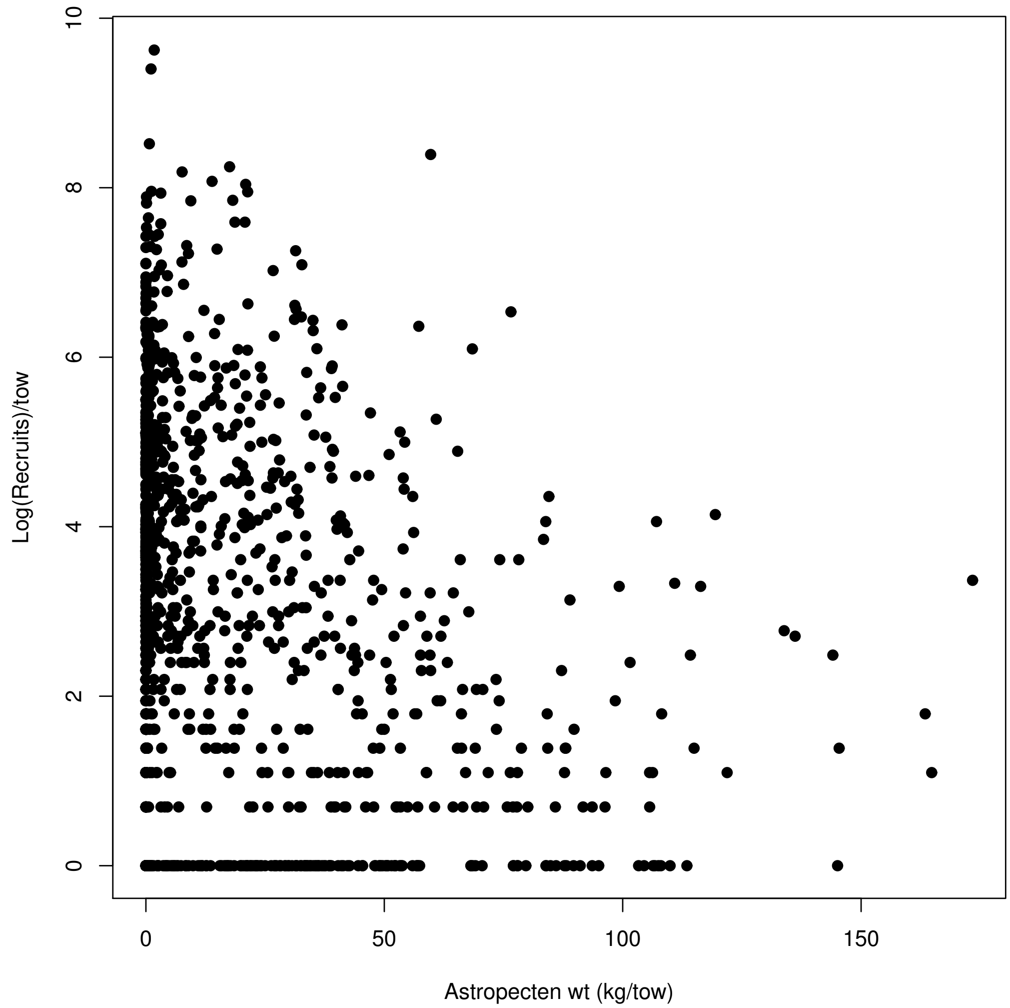


Figure A8.1. Relationship between *Astropecten americanus* biomass and scallop recruitment ($\log(x+1)$ transformed) in the Mid-Atlantic Bight, based on dredge data from 2000-2017.

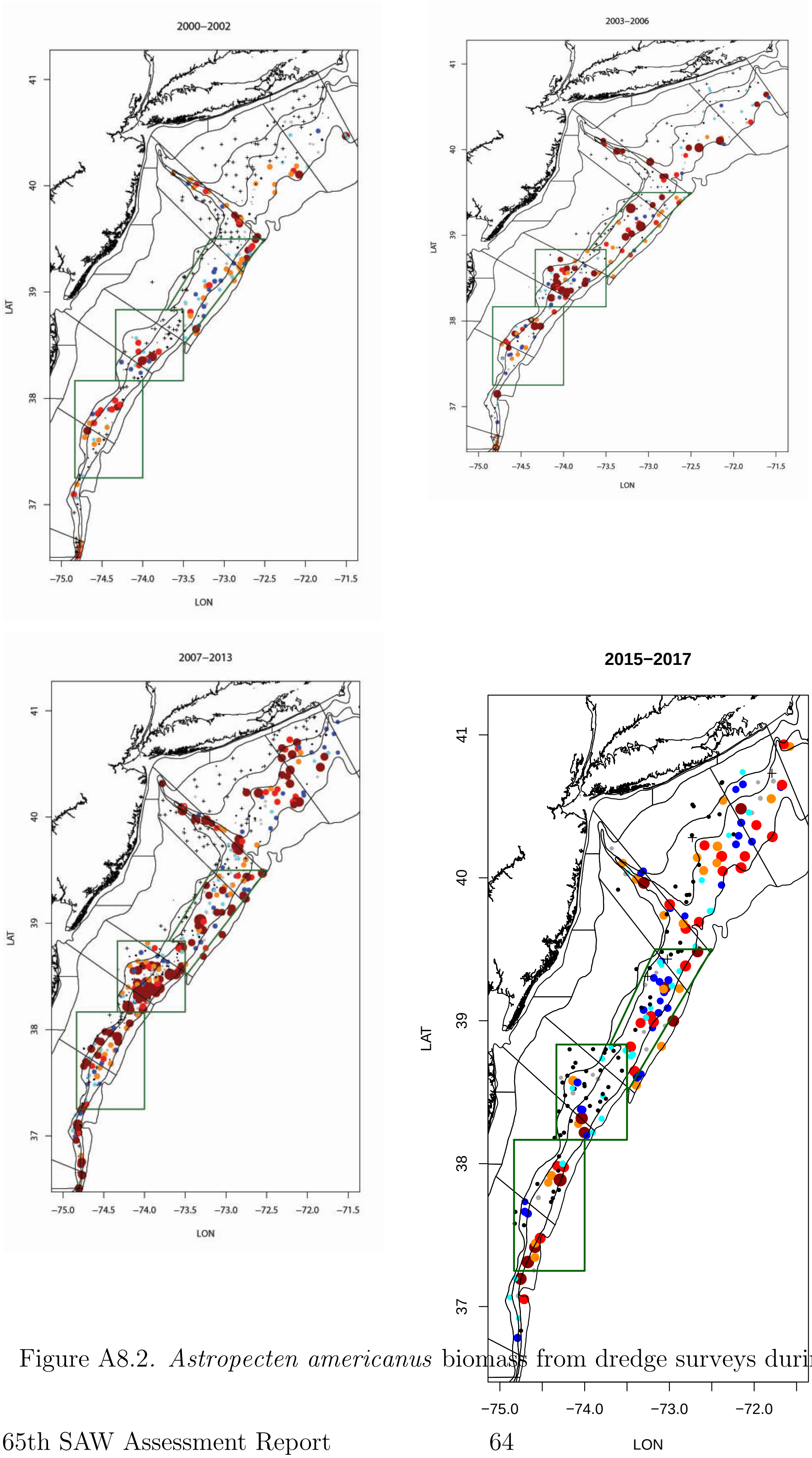


Figure A8.2. *Astropecten americanus* biomass from dredge surveys during four periods.

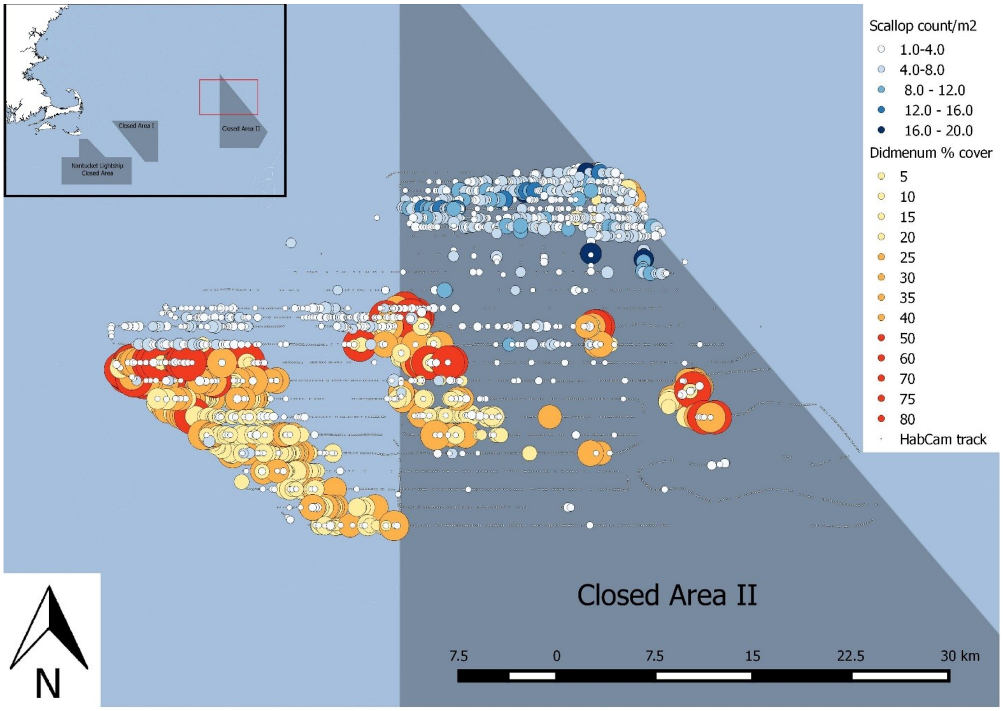


Figure A8.3. Chart of sea scallop (blue dots) and *Didemnum vexillum* (yellow to red dots) observations, from an intensive 2012 Habcam survey of the northern edge of Georges Bank, from Kaplan et al. 2017.

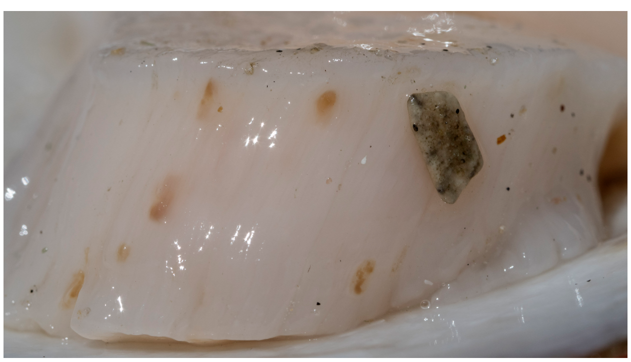


Figure A8.4. A scallop meat, taken from the Elephant Trunk area in the Mid-Atlantic in 2017, heavily infected with nematode larvae, as indicated by the numerous dark orange lesions.

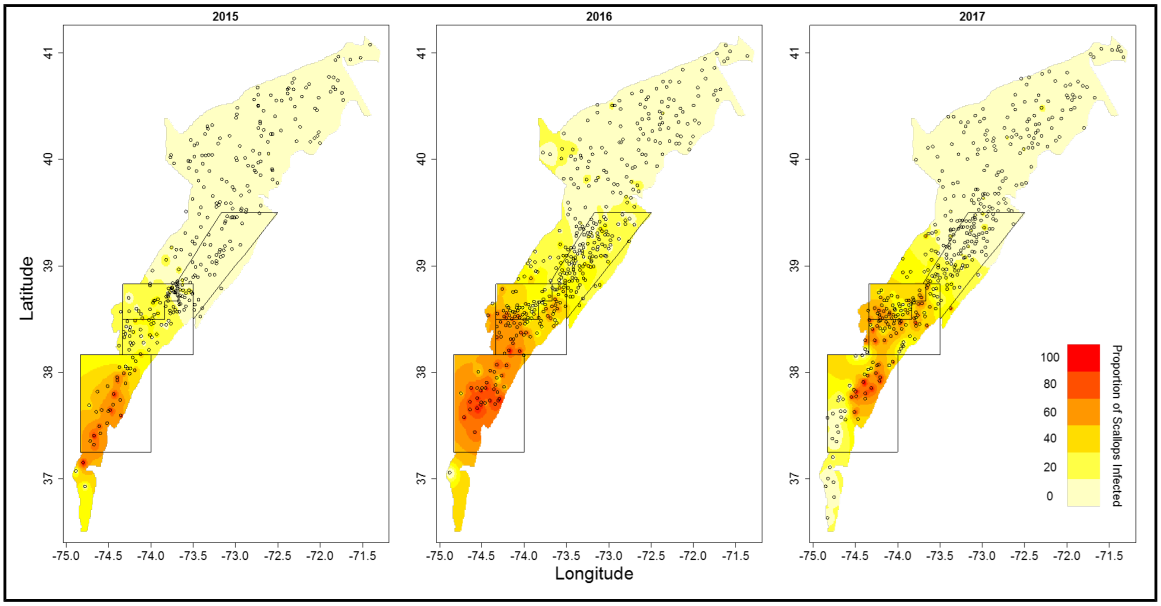


Figure A8.5. Prevalence of larval nematode *Sulcascaris sulcata* infection in scallops in the Mid-Atlantic Bight, based on a GAM model using observations from dredge surveys (Rudders and Roman 2018). The colors indicate predicted proportions of scallops infected, and the circles indicate dredge station locations where nematode sampling occurred.

9 Fishing Mortality, Biomass, and Recruitment Estimates (TOR 5)

Model Formulation

A catch-at-size-analysis (CASA, Sullivan et al. 1990) has been used as the primary assessment estimation model for US sea scallop assessments since 2007 (NEFSC 2007, 2010, 2014). It performed well in simulation testing using SAMS as the operating model (NEFSC 2007;

Hart et al. 2013). A detailed description of this model can be found in Appendix A6.

Prior to 2014, Georges Bank sea scallops were assessed as a whole. This required the used of domed fishery selectivity patterns during periods when there was little or no fishing in the closed areas . During these times, large scallops accumulated in the closures, which caused fishing mortality on large individuals, which were mainly in the closed areas, to be less than on intermediate sized ones (Sampson and Scott 2011). 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. Thus, as in NEFSC (2014), separate models were used for Georges Bank open and closed areas for this assessment. As in previous assessments, scallops in the Mid-Atlantic were assessed using a single CASA model.

All three CASA models (Georges Bank open, Georges Bank closed, and Mid-Atlantic) were run from 1975-2017. Shell heights were modelled with 5 mm shell height bins. Previous assessments began modeling the population at 20 mm, but because we were interested in modeling juvenile natural mortality, this assessment modelled the population starting at 0-5 mm, so that we can estimate recruitment at age 1, instead of age 2 as in previous assessments. However, as in previous assessments, only scallops larger than 40 mm were used in tuning because smaller scallops are not fully selected by any of the surveys. The lined dredge, Habcam, and SMAST digital camera surveys were assumed to have flat selectivity for scallops 40+ mm. Selectivities of the large drop camera, unlined dredge, and winter trawl surveys were fixed at experimentally determined values (NEFSC 2014).

Population shell height/meat weight conversions used parameters estimated from 2001-2017 research vessel survey data (A4.2). Fishery meat weights were adjusted based on estimated seasonal anomalies and the seasonal distribution of landings in that year (Appendix A2). Commercial shell height (size composition) data for 1975-1984 was from port samples, and 1992-2017 data were from sea samples (observers). The final (plus) group included L_∞. The meat weights for the plus group bin in a given year was 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 A9.1).

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 NEFSC (2007 and 2010), transition matrices were derived directly from shell increment data. This has the advantage of not assuming a specific form of growth, but it requires a large amount of data, and the modelled growth of sizes with few observations may not be reliable. As in NEFSC (2014), growth matrices were constructed in this assessment based on von Bertanlaffy growth parameters and their variances among individuals, estimated from growth increment data using mixed-effects models (Hart and Chute 2009b, Table A4.1). The matrices were constructed by drawing L_∞ and K values from independent normal distributions with means and variances among individuals estimated by the mixed-effects 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 prior to 2014. There is strong evidence that growth has changed over time (Table A4.1, Appendix A1). To account for this, several growth transition matrices were used to represent growth in different time periods, as in NEFSC (2014).

In the previous assessments, scallop abundance estimates from the CASA model trended below the surveys when strong recruitment was observed in the surveys (NEFSC 2014). This suggests that natural mortality of juveniles may increase at high density. If this were the case, CASA model estimates would be below the surveys for those years because observations of the strong year class in subsequent years would indicate fewer scallops than would be expected based on observed declines in the year class that cannot be accounted for by fishing. High natural mortality on large year classes of juveniles that are not captured in the model would likely induce retrospective patterns as has been observed in previous assessments, where estimates of strong year classes and abundance would decline as more years of data were added.

There is some experimental evidence of higher natural mortality on juvenile sea scallops especially at high densities (see Section 8). *Cancer* crabs are potential agents of density dependence in juvenile sea scallops as they primarily consume scallops less than 70 mm and rarely eat scallops greater than 90 mm (Elner and Jamieson 1979; Lake et al. 1987, Hart and Shank 2011, Bethany et al. 2016).

In order to differentially model juvenile and adult natural mortality, the estimation of total natural mortality in the CASA model in this assessment is calculated as the sum of juvenile and adult mortality components:

$$M_{(L,y)} = Ju_y\alpha_L + Av_y(1 - \alpha_L), \tag{A9.1}$$

where J and A are the mean natural mortality parameters for all years (y) and sizes (L) for juveniles and adults, and u and v are the year-specific deviation parameters for juveniles and adults, respectively, and α_L is a descending logistic function based on size. The logistic function is used to partition natural mortalities between juveniles and adults, thus determining how the natural mortality changes with size. Juvenile and adult natural mortality were estimated separately as discussed above for the Georges Bank open and Mid-Atlantic stocks but not the Georges Bank closed stock due to a lack of evidence of high juvenile natural mortality events in this area. Instead, natural mortality was estimated by year for all size classes in this model.

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, large drop camera, SMAST digital 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 0.1-0.15 for SMAST and the dredge survey and 0.1 for Habcam. The CV for Habcam is the smallest because it is expected to give the most accurate scaling (NEFSC 2014).

For use with priors, the dredge survey was expanded to survey abundance assuming flat selectivity, experimentally derived estimates of capture efficiency, and best estimates of stock area and areas swept by the dredge tows (Miller et al. 2018). large drop camera survey data were expanded after using the experimentally derived selectivity curve to adjust for reduced selectivity of small scallops. After this adjustment, large drop camera abundance and size data were expanded assuming flat selectivity and 100% capture efficiency. Expansion of the Habcam and SMAST digital camera surveys assumed 100% detectability of scallops >40 mm 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 (expanded by swept area and assumed gear efficiency), N is abundance available to the survey and q (with expected value 1) is the catchability parameter. Relatively high estimates of q indicate relatively low estimated abundance and vice versa because abundance $N = \frac{I}{q}$.

The catchability parameters estimates described above could, in principal, be larger or smaller than 1 but beta distributions used in CASA do not allow values larger than 1. Moreover, the parameter is estimated as a symmetrical beta distribution so that the probability of being slightly larger or smaller than the expected value was the same. This is accomplished 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 for catchability by 0.5 with no other effect on model estimates.

CVs for survey data and effective sample 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. Historical retrospective, profile likelihood, and sensitivity analyses were also used to describe uncertainty.

CASA Model for Georges Bank Closed Areas

The model was tuned to the lined dredge survey (1979-2017), the SMAST large video camera survey (2003-2012, and 2014), the SMAST digital camera survey (2015 and 2017), the Habcam survey (2011-2017), and the unlined dredge survey (1975 and 1977). Two fishery selectivity periods were used, for the years prior and after these areas were closed.

Natural mortality was not estimated in previous assessments and was fixed at 0.16 (0.24 on the plus group) in NEFSC 2014. In this assessment, mean total natural mortality and the annual deviations of total natural mortality were estimated in the model. Estimation of juvenile natural mortalities were explored during preliminary model runs and were found not to be significantly different than adults for this stock. The estimated mean natural mortality is 0.23; peak M was estimated at 0.46 for 2011 followed by 0.37 for 2012 (Figure A9.2). These years coincide with observations of gray meat disease, in particular in Closed Area I (Levesque et al. 2016). Incidental fishing mortality was set at 0.11 times fully recruited fishing mortality for each size group as described elsewhere in this report.

Five growth matrices were used in the model: matrix 1 for 1993-1996, matrix 2 for 2000-2006, matrix 3 for 1975-1992 and 1997-1999, matrix 4 for 2007-2011 and 2014-2017, and matrix 5 for 2012-2013. The matrices are ordered by slow to fast growth, and were estimated using shell growth increments from those years. The slow growing strong 2012 year class is underrepresented in the shell sample collections so that matrix 5 overestimated growth for 2014-2017. Therefore, matrix 4 was used instead of matrix 5 for this period.

The basecase model fits survey trends reasonably well (Table A9.1, Figures A9.3 to A9.17). Mean estimated posterior efficiencies for the dredge, large drop camera, Habcam, and SMAST digital camera surveys ranged from 0.47-0.56 (compared to the prior mean of 0.5; Figure A9.11). When the 2014 CASA model configuration (M fixed at 0.16) was used, the estimated abundance was below survey abundance for 1998-2010, which is similar to the model results in the 2014 assessment (NEFSC 2014; Figure A9.18). The estimated recruitment was highest in 2013 when the strongest recruitment event was observed in this area (Figure A9.13). Model estimates of shell heights generally fit the data well, except for the SMAST digital camera survey (Figures A9.4 to A9.9).

Model estimates of fishing mortality were consistent in scale with the Beverton-Holt (1956) length-based equilibrium estimator, which is used as a diagnostic for the CASA model (Figure A9.17). Estimated fishing mortalities increased from 1975-1993 (Figures A9.13 and A9.15) and were low or zero afterward. This resulted in a dramatic increase in biomass during 1994-2004, and a build-up of large scallops (Figures A9.13 and A9.14). Fishery selectivity since 1999 has been shifted strongly towards large scallops (Figures A9.12 and A9.15), even more so than in the open areas, because scallopers often target the largest market category (U-10s) which typically command a price premium. Maximum abundance occurred in 2016 (5,519 millions) and and biomass in 2017 (150,951 mt), primarily driven by the strong 2012 year class (Figure A9.16).

The model for Georges Bank closed areas has a moderate retrospective pattern with a 7 year peel (Mohn's $\rho = 0.45$ for biomass and -0.12 for fishing mortality; Figure A9.19), where estimates of biomass decrease as more years of data are added, especially for 2010-2013. The retrospective pattern disappeared after 2014 as more data on the strong 2012 year class became available.

CASA Model for Georges Bank Open Areas

The model was tuned to the same surveys as used for Georges Bank closed area. There were three commercial fishery selectivity periods: 1975-1998, 1999-2004, and 2005-2017.

As was the case for Georges Bank Closed, natural mortality was set at 0.16 (0.24 on the plus group) in the 2014 assessment (NEFSC 2014). In this assessment, the mean natural mortality for juveniles and adults were not estimated and set at 0.2 based on the natural mortality estimated for the Georges Bank closed stock. The value of 0.2 was used instead of 0.23 because a portion of the estimated natural mortality in the closed areas may be aliasing for other processes, including poaching, highgrading practices, and effects of spatially heterogeneous fishing mortality (Hart 2001; Truesdell et al. 2016). Likelihood profiles did not indicate that plus group mortality was higher than smaller sizes, so M was set at 0.2 for the plus group as well. The annual deviations of juvenile natural mortality were estimated in the model. The logistic curve used to partition juvenile and adult natural mortalities were not estimated and set at $L_{50} = 65$ mm (equivalent to average size at age 2.5) and slope=0.1 (Figure A9.20). The estimated juvenile natural mortalities were around 0.2 for most years and sizes, except in 2014 when the natural mortality for small size groups (0-65 mm) was estimated at 1.42 (Figure A9.21). The agrees with observations of a large year class in 2014, most particularly in the dredge survey, much of which was not observed in subsequent years. Incidental fishing mortality was set at 0.11 times fully recruited fishing mortality for each size group.

As in the Georges Bank closed stock, five discontinuous growth periods were used for Georges Bank open stock (period 1 from 1993-1996; period 2 from 2000-2006 period 3 from 1975-1992 and 1997-1999; period 4 from 2007-2011 and 2014-2017; and period 5 from 2012-2013).

The resulting basecase model generally fit survey trends well except for recent years when high abundances were observed (2013-2014; Table A9.2, Figures A9.22 to A9.36). Mean estimated posterior efficiencies were 0.57 for the dredge and large drop camera, 0.83 for Habcam, and 0.38 for SMAST digital camera surveys so that the dredge, large drop camera and Habcam surveys were above the prior mean of 0.5 whereas the SMAST digital camera was below (Figure A9.30). When the 2014 CASA model configuration (M fixed at 0.16) was used, the underestimation of abundance in recent years was worse (Figure A9.37). The underestimation of abundance in those years is likely due to a discrepancy between survey observations and the underrepresentation of slow-growth scallops in the model. This also likely caused the model to partition the one-year recruitment event into two years as the estimated recruitment was highest in 2013 but 2014 was almost as high (Figure A9.32). Model estimates of shell heights generally fit the data well, except for the SMAST digital camera survey (Figures A9.23 to A9.28).

Model estimates of fishing mortality were consistent in scale with the Beverton-Holt (1956) length-based equilibrium estimator (Figure A9.36). Fishery selectivity strongly shifted over time toward larger shell heights, reflecting changes in gear and targeting practices (Figure A9.31). The size at 50% selectivity moved from about 75 mm before 1999, to 90 mm during 1999-2004, and 100 mm since 2005 (Figure A9.31). Biomass and abundance generally declined and fishing mortality increased during 1975-1995, with these trends reversing themselves after 1995 (Figure A9.32). As a result of the changes in selectivity and fully recruited fishing mortality, survival to large shell heights has increased substantially in recent years (Figure A9.33). Estimated biomass and fully recruited fishing mortality for 2017 were 21,119 mt and 0.13, respectively.

The Georges Bank Open runs show moderate retrospective patterns with a 7 year peel (Mohn's $\rho = 0.54$ for biomass and -0.29 for fishing mortality; Figure A9.38), where estimates of biomass decrease and fishing mortality increase as more years of data are added.

CASA Model for combined Georges Bank Closed and Open Areas

Biomass and fishing mortality estimates for Georges Bank open and closed combined (Figure A9.39) show generally decreasing biomass and increasing fishing mortality from 1975-1992, with peak fishing mortality of 1.44 in 1991, and minimum biomass of 10,908 mt in 1985. Fishing mortality since 1995 has generally been between 0.2 and 0.4. Biomass increased substantially between 1994 and 2003 and has further increased since 2014. Estimated 2017 biomass and fishing mortality were 172,070 mt and 0.06, respectively, the highest and lowest values of the entire time series. Retrospective scores for the combined Georges Bank sea scallop stocks were similar to the scores for individual regions (Mohn's $\rho = 0.45$ for biomass and -0.28 for fishing mortality; Figure A9.40).

CASA Model for Mid-Atlantic Areas

Surveys used in the Mid-Atlantic CASA model were all the surveys used for the Georges Bank model plus the NEFSC winter bottom trawl survey, which was conducted between 1992-2007. 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 modelled using a ascending logistic curve with parameters fixed at values estimated by the 2014 assessment (NEFSC 2014). Survey catchability 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 2005-2017. The first period was modelled as domed (double logistic) selectivity, due to indications in the data of higher mortality on intermediate sized scallops. The domed selectivity was likely caused by fishing effort concentrating on only a portion of the stock, with most large scallops outside the intensively fished area (Samson and Scott 2011). All the other periods were assumed to have logistic selectivity.

Natural mortality was set at 0.2 and 0.3 on the plus group in the 2014 assessment (NEFSC 2014). In this assessment, the natural mortality is estimated the same way as the Georges Bank open stock. The mean juvenile and adult natural mortalities were not estimated and set at 0.25 based on the natural mortality estimated for the Georges Bank closed areas together with the assumption that K/M is invariant. Likelihood profiles did not indicate that plus group mortality was higher than smaller sizes, so M was set at 0.25 for the plus group as well. The annual deviations of juvenile natural mortality were estimated in the model. The logistic curve used to partition natural mortalities for juveniles and adults were not estimated and set at $L_{50} = 70$ (equivalent to average size at age 2.5) and slope=0.1 (Figure A9.41). The estimated natural mortalities were around 0.25 for most years and sizes, except in 2003, 2004, and 2015 when high recruitment were observed (Figure A9.42). The average natural mortality for the small size groups (0-70 mm) was estimated as 0.62 in 2003, 0.46 in 2004, and 0.49 in 2015 (Figure A9.42); all of these years are associated with large juvenile year classes. Elevated natural mortality for these years is consistent with declines observed in surveys. The incidental fishing mortality was set at 0.06 times fully recruited fishing mortality for each size group.

Three discontinuous growth periods were used for the Mid-Atlantic stock: 1975-1977, 1987-2003, and 2006 (matrix 1, slow growth), 1978, 1983-1986, 2004-2005, and 2007 (matrix 2, moderate growth), and 1979-1982 and 2008-2012 (matrix 3, fast growth). No shells were collected in the Mid-Atlantic area after 2013 so growth matrices used for 2013-2017 were selected from the above three matrices (matrix 1: 2015-2017; matrix 2: 2014; matrix 3: 2013) based on apparent growth from surveys in those years.

The basecase model generally fit survey trends and size frequency data well but trended below survey estimates during and after strong recruitment events (Table A9.3, Figures A9.43 and A9.57). This was especially apparent after 2003 and 2015 when very strong year classes were observed in surveys (Figure A9.56). Mean estimated posterior efficiencies were 0.69 for the dredge and SMAST digital camera, 0.78 for large drop camera, and 0.6 for Habcam surveys (relative to the prior target 0.5; Figure A9.51), indicating that CASA abundance estimates were slightly lower than the survey abundance data on average (Figure A9.56). When the 2014 CASA model configuration was used instead (M fixed at 0.2), the underestimation of abundance was worse (Figure A9.58). The model correctly identified the two high recruitment events in 2002 and 2014 (Figure A9.53).

Model estimates of fishing mortality were consistent in scale with the Beverton-Holt (1956) length-based equilibrium estimator (Figure A9.57). 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 A9.52). By 2005-2017, only the plus group was fully selected (Figure A9.52). Model estimated fishing mortality on larger scallops generally increased during 1975-1995, reaching a maximum fully recruited fishing mortality of about 1.2 in 1995, and then declined (Figure A9.55). 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 (Figures A9.53 and A9.55). Abundance and biomass were relatively low during 1975-1998, rapidly increased from 1998-2003, and increased dramatically again in 2015 primary driven by the strong 2013 year class (Figure A9.56). Biomass and abundance declined during 2009-2012, primarily as a result of poor recruitment compared to previous years (Figure A9.53). In general, recruitment appears to have been substantially stronger since 1998 (Figure A9.53). Estimated biomass and fully recruited fishing mortality for 2017 were 145,265 mt and 0.17, respectively.

The Mid-Atlantic model showed a minor retrospective pattern with a 7 year peel (Mohn's ρ = 0.14 for biomass and 0.2 for fishing mortality; Figure A9.59). However, there is a tendency for the model to slightly overestimate biomass as more years of data are added.

Whole Stock Biomass, Abundance and Mortality

Biomass, SSB, 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 and Georges Bank open and closed (Table A9.4). For example, whole stock fishing mortality rates for each year were calculated as

$$F = \frac{C_{MA} + C_{Go} + C_{Gc}}{\bar{N}_{MA} + \bar{N}_{Go} + \bar{N}_{Gc}}, \tag{A9.2}$$

where C_{MA}, C_{Go}, and C_{Gc} are catch numbers for the Mid-Atlantic, Georges Bank open, and Georges Bank closed areas, respectively. Terms in the denominator are average fishable abundances during each year calculated in the CASA model as

$$\bar{N} = \sum_{L} \frac{N_L(1 - e^{-Z_L})}{Z_L} \tag{A9.3}$$

The simple ratio formula used to calculate whole stock F is an "exact" solution because the catch equation can be written as $C = F\bar{N}$. Whole stock variances were calculated assuming that estimation errors for Georges Bank open and closed, and the Mid-Atlantic were independent. In particular, variances for biomass, abundance and catch estimates were the sum of the variances for Georges Bank open and closed and Mid-Atlantic. CVs for the ratios estimating whole stock F were approximated by:

$$CV_F = \sqrt{CV_C^2 + CV_{\bar{N}}^2}, \tag{A9.4}$$

which is exact if catch number C_N and average abundance $\bar{N}$ 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.

As in the individual models, whole-stock fishing mortality generally increased from 1975-1992, then strongly declined during 1992-1995 and stayed low ever since (Figure A9.39). Whole stock biomass, abundance, and fishing mortality in 2017 were 317,335 mt meats (153,182 mt gonads), 14,171 million scallops and 0.12, respectively (Figure A9.39), not including the DSENLS scallops. The biomass in 2017 and the abundance in 2015 were the highest in the 1975-2017 time series (Figure A9.39). Retrospective scores for the entire sea scallop stocks were better compared to the scores for individual regions (Mohn's $\rho = 0.32$ for biomass and -0.21 for fishing mortality; Figure A9.60). Adding in the DSENLS scallop estimates (see Table A6.4), the total stock estimates are: 380,389 mt meats (223,269 mt gonads), and 22,422 million scallops.

The standard errors estimated by the CASA model in this assessment are too small and do not capture all of the underlying uncertainties (Figure A9.61). In particular, the uncertainties calculated by CASA are based on the assumptions underlying the model being exactly correct. The CASA model is in actuality a simplification of reality, and many processes are simplified. 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.

Likelihood Profile Analysis

Likelihood profiles were conducted on four sources of mortality: mean natural mortality, plus group natural mortality, catch, and incidental mortality. Likelihood profiles were constructed on mean natural mortality (both juveniles and adults) for Georges Bank open and Mid-Atlantic stocks but not for Georges Bank closed stock because natural mortality was estimated in that model. For both stocks, total negative log likelihood decreased when natural mortality increased (Figures A9.63 and A9.64). The survey trend, size frequency, and Q prior components of the likelihood (sum over all surveys) all tend to be lower at lower M values (Figures A9.63 and A9.64).

Another likelihood profile analysis was constructed for natural mortality of the plus group. Natural mortality for the smaller size groups was estimated the same way as in the basecase models. For the Georges Bank closed stock, the total negative log likelihood and the survey trend, size frequency, and Q prior components of the likelihood decreased slightly when plus group natural morality increased (Figure A9.62). However, for Georges Bank open and Mid-Atlantic stocks, total negative log likelihood and survey trend, size frequency, and Q prior components of the likelihoods increased with higher plus group natural mortality, except for the Q priors for Mid-Atlantic stock (Figures A9.63 and A9.64). Therefore, unlike the 2014 assessment where the plus group natural mortality was taken to be 1.5 times that of smaller size groups, plus group natural mortality the same as other adults in this assessment.

Likelihood profile analysis was also conducted for situations where catch was increased by 5%-40%. Total negative likelihood decreased with increasing catch for all three stocks (Figures A9.62 to A9.64). The survey trend, size frequency, and Q prior components of the likelihood were all tend to be lower at higher catch, except for the Q prior component for Georges Bank closed stock and the survey trend component for Georges Bank open stock (Figures A9.62 to A9.64).

The final set of likelihood profile analysis was constructed on incidental mortality. For the Georges Bank open stock, the total negative log likelihood and the survey trend, size frequency, and Q prior components of the likelihood decreased with increasing incidental mortality until about 0.2 (Figure A9.63). Total negative likelihood increased for Georges Bank closed stock and decreased for Mid-Atlantic stock with increasing incidental mortality (Figures A9.62 and A9.64). For both stocks, the likelihood components of survey trend and Q prior tend to be lower while size frequency was higher at higher incidental mortality (Figures A9.62 and A9.64).

Sensitivity Analysis

To test the sensitivity of the model outputs to key assumptions, CASA model runs were conducted with three alternative assumptions: removing recent dredge survey indices, doubling the CVs on the prior of survey efficiencies (loose priors), and estimating large drop camera survey selectivities in the model.

In the first sensitivity run, dredge indices from 2014-2017 were dropped for Georges Bank closed stock and 2015-2017 for Mid-Atlantic stock, based on evidence of reduced dredge efficiency for those years. Biomass estimates in last few years increased slightly in this case in both the Georges Bank closed and Mid-Atlantic models (Figure A9.65). Loose survey priors decreased biomass estimates in all three stocks, mainly in the last several years for Georges Bank open and Mid-Atlantic stocks and since 1995 for Georges Bank closed stock (Figure A9.65). The Georges Bank closed stock model run was unable to converge while estimating large drop camera selectivity. For the Georges Bank open stock, the small scallops (<45 mm) have reduced selectivity but relatively large scallops (>45 mm) are more fully selected compared to what was assumed in the basecase model (Figure A9.65). The large drop camera abundance indices converted using CASA model estimated selectivity was less consistent with other surveys in earlier years and more consistent in recent years, suggesting that the large drop camera selectivity might be changing over time in the Georges Bank open area (Figure A9.65). For Mid-Atlantic, the model estimated selectivities were higher for all size scallops compared to the selectivities used in the basecase model (Figure A9.65). The large drop camera abundance indices converted using CASA estimated selectivity are more consistent with indices of other surveys in the Mid-Atlantic area (Figure A9.65). Overall, estimating the drop camera selectivities in the model has little effect on the biomass estimates (Figure A9.65). Fishing mortalities were insensitive to all the alternative assumptions tested above (Figure A9.65).

Historical Retrospective Analysis

The current CASA model estimates can be compared to those from the last three benchmark assessments (SARC-45/NEFSC (2007), SARC-50/NEFSC (2010), and SARC-59/NEFSC (2014); Figure A9.67). The biomass estimates for this assessment tend to be higher while fishing mortalities tend to be lower than previous assessments for both Georges Bank and Mid-Atlantic stocks (Figure A9.67). These differences are more for the Georges Bank model during 1995-2003 (Figure A9.67).

The current CASA model estimates can be compared to the SARC-59 configuration updated through 2017 to the present model. The biomass and fishing mortality plots indicate modest differences between the two configurations for all three stocks (Figure A9.65). The retrospective patterns were higher when using 2014 CASA model configurations (Mohn's $\rho = 0.37$ for biomass and -0.24 for fishing mortality; Figure A9.68).

Tables

Table A9.1. CASA estimates for Georges Bank Closed Areas, excluding DSENLS scallops.

Year	Abundance millions	CV	SSB mt gonads	CV	Bms mt meats	CV	ExplBms mt meats	CV	F	CV
1975	705	0.11	6333	0.24	13526	0.11	12196	0.12	0.09	1.28
1976	615	0.10	6851	0.21	14739	0.10	13797	0.11	0.13	0.79
1977	584	0.09	6338	0.21	14090	0.09	13335	0.10	0.27	0.34
1978	577	0.09	5466	0.19	12412	0.09	11327	0.09	0.34	0.26
1979	388	0.09	3941	0.21	9359	0.09	8981	0.09	0.56	0.15
1980	443	0.10	3208	0.22	7413	0.10	6300	0.11	0.48	0.20
1981	397	0.11	2794	0.25	6575	0.11	5670	0.12	0.56	0.18
1982	358	0.12	2561	0.27	5944	0.12	5174	0.13	0.45	0.24
1983	294	0.14	2280	0.31	5283	0.14	4737	0.15	0.46	0.26
1984	357	0.13	2516	0.29	5531	0.13	4593	0.16	0.21	0.63
1985	352	0.12	2654	0.28	5944	0.13	5165	0.14	0.36	0.35
1986	492	0.11	2704	0.27	6261	0.12	4790	0.16	0.59	0.23
1987	539	0.12	2800	0.29	6534	0.12	4956	0.16	0.75	0.18
1988	757	0.12	3253	0.29	7600	0.12	5063	0.19	0.88	0.18
1989	976	0.09	4411	0.20	10131	0.09	6899	0.13	0.68	0.21
1990	920	0.08	3773	0.21	9686	0.08	7131	0.11	1.40	0.08
1991	876	0.09	3174	0.24	8162	0.09	5435	0.14	1.54	0.07
1992	795	0.15	3291	0.36	7891	0.15	5422	0.22	1.21	0.11
1993	1453	0.17	6046	0.38	13336	0.17	7959	0.29	0.34	0.50
1994	1379	0.16	9641	0.35	20248	0.17	17033	0.20	0.04	3.45
1995	1711	0.15	13698	0.31	28590	0.15	25215	0.17	0.00	0.00
1996	1757	0.14	17381	0.28	36310	0.14	33527	0.15	0.00	0.00
1997	1790	0.13	20408	0.26	42316	0.12	40128	0.13	0.00	0.00
1998	2413	0.12	24824	0.24	51831	0.12	35274	0.17	0.00	62.28
1999	2192	0.11	27507	0.23	58059	0.11	40224	0.16	0.09	1.22
2000	2681	0.11	29968	0.23	63250	0.11	44662	0.15	0.07	1.43
2001	2956	0.10	34118	0.21	71699	0.10	49519	0.15	0.01	6.98
2002	2756	0.09	37495	0.19	78581	0.09	56758	0.13	0.00	0.00
2003	2861	0.08	39467	0.17	82976	0.08	64866	0.10	0.00	0.00
2004	2515	0.07	38448	0.16	81802	0.07	66849	0.09	0.06	1.39
2005	2152	0.07	33923	0.15	73463	0.07	62431	0.08	0.14	0.57
2006	1723	0.08	27105	0.17	59839	0.08	51974	0.09	0.24	0.36
2007	1688	0.08	23035	0.18	49977	0.08	41640	0.10	0.15	0.61
2008	1597	0.08	22662	0.18	48206	0.08	38262	0.10	0.07	1.45
2009	1766	0.09	22954	0.18	48838	0.08	37693	0.11	0.05	2.00
2010	1877	0.09	22372	0.19	48643	0.09	35926	0.12	0.10	1.09
2011	1866	0.09	18506	0.21	42867	0.09	29657	0.13	0.23	0.44
2012	1701	0.11	15464	0.26	35688	0.11	22332	0.18	0.36	0.35
2013	1701	0.12	18013	0.24	37374	0.12	23355	0.19	0.11	1.22
2014	5064	0.10	30556	0.21	64481	0.10	29512	0.21	0.07	1.69
2015	5723	0.09	48077	0.18	101319	0.09	41507	0.21	0.00	29.09
2016	5896	0.08	63153	0.16	133582	0.08	76928	0.13	0.00	55.26
2017	5519	0.08	70664	0.17	150951	0.08	109162	0.11	0.05	2.06

Table A9.2. CASA Estimates for Georges Bank Open Areas.

Year	Abundance millions	CV	SSB mt gonads	CV	Bms mt meats	CV	ExplBms mt meats	CV	F	CV
1975	1425	0.03	12172	0.07	25818	0.03	23473	0.03	0.07	0.64
1976	1270	0.03	12322	0.07	26818	0.03	25023	0.04	0.15	0.29
1977	1103	0.04	10767	0.08	24072	0.04	23227	0.04	0.27	0.18
1978	1017	0.04	8938	0.09	20147	0.04	18808	0.04	0.31	0.16
1979	804	0.04	7031	0.09	16178	0.04	15195	0.04	0.41	0.12
1980	957	0.03	6055	0.09	13742	0.04	11952	0.04	0.38	0.14
1981	781	0.04	5071	0.08	11994	0.03	10582	0.04	0.56	0.08
1982	656	0.04	3617	0.10	8949	0.04	7953	0.04	0.77	0.06
1983	452	0.05	2769	0.12	6682	0.05	5934	0.05	0.65	0.08
1984	375	0.05	2341	0.14	5456	0.06	4877	0.07	0.51	0.13
1985	448	0.06	2123	0.14	4964	0.06	3986	0.08	0.63	0.13
1986	722	0.05	2129	0.14	5193	0.06	3247	0.09	1.15	0.08
1987	783	0.04	2788	0.12	6603	0.05	4471	0.07	0.78	0.09
1988	622	0.05	2810	0.14	6864	0.06	5612	0.07	0.80	0.09
1989	626	0.05	2682	0.14	6408	0.06	5087	0.07	0.73	0.11
1990	699	0.04	2539	0.12	6214	0.05	4521	0.07	0.97	0.08
1991	851	0.04	2450	0.09	6187	0.04	3971	0.06	1.31	0.05
1992	532	0.03	1923	0.09	5172	0.03	4011	0.04	1.41	0.03
1993	335	0.04	1610	0.11	3962	0.04	3393	0.05	0.80	0.07
1994	352	0.05	1614	0.12	3662	0.05	2912	0.07	0.50	0.13
1995	655	0.04	2144	0.09	4829	0.04	3042	0.06	0.56	0.12
1996	626	0.04	2431	0.09	5833	0.04	4144	0.05	0.81	0.07
1997	543	0.05	2291	0.11	5515	0.05	4386	0.06	0.79	0.07
1998	795	0.05	2841	0.11	6515	0.05	4443	0.07	0.65	0.10
1999	1187	0.04	4137	0.09	9407	0.04	3776	0.10	0.90	0.10
2000	1494	0.03	6283	0.08	14023	0.03	6666	0.07	0.56	0.14
2001	1653	0.03	8287	0.06	18992	0.03	11464	0.05	0.58	0.10
2002	1352	0.03	8193	0.07	19389	0.03	13910	0.04	0.68	0.07
2003	1342	0.03	7792	0.07	18144	0.03	13539	0.04	0.57	0.09
2004	1259	0.03	8640	0.06	19143	0.03	14052	0.04	0.29	0.17
2005	1252	0.03	9149	0.06	20406	0.03	14219	0.04	0.35	0.14
2006	1073	0.04	6933	0.08	16912	0.03	11913	0.04	0.90	0.05
2007	1470	0.03	6835	0.07	15781	0.03	8820	0.06	0.70	0.08
2008	1561	0.03	8417	0.07	19193	0.03	9342	0.06	0.66	0.09
2009	1632	0.03	9842	0.06	22362	0.03	13366	0.05	0.52	0.10
2010	1541	0.03	11544	0.06	25423	0.03	17237	0.04	0.27	0.18
2011	1450	0.02	12827	0.05	27946	0.02	21570	0.03	0.19	0.23
2012	1416	0.02	11523	0.06	26521	0.03	21084	0.03	0.46	0.10
2013	1217	0.03	8478	0.08	20695	0.03	15540	0.04	0.84	0.06
2014	1701	0.05	9108	0.08	21138	0.04	13072	0.06	0.35	0.17
2015	1314	0.05	8694	0.11	19906	0.05	12557	0.07	0.52	0.13
2016	1210	0.05	8938	0.12	19944	0.06	13727	0.08	0.34	0.21
2017	1075	0.06	9819	0.13	21118	0.06	16030	0.08	0.13	0.55

Table A9.3. CASA Estimates for Mid-Atlantic.

Year	Abundance millions	CV	SSB mt gonads	CV	Bms mt meats	CV	ExplBms mt meats	CV	F	CV
1975	1025	0.05	5962	0.10	11843	0.05	4390	0.13	0.40	0.23
1976	772	0.05	6708	0.11	13388	0.06	5824	0.13	0.64	0.14
1977	816	0.05	6282	0.11	12519	0.06	3867	0.18	0.73	0.17
1978	654	0.04	5384	0.10	10744	0.05	3681	0.15	1.27	0.09
1979	398	0.05	4074	0.10	8134	0.05	2655	0.16	1.23	0.10
1980	369	0.05	3327	0.11	6638	0.06	5868	0.06	0.39	0.19
1981	404	0.05	3463	0.11	6912	0.05	5999	0.06	0.14	0.49
1982	422	0.05	3718	0.10	7415	0.05	6572	0.06	0.27	0.24
1983	483	0.05	3536	0.10	7051	0.05	5879	0.06	0.59	0.12
1984	433	0.06	2868	0.13	5694	0.07	4700	0.08	0.84	0.10
1985	763	0.07	3174	0.14	6276	0.07	3689	0.12	0.92	0.12
1986	1192	0.05	5258	0.11	10423	0.06	6393	0.09	0.55	0.17
1987	1502	0.04	6880	0.10	13675	0.05	9075	0.08	1.02	0.08
1988	1439	0.04	7537	0.10	14969	0.05	10912	0.07	0.69	0.11
1989	1496	0.04	7302	0.10	14497	0.05	10384	0.07	1.02	0.08
1990	1336	0.04	6967	0.08	13862	0.04	10077	0.06	0.86	0.08
1991	903	0.04	5431	0.08	10823	0.04	8820	0.05	1.10	0.06
1992	541	0.06	3474	0.12	6868	0.06	5817	0.07	1.13	0.06
1993	1234	0.05	4144	0.10	8160	0.05	3937	0.11	0.87	0.12
1994	1352	0.04	5606	0.09	11114	0.04	6632	0.07	1.25	0.07
1995	1205	0.04	5767	0.08	11477	0.04	8002	0.06	1.05	0.07
1996	727	0.05	5089	0.10	10147	0.05	8582	0.06	0.72	0.09
1997	703	0.07	4538	0.13	8949	0.07	7435	0.08	0.48	0.16
1998	2115	0.06	7433	0.11	14608	0.06	5598	0.15	0.60	0.19
1999	3603	0.05	15200	0.09	30136	0.05	10332	0.13	0.48	0.25
2000	4211	0.04	23628	0.08	47021	0.04	24429	0.07	0.41	0.24
2001	4518	0.03	28644	0.07	57080	0.03	36561	0.05	0.46	0.17
2002	4076	0.03	29856	0.06	59388	0.03	36932	0.05	0.50	0.14
2003	5751	0.07	31814	0.07	63280	0.04	37717	0.06	0.57	0.12
2004	4267	0.04	29622	0.06	59071	0.03	31418	0.06	0.80	0.08
2005	3791	0.03	29182	0.06	58224	0.03	31623	0.05	0.54	0.12
2006	3941	0.03	31493	0.05	62869	0.03	38228	0.04	0.25	0.24
2007	3444	0.03	29845	0.05	59497	0.03	37118	0.04	0.49	0.12
2008	4213	0.03	28884	0.05	57626	0.03	34163	0.04	0.56	0.11
2009	3434	0.02	29679	0.05	59294	0.03	33053	0.05	0.61	0.10
2010	2705	0.03	26760	0.05	53476	0.03	37597	0.04	0.53	0.11
2011	1911	0.03	21145	0.07	42157	0.03	32298	0.04	0.56	0.10
2012	3116	0.04	20312	0.08	40494	0.04	25648	0.06	0.46	0.14
2013	3111	0.04	25158	0.08	50130	0.04	26038	0.08	0.23	0.28
2014	4773	0.04	33135	0.08	65750	0.04	38482	0.07	0.24	0.27
2015	10054	0.04	50934	0.08	101435	0.04	42909	0.09	0.24	0.28
2016	8902	0.05	65902	0.09	131588	0.04	55984	0.10	0.23	0.30
2017	7578	0.05	72698	0.09	145265	0.05	89728	0.08	0.17	0.42

Table A9.4. CASA Estimates for Georges Bank and Mid-Atlantic combined, excluding DSENLS scallops.

Year	Abundance millions	CV	SSB mt gonads	CV	Bms mt meats	CV	ExplBms mt meats	CV	F	CV
1975	3155	0.05	24467	0.12	51187	0.06	40059	0.07	0.14	0.06
1976	2656	0.05	25882	0.12	54945	0.06	44644	0.07	0.25	0.06
1977	2503	0.05	23387	0.12	50681	0.06	40429	0.07	0.33	0.06
1978	2247	0.05	19788	0.12	43303	0.06	33816	0.07	0.49	0.07
1979	1590	0.05	15045	0.13	33670	0.06	26831	0.07	0.57	0.07
1980	1770	0.05	12590	0.13	27793	0.06	24120	0.07	0.41	0.06
1981	1582	0.06	11328	0.13	25481	0.06	22251	0.07	0.46	0.06
1982	1436	0.06	9895	0.14	22308	0.06	19699	0.07	0.54	0.07
1983	1228	0.07	8585	0.16	19016	0.07	16550	0.08	0.58	0.08
1984	1165	0.08	7726	0.19	16680	0.09	14170	0.10	0.53	0.09
1985	1564	0.08	7951	0.19	17184	0.09	12841	0.12	0.62	0.10
1986	2407	0.06	10091	0.16	21877	0.07	14430	0.11	0.70	0.10
1987	2824	0.06	12468	0.15	26813	0.07	18502	0.10	0.89	0.09
1988	2818	0.07	13599	0.15	29433	0.07	21588	0.10	0.76	0.09
1989	3098	0.06	14395	0.14	31035	0.06	22370	0.09	0.85	0.09
1990	2954	0.05	13278	0.13	29763	0.06	21729	0.08	1.06	0.08
1991	2629	0.06	11055	0.13	25171	0.06	18226	0.08	1.28	0.07
1992	1869	0.09	8688	0.20	19932	0.09	15250	0.12	1.24	0.09
1993	3023	0.10	11799	0.25	25459	0.11	15289	0.19	0.57	0.13
1994	3083	0.09	16861	0.24	35024	0.11	26576	0.15	0.43	0.13
1995	3571	0.09	21609	0.23	44896	0.11	36259	0.14	0.34	0.12
1996	3109	0.10	24901	0.23	52290	0.11	46252	0.12	0.27	0.11
1997	3036	0.10	27237	0.23	56780	0.11	51949	0.12	0.18	0.11
1998	5324	0.09	35097	0.20	72954	0.10	45315	0.16	0.21	0.11
1999	6981	0.07	46844	0.18	97602	0.08	54332	0.15	0.30	0.12
2000	8386	0.06	59879	0.15	124294	0.07	75756	0.12	0.29	0.10
2001	9127	0.05	71049	0.14	147771	0.07	97544	0.10	0.32	0.08
2002	8184	0.05	75545	0.13	157358	0.06	107600	0.09	0.33	0.07
2003	9954	0.07	79073	0.12	164400	0.06	116122	0.08	0.32	0.07
2004	8041	0.05	76710	0.11	160016	0.05	112320	0.07	0.37	0.06
2005	7195	0.04	72255	0.10	152093	0.05	108273	0.07	0.32	0.06
2006	6737	0.04	65531	0.10	139620	0.05	102115	0.07	0.34	0.06
2007	6601	0.04	59716	0.10	125255	0.05	87578	0.07	0.39	0.06
2008	7370	0.04	59962	0.10	125025	0.05	81766	0.07	0.40	0.06
2009	6832	0.04	62474	0.10	130493	0.05	84113	0.07	0.40	0.06
2010	6124	0.05	60677	0.11	127542	0.05	90761	0.07	0.34	0.06
2011	5226	0.05	52478	0.11	112971	0.05	83525	0.07	0.36	0.06
2012	6233	0.06	47299	0.13	102703	0.06	69064	0.09	0.43	0.07
2013	6029	0.06	51650	0.14	108199	0.06	64933	0.11	0.35	0.07
2014	11538	0.07	72799	0.13	151369	0.06	81066	0.12	0.21	0.07
2015	17091	0.06	107706	0.12	222660	0.06	96973	0.14	0.19	0.07
2016	16008	0.06	137994	0.12	285114	0.06	146639	0.12	0.14	0.08
2017	14171	0.06	153182	0.13	317334	0.06	214921	0.09	0.12	0.07

Figures

Figure A9.1. Estimated plus group meat weights for the population and fishery in the open and closed portions of Georges Bank, and in the Mid-Atlantic. The plus group represents scallops in the largest bin which contained L_∞. Dash line is the meat weight for the last size bin calculated using the shell height meat weight parameters used in the CASA model.

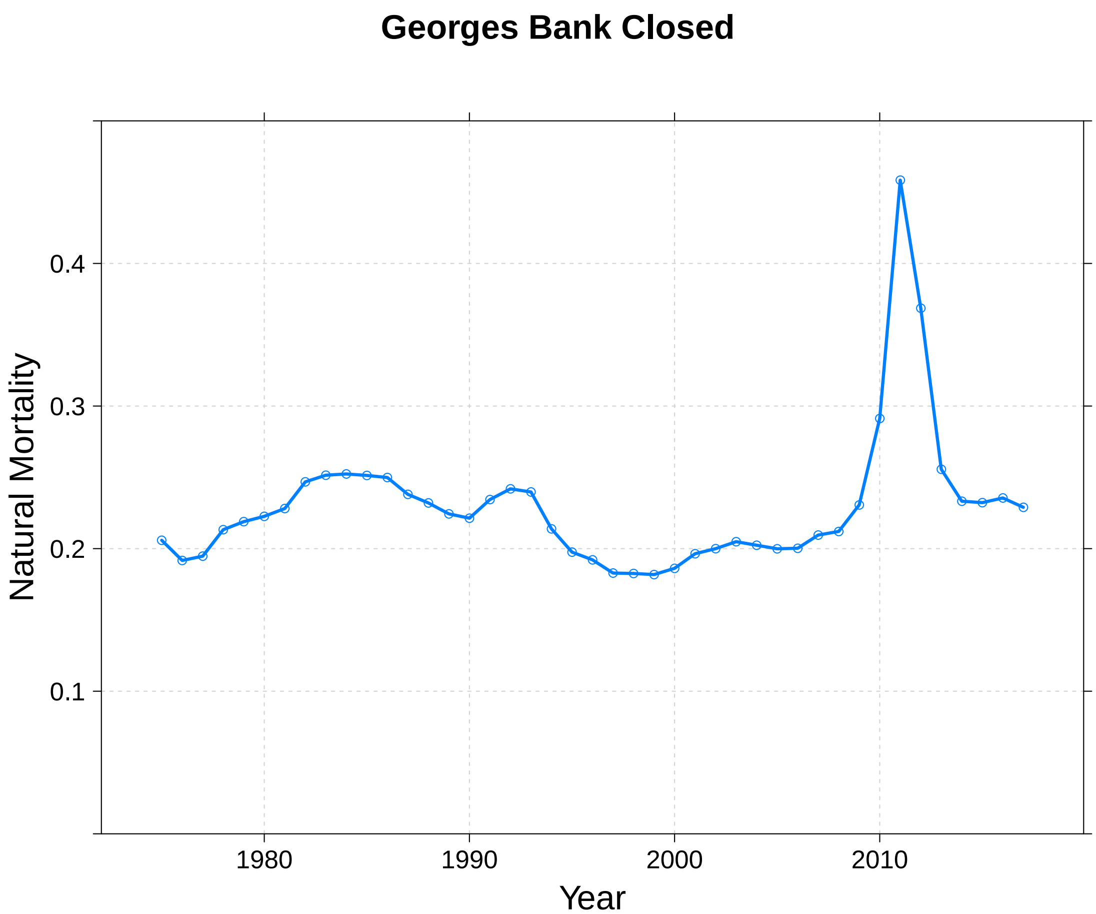


Figure A9.2. Estimated natural mortality from 1975 to 2017 for Georges Bank closed areas.

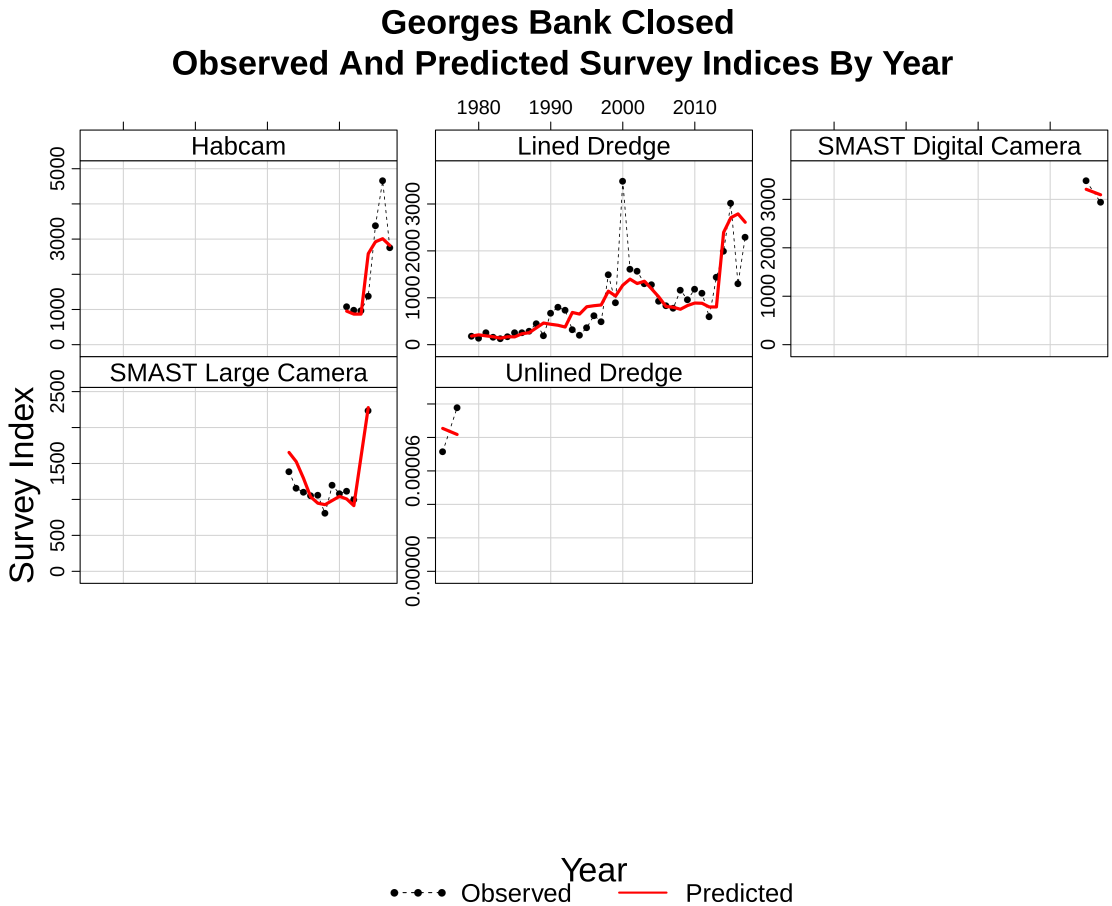
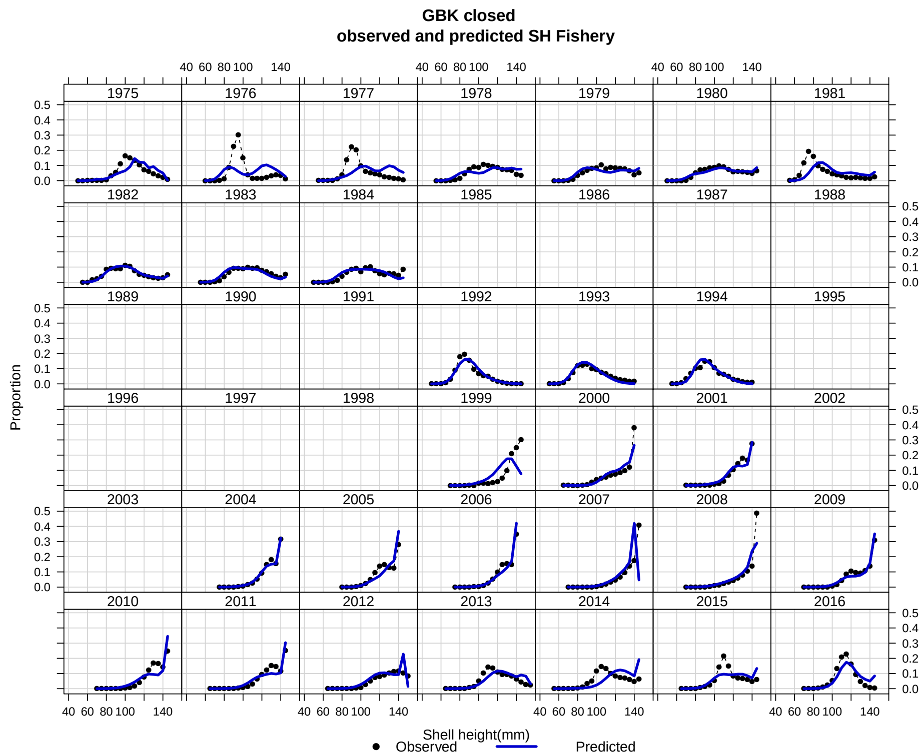


Figure A9.3. Observed survey trend (solid circles) and corresponding model estimates (lines) for the SMAST digital camera (top left), lined dredge (top middle), Habcam (top right), large drop camera (bottom left), and unlined dredge (bottom middle) surveys on Georges Bank closed areas.



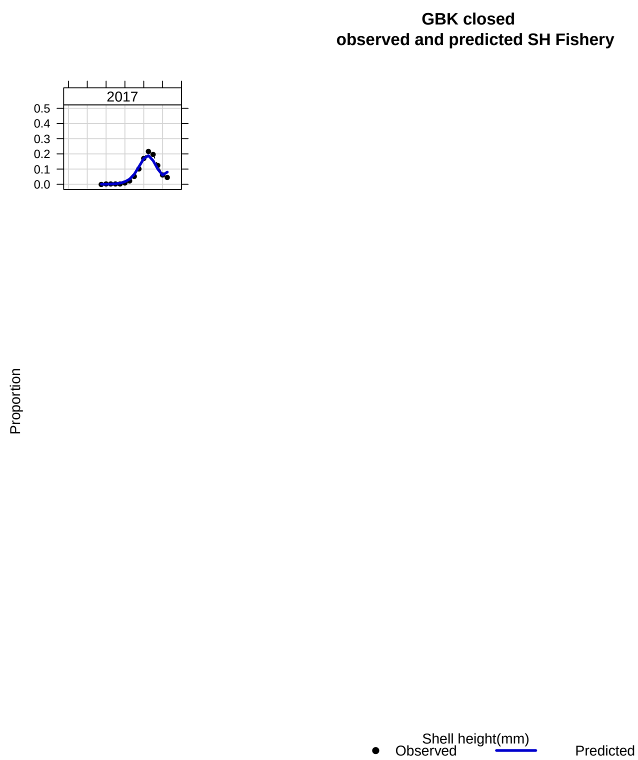


Figure A9.4. Comparison of observed fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Georges Bank closed areas.

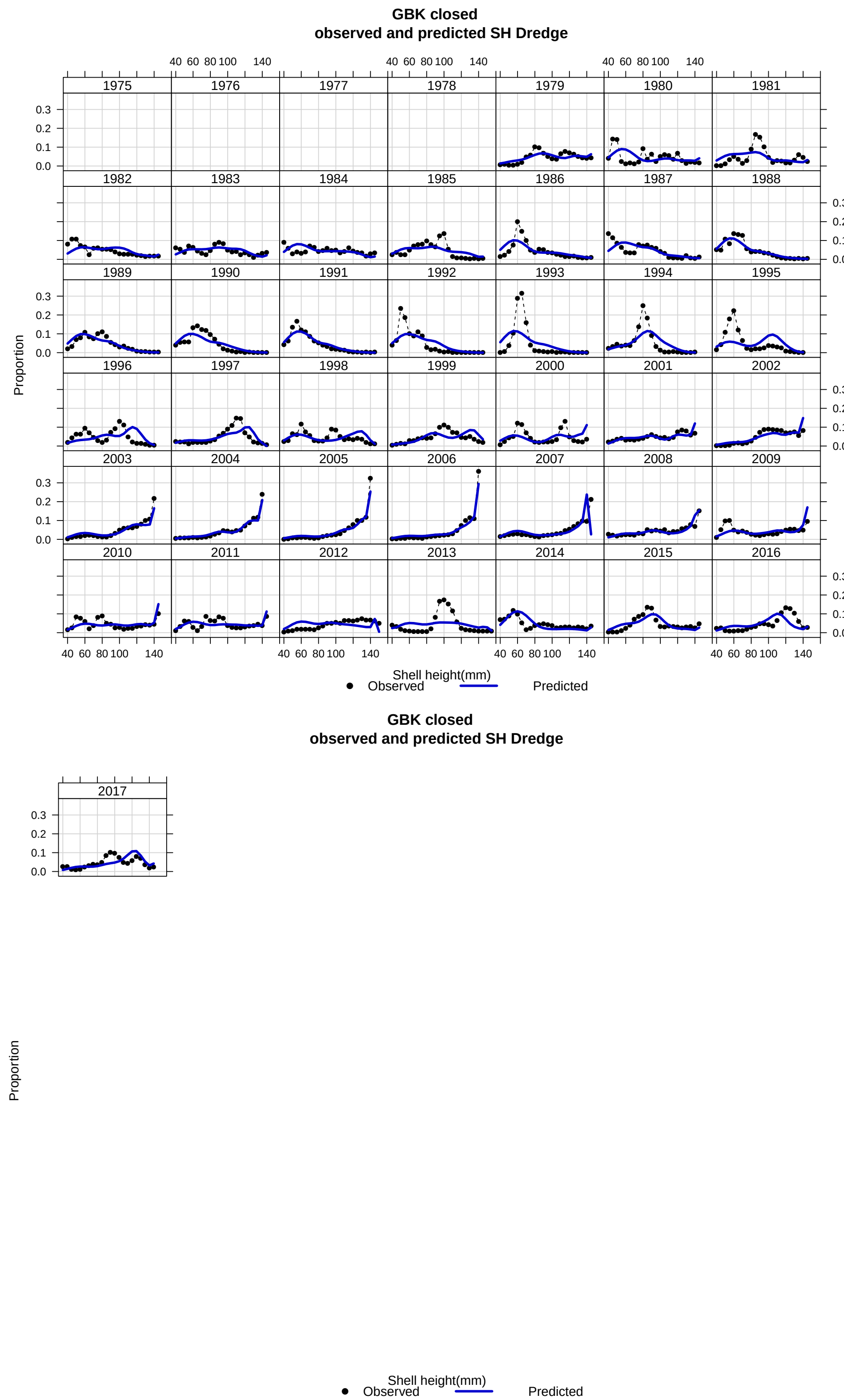
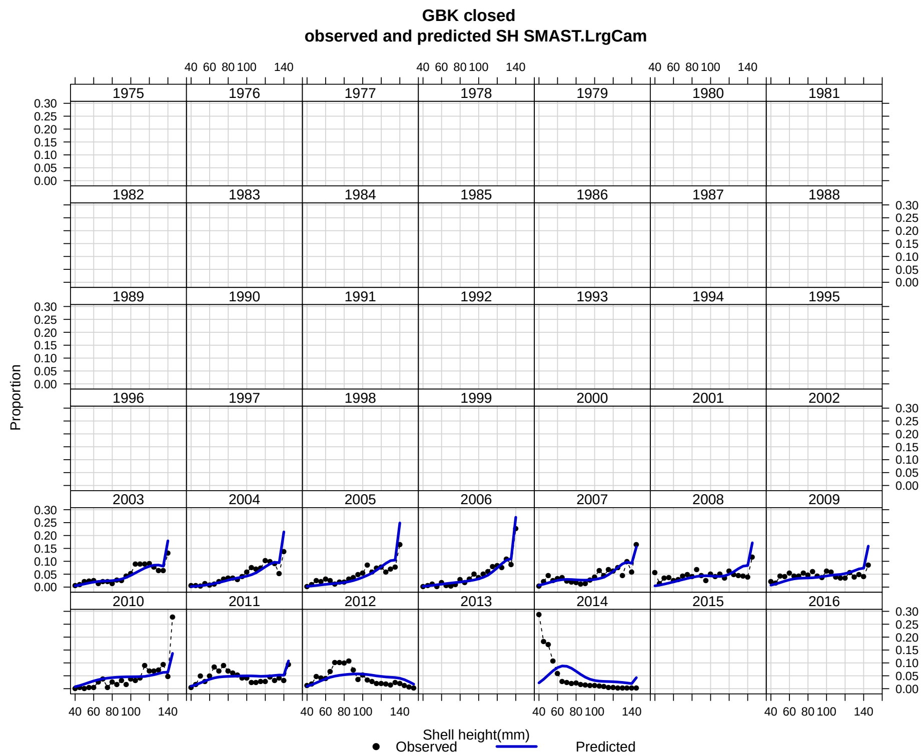


Figure A9.5. Comparison of lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.



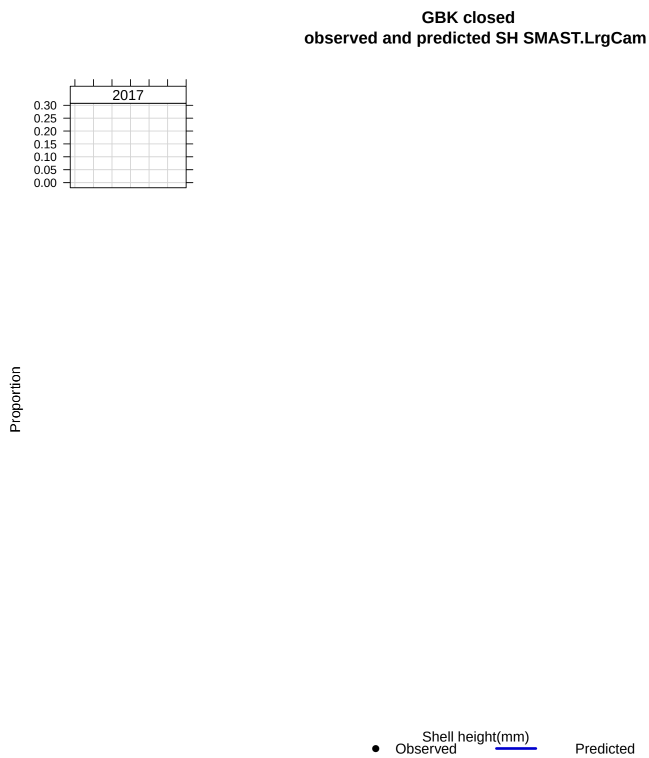
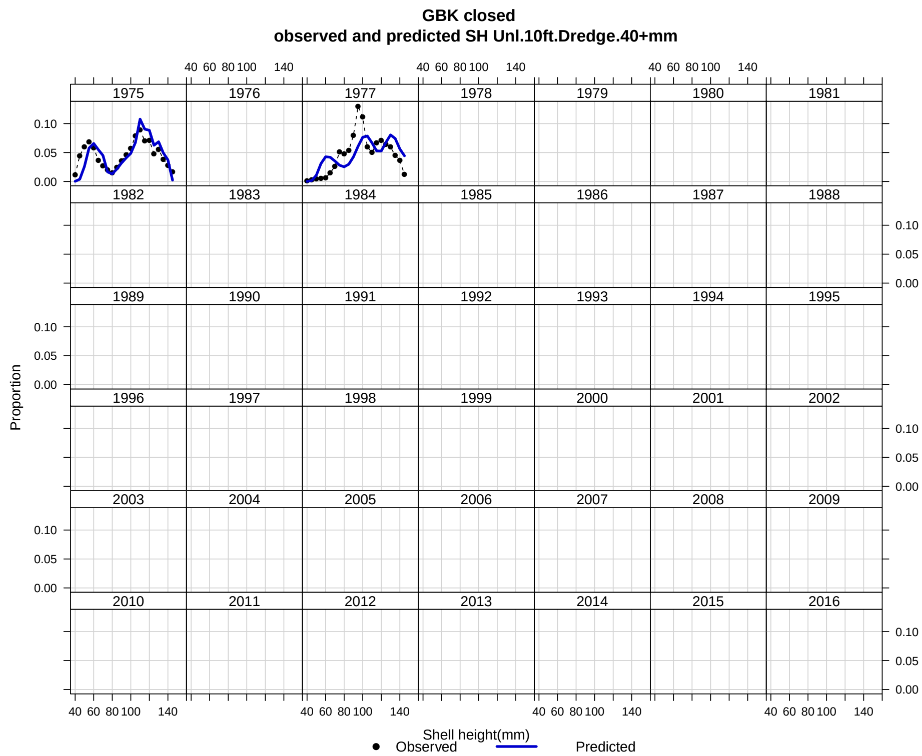


Figure A9.6. Comparison of large drop camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.



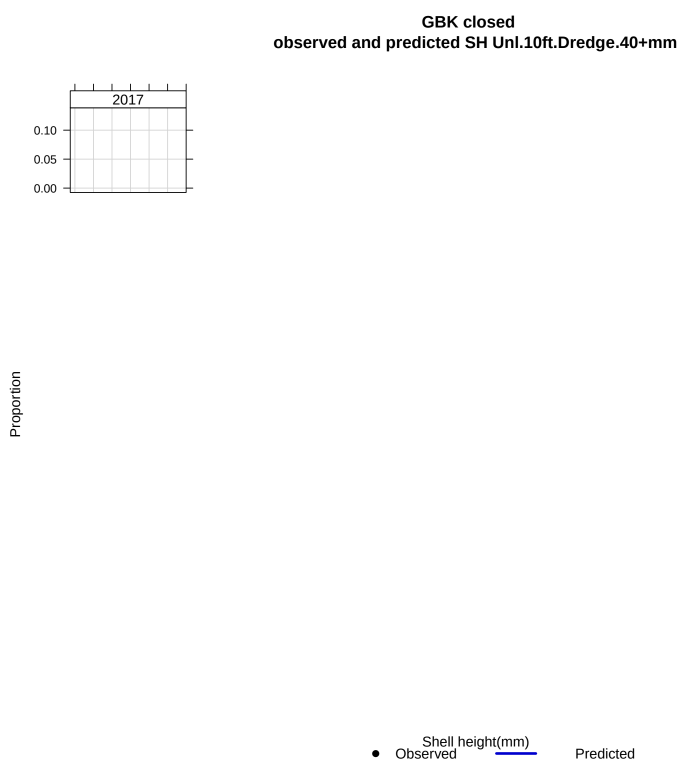
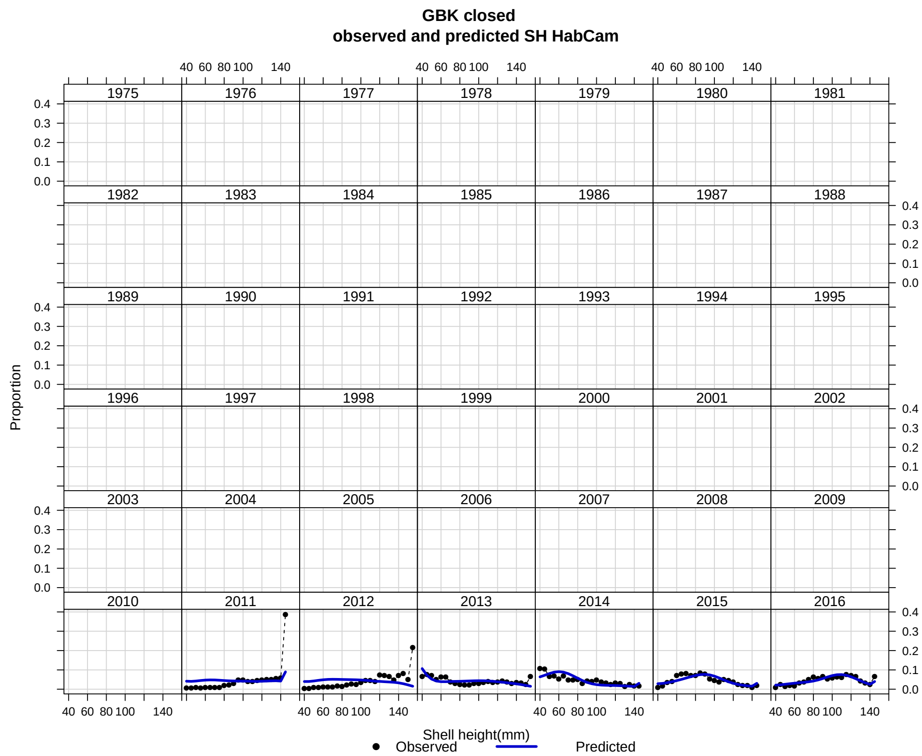


Figure A9.7. Comparison of unlined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.



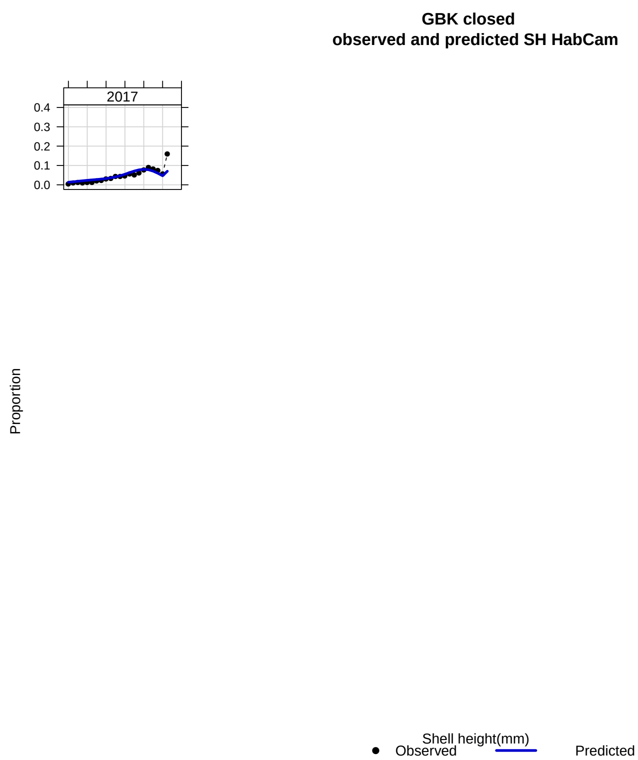
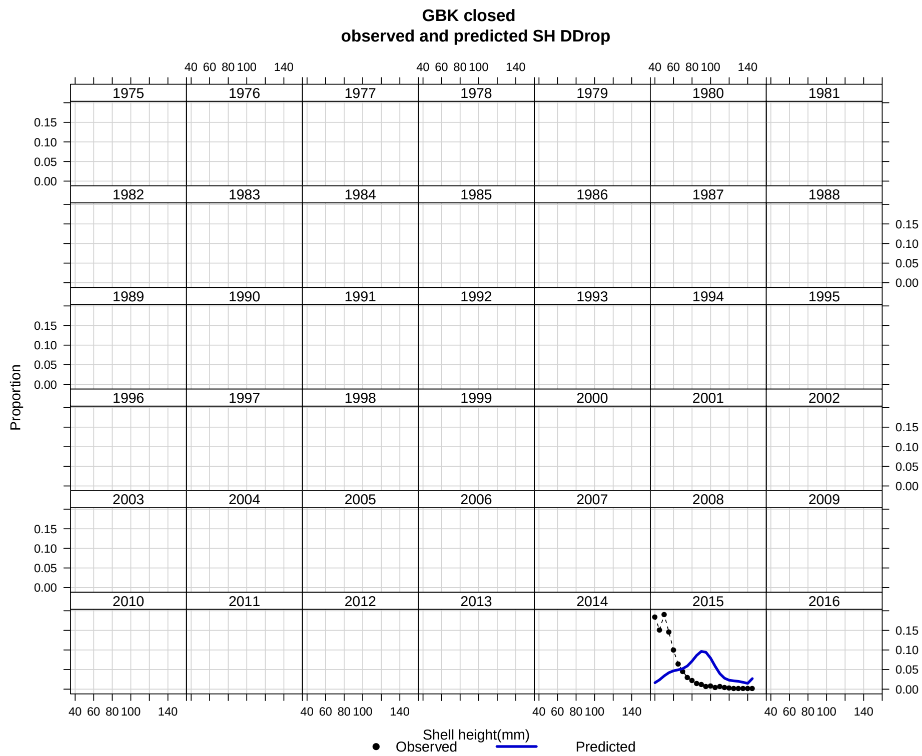


Figure A9.8. Comparison of Habcam survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.



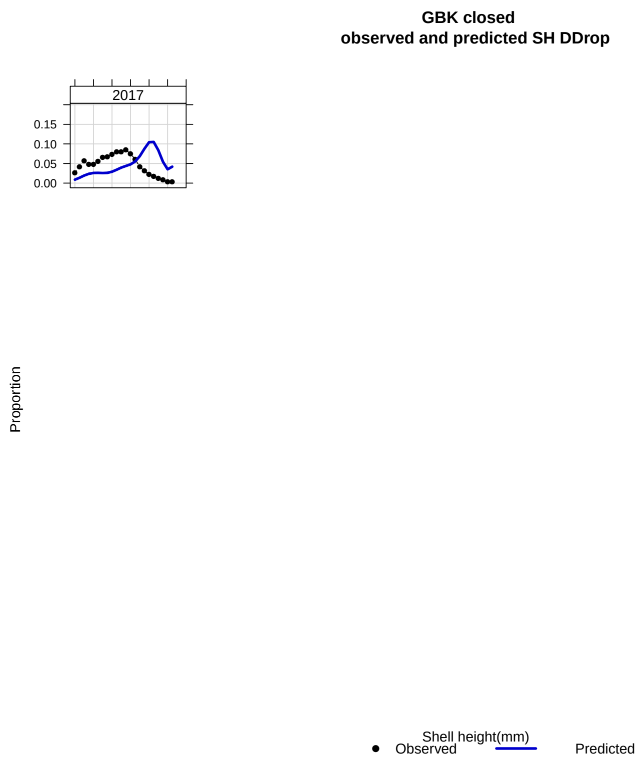


Figure A9.9. Comparison of SMAST digital camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.

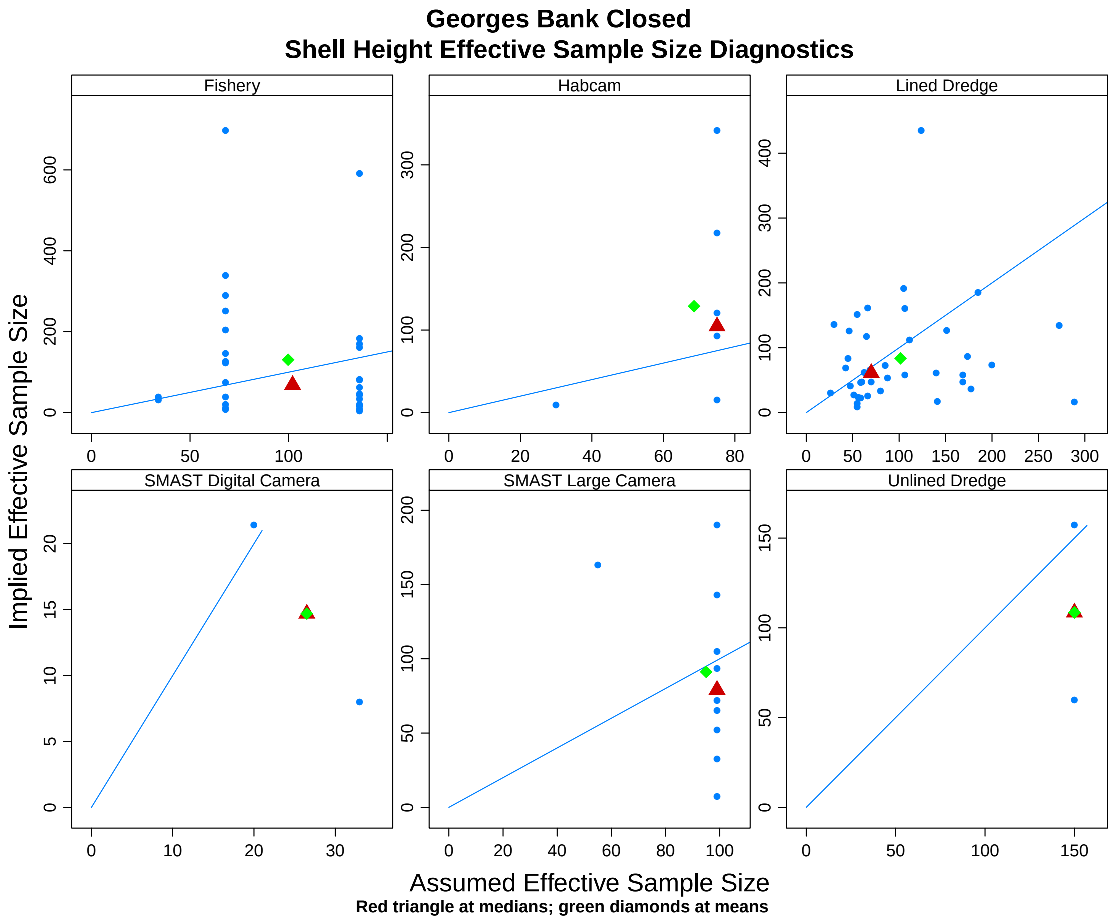


Figure A9.10. Assumed and model implied effective sample sizes for SMAST digital camera (top left), lined dredge (top middle), Habcam (bottom left), large drop camera (bottom middle), and unlined dredge (bottom right) surveys, and the fishery (top right) shell height compositions for Georges Bank closed areas. The triangle is the median and the diamond is the mean.

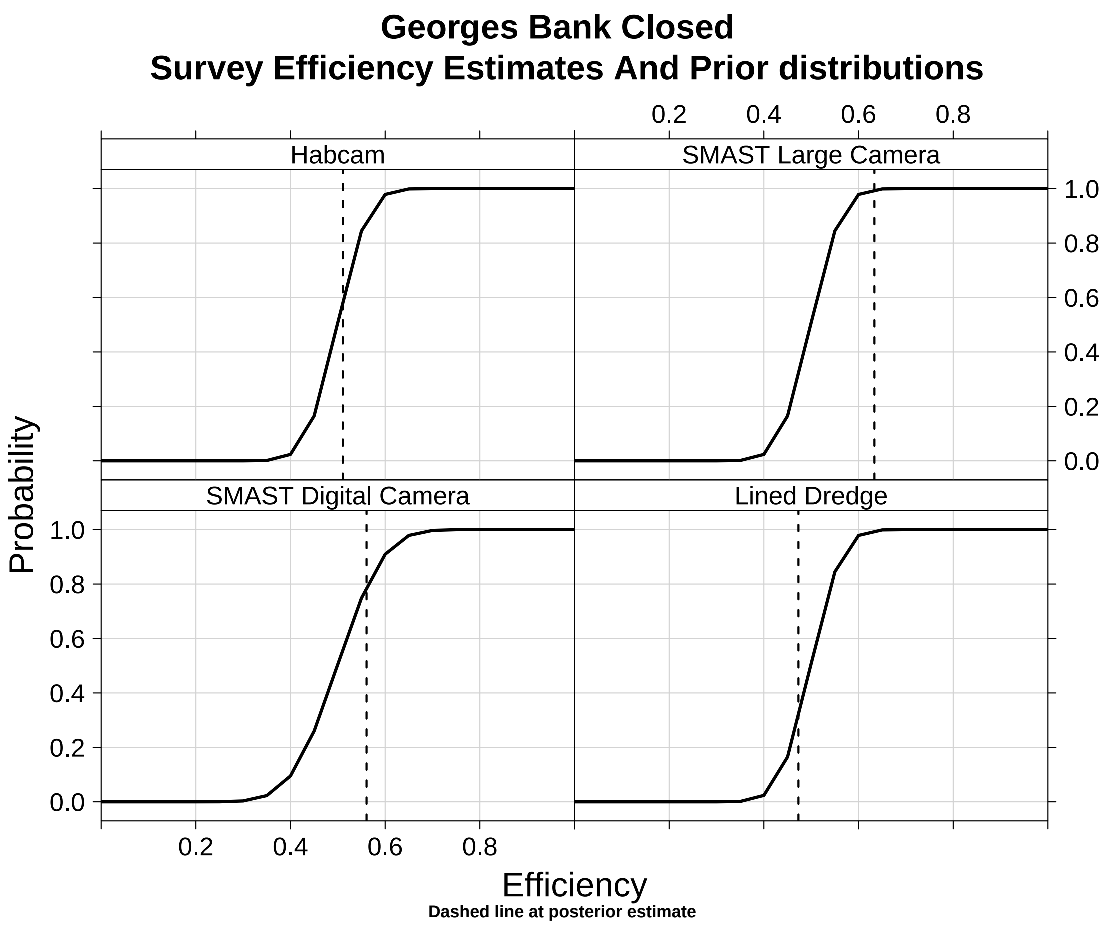


Figure A9.11. Prior cumulative distributions for catchability of Habcam (top left), large drop camera (top right), SMAST digital camera (bottom left), and lined dredge (bottom right) surveys for Georges Bank closed areas. The dashed lines are the mean posterior estimate for survey catchability. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5.

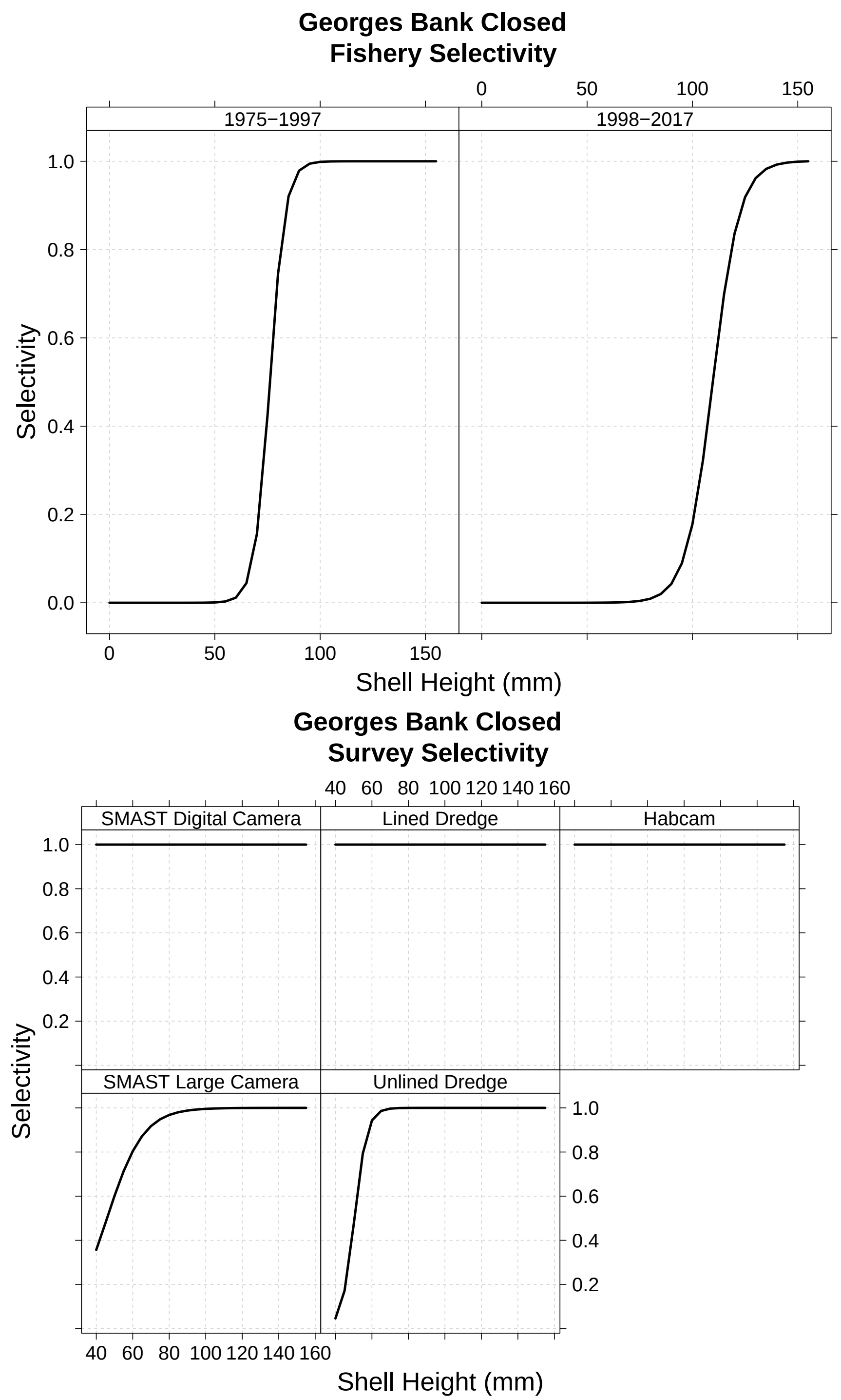


Figure A9.12. Estimated fishery selectivity curves (top) and assumed selectivity curves (bottom) for SMAST digital camera (bottom panel; top left), lined dredge (bottom panel; top middle), Habcam (bottom panel; top right), large drop camera (bottom panel; bottom left), and unlined dredge (bottom panel; bottom middle) surveys for Georges Bank closed areas.

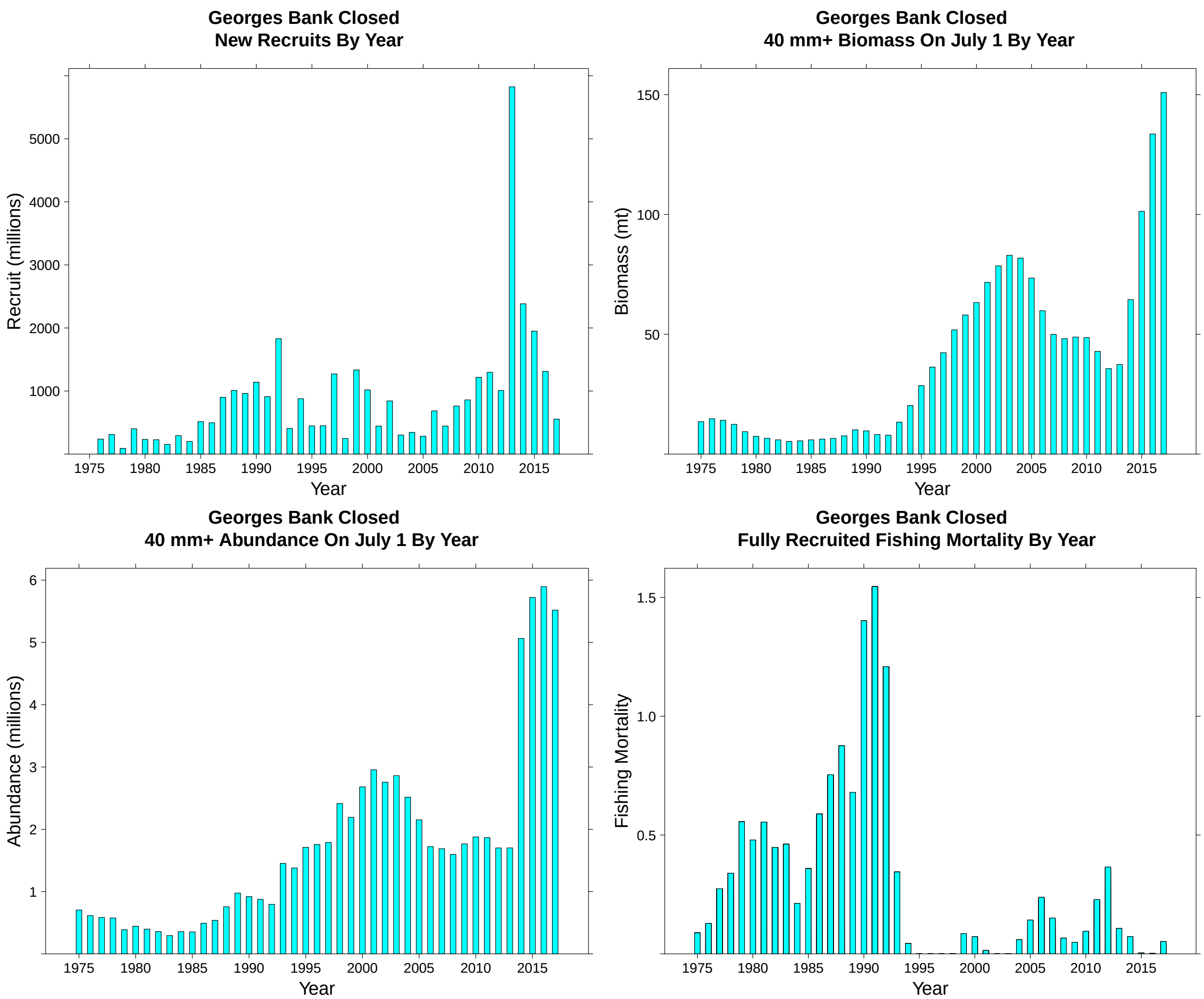


Figure A9.13. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left) and fully recruited fishing mortality (bottom right) for Georges Bank closed areas.

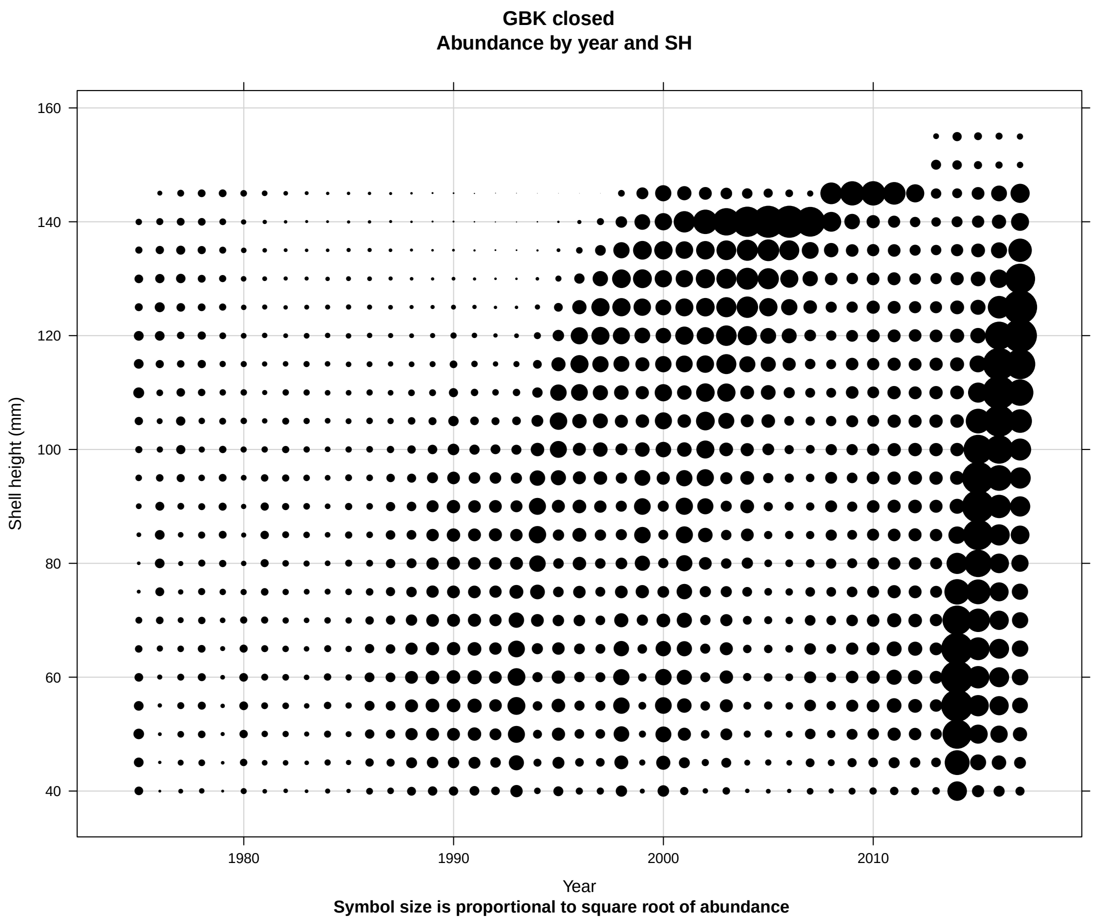


Figure A9.14. CASA model estimated abundances at shell height for Georges Bank closed areas. Symbol areas are proportional to abundance.

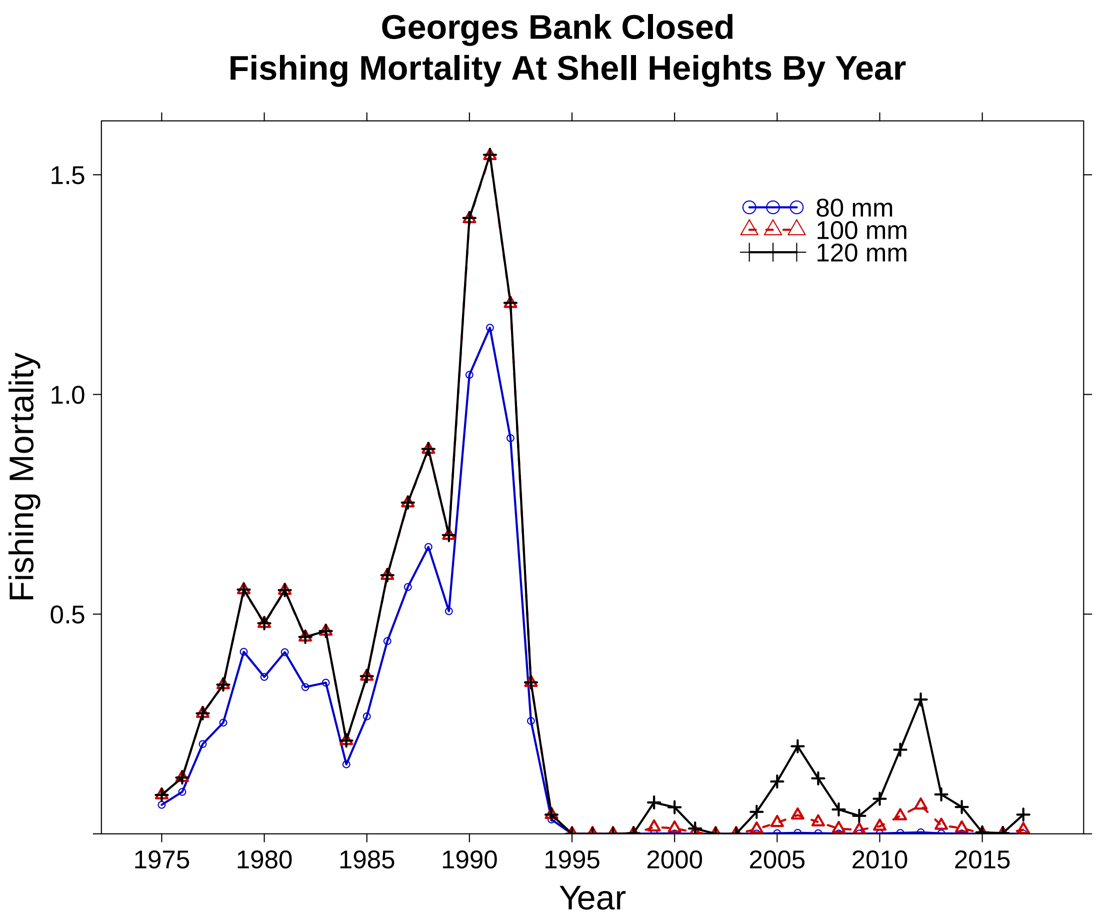


Figure A9.15. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles), and 120 mm SH (dashed line with crosses) for Georges Bank closed areas.

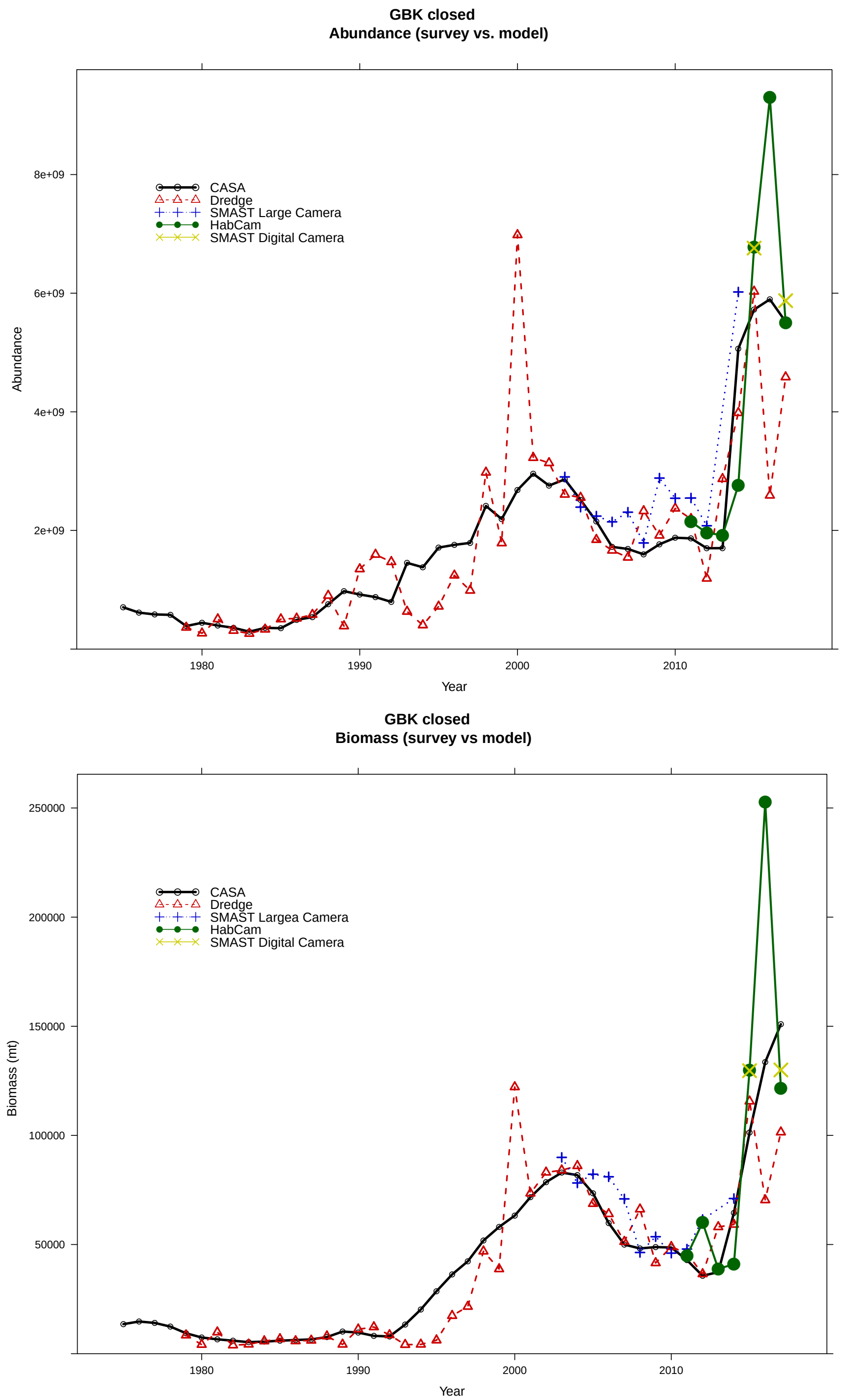


Figure A9.16. Comparison of CASA model estimated abundance (top) and biomass (bottom) with expanded estimates from the lined dredge (red), large drop camera (blue), HabCam (green), and SMAST digital camera (light green) for Georges Bank closed areas.

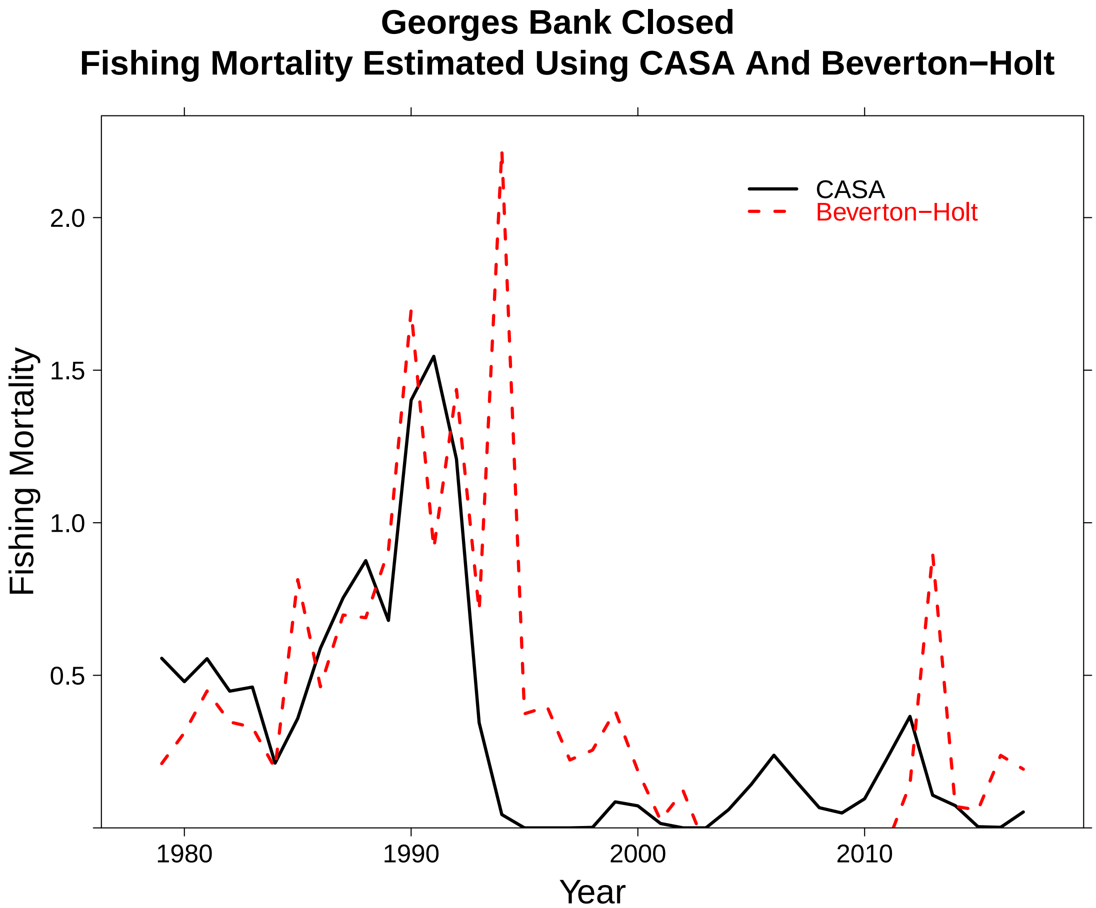


Figure A9.17. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium length based estimator for Georges Bank closed areas.

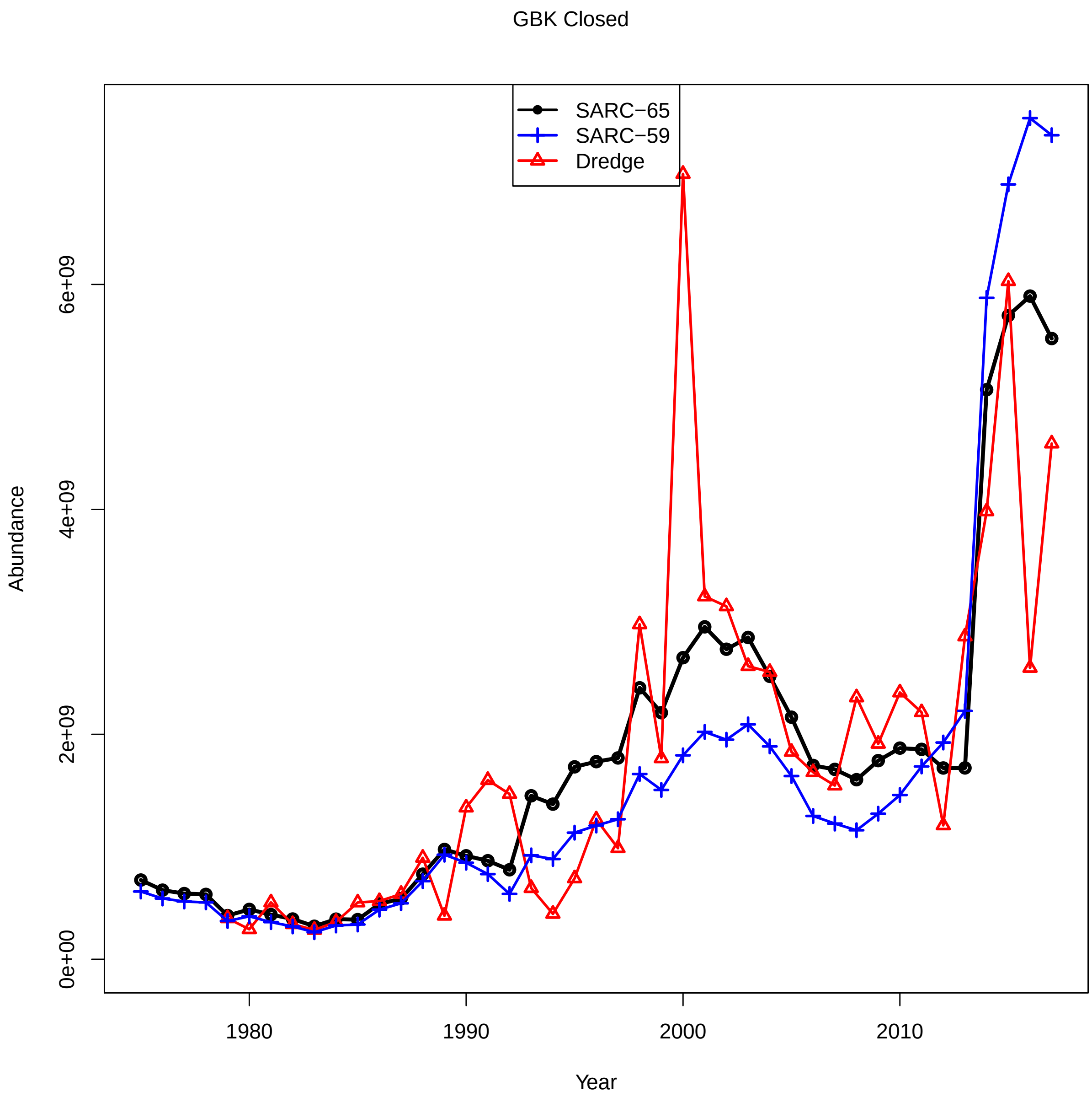


Figure A9.18. Comparison of estimated abundance from current and SARC-59 CASA model configurations, along with lined dredge survey abundance index for Georges Bank closed areas.

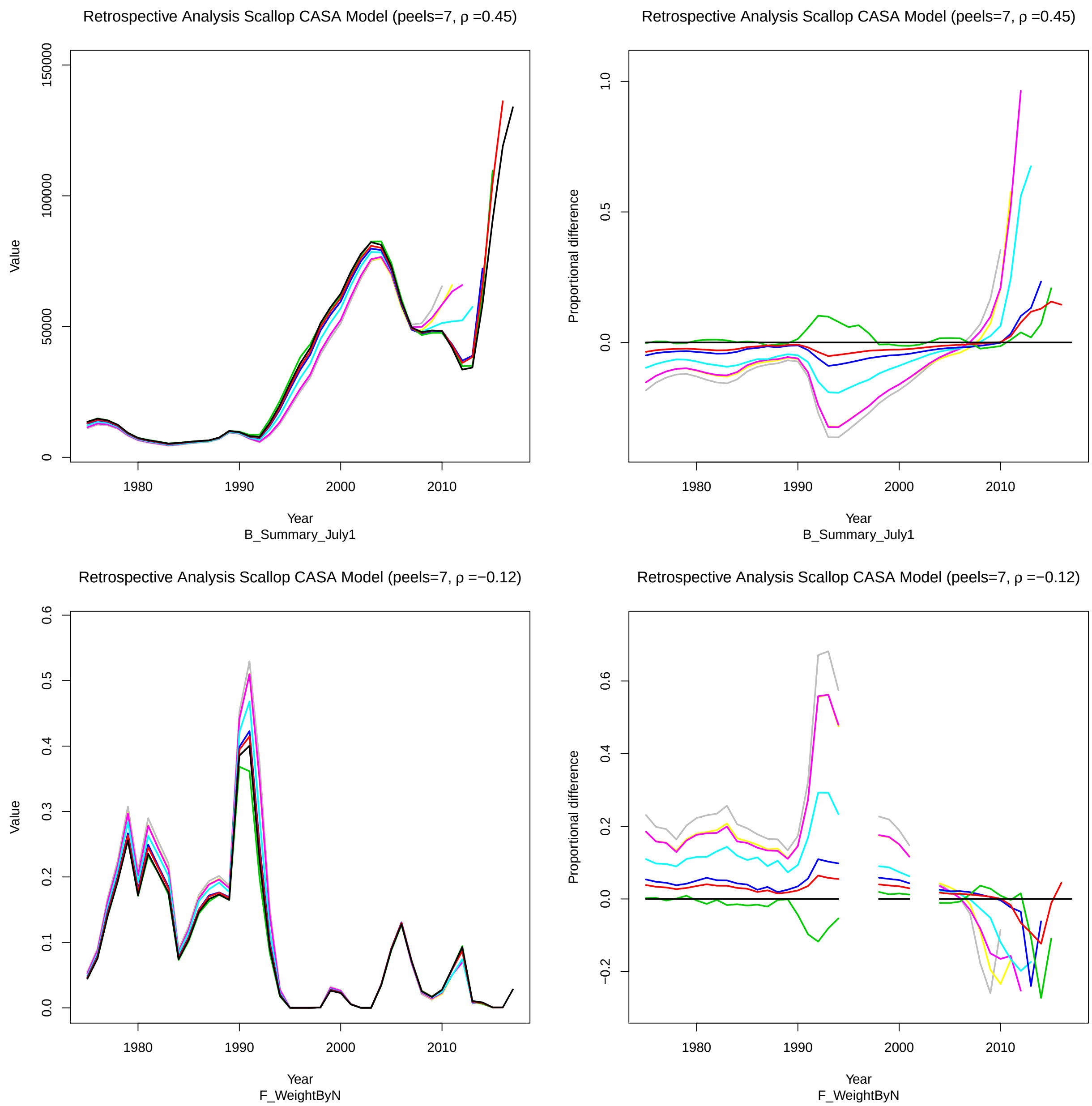


Figure A9.19. Retrospective plots for biomass and fishing mortality for Georges Bank closed areas. Retrospectives are shown on both absolute and relative scales.

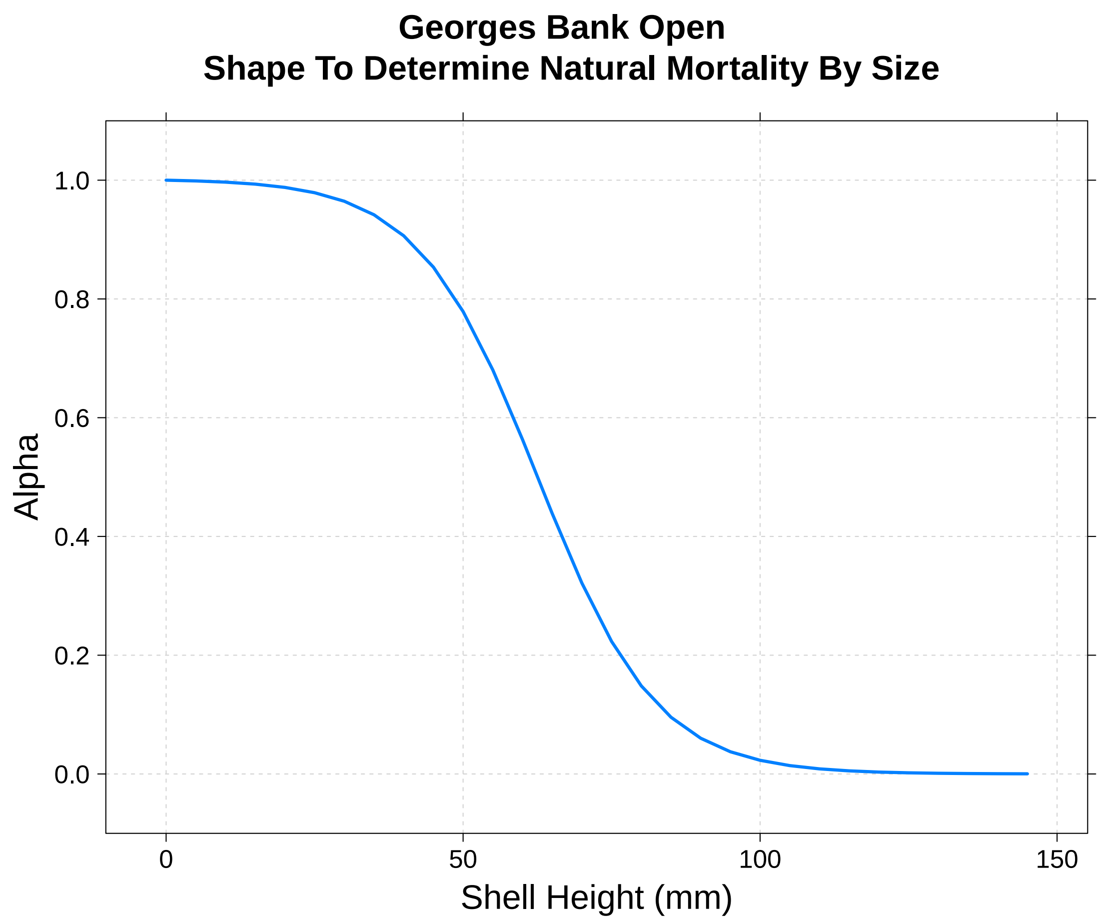


Figure A9.20. Logistic curve used to partition juvenile and adult natural mortality for Georges Bank open areas.

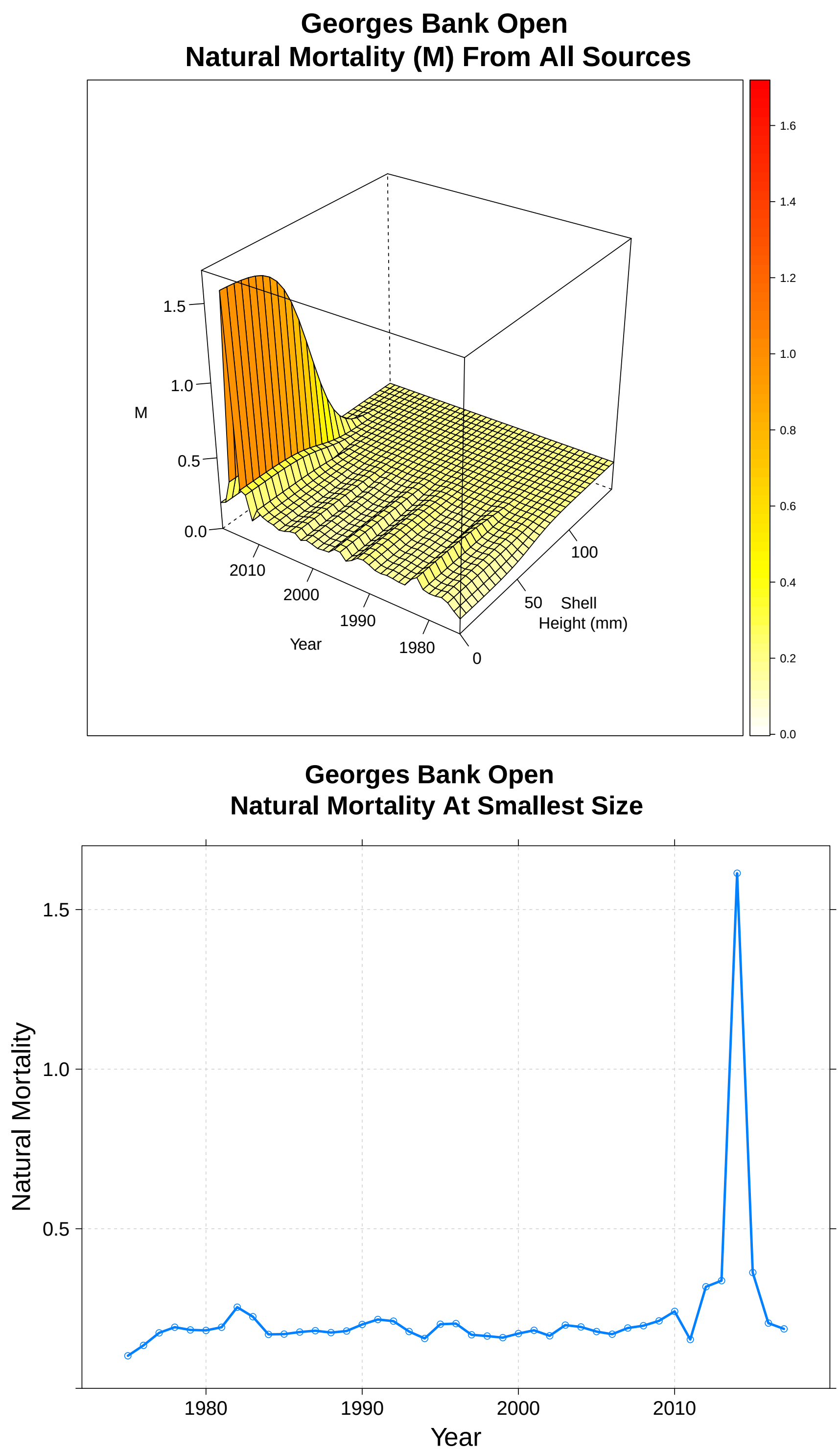


Figure A9.21. Estimated natural mortality by size (top) and for smallest size group (bottom) from 1975 to 2017 for Georges Bank open areas.

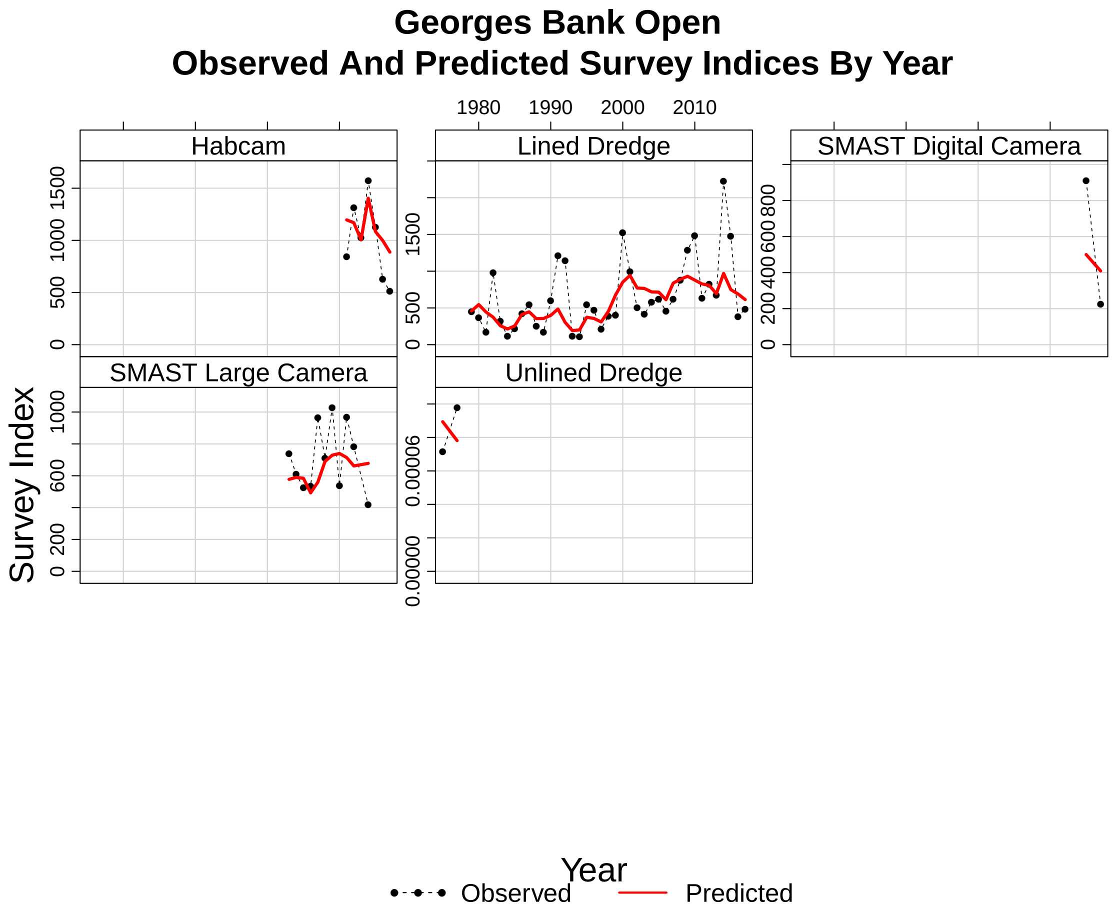
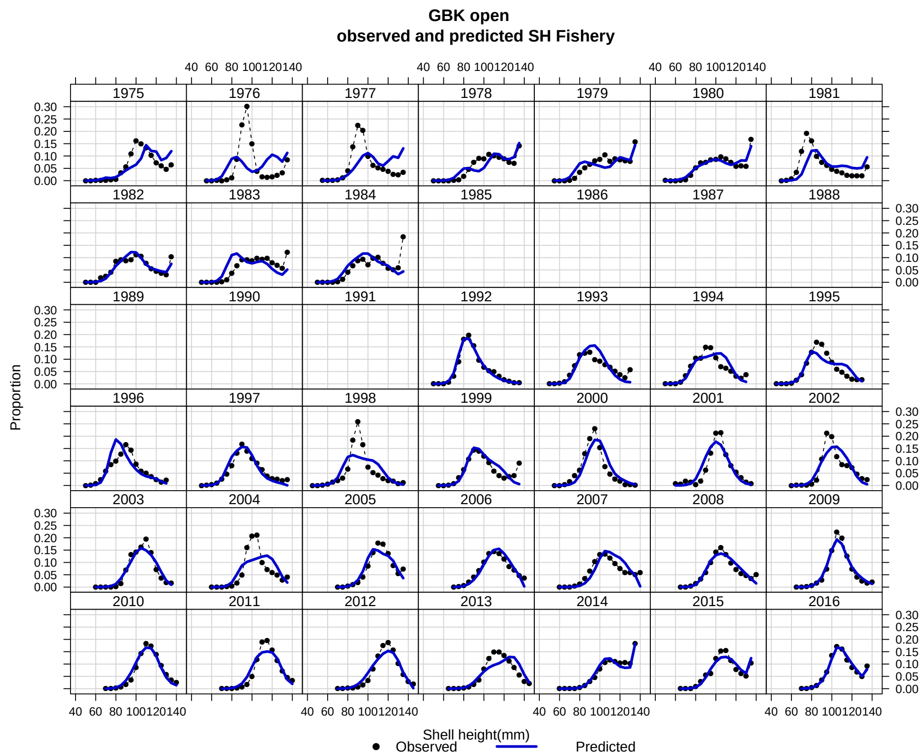


Figure A9.22. Observed survey trend (solid circles) and corresponding model estimates (lines) for the SMAST digital camera (top left), lined dredge (top middle), Habcam (top right), large drop camera (bottom left), and unlined dredge (bottom middle) surveys on Georges Bank open areas.



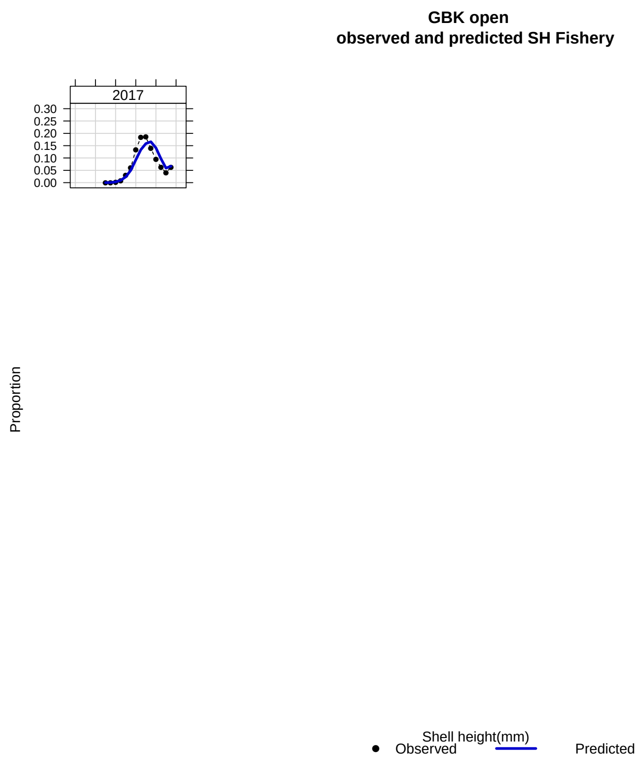
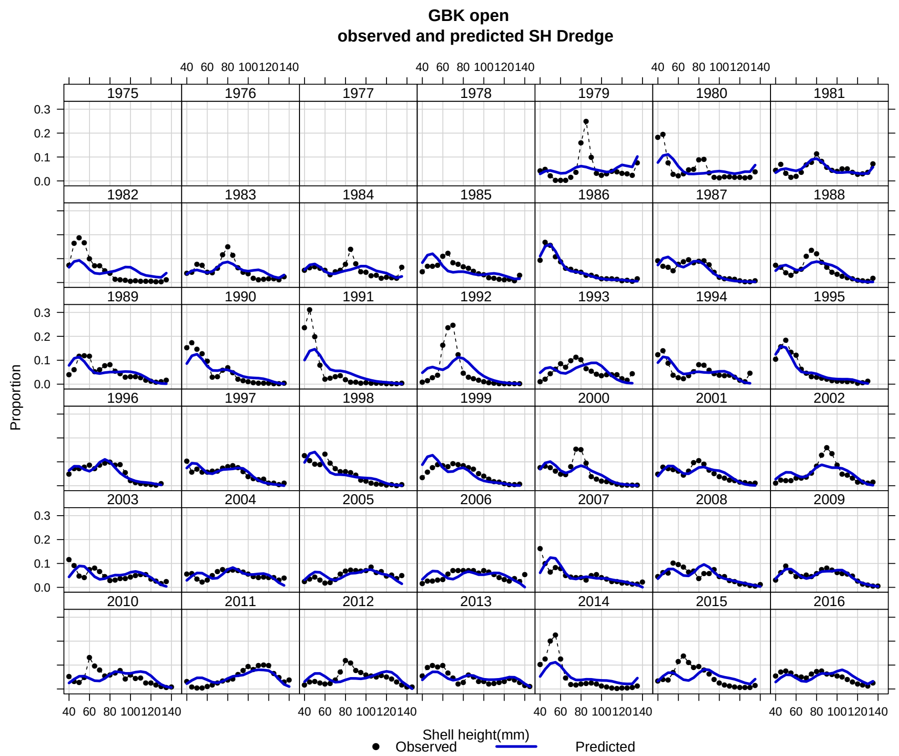


Figure A9.23. Comparison of observed fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Georges Bank open areas.



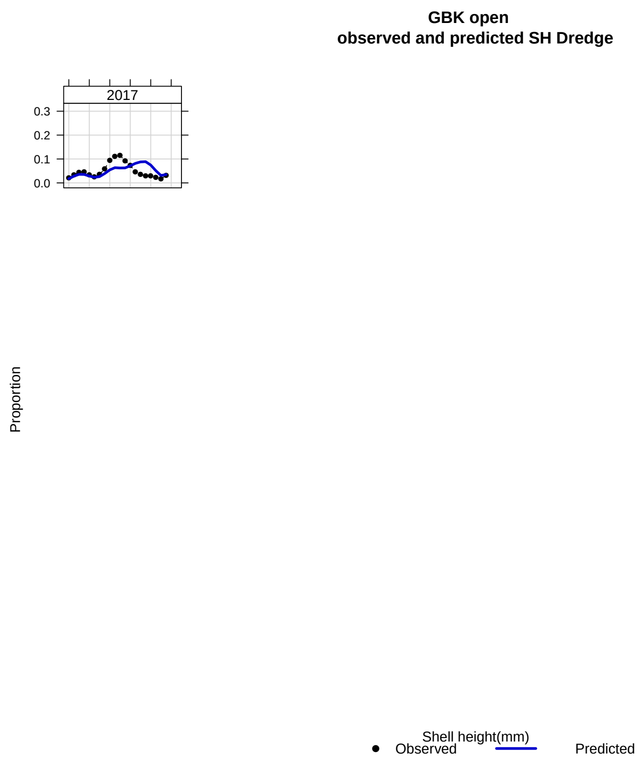
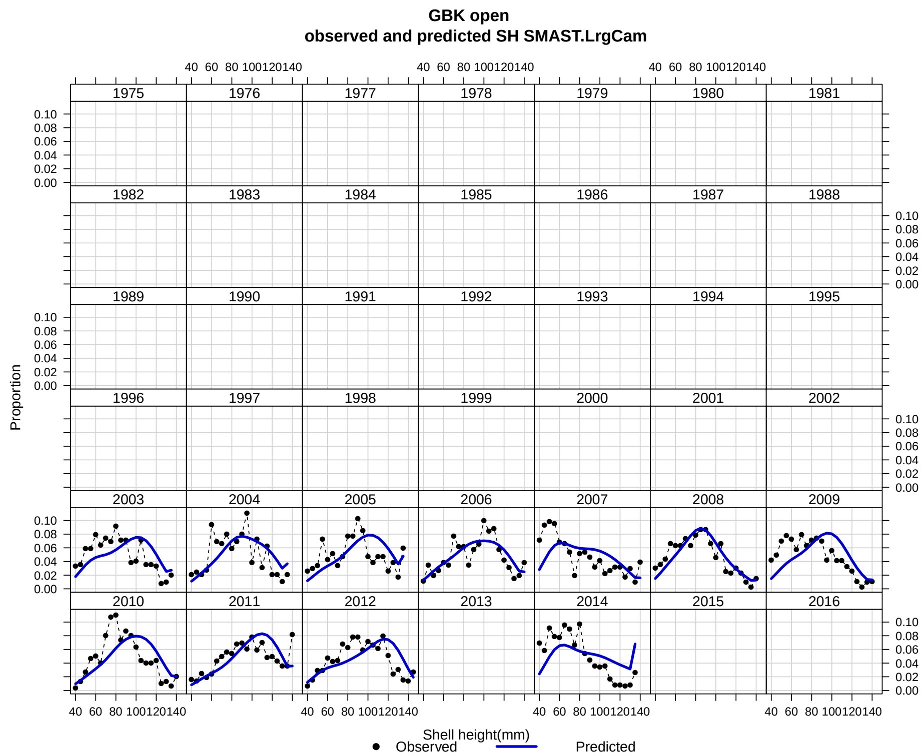


Figure A9.24. Comparison of lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.



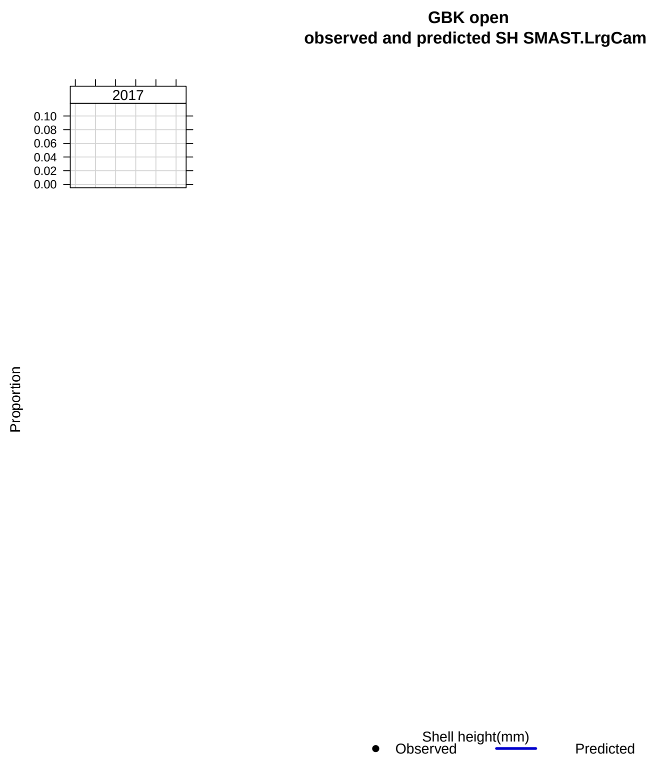
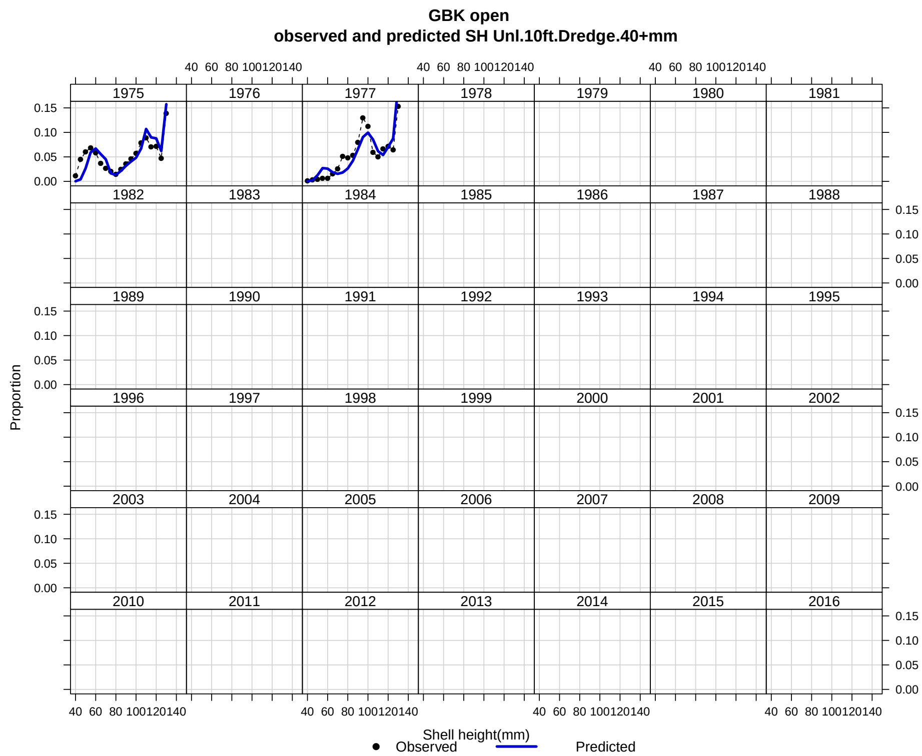


Figure A9.25. Comparison of large drop camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.



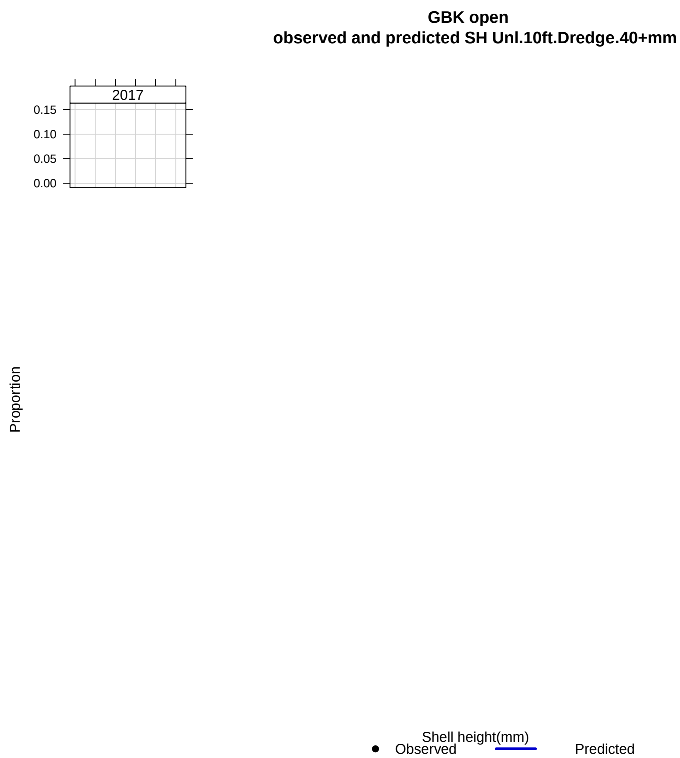


Figure A9.26. Comparison of unlined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.

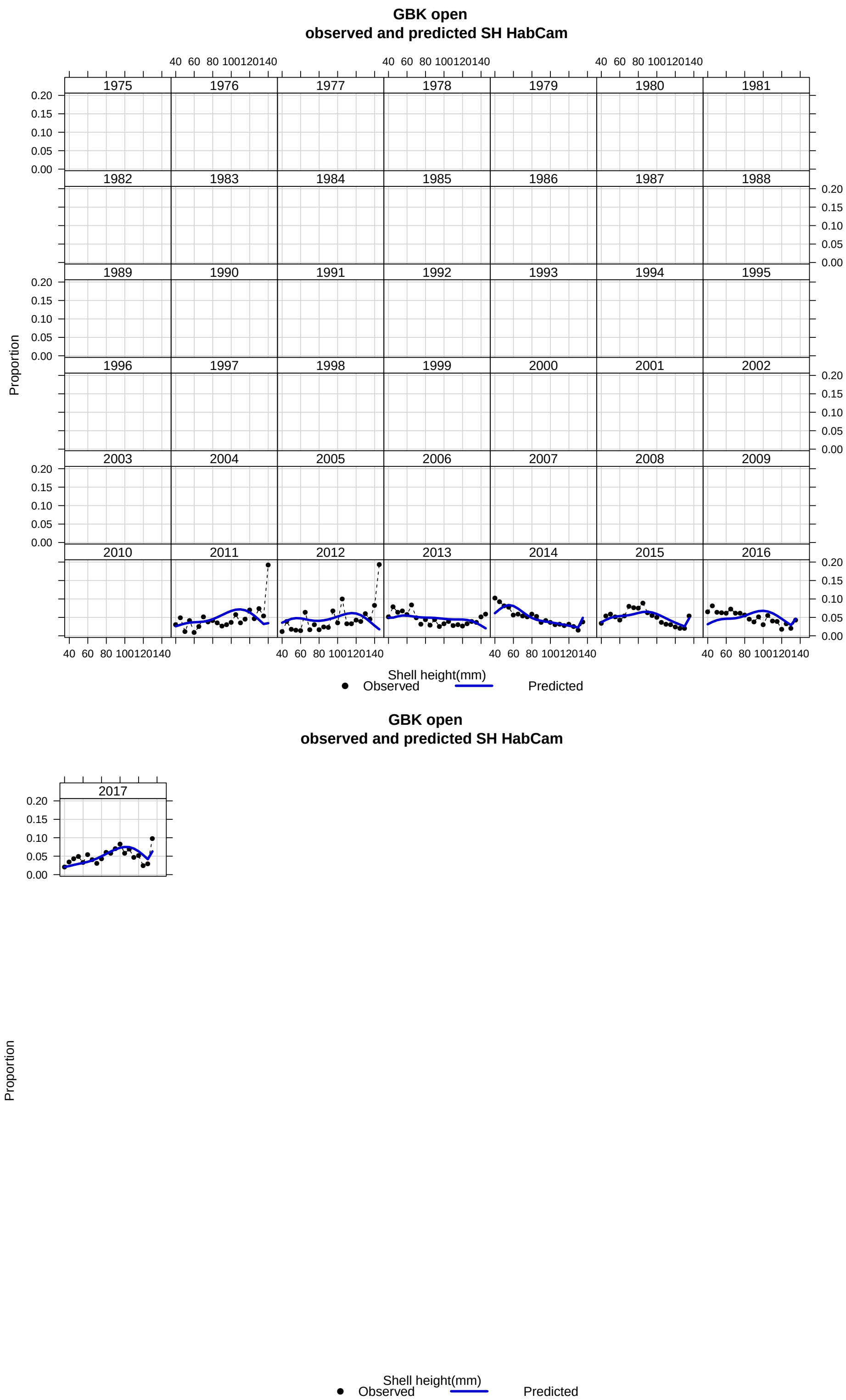
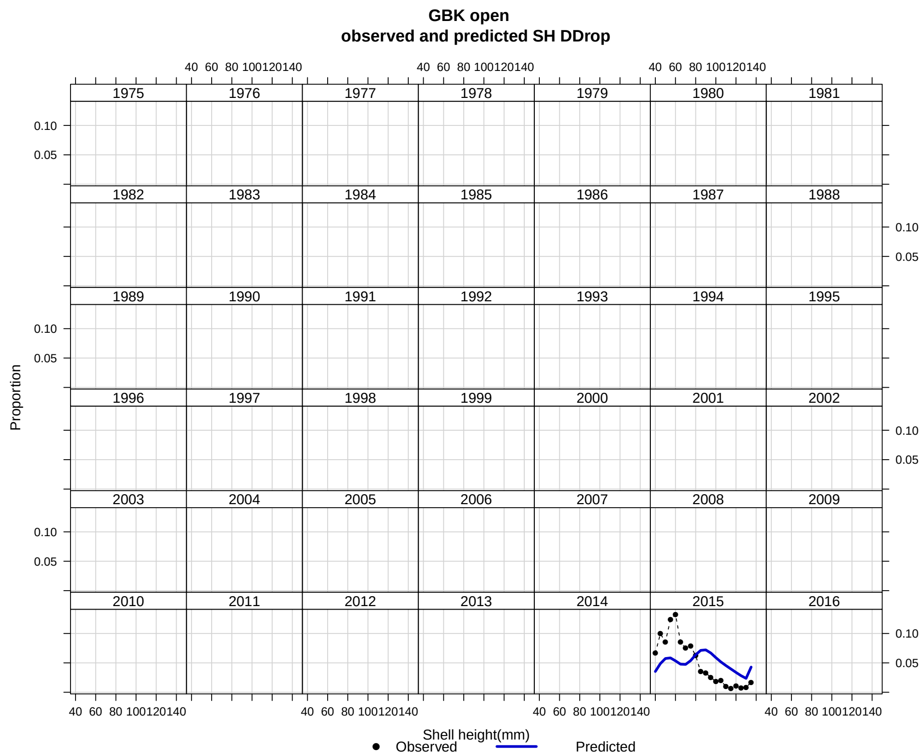


Figure A9.27. Comparison of Habcam survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.



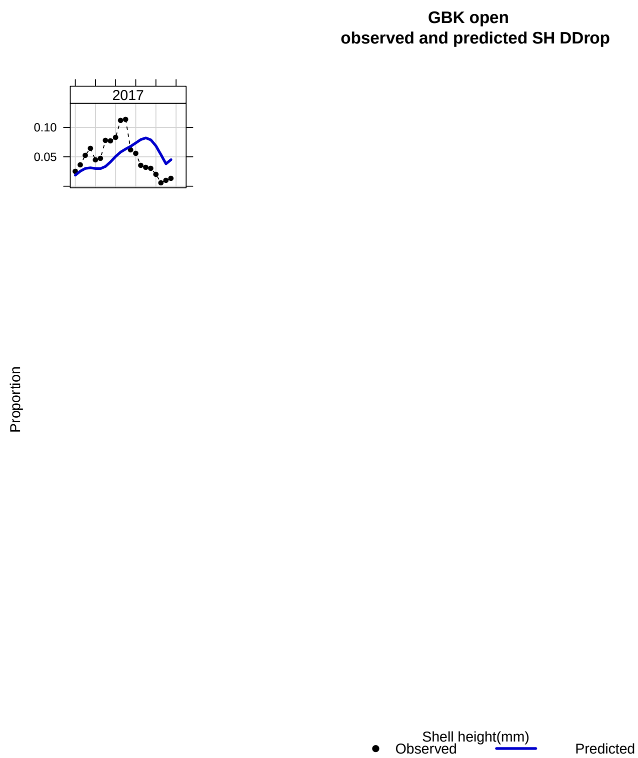


Figure A9.28. Comparison of SMAST digital camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.

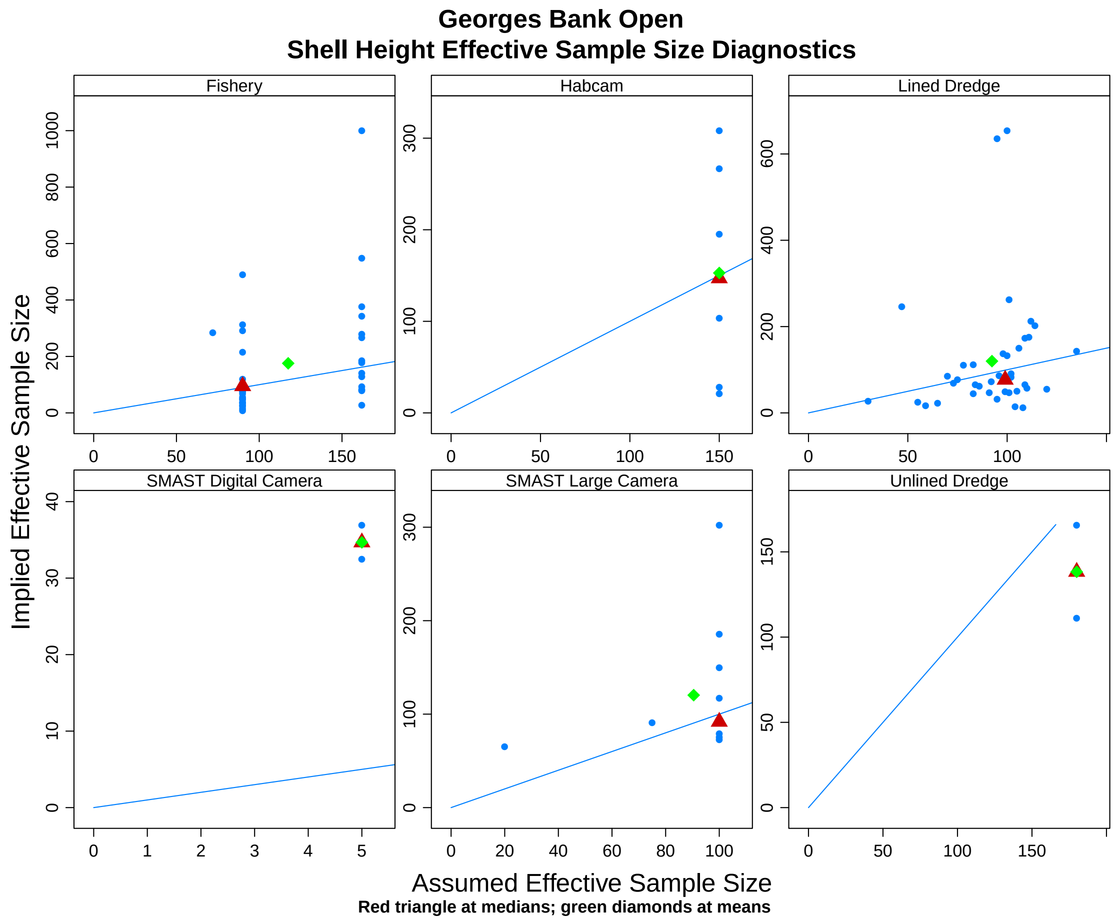


Figure A9.29. Assumed and model implied effective sample sizes for SMAST digital camera (top left), lined dredge (top middle), Habcam (bottom left), large drop camera (bottom middle), and unlined dredge (bottom right) surveys, and the fishery (top right) shell height compositions for Georges Bank open areas. The triangle is the median and the diamond is the mean.

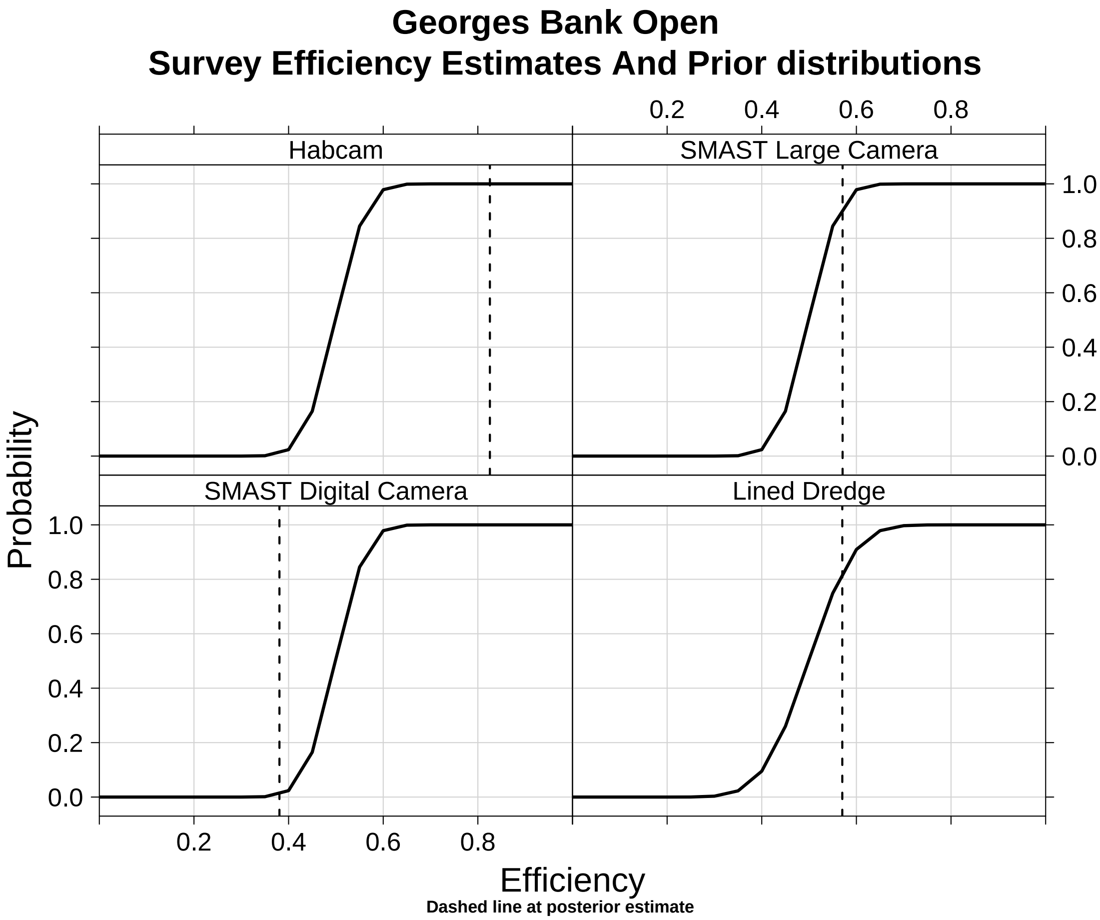


Figure A9.30. Prior cumulative distributions for catchability of Habcam (top left), large drop camera (top right), SMAST digital camera (bottom left), and lined dredge (bottom right) surveys for Georges Bank open areas. The dashed lines are the mean posterior estimate for survey catchability. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5.

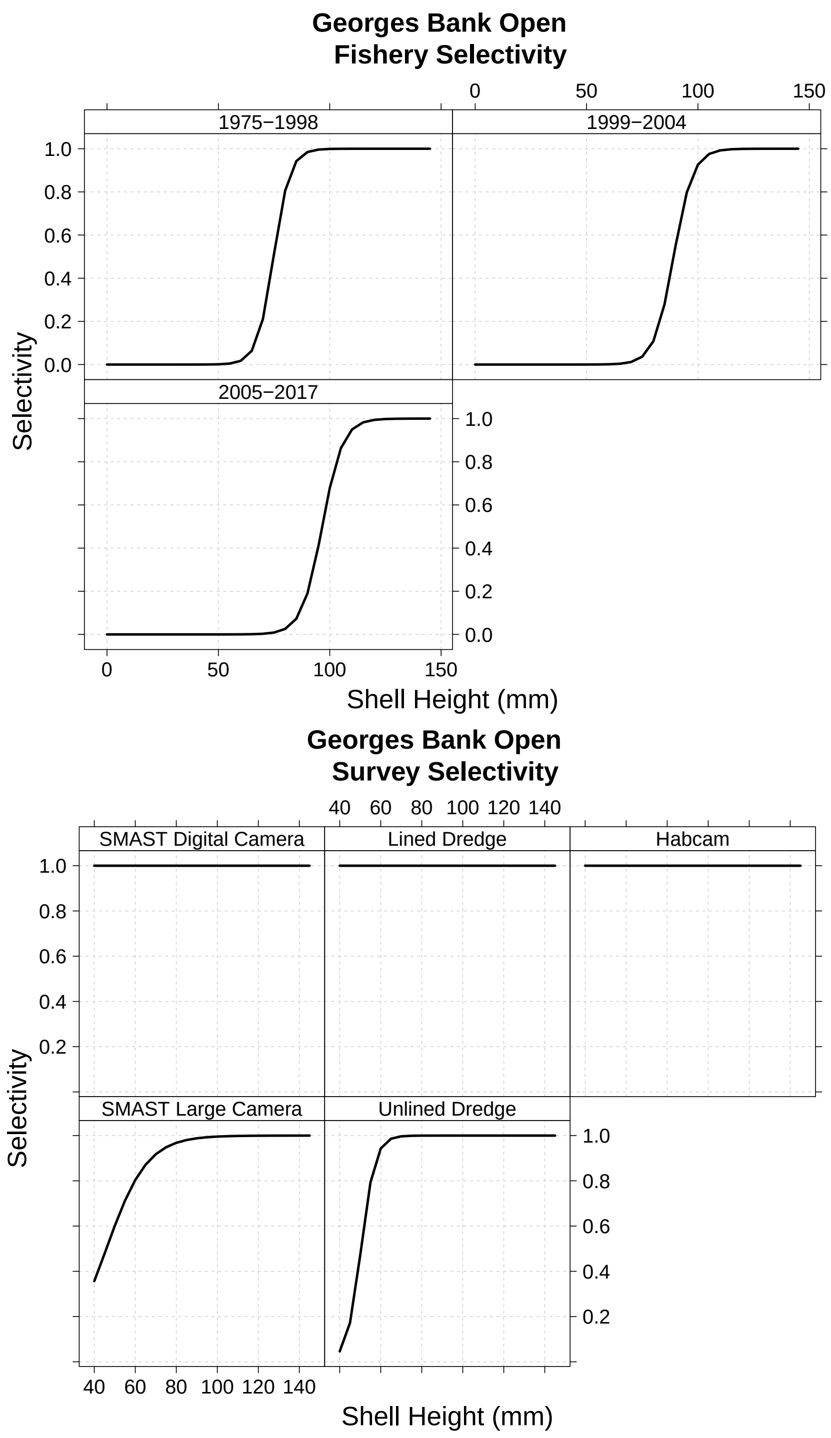


Figure A9.31. Estimated fishery selectivity curves (top) and assumed selectivity curves (bottom) for SMAST digital camera (bottom panel; top left), lined dredge (bottom panel; top middle), Habcam (bottom panel; top right), large drop camera (bottom panel; bottom left), and unlined dredge (bottom panel; bottom middle) surveys for Georges Bank open areas.

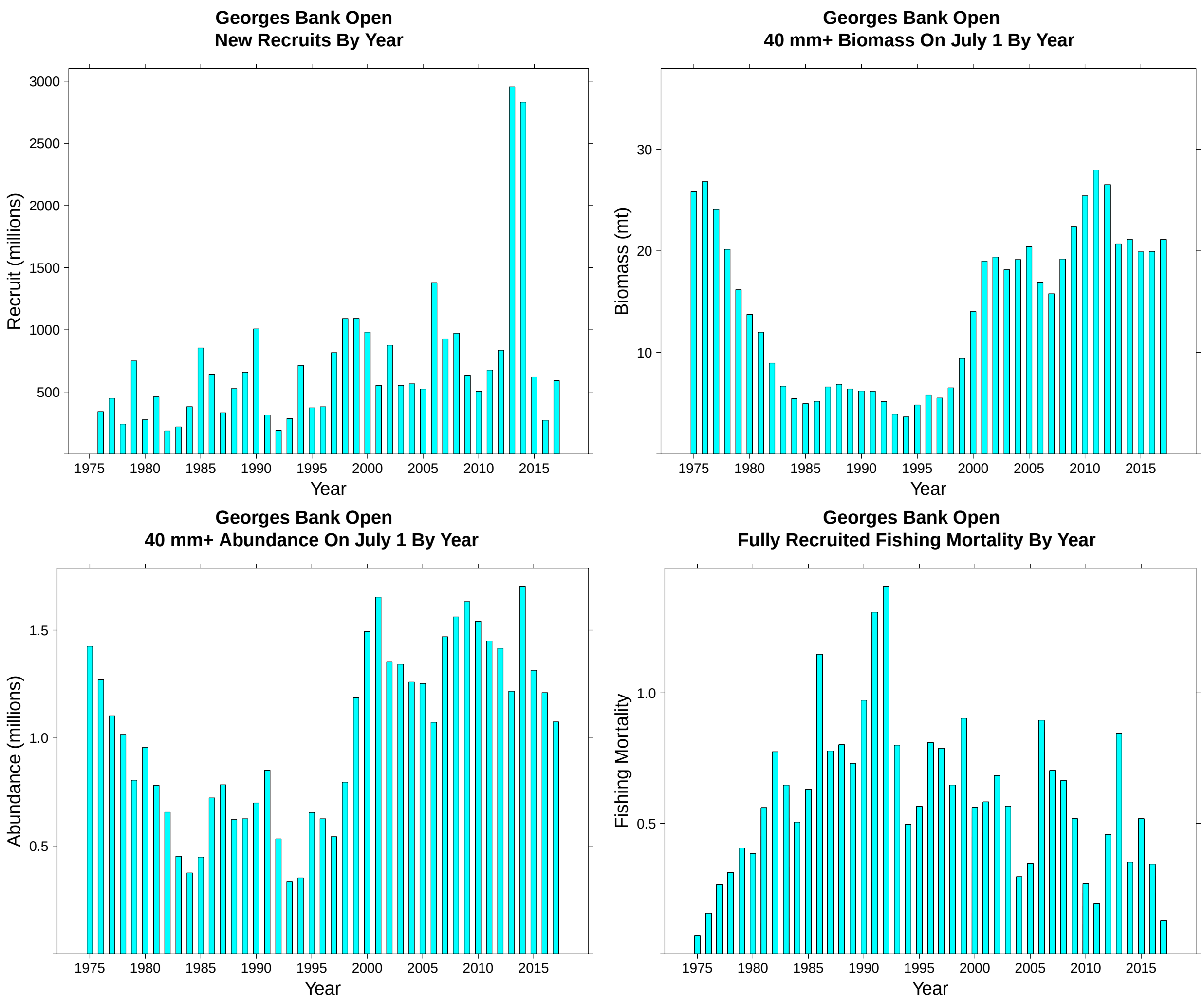


Figure A9.32. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left) and fully recruited fishing mortality (bottom right) for Georges Bank open areas.

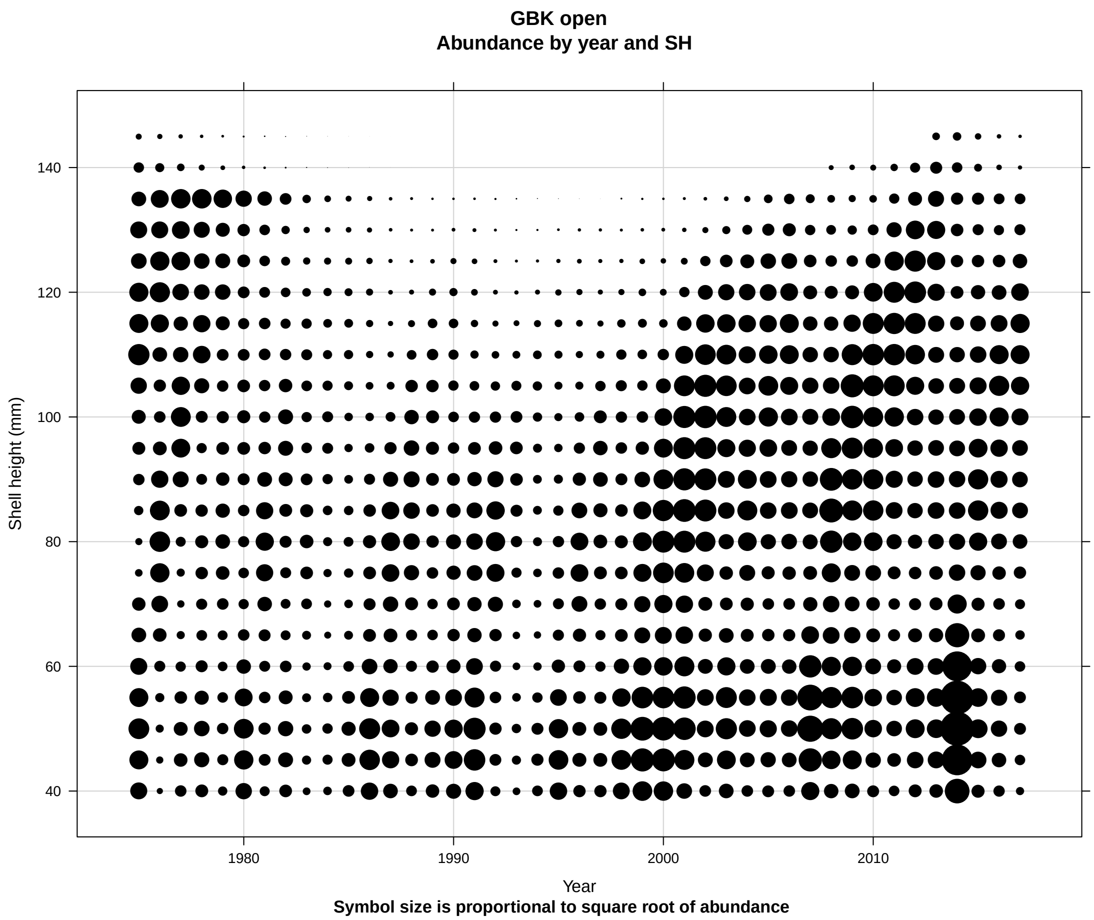


Figure A9.33CASA model estimated abundances at shell height for Georges Bank open areas. Symbol areas are proportional to abundance.

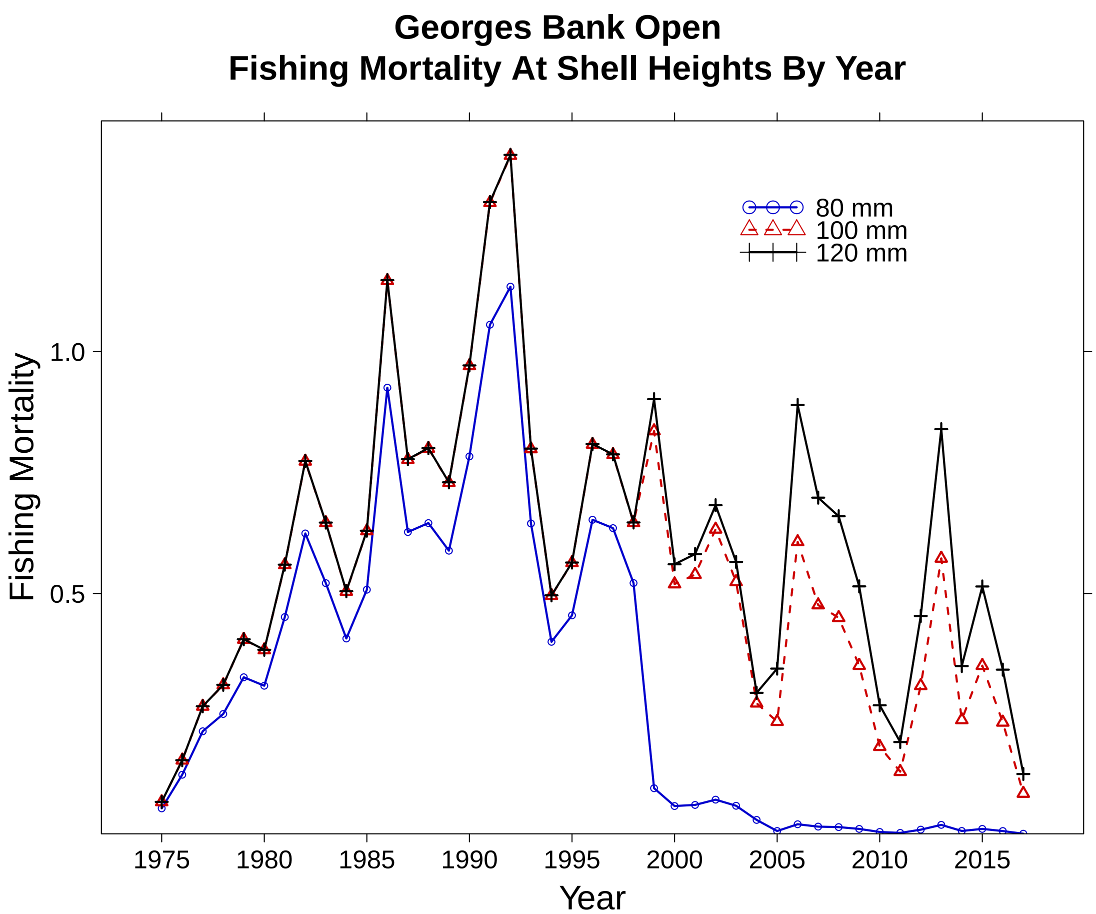


Figure A9.34. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles), and 120 mm SH (dashed line with crosses) for Georges Bank open areas.

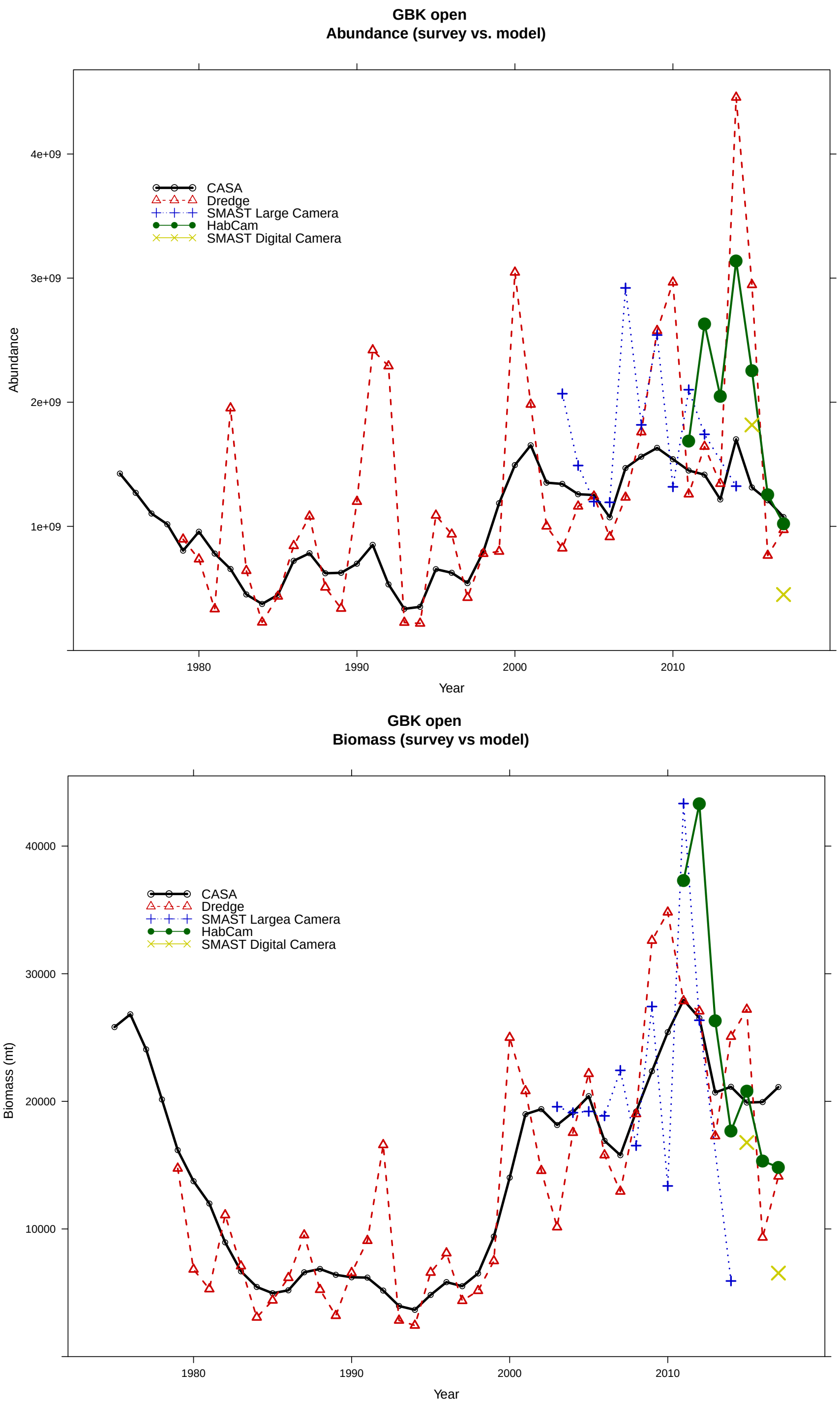


Figure A9.35. Comparison of CASA model estimated abundance (top) and biomass (bottom) with expanded estimates from the lined dredge (red), large drop camera (blue), HabCam (green), and SMAST digital camera (light green) for Georges Bank open areas.

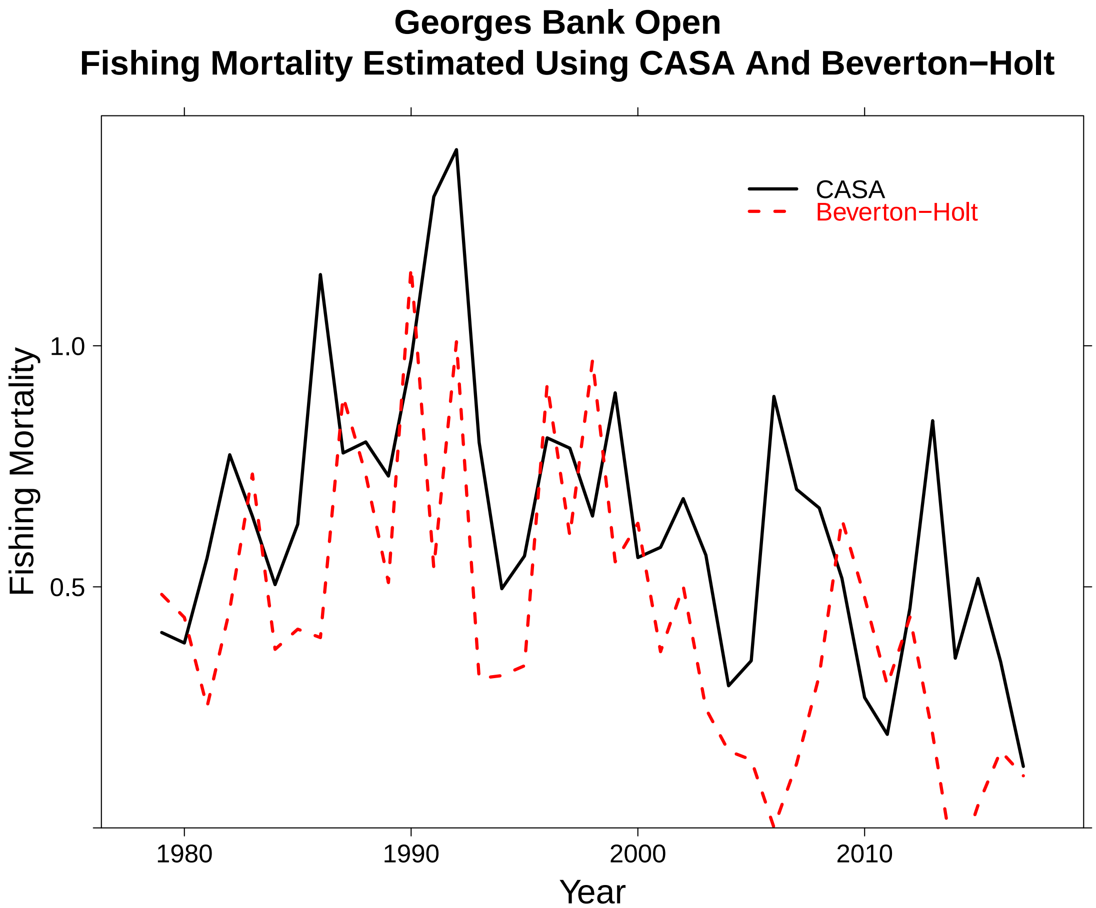


Figure A9.36. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium length based estimator for Georges Bank open areas.

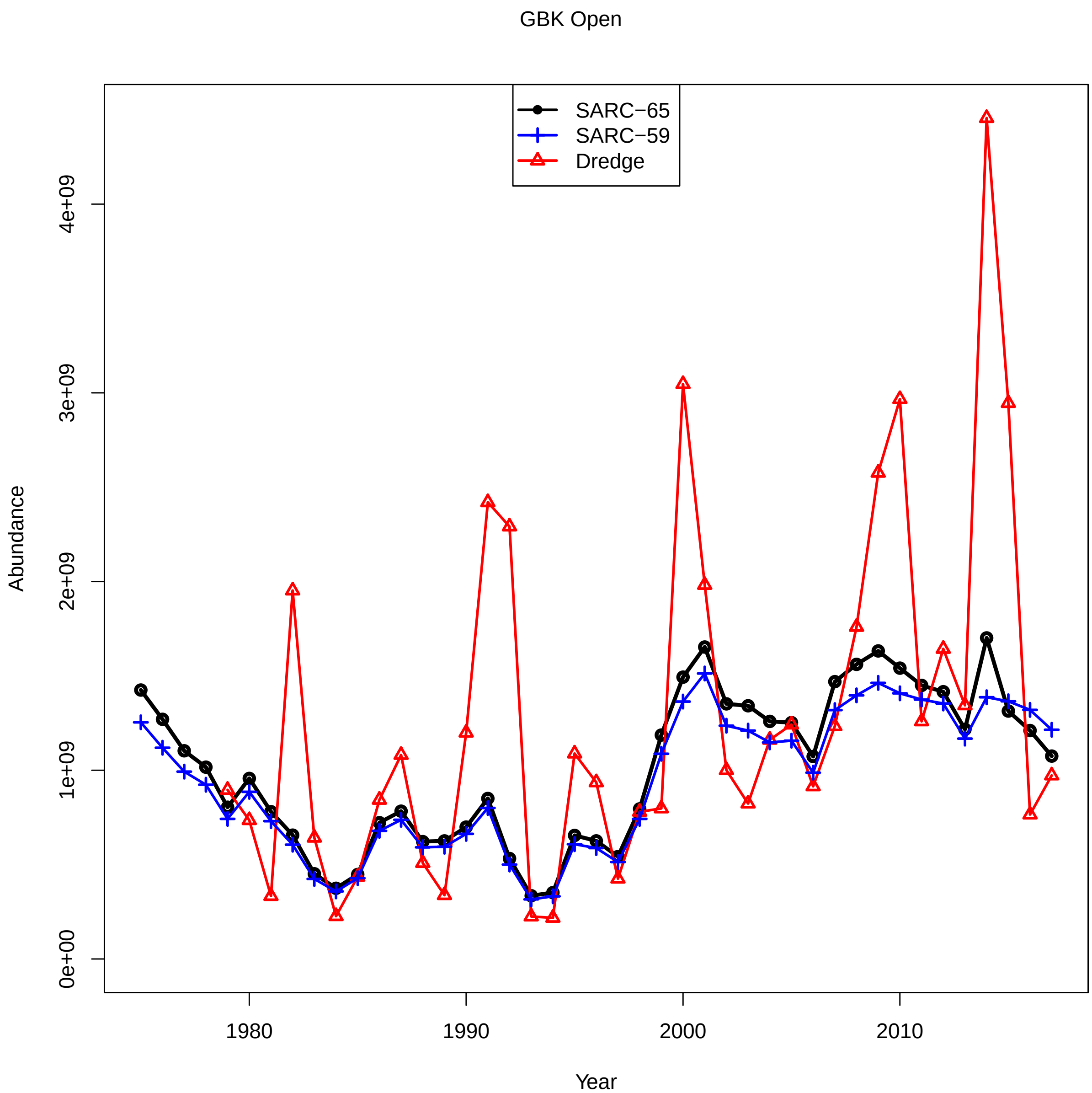


Figure A9.37. Comparison of estimated abundance from current and SARC-59 CASA model configurations, along with lined dredge survey abundance index for Georges Bank open areas.

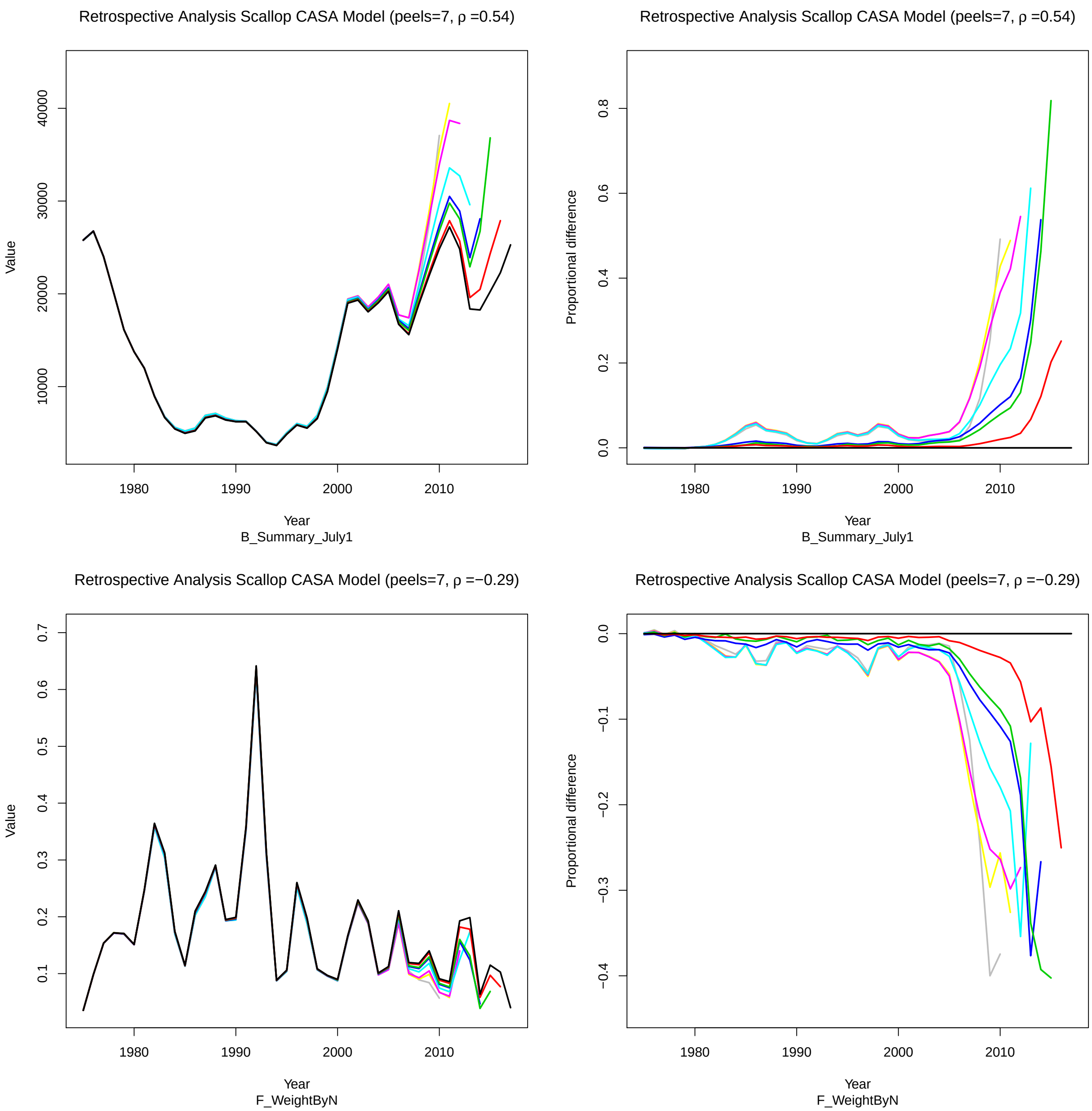


Figure A9.38. Retrospective plots for biomass and fishing mortality for Georges Bank open areas. Retrospectives are shown on both absolute and relative scales.

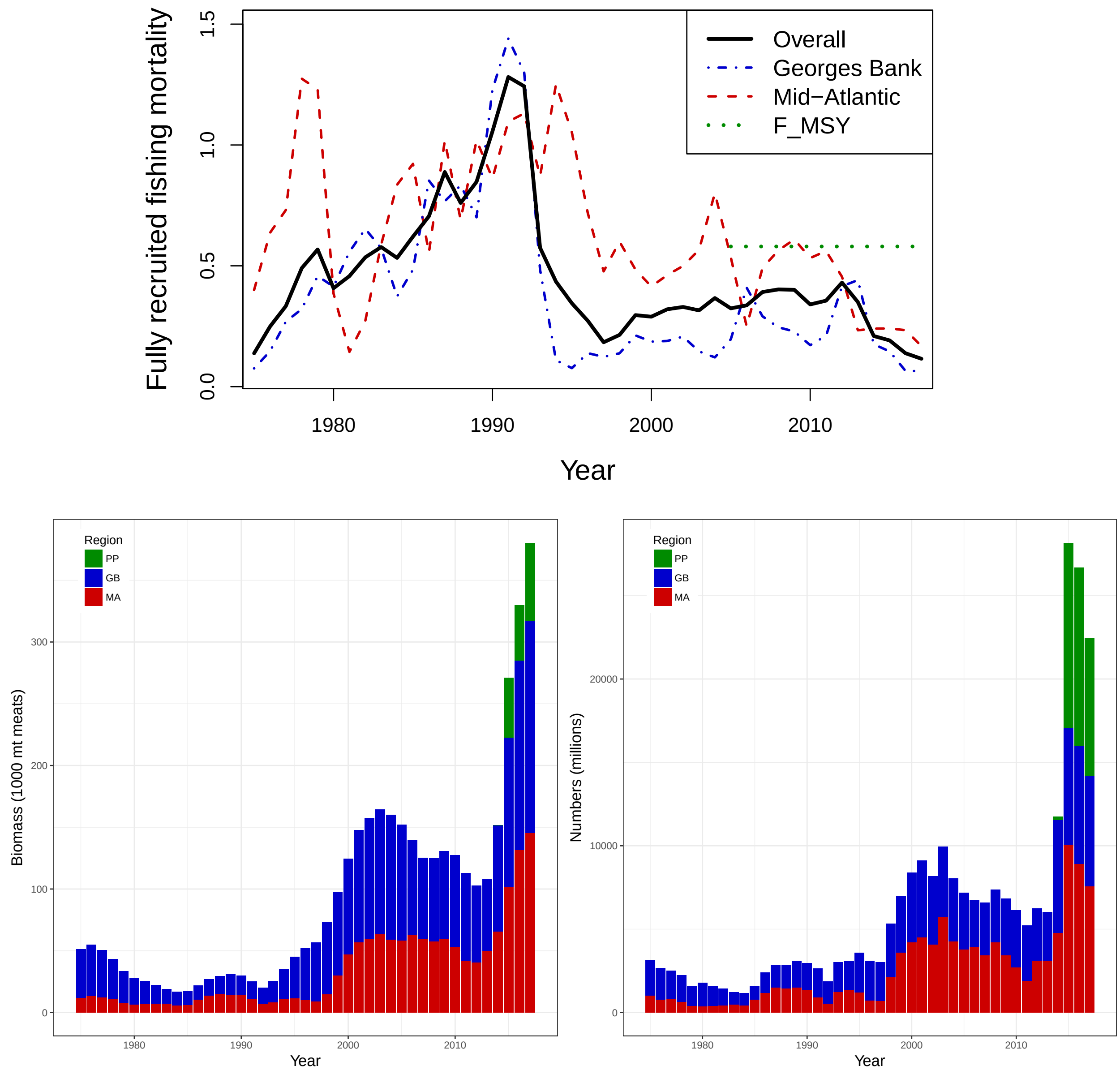


Figure A9.39. Estimated fully recruited fishing mortality (top), biomass (bottom left), and abundance (bottom right) including Habcam biomass and abundance estimates of DSENLS scallops (pp) for Georges Bank (open and closed combined) and Mid-Atlantic sea scallops.

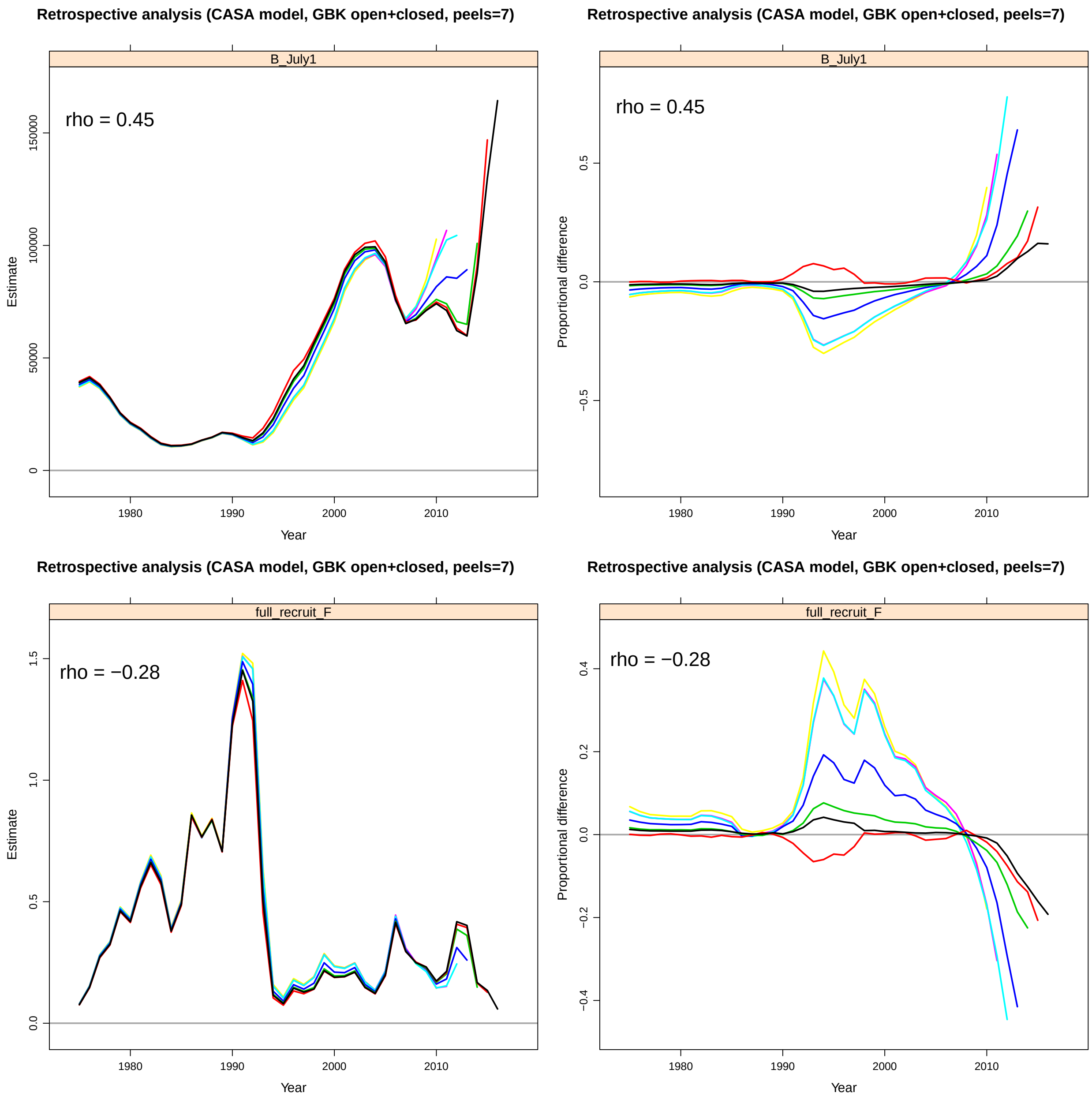


Figure A9.40. Retrospective plots for biomass and fishing mortality for the combined Georges Bank stock. Retrospectives are shown on both absolute and relative scales.

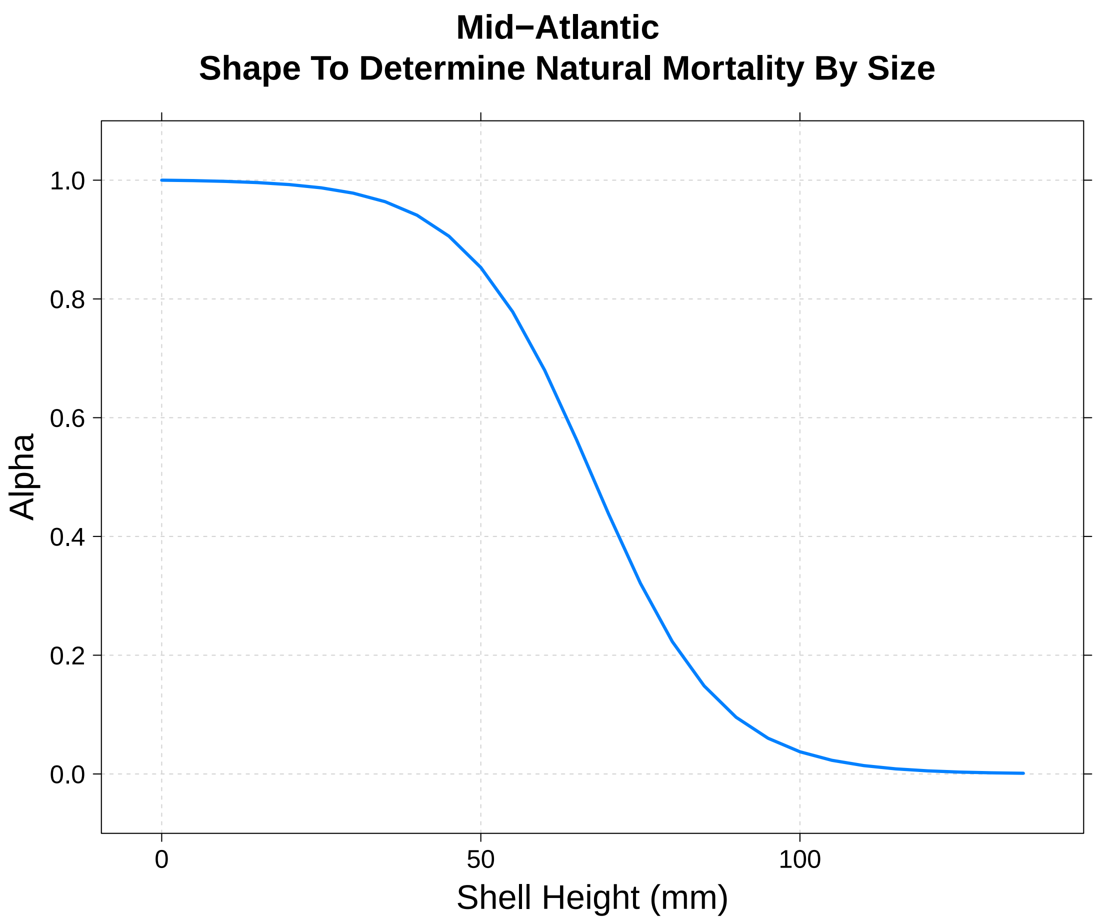


Figure A9.41. Logistic curve used to partition juvenile and adult natural mortality for Mid-Atlantic areas.

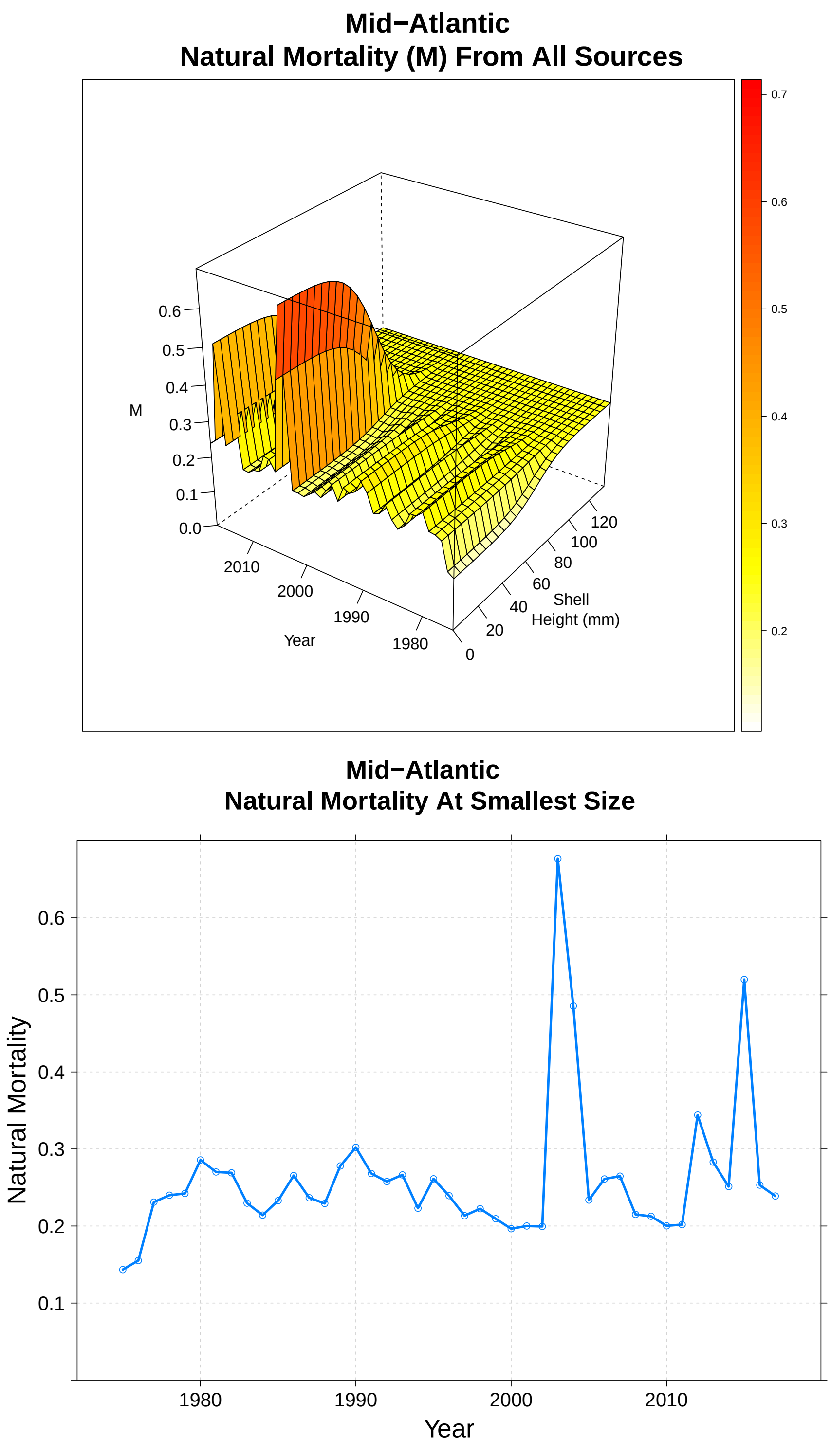


Figure A9.42. Estimated natural mortality by size (top) and for smallest size group (bottom) from 1975 to 2017 for Mid-Atlantic areas.

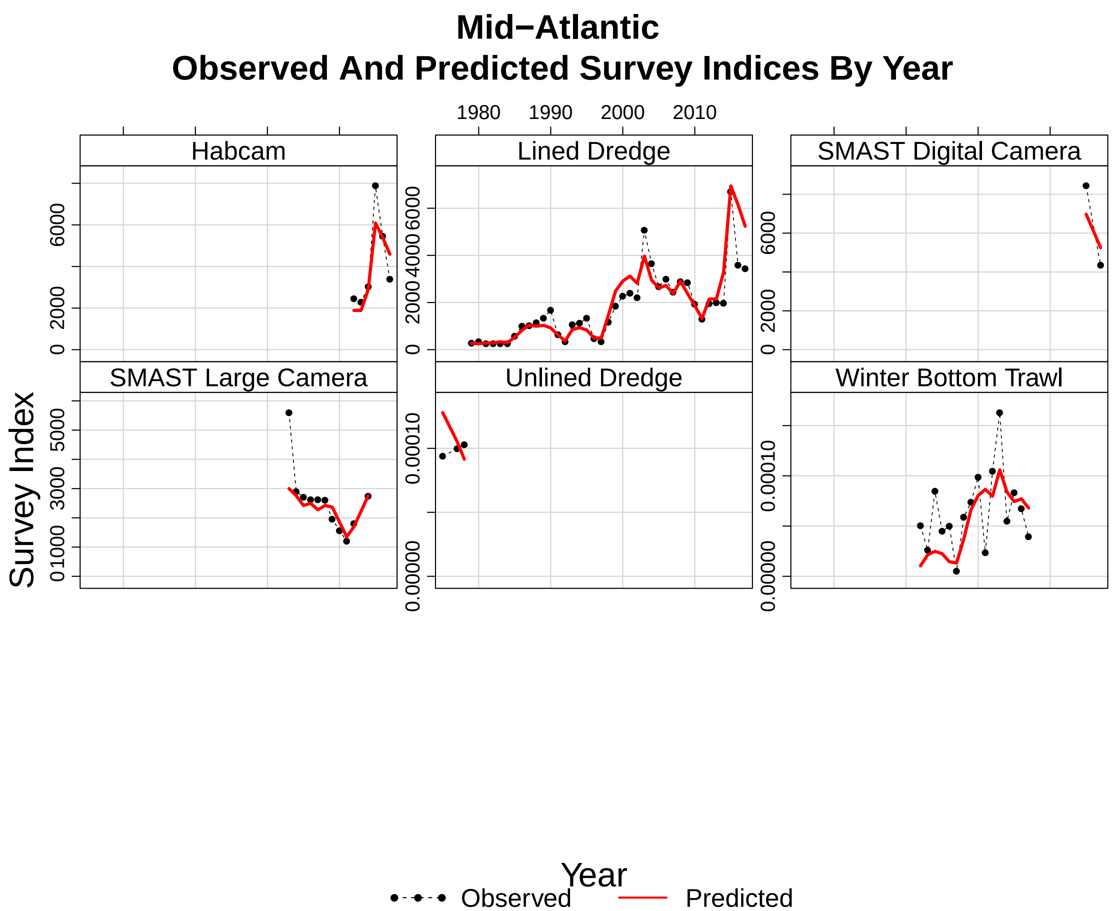


Figure A9.43. Observed survey trend (solid circles) and corresponding model estimates (lines) for the SMAST digital camera (top left), lined dredge (top middle), Habcam (top right), large drop camera (bottom left), unlined dredge (bottom middle), and winter bottom trawl (bottom right) surveys on Mid-Atlantic areas.

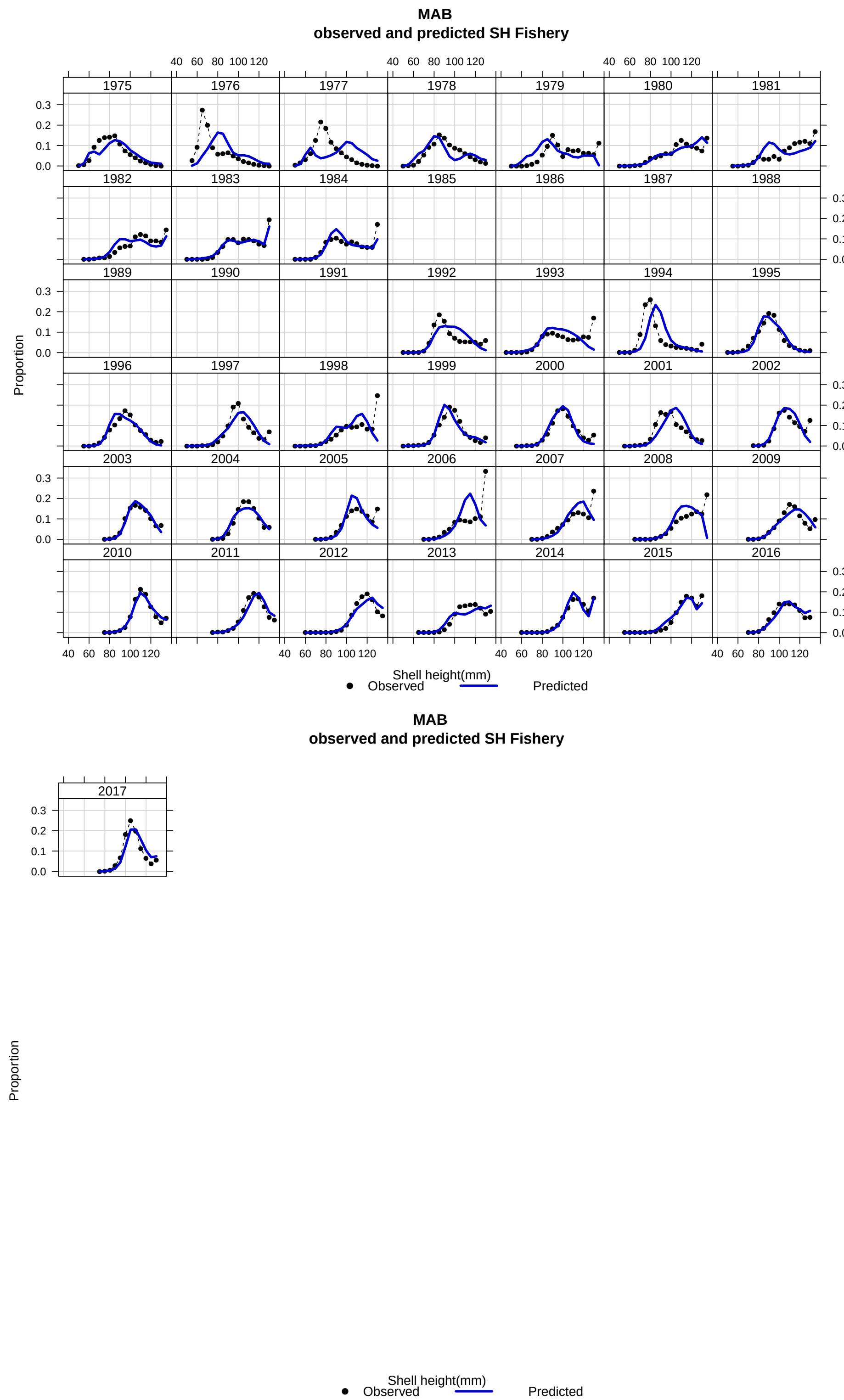
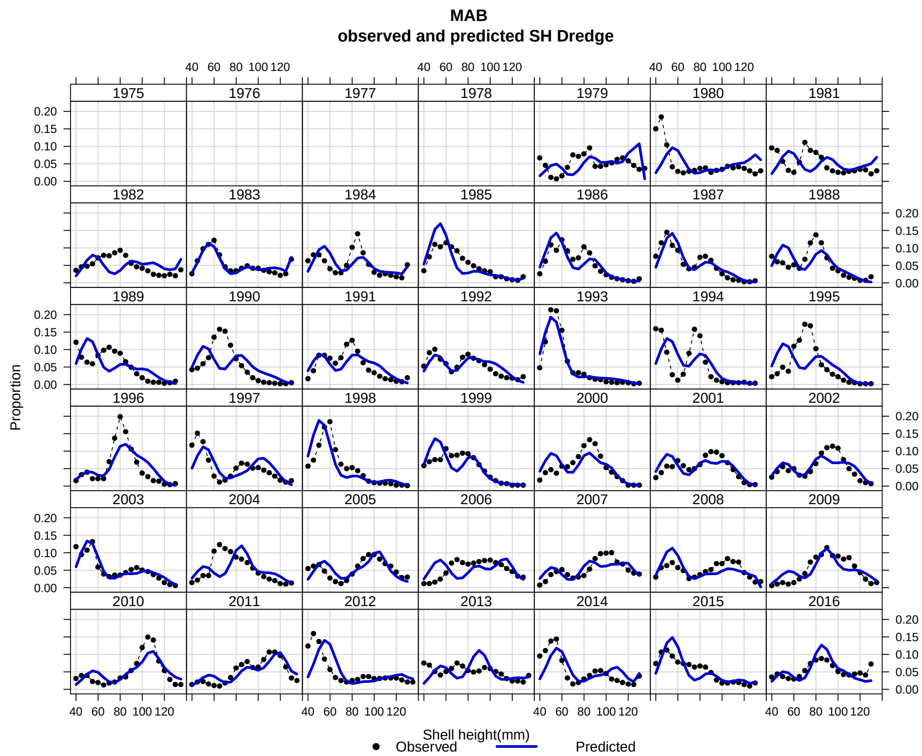


Figure A9.44. Comparison of observed fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Mid-Atlantic areas.



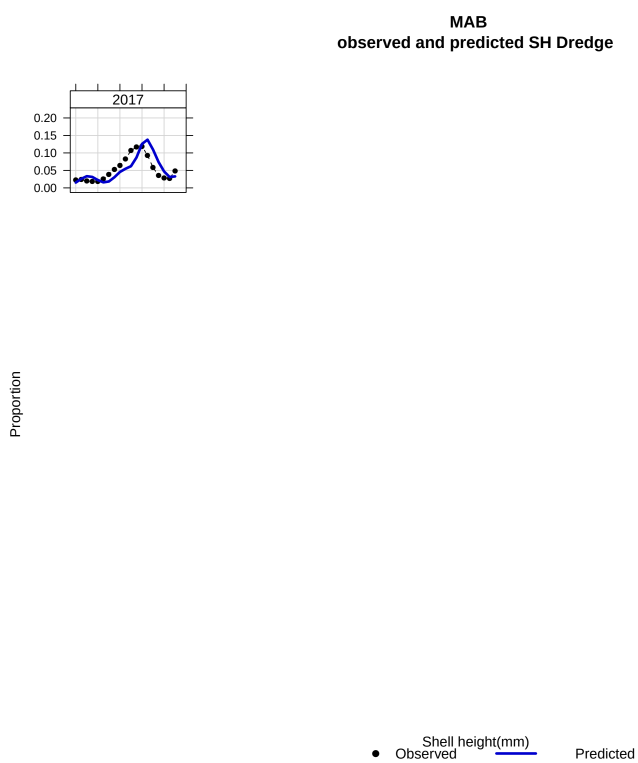
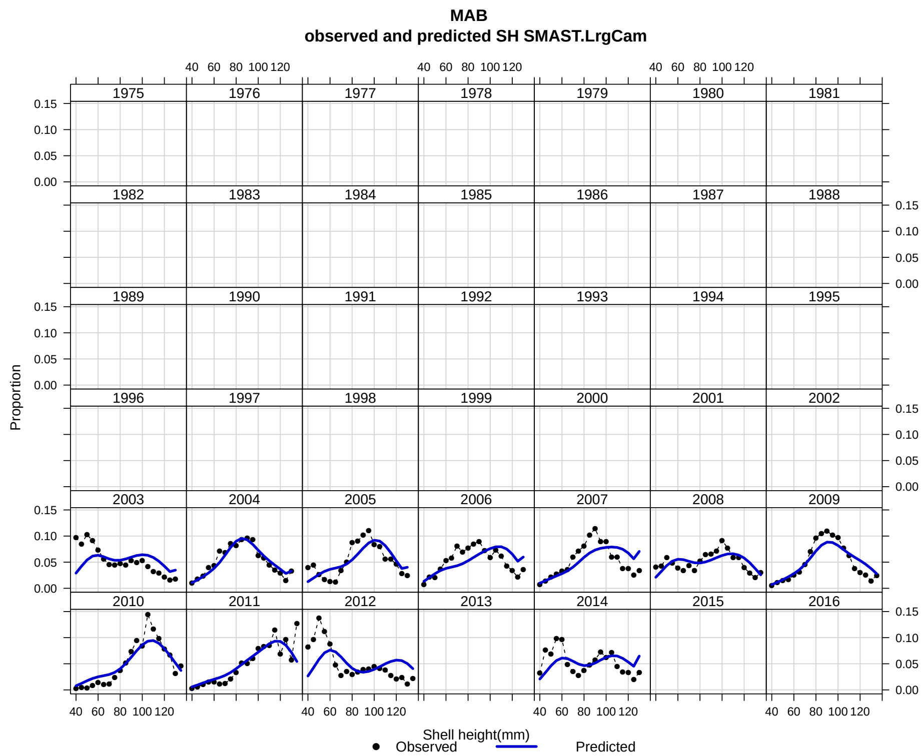


Figure A9.45. Comparison of lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Mid-Atlantic areas.



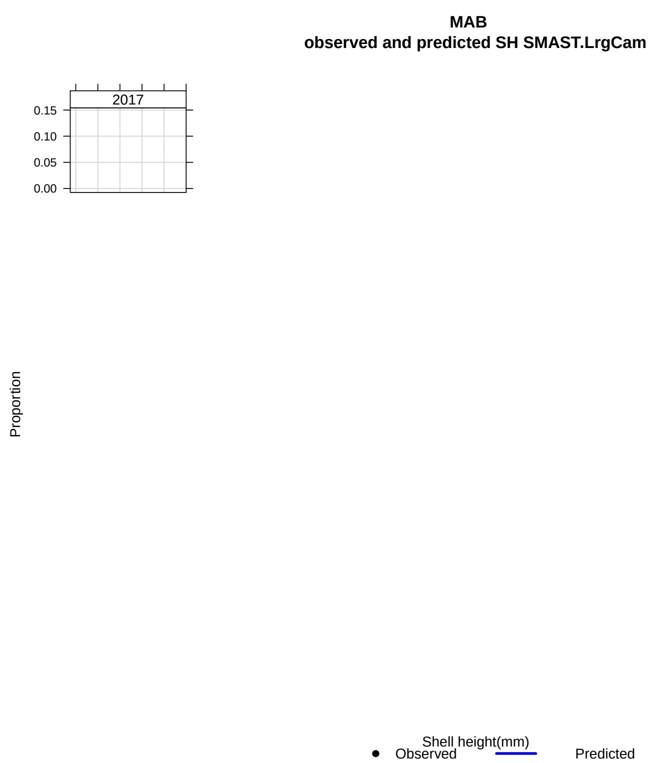


Figure A9.46. Comparison of large drop camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Mid-Atlantic areas.

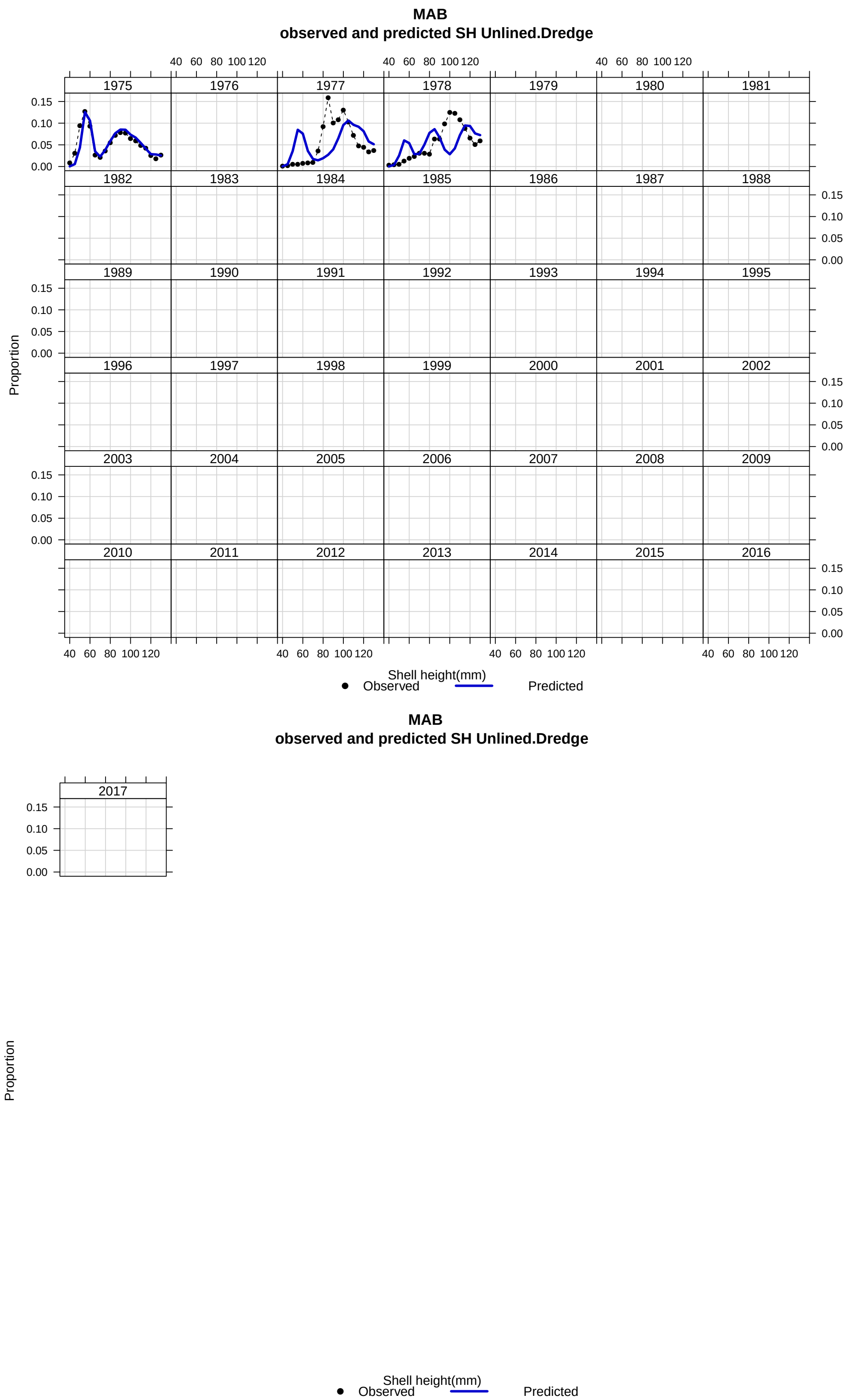


Figure A9.47. Comparison of unlined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Mid-Atlantic areas.

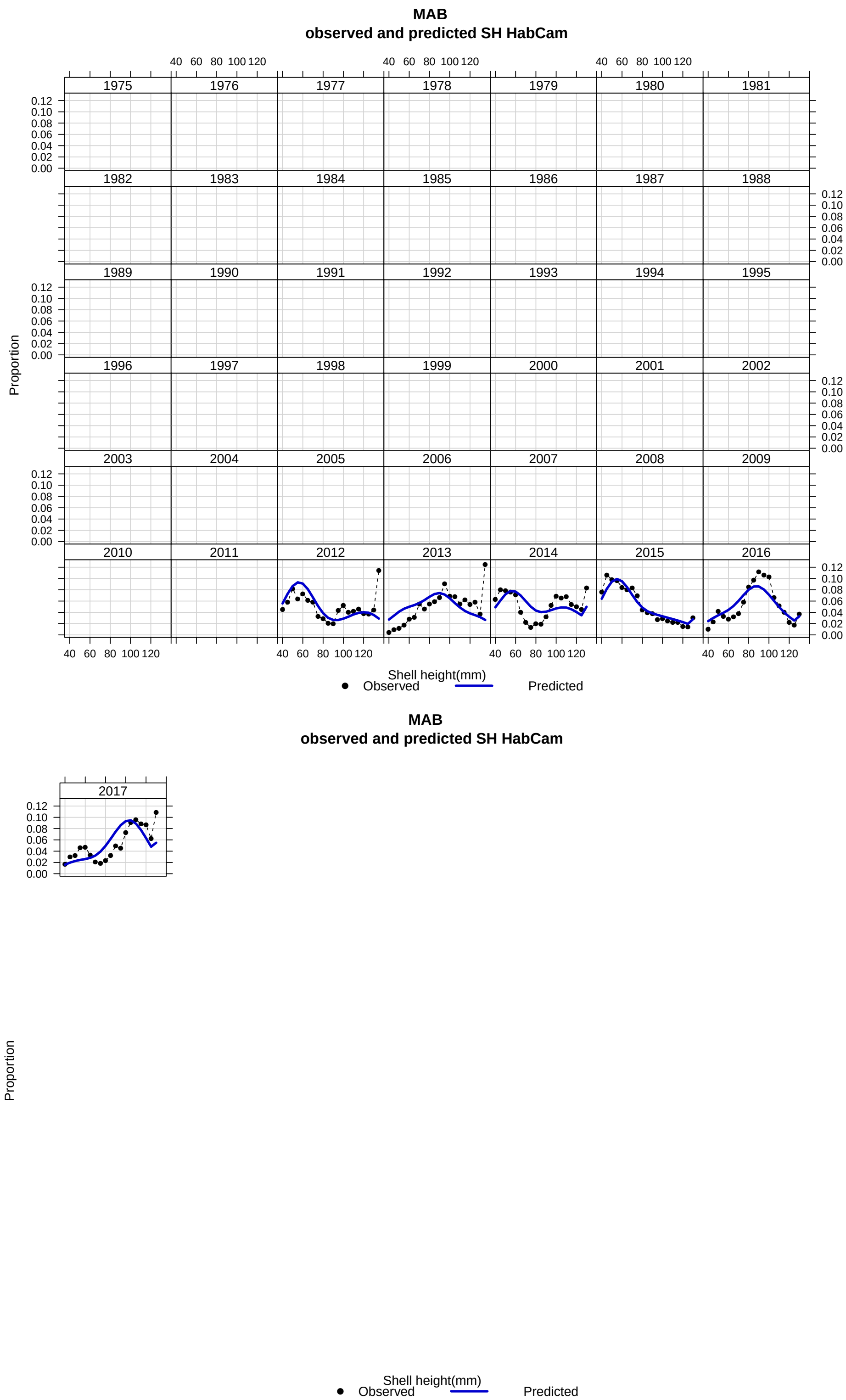
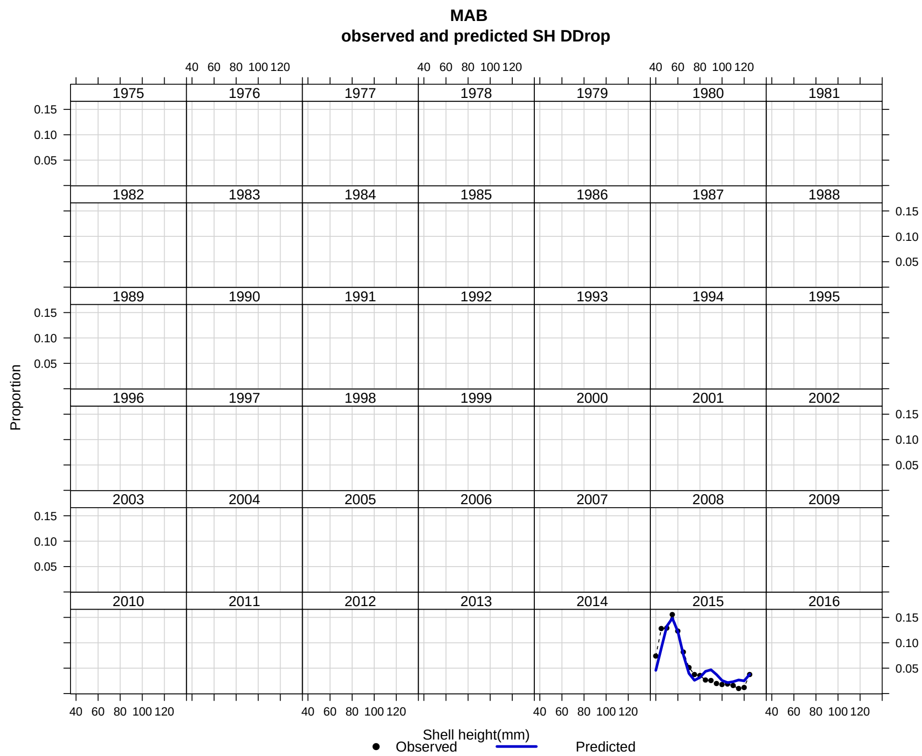


Figure A9.48. Comparison of Habcam survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Mid-Atlantic areas.



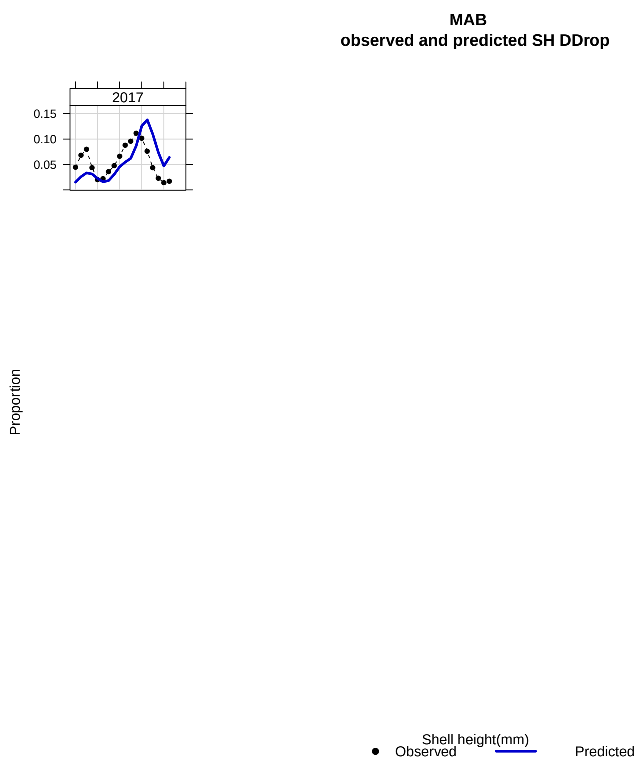


Figure A9.49. Comparison of SMAST digital camera survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Mid-Atlantic areas.

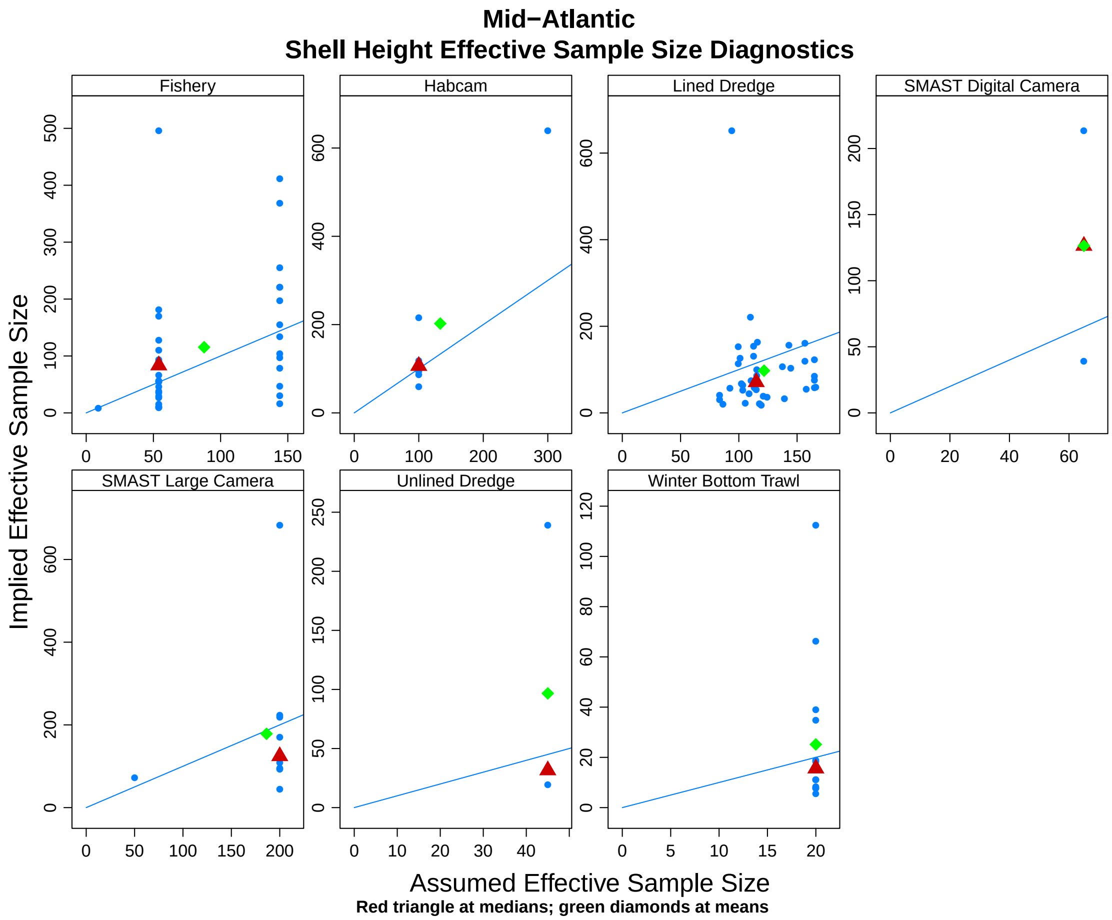


Figure A9.50. Assumed and model implied effective sample sizes for SMAST digital camera, lined dredge, Habcam (top from left to right), large drop camera, unlined dredge, and winter bottom trawl (bottom from left to right) surveys, and the fishery (top third from left) shell height compositions for Mid-Atlantic areas. The triangle is the median and the diamond is the mean.

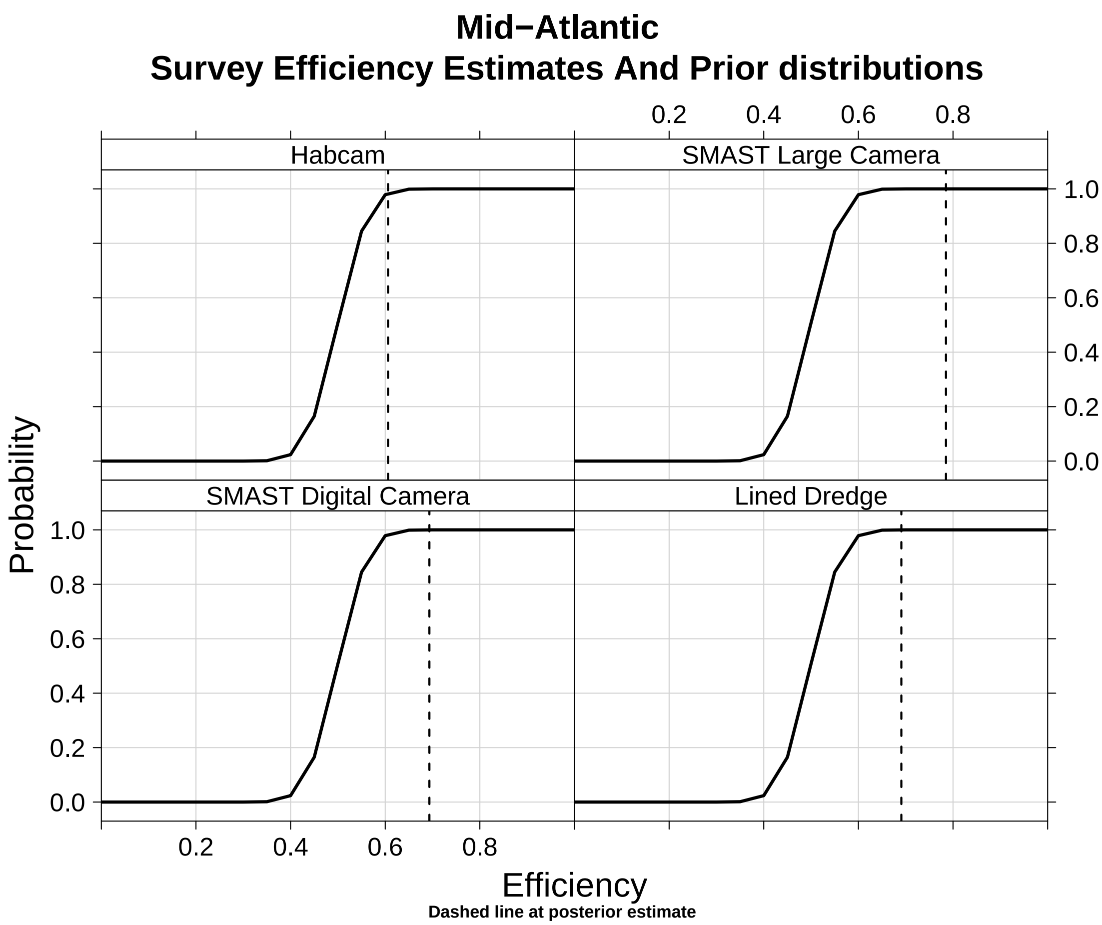


Figure A9.51. Prior cumulative distributions for catchability of Habcam (top left), large drop camera (top right), SMAST digital camera (bottom left), and lined dredge (bottom right) surveys for Mid-Atlantic areas. The dashed lines are the mean posterior estimate for survey catchability. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5.

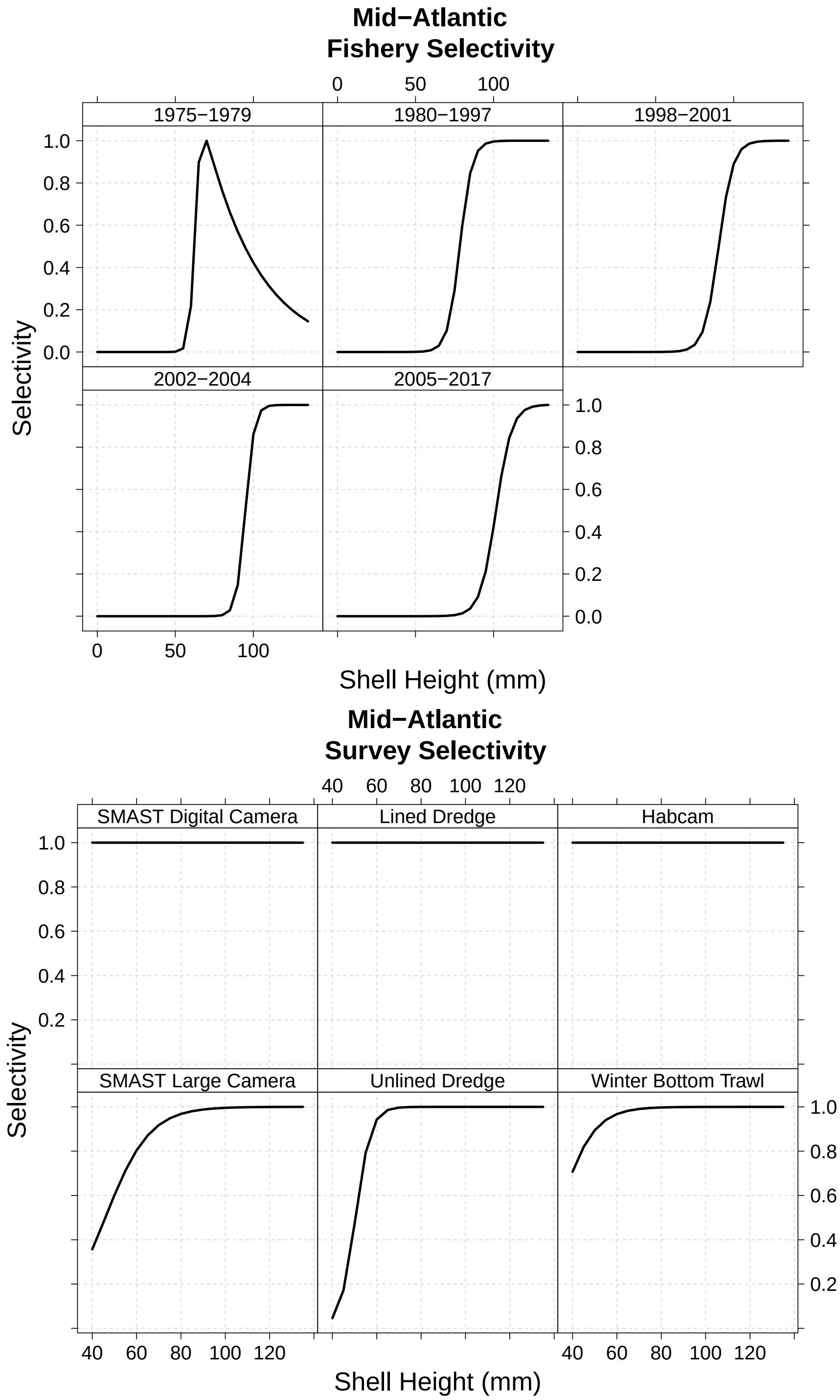


Figure A9.52. Estimated fishery selectivity curves (top) and assumed selectivity curves (bottom) for SMAST digital camera (bottom panel; top left), lined dredge (bottom panel; top middle), Habcam (bottom panel; top right), large drop camera (bottom panel; bottom left), and unlined dredge (bottom panel; bottom middle) surveys for Mid-Atlantic areas.

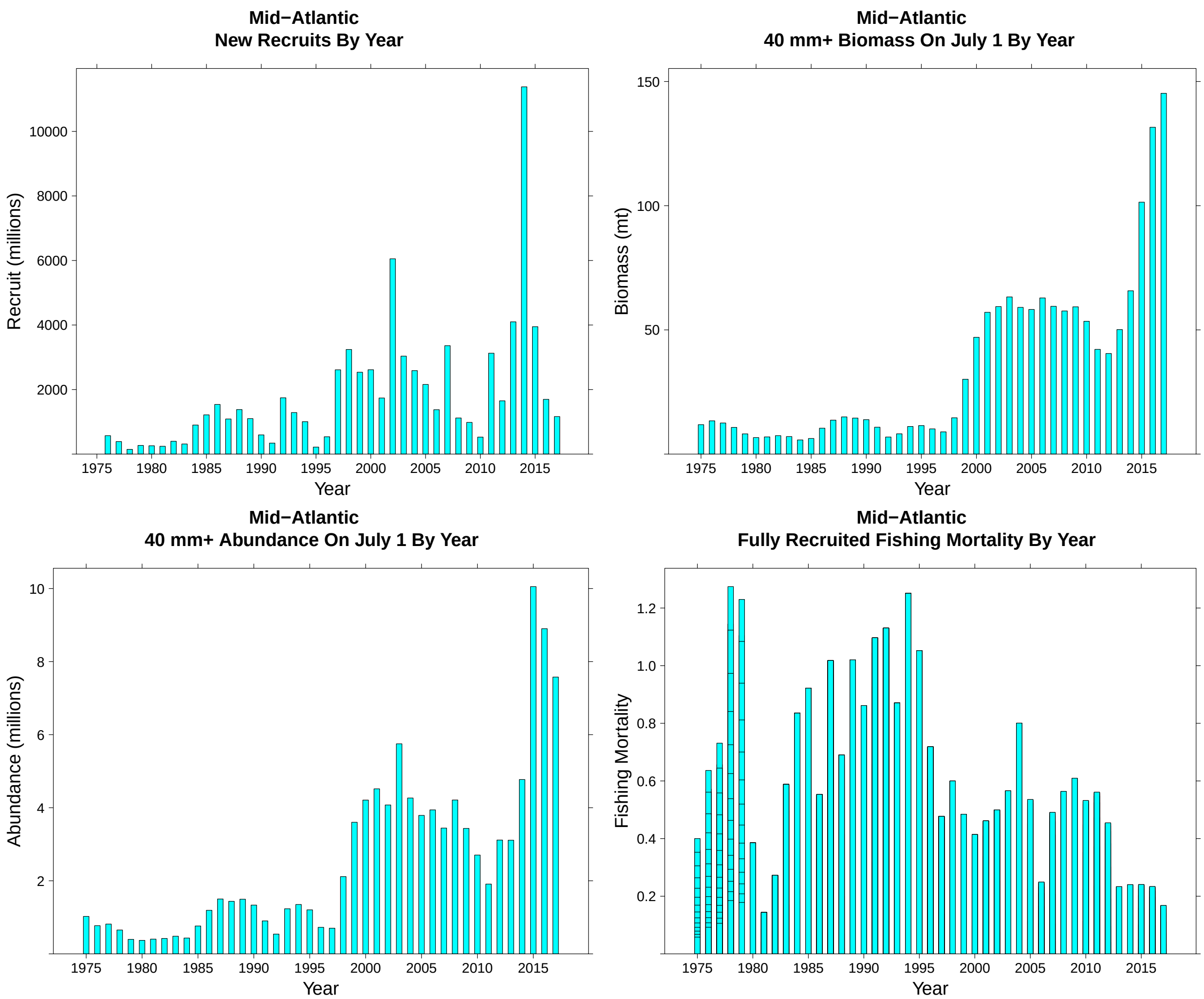


Figure A9.53. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left) and fully recruited fishing mortality (bottom right) for Mid-Atlantic areas.

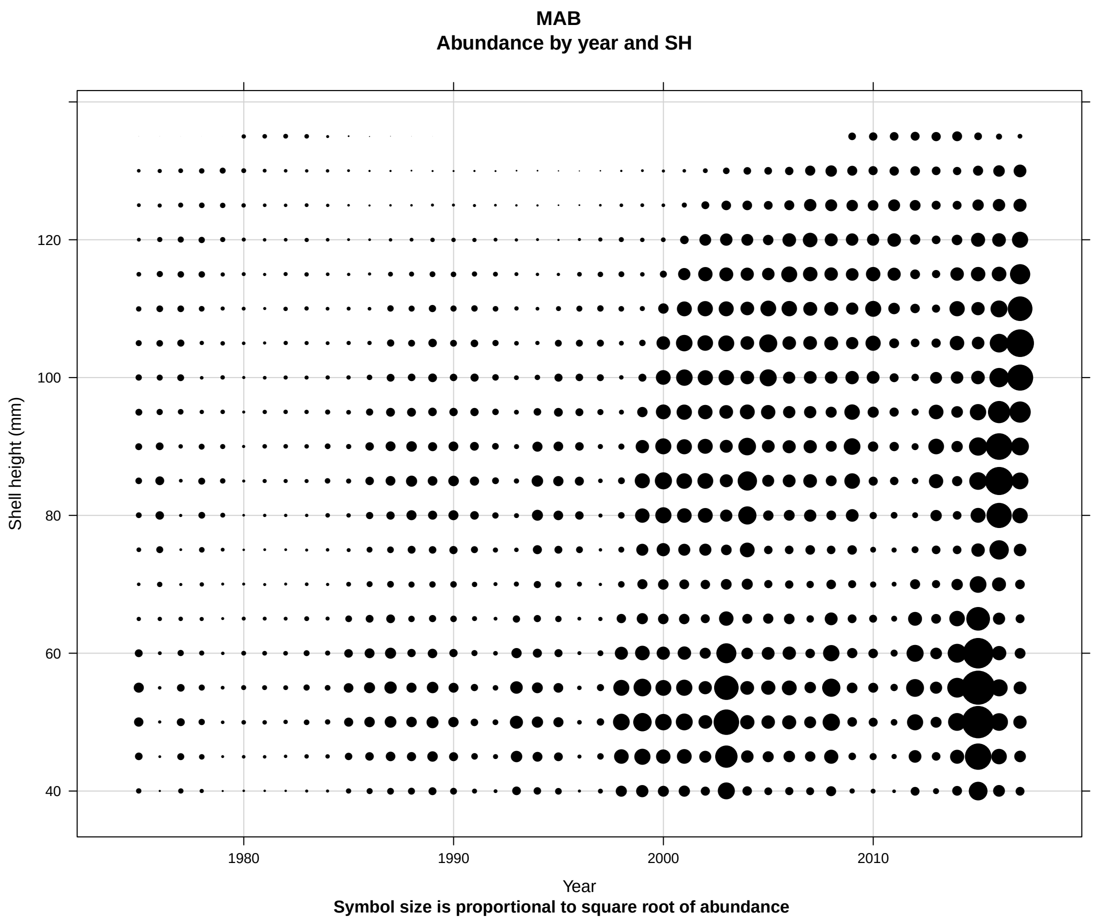


Figure A9.54. CASA model estimated abundances at shell height for Mid-Atlantic areas. Symbol areas are proportional to abundance.

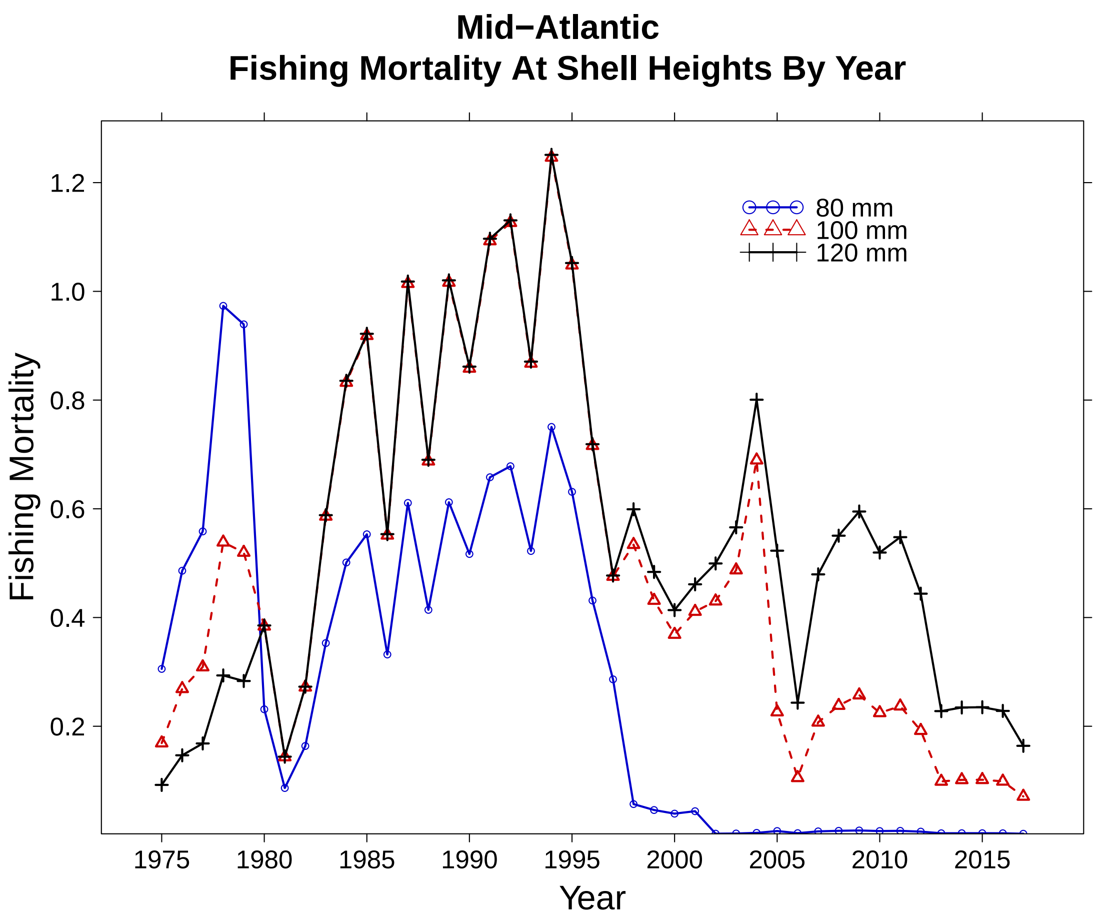
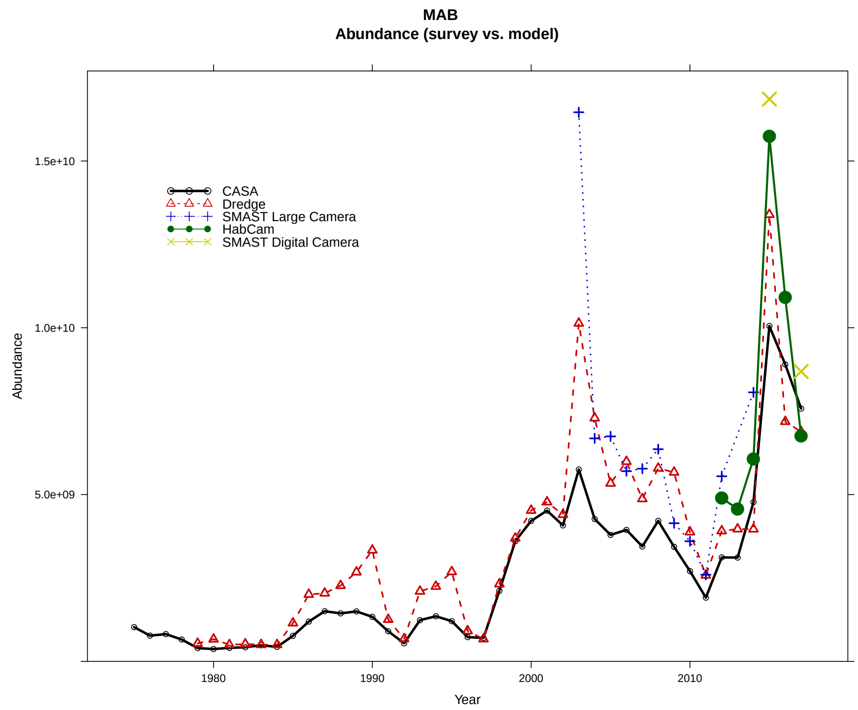


Figure A9.55. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles), and 120 mm SH (dashed line with crosses) for Mid-Atlantic areas.



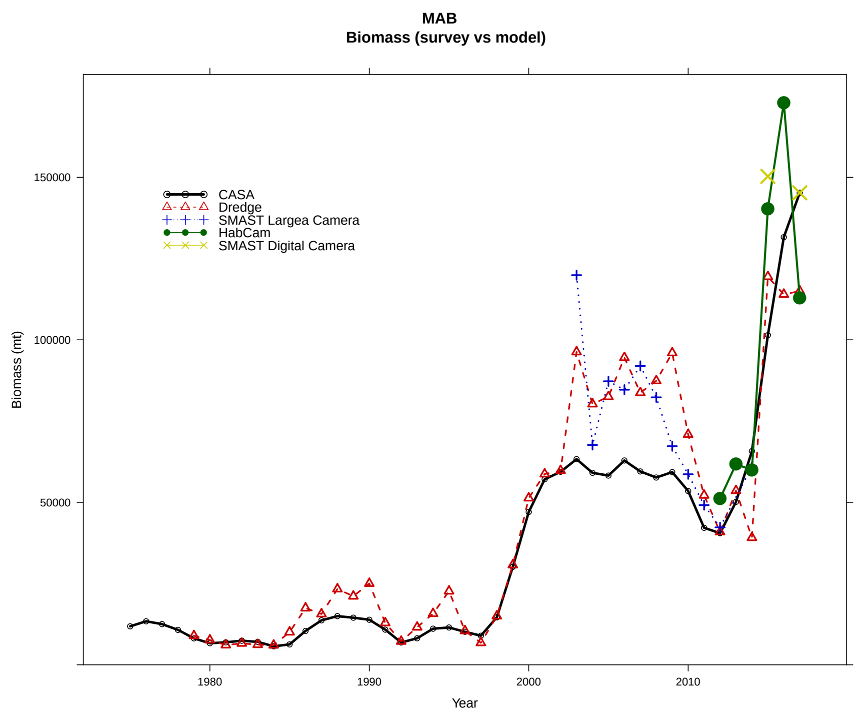


Figure A9.56. Comparison of CASA model estimated abundance (top) and biomass (bottom) with expanded estimates from the lined dredge (red), large drop camera (blue), HabCam (green), and SMAST digital camera (light green) for Mid-Atlantic areas.

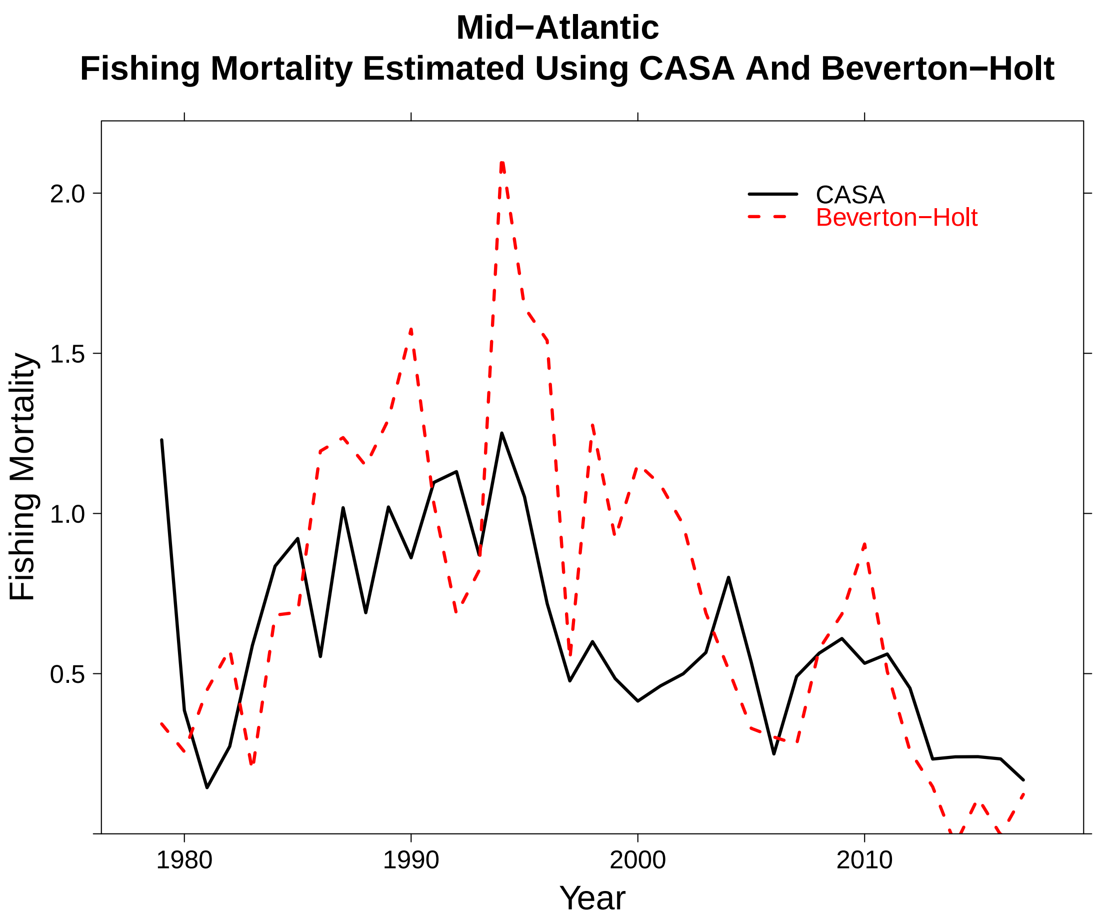


Figure A9.57. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium length based estimator for Mid-Atlantic areas.

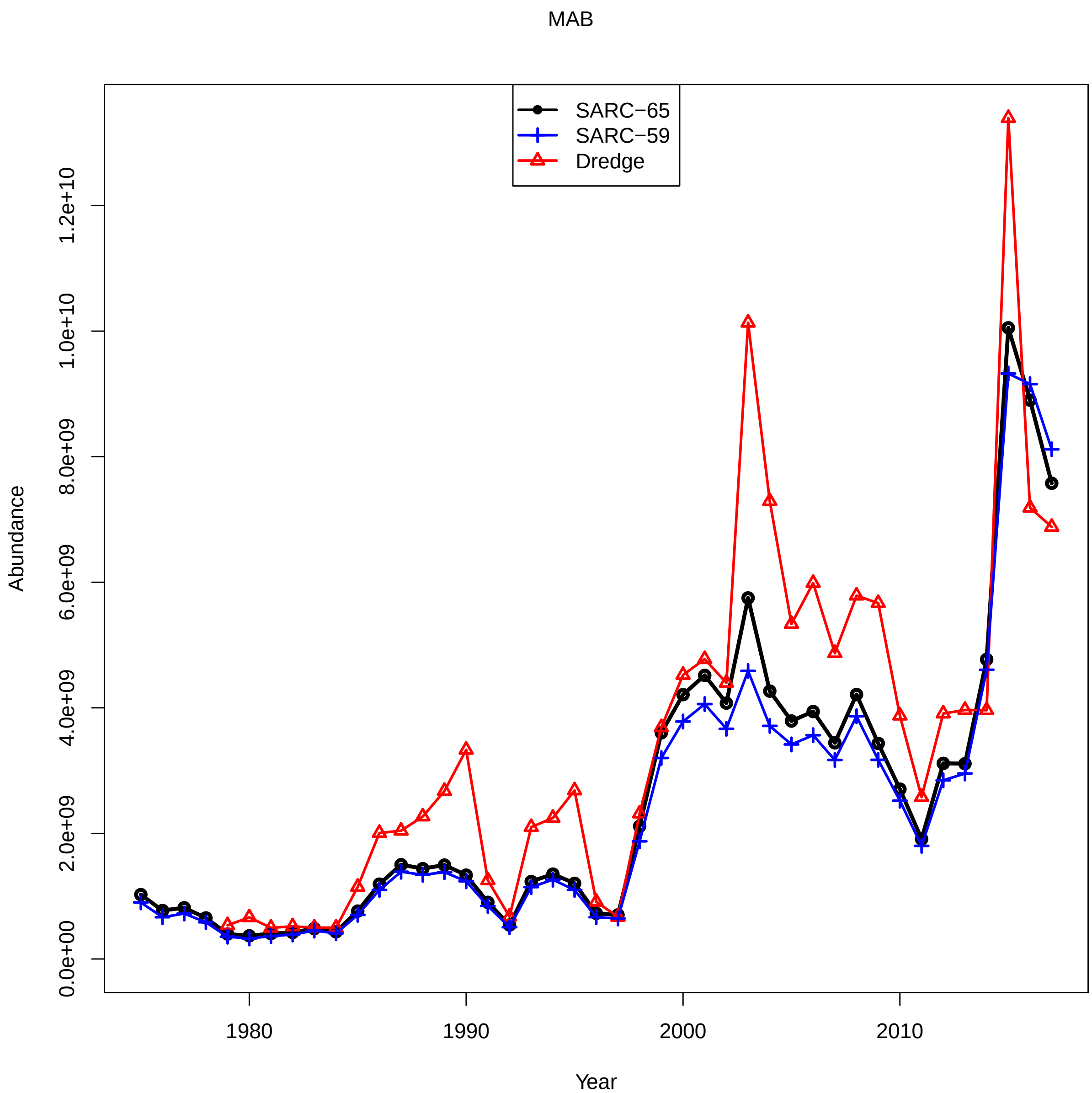


Figure A9.58. Comparison of estimated abundance from current and SARC-59 CASA model configurations, along with lined dredge survey abundance index for Mid-Atlantic areas.

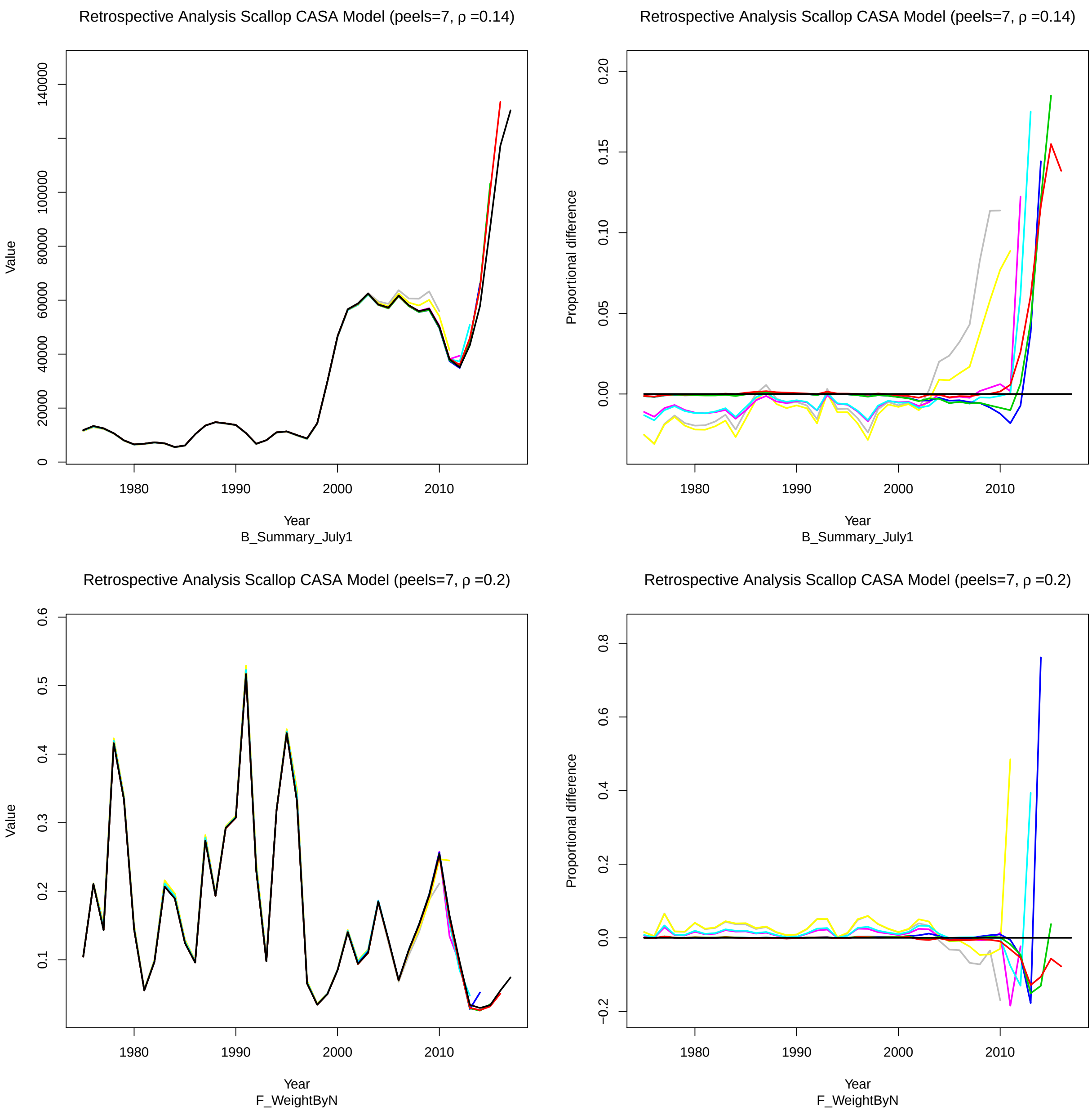


Figure A9.59. Retrospective plots for biomass and fishing mortality for Mid-Atlantic areas. Retrospectives are shown on both absolute and relative scales.

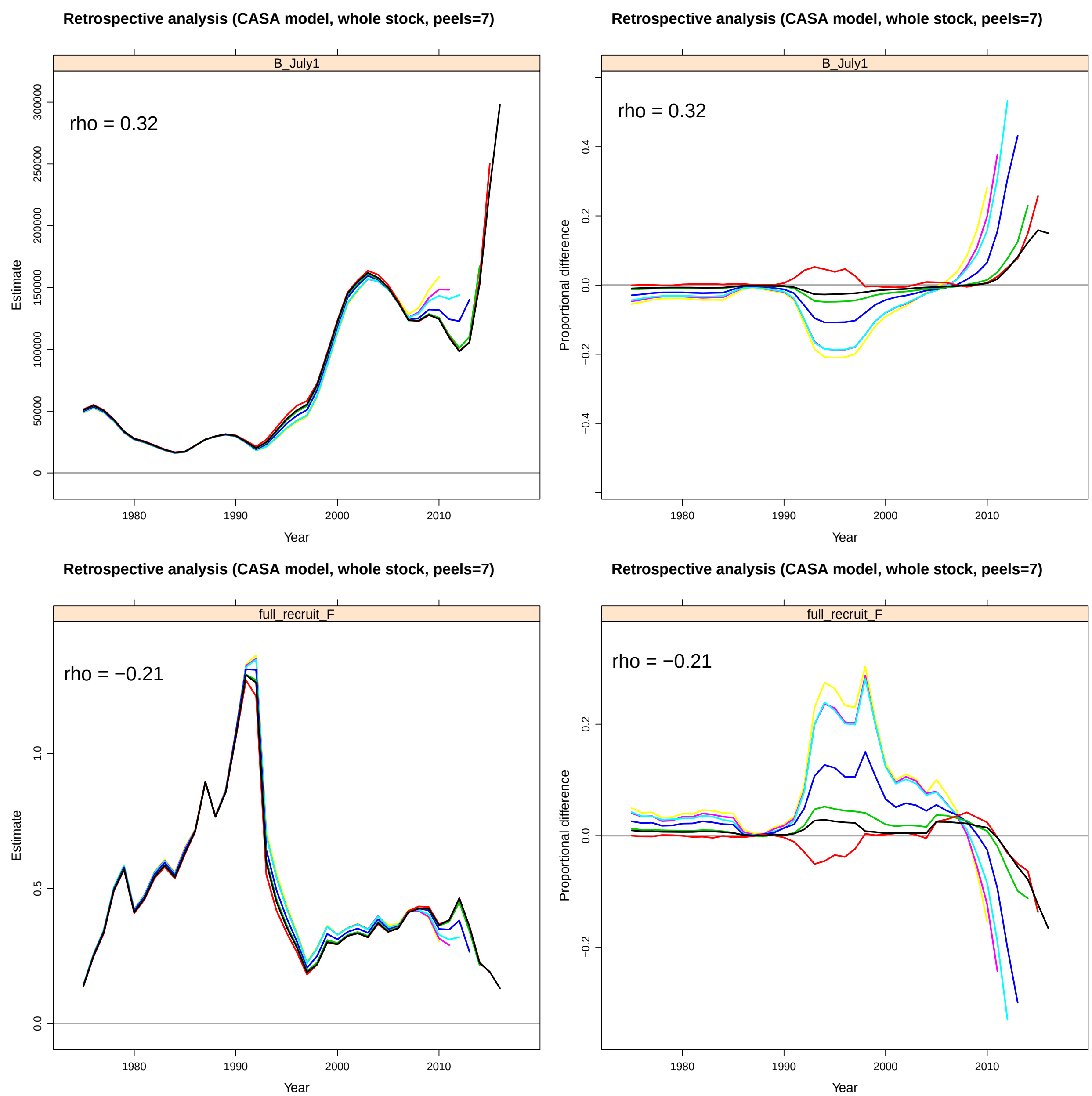


Figure A9.60. Retrospective plots for biomass and fishing mortality for all three stocks combined. Retrospectives are shown on both absolute and relative scales.

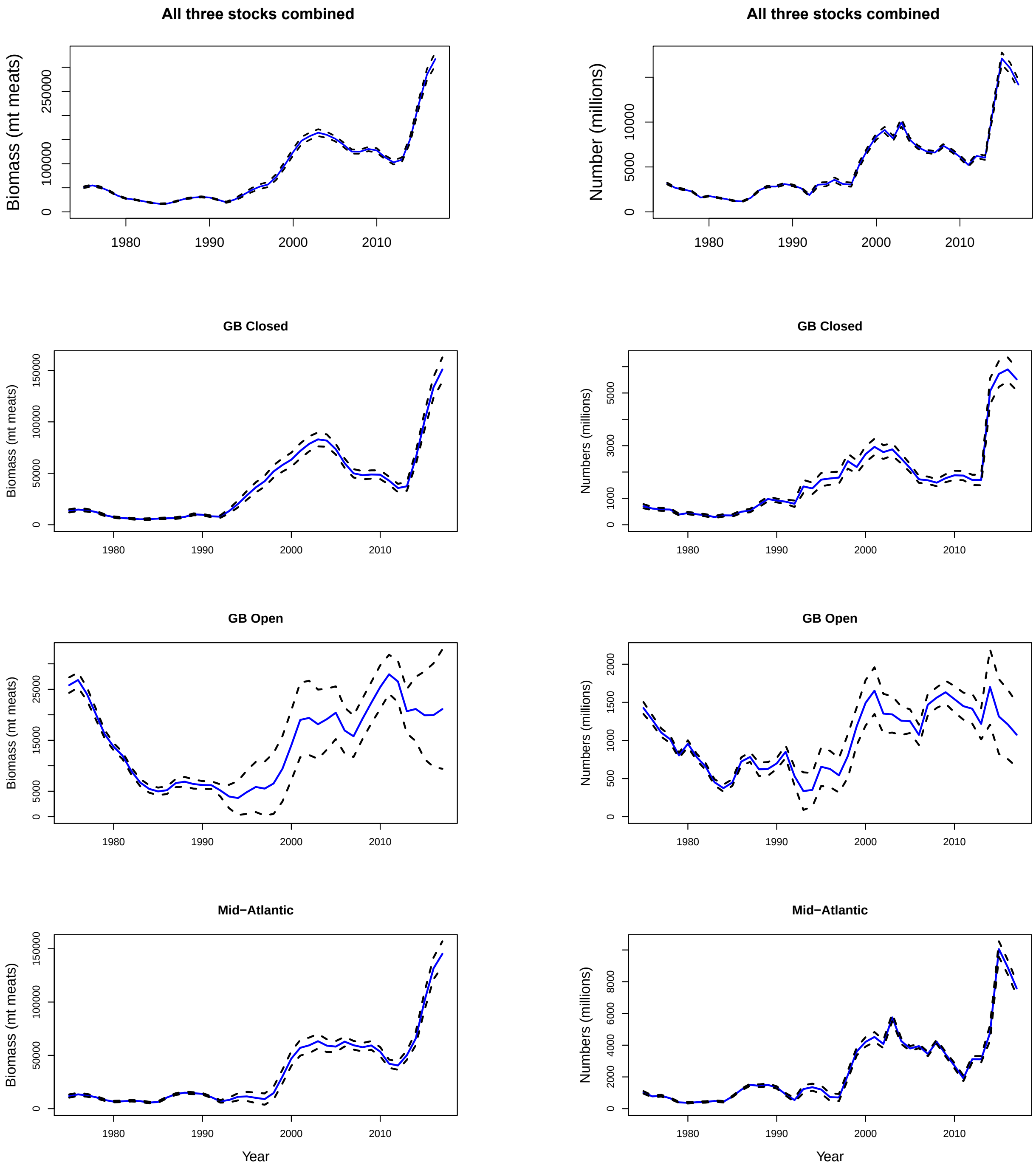


Figure A9.61. CASA estimated biomass and abundance with standard error for Georges Bank closed, Georges Bank open, and Mid-Atlantic stocks and all three stocks combined.

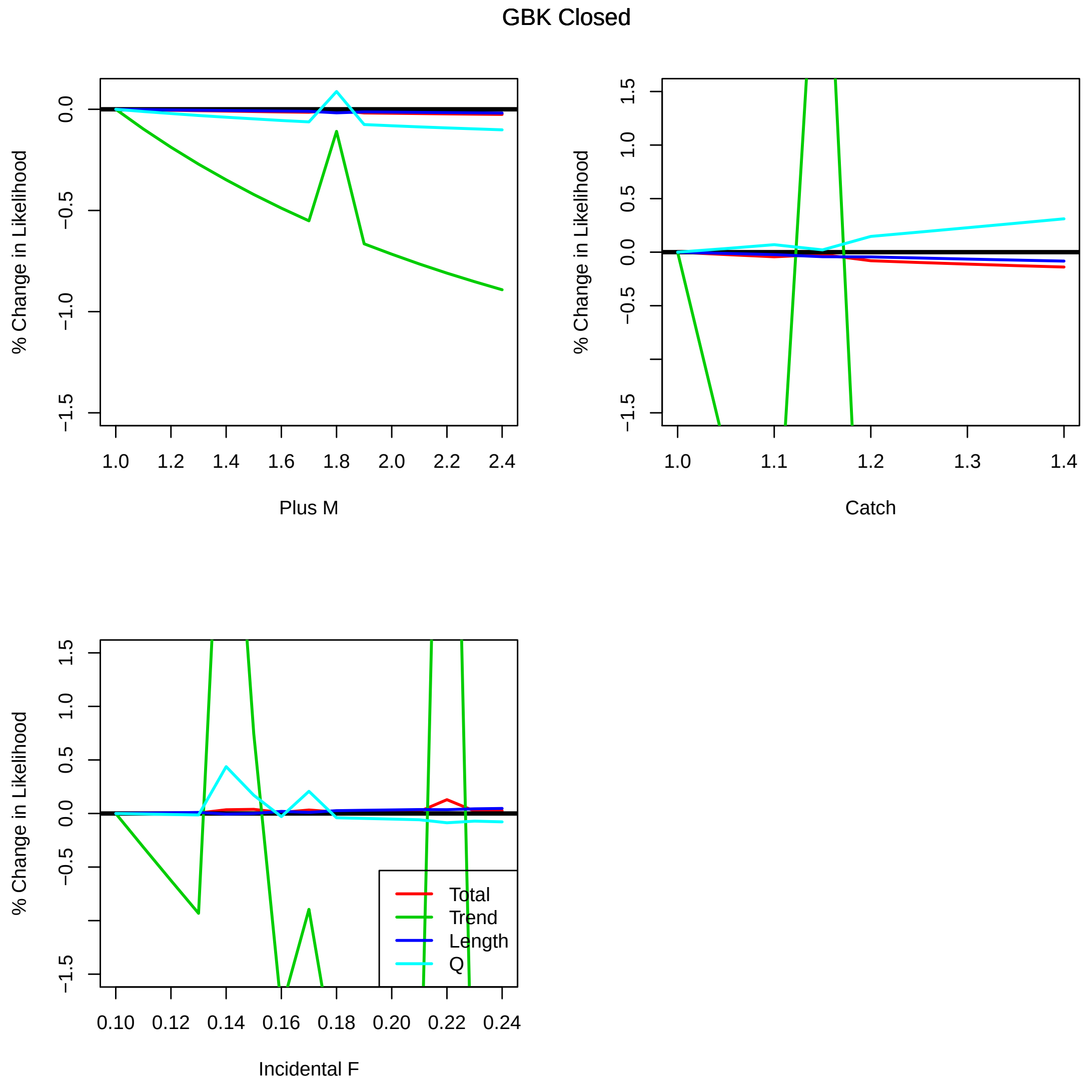


Figure A9.62. Likelihood profiles (red: total negative log likelihood; green: survey trend likelihood; blue: size frequency likelihood; light blue: Q prior likelihood; each of the three likelihood components was summed over all surveys) over plus group natural mortality, catch, and incidental mortality for Georges Bank closed stock. The % changes in likelihood were plotted from -1.5% to 1.5%.

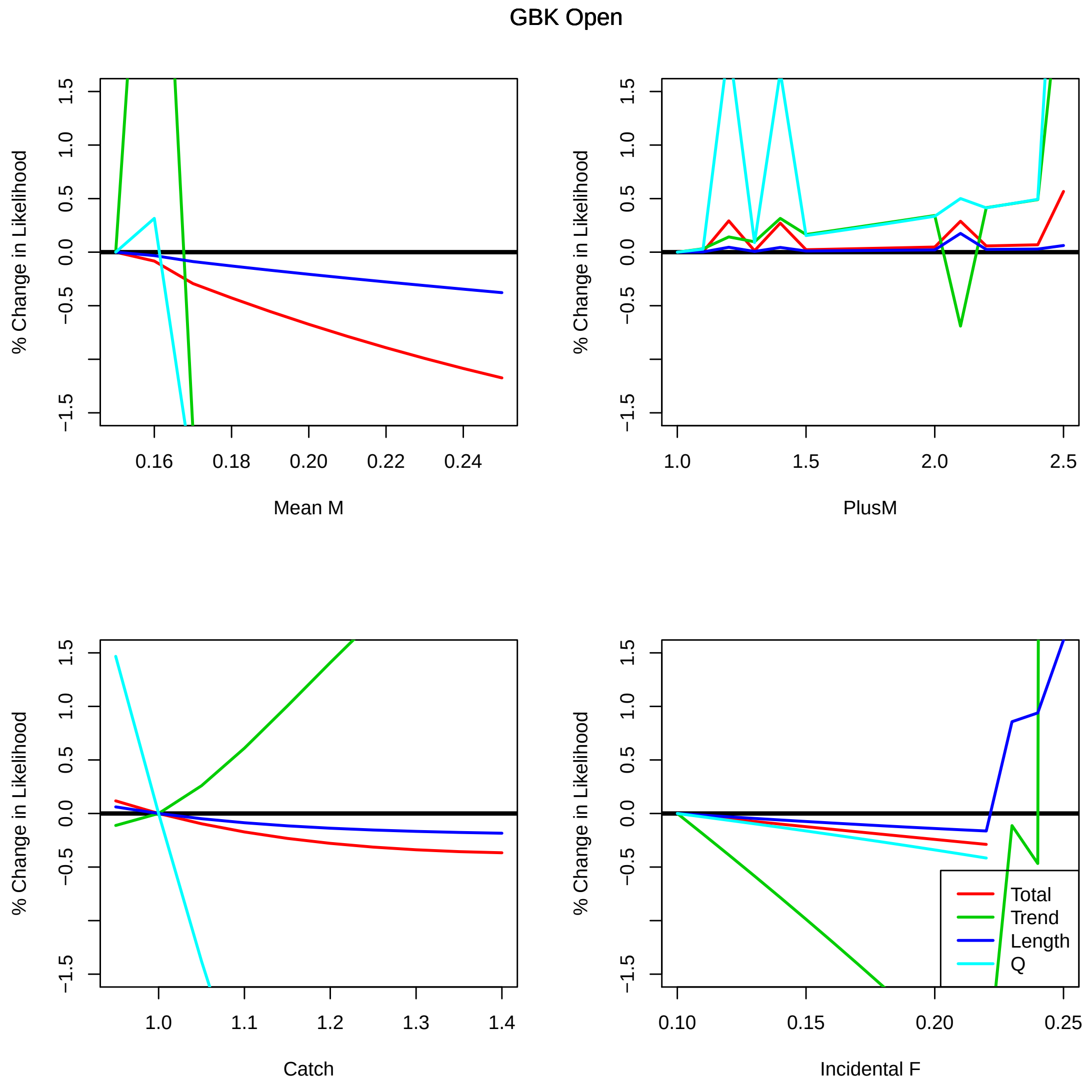


Figure A9.63. Likelihood profiles (red: total negative log likelihood; green: survey trend likelihood; blue: size frequency likelihood; light blue: Q prior likelihood; each of the three likelihood components was summed over all surveys) over mean natural mortality, plus group natural mortality, catch, and incidental mortality for Georges Bank open stock. The % changes in likelihood were plotted from -1.5% to 1.5%.

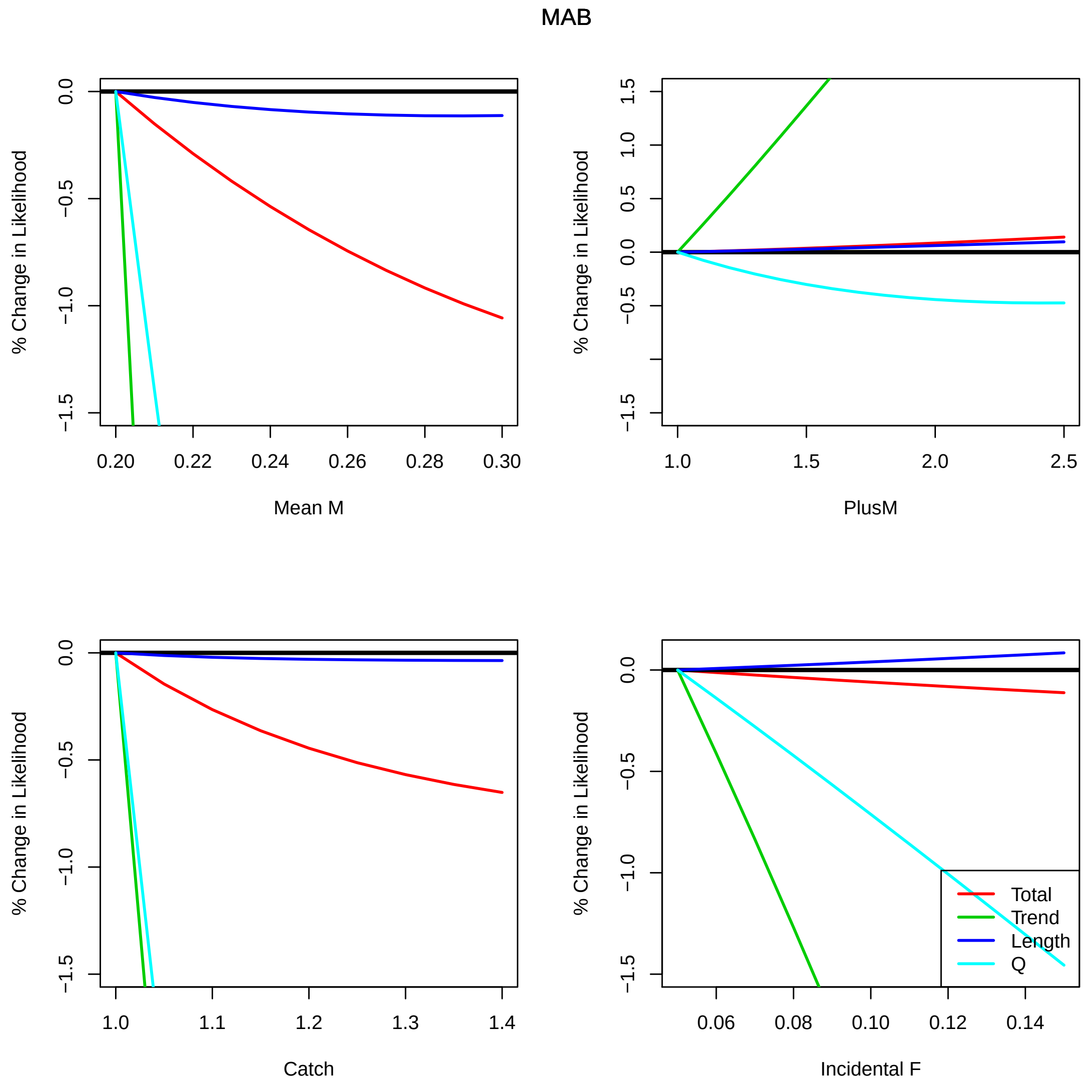


Figure A9.64. Likelihood profiles (red: total negative log likelihood; green: survey trend likelihood; blue: size frequency likelihood; light blue: Q prior likelihood; each of the three likelihood components was summed over all surveys) over mean natural mortality, plus group natural mortality, catch, and incidental mortality for Mid-Atlantic stock. The % changes in likelihood were plotted from -1.5% to 1.5%.

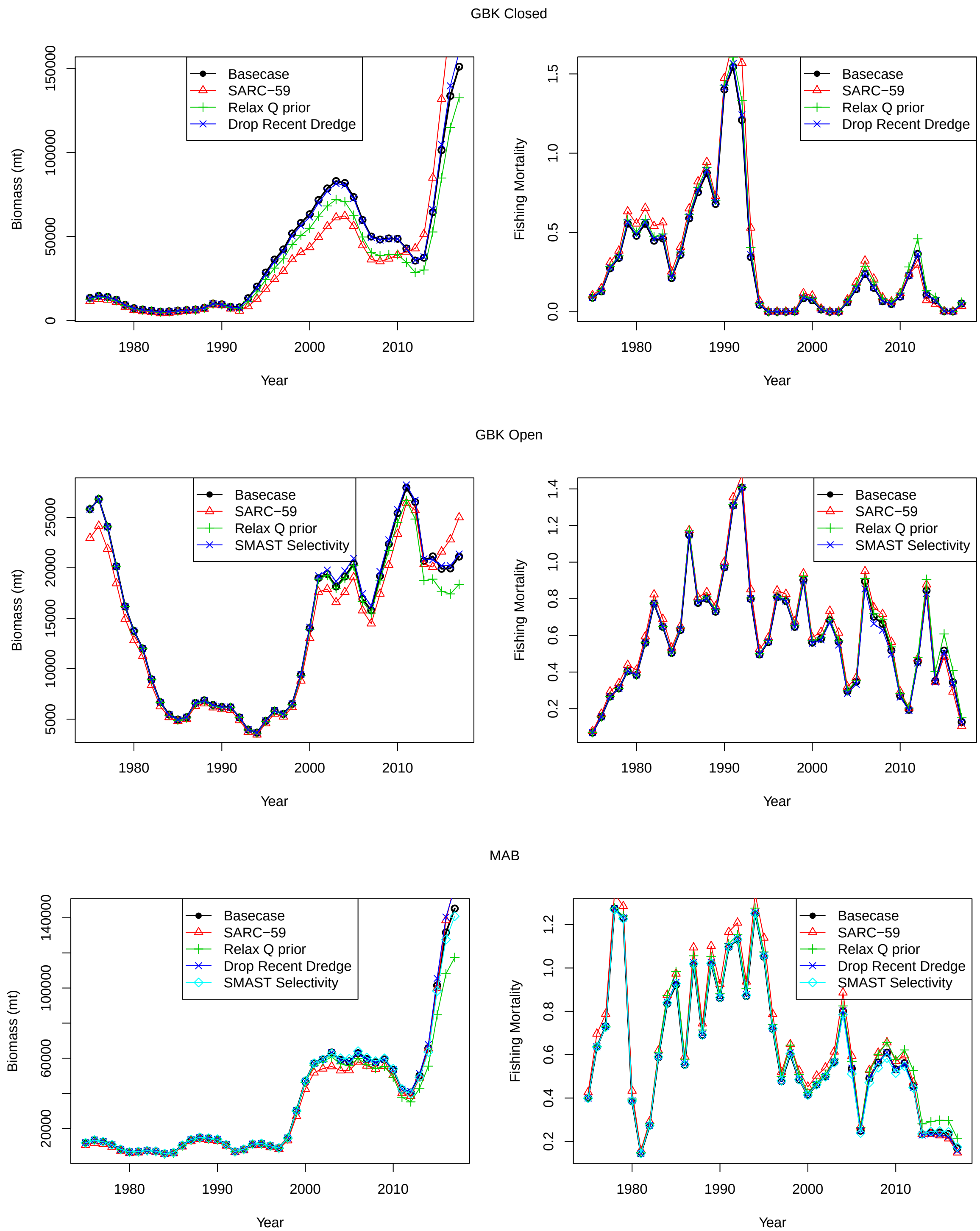


Figure A9.65. Sensitivity of CASA estimated biomass and fishing mortality to assumptions regarding CASA model configuration (SARC-59), Q prior, and large drop camera survey selectivity for Georges Bank closed areas.

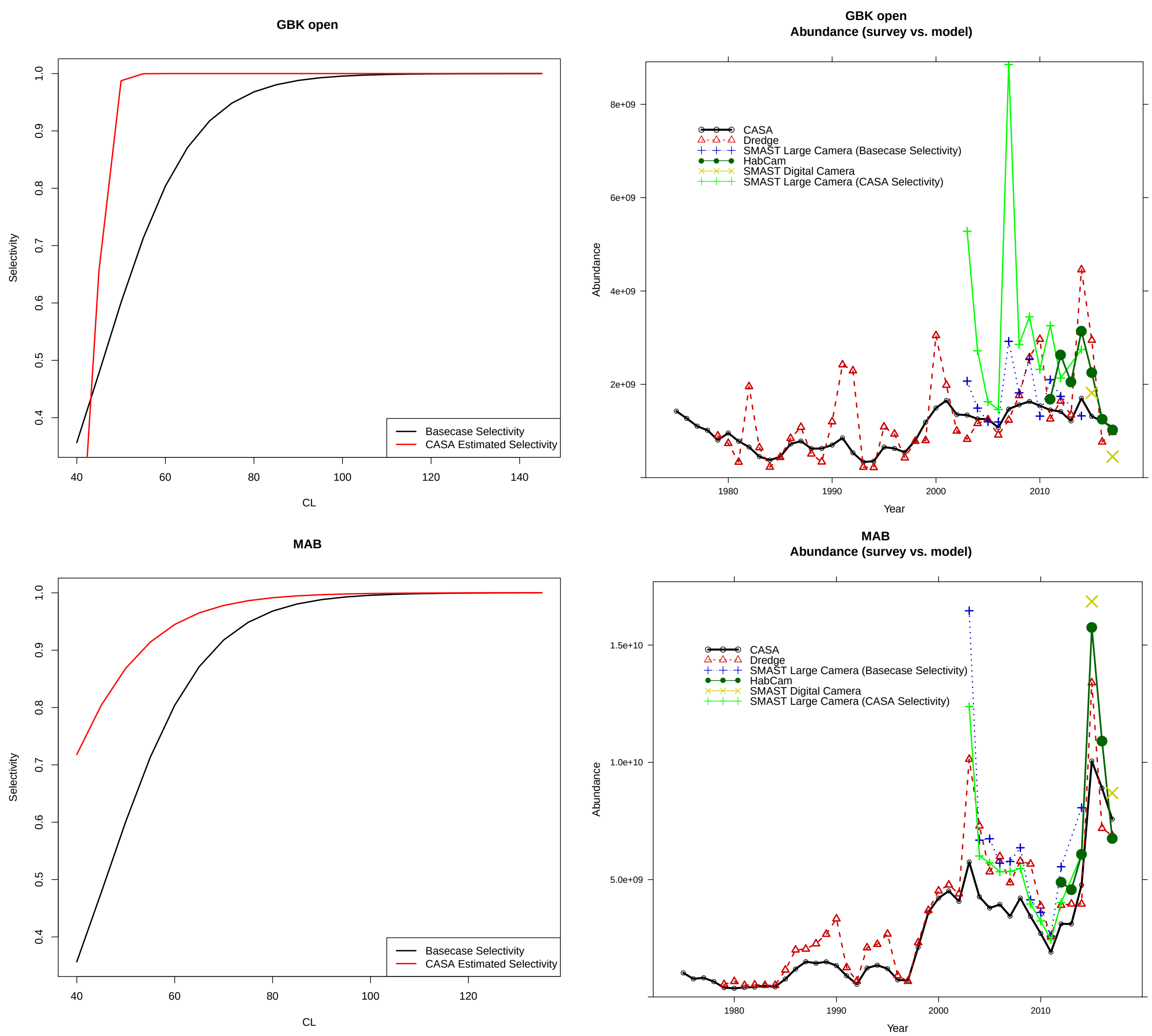


Figure A9.66. CASA model estimated selectivity for large drop camera survey and abundance indices converted using the model estimated selectivity for Georges Bank open and Mid-Atlanctic areas.

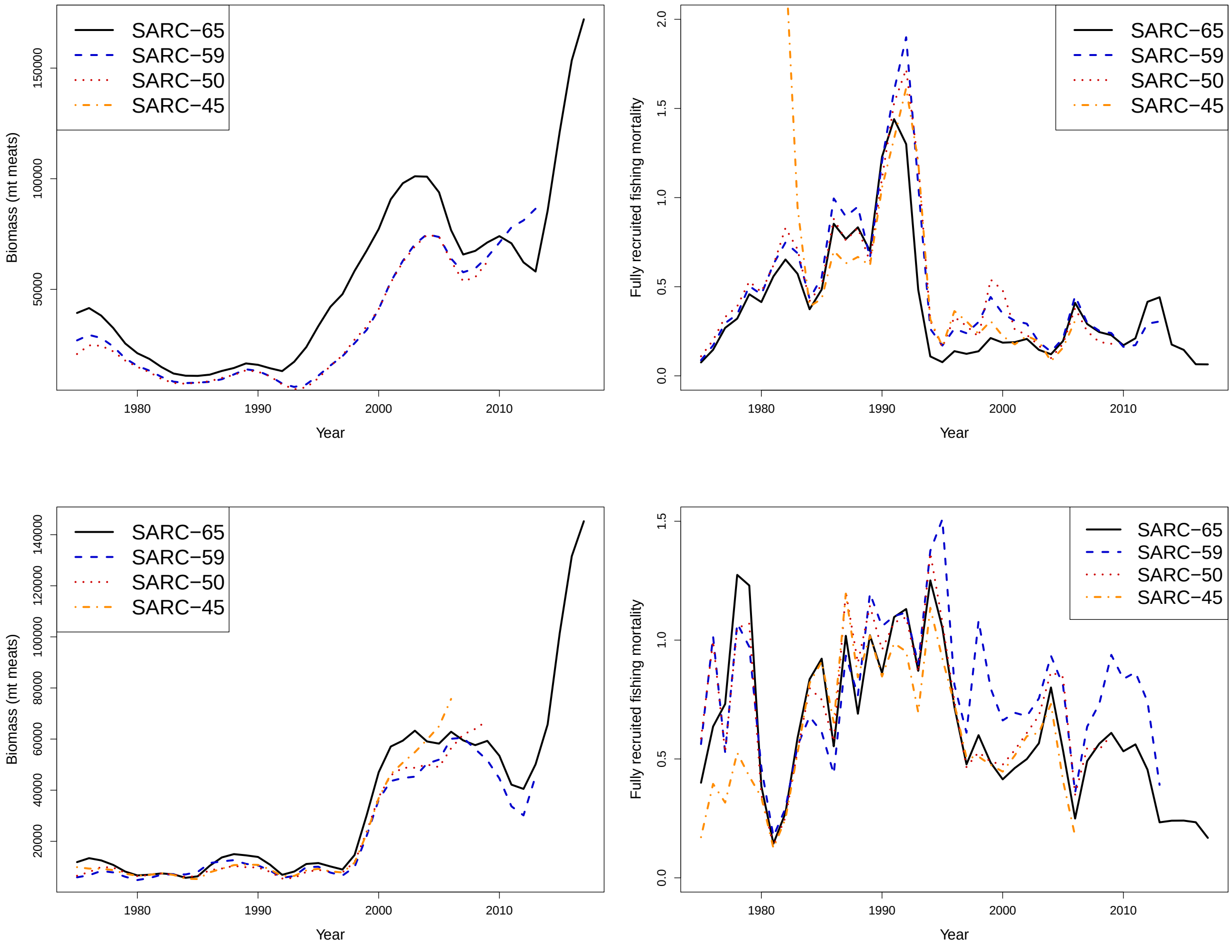


Figure A9.67. Comparison of current CASA model estimates of biomass (left) and fishing mortality (right) to previous CASA model estimates for Georges Bank (top) and Mid-Atlantic (bottom) sea scallops.

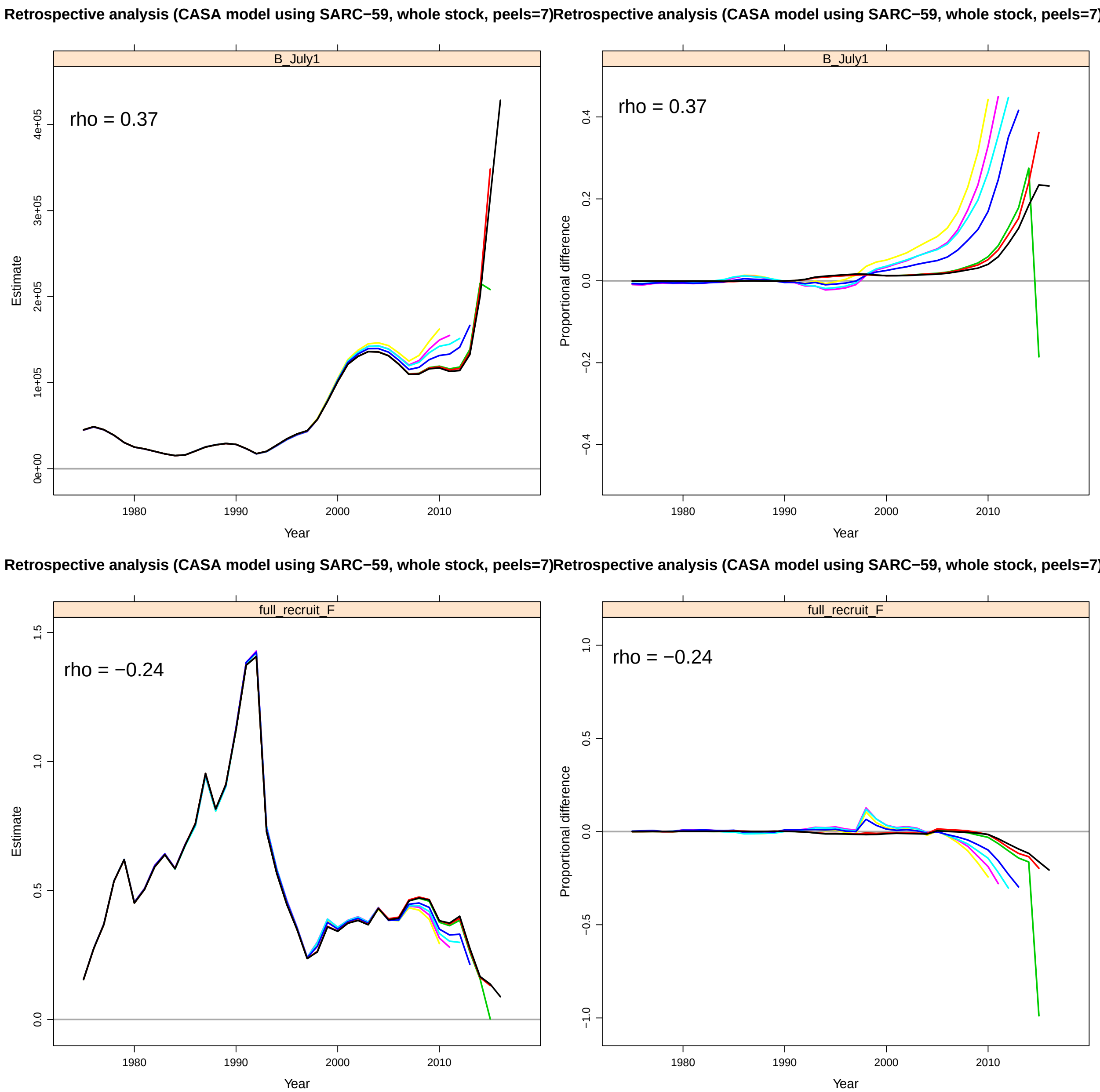


Figure A9.68. Retrospective plots for biomass and fishing mortality estimated using SARC-59 CASA model configuration for all three stocks combined. Retrospectives are shown on both absolute and relative scales.

10 Reference points (TOR 6)

Introduction

Per recruit reference points were used as proxies for F_{MSY} and B_{MSY} in scallop assessments prior to 2010. The per recruit reference point F_{MAX}, the fully recruited fishing mortality rate that generates maximum yield-per-recruit, was used as a proxy for F_{MSY}. The biomass reference point was defined as the product of B_{MAX} (biomass per recruit at $F = F_{\mathrm{MAX}}$) 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, NEFSC (2010) 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, F_{MSY} and B_{MSY}.

NEFSC (2014) estimated $F_{\mathrm{MSY}} = 0.48$ and $B_{\mathrm{MSY}} = 96,480$ mt meats. According to the sea scallop fishery management plan, overfishing is occuring if the whole stock fully recruited fishing mortality exceeds F_{MSY}, and the stock is overfished if biomass is less than half of B_{MSY}.

Methods

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. In particular, growth was modelled using stochastic growth matrices based on the most recent period. However, because the SYM model (unlike CASA) uses a stock-recruit relationship, the Georges Bank open and closed areas were combined, so that two SYM models were used, one for Mid-Atlantic and one for Georges Bank.

Uncertainties in the SYM model can be divided into uncertainty in the per recruit models, and that from the stock recruit relationship. Per recruit calculations depend on a number of parameters that each carry a level of uncertainty, including shell height/meat weight (and gonad weight) parameters, fishery selectivity, cull size and the fraction of discards that survive, the incidental mortality rate, and the natural mortality rate M. Each of these was modelled by specifying a distribution, together with parameters for that distribution, typically a mean and variance. A more detailed discussion of all these parameters can be found in Hart (2013). The form of the distributions used are the same as in Hart (2013), but point estimates of some of the parameters, such as selectivity and shell height/meat weight, have been updated as discussed previously.

Of all the input parameters to the per recruit modeling, by far the most uncertain is natural mortality. Natural mortality was modelled as an inverse gamma distribution (i.e., $1/\gamma$, where γ is a gamma distribution). This makes sense for several reasons. First, in many methods to estimate natural mortality, such as maximum longevity or “clapper” ratios for scallops, the uncertain quantity is in the denominator. Secondly, inverse gamma distributions are skewed to the right, which makes sense for natural mortality. For example, it is possible that the true natural mortality is twice or more than its point estimate, but it cannot be zero or less. For this assessment, the mean M was taken as 0.2 on Georges Bank and 0.25 in the Mid-Atlantic, with standard deviation $\sigma = 0.082$ in both cases (Figure A10.1); this is the same value of σ that was used in previous assessments.

Beverton-Holt stock-recruit curves were fitted to spawning stock biomass B (using meat weight; gonad weight fits are shown in Appendix 10) and recruitment estimates from base case CASA model runs

$$R = \frac{sB}{h + B} \tag{A10.1}$$

assuming square-root-normal errors, where s is the expected asymptotic recruitment, and h is the spawning stock biomass where the expected recruitment is half its asymptotic value. Standard errors of the stock-recruit parameters and their correlation were estimated using the delta method.

For this assessment, CASA now estimates one year old recruits (in previous assessments, it estimated two year olds). Estimated juvenile natural mortality varies by year, and is apparently correlated with density. Because time varying or density-dependent natural mortality cannot be modelled in a per recruit framework, we calculated recruits at three years old by applying two years of natural and incidental mortality to one year old recruits, as estimated by CASA. For this reason, the SYM model now starts its per recruit calculations at 80 mm, approximately three years old, and the stock-recruit curve (Equation A10.1, Figure A10.2) was estimated in terms of three year old recruits. Meat weight was used as the primary surrogate for spawning stock biomass (SSB); the use of gonad weight instead of meat weight is explored in Appendix 10. Although in principle gonad weight is a better indicator of egg production than meat weight, a number of technical issues need to be resolved. For example, shell height to gonad weights are only available during the times of the dredge survey, in late spring or early summer. During these times, Mid-Atlantic scallops are typically recovering from their spring spawn, and hence their gonads weigh less that those in Georges Bank during those times. Thus, Georges Bank appears to have a much higher SSB as measured by gonad weight than the Mid-Atlantic.

At each iteration of the simulation model, parameter values were drawn from their corresponding distributions, and per recruit and yield curves were calculated. This was repeated $100,000$ times and the results of each iteration were stored. The stock-recruit parameters were simulated as correlated square-root normals.

For each run, equilibrium recruitment at fishing mortality F is given by:

$$R = s - h/b(F) \tag{A10.2}$$

where $b(F)$ is SSB per recruit at fishing mortality F, and s and h are as in equation (A10.1). Total yield $Y(F)$ is therefore:

$$Y(F) = y(F)R = y(F)[s - h/b(F)] \tag{A10.3}$$

where $y(F)$ is yield per recruit at fishing mortality F.

Mean yield curves calculated by this method can be disproportionately influenced by outliers, both in cases where the population collapses at zero fishing and where predicted yields are unrealistically high (Hart 2013). For this reason, we used the median of the 100,000 runs to obtain point estimates of yield at each fishing mortality. The probabilistic F_{MSY} was taken as the fishing mortality that maximizes the median yield curve; MSY and B_{MSY} are the median yield and SSB at F_{MSY}. In the previous assessment, the 10% trimmed mean, rather than the median curves, were used. However, the large year classes observed since the last assessment have caused more variability in the stock recruit relationship, which propagates to the SYM model. For this reason, the Working Group felt that the median was a more appropriate measure of central tendency.

Results

Both regions show evidence for higher recruitment at higher SSB, but this pattern is more clear in the Mid-Atlantic (Figure A10.2). Estimated stock recruit curves were more stable on Georges Bank, where it appears that most of the uncertainty is in the asymptote s. In the Mid-Atlantic, the stock-recruit relationship in some iterations continues to increase well beyond the observed SSB. Although Y_{MAX} and B_{MAX} values were generally well defined, F_{MAX} was highly uncertain in both regions, and hit the $F = 2$ bound in a majority of the simulations in the Mid-Atlantic and in about 15% of the cases on Georges Bank (Figures A10.3, A10.4). MSY-based reference points are somewhat better defined, as the stock-recruit relationships tend to constrain F_{MSY} (Figures A10.5, A10.6).

Estimates for the combined (Georges Bank and Mid-Atlantic) MSY range from low values to over 150,000 mt meats (Figure A10.7, A10.8), and B_{MSY} also ranged to over 150,000 mt meats. F_{MSY} values for the combined stock are highly uncertain. Median mean yield curves have a maximum at $F = 0.57$ on Georges Bank, and $F = 0.73$ in the Mid-Atlantic, with corresponding MSY values of 19,105 and 27,696 mt meats, respectively (Table A10.1). One complication with using median curves is that the sum of the median curves for Mid-Atlantic and Georges Bank is not equal to the median curve of the sum of the Georges Bank and Mid-Atlantic SYM runs (Figure A10.9). In Table A10.1, Med. $\mathrm{Yield}_{0.64}$ and Med. $\mathrm{SSB}_{0.64}$ for the combined stock represent the value of the combined median yield curve at $F_{\mathrm{MSY}} = 0.64$, which is different than the sum of the corresponding values for each region. The Working Group chose to use the sum of the regional yields and SSBs at $F_{\mathrm{MSY}} = 0.64$ as the value of the combined MSY and B_{MSY}.

The probabilistic approach to reference points employed here can be used to examine the tradeoff between the risk of overfishing and the expected loss of yield (Hart 2013). At a fixed fishing mortality F, the probability that this F exceeds the true but unknown F_{MSY} is simply one minus the cdf of the distribution of the F_{MSY}s (Figure A10.10, red dashed line). On the other hand, fishing at an F other than the estimated F_{MSY} will result in a loss of expected median yield (Figure A10.10, blue solid line). According to Amendment 15 to the Sea Scallop Fishery Management Plan (NEFMC 2011), the Allowable Biological Catch and Annual Catch Limit (ABC and ACL) is estimated using the fishing mortality corresponding to the 25th percentile of the distribution of the combined F_{MSY}, which in this case is $F_{\mathrm{ACL}} = 0.51$. Fishing at this rate considerably reduces the risk of overfishing (i.e., fishing greater than the true F_{MSY}), with little cost in terms of loss of median expected yield (Figure A10.10). ACL/ABCs are computed by the SAMS model, with $F = F_{\mathrm{ACL}}$ for all areas, projected forward one or two years. The ABCs/ACLs for 2018/2019 will be estimated in the autumn of 2018, using the results from the 2018 surveys.

Discussion

Although the recommended F_{MSY} appears to be high, fishery selectivity has shifted substantially to the right in the last twenty years, so that most scallops will never experience fully recruited fishing mortality. For example, a 100 mm scallop would experience only about half the fully recruited F in the Mid-Atlantic or Georges Bank open, and even less in Georges Bank closed.

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, Truesdell et al. 2015, 2017). 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, Truesdell et al. 2015). 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).

Table A10.1. Summary of reference points for Georges Bank, Mid-Atlantic and combined. Med. $\text{Yield}_{0.64}$ and Med. $\text{SSB}_{0.64}$ are the yield and SSB from the median curves at the combined $F_{\text{MSY}} = 0.64$.

Region	MSY	F_{MSY}	B_{MSY}	Med. $\text{Yield}_{0.64}$	Med. $\text{Bms}_{0.64}$
GB	19,105	0.57	49,667	18,979	44,931
MA	27,696	0.73	65,286	27,552	71,835
Combined	46,531	0.64	116,766	49,433	116,766

Figures

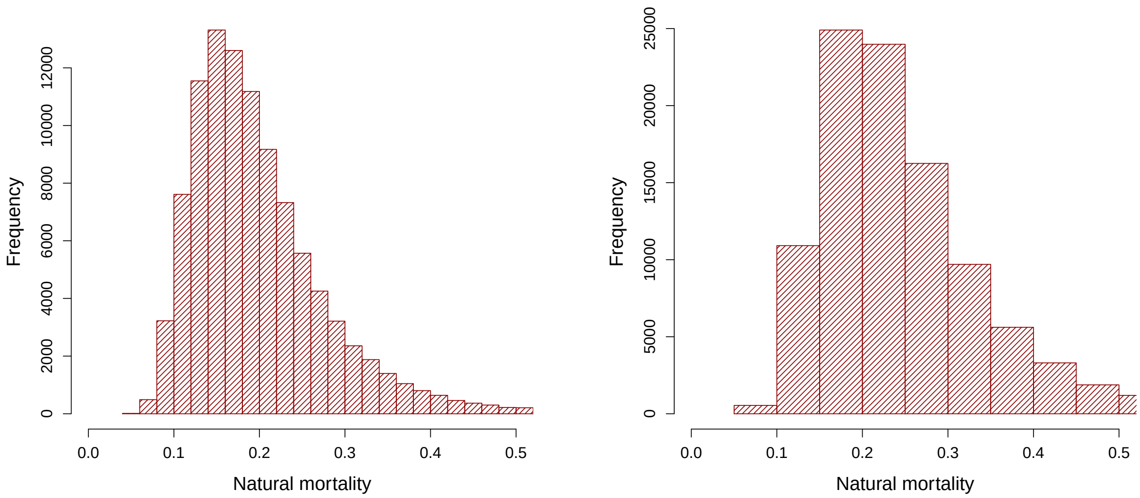


Figure A10.1. Distribution of the natural mortality parameters used in the SYM model for Georges Bank (left) and Mid-Atlantic (right)

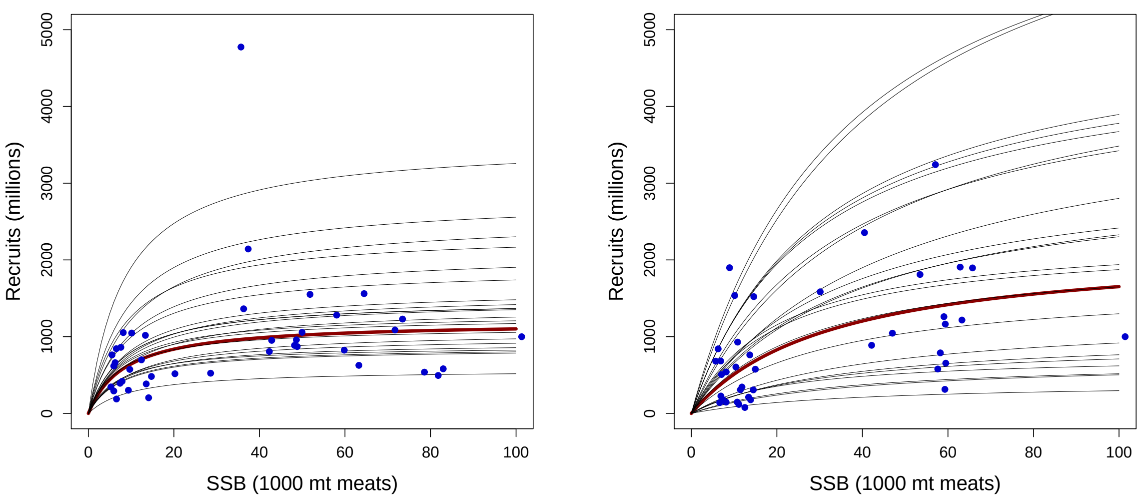


Figure A10.2. Stock-recruit plots for Georges Bank (left), and Mid-Atlantic (right) (blue dots), with best fits to the data (think red line), and 25 example fits (thin black lines) from the SYM model.

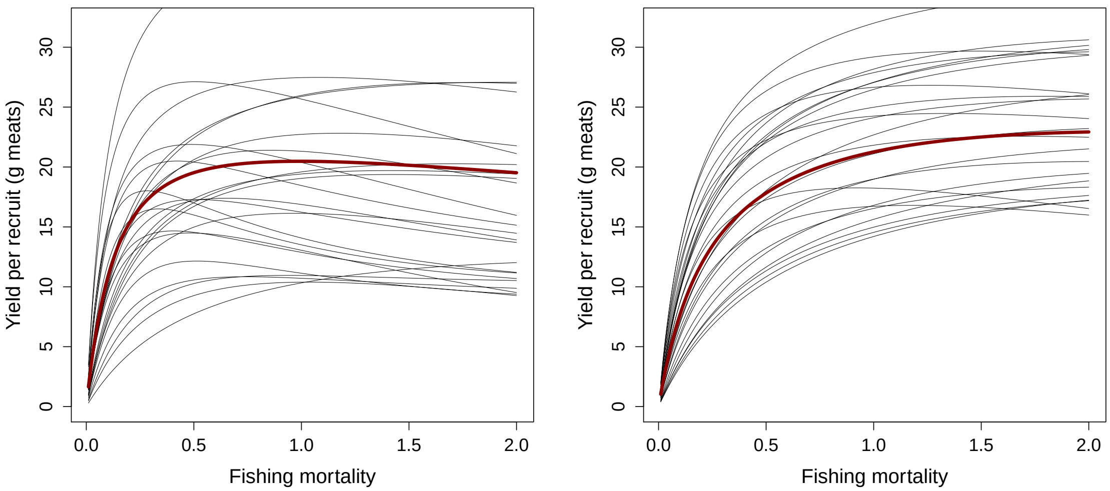


Figure A10.3. Mean yield per recruit plot (dark red line) together with 25 example yield per recruit plots (thin black lines) from the SYM model for Georges Bank (left) and the Mid-Atlantic (right).

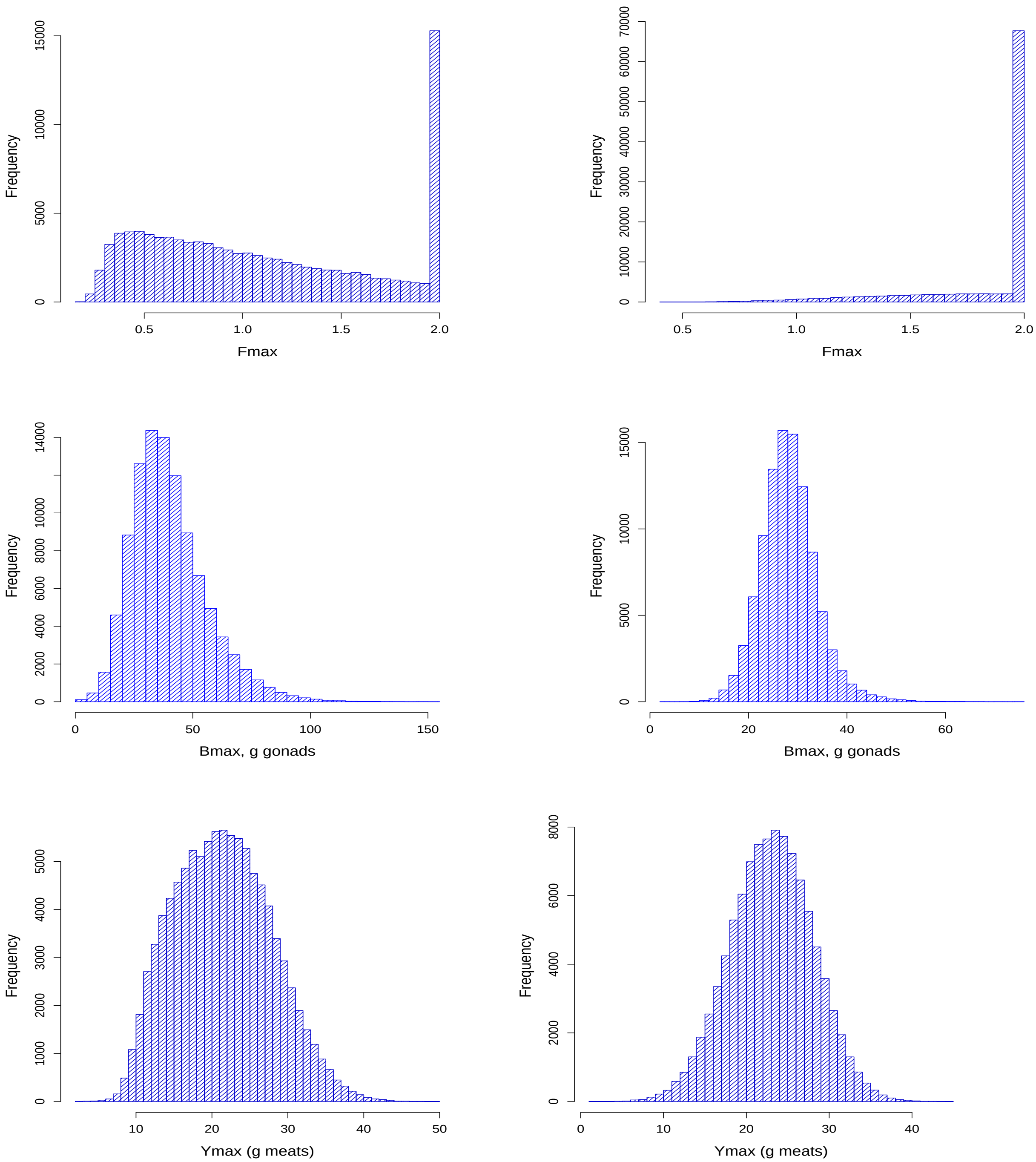


Figure A10.4. Distribution of the yield per recruit reference point F_{MAX}, B_{MAX} and Y_{MAX} for Georges Bank (left), and the Mid-Atlantic (right).

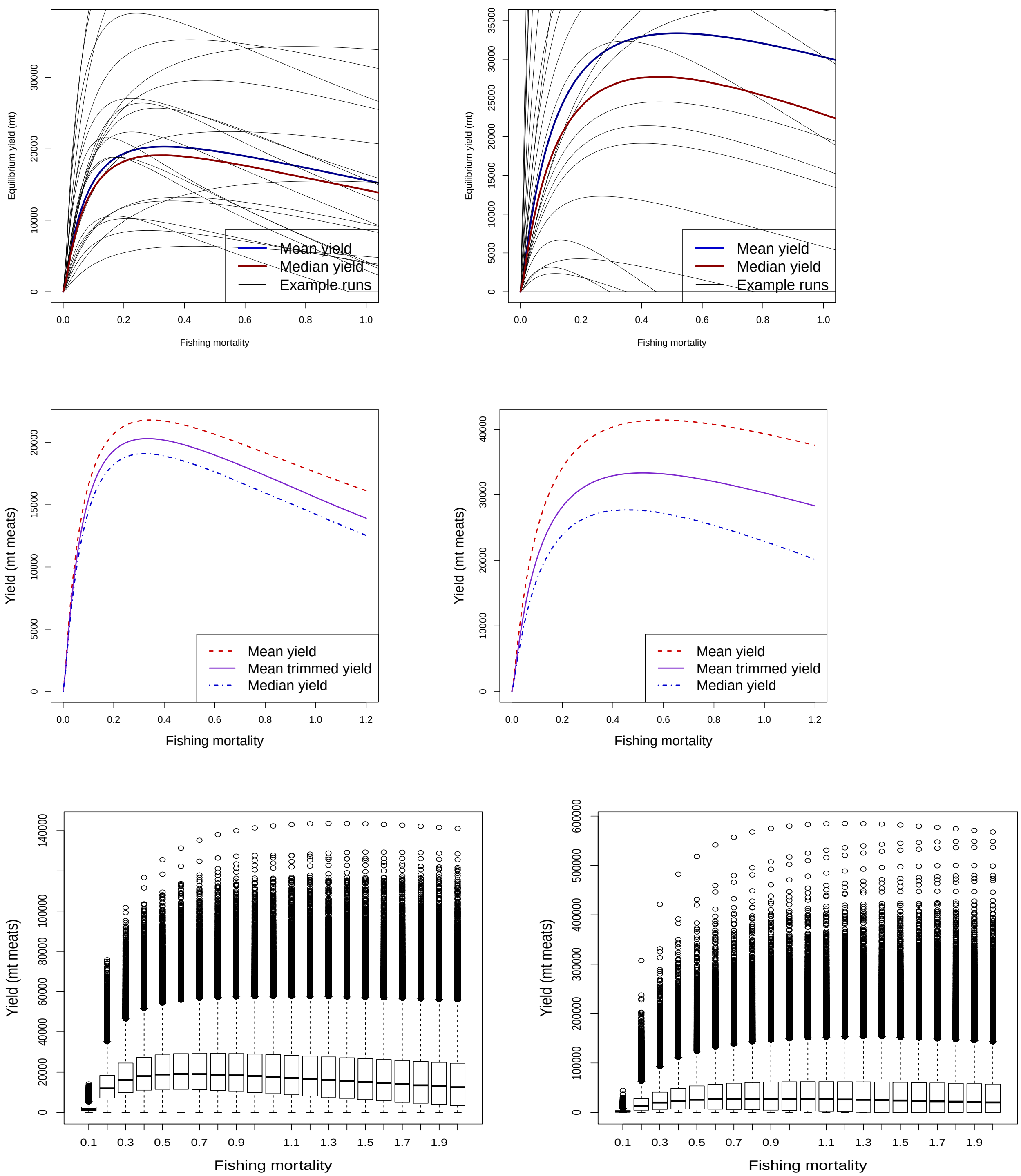


Figure A10.5. Estimated mean and median yield curves with example runs (top), mean, trimmed mean, and median yield curves (middle), and yield box plots (below) for Georges Bank (left) and Mid-Atlantic (right).

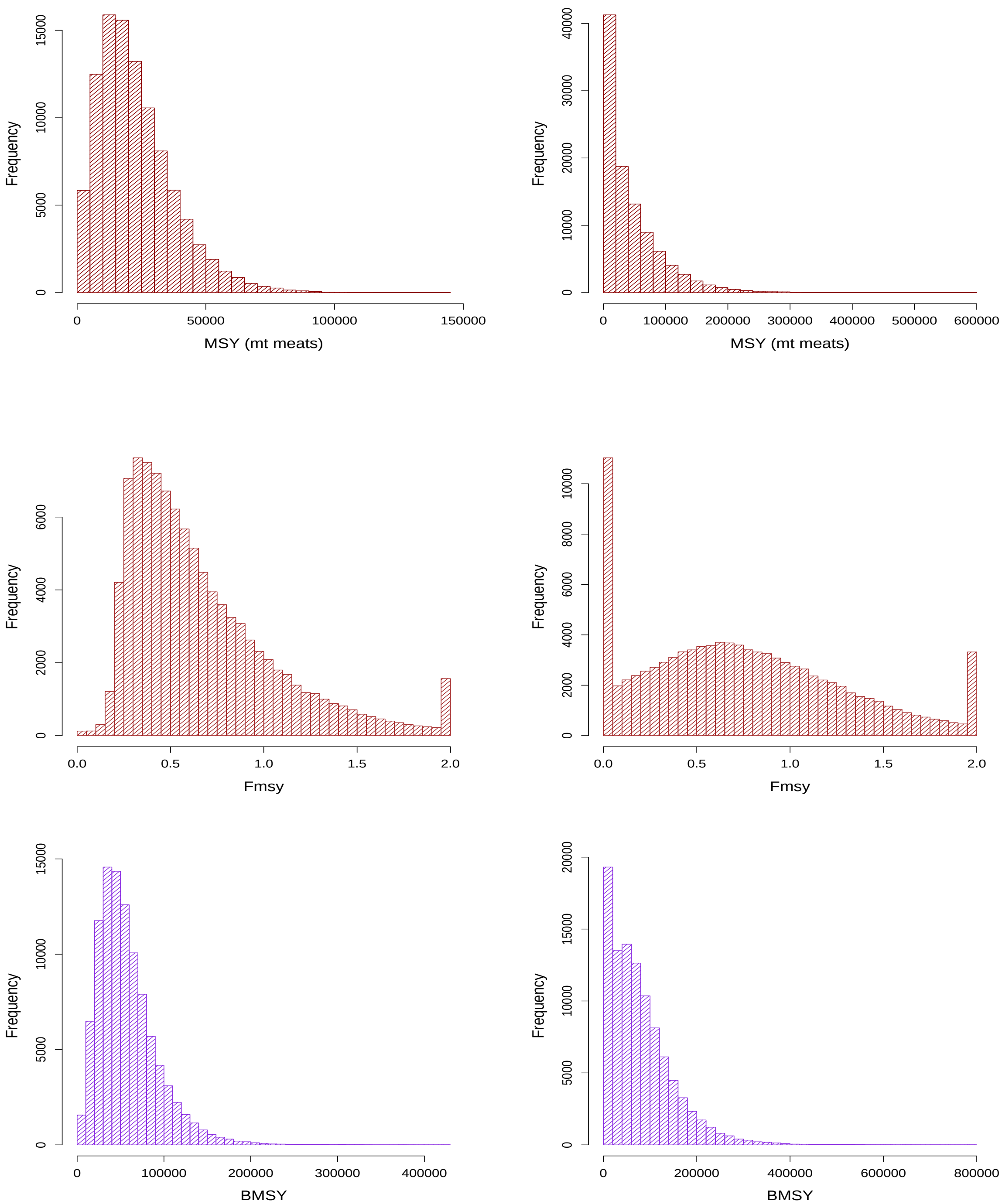


Figure A10.6. Distributions of MSY, F_{MSY} and B_{MSY} for Georges Bank (left) and Mid-Atlantic (right).

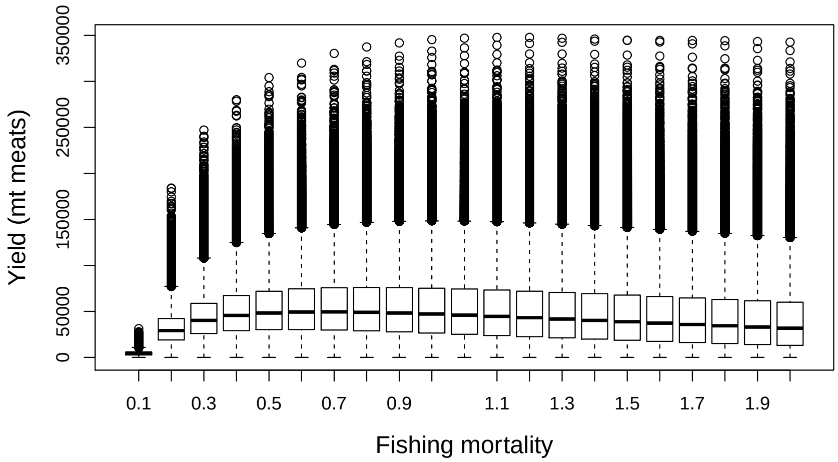


Figure A10.7. Boxplots of SYM yields for Georges Bank and Mid-Atlantic combined.

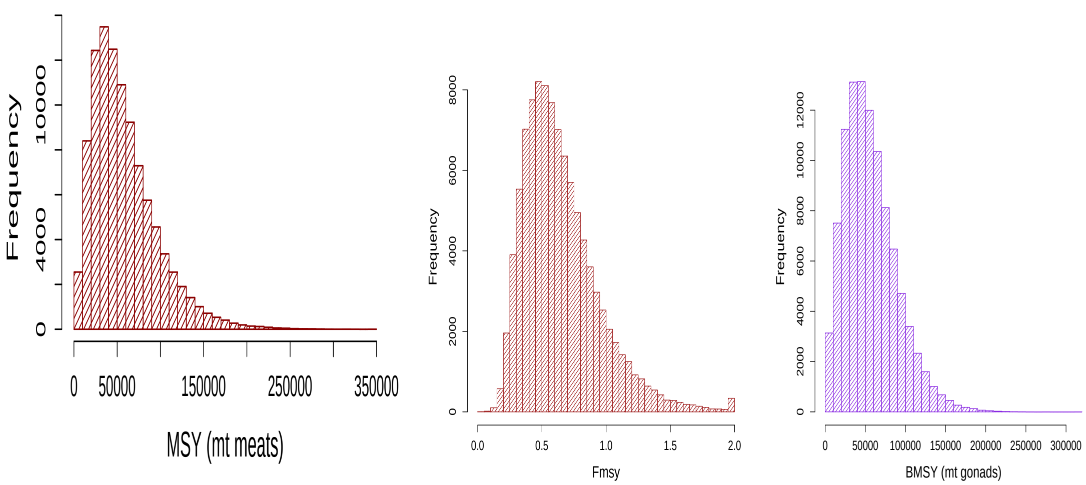


Figure A10.8. Distributions of MSY, F_{MSY} and B_{MSY} for Mid-Atlantic and Georges Bank combined.

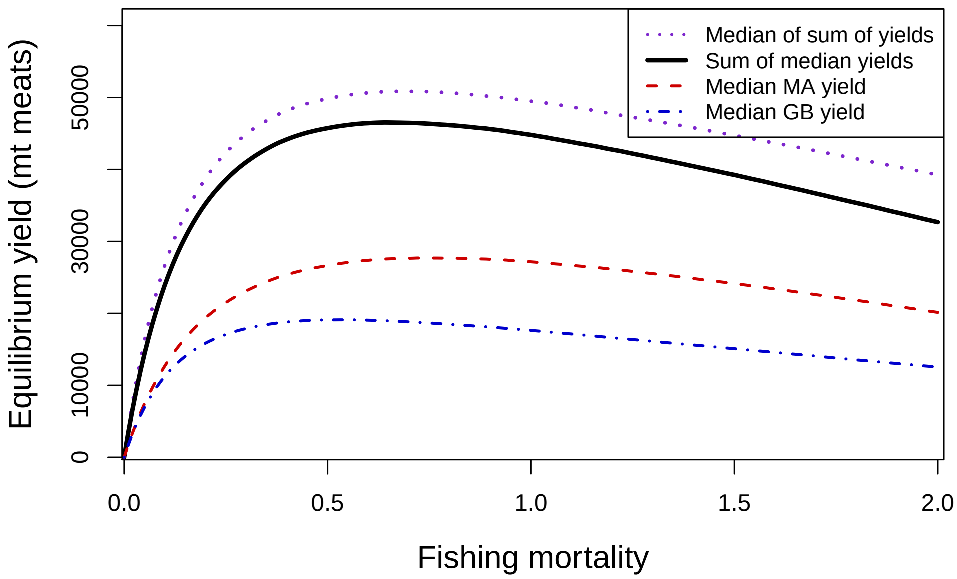


Figure A10.9. Median yield curves for Georges Bank (dashed-dotted line) and the Mid-Atlantic (dashed line) together with the sum of the regional median yields (solid line) and the median of the sum of the Georges Bank and Mid-Atlantic SYM runs (dotted line).

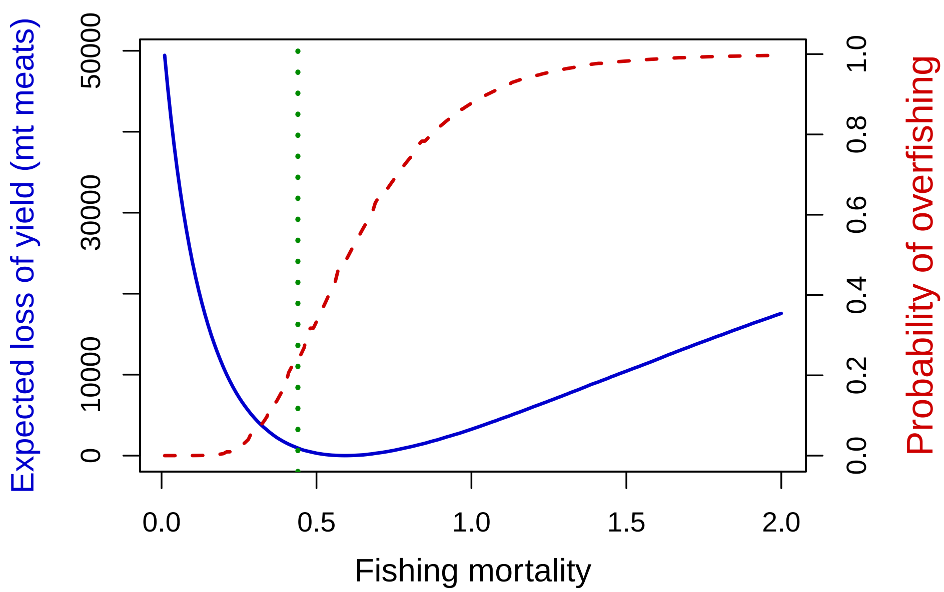


Figure A10.10. The tradeoff between overfishing risk (red dashed lines) and loss of expected yield (solid blue line). The vertical green line marks the fishing mortality where the risk of overfishing is 0.25, which is the fishing mortality corresponding to the ACL/ABC.

11 Stock Status (TOR 7)

According to the sea scallop fishery management plan (NEFMC 2003), sea scallops are overfished when the survey biomass index for the whole stock falls below $\frac{1}{2}B_{\mathrm{TARGET}}$, with B_{TARGET} set equal to B_{MSY} or its proxy. The target biomass estimated in NEFSC (2014) was $B_{\mathrm{TARGET}} = B_{\mathrm{MSY}} = 96,480$ mt meats corresponding to a fishing mortality reference point of $F_{\mathrm{MSY}} = 0.48$. Using the NEFSC (2014) model, the estimated biomass in 2017 was 458,665 mt meats (395,610 meats not including DSENLS scallops), over four times the corresponding B_{MSY} value. The estimated $B_{\mathrm{MSY}} = B_{\mathrm{TARGET}}$ using the models from this assessment is $116,766$ mt meats. The estimated combined stock biomass in 2017 was 380,389 mt meats (317,334 mt meats excluding DSENLS scallops), well above both the recommended new B_{TARGET} values. Using either modeling approach, the probability that the biomass is above $B_{\mathrm{TARGET}}/2$ is almost exactly 1, so there is essentially no chance that the stock is overfished.

Overall fishing mortality was 0.12 in 2017 (CV= 0.07), which is well below both the NEFSC (2014) estimate of $F_{\mathrm{MSY}} = 0.48$ and the new estimate of $F_{\mathrm{MSY}} = 0.64$. Therefore, overfishing was not occurring in 2017 according to both definitions. According to the CASA model, the probability that overall fishing mortality is greater than 0.64 is essentially zero. The SYM model runs indicate that there is a probability of 0.003 that the true F_{MSY} is below 0.12. In conclusion, it is almost certain the that stock is not overfished, and overfishing is not occurring.

12 Stock Projections (TOR 8)

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 has 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 A7). Various versions of SAMS have been used since 1999. In the current version of SAMS used here, the resource is divided into 21 separate subareas, 8 in the Mid-Atlantic and 13 on Georges Bank (Figure A12.1).

Growth is modelled in SAMS and CASA in a similar manner, except that each subarea in SAMS has its own stochastic growth transition matrix derived from the shell increments collected in that area from the most recent growth period. Natural and fishing mortality, recruitment, and shell height to meat weight relationships are also area-specific. Fishing mortality in a subarea can either be explicitly specified in each area or calculated using a simple fleet dynamics model which assumes fishing effort is proportional to estimated LPUE. Adult natural mortality for these runs was set at 0.2 for the Georges Bank areas, and 0.25 for the Mid-Atlantic. Juvenile natural mortality was calculated as a piecewise linear function of density in that subarea, based on CASA model runs, which suggest that juvenile natural mortality is greater at high densities. For the Mid-Atlantic, juvenile natural mortality was $M_J = 0.25$ except when juvenile density (as measured by dredge units for scallops 40-75 mm) was above 261 scallops per tow (approximately the 90th percentile of observations), above which juvenile natural mortality is calculated as $M_J = M_0 + m(D_J - D_0)$, where $M_0 = 0.25$, $m = 0.0001495$, D_J is the juvenile density in the subarea (in dredge survey units), and $D_0 = 261$. These parameters were estimated based on CASA estimated juvenile natural mortality, and observed dredge survey densities in the Elephant Trunk region, where most of the density dependent natural mortality probably occurred. The evidence for density dependent juvenile M on Georges Bank is less clear; it was modelled the same way, but with $M_0 = 0.2$, $m = 0.0001$, and $D_0 = 500$/tow, making the effects of density dependence weaker than in the Mid-Atlantic.

In previous assessments, projected recruitment is modelled stochastically with the log-transformed mean and covariance for recruitment in each area matching that observed in NEFSC dredge survey time series. For this assessment, recruitment was also scaled to a region-wide Beverton-Holt stock recruitment relationship, making the SAMS model more comparable to the SYM reference points model. Thus, the recruitment as previously estimated is multiplied by $s_r(B_r)/s_r(B_{\mathrm{mean}})$, where s_r is the (mean) regional stock recruit relationship (for two years old) used in the SYM model, B_r is the current regional biomass, and B_{mean} is the mean overall biomass. Because of this addition, estimated recruitment for the projections presented here increased by about 20%, due to the high current biomass.

In the example projections shown here, initial conditions are based on shell height data from the 2017 dredge surveys. Initial values in each subarea are bootstrapped based on the standard errors of the surveys. Further details regarding the SAMS model are given in Appendix A7.

Because the "open" areas are managed through days at sea, LPUE needs to be estimated in these areas. This is done using a linear regression between mean exploitable biomass in the open areas, and observed LPUE in these areas (Figure A12.2). The SAMS model gives projections of mean open area exploitable biomass, which then are translated into projected LPUE using the regression. A more complex model was presented during the working group meetings (Appendix A8), but was not used due to in-sample data fits being comparable to the models historically used (the current approach has only been used since 2016), and much poorer out of sample predictions for 2017 and 2018 than the current approach.

Example SAMS runs

For the example simulations, the stock area was split into 21 subareas (Figure A12.1), eight in the Mid-Atlantic (Virginia, Delmarva, Elephant Trunk flex, Elephant Trunk open, Hudson Canyon South, New York Bight, Long Island, and New York Bight inshore) and thirteen on Georges Bank, including 8 in the groundfish closed areas (Closed Area I, II and Nantucket Lightship EFH closures, Closed Area I, II access areas, Nantucket Lightship access area north, south/shallow, and south deep (where the DSENLS scallops are located), and five in the "open areas: Great South Channel, Northern Edge, Southern Flank, Nantucket Lightship Extension (east of the Nantucket Lightship Closed Area), and Closed Area II extension (south of Closed Area II). Currently, the Gulf of Maine is not included as a SAMS areas, although a similar method was used to give catch advice for that area (see Appendix A4).

Basic example projections assumed the stock is fished uniformly at $F = 0.58$ in all areas. A total of $n = 1000$ projection runs were performed in this example with stochastic initial conditions, and recruitment. Four different sets of projections were run: A base case, one like the base but with the bootstrapping turned off, one that assumes adult natural mortality was as in SARC-59 ($M = 0.16$ on Georges Bank and $M = 0.2$ in the Mid-Atlantic), and one where the growth parameter L_∞ was reduced by 5% in all areas. The alternative runs were the same as the base case except as specified.

Results indicate that projected mean biomass is expected to decline as the very large year classes currently in both regions die due to both fishing and natural mortality (Figure A12.3). Catches under the assumed fishing mortalities are also expected to decline similarly to biomass. However, even in 2020, all but one of the 1000 runs remained above the estimated $B_{\text{MSY}} = 116,766$ mt meats (Figure A12.4). Thus, it is highly unlikely that this stock will become overfished in the near future, unless something not anticipated in the model (e.g., mass mortality or much higher fishing mortality) occurs.

Variability in SSB and catch in 2018 is considerably reduced when bootstrapping is turned off (Figure A12.5). However, the variability increases with year in this run, indicating that recruitment variability becomes more important for later years, whereas the initial conditions drive most of the variability for the first year or two. Assuming decreased natural mortality slightly increases catches and SSB, whereas decreasing growth sightly decreases SSB and catch (Figures A12.6 and A12.7).

None of these projections are fully realistic, since they assume fishing at F_{MSY} with no spatial management. To give an example of a more realistic projection, the preferred model run from the most recent management action (Framework 29, NEFMC 2017) is presented here, except with updated parameters and model structure, as discussed above. These models are typically run for 15 years, to give an idea of their long-term consequences. Open area fishing mortality was set at $F = 0.4$ in 2018, and $F = 0.48$ for later years. One area was assumed to be closed for the duration of the simulations (the northern portion of Closed Area II), consistent with current policy. Landings are lower than the other projections, due to lower fishing mortalities and the closed area. This leads to a more gentle decline in SSB (Figure A12.8). Longer term, mean projected SSB gradually declines and then levels off above the estimated B_{MSY}. A few of the runs dropped below B_{MSY} by the end of the simulated time period, but none crossed the $B_{\text{MSY}}/2$ overfishing threshold.

Figure (A12.9) compares landings predicted by SAMS to actual landings. SAMS somewhat underpredicted landings in three years, overestimated in three years, and was almost exactly correct in one. This suggests that the SAMS forecasted landings are close to unbiased with a reasonable degree of precision.

Sea scallops are highly fecund, and have a much higher K/M (Brody growth coefficient/natural mortality) ratio than is typical for finfish. For these reasons, US sea scallops rapidly recovered from overfishing after fishing mortality was reduced and areas were closed. Under the current management restrictions, this stock is therefore unlikely to become overfished. However, because they are very valuable, it would be expected that if these restrictions were lifted or made ineffective, sea scallops could become overfished, as was observed in the 1980s and 1990s.

Figures

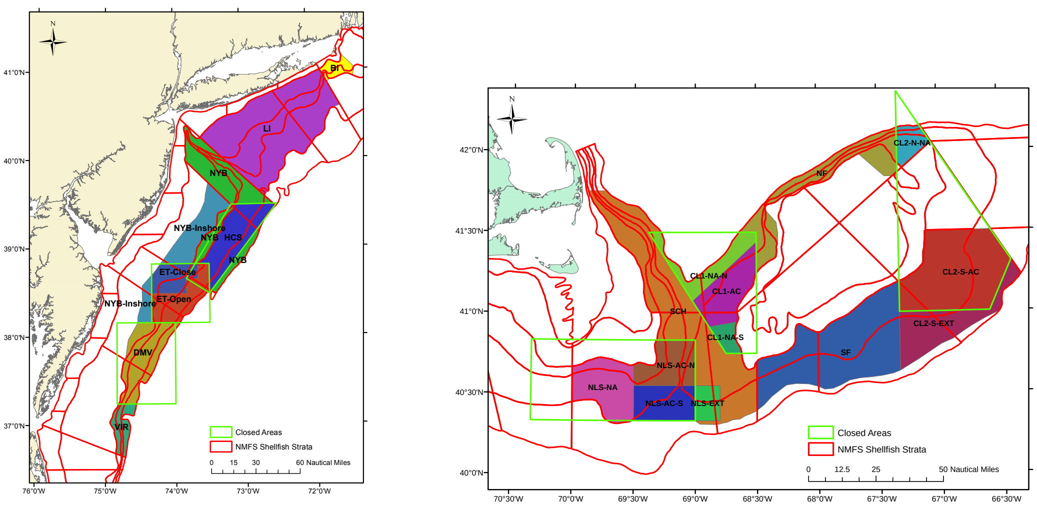


Figure A12.1. Charts of SAMS subareas in the Mid-Atlantic (left), and Georges Bank (right).

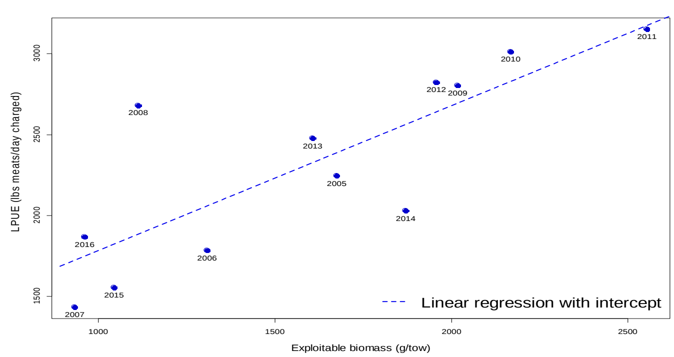


Figure A12.2. Regression of open area LPUE vs. open area exploitable biomass.

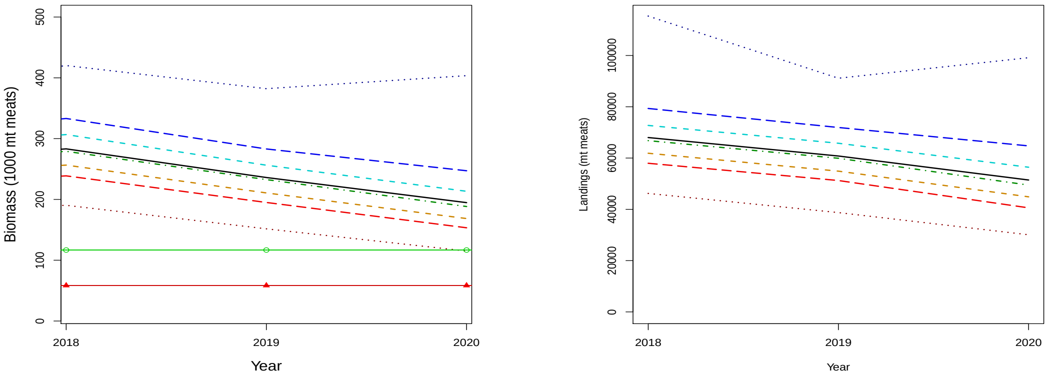


Figure A12.3. Biomass (left) and landings (right) from the base projections. Maximum and minimums of the 1000 runs (dotted lines), 10th and 90th percentiles (long dashed lines), 25th and 75th percentiles (short dashed lines), median (dashed-dotted line), and mean (solid line) are shown. The green line with circles is B_{MSY} and the red line with triangles is the overfishing threshold $B_{\mathrm{MSY}}/2$.

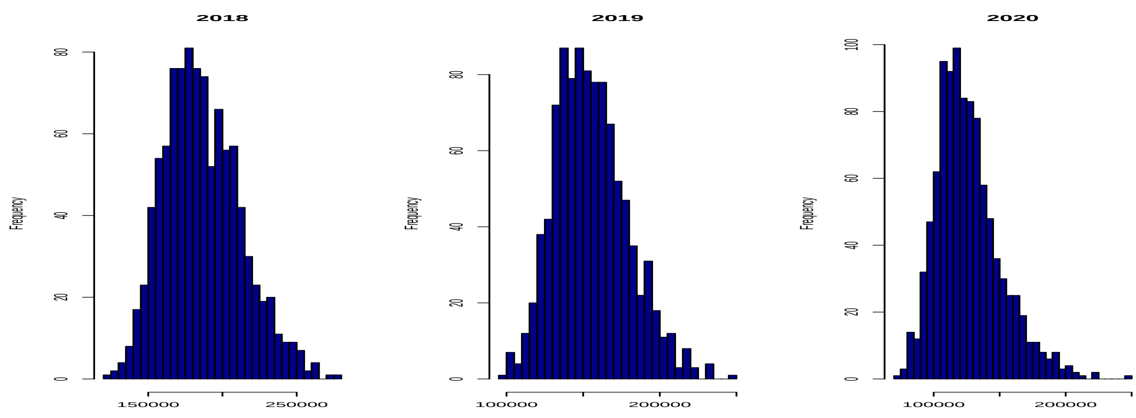


Figure A12.4. Histogram of the probability distributions of biomass for 2018, 2019 and 2020 from the base projection.

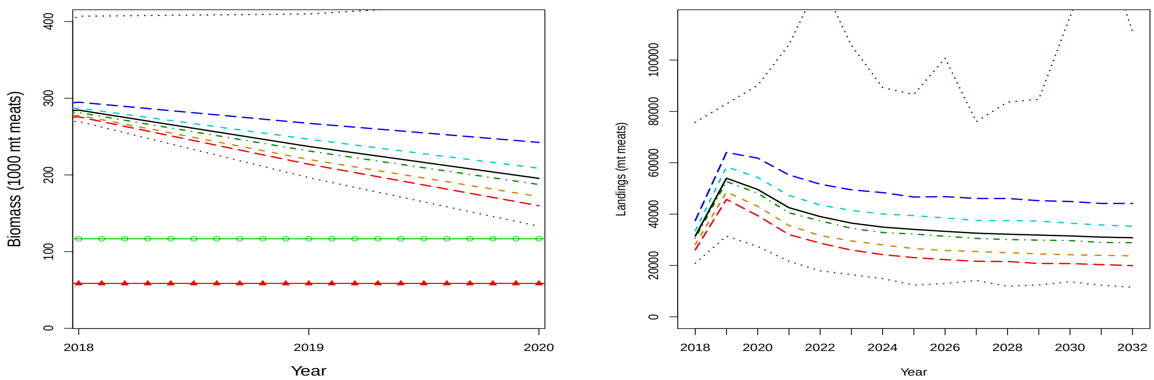


Figure A12.5. Biomass (left) and landings (right) from the no bootstrap projections. Maximum and minimums of the 1000 runs (dotted lines), 10th and 90th percentiles (long dashed lines), 25th and 75th percentiles (short dashed lines), median (dashed-dotted line), and mean (solid line) are shown. The green line with circles is B_{MSY} and the red line with triangles is the overfishing threshold $B_{\text{MSY}}/2$.

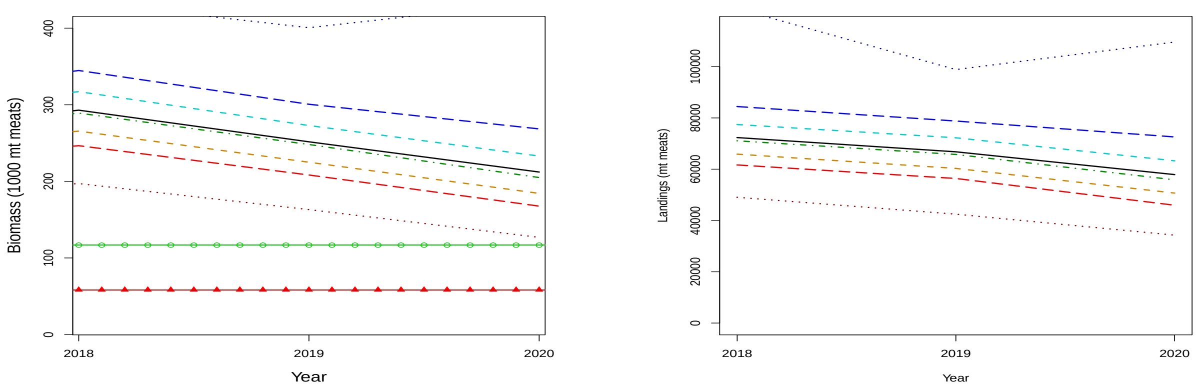


Figure A12.6. Biomass (left) and landings (right) from the lower natural mortality projections. Maximum and minimums of the 1000 runs (dotted lines), 10th and 90th percentiles (long dashed lines), 25th and 75th percentiles (short dashed lines), median (dashed-dotted line), and mean (solid line) are shown. The green line with circles is B_{MSY} and the red line with triangles is the overfishing threshold $B_{\text{MSY}}/2$

.

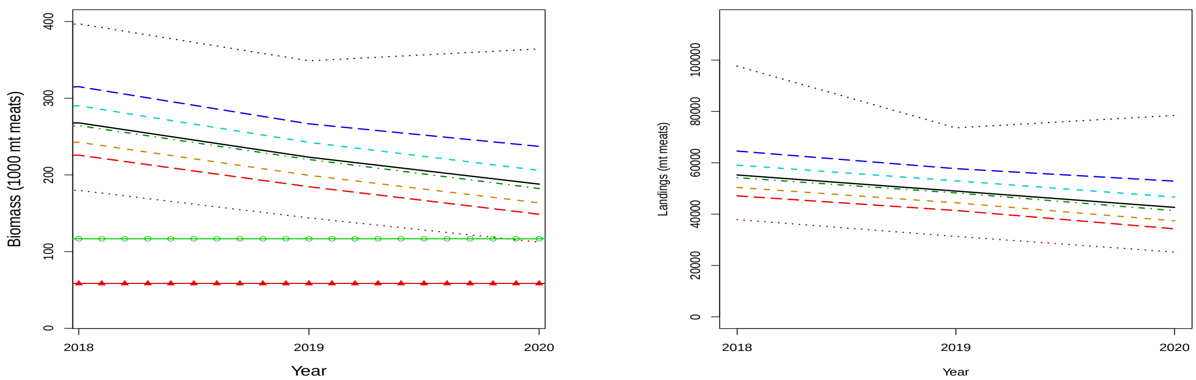


Figure A12.7. Biomass (left) and landings (right) from the slower growth projections. Maximum and minimums of the 1000 runs (dotted lines), 10th and 90th percentiles (long dashed lines), 25th and 75th percentiles (short dashed lines), median (dashed-dotted line), and mean (solid line) are shown.

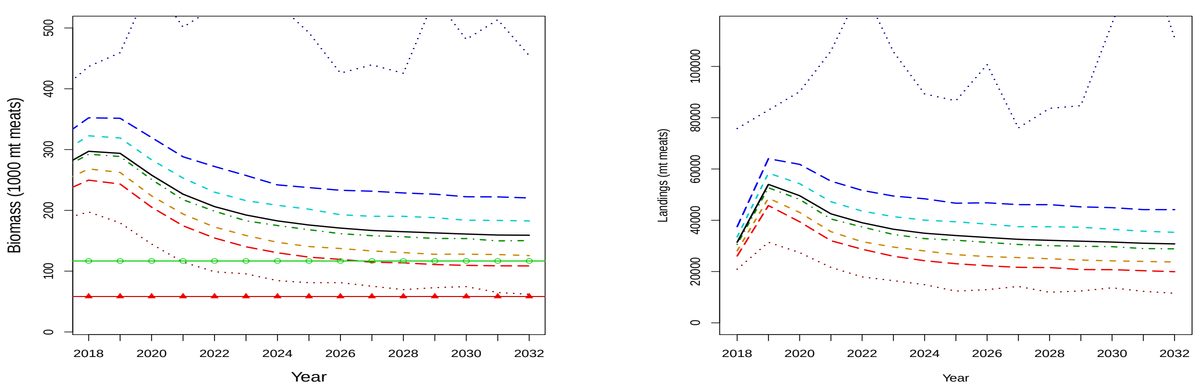


Figure A12.8. Biomass (left) and landings (right) for the long term Framework 29 projections. Maximum and minimums of the 1000 runs (dotted lines), 10th and 90th percentiles (long dashed lines), 25th and 75th percentiles (short dashed lines), median (dashed-dotted line), and mean (solid line) are shown. The green line with circles is B_{MSY} and the red line with triangles is the overfishing threshold $B_{\mathrm{MSY}}/2$.

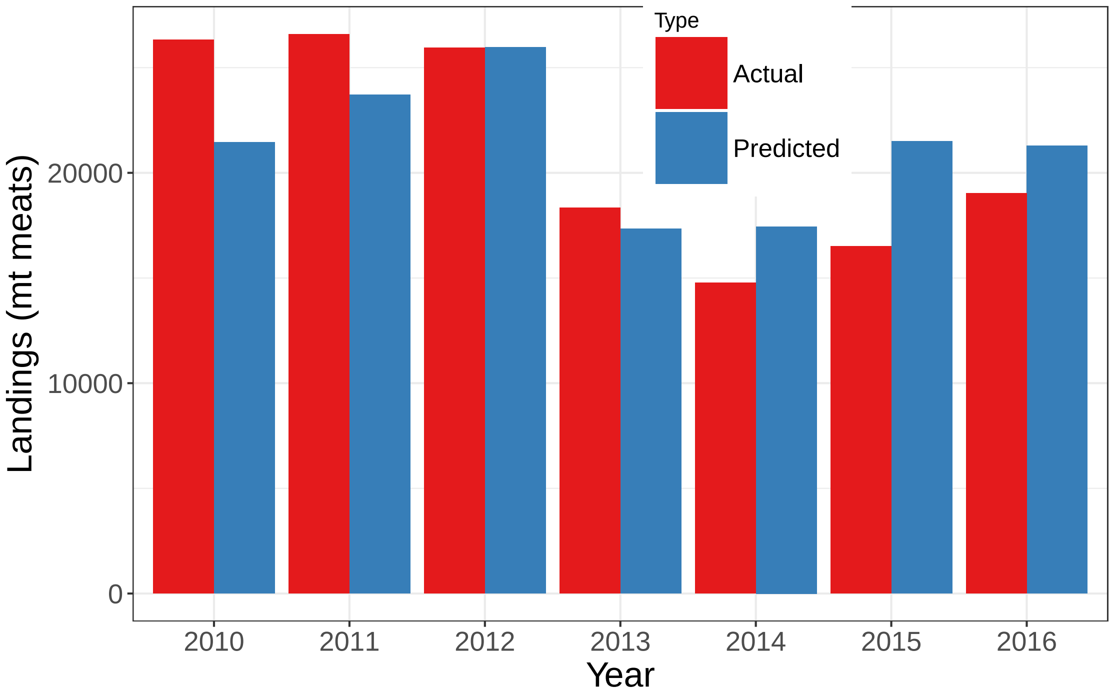


Figure A12.9. Comparison of predicted landings by the SAMS model compared to actual landings, 2010-2016.

13 Research Recommendations (TOR 9)

Previous recommendations from NEFSC (2014)

1. *Investigate methods for better survey coordination between the various survey programs.*
   Some progress has been made coordinating the RSA dredge and Habcam surveys with NEFSC surveys.

2. *Evaluate effects of uncertainty in identifying dead scallops in optical surveys and improve procedures for identifying dead scallops.*
   No real progress have been made aside from improving quality of imagery in optical surveys.

3. *Collect data to refine estimates of incidental mortality. Analytical procedures were improved this assessment but further progress awaits collection of more data.*
   Several studies have been funded and reported in this assessment and the incidental mortalities were updated. However, there is still need for better estimates of incidental mortality on hard-bottom habitats on Georges Bank and the Gulf of Maine.

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.*

Some progress has been made on improved training for NEFSC Habcam annotators but there is still a need to develop formal QA/QC procedures.

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.*
   Some progress have been made on survey designs and the analytical procedures have been automated. In addition, automated annotation software utilizing deep learning computer vision algorithms are under development (e.g., Chang et al. 2016).

6. *Quantify and improve accuracy of SAMS projection models used to specify harvest levels. Recent projections appear to overestimate stock size to some extent.*
   New features such as density-dependent juvenile natural mortality and increased adult natural mortality will reduce the degree of overestimation on stock size.

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.*
   Interannual variations in juvenile natural mortality is now explicitly included in this assessment.

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.*
   Archived shells from the 1980s and 1990s were analyzed for this assessment, which documents changes in growth over time.

9. (1) *Refine models that predict scallop recruitment based on chlorophyll and predator data in order to improve estimates from stock assessment and projection models.*
   A funded project is investigating the effects of changes in climate and predator abundance on scallop recruitment, but was not completed in time for this assessment.
   (2) *Investigate statistical approaches to estimating year class strength directly from survey data.*
   No progress.

10. *Investigate and quantify the utility of multiple scallop surveys.*
    This was discussed during the 2015 sea scallop survey review.

New research recommendations

1. Further investigate methods for better survey coordination between the various survey programs, including survey design, timing, and standardized data formatting for easier sharing.

2. Investigate changes in dredge efficiency and saturation due to high scallop densities or high bycatch rates.

3. Analyze past juvenile scallop mortality events and develop better methods to model time-varying mortality in the assessment models.
4. Collect information needed for the management of the GOM fishery and development of appropriate reference points including biological parameters, fishery-independent surveys, and fishery-dependent data.
5. Continue development of scallop ageing methods and examination of scallop growth processes including density dependent effects.
6. Improve training of annotators used in optical surveys and develop standardized QA/QC procedures for data collected from imagery.
7. Investigate use of software for automated annotation of imagery from optical surveys.
8. Investigate methods to better estimating biomass and abundance variances from Habcam optical surveys including development of Bayesian geostatistical methods.
9. Investigate and estimate current and historical unreported landings and effects of spatially heterogeneous fishing mortality on mortality estimates.
10. Develop a spatially-explicit methodology for forecasting the abundance and distribution of sea scallops by incorporating spatial data from surveys, landings, and fleet effort (aka GEOSAMS).
11. Investigate and parameterize sub-lethal effects of disease, parasites, or discarding on mortality, growth, and landings.
12. Revive and streamline previously-developed methods for interpreting VMS data.
13. Further refine and test methods for forecasting LPUE.
14. Continued investigation of discard mortality, particularly during warm water periods, by incorporating environmental data.
15. Continue improvements of observer recordings for vessel fishing behavior including deck loading and shucking dynamics in responses to disease or poor scallop health.
16. Continue investigating the extent of incidental fishing mortality, particularly on hard bottom habitats.

References

Almeida F, T Sheehan, Smolowitz R (1994) Atlantic sea scallop, *Placopecten magellanicus*, maturation on Georges Bank during 1993. NEFSC Ref Doc 94-13

Andersen S, Grefsrud ES, Harboe T (2013) Effect of increased pCO_2 on early shell development in great scallop (*Pecten maximus* Lamarck) larvae. Biogeosciences Discuss 10(2)

Barbeau MA, Scheibling RE, Hatcher BG (1998) Behavioural responses of predatory crabs and sea stars to varying density of juvenile sea scallops. Aquaculture 169:87-98

Barber BJ, Getchell R, Shumway S, Schick D (1988) Reduced fecundity in a deep-water population of the giant scallop *Placopecten magellanicus* in the Gulf of Maine, USA. Mar Ecol Prog Ser 207-212

Bethoney ND, Asci S, Stokesbury KDE (2016) Implications of extremely high recruitment events into the US sea scallop fishery. Mar Ecol Prog Ser 547:137-147.

Bethoney ND, Stokesbury KDE (2018) Methods for image-based surveys of benthic macroinvertebrates and their habitat exemplified by the drop camera survey for the Atlantic sea scallop. Journal of visualized experiments: JoVE, (137).

Beverton RJH, Holt SJ (1956) A review of methods for estimating mortality rates in fish populations, with special reference to sources of bias in catch sampling. Rapp P-v Reun Cons Int Explor Mer 140:67-83

Beverton RJH, Holt SJ (1957) On the Dynamics of Exploited Fish Populations. Chapman and Hall, London

Beverton RJH, Holt SJ (1959) A review of the lifespans and mortality rates of fish in nature and the relation to growth and other physiological characteristics. 142-177p in:Ciba Foundation colloquia in ageing. Churchill, London

Caddy JF (1973) Underwater observations on tracks of dredges and trawls and some effects of dredging on a scallop ground. J Fish Res Bd Can 30:173-180

Caddy JF, Radley-Walters C (1972) Estimating count per pound of scallop meats by volumetric measurement. Fisheries Research Board of Canada Manuscript Report Series, Biological Station, St. Andrews, N.B. 11 p.

Chang JH, Hart DR, Shank BV, Gallager SM, Honig P, York AD (2016) Combining imperfect automated annotations of underwater images with human annotations to obtain precise and unbiased population estimates. Methods Oceanogr 17:169-186

Chang JH, Shank BV, Hart DR (2017) A comparison of methods to estimate abundance and biomass from belt transect surveys. Limnol Oceanogr Methods 15:480-494

Charnov EL (1993) Life history invariants:Some explorations of symmetry in evolutionary ecology. Oxford Press, Oxford

Chute AS, Wainright SC, Hart DR (2012) Timing of shell ring formation and patterns of shell growth in the sea scallop *Placopecten magellanicus* based on stable oxygen isotopes. J Shellfish Res 31(3):649-662

Claereboudt MR, Himmelman JH (1996) Recruitment, growth and production of giant scallops (*Placopecten magellanicus*) along an environmental gradient in Baie des Chaleurs, eastern Canada. Mar Biol 124(4):661-670

Cochran WG (1977) Sampling Techniques, 3rd ed. Wiley & Sons, New York. 428p

Cooley SR, Rheuban JE, Hart DR, Luu V, Glover DM, Hare JA, Doney SC (2015) An integrated assessment model for helping the United States sea scallop (*Placopecten magellanicus*) fishery plan ahead for ocean acidification and warming. PLoS One 10(5) p.e0124145

Deming WE (1960) Sample design in business research. Wiley & Sons, Inc. New York. 517p

Dibacco C, Robert G, Grant J (1995) Reproductive cycle of the sea scallop, *Placopecten magellanicus* (Gmelin, 1791), on northeastern Georges Bank. J Shellfish Res 14:59-69

Dickie LM (1955) Fluctuations in abundance of the giant scallop, *Placopecten magellanicus* (Gmelin), in the Digby Area of the Bay of Fundy. J Fish Res Bd Can 12:797-857

DuPaul WD, Fisher RA, Kirkley JE (1990) An evaluation of at-sea handling practices:Effects on sea scallop meat quality, volume and integrity. Virginia Sea Grant Marine Advisory Program

DuPaul WD, Kirkley JE, Schmitzer AC (1989) Evidence of a semiannual reproductive cycle for the sea scallop, *Placopecten magellanicus* (Gmelin, 1791), in the Mid-Atlantic region. J Shellfish Res 8:173-178

Elner RW, Jamieson GS (1979) Predation of sea scallops, *Placopecten magellanicus*, by the rock crab, *Cancer irroratus*, and the American lobster, *Homarus americanus*. J Fish Res Board Can 36:537-543

Ferraro DM, Trembanis AC, Miller DC, Rudders DB (2017) Estimates of sea scallop (*Placopecten magellanicus*) incidental mortality from photographic multiple before-After-Control-Impact Surveys. J Shellfish Res 36(3):615-626

Franz DR, Worley EK (1982) Seasonal variability of prey in the stomachs of *Astropecten americanus* (Echinodermata Asteroidea) from off Southern New England, USA. Estuar Coast Shelf Sci 14(4):355-368

Franz DR, Worley EK, Merrill AS (1981) Distribution patterns of common seastars of the middle Atlantic continental shelf of the northwest Atlantic (Gulf of Maine to Cape Hatteras). Biol Bull 160(3):394-418

Hart DR (2001) Individual-based yield-per-recruit analysis, with an application to the Atlantic sea scallop, *Placopecten magellanicus*. Can J Fish Aquat Sci 58:2351-2358

Hart DR (2003) Yield- and biomass-per-recruit analysis for rotational fisheries, with an application to the Atlantic sea scallop (*Placopecten magellanicus*). Fish Bull 101:44-57

Hart DR (2006) Effects of sea stars and crabs on sea scallop *Placopecten magellanicus* recruitment in the Mid-Atlantic Bight (USA). Mar Ecol Prog Ser 306:209-221.

Hart DR (2013) Quantifying the tradeoff between precaution and yield in fishery reference points. ICES J Mar Sci 70(3):591-603

Hart DR, Chute AS (2004) Essential fish habitat source document:Sea scallop, *Placopecten magellanicus*, life history and habitat characteristics, 2nd ed. NOAA Technical Memorandum NMFS NE-189

Hart DR, Chute AS (2009a) Verification of Atlantic sea scallop, *Placopecten magellanicus*, shell growth rings by tracking cohorts in fishery closed areas. Can J Fish Aquat Sci 66:751-758

Hart DR, Chute AS (2009b) Estimating von Bertalanffy growth parameters from growth increment data using a linear mixed-effects model, with an application to the sea scallop *Placopecten magellanicus*. ICES J Mar Sci 66:2165-2175

Hart DR, Jacobson LD, Tang J (2013) To split or not to split:Assessment of Georges Bank sea scallops in the presence of marine protected areas. Fish Res 144:74-83

Hart DR, Rago PJ (2006) Long-term dynamics of US Atlantic sea scallop *Placopecten magellanicus* populations. N Amer J Fish Management 26:490-501

Hart, DR and Shank BV (2011) Mortality of sea scallops *Placopecten magellanicus* in the Mid-Atlantic Bight: Comment on Stokesbury et al. (2011). Mar Ecol Prog Ser 443:293-297.

Hatcher BG, Scheibling RE, Barbeau MA, Hennigar AW, Taylor LH, Windust AJ (1996) Dispersion and mortality of a population of sea scallop (*Placopecten magellanicus*) seeded in a tidal channel. Can J Fish Aquat Sci 53:38-54

Hennen DR, Hart DR (2012) Shell height-to-weight relationships for Atlantic sea scallops (*Placopecten magellanicus*) in offshore US waters. J Shellfish Res 31(4):1133-1144

Inglis SD, Kristmundsson Á, Freeman MA, Levesque M, Stokesbury K (2016). Gray meat in the Atlantic sea scallop, *Placopecten magellanicus*, and the identification of a known pathogenic scallop apicomplexan. Journal of invertebrate pathology 141:66-75.

Jacobson LD, Stokesbury KDE, Allard MA, Chute AS, Harris BP, Hart DR, Jaffarian T, Marino MC, Nogueira JI, Rago PJ (2010) Measurement errors in body size of sea scallops (*Placopecten magellanicus*) and their effect on stock assessment models. Fish Bull 108(2):233-247

Jenkins SR, Brand AR (2001) The effect of dredge capture on the escape response of the great scallop, *Pecten maximus* (L.):implications for the survival of undersized discards. J Exp Mar Biol Ecol 266:33-50

Kaplan KA, Hart DR, Hopkins K, Gallager S, York A, Taylor R, Sullivan PJ (2017) Evaluating the interaction of the invasive tunicate *Didemnum vexillum* with the Atlantic sea scallop *Placopecten magellanicus* on open and closed fishing grounds of Georges Bank. ICES J Mar Sci 74(9):2470-2479

Kaplan KA, Hart DR, Hopkins K, Gallager S, York A, Taylor R, Sullivan PJ (2018) Invasive tunicate restructures invertebrate community on fishing grounds and a large protected area on Georges Bank. Biol Invasions 20:87-103

Lake NCH, Jones MB, Paul JD (1987) Crab predation on scallop (*Pecten maximus*) and its implication for scallop cultivation. J Mar Biol Assoc UK 67:55-64

Lester RJG, Blair D, Heald D (1980) Nematodes from scallops and turtles from Shark Bay, Western Australia. Mar Freshwater Res 31:713-717

Levesque MM, Inglis SD, Shumway SE, Stokesbury KD (2016). Mortality assessment of Atlantic sea scallops (Placopecten magellanicus) from gray-meat disease. J Shellfish Res 35:295-305.

Lichtenfels JR, Bier JW, Madden PA (1978) Larval anisakid (*Sulcascaris*) nematodes from Atlantic molluscs with marine turtles as definitive hosts. Trans Am Microscop Soc 97:199-207

Lichtenfels JR, Sawyer TK, Miller GC (1980) New hosts for larval *Sulcascaris* sp.(Nematoda, Anisakidae) and prevalence in the calico scallop (*Argopecten gibbus*). Trans Am Microscop Soc 99:448-451

MacDonald BA, Thompson RJ (1985) Influence of temperature and food availability on the ecological energetics of the giant scallop *Placopecten magellanicus*. II. Reproductive output and total production. Mar Ecol Prog Ser 25:295-303

MacDonald BA, Thompson RJ (1986a) Influence of temperature and food availability on the ecological energetics of the giant scallop *Placopecten magellanicus*. Mar Biol 93(1):37-48

MacDonald BA, Thompson RJ (1986b) Production, dynamics and energy partitioning in two populations of the giant scallop *Placopecten magellanicus* (Gmelin). J Exp Mar Biol Ecol 101:285-299

MacDonald BA, Thompson RJ, Bayne BL (1987) Influence of temperature and food availability on the ecological energetics of the giant scallop *Placopecten magellanicus*. Oecologia 72(4):550-556

Merrill AS, Posgay JA (1964) Estimating the natural mortality rate of sea scallop. Res Bull Int Comm NW Atlantic Fish 1:88-106

Merrill AS, Posgay JA, Nichy F (1966) Annual marks on shell and ligament of sea scallop (*Placopecten magellanicus*). Fish Bull 65:299-311

Miller TJ, Hart DR, Hopkins K, Vine NH, Taylor R, York AD, Gallager SM (2018) Estimation of the capture efficiency and abundance of Atlantic sea scallops (*Placopecten magellanicus*) from paired photographic-dredge tows using hierarchical models. Can J Fish Aquat Sci, in press

Morris JA, Carman MR (2012) Fragment reattachment, reproductive status, and health indicators of the invasive colonial tunicate Didemnum vexillum with implications for dispersal. Biol Invasions 14(10):2133-2140

Morris Jr JA, Carman MR, Hoagland KE, Green-Beach ER, Karney RC (2009) Impact of the invasive colonial tunicate Didemnum vexillum on the recruitment of the bay scallop (*Argopecten irradians*) and implications for recruitment of the sea scallop (*Placopecten magellanicus*) on Georges Bank. Aquat Invasions 4(1):207-211

Murawski SA, Brown R, Lai HL, Rago PJ, Hendrickson L (2000) Large-scale closed areas as a fishery-management tool in temperate marine systems: the Georges Bank experience. Bull Mar Sci 66:775-798.

Murawski SA, Serchuk FM (1989) Environmental effects of offshore dredge fisheries for bivalves. ICES CM K:27

Naidu KS (1987) Efficiency of meat recovery from Iceland scallops (*Chlamys islandica*) and sea scallops (*Placopecten magellanicus*) in the Canadian offshore fishery. J Northw Atl Fish Sci 7:131-136

NEFMC (1993) Amendment #4 and supplemental environmental impact statement to the sea scallop fishery management plan. New England Fisheries Management Council, Saugus, MA

NEFMC (2003) Final Amendment 10 to the Atlantic sea scallop fishery management plan with a supplemental environmental impact statement, regulatory impact review, and regulatory flexibility analysis. New England Fisheries Management Council, Newburyport, MA

NEFSC (1999) 29th Northeast Regional Stock Assessment Workshop (29th SAW). Stock Assessment Review Committee (SARC) consensus summary of assessments. Northeast Fish Sci Cent Ref Doc 99-14, Woods Hole, MA

NEFSC (2001) 32th Northeast Regional Stock Assessment Workshop (32th SAW). Stock Assessment Review Committee (SARC) consensus summary of assessments. Northeast Fish Sci Cent Ref Doc 01-05, Woods Hole, MA

NEFSC (2004) 39th Northeast Regional Stock Assessment Workshop (39th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc 04-10, Woods Hole, MA

NEFSC (2007) 45th Northeast Regional Stock Assessment Workshop (45th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc 07-16, Woods Hole, MA

NEFSC (2010) 50th Northeast Regional Stock Assessment Workshop (50th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc 10-17, 844p, Woods Hole, MA

NEFSC (2014) 59th Northeast Regional Stock Assessment Workshop (59th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc 14-09, 782p, Woods Hole, MA

Palmer MC, Wigley SE (2009) Using positional data from vessel monitoring systems to validate the logbookreported area fished and the stock allocation of commercial fisheries landings. N Am J Fish Manag 29:928-942

Rudders DB, Roman S (2018) A cooperative high precision dredge survey to assess the mid-Atlantic sea scallop resource. VIMS Marine Resource Report No. 2018-05 57p

Sampson DB, Scott RD (2011) A spatial model for fishery age-selection at the population level. Can J Fish Aquat Sci 68(6):1077-1086

Schalkhausser B, Bock C, Stemmer K, Brey T, Prtner HO, Lannig G (2013) Impact of ocean acidification on escape performance of the king scallop, *Pecten maximus*, from Norway. Mar Biol 160(8):1995-2006

Serchuk FM, Wigley SE (1989) Current resource conditions in USA Georges Bank and Mid-Atlantic sea scallop populations:Results of the 1989 NMFS sea scallop research vessel survey. NEFSC SAW-9 Working Paper No. 9 52p.

Shepherd JG (1982) A versatile new stock-recruitment relationship for fisheries, and the construction of sustainable yield curves. ICES J Mar Sci 40(1):67-75

Shumway SE, Selvin R, Schick DF (1987) Food resources related to habitat in the scallop *Placopecten magellanicus* (Gmelin, 1791):a qualitative study. J Shellfish Res 6(2):89-95

Smith SJ (1997) Bootstrap confidence limits for groundfish trawl survey estimates of mean abundance. Can J Fish Aquat Sci 54:616-630

Smith SJ, Rago P (2004) Biological reference points for sea scallops (*Placopecten magellanicus*):the benefits and costs of being nearly sessile. Can J Fish Aquat Sci 61:1338-1354

Smolowitz, RJ, Serchuk FM, Redman RJ (1989) The use of a volumetric measure for determining sea scallop meat count. J Shellfish Res 8:151 − 172.

Stevenson JA, Dickie LM (1954) Annual growth rings and rate of growth of the giant scallop *Placopecten magellanicus* (Gmelin) in the Digby Area of the Bay of Fundy. J Fish Res Board Can 11:660-671

Stokesbury KDE, Harris BP, Marino II MC, Nogueira JI (2004) Estimation of sea scallop abundance using a video survey in off-shore U.S. waters. J Shellfish Res 23:33-40

Sullivan PJ, Lai HL, Gallucci VF (1990) A catch-at-length analysis that incorporates a stochastic model of growth. Can J Fish Aquat Sci 47:184-198

Talmage SC, Gobler CJ (2009) The effects of elevated carbon dioxide concentrations on the metamorphosis, size, and survival of larval hard clams (*Mercenaria mercenaria*), bay scallops (*Argopecten irradians*), and Eastern oysters (*Crassostrea virginica*). Limnol Oceanogr 54(6):2072-2080

Tan FC, Cai D, Roddick DL (1988) Oxygen isotope studies on sea scallops, *Placopecten magellanicus*, from Browns Bank, Nova Scotia. Can J Fish Aquat Sci 45(8):1378-1386

Then AY, Hoenig JM, Hall NG, Hewitt DA (2014) Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. ICES J Mar Sci 72:82-92

Thompson KJ, Inglis SD, Stokesbury KDE (2014) Identifying events of the sea scallop *Placopecten magellanicus* on Georges Bank. J Shellfish Res 33(1):77-87

Truesdell SB, Hart DR, Chen Y (2015) Effects of spatial heterogeneity in growth and fishing effort on yield-per-recruit models:an application to the US Atlantic sea scallop fishery. ICES J Mar Sci 73(4):1062-1073

Truesdell SB, Hart DR, Chen Y (2017) Effects of unequal capture probability on stock assessment abundance and mortality estimates:an example using the US Atlantic sea scallop fishery. Can J Fish Aquat Sci 74(11):1904-1917

Veale LO, Hill AS, Brand AR (2000) An in situ study of predator aggregations on scallop (*Pecten maximus* (L.)) dredge discards using a static time-lapse camera system. J Exp Mar Biol Ecol 255:111-129

White MM, McCorkle DC, Mullineaux LS, Cohen AL (2013) Early exposure of bay scallops (*Argopecten irradians*) to high CO2 causes a decrease in larval shell growth. PloS One 8(4) p.e61065

Wigley SE, Hersey P, Palmer JE (2008) A sescription of the allocation procedure applied to the 1994 to 2007 commercial landings data. NEFSC Ref Doc 08-18

Wong MC, Barbeau MA (2005) Prey selection and the functional response of sea stars (*Asterias vulgaris* Verrill) and rock crabs (*Cancer irroratus* Say) preying on juvenile sea scallops (*Placopecten magellanicus* (Gmelin)), and blue mussels (*Mytilus edulis* Linnaeus). J Exp Mar Biol Ecol 327:1-21

Wong MC, Barbeau MA, Hennigar AW, Robinson SMC (2005) Protective refuges for seeded juvenile scallops (*Placopecten magellanicus*) from sea star (*Asterias* spp.) and crab (*Cancer irroratus* and *Carcinus maenas*) predation. Can J Fish Aquat Sci 62:1766-1781

Yochum N, DuPaul WD (2008) Size selectivity of the northwest Atlantic sea scallop (*Placopecten magellanicus*) dredge. J Shellfish Res 27:265-271

SAW 65 Sea Scallop Assessment Report APPENDICES

APPENDIX A1 Sea Scallop Growth

APPENDIX A2 Sea Scallop Shell Height to Meat Weight Relationships

APPENDIX A3 Gulf of Maine Sea Scallop Fishery

APPENDIX A4 Scallop Landing and Discard Estimation

APPENDIX A5 Updated Selectivity of Large Camera Data from the SMAST Drop Camera Survey Using Digital Still Images

APPENDIX A6 CASA Model Technical Details

APPENDIX A7 SAMS Model Technical Details

APPENDIX A8 Alternative LPUE Model

APPENDIX A9 Survey Coverage in Comparison to Scallop Habitat Identified Using Tree Models

APPENDIX A10 Estimation of Reference Points Using Gonad Weights to Represent Spawning Stock Biomass

Appendix A1 - Sea Scallop Growth

Deborah R. Hart and Antonie S. Chute,
Northeast Fisheries Science Center, 166 Water St., Woods Hole MA 02543
Chase Long and Roger Mann,
Virginia Institute of Marine Science, Gloucester Point VA 23062

This is an update on the growth estimates of Hart and Chute (2009b), and follows the methods there except as specifically noted. In brief, the data consist of the shell heights of successive growth rings on the upper valve of sea scallop shells. Because early growth rings may be obscure or missing, these data are used here as a measure of annual growth increments, rather than an estimate of absolute age. The growth rings have been verified to be annual (Hart and Chute 2009a).

A linear mixed-effects model (using lme4 in R) that predicts the shell height of the larger ring of an increment r_2 from the starting ring shell height r_1 is used to estimate the von Bertalanffy coefficients L_∞ and K. Random effects on both the slope and intercept, on the level of individual scallop, take into account individual variability in both K and L_∞, which otherwise would cause bias and artificially inflate sample size. Estimates of individual variation from the random effects are used, together with the estimates of the means of these parameters, to construct stochastic growth matrices for the CASA, SAMS and SYM models. The von Bertalanffy parameter t_0 is not estimatable by these methods, but it is not needed for a size-structured assessment.

A station-level random effect on the intercept only was included for the first time here, so that there are now nested station and individual scallop random effects. The term was included to take into account that scallops within a station are more likely to have similar growth. Including the station-level effect mainly affected the estimated variability in L_∞; not including it, as was done in the past, biases this variability slightly low. In addition, not including the station effect biases the standard errors low since sample size is somewhat inflated. However, the station-level random effect did not measurably affect the estimates of the mean L_∞ and K parameters.

The Hart and Chute (2009b) analysis was based on shells collected in 2001-2007. There were shells collected previous to 2001, but they were archived and not analyzed. NEFSC (2014) updated the Hart and Chute (2009b) analysis with shells collected from 2008-2012 and also archived shells from 1988 and 1993. Growth from these archived shells was slower than that observed in the more recent years. It is probable that this is at least in part a fishery effect. Scallopers tend to target faster growing scallops, leaving a fished population with a disproportionately high level of slower growers. This effect would be expected to become more pronounced with higher fishing mortality; estimates of F for 1988 and 1993 are much higher than in most recent years.

For this assessment, many more archived shells were analyzed; these were collected between 1982 and 2000. In order to discern temporal changes in growth, we define the increment year (iyear) as the year when the second (larger) ring defining that increment was laid down. For example, the iyear of the last increment on a shell collected in 2017 is 2016, since shells are collected on surveys prior to the formation of the annual ring (shell rings in US waters are laid down near the temperature maximum, in fall or late summer, Chute et al. 2012). For Georges Bank (excluding shells collected from Canadian waters and from the DSENLS area), there were 7298 shells analyzed from 1502 stations, which contained a total of 31480 growth increments, with samples spread out across the region (Figure App A1-1). For the Mid-Atlantic, 6486 scallops from 1469 stations were analyzed, comprising 16231 increments, which also covered a wide geographic range. However, no shells were analyzed during 2014-2017.

As an initial exploratory exercise, we fit the following linear model, with, as discussed above, random effects on the individual and station level, to the Georges Bank and Mid-Atlantic growth ring data:

$$r_2 = a_0 + a_1 r_1 + a_2 D + a_{12} r_1 D + a_3 I_{\text{cl}} + a_y \text{as.factor (iyear)} \qquad \text{(App A1-1)}$$

where D is depth, I_{cl} is 1 if the scallop was collected from one of the closed or rotational areas, and 0 if not, and the increment year effects were plotted (FigureApp A1-2). The covariates were included to isolate the year effect from potential confounding effects such as depth. As can be seen from Figure App A1-2, growth varied by year, and there is a negative correlation between fishing mortality (see main document) and growth.

Based on these plots, we split Georges Bank growth into 5 periods, from slowest to fastest: (1) 1993-1996, (2) 1975-1992 and 1997-1999, (3) 2000-2006, (4) 2007-2011, and (5) 2012-2017. Similarly, for the Mid-Atlantic, growth was split into three periods: (Slow) 1975-1977, 1987-2003, and 2006, (Medium) 1978, 1983-1986, 2004-2005, and 2007, and (Fast) 2008-2012. The shell increments were split into these periods, and separate growth curves and matrices were computed for each period. These periods were used to model time-varying growth in the CASA models.

For each region, period was included as a factor in the basic mixed-effects regression:

$$r_2 = a_0 + b_0 r_1 + a_p \text{as.factor(period)} + b_p \text{as.factor(period)} r_1 \qquad \text{(App A1-2)}$$

The slope terms for each period (b_p) were tested using AIC to determine whether the slope (which determines K) was different in that period, and only those b_p that reduced AIC were retained in the final model. Results are shown in Table App A1-1 and Figure App A1-3.

Subarea specific growth estimates are desirable for the SAMS model. Because SAMS is a foward projection model, data were taken only from the latest growth period in each region (i.e., Mid-Atlantic or Georges Bank). For each region, area-specific covariates were added to the basic mixed-effects regression model:

$$r_2 = a_0 + b_0 r_1 + a_s \text{as.factor(period)} + b_s \text{as.factor(period)} r_1 \qquad \text{(App A1-3)}$$

Similar to that done with periods, only the b_s coefficients on the slope that reduced AIC were included in the final model. Additionally, a total of 75 shells from DSENLS scallops were also analyzed. Because their growth is so different than other areas, its growth parameters were estimated seperately, with no station level random effect (due to the paucity of data). Results are given in and Table App A1-2 and Figure App A1-4.

Tables

Table App A1-1. Estimated regional von Bertalanffy parameters, by temporal periods.

Years	L_∞	SDL_∞	K	SDK	SEL_∞	SEK
GB All						
75-93; 97-99	141.5	9.9	0.429	0.109	0.5	0.003
93-96	136.6	9.3	0.429	0.109	0.5	0.003
00-06	140.2	9.0	0.478	0.114	0.4	0.003
07-11	144.6	9.8	0.464	0.113	0.4	0.003
12-16	151.5	11.4	0.429	0.109	0.5	0.003
GB Open						
go 75-93; 97-99	136.8	9.1	0.442	0.110	0.6	0.003
93-96	132.1	8.5	0.442	0.110	0.6	0.003
00-06	135.8	8.2	0.494	0.116	0.6	0.004
07-11	140.1	8.9	0.479	0.114	0.6	0.004
12-16	146.7	10.4	0.442	0.110	0.7	0.003
GB Closed						
75-93; 97-99	145.7	10.7	0.420	0.108	0.7	0.003
93-96	140.8	10.0	0.420	0.108	0.7	0.003
00-06	143.8	9.5	0.470	0.113	0.6	0.004
07-11	148.5	10.4	0.455	0.111	0.7	0.004
12-16	156.0	12.3	0.420	0.108	0.7	0.003
Mid-Atlantic						
75-77; 87-03; 06	131.4	8.2	0.534	0.117	0.4	0.003
78; 83-86; 04-05; 07	133.4	8.1	0.564	0.121	0.4	0.003
79-82; 08-12	137.4	8.6	0.564	0.121	0.4	0.003

Table App A1-2. Estimated subarea specific von Bertalanffy parameters in the most recent growth period. Georges Bank areas are: South Channel (Sch), Northern Edge (NE), Southern Flank (SF), Closed Area I (CA-I), Closed Area II (CA-II), Nantucket Lightship Closed Area (NLS), and the deep water southeast portion of the Nantucket Lightship area (DSENLS). Mid-Atlantic areas are Delmarva (DMV),Elephant Trunk (ET), Hudson Canyon South (HCS), New York Bight (NYB), Long Island (LI), and Inshore.

Region	Year	L_∞	K	SDL_∞	SDK	SEL_∞	SEK
Georges Bank							
Sch	12-16	150.3	0.3966	9.58	0.1173	4.4	0.0055
NE	12-16	148.77	0.3966	9.62	0.1173	4.4	0.0055
SF	12-16	137.29	0.4641	6.74	0.1254	3.9	0.0058
CA-I	12-16	149.36	0.3966	9.73	0.1173	4.4	0.0055
CA-II	12-16	146.89	0.3966	9.27	0.1173	4.4	0.0055
NLS	12-16	151.15	0.3966	10.08	0.1173	4.4	0.0055
DSENLS	15-16	110.3	0.423	18.11	0.101	2.7	0.023
Mid-Atlantic							
DMV	08-12	136.41	0.5467	8.08	0.1362	1.3	0.0057
ET	08-12	137.92	0.5467	8.28	0.1362	1.3	0.0057
HCS	08-12	129.54	0.5467	7.21	0.1362	1.3	0.0057
NYB	08-12	140.79	0.5467	8.69	0.1362	1.3	0.0057
LI	08-12	139.6	0.5467	8.52	0.1362	1.3	0.0057
Inshore	08-12	147.28	0.5467	9.64	0.1362	1.34	0.0057

Figures

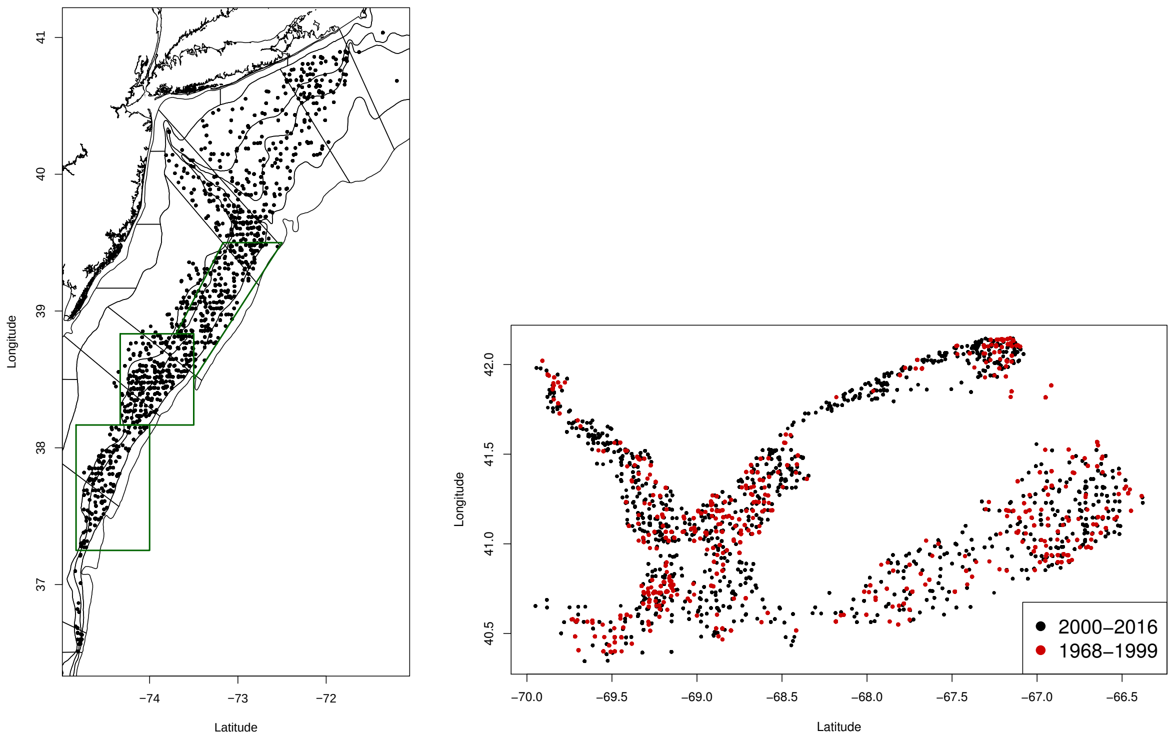
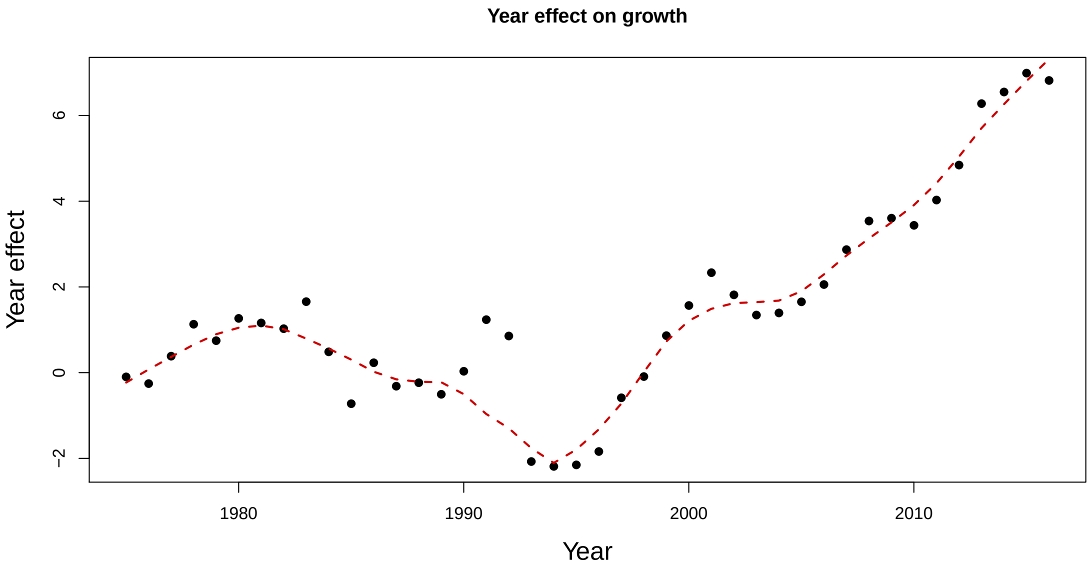


Figure App A1-1. Locations where shells samples were taken in the Mid-Atlantic (left) and Georges Bank (right)



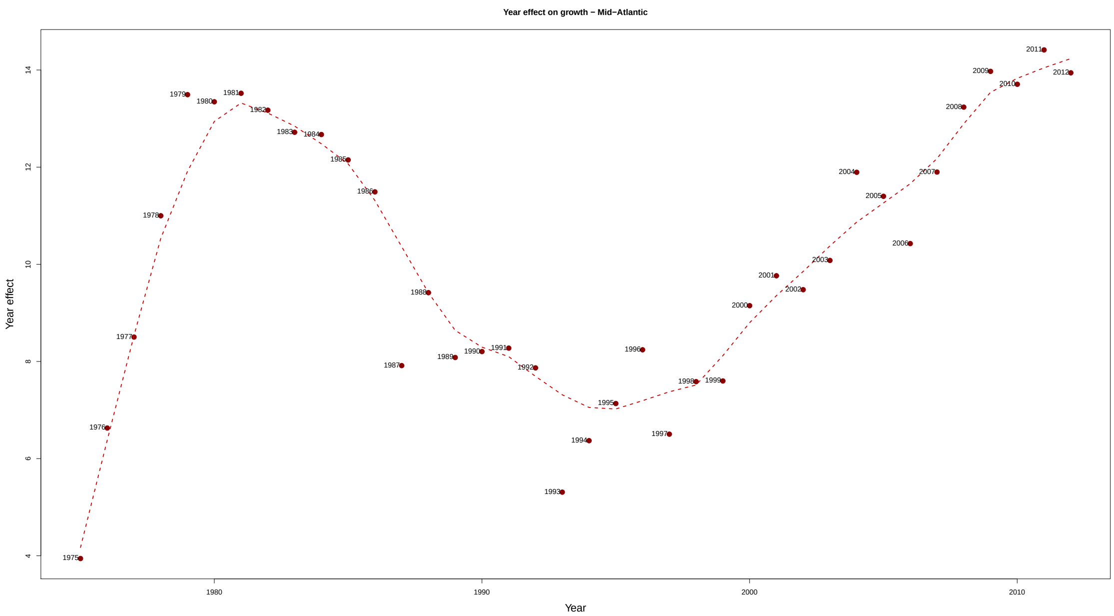
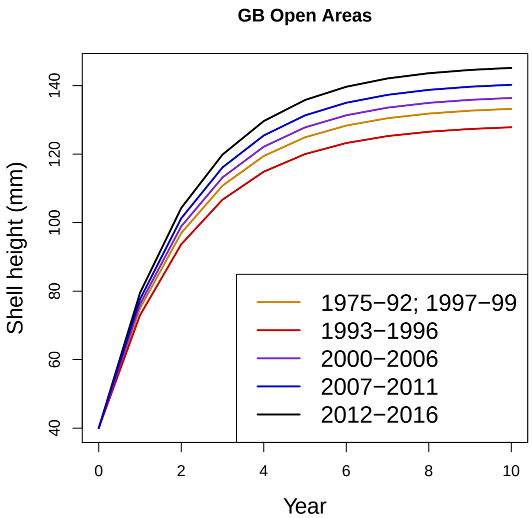
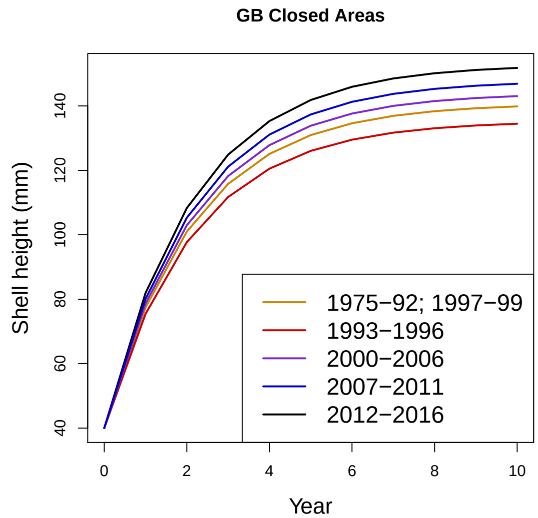


Figure App A1-2. Increment year effects, on Georges Bank (above) and Mid-Atlantic (below)





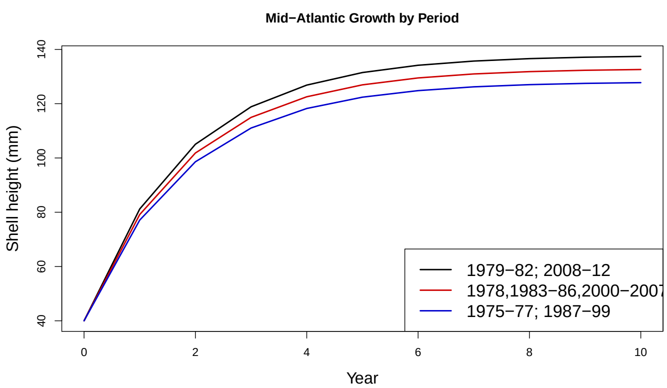


Figure App A1-3. Estimated growth of a 40 mm scallop in each of the growth periods for Georges Bank Open (left), Georges Bank Closed (right), and Mid-Atlantic, below.

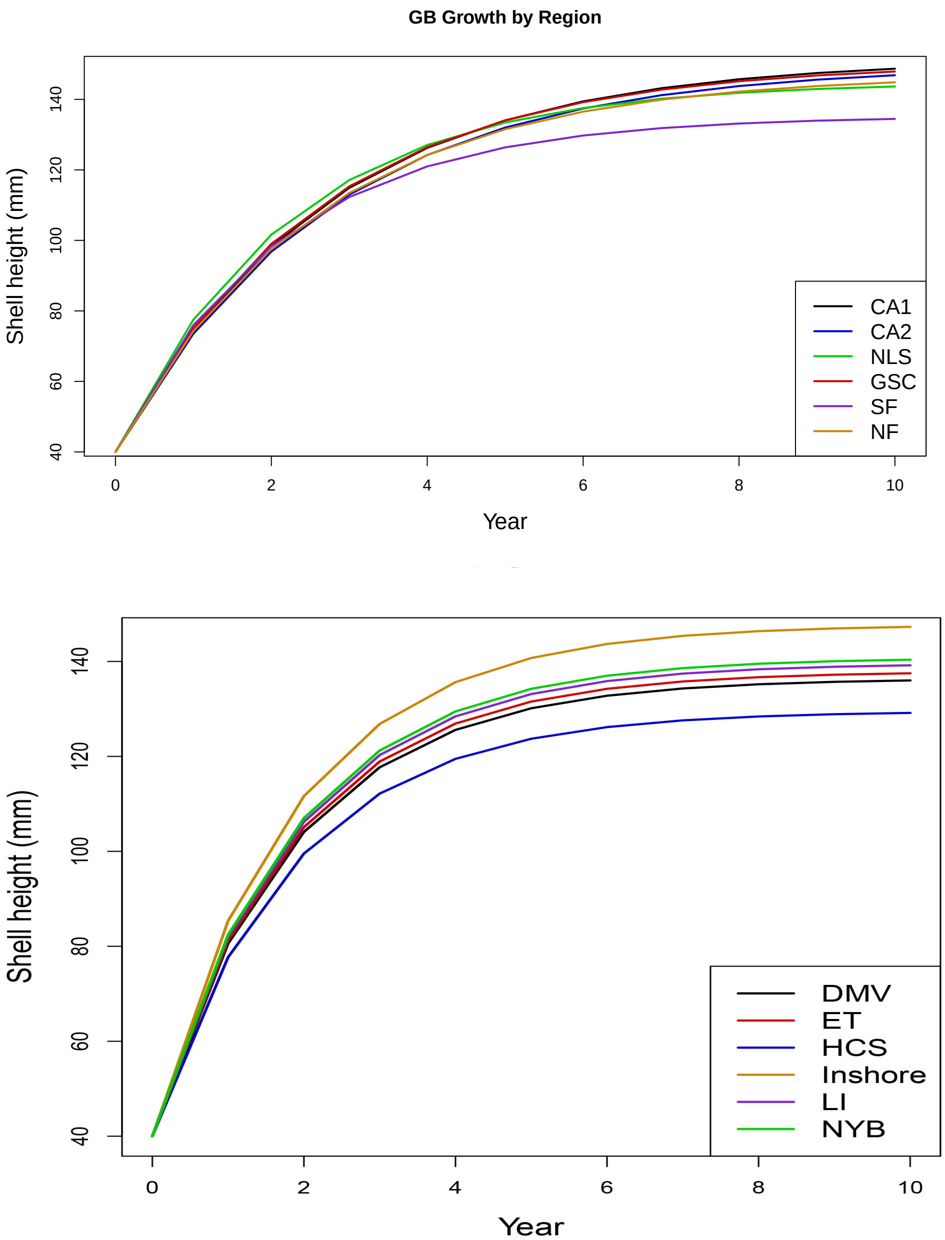


Figure App A1-4. Area-specific growth for the most recent growth period for Georges Bank and the Mid-Atlantic. Georges Bank areas are: South Channel (Sch), Northern Edge (NE), Southern Flank (SF), Closed Area I (CA-I), Closed Area II (CA-II), and Nantucket Lightship Closed Area (NLS). Mid-Atlantic areas are Delmarva (DMV),Elephant Trunk (ET), Hudson Canyon South (HCS), New York Bight (NYB), Long Island (LI), and Inshore.

Appendix A2
Sea Scallop Shell Height/Meat Weight Relationships

Deborah R. Hart
Northeast Fisheries Science Center
166 Water St., Woods Hole MA 02543

Meat weight data is obtained from two sources: surveys and observers. The survey data will be analyzed first.

Shell height to meat weight relationships from survey data

Since 2001, a sample of scallops from the catch (typically 3-10 scallops on about half the stations) on the NEFSC and VIMS dredge surveys are dissected, and their shell height, whole weight, gonad weight and meat weights are measured (see Hennen and Hart 2012). Meat weight is estimated from a generalized linear mixed-effects model, with a random effect for station, a log link, and a gamma distribution:

$$W = \exp(a + b\ln(H)) \qquad \text{(App A2-1)}$$

where W is meat weight and H is shell height. Hennen and Hart (2012) and previous assessments used the lme4 package in R for this analysis, but for the analysis in this assessment, models using this package were often unstable and/or did not converge. Thus, the r2glmm package was used instead which proved much more stable. In cases where using both models converged, they gave similar answers.

Meat weights vary seasonally, and the survey timing has varied. Prior to 2009, the Mid-Atlantic survey was conducted in late June and/or July, but since then, the survey has been conducted in May and/or early June (Figure App A2-1). The Georges Bank component was conducted in July to early August, but now is conducted in June. Examination of the effects of this timing difference showed no significant effect on Georges Bank, but substantial effects in the Mid-Atlantic. Sea scallops typically lose up to 20% of their meat weight around the time that they spawn, and Mid-Atlantic scallops often have a strong spring spawn in April to early May. Thus, these scallops are recovering from their spawn during the time of the survey, and it is to be expected that meat weights at size will be less in May than June or July. To model this, the covariate "mday", the number of days after April 30, was included in the Mid-Atlantic model, together with a non-linear mday^2 term and an interaction of mday with shell height:

$$W = \exp(a_0 + a_1\mathrm{mday} + a_2\mathrm{mday}^2 + (b_0 + b_1\mathrm{mday})\ln(H)). \qquad \text{(App A2-2)}$$

As expected, meat weights at size in the Mid-Atlantic are increasing with mday at least through mid-June (Figure App A2-2). The stock assessment models employ a single (regional) shell height to meat weight relationship, although, as discussed below, it is modified by annual anomalies. For the purposes of this assessment, the Working Group chose to use the Mid-Atlantic relationship with mday= 21, i.e., the predicted sh/mw relatioship on May

21. As discussed above, there was no significant seasonality for Georges Bank during the surveyed time of June-August, so the basic relationship was simply estimated using Equation (App A2-1); see Table App A2-1 and Figure App A2-3. Estimates based on limited samples of D scallops are also presented.

Covariates such as depth, latitude, and whether a sample is in a open or closed area ("clop") can substantially affect predicted meat weights at shell height (Hennen and Hart 2012). This information is used to estimate biomass from a survey. Additionally, it is used to compute shell height to meat weight anomalies discussed in the next section.

It is also useful, particularly for use in the SAMS forecasting model, to estimate shell height to meat weight relationships by area (Table refRegSHMWTable). On Georges Bank, only regional differences in the intercept reduced AIC, so the slope was the same for all areas. For the Mid-Atlantic, regional differences in both slope and intercepts reduced AIC; these estimates were based on collection date of May 21.

Commercial shell height/meat weight data

Shell height to meat weight relationships on commercial vessels can vary from the survey relationship for two primary reasons. First, as discussed above, meat weights vary seasonally, and the fishery is conducted year-round, not just when the survey occurs. Secondly, scientist on the survey carefully shuck the scallops, taking all the meat. Scallopers shuck for speed, often leaving a small portion of the meat in the shell.

For these reasons, commercial meat weights are monitored by fishery observers. Once a watch, the observers take about 100 scallops retained for landing, and measure their shell heights. The scallops are then given to a fisherman to shuck, and the scallop meats are put into a graduated cylinder to get a volumetric measure. The volume is converted to a weight using a specific weight of 1.05 (Caddy and Radley-Walters 1972, Smolowitz et al. 1989). Only samples with between 50 and 200 meats were used. Additionally, a few outliers, where the meat weight anomalies (see below) were less than -0.67 or greater than 0.67 were removed. All together, about 3% of the data were removed for one of these reasons; the analysis below is based on the cleaned data set, which comprises about 97% of the original data.

The shell height measurements are used to compute the predicted meat weight, using the estimates from survey data with covariates (Table App A2-2). The data are then aggregated by month and region and "clop" (Georges Bank and Mid-Atlantic, open or rotational access area), to compute an empirical monthly meat weight anomaly A_m (Hennen and Hart 2012):

$$A = \frac{\text{predicted} - \text{observed}}{\text{predicted}}. \qquad \text{(App A2-3)}$$

Computed empirical meat weight anomalies show both a strong seasonal cycle and a long-term trend towards larger meat weights at size since 1994, when this sampling began (Figure App A2-4). These trends is similar to the trends found in growth (see Appendix A1), and likely have been caused by similar factors. Besides potential environ mental signals (such as warming temperatures, Cooley et al. 2015), fishing effects have likely contributed to these trends. Scallopers tend to target scallops with larger meats at size, which correlates to faster growth as well, leaving a disproportionately high percentage of scallops with smaller meats at size in the population. Such effects would be expected to increase at higher fishing mortality rates, which at least in part, explains the increasing trends, as fishing effort and mortality has generally declined since early 1990s.

To further explore the cycles and trends in these data, a generalized additive model (GAM) was developed for predicting the meat weight anomaly:

$$\text{MWanomaly} \sim s(\text{Year,Month}) + \text{nscallops} + \text{Clop}, \qquad \text{(App A2-4)}$$

where s is a two dimensional smoother on Year and Month, nscallops in the number of meats in the observer's sample, and Clop is 1 if the sample was taken in a rotational access area, and 0 if it was taken in an open area. The term "nscallops" was used since there is a slight tendency for the anomalies to decline with the number of scallops in the sample. This might occur if the probability that some of the measured scallops didn't get into the graduated cylinder increases with number of scallops measured. For the purposes of the predictions given below, I used nscallops = 100, which is both what is specified in the protocol, and also the most common number measured. Using the two dimensional smoother rather than separate smoothers on Year and Month substantially improved the fit and reduced AIC. This indicates that there was a significant Year/Month interaction, i.e., the annual meat weight cycle significantly varied by year (Figure App A2-5).

The predicted GAM anomalies (Figure App A2-6) also show strong seasonality and trends towards larger meats with time. Weighted (by landings) averages over each year from these anomalies were calculated and then used to adjust the CASA survey shell height to meat weight relationships by year. Yearly meat weight anomalies prior to 1994 were calculated as the weighted (by landings) average monthly anomaly from 1994-98.

Mean monthly anomalies from the GAM show the strong seasonal cycle, with meats varying by 20% or more in both regions (Figure App A2-7) over an average year. Year effects show strong trends, as discussed previously (Figure App A2-8).

Table App A2-1: Basic shell height to meat weight estimates.

Region	Estimate	Std. Error
Georges Bank All*		
a (intercept)	-9.67	0.09
b (Slope)	2.732	0.019
Georges Bank Open		
a (intercept)	-10.39	0.13
b (Slope)	2.87	0.03
Georges Bank Closed*		
a (intercept)	-10.3	0.13
b (Slope)	2.86	0.05
DSENLS only		
a (intercept)	-11.84	0.69
b (Slope)	3.167	0.15
Mid-Atlantic		
a (intercept)	-9.417	0.13
b (Slope)	2.435	0.022
mday	-0.0235	0.0027
mday^2	-9.20e-5	7.2e-6
Slope:mday	0.0068	0.0006

*Without DSENLS scallops

Table App A2-2: Shell height to meat weight relationships with covariates. "Clop" is 1 if a sample is in a closed or rotational area, and 0 otherwise. Only covariates that reduced AIC are included.

Georges Bank

Variable	Estimate	Std. Error
Intercept	-6.69	0.38
ln sh	2.878	0.027
Depth	-0.0073	0.0003
Lat	-0.073	0.009
Clop	1.28	0.17
ln sh:Clop	-0.25	0.04

Mid-Atlantic

Variable	Estimate	Std. Error
Intercept	-9.48	0.24
ln sh	2.51	0.026
mday	-0.0083	0.0086
mday^2	-0.000134	0.000005
Depth	-0.0033	0.00045
Lat	0.021	0.005
Clop	-0.031	0.008
ln sh:mday	0.00525	0.0005
mday:Depth	-6.5e-5	9.6e-6

Table App A2-3: Regional shell height to meat weight coefficients. Georges Bank regions are Closed Area I Access, Closed Area I No Access, Closed Area II South (S of 41.5° N), Closed Area II North, Closed Area II Extension, Great South Channel, Northern Flank, Nantucket Lightship East (east of 69.5° W), Nantucket Lightship West, Southern Flank. Mid-Atlantic regions are Delmarva (including Virginia), Elephant Trunk, Hudson Canyon South, Inshore, Long Island (including Block Island), New York Bight.

Georges Bank

Region	Intercept	SE	Slope	SE
C1ACC	-9.60	0.10	2.73	0.02
C1NA	-9.81	0.10	2.73	0.02
C2S	-9.65	0.10	2.73	0.02
C2N	-9.68	0.10	2.73	0.02
C2Ext	-9.90	0.10	2.73	0.02
GSC	-9.65	0.10	2.73	0.02
NF	-9.69	0.10	2.73	0.02
NLSE	-9.52	0.10	2.73	0.02
NLSW	-9.64	0.10	2.73	0.02
SF	-9.73	0.10	2.73	0.02

Mid-Atlantic

Region	Intercept	SE	Slope	SE
DMV	-9.58	0.06	2.41	0.14
ET	-9.52	0.06	2.72	0.14
HCS	-9.50	0.06	2.69	0.14
Inshore	-9.36	0.08	2.60	0.20
LI	-9.44	0.06	2.48	0.14
NYB	-9.44	0.06	2.63	0.14

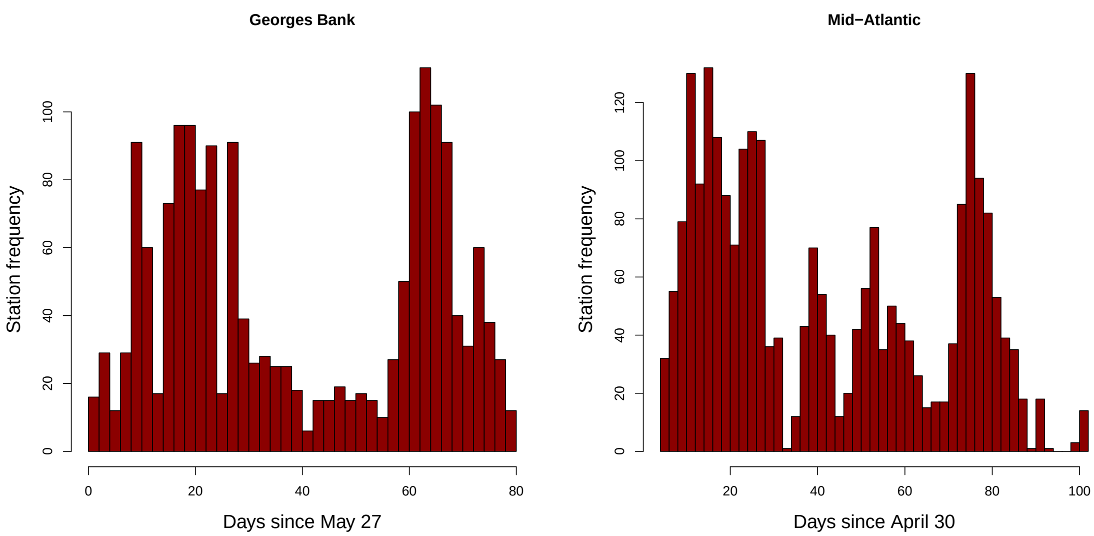


Figure App A2-1: Histograms of the timing of meat weight samples in Georges Bank (left) and the Mid-Atlantic (right).

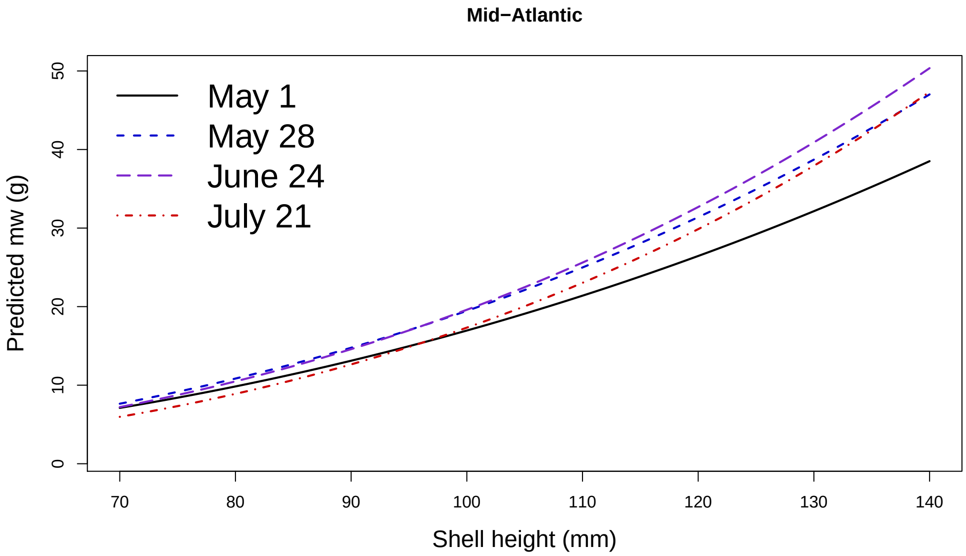
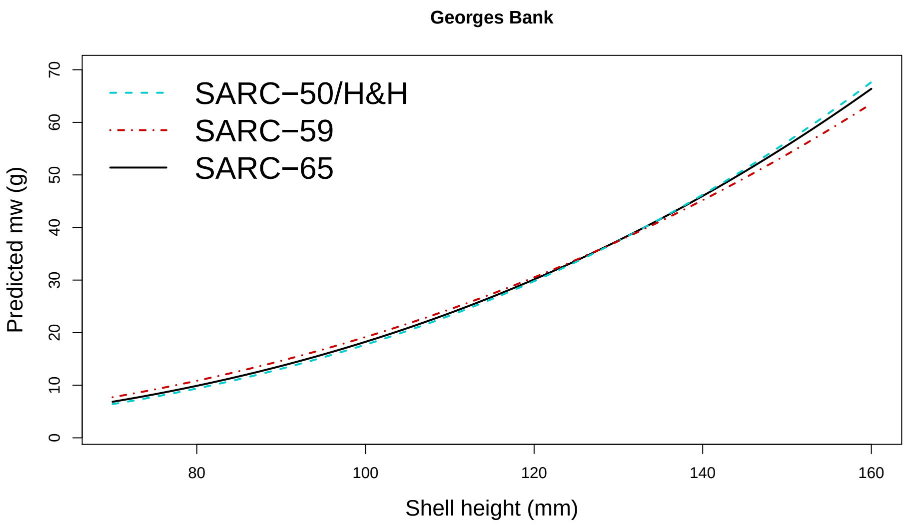


Figure App A2-2: Mid-Atlantic shell height meat weight relationships on various dates as estimated by survey data.



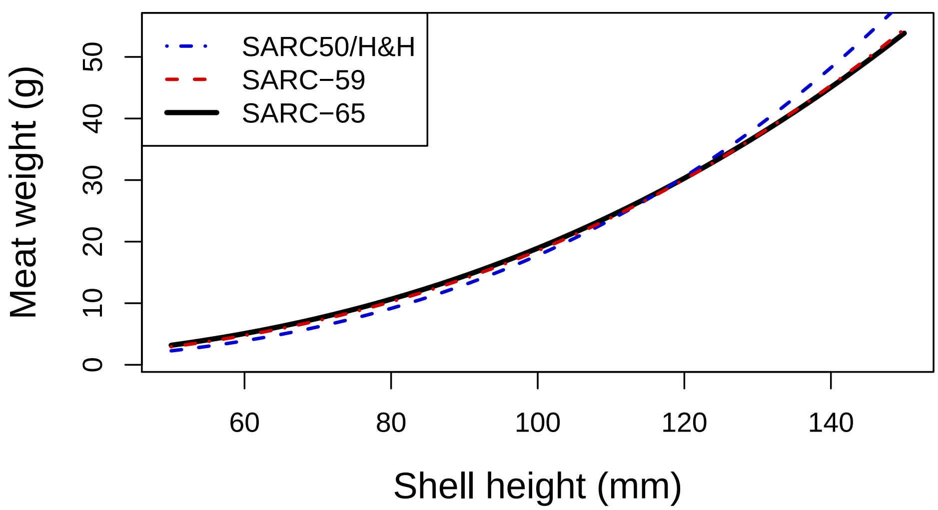


Figure App A2-3: Basic Georges Bank (above) and Mid-Atlantic (below) shell height to meat weight relationships, compared to those from the last two benchmark assessments.

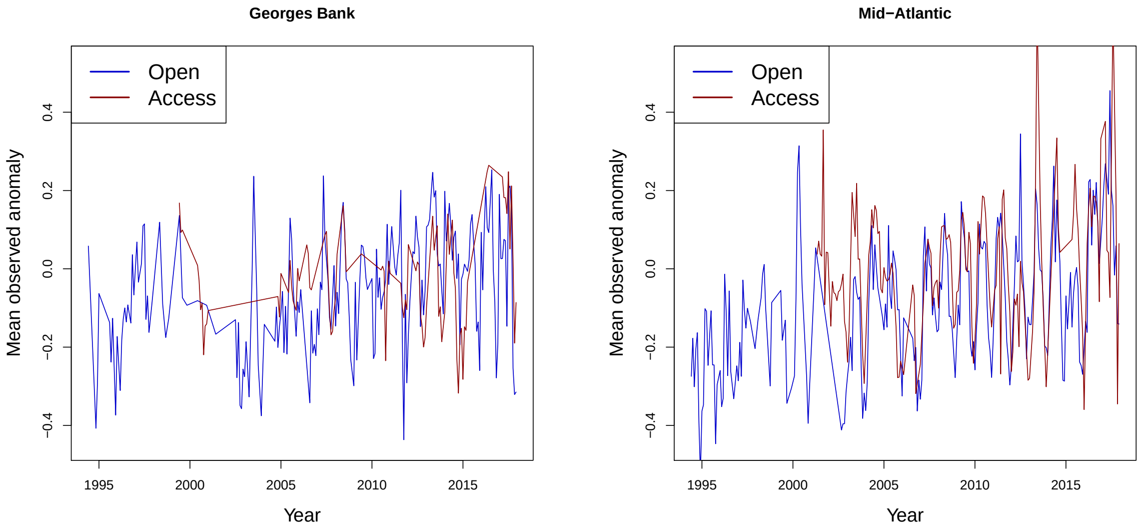


Figure App A2-4: Empirical meat weight anomalies, computed for each month and year, using equation (App A2-3) for Georges Bank (left) and Mid-Atlantic (right) open (blue) and access areas (black).

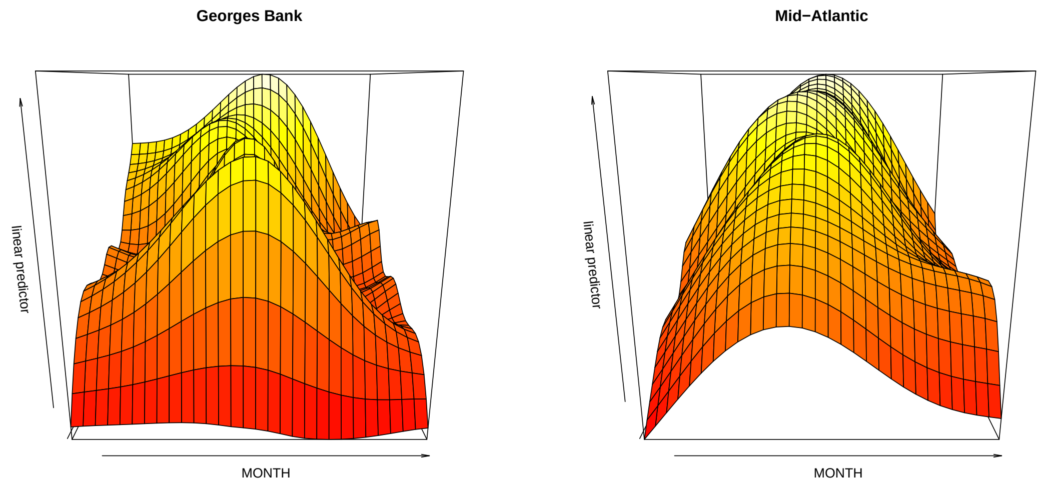


Figure App A2-5: GAM predictions as a function of month and year.

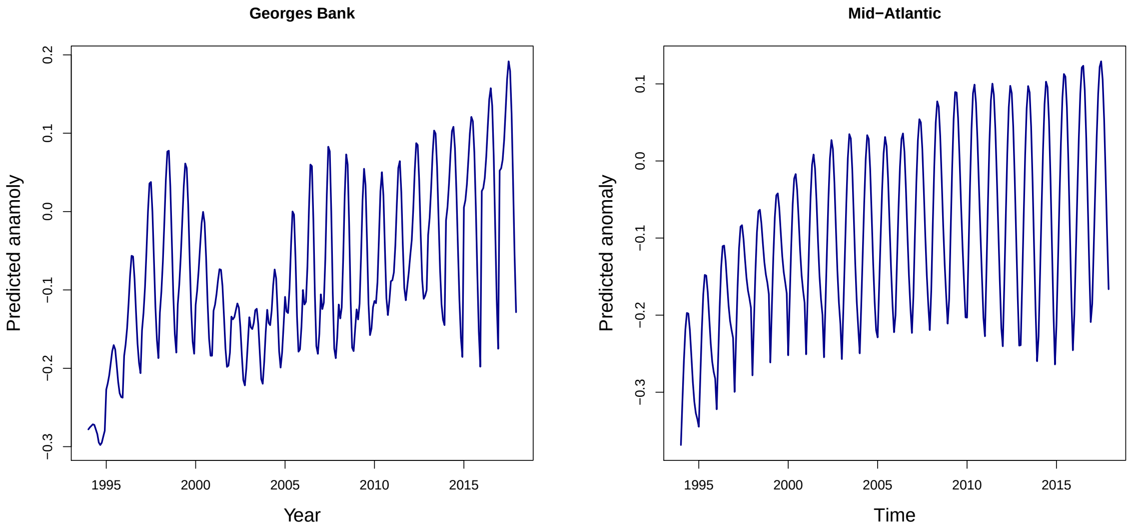


Figure App A2-6: Predicted meat weight anomalies from the GAM for Georgers Bank (left) and Mid-Atlantic (right) open areas

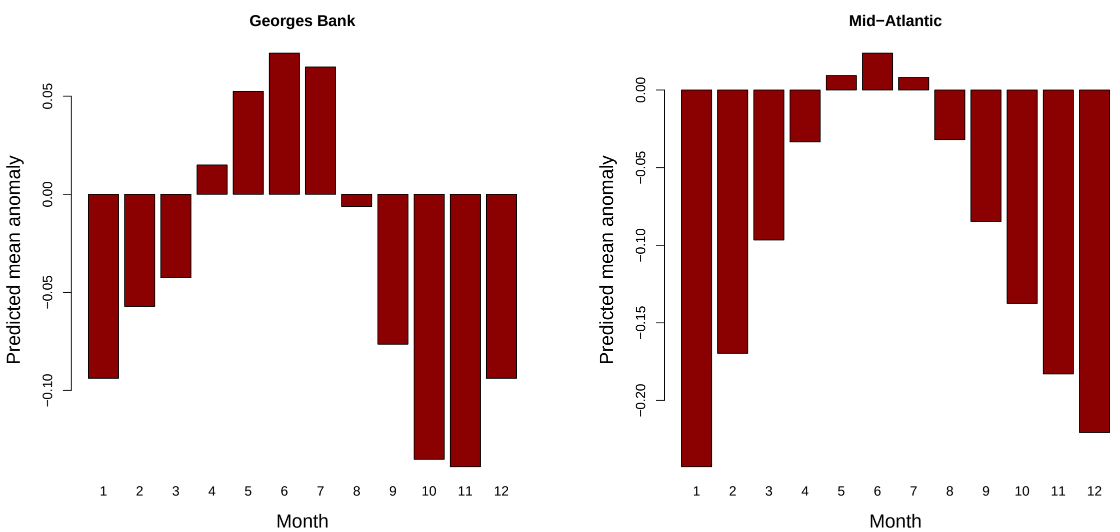


Figure App A2-7: Mean monthly meat weight anomalies on Georges Bank (left) and Mid-Atlantic (right) open areas from GAM predictions.

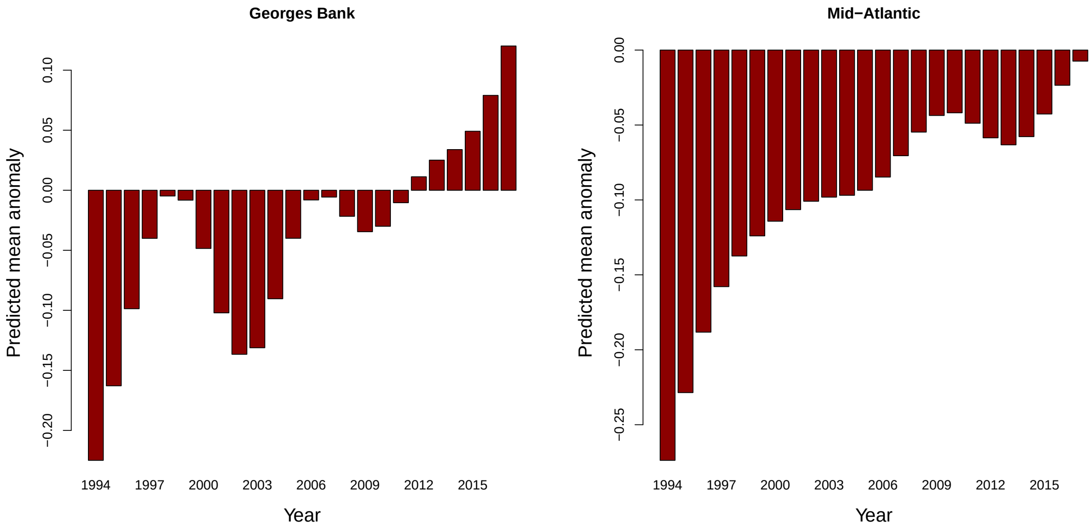


Figure App A2-8: Mean annual meat weight anomalies on Georges Bank (left) and Mid-Atlantic (right) open areas, from GAM predictions.

Appendix A3: An Overview of the Atlantic Sea Scallop Resource in the Gulf of Maine

Jonathon Peros (New England Fishery Management Council, Newburyport, MA)

Samuel Asci (New England Fishery Management Council, Newburyport, MA)

N. David Bethoney (School for Marine Science and Technology, New Bedford, MA)

Jason Clermont (Coonamesset Farm Foundation, Falmouth, MA)

Benjamin Galuardi (NOAA Fisheries – Greater Atlantic Regional Office, Gloucester, MA)

Dvora Hart (NOAA Fisheries – Northeast Fisheries Science Center, Woods Hole, MA)

Kevin Kelly (Maine Department of Marine Resources, W. Boothbay Harbor, ME)

Burton Shank (NOAA Fisheries – Northeast Fisheries Science Center, Woods Hole, MA)

Tyler Staples (NOAA Fisheries – Northeast Fisheries Science Center Observer Program, Falmouth, MA)

Michael Torre (University of Maine, Orono, ME)

1.0 INTRODUCTION

The Atlantic sea scallop (*Placopecten magellanicus*) ranges Cape Hatteras to the Gulf of St. Lawrence. The scallop fishery is primarily prosecuted in concentrated areas in and around Georges Bank and off the Mid-Atlantic coast, in waters extending from the near-coast out to the edge of the continental shelf. Atlantic sea scallops occur primarily in depths less than 110 meters on sand, gravel, shells, and cobble substrates (Hart and Chute 2004). While the majority of the Atlantic sea scallop resource is found on Georges Bank and in the Mid-Atlantic, sea scallops also occur in the Gulf of Maine (GOM) in both state and federal waters. The federal scallop resource in the GOM is managed by the New England Fishery Management Council and NOAA Fisheries.

1.1 RELEVANT TERMS OF REFERENCE (SCALLOPS - A3)

This appendix is intended to address Term of Reference (TOR) A3 for SAW/SARC 65:

> *Summarize existing data, and characterize trends if possible, and define what data should be collected from the Gulf of Maine area to describe the condition of the resource. If possible, provide a basis for developing catch advice for this area.*

1.2 SUMMARY OF EXISTING DATA

Table 1 summarizes the existing data streams available for the Gulf of Maine. Examples and/or analyses of each data set are shown in the corresponding section listed below.

Table 1 - Summary of existing data for Atlantic Sea Scallops in Gulf of Maine

Existing Data	Section	Details
Dedicated scallop surveys	2.1 & 2.2	There have been periodic dredge and optical scallop surveys in portions of the Gulf of Maine. Section 2.1 provides an overview of dedicated scallop surveys, while Section 2.2 covers the results of dedicated survey work in 2016 and 2017.
Scallop catch in regional trawl surveys	2.3	Records of scallop catch in several regional trawl surveys (state, federal, shrimp). Indices of spring and fall NMFS bottom trawl surveys provided in Section 2.3.1. Spatial/temporal distribution of scallop catch in regional trawl surveys is shown in Section 2.3.2.
Observer data	3.2	Low number of observed hauls in the Gulf of Maine over the last 10 years. Majority of data collected in Stellwagen Bank region. Section 3.2.1 summarizes data from observed LA trips in the NGOM in 2017.
Vessel monitoring system data	3.1	Multiple uses: 1) Spatial extent of fishing activity in the region; 2) Required daily reports of catch. All vessels in the federal scallop fishery must have an active VMS unit to participate. Units ping vessel's location every 30 minutes and can be used to transmit fishing reports.
Vessel trip report data	3.3.2.2	Reports completed by vessels prior to landing. VTRs provide estimates of fishing location and landings. Estimated landing by statistical reporting area from VTR reports are provided for reference.
Landings data	3.3	Data available from both state and federal fisheries. Dealer reports should represent a census of landings.

Figure 1. Major physiographic features of the Gulf of Maine relative to the territorial waters boundary of the US and Canada (EEZ, red dotted line).

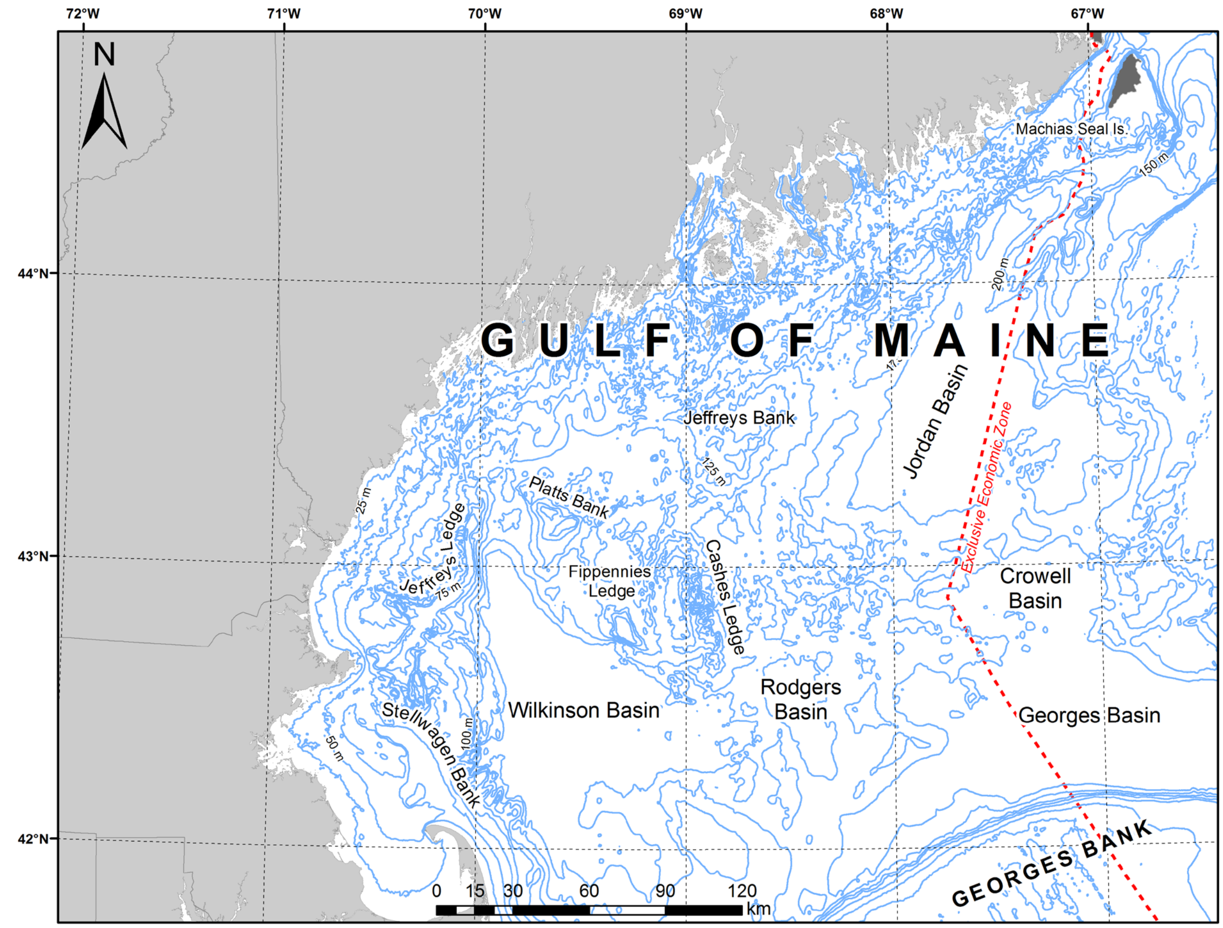


Figure 2. Management boundaries in the Gulf of Maine region including boundaries of the Northern Gulf of Maine Management Area (NGOM, in blue), Habitat Management Areas, Dedicated Habitat Research Areas, Groundfish Closures, VMS Demarcation line, State Waters, and Statistical Reporting Areas (grey lines and labels).

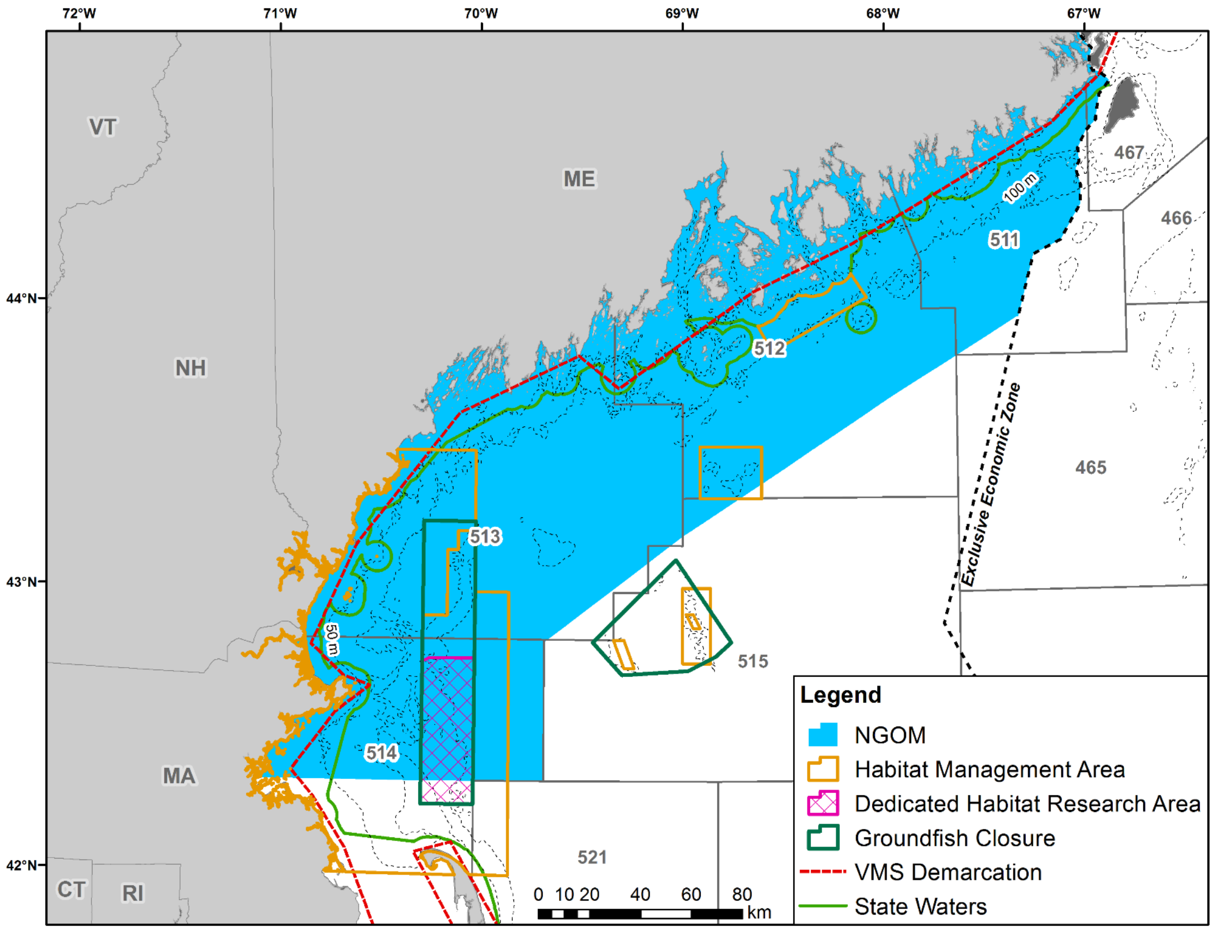
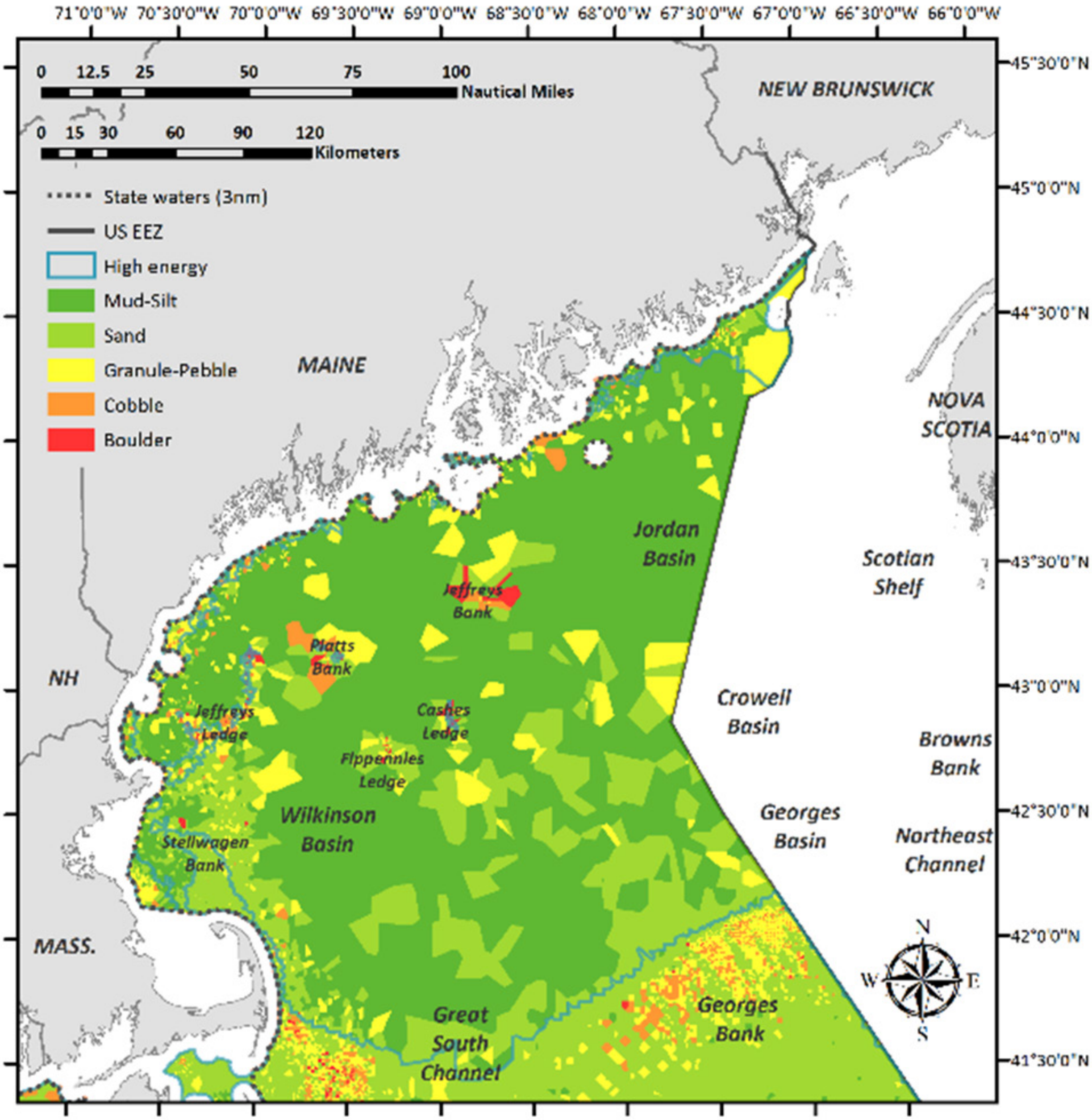


Figure 3 -Sedimentary features of the Gulf of Maine (NEFMC, 2016).



2.0 RESOURCE SURVEYS

Scallops are captured in both dedicated resource surveys and incidentally in trawl surveys. This section describes the history of both dedicated survey work, as well as the available information from trawl surveys.

2.1 DEDICATED SCALLOP SURVEYS IN THE GULF OF MAINE

Truesdell et al (2014) noted in SAW59 that there was a limited amount of fishery independent data available for the Gulf of Maine Region (). Since the last scallop assessment additional dredge and optical surveys have been completed since the last benchmark, however the region remains data-limited relative to Georges Bank and the Mid-Atlantic. Some of the earliest dedicated scallop surveys of the region took place in the early 1980's (Serchuk, 1983). Later, dredge surveys focusing on the Northern Gulf of Maine Management Area were conducted by Maine Department of Marine Resources/University of Maine in 2002, 2006, 2010, 2012, and 2016 (Table 2). Coverage of this survey has varied each year and recently has focused mostly on areas with known aggregations of scallops commonly targeted in the NGOM. Additional drop camera surveys were conducted by the School for Marine Science and Technology (University of Massachusetts Dartmouth) in 2009, 2010, 2011, 2013, 2014, and 2015 (Table 2; Stokesbury et al. 2010, Bethoney et al. 2016, Asci et al. 2018). SMAST drop camera surveys covered areas that have been closed to fishing (Jeffreys Ledge, Cashes Ledge, and Fippennies Ledge) and covered one area open to fishing in the NGOM (Platts Bank).

Data collected from these surveys have been useful in estimating localized scallop abundance, size distribution, and exploitable biomass; however, the relatively small proportion of the Gulf of Maine actually surveyed and lack of annual survey effort suggests our knowledge of the Gulf of Maine scallop population is highly uncertain.

Table 2 - Northern Gulf of Maine scallop surveys, 2009-2017

	Northern Gulf of Maine (NGOM) scallop surveys, 2009-2017											
	Year	2009		2010	2011	2012	2013	2014	2015	2016	2017	
	Month(s)	June-July	August	August	August	May-June	August	August	July-August	May-June	July	July
	Organization	ME DMR/ UMaine	SMAST	SMAST	SMAST	ME DMR/ UMaine	SMAST	SMAST	SMAST	ME DMR/ UMaine	SMAST	CFF
	Method	dredge	drop camera	drop camera	drop camera	dredge	drop camera	drop camera	drop camera	dredge	drop camera	HabCam/ dredge
Area(s) surveyed	**northern Stellwagen**	X				X				X	X	X
	parts of Stellwagen		X				X					
	southern Jeffreys Ledge	X				X	X	X	X	X		X
	Ipswich Bay	X				X				X		
	northern Jeffreys Ledge(closed)		X	X	X		X	X	X			
	Platts Bank	X	X	X	X	X	X	X	X	X		
	Fippennies Ledge (closed)		X	X	X	X	X	X	X	X		
	Cashes Ledge (closed)		X	X	X		X	X	X			
	Jeffreys Bank (closed)		X	X								
	Mt. Desert/ Isle au Haut	X				X				X		
	eastern ME	X				X				X		

2.2 RECENT DEDICATED SCALLOP SURVEYS

The following section briefly summarizes findings from 2016 and 2017 scallop surveys in the Gulf of Maine. The specific areas covered by these efforts are shown in Table 2.

2.2.1 2017 SMAST Drop Camera Survey

The SMAST drop camera was used to survey the portion of Stellwagen Bank that was fished by Limited Access, LAGC IFQ, and LAGC NGOM vessels in FY2017 (Figure 4). The survey stations were fixed on a high-resolution (1.5 km^2) grid and were sampled between July 7th and July 13th, 2017.

Survey findings suggest very few smaller scallops (< 75 mm) were in the area and that most scallops observed were approximately 100 mm in length (Figure 2). Density was estimated to be roughly 0.1 +/- 0.02 scallops m^{-2}, which translates to a harvestable density similar to what would be seen on Georges Bank. Total biomass was estimated at 356 mt and exploitable biomass was estimated at 228 mt. SARC 50 SH:MW parameter estimates were used when calculating total and exploitable biomass.

Figure 4. Observed scallop density m^{-2} at survey stations completed by the SMAST drop camera in 2017.

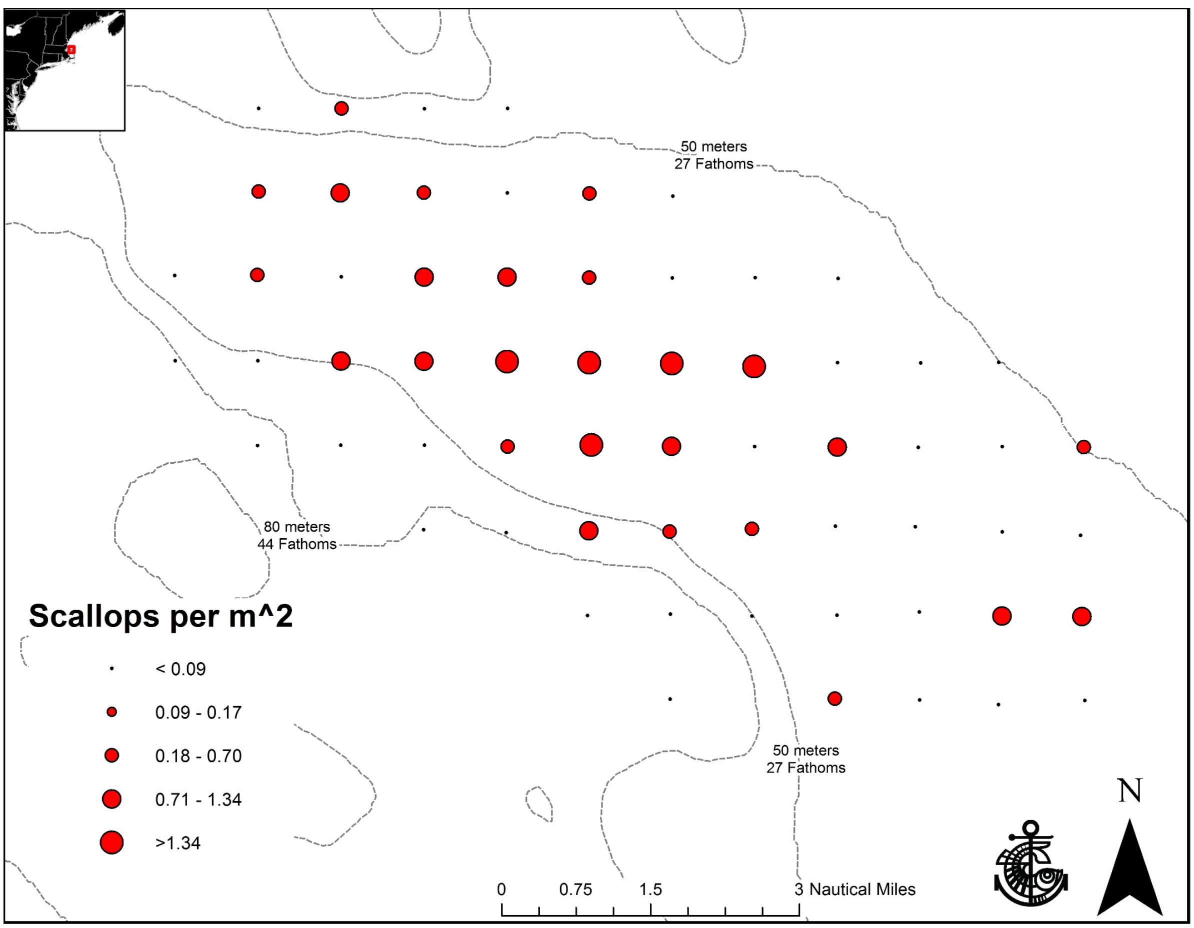
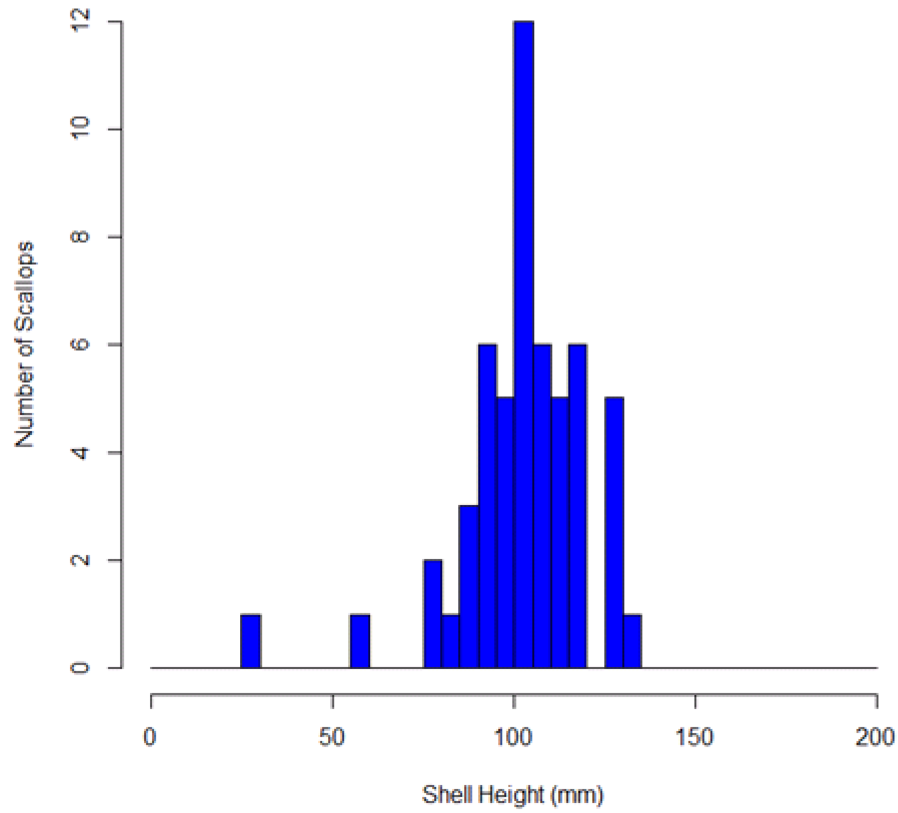


Figure 2. Shell height distribution of scallops in the Stellwagen Bank priority area based on digital still camera images.



2.2.2 2017 RSA HabCam v3 Survey, Northern Gulf of Maine

Between July 8th and July 9th, 2017, the Coonamessett Farm Foundation surveyed portions of the NGOM that were fished in FY2016 (southern Jeffreys Ledge, see Figure 6) and FY2017 (Stellwagen Bank, see Figure 8) using HabCam v3 and a survey dredge. HabCam tracks covered approximately 67 NM on Stellwagen Bank and 22 NMon southern Jeffreys Ledge. Six dredge tows were conducted on Stellwagen to collect biological samples and length frequency data. Due to the high density of fixed gear on Jeffreys Ledge, it was not possible to complete dredge tows in this area.

Both HabCam and dredge data indicated very few recruits (<= 75 mm) on Stellwagen Bank. Scallops > 75 mm appeared to be spread out across the top of Stellwagen (Figure 7). There was some evidence of recruits in the southwest part of Jeffreys Ledge, and scallops > 75 mm seemed to be distributed across the survey area (Figure 5). The smaller scallops on Jeffreys Ledge were observed with notable densities of sea stars. Total biomass on Stellwagen Bank was estimated to be roughly 459 mt and biomass on southern Jeffreys Ledge was estimated to be roughly 152 mt.

Figure 5. Scallop size distribution observed by the CFF HabCam survey of Jeffreys Ledge in 2017.

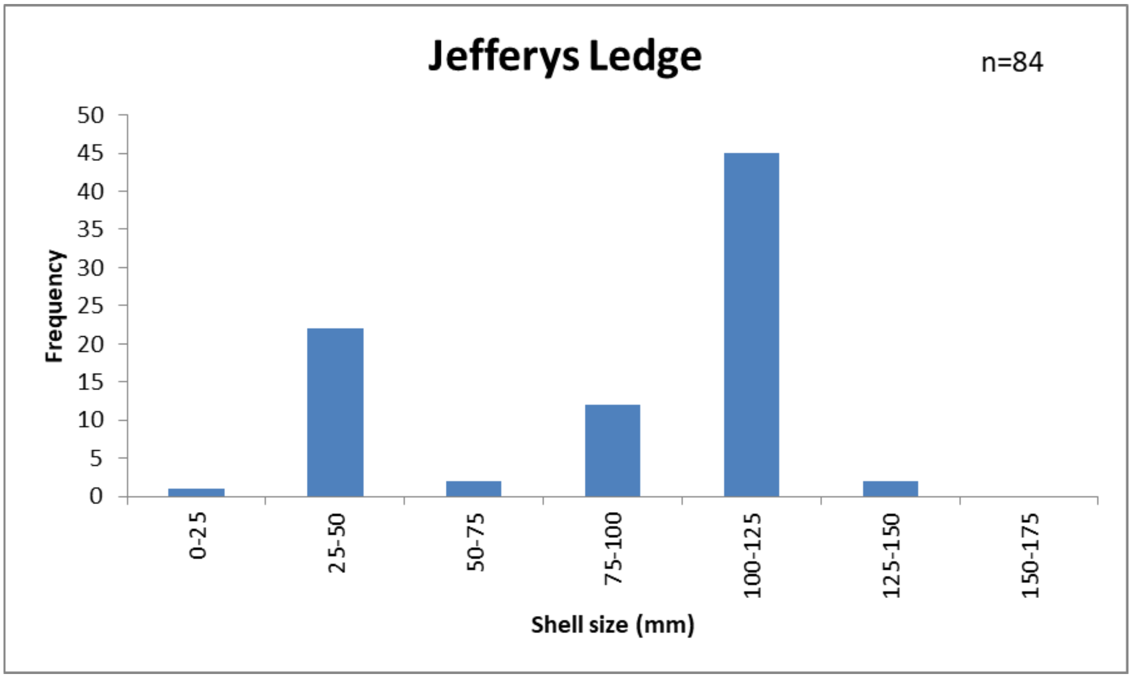


Figure 6. Biomass heatmap from the CFF HabCam survey of Jeffreys Ledge in 2017.

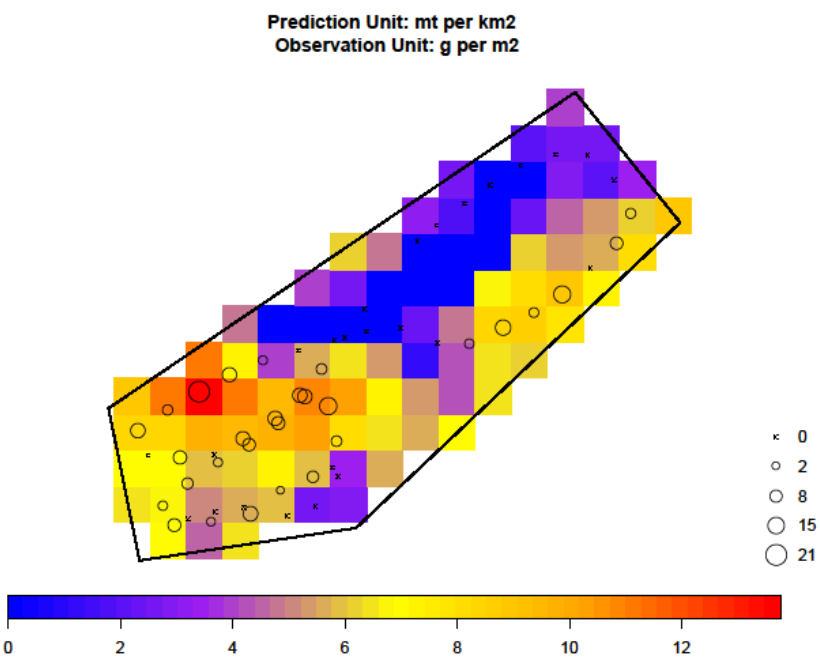


Figure 7. Scallop size distribution observed by the CFF HabCam survey of Stellwagen Bank in 2017.

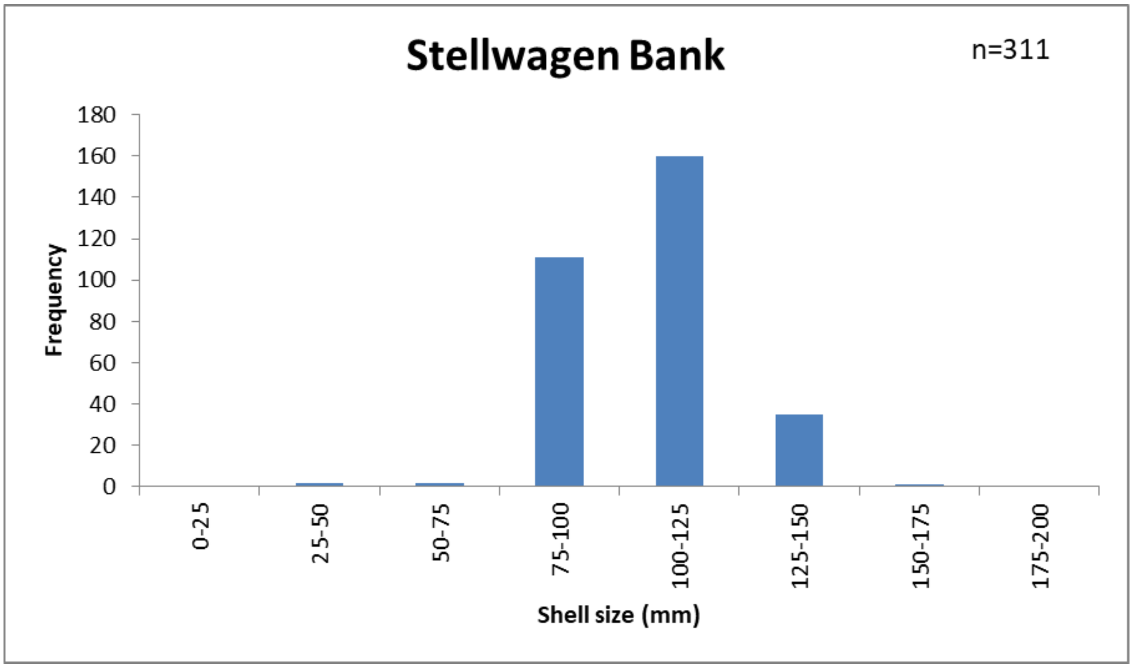
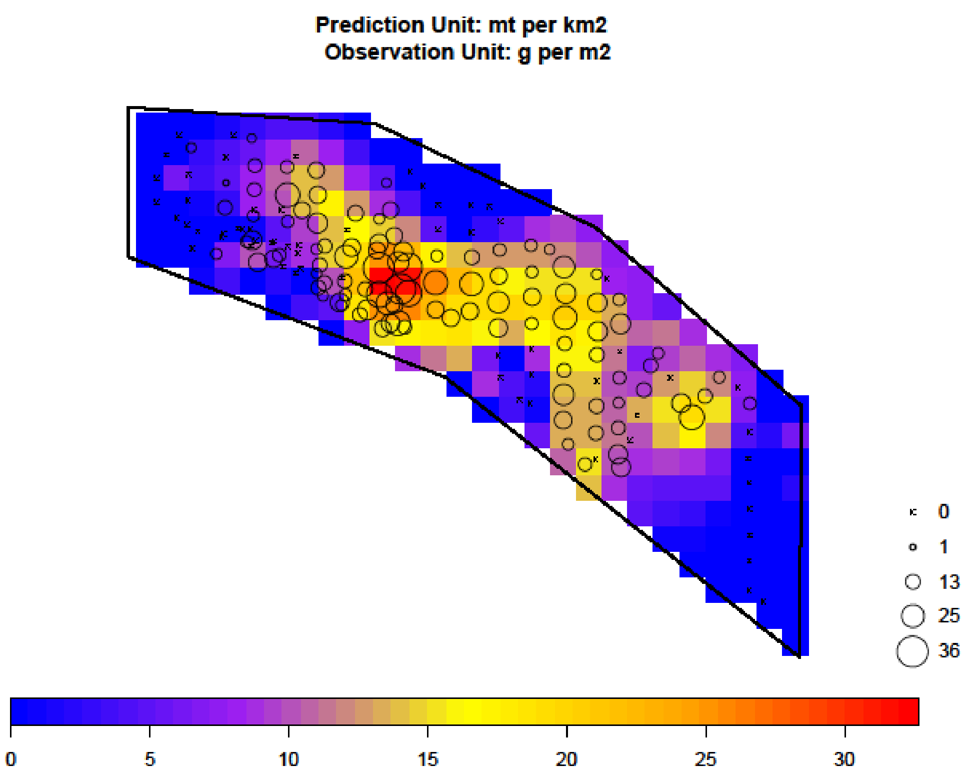


Figure 8. Biomass heatmap from the CFF HabCam survey of Stellwagen Bank in 2017.



2.2.3 2016 UMaine/Maine DMR Dredge Survey of Northern Gulf of Maine

In order to effectively allocate survey effort to areas with high scallop biomass, the southern three survey areas (Ipswich Bay, Jeffreys Ledge and Northern Stellwagen Bank; the areas of highest fishing activity within the NGOM) were subdivided into high, medium, and low density sub-strata. The delineation of these substrata was informed by fishermen input, VTR, and VMS, as well as previous survey data from 2009 and 2012. Tows were allocated among these sub-strata according to the Neyman's allocation, which ensures that sampling effort is allocated to areas of high variance to increase precision of abundance indices and refine the resulting biomass estimates:

$$n_h = n\frac{W_h S_h}{\sum_{h=1}^{H} W_h S_h}$$

where n is the total number of sampling stations for the survey area, H is the total number of strata, W_h is the proportion of stratum h area over the survey area, and S_h is the estimated standard deviation of historical data in stratum h.

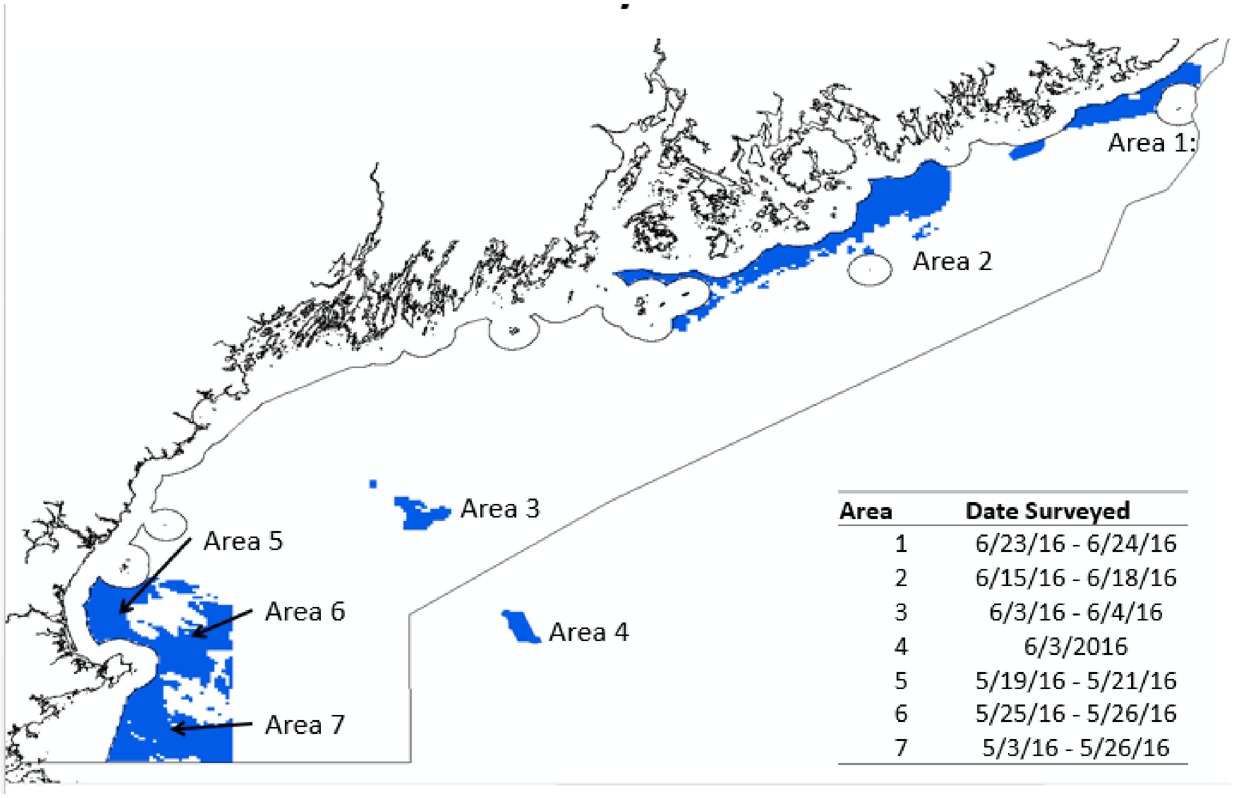
The highest scallop biomass observed in the 2016 ME DMR/UMaine dredge survey was on Stellwagen Bank (Figure 10, Figure 11). Smaller concentrations of biomass (>101mm) were seen in Machias/Seal Island, and on Platts Bank. Total biomass in the NGOM, assuming a dredge efficiency of 0.4 and selecting a conservative value (q0.10 on the bootstrapped distribution), was estimated at 1.75 million lbs. (Table 3). Using an exploitation rate of 0.2, the removable biomass was calculated to be approximately 350,000 lbs. (Table 3). This information was used to inform a TAC for the NGOM for FY 2017.

Biomass estimates were substantially higher in 2016 (**Table 3**) than they were in 2012 (approximately 416,000 lbs). Biomass estimates were presented to managers in 2016 using an exploitation rate of 0.38 and an 0.26. Managers requested a new model run using an exploitation rate equal to 0.2, with estimates at the q0.25 and q0.10.

Table 3 - Biomass estimates from 2016 NGOM survey (F=0.2, Dredge Efficiency=0.4).

Exploitation Rate = 0.20 **Dredge Efficiency = 0.40**	**q0.05**	**q0.10**	**q0.15**	**q0.20**	**q0.25**	**Mean**
Biomass Estimate (MT)	657	795	932	1018	1090	1651
TAC(MT)	131	159	186	204	218	330
Biomass Estimate (lbs)	1,447,797	1,751,822	2,055,240	2,244,263	2,402,140	3,640,385
TAC(lbs)	289,559	350,364	411,048	448,853	480,428	728,077

Figure 9 - 2016 ME DMR NGOM Survey Areas.



Area	Date Surveyed
1	6/23/16 - 6/24/16
2	6/15/16 - 6/18/16
3	6/3/16 - 6/4/16
4	6/3/2016
5	5/19/16 - 5/21/16
6	5/25/16 - 5/26/16
7	5/3/16 - 5/26/16

Figure 10 - 2016 ME DMR NGOM survey - estimates of harvestable biomass from each survey area.

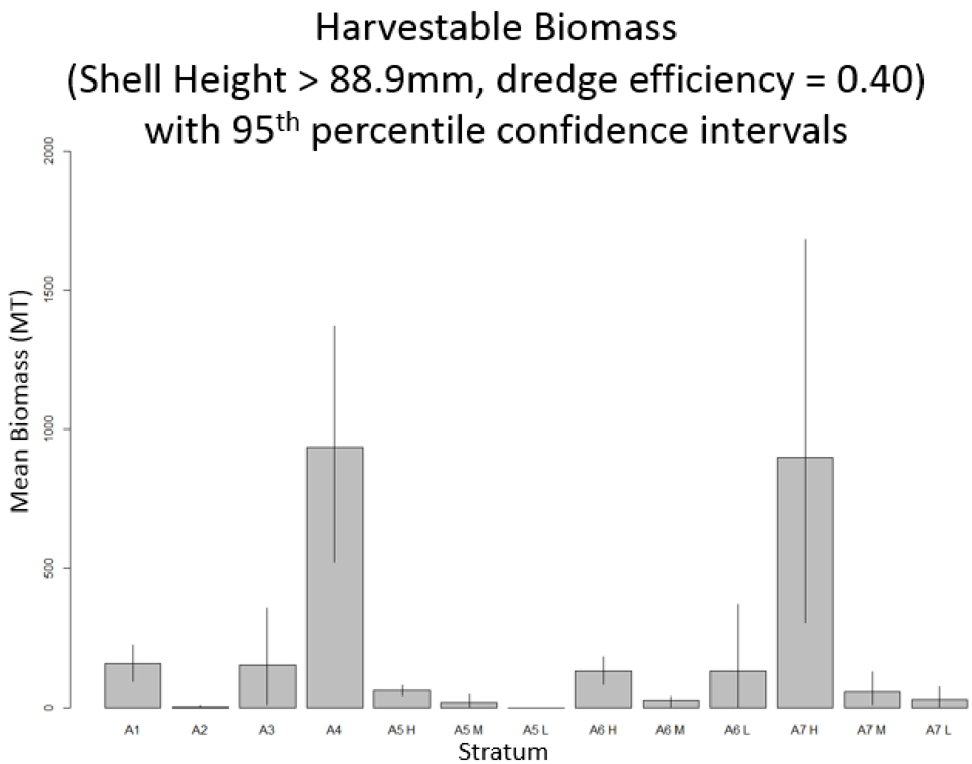
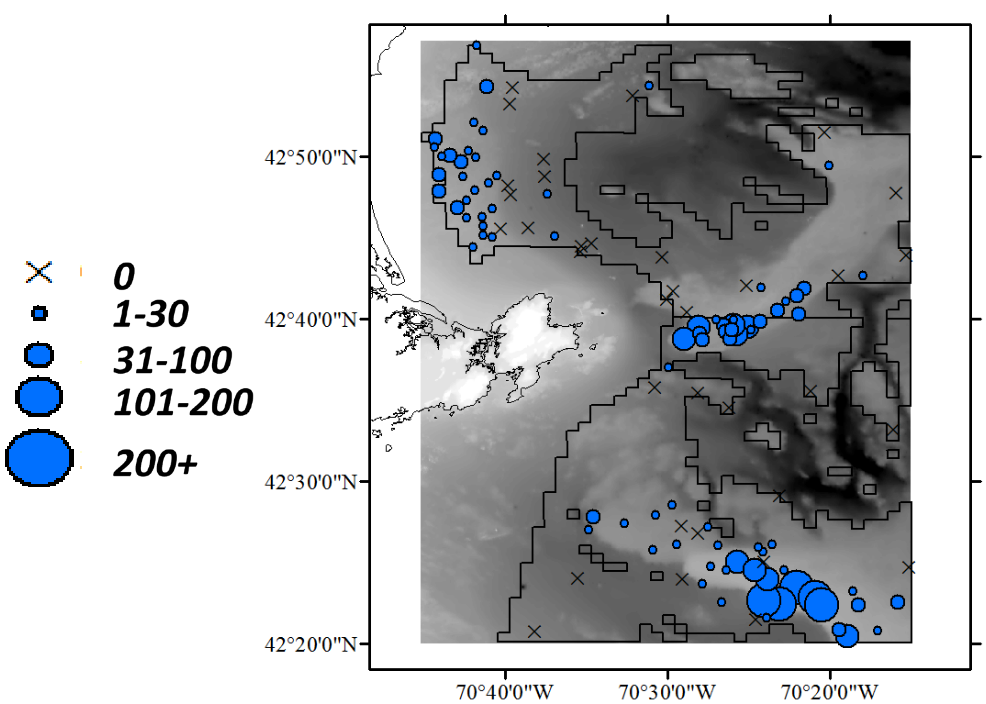


Figure 11 - 2016 ME DMR NGOM survey – Distribution of scallop abundance on Ipswich Bay, Jeffreys Ledge and Stellwagen Bank.



2.3 OVERVIEW OF SCALLOP CATCH IN REGIONAL TRAWL SURVEYS

In addition to dedicated dredge and optical surveys, Atlantic sea scallops are captured in several regional trawl surveys. To summarize existing data, a time series of scallop catch in Gulf of Maine strata was compiled using the NEFSC bottom trawl survey (Section 2.3.1). The spatial distribution of scallops in the Gulf of Maine was examined in Section 2.3.2 using data from the NEFSC bottom trawl survey, the NEFSC shrimp trawl survey, the Massachusetts Division of Marine Fisheries trawl survey, and the New Hampshire/Maine Department of Marine Resources inshore trawl survey.

There are several important caveats to be aware of when reviewing the following figures. First, note that these trawl surveys were not targeting popular scallop grounds or areas with known scallop habitat. Second, trawl survey gear has different selectivity and efficiency compared to dredge survey gear, meaning scallop catches in trawl surveys are not directly comparable to scallop catches in dredge surveys. It is worth noting that the field methods and gear configuration varies between each of these trawl surveys and also that these trawl surveys have evolved over the roughly 50-year time series being considered.

2.3.1 Scallop catch in NEFSC bottom trawl survey

Indices of scallop catch from the spring and fall Northeast Fisheries Science Center (NEFSC) bottom trawl surveys in the Gulf of Maine are shown in Figure 12 and . Note that there is a break in the time series between 2008 and 2009 when the survey was shifted from the R/V Albatross to the R/V Bigelow. The survey indices appear to detect recent recruitment events in the Gulf of Maine that supported increases in fishing activity and removals from the region in 2015, 2016, and 2017 (see also Figure 20 and Figure 22).

Figure 12 - GOM Survey Indices (40mm+) from NMFS Bottom Trawl Survey (1978 – 2017)

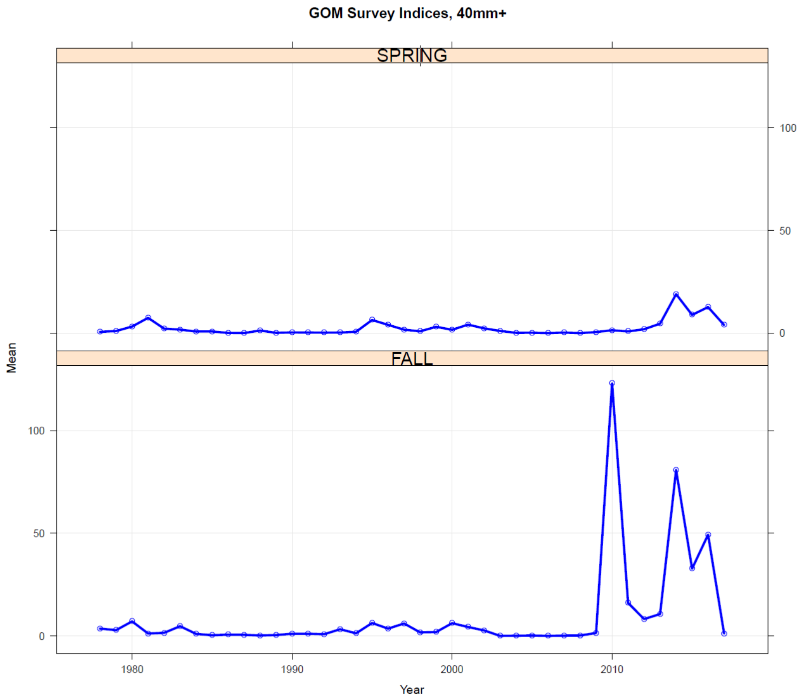


Figure 13 - GOM Survey Indices (80mm+) from NMFS Bottom Trawl Survey (1978 – 2017)

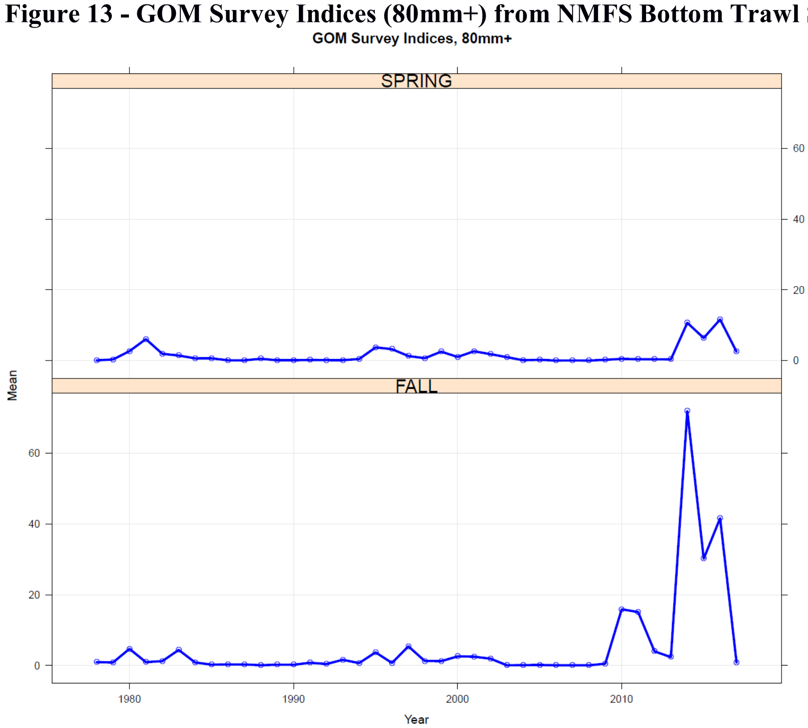
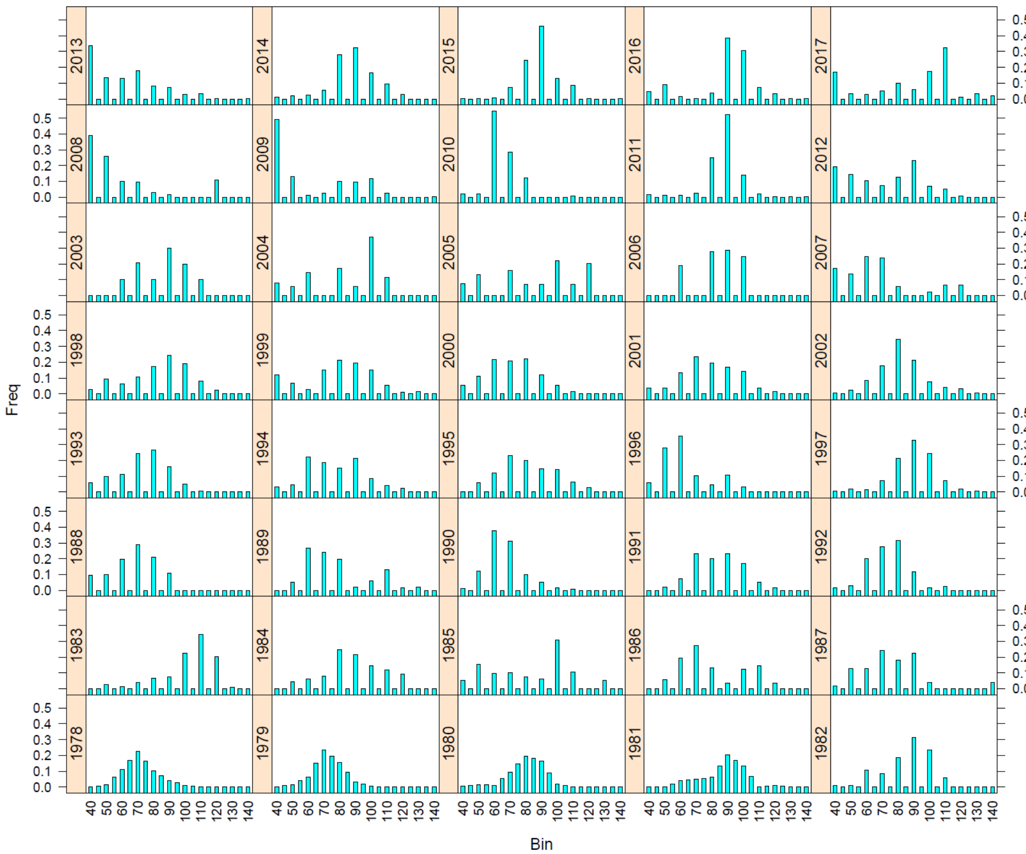


Figure 14 – Length composition of scallops captured in the GOM during the NMFS Fall Bottom Trawl Survey.



2.3.2 Spatial distribution of scallop catches in historic GOM surveys

Data from four regional trawl surveys in the Gulf of Maine were compiled to describe historic observations of scallop catch across the region. The survey data series included:

1. National Marine Fisheries Service Bottom Trawl Survey (1963-2014)
2. National Marine Fisheries Service Shrimp Trawl Survey (1983-2014)
3. Massachusetts Division of Marine Fisheries Bottom Trawl Survey (1978-2014)
4. New Hampshire/Maine Dept. of Marine Resources Inshore Trawl Survey (2000-2012)

There were 17,043 tows recorded in the Gulf of Maine between the NMFS Bottom Trawl, NMFS Shrimp Trawl, and MADMF Bottom Trawl surveys. Of the 17,043 recorded tows, 2,962 observed scallops. Tows with scallop catch were mostly attributed to inshore areas or to shallower ledges and banks in the central Gulf of Maine. These numbers do not include the NH/ME DMR Inshore Trawl survey because of its much shorter time series. However, NH/ME DMR Inshore Trawl survey catches were included in the figures to offer an additional look at regional scallop catches.

The National Marine Fisheries Service (NMFS) Dredge Survey does not include survey strata in the Gulf of Maine; however, the NMFS Dredge Survey did some exploratory surveying in the Gulf of Maine in 1983, 1984, and 1987. Dredge stations in these exploratory surveys mostly covered known scallop fishing grounds in the Gulf of Maine, such as Fippennies Ledge, Jeffreys Ledge, Stellwagen Bank, and the deeper water north east of Platts Bank. 84 of the 102 non-random dredge tows observed scallops, primarily on southern Jeffreys Ledge and Fippennies Ledge.

Though drawing quantitative conclusions from compiled trawl and dredge surveys in the Gulf of Maine may be difficult, some qualitative points regarding scallop distribution over time can be made:

1) Trawl surveys have consistently observed scallops in the area north of Provincetown, MA, extending towards the southern edge of Stellwagen Bank. This finding is consistent with our knowledge of this area being a popular fishing ground based on the VMS data described in Section 3.1.
2) All trawl surveys have consistently observed scallops in Ipswich Bay over the time-series meaning and may be an area of interest for future surveying.

Figure 15. Compiled scallop catches from the NMFS Bottom Trawl Survey (1963-2014), NMFS Shrimp Trawl Survey (1983-2014), MA DMF Bottom Trawl Survey (1978-2014), and NH/ME DMR Inshore Trawl Survey (2000-2012). The black "x" marks show survey tows with no scallops.

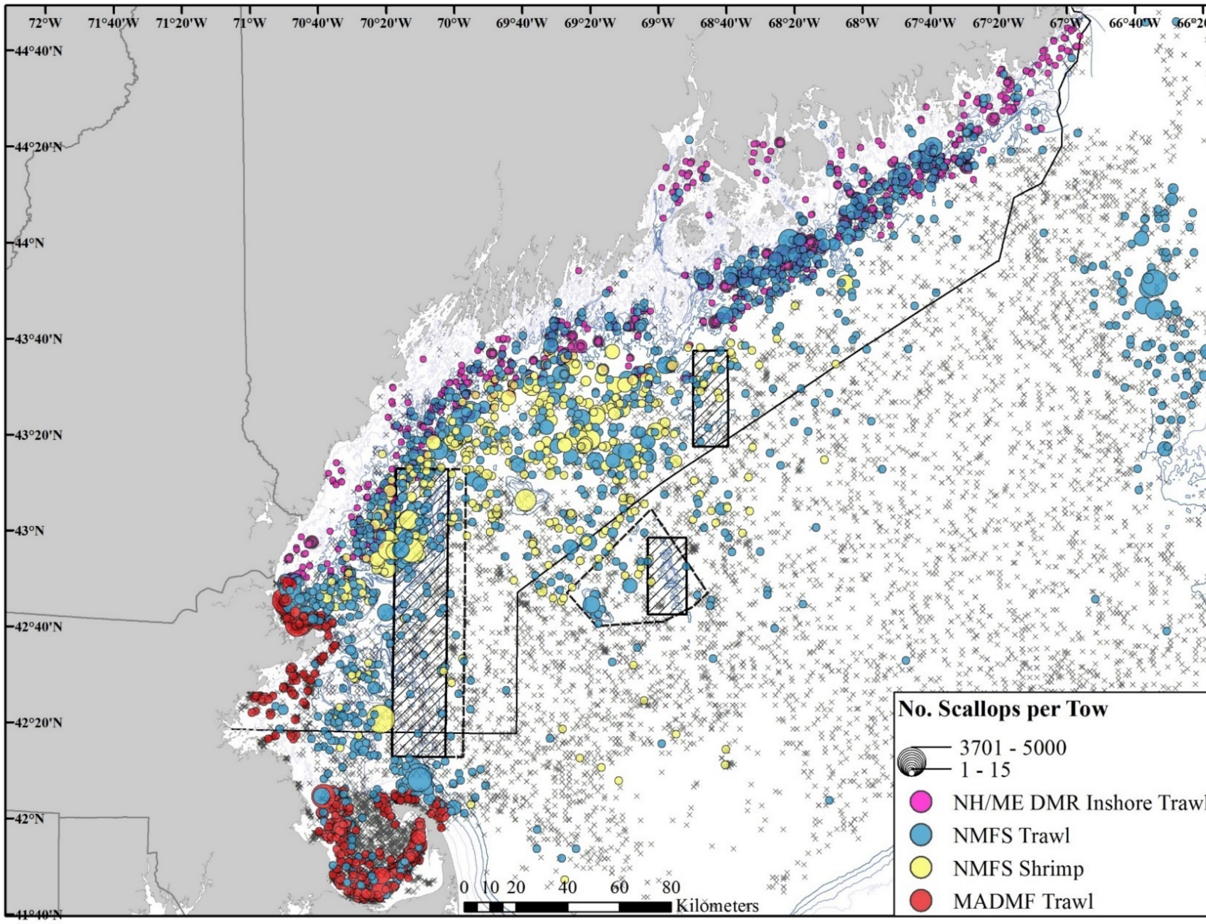


Figure 16 Compiled scallop catches from the NMFS Bottom Trawl Survey (1963-2014), NMFS Shrimp Trawl Survey (1983-2014), MA DMF Bottom Trawl Survey (1978-2014), and NH/ME DMR Inshore Trawl Survey (2000-2012), excluding survey tows with zero scallop catch.

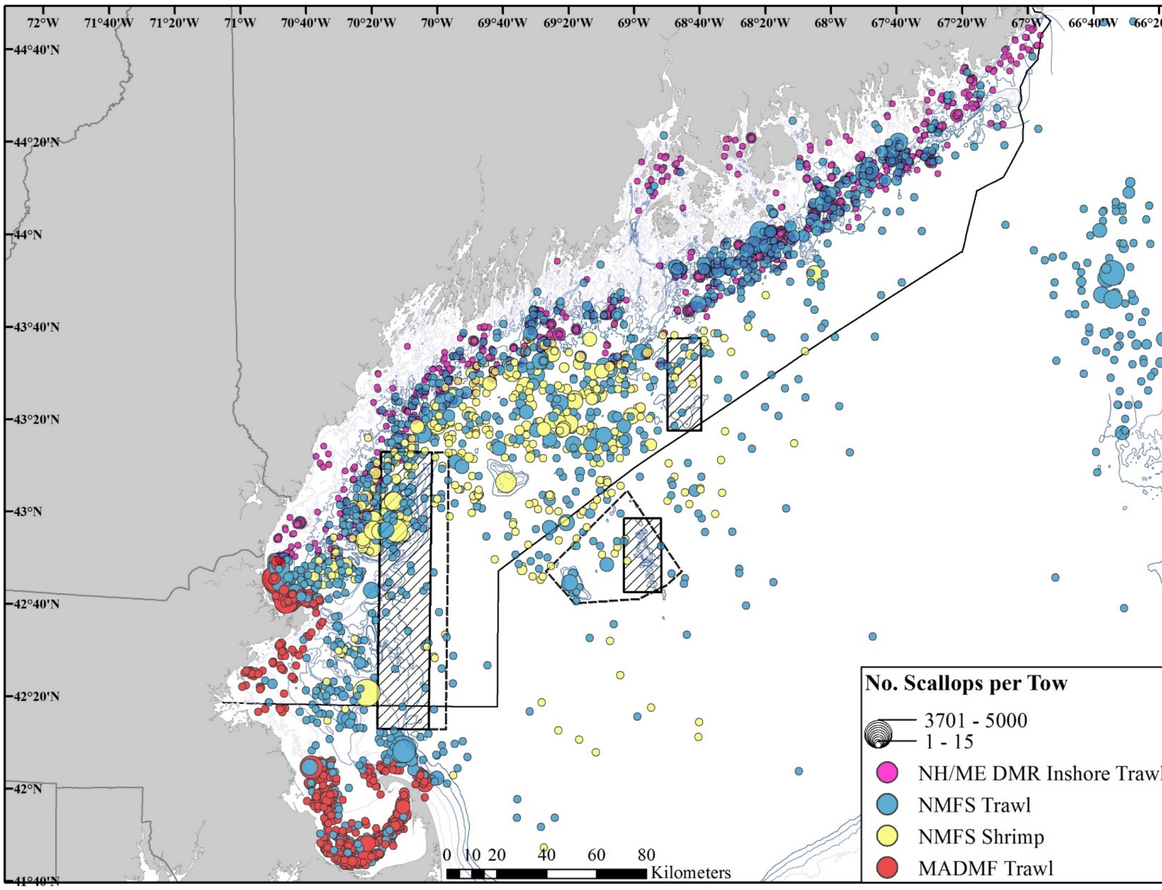


Figure 17. Compiled scallop catches from the NMFS Bottom Trawl Survey (1963-2014), NMFS Shrimp Trawl Survey (1983-2014), MA DMF Bottom Trawl Survey (1978-2014), and NH/ME DMR Inshore Trawl Survey (2000-2012) relative to 2016 and 2017 scallop fishery activity. Black "x" marks show tows with zero scallops. (NEXT PAGE)

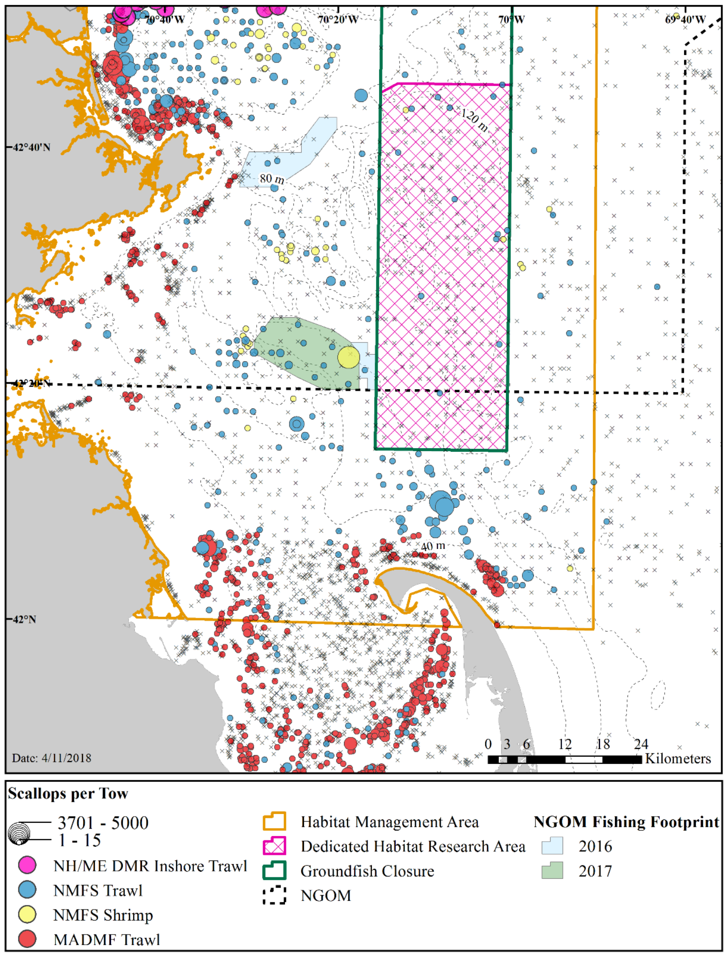
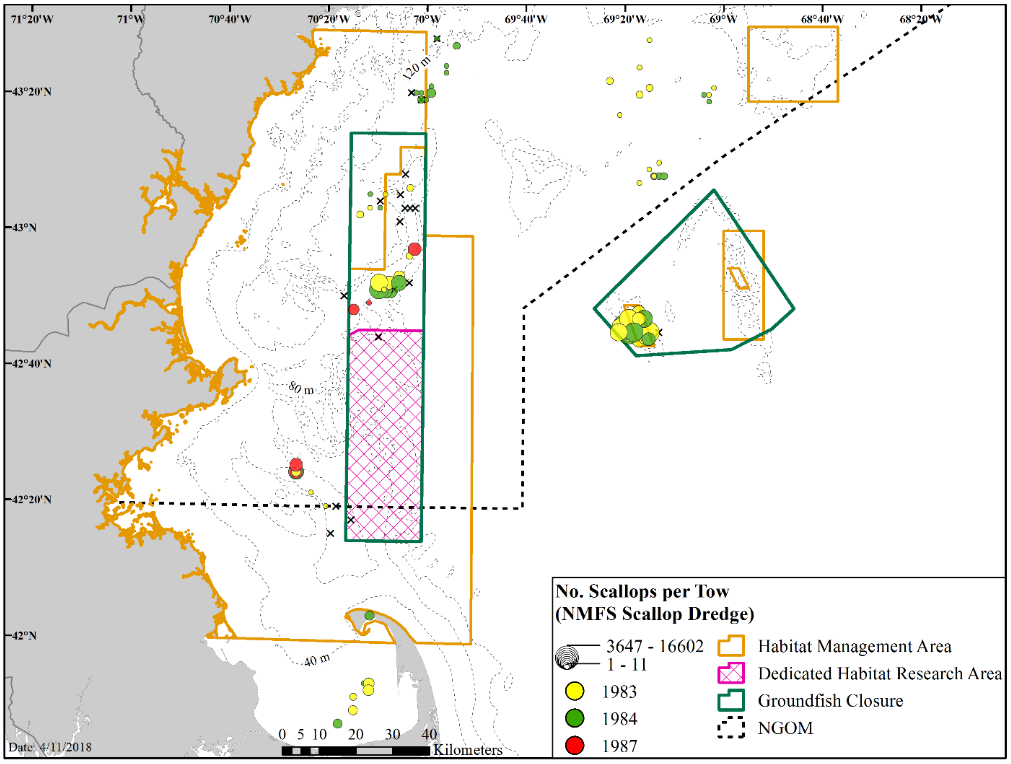


Figure 18 - Scallop catch per tow in the 1983, 1984, and 1987 NMFS dredge surveys in the Gulf of Maine



3.0 FISHERY DEPENDENT DATA

Several streams of fishery dependent data were used to characterize the sea scallop resource in the Gulf of Maine. Section 3.1 describes fishery trends using Vessel Monitoring System (VMS data) and Section 3.2 describes existing observer data in the Gulf of Maine.

3.1 FISHING EFFORT (VESSEL MONITORING SYSTEM DATA)

Federally permitted scallop vessels are required to carry vessel monitoring systems (VMS) that track the position of the vessel every 30 minutes ("pings"). VMS data can be pooled to determine activity by fleets in spatially distinct regions. Figure 19 illustrates the relative fishing activity in six areas of the Gulf of Maine: Stellwagen Bank north and south of 42' 20°, Platts Bank, Jeffreys Ledge, New Hampshire/Northern Massachusetts state waters (includes Ipswich Bay), and other regions of the Gulf of Maine that are marked as unclassified. The majority of fishing activity in the Gulf of Maine region occurs in the Stellwagen-South region, located due north of Provincetown, which is outside of the Northern Gulf of Maine Management area.

Figure 19 - Total number of VMS pings per region (thousands) by year in Gulf of Maine.

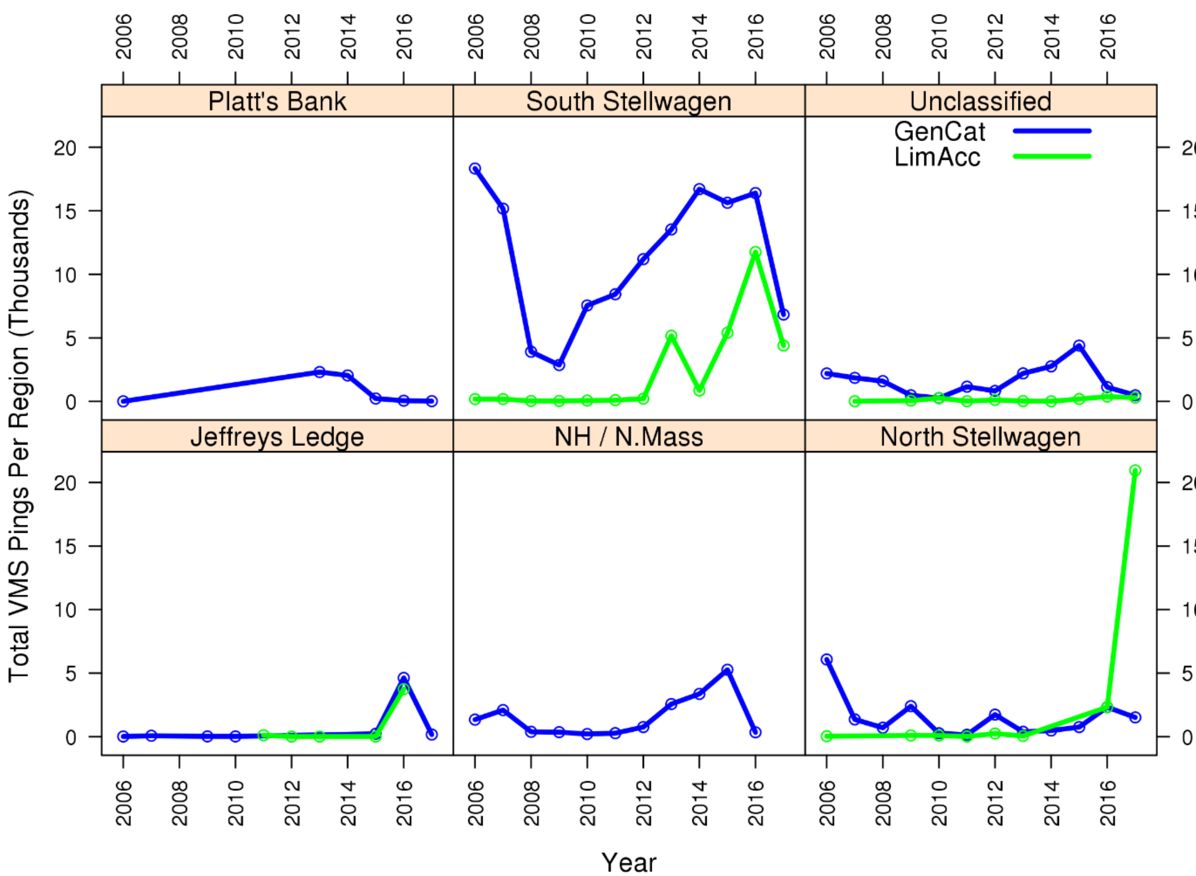


Figure 20 - Limited Access VMS fishing effort on Stellwagen Bank (2014 - 2017)

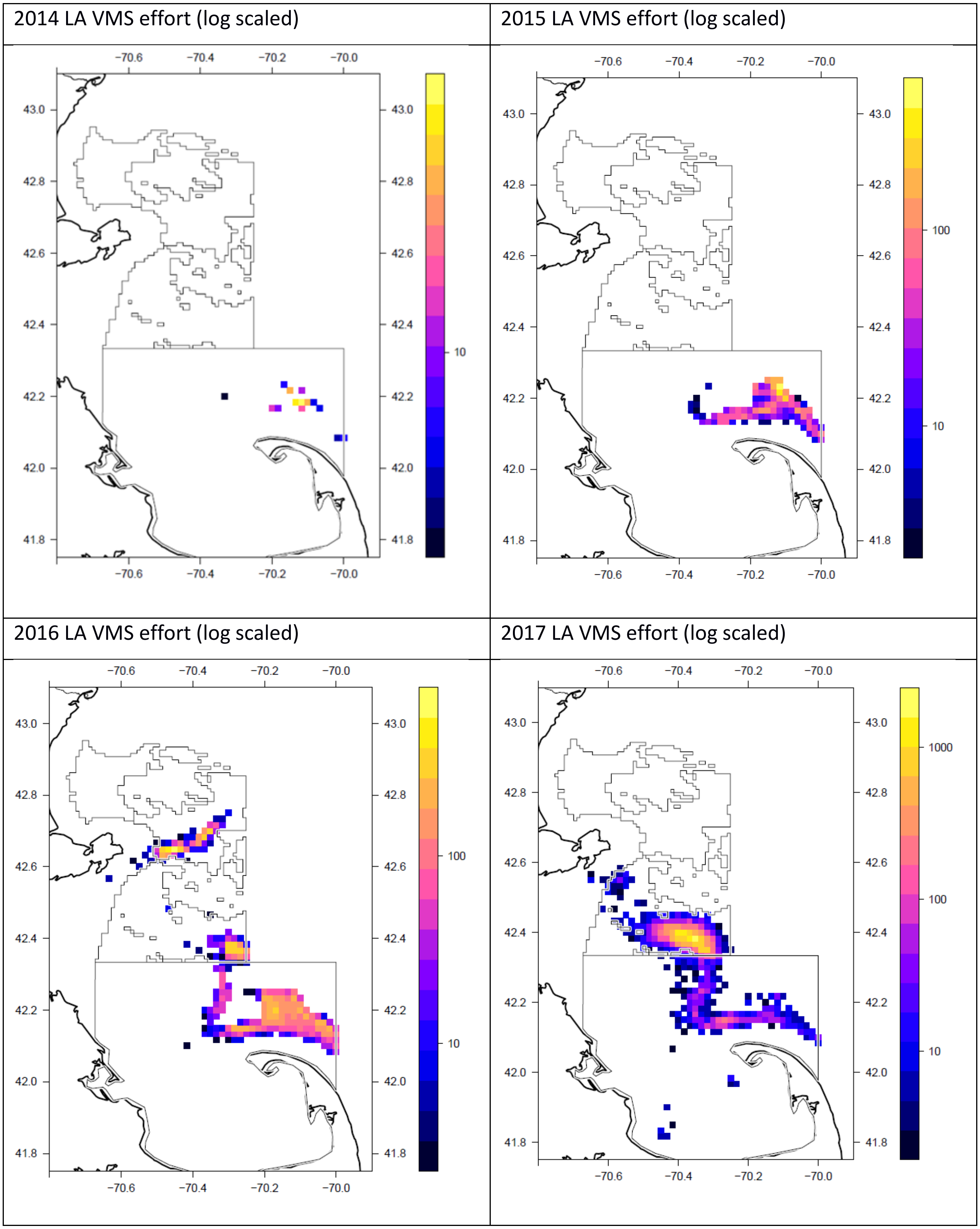
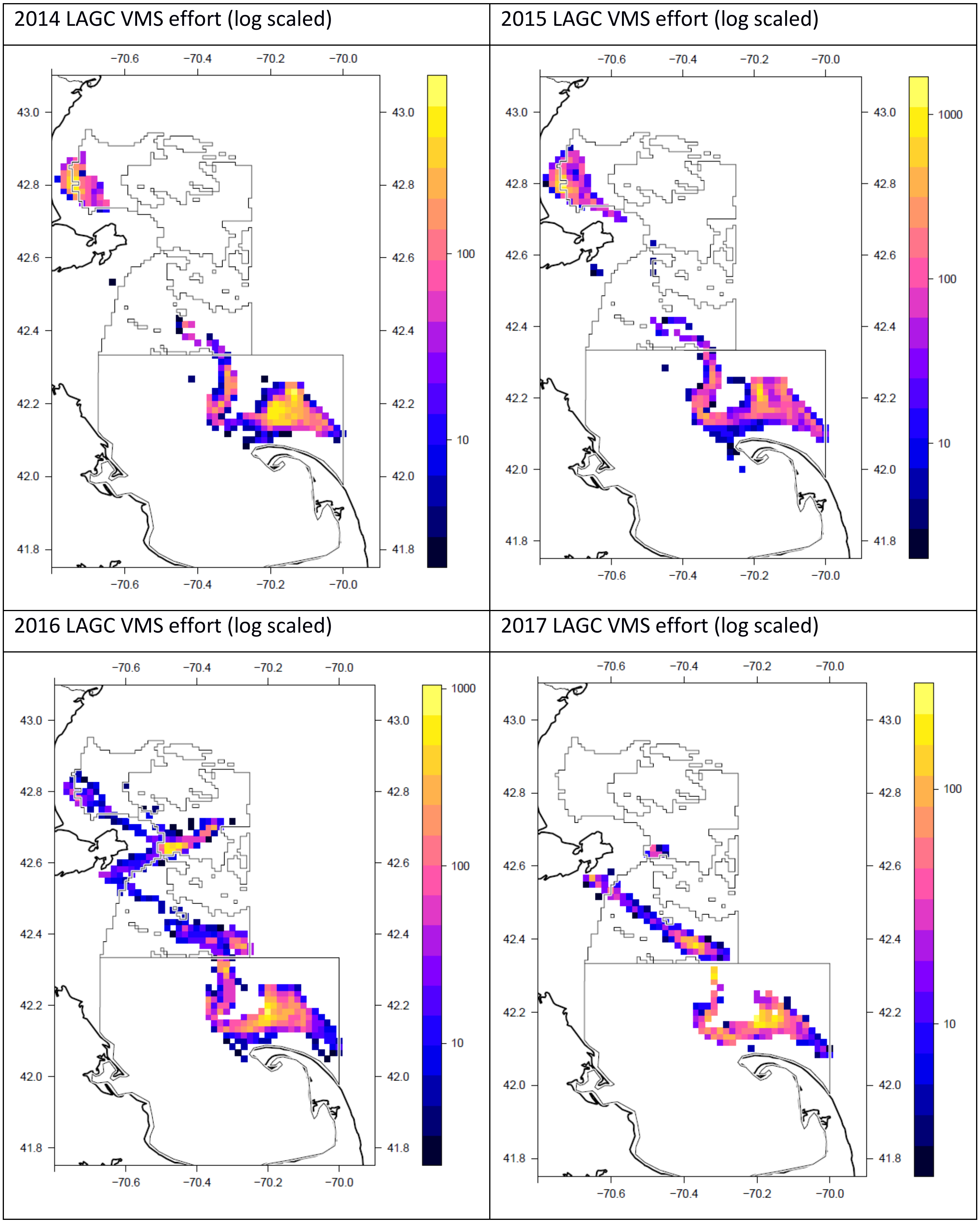


Figure 21 - LAGC VMS fishing effort on Stellwagen Bank (2014 – 2017)

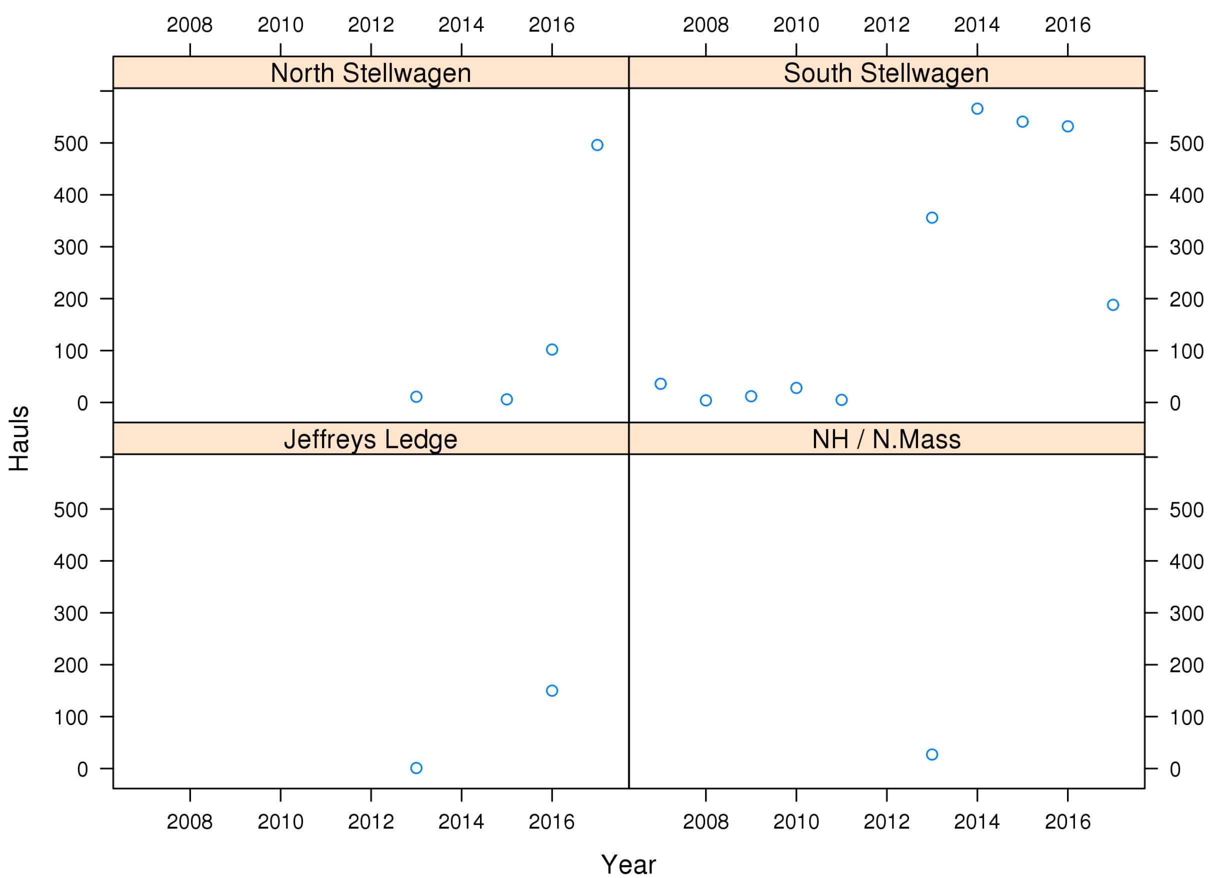


3.2 OBSERVER DATA

At-sea observers are deployed on federally permitted scallop vessels through the Northeast Fisheries Observer Program (NEFOP). In general, observers collect information on catch and bycatch (species, counts, weights), the location of fishing activity, the amount of fishing effort (number of hauls), and gear configurations. Observers are currently deployed on two classes of scallop vessels (Limited Access and Limited Access General Category IFQ permitted vessels).

Overall, a limited amount of observer data is available from directed scallop fishing in the Gulf of Maine. This is in part due to less scallop fishing occurring in the Gulf of Maine compared to Georges Bank and the Mid-Atlantic, and also because there is no at-sea observer coverage on directed LAGC scallop trips in the Northern Gulf of Maine Management Area. Figure 22 describes the number of hauls in sub-regions of the Gulf of Maine by year. The majority of available information is specific to Stellwagen Bank.

Figure 22 – Number of observed hauls by year and Gulf of Maine Region



3.2.1 FY2017 Observed LA trips in the NGOM

As previously noted, NEFOP does not currently assign at-sea monitors to LAGC IFQ or LAGC NGOM vessels fishing under a Northern Gulf of Maine (NGOM) management area declaration; however, in FY2017, several LA vessels fishing under DAS management in the NGOM had on-board observers.

The FY2017 NGOM fishery was concentrated on Stellwagen Bank, in shallow water just north of 42° 20’ N (i.e. the southern boundary of the NGOM management area). NEFOP assigned at-sea monitors to 18 LA scallop trips that directed effort in this part of the NGOM, resulting in a combined total of 443 observed hauls. Catch and discard data from the observed hauls are shown in Table 4.

Table 4. Summary of kept and discarded catch (lbs) from the 443 observed hauls recorded on LA vessels fishing in the NGOM in FY2017.

COMNAME	LBS	K/D	D/R		COMNAME	LBS	K/D	D/R
DEBRIS, ROCK	209790	DISCARD	ROUND		FLOUNDER, NK	17	DISCARD	ROUND
SCALLOP, SEA	164039	KEPT	DRESSED		COD, ATLANTIC	15	DISCARD	ROUND
SCALLOP, SEA	71432	DISCARD	ROUND		DEBRIS, PLASTIC	13	DISCARD	ROUND
SHELL, SCALLOP	15335	DISCARD	ROUND		OCEAN POUT	10	DISCARD	ROUND
SAND DOLLAR	5227	DISCARD	ROUND		WOLFFISH, ATLANTIC	8	DISCARD	ROUND
FISH, NK	3619	DISCARD	UNKNOWN		SQUID, ATL LONG-FIN	8	DISCARD	ROUND
SHELL, NK	3579	DISCARD	ROUND		FLOUNDER, WINTER (BLACKBACK)	8	KEPT	ROUND
FLOUNDER, WINTER (BLACKBACK)	1729	DISCARD	ROUND		CRAB, HERMIT, NK	6	DISCARD	ROUND
SCULPIN, LONGHORN	1336	DISCARD	ROUND		SKATE, LITTLE/WINTER, NK	4	DISCARD	ROUND
FLOUNDER, YELLOWTAIL	1005	DISCARD	ROUND		HALIBUT, ATLANTIC	4	KEPT	ROUND
FLOUNDER, SAND DAB (WINDOWPANE)	451	DISCARD	ROUND		SEAWEED, NK	4	DISCARD	ROUND
SKATE, LITTLE	403	DISCARD	ROUND		EEL, SAND LANCE, NK	3	DISCARD	ROUND
CRAB, JONAH	249	DISCARD	ROUND		DEBRIS, NK	3	DISCARD	ROUND
SKATE, WINTER (BIG)	113	DISCARD	ROUND		FLOUNDER, AMERICAN PLAICE	3	DISCARD	ROUND
STARFISH, SEASTAR,NK	97	DISCARD	ROUND		ANEMONE, NK	2	DISCARD	ROUND
CRAB, ROCK	90	DISCARD	ROUND		FLOUNDER, WITCH (GREY SOLE)	2	KEPT	ROUND
DEBRIS, FISHING GEAR	53	DISCARD	ROUND		FLOUNDER, SUMMER (FLUKE)	1	DISCARD	ROUND
SPONGE, NK	52	DISCARD	ROUND		QUAHOG, OCEAN (BLACK CLAM)	1	DISCARD	ROUND
MUSSEL, NK	48	DISCARD	ROUND		DEBRIS, METAL	1	DISCARD	ROUND
RAVEN, SEA	46	DISCARD	ROUND		HAKE, SILVER (WHITING)	0.6	DISCARD	ROUND
SNAIL, NK	38	DISCARD	ROUND		HERRING, ATLANTIC	0.6	DISCARD	ROUND
MONKFISH (GOOSEFISH)	29	DISCARD	ROUND		MACKEREL, ATLANTIC	0.3	DISCARD	ROUND
CLAPPER, SCALLOP	26	DISCARD	ROUND		FLOUNDER, WITCH (GREY SOLE)	0.3	DISCARD	ROUND
FISH, NK	25	DISCARD	ROUND		CORAL, SOFT, NK	0.2	DISCARD	ROUND
CLAM, NK	23	DISCARD	ROUND		SEA ROBIN, NORTHERN	0.2	DISCARD	ROUND

At-sea monitors also record measurement data (i.e. shell height, meat weight) of kept and discarded scallops for a minimum of 25% of observed hauls on a LA scallop trip. A total of 32,790 kept scallops were measured from observed LA hauls in the NGOM in FY2017, amounting to 164,039 pounds of observed kept meats.

Figure 23 displays the shell-height distribution of kept scallops from these observed hauls. Kept scallops ranged from approximately 75 mm to 160 mm, though the majority were between 100 mm and 115 mm.

Figure 24 displays the shell-height distribution of discarded scallops and Table 5 summarizes discarded meat weights by discard code. Of the 71,432 lbs of discarded meats, the majority were smaller scallops that did not have as high a market value as larger scallops being caught in the same hauls.

Figure 23 - Recorded shell-heights of kept scallops from observed LA hauls in the NGOM in FY2017

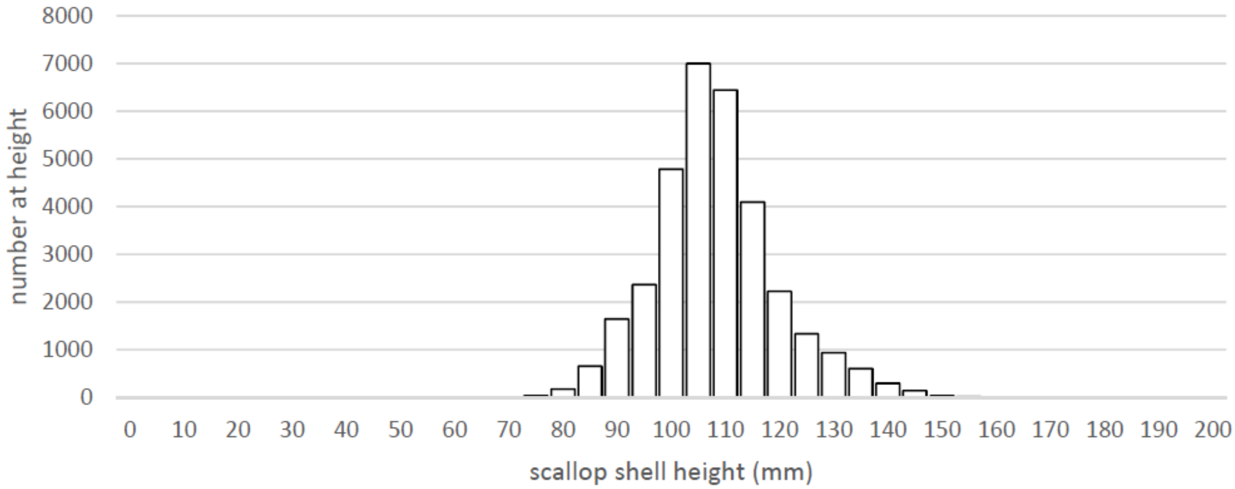


Figure 24 - Recorded shell-heights of kept (white bars) and discarded (black bars) scallops from observed LA hauls in the NGOM in FY2017.

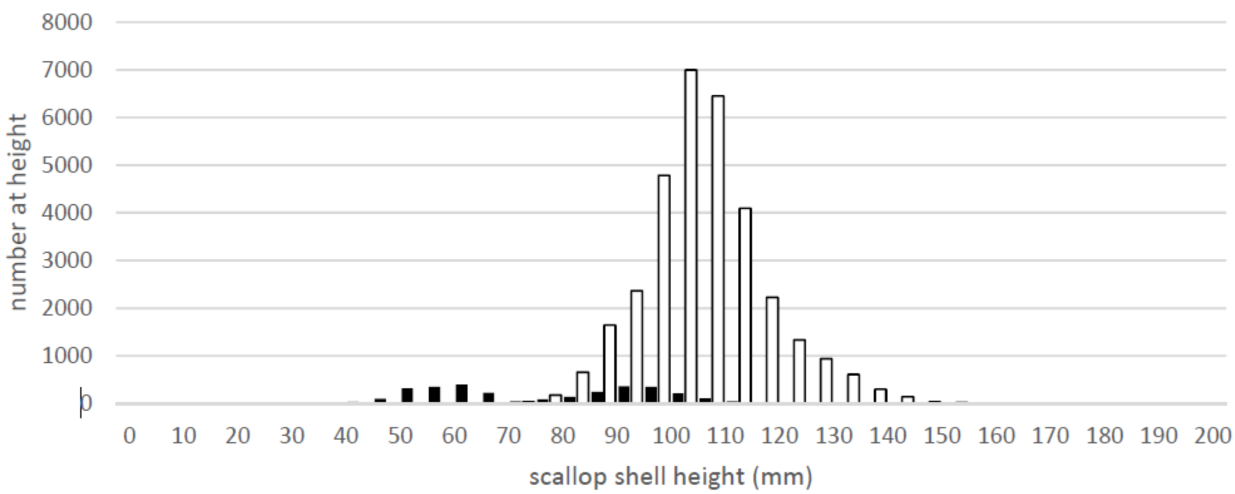


Table 5. Discarded scallop meats (in pounds) by discard code from observed NGOM LA hauls in FY2017.

Discard Code	scallop meats (lbs)
NO MARKET, TOO SMALL	38,050
POOR QUALITY, GEAR DAMAGE	16,226
DISCARDED, OTHER	17,156
Total	71,432

3.3 LANDINGS DATA

3.3.1 State Waters Landings Data

Scallop removals from the Gulf of Maine come from separate state and federal water fisheries. The largest state waters fishery in the region takes place in Maine, where landings exceeded three million pounds in the early 1980's (Figure 26). Recent landings have increased from around 100,000 lbs in 2008 and 2009 to nearly 800,000 pounds in 2017 (Figure 25).

Figure 25 - State of Maine state waters scallop landings (2008 - 2017) and average price per pound

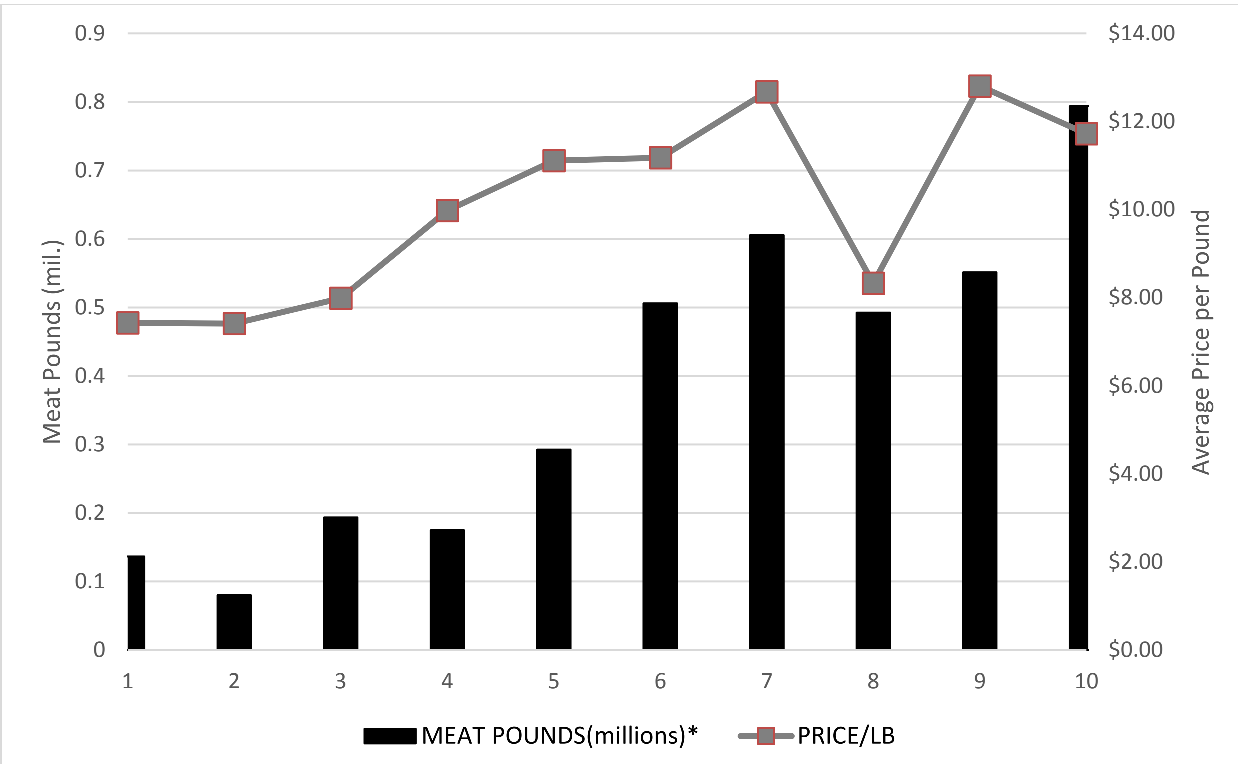
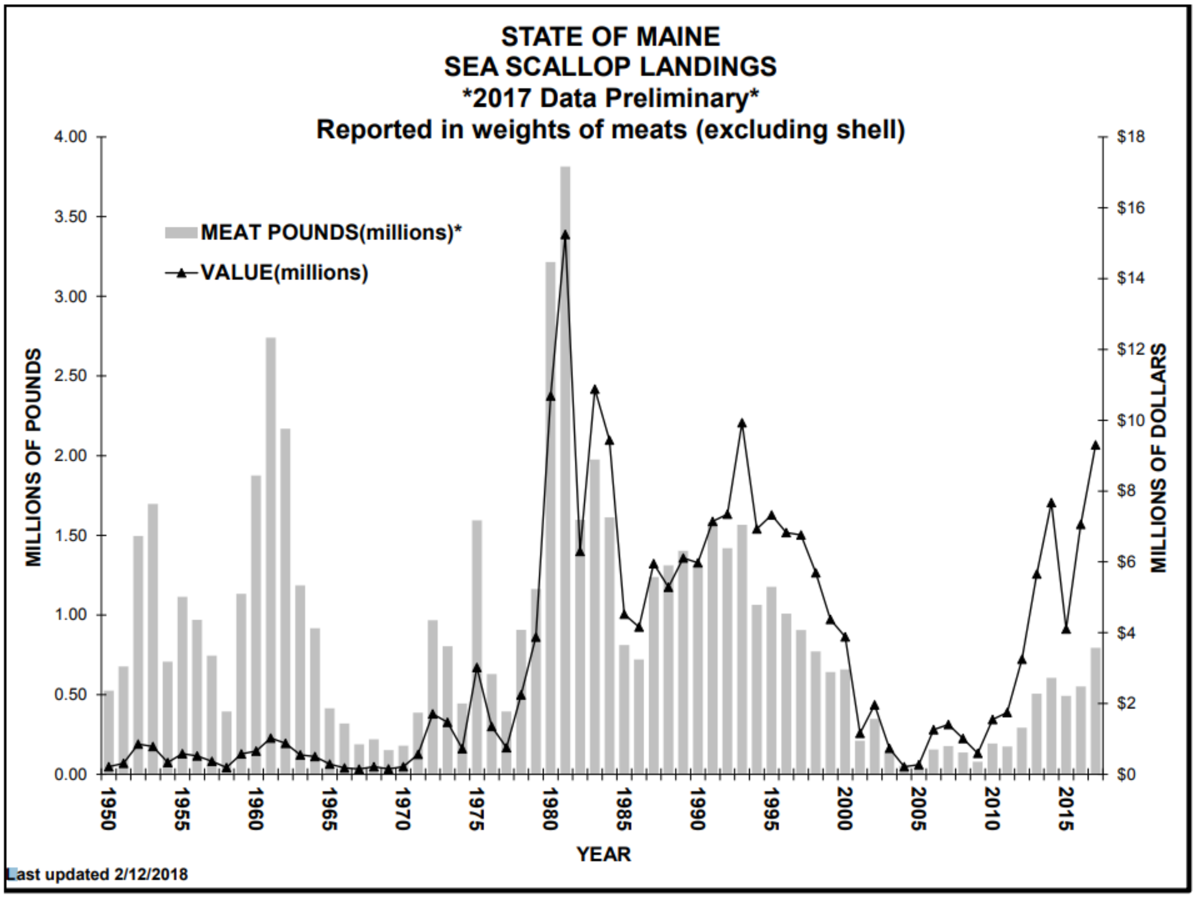


Figure 26 – State of Maine Sea Scallop Landings



3.3.2 Landings Data – Federal Fishery

3.3.2.1 Federal Dealer Landings from Northern Gulf of Maine Management Area

The majority of federal waters in the Gulf of Maine region are bounded by the Northern Gulf of Maine scallop management area, which is the found north of 42º 20' N and within the boundaries of the Gulf of Maine Scallop Dredge Exemption Area.

This area was created by the New England Fishery Management Council in 2008 to allow vessels that did not qualify for an LAGC IFQ permit the ability to fish at a conservative level (NEFMC, 2008). When fishing in the NGOM management area, LAGC vessels (i.e. LAGC IFQ, LAGC NGOM) adhere to a 200-pound possession limit and may fish until the annual Total Allowable Catch (TAC) limit is met and the area is closed by NMFS. Until the 2018 fishing year, LA vessels could operate in the NGOM under DAS management until the LAGC TAC was met and the management area was closed to all federally permitted scallop vessels. In 2018, the Council modified harvest controls in the NGOM so that an overall TAC was split between LAGC and LA components, and the LA share was dedicated to research compensation fishing (NEFMC, 2018). The following sub-sections are based on information reported in Framework 29 to the Scallop FMP (NEFMC 2018), and briefly summarize NGOM landings from fishing years 2008 to 2016 (Section 3.3.2.1.1) and in fishing year 2017 (Section 3.3.2.1.2).

3.3.2.1.1 NGOM Landings, 2008 to 2016

Before FY2013, combined annual landings by LAGC IFQ and LAGC NGOM vessels filled a small portion of the NGOM TAC, in several years landing less than 20% (Table 6). A strong year class of scallops on Platts Bank in FY2013 was followed by an increased LAGC NGOM fishing effort in this area through FY2014. LAGC IFQ vessels have typically focused effort to the southern portion of the management area, namely east and southeast of Cape Ann. IFQ landings nearly doubled between FY2014 and FY2015, with LAGC IFQ vessels working on aggregations of scallops located in Ipswich Bay and to the east and southeast of Cape Ann. FY 2015 marked the first year that the NGOM TAC was reached (overage of approximately 2,500 lbs). In FY2016, the NGOM TAC was exceeded by approximately 21,000 lbs.

From FY2008-FY2015, all NGOM landings came from the LAGC IFQ and LAGC NGOM fleets, during which time landings did not exceed the TAC (70,000 lbs) (Table 7). FY2016 marked a high point in landings by all permit types fishing in NGOM since the area was established in 2008, collectively totaling over 381,000 lbs (Table 7); LA vessels fishing under DAS management in the NGOM harvested roughly 293,000 lbs while approximately 89,000 lbs were landed by LAGC IFQ and LAGC NGOM vessels.

Table 6. Combined annual landings and percent of the NGOM TAC used in the management area.

FY	NGOM & IFQ Landings	TAC	Percent of TAC used
2008	9,936	70,000	14%
2009	15,534	70,000	22%
2010	8,639	70,000	12%
2011	6,908	70,000	10%
2012	7,440	70,000	11%
2013	55,450	70,000	79%
2014	57,842	70,000	83%
2015	72,546	70,000	104%
2016	89,083	67,454	132%

Table 7. Total landings attributed to the NGOM Management Area by permit type.

	Landings by Permit Category			Total NGOM Landings	NGOM closure date, (days open)
FY	LAGC IFQ	LAGC NGOM	LA		
2009	0	5,793	0	5,793	n/a, (entire FY year)
2010	4,762	3,877	0	8,639	n/a, (entire FY year)
2011	6,092	816	0	6,908	n/a, (entire FY year)
2012	894	6,546	0	7,440	n/a, (entire FY year)
2013	8,907	46,543	0	55,450	n/a, (entire FY year)
2014	11,521	46,321	0	57,842	n/a, (entire FY year)
2015	26,395	46,151	0	72,546	n/a, (entire FY year)
2016	26,484	62,599	292,517	381,600	May 13, (74 days)

3.3.2.1.2 NGOM Landings, 2017

FY2017 fishing by both LA and LAGC components was heavily concentrated along the southern boundary of the NGOM management area on Stellwagen Bank. VMS daily catch reports indicated 67 Limited Access vessels fished within the management area along with 38 LAGC vessels; the upper limit of total FY2017 removals from the NGOM was estimated to be roughly 1.6 million lbs. (Table 8). The NGOM management area was closed to all federally permitted scallop vessels effective at 12:01 AM on March 23rd, 2017. Upon closure, LA vessels were prohibited from fishing within the NGOM, but could continue fishing outside of the management area using open-area DAS.

Table 8. The number of active permits and estimated landings by Limited Access and LAGC vessels operating in the NGOM in FY2017.

Component	Active permits	scallop landings (lb)
LA	67	1,578,020
LAGC	38	47,437
Grand Total	105	1,625,457

3.3.2.2 Vessel Trip Report Landings Data

Federally permited vessels are required to report catch from each trip on vessel trip reports (VTRs). Vessels report the statistial reporting area they fished in, as well as the amout of kept catch by species. Figure 28 highlights statistical reporting areas in the Gulf of Maine. Figure 27 depicts estimated scallop landings from four areas in the Gulf of Maine by limited access (LA) and general cateogry (LAGC) vessels fom 1996 – 2017. The majority of the scallop removals over the past 20 years have come from statistical reporting area 514, which covers several important scallop grounds in the Gulf of Maine (Stellwagen Bank, Jeffreys Ledge, Ipswich Bay). This statisitcal reporting area is also bisected by the Nothern Gulf of Maine Management Area at 42°40'N latitude, meaning that harvest in this statistical area is regulated differently north and south of NGOM boundary line. Statistical reporting area 511 generally covers grounds in Eastern Maine, such as Machias/Seal Island, while 512 covers the Mid-Coast of Maine and federal waters south of Penobscot Bay. Area 513 covers Platts Bank, as well as inshore areas off of New Hanpshire and southern Maine. VTR data suggests that areas 511, 512, and 513 have supported both Limited Access and Limited Access Genenal Category fishing between 1996 and 2017.

Statistical reporting area 515 is considered part of the Gulf of Maine and covers Fippennies Ledge, which supported scallop effort in the mid-1980s to early 1990s; however, SRA 515 was excluded from this analysis because: 1) scallop fishing in this area generally pre-dated the use of VTRs and, 2) the majority of scallop habitat in SRA 515 (i.e. Fippennies Ledge, parts of Cashes Ledge), has been subject to a permananent groundfish mortality closure since 1998.

Figure 27 - VTR landings from Gulf of Maine statistical reporting areas (1996 - 2017) by fishery component.

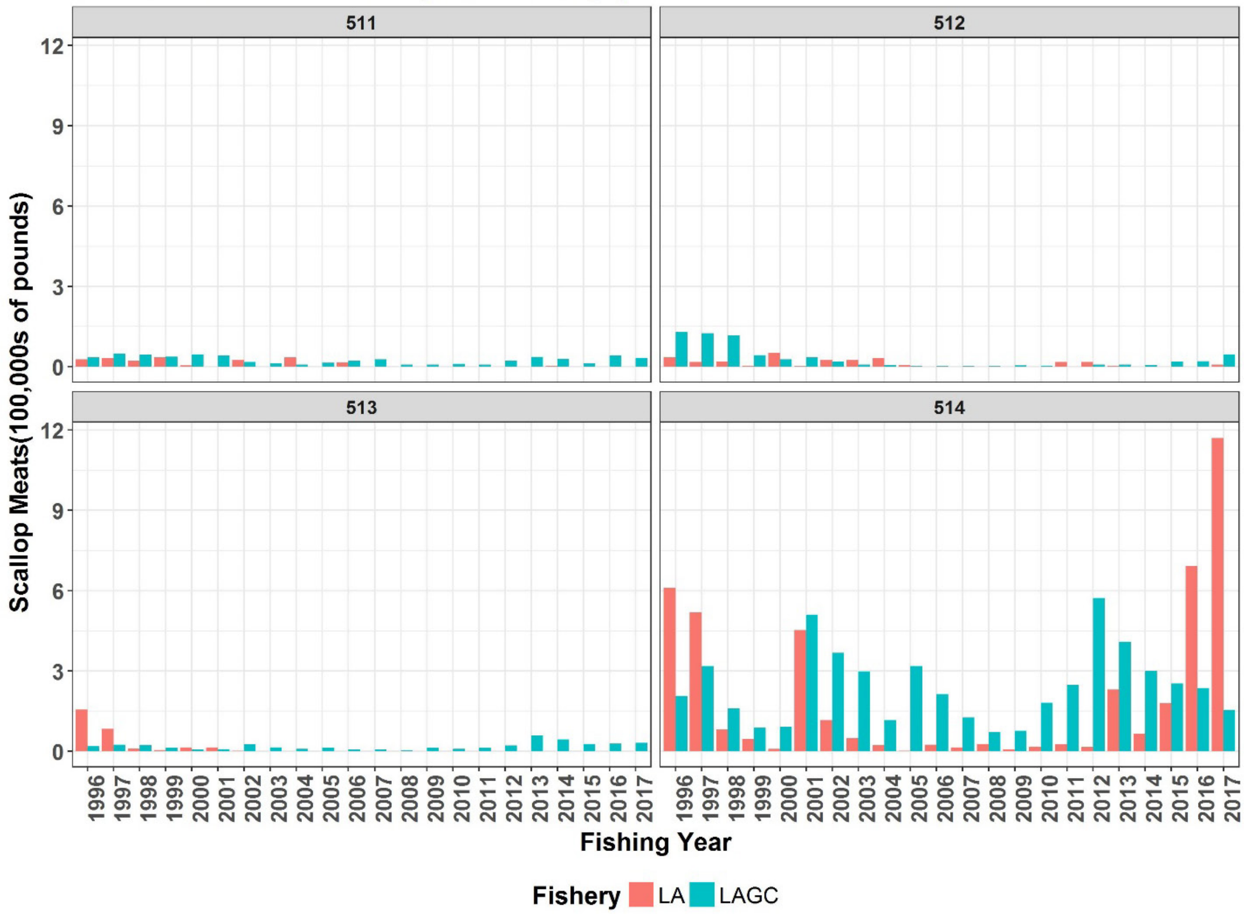
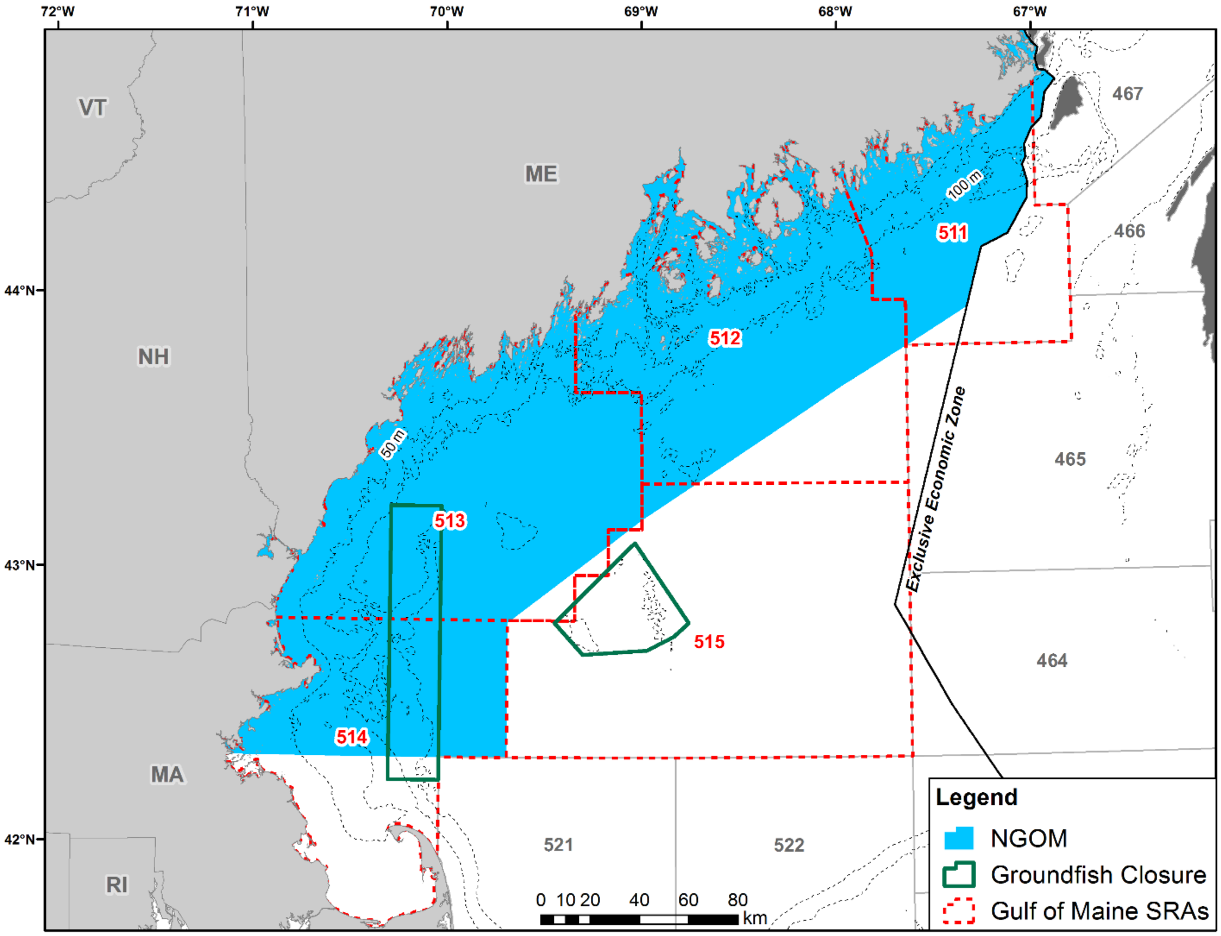


Figure 28. Statistical reporting areas (SRAs) in the Gulf of Maine relative to the NGOM management area and groundfish closures.



4.0 METHODS FOR DEVELOPING CATCH ADVICE

The current configuration of the Scallop Area Management Simulator (SAMS) provides forecasts for 21 sub-areas on Georges Bank and Mid-Atlantic, but does not cover portions of the Gulf of Maine. Since 2008, other methods for developing catch advice have been used to facilitate harvest of the resource in the Northern Gulf of Maine Management area, while harvest in areas of the Gulf of Maine south of 42°20 counts against fishery wide allocations. While some directed scallop fishing occurs in the Gulf of Maine, harvest from federal waters in this region have historically been a small proportion of total fishery removals. Table 9 describes the ways catch advice has been developed for the NGOM since 2008. Section 4.1 describes the methods used to develop catch advice for the NGOM in 2018.

Table 9 - Methods used to develop catch advice for Northern Gulf of Maine Management Area (2008 - 2018)

Method for developing catch advice for NGOM	Fishing Year(s) used
Historic Catch (TAC set at 70,000 lbs)	2008 – 2016
Simple exploitation of 2016 dredge survey estimates	2017
Forward projecting model using 2017 optical survey data	2018

4.1 2018 CATCH ADVICE - PROJECTION MODELING

A projection was used to develop catch advice for the Northern Gulf of Maine Management Area for the first time for the 2018 fishing year. For the 2018 exploitable biomass estimates, 2017 survey shell heights were projected forward 9 months for both Stellwagen (Figure 29) and southern Jeffreys Ledge (Figure 30) using growth parameters $L_\infty = 134.7$, $K = 0.433$, and $M = 0.16$. Projections were made using shell height to meat weight parameter estimates from the 2016 UMaine/Maine DMR survey of Stellwagen Bank. 2018 exploitable biomass was estimated at approximately 360 mt on Stellwagen Bank and approximately 101 mt on southern Jeffreys Ledge, translating to a combined exploitable biomass of approximately 461 mt.

The 2018 catch advice for the NGOM management area was calculated based on the projected 2018 exploitable biomass of southern Jeffreys Ledge and Stellwagen because these are the parts of the NGOM management area that were expected to be fished in 2018. In light of this, and acknowledging that this approach does not include biomass from other parts of the NGOM, the 2018 TAC was estimated at a conservative level using a fishing mortality target (F_{target}) of 0.18, which is less than 70% of the F_{MSY} reference point used for Georges Bank ($F_{MSY} = 0.3$) in SARC 59. Table 10 shows the TAC calculation for southern Jeffreys at varying fishing mortality targets. Table 11 shows the TAC calculation for Stellwagen Bank at varying fishing mortality targets. The combined 2018 TAC calculation (both southern Jeffreys and Stellwagen) is shown in Table 12.

Figure 29. Observed shell heights on Stellwagen Bank in 2017 and projected shell heights for April 2018.

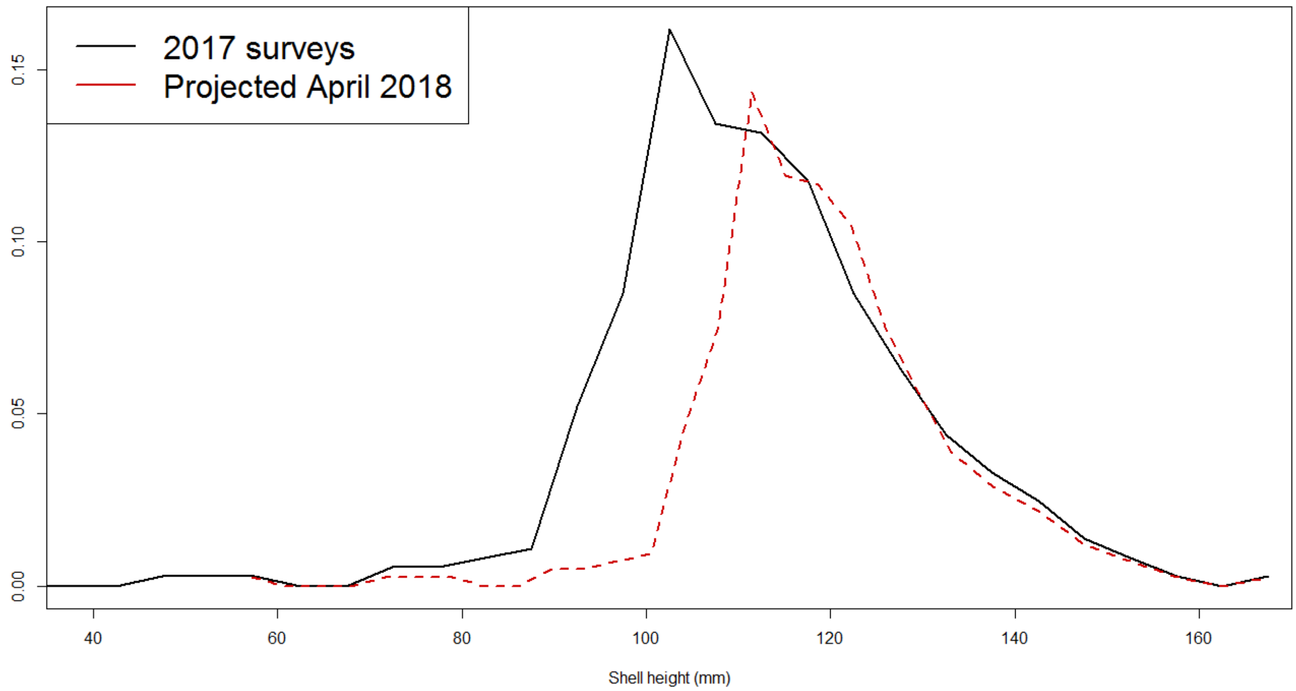


Figure 30. Observed shell heights on southern Jeffreys in 2017 and projected shell heights for April 2018.

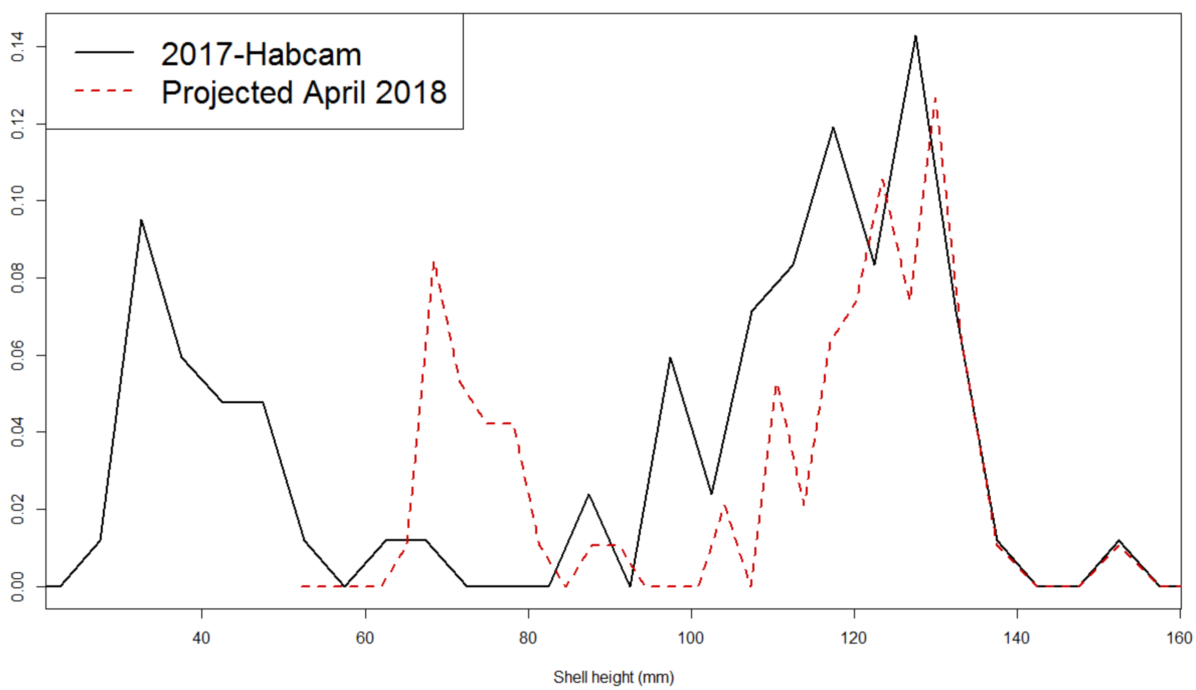


Table 10. Estimated TAC for southern Jeffreys Ledge in 2018 at varying fishing mortality target levels, based on projected exploitable biomass of 101 mt.

F_{target}	TAC (mt)	TAC (thousand lbs)
0.15	15.2	33.5
0.18	18.2	40.2
0.2	20.2	40.6

Table 11. Estimated TAC for Stellwagen Bank in 2018 at varying fishing mortality target levels, based on projected exploitable biomass of 360 mt.

F_{target}	TAC (mt)	TAC (thousand lbs)
0.15	54	119
0.18	64.8	142.9
0.2	72	158.7

Table 12. Estimated TAC for the NGOM in 2018 at varying fishing mortality target levels, based on combined projected exploitable biomass from southern Jeffreys Ledge and Stellwagen Bank.

F_{target}	TAC (mt)	TAC (thousand lbs)
0.15	69.2	152.6
0.18	83	183
0.2	90.2	198.9

4.2 POTENTIAL METHODS FOR DEVELOPING CATCH ADVICE AND DATA NEEDS

4.2.1 Estimating Yield-Per-Recruit, CASA, and Reference Points

The SAW 65 workgroup attempted to estimate yield-per-recruit for the Gulf of Maine using the stochastic yield model (SYM) that is used to develop reference points for Georges Bank and the Mid-Atlantic. Shell-height meat-weight data from UMaine/Maine DMR trawl surveys were used in the SYM model, however the model was determined to be unstable.

Expanding the CASA model to the GOM region would require estimates of abundance, biomass, growth, natural mortality, incidental and discard mortality, as well as landings. Additional biological and life-history information from animals in the Gulf of Maine is needed to estimate yield-per-recruit in this region, and to develop reference points for status determination.

4.2.2 Expanding Scallop Area Management Simulator (SAMS) Model

The SAMS model (forecasting) used to develop catch advice for Georges Bank and the Mid-Atlantic could be expanded to cover portions of the Gulf of Maine. The SAMS model could be used to develop catch advice for areas inside and outside of the Northern Gulf of Maine Management Area. The SAW workgroup recommended this approach when sufficient data is available.

Adding portions of the GOM into the SAMS model requires estimating a "starting" biomass for the current year that can be projected forward. When the model is run, it is typically initialized using the most recent survey data. For example, survey results from 2017 were used to initialize the model for 2018 projections. As the SAMS model can be used to make multi-year projections, catch advice could be made for two or three years using the model, depending on the availability/ frequency of survey information.

The additional data needed to expand the SAMS model are similar to the data needs of the CASA model and include sub-regional shell-height meat-weight relationships and growth, estimates of natural mortality, fishing mortality, and recruitment. If this data is not available for the Gulf of Maine, it may be reasonable to use data from Georges Bank to develop catch advice in the SAMS model for the Gulf of Maine.

4.2.3 Other methods for developing catch advice

The Gulf of Maine is outside of the federal shellfish survey strata, and there is not a dedicated federal survey of resource in this region. The majority of recent dedicated scallop survey work in the Gulf of Maine has been supported through research from the Scallop Research Set-Aside program, or through industry support.

As noted above, the distribution of the scallop resource in the Gulf of Maine is generally patchy, with the majority of animals in federal waters occurring on offshore ledges and banks. Over the past ten years the federal fishery has operated in spatially discrete areas such as Platts Bank, Jeffreys Ledge, Stellwagen Bank, and Ipswich Bay following periodic recruitment in these areas. Landings in the federal scallop fishery from the Gulf of Maine generally ~2% or less of total fishery landings.

The distribution of animals in the region, infrequency of surveys, and lack of key biological information means that existing models used to evaluate the status of the scallop resource (CASA, SYM) and forecast for catch advice (SAMS) cannot currently be applied to the entire Gulf of Maine region. Given the unique characteristics of this region relative to the rest of the resource, other methods of setting catch advice could be considered that require less fishery independent information or draw upon fishery dependent data streams. For example, fishery monitoring programs could be developed or expanded to support catch setting. Depletion modeling could be considered for developing or supporting catch advice in discrete areas of the Gulf of Maine.

5.0 LIST OF ACROYNMS

CASA	Catch-at-Size Analysis
CFF	Coonamessett Farm Foundation
GB	Georges Bank
GOM	Gulf of Maine
LA	Limited Access
LAGC	Limited Access General Category
MA	Mid-Atlantic
MA DMF	Massachusetts Division of Marine Fisheries
ME DMR	Maine Deprartment of Marine Resouces
NEFSC	Northeast Fisheries Science Center
NGOM	Northern Gulf of Maine Management Area
NMFS	National Marine Fisheries Service
SAMS	Scallop Area Management Simulator
SMAST	School for Marine Science and Technology
SYM	Stochastic yield model
TAC	Total Allowable Catch
VMS	Vessel Monitoring System
VTR	Vessel Trip Report
YPR	Yield-per-recruit

6.0 REFERENCES

Asci, S.C., Langton, R.W., and Stokesbury, K.D. 2018. Estimating similarity in benthic communities over decades and in areas open and closed to fishing in the central Gulf of Maine, USA. *Marine Ecology Progress Series* 595, 15-26. Online publication date: 14-May-2018.

Bethoney, N.D, Asci, S.C., and Stokesbury, K.D. 2016. Implications of extremely high recruitment events into the US sea scallop fishery. *Marine Ecology Progress Series* 547, 137-147.

Hart, D.R., and Chute, A.S. 2004. Essential fish habitat source document: Sea scallop, Placopecten magellanicus, life history and habitat characteristics, 2nd ed. NOAA Technical Memorandum NMFS NE-189.

New England Fisheries Management Council. 2007. Amendment 11 to the Atlantic Sea Scallop FMP, including a Supplimental Environmental Impact Statement and Initial Regulatory Flexibility Analysis.
http://s3.amazonaws.com/nefmc.org/Amendment_11FSEIS_0709_Submission_v1.pdf

New England Fishery Management Council. 2016. Omnibus Essential Fish Habitat Amendment. Volume 1: Executive Summary, Background and purpose, and Description of the affected environment.
http://s3.amazonaws.com/nefmc.org/OA2-FEIS_Vol_1_FINAL_161208.pdf

New England Fishery Management Council. 2018. Framework 29 to the Atlantic Sea Scallop FMP, including an Environmental Assessment, an Initial Regulatory Flexibility Analysis and Stock Assessment and Fishery Evaluation (SAFE) Report:
http://s3.amazonaws.com/nefmc.org/Scallop-FW29-for-final-submission.pdf

Serchuk, F.M. 1983. Results of the 1983 USA Sea Scallop research survey: distribution and abundance of Sea Scallops in the Georges Bank, Mid-Atlantic and Gulf of Maine Regions and biological characteristics of Iceland Scallops off the coast of Massachusetts. Northeast Fisheries Science Center Reference Document 83-37, Woods Hole, Massachusetts.

Stokesbury, K.D., Carey, J.D., Harris, B.P. and O'Keefe, C.E. 2010. High Densities of Juvenile Sea Scallop (*Placopecten Magellanicus*) on Banks and Ledges in the Central Gulf of Maine. *Journal of Shellfish Research*, *29*(2), pp.369-372.

Truesdell, S.B., Kelly, K.H., and Yong Chen. 2014. An assessment of the sea scallop resource in the Northern Gulf of Maine management area. Appendix B7 to the 59th Northeast Regional Stock Assessment Workshop (59th SAW) assessment report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 14-09; 782 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at:
http://www.nefsc.noaa.gov/publications/crd/crd1409/

Appendix A4 - Scallop Landing and Discard Estimation

Ben Galuardi (NOAA/NMFS/GARFO/APSD)

September 6, 2018

This paper presents discard estimates for the Atlantic sea scallop fishery (*Placopecten magellacanicus*), calculated using the Standard Bycatch Reporting Methodology (Wigley et al., 2007) and the discaRd package for R (Galuardi et al., 2016; Linden et al., 2016). This analysis is provided as an update to discard estimates calculated during the previous assessment (NEFSC, 2014).

Methods

Estimates of scallop discards (mt) were derived for scallop fleets using Northeast Fishery Observer Program (NEFOP) and commercial landings (dealer) information for calendar years 2010-2017. Stratifications for discard estimation from scallop trips were based on region, gear and Access/Open area fishing. Discards from non-scallop trips also utilized At-sea monitoring (ASM) information.

Gulf of Maine (GOM), Georges Bank (GB), Southern New England (SNE), and Mid-Atlantic (MA) regions were defined according to statistical area (Figure App A4-1). Gear was classified broadly into Scallop Dredge (SD) and Scallop Trawl (ST). Trawl encompasses any trawl gear used on a scallop declared trip. As there was very little trawl fishing during 2010-2017, trawl mesh size was not included as a separate stratification in this analysis. Calendar quarter was not included as a stratification.

Access/Open area fishing was determined based on vessel monitoring system (VMS) declarations for commercial trips, and by Program Code from NEFOP data. Access areas fished in 2010-2017 were Closed Area I (CAI), Closed Area II (CAII), Nantucket Lightship (NLS), Hudson Canyon (HC), Elephant Trunk (ET), Elephant Trunk Flex (ETF), DelMarVa (DMV), and The Mid-Atlantic Access Area (MA; which is HC, ET/EF and DMV combined). This resulted in 16 stratifications for each year which are, with the exception of the addition of the GOM and SNE regional stratifications, nearly identical to those used for in-season bycatch monitoring of flounder species in the scallop fishery.

The Data Matching and Imputation System (DMIS) is a set of tables used by the Greater Atlantic Regional Fisheries Office (GARFO) for quota monitoring and data requests. These tables contain matched Vessel Trip Report (VTR), dealer, and VMS declarations. These data were used in this analysis since they accurately stratify trips and effort based on Open vs. Access Area and contain all necessary fields from VTR and dealer records.

Discards were estimated using a ratio estimator (Cochran, 1977) where stratified rates of observed discards to kept all (k_{all}) and multiplied by the total estimated landings (K) from commerical trips in that strata (see eq. App A4-1 and eq. App A4-2). Coefficients of variation (CV) were calculated as the ratio of the standard error of the discards (eq. App A4-3) divided by the discards eq. App A4-4.

The discaRd package for R was developed during the Discard Methodolgy Review (see `https://www.greateratlantic.fisheries.noaa.gov/aps/discard/review/index.html`). The package uses the Cochran ratio estimator equations to generate discard and CV estimates, as well as determine the number of seadays required to reach a target CV within (or without) specified stratifications. This package was used for all discard and CV calculations presented.

$$r_{jh} = \frac{\sum_{i=1}^{n_h} d_{jih}}{\sum_{i=1}^{n_h} k_{ih}} \tag{App A4-1}$$

$$\hat{D} = \sum_{h=1}^{L} K_h r_{jh} \tag{App A4-2}$$

$$V(\hat{D}_j) = \sum_{h=1}^{L} K_h^2 \quad \left. \frac{N_h}{n_h} \right) \frac{1}{\left(\frac{\sum_{i=1}^{n_h} k_{ih}}{n_h} \right)^2} \left[\frac{\sum_{i=1}^{n_h} (d_{jih}^2 + r_{jh}^2 k_{ih} - 2 r_{jh} d_{jih} k_{ih})}{n_h - 1} \right] \tag{App A4-3}$$

$$CV(\hat{D}_j) = \frac{\sqrt{V(\hat{D}_j)}}{(\hat{D}_j)} \tag{App A4-4}$$

Where

$\hat{D}_j$ is the total estimated discarded pounds for species j

K_h is the total kept pounds of all species in stratum h

r_{jh} is the separate ratio for species j in stratum h

d_{jih} is discards of species j from observed trip i in stratum h

k_{ih} is the kept pounds of all species on observed trip i

n_h is the number of observed trips in stratum h

L is the number of strata

Results

Landings of Atlantic sea scallops, by calendar year, are shown in figures App A4-2, App A4-3 and in table App A4-5. Discard estimates are presented for directed scallop and non-directed scallop trips in figures App A4-4 and App A4-5, respectively. Figure App A4-6 shows total removals and highlights the scale of removals from scallop and non-scallop trips against landings.

Tables App A4-1-App A4-4 show discard estimates, broken down by Open vs Access Area fishing and by dredge/trawl gear used by region: Georges Bank (GB, table App A4-1), Gulf of Maine (GOM, table App A4-2), Mid-Atlantic (MA, table App A4-3), and Southern New England (SNE, table App A4-4, respectively. Coefficients of variation (CV) are presented for each stratification.

Discards in the scallop fishery during 2010-2017 were between 3-8%, peaking in 2016 at 11%. Differences in discard by region were quite pronounced, especially in 2016 where grade and nematode issues likely played a role in higher than average discard rates.

Table App A4-6 shows the overall observer rates, discard rates and CV for each calendar year. When examining individual strata (App A4-1-App A4-4) it is clear some strata are observed more than others, and that rates and CVs are highly variable. Overall, however, the annual CV is quite low, as the fishery has adequate observer coverage for a stable estimate of discards.

Discussion

There are several differences between the current approach and the previous assessment. First, this analysis was only done for most recent seven years of fisheries data. Instead of repeating the extensive mining of fisheries and observer data back to 1989, which was not likely to be different, efforts were spent on providing the most accurate estimates of discards and landings for the most recent time period (2010-2017).

Access vs. Open area stratifications in the 59th SAW were based on VTR location (NEFSC, 2014). Given data limitations in observer and catch records, these methods were appropriate, especially for older records. For more recent information, including the most recent calendar year (2017), it is possible to stratify Access Area and Open area fishing based on VMS declaration. This is a far superior approach as VTR locations are self-reported vs. a specific designation of intended fishing area.

Regional stratification used in the current analysis added SNE and GOM as regional definitions, which differs from previous assessments (NEFSC, 2014). Statistical area, and expert consensus, were the basis for regional definition in each case. Finally, the stratification scheme employed here closely resembles that used for in-season discard estimation of flounder species in the Atlantic scallop fishery (see `https://www.greateratlantic.fisheries.noaa.gov/ro/fso/Reports/ScallopProgram/ACL_YT_catch_estimation_20120815.pdf`)

Acknowledgments

I wish to thank Jessica Blaylock and Dvora Hart for guidance during this analysis. The NEFOP observers deserve special thanks in the data collection so imperative to discard estimation.

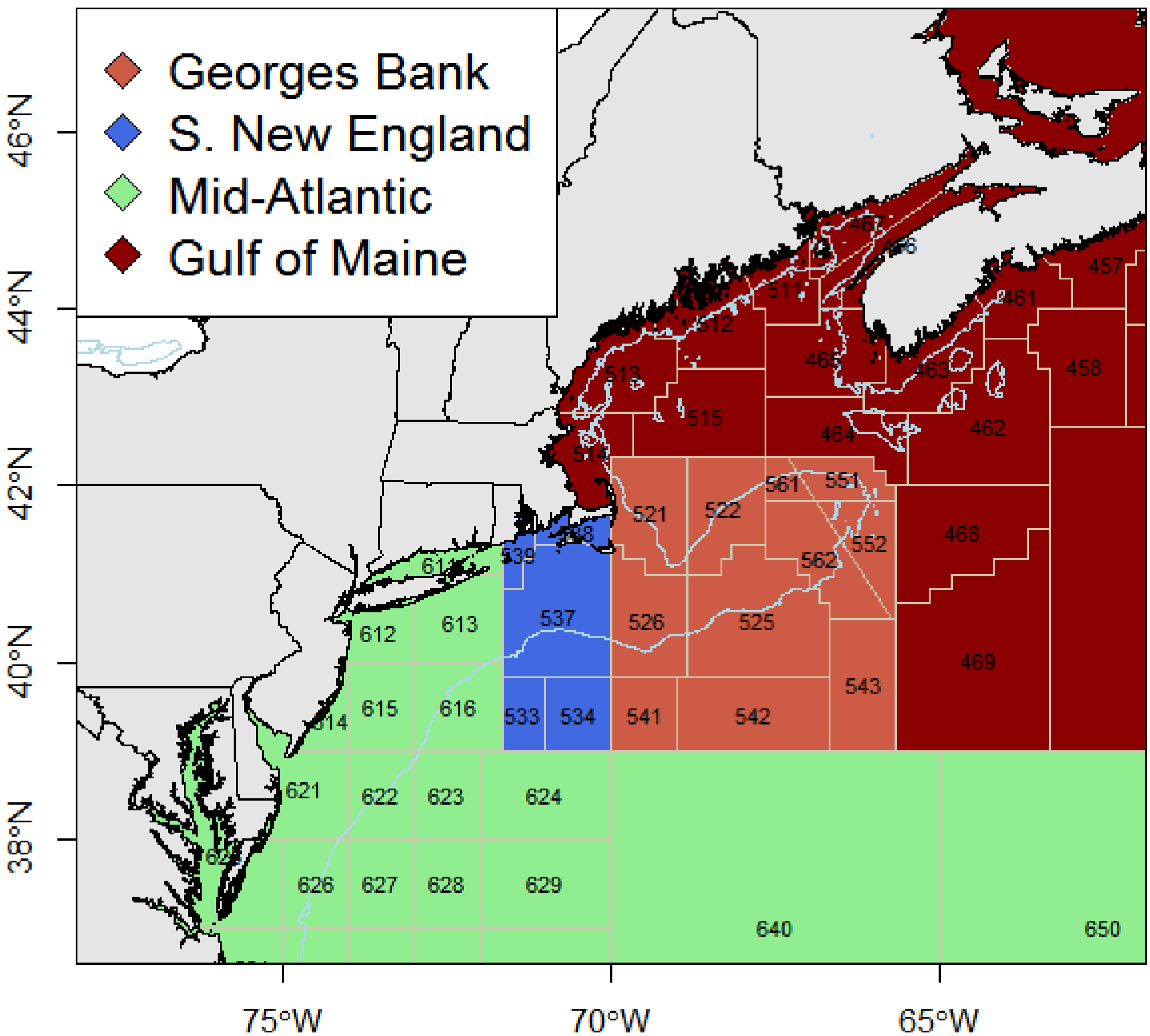


Figure App A4-1: Regional Definitions

Figure App A4-2: Landings by region within Open and Access Areas, by calendar year.

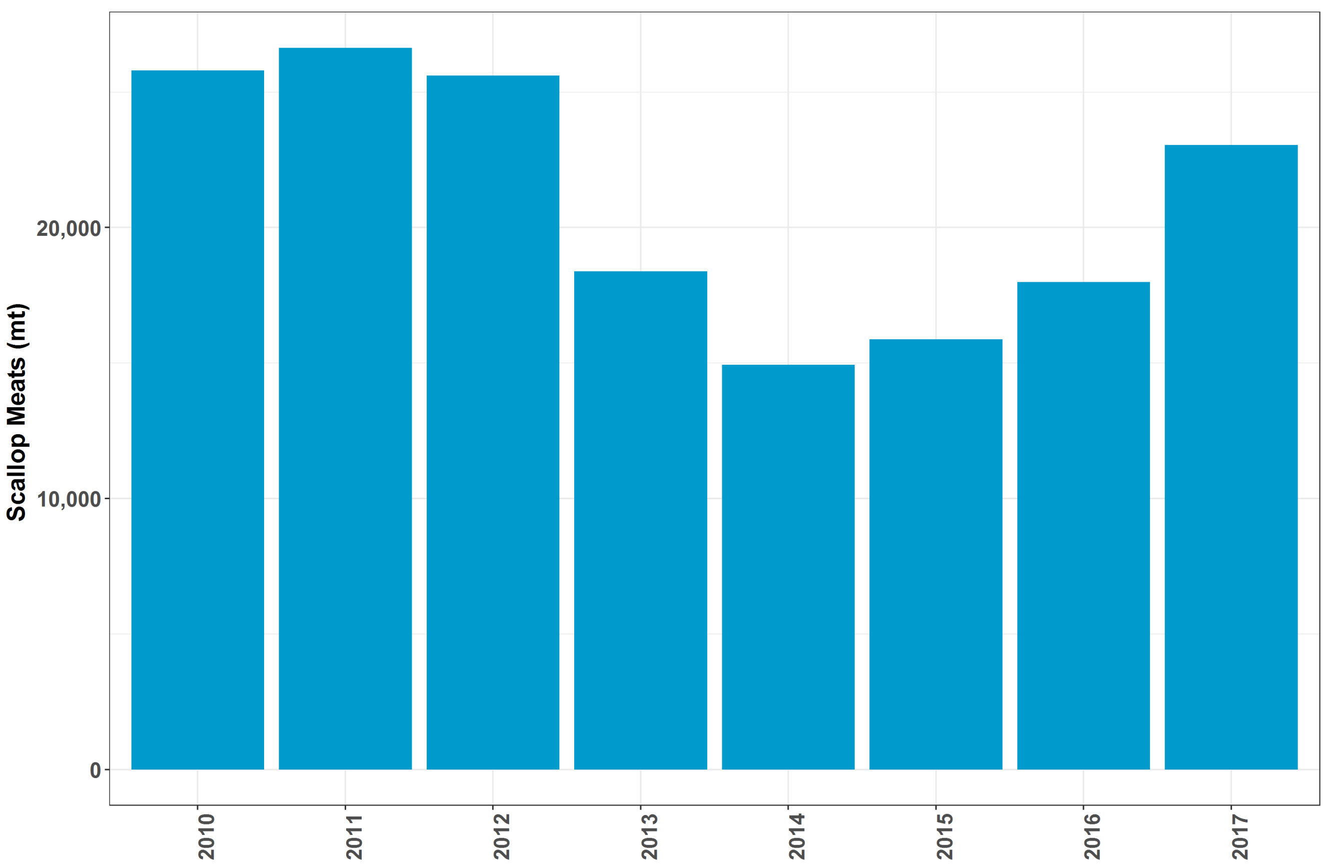


Figure App A4-3: Total landings by calendar year.

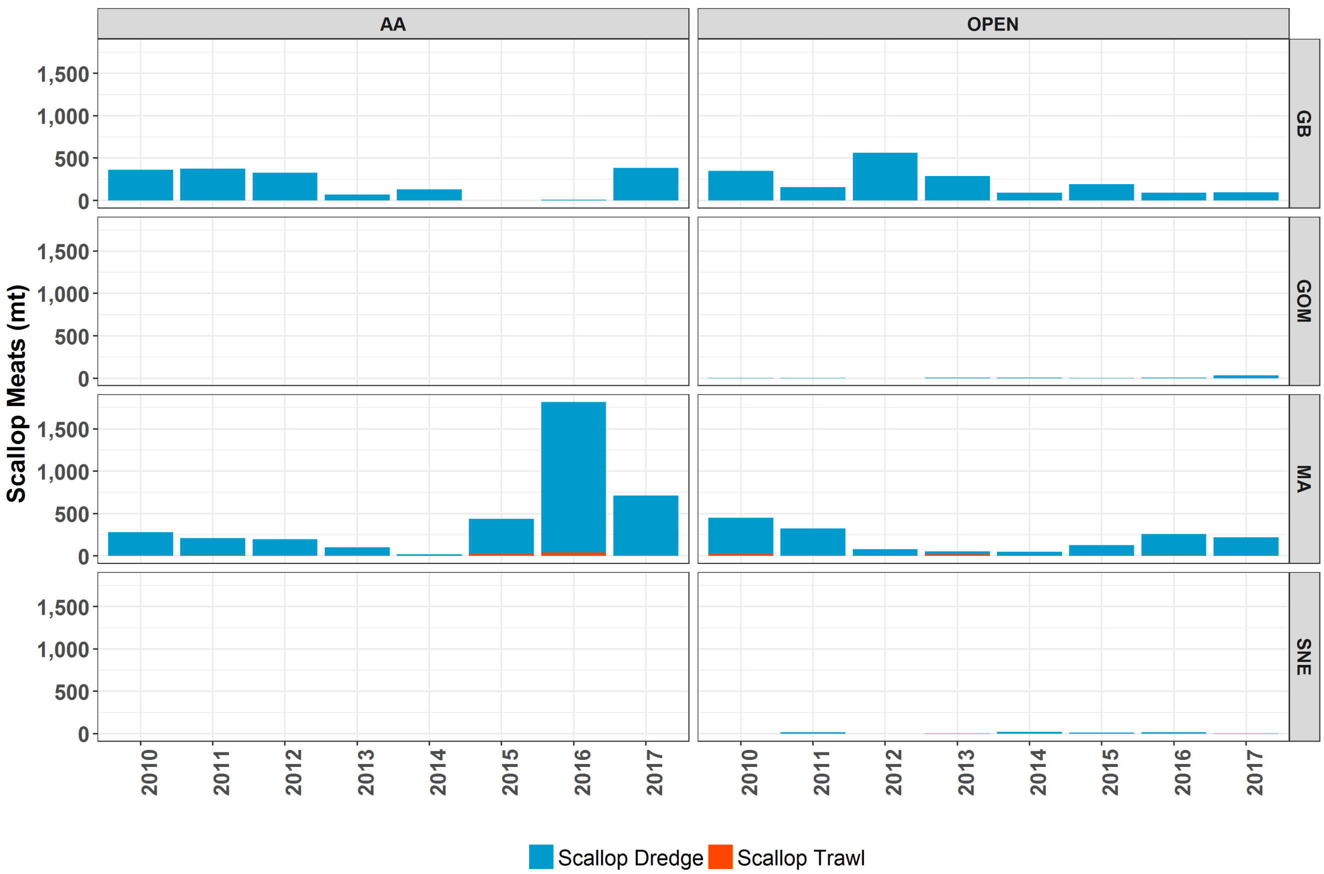


Figure App A4-4: Discards of scallops on directed scallop trips by calendar year for scallop dredge and trawl.

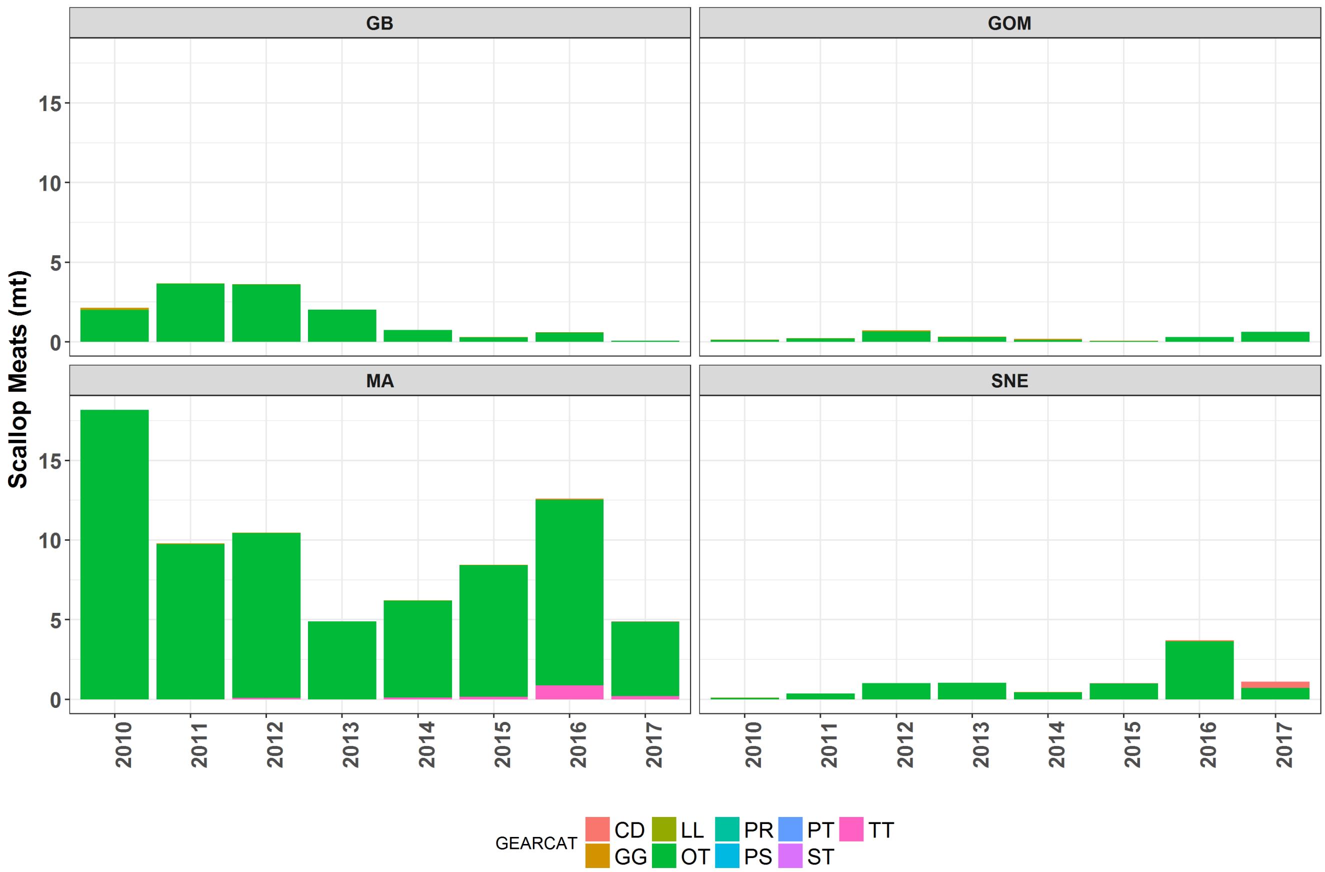


Figure App A4-5: Discards of scallops on non-directed scallop trips by calendar year. Colors represent gear category. Otter Trawl (OT) and turtle trawl (TT) are clearly dominant, with all other gear types having extremely small estimated discard amounts.

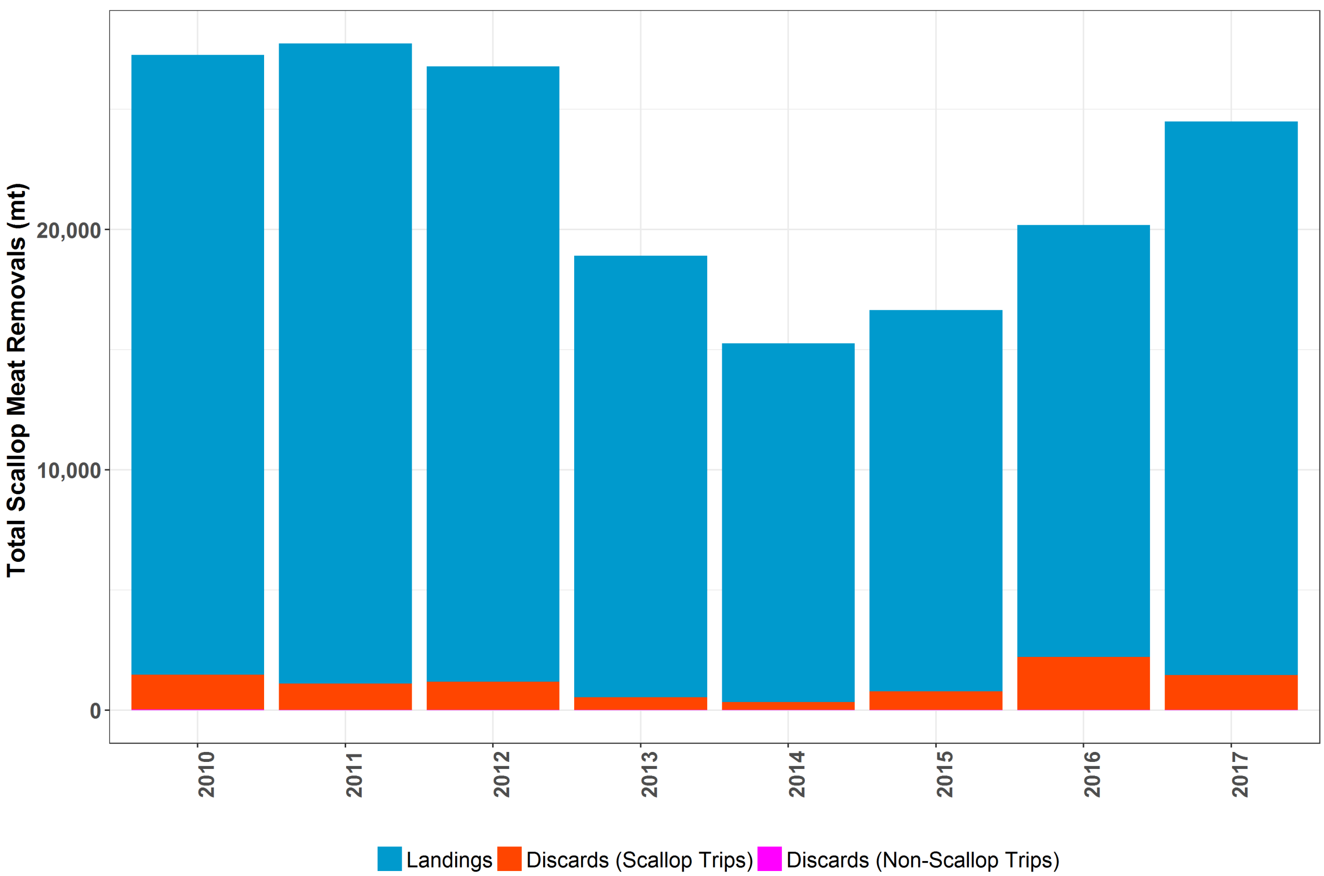


Figure App A4-6: Total Removals. Discards from non-scallop trips are too small to be seen at this scale.

Table App A4-1: Estimated scallop discards and coeffcient of variation (CV) for Georges Bank (GB) Region

Year	Area Type	Gear	Observed Trips	Discard Rate	Discard (mt)	CV
2010	AA	SD	54	0.13	364	0.21
2010	OPEN	SD	42	0.11	351	0.27
2010	OPEN	ST	0			
2011	AA	SD	73	0.07	377	0.27
2011	OPEN	SD	63	0.05	159	0.33
2011	OPEN	ST	0			
2012	AA	SD	121	0.05	328	0.15
2012	AA	ST	0			
2012	OPEN	SD	98	0.08	562	0.13
2012	OPEN	ST	0			
2013	AA	SD	67	0.02	68	0.22
2013	AA	ST	0			
2013	OPEN	SD	175	0.03	289	0.16
2013	OPEN	ST	0			
2014	AA	SD	51	0.07	129	0.18
2014	AA	ST	0			
2014	OPEN	SD	98	0.03	91	0.28
2015	AA	SD	12	0	1	0.36
2015	OPEN	SD	127	0.04	190	0.18
2015	OPEN	ST	0			
2016	AA	SD	36	0.05	7	0.19
2016	OPEN	SD	106	0.02	91	0.21
2017	AA	SD	181	0.08	382	0.12
2017	AA	ST	0			
2017	OPEN	SD	50	0.05	97	0.27

Table App A4-2: Estimated scallop discards and coeffcient of variation (CV) for Gulf of Maine (GOM) Region

Year	Area Type	Gear	Observed Trips	Discard Rate	Discard (mt)	CV
2010	AA	SD	0			
2010	OPEN	SD	3	0.04	3	0.67
2010	OPEN	ST	0			
2011	AA	SD	0			
2011	OPEN	SD	1	0.03	2	
2011	OPEN	ST	0			
2012	AA	SD	0			
2012	OPEN	SD	0			
2012	OPEN	ST	0			
2013	AA	SD	0			
2013	OPEN	SD	33	0.02	6	0.31
2013	OPEN	ST	0			
2014	AA	SD	0			
2014	OPEN	SD	42	0.05	9	0.4
2014	OPEN	ST	0			
2015	AA	SD	0			
2015	OPEN	SD	37	0.01	2	0.21
2015	OPEN	ST	0			
2016	AA	SD	0			
2016	AA	ST	0			
2016	OPEN	SD	40	0.01	6	0.3
2016	OPEN	ST	0			
2017	AA	SD	0			
2017	OPEN	SD	29	0.05	34	0.22
2017	OPEN	ST	0			

Table App A4-3: Estimated scallop discards and coeffcient of variation (CV) for Mid-Atlantic (MA) Region

Year	Area Type	Gear	Observed Trips	Discard Rate	Discard (mt)	CV
2010	AA	SD	108	0.04	278	0.21
2010	AA	ST	10	0.02	2	0.39
2010	OPEN	SD	130	0.04	425	0.28
2010	OPEN	ST	19	0.15	25	0.35
2011	AA	SD	106	0.03	203	0.25
2011	AA	ST	5	0.12	10	0.58
2011	OPEN	SD	144	0.03	324	0.17
2011	OPEN	ST	2	0	0	0.92
2012	AA	SD	101	0.05	197	0.13
2012	AA	ST	2	0.01	0	0.77
2012	OPEN	SD	100	0.01	77	0.17
2012	OPEN	ST	17	0.01	4	0.61
2013	AA	SD	45	0.07	103	0.24
2013	AA	ST	0			
2013	OPEN	SD	137	0.01	34	0.22
2013	OPEN	ST	19	0.11	20	0.37
2014	AA	SD	40	0.01	20	0.31
2014	AA	ST	0			
2014	OPEN	SD	216	0.01	45	0.21
2014	OPEN	ST	19	0.04	5	0.31
2015	AA	SD	185	0.05	414	0.13
2015	AA	ST	3	0.32	22	0.67
2015	OPEN	SD	136	0.05	121	0.22
2015	OPEN	ST	7	0.1	5	0.5
2016	AA	SD	224	0.24	1774	0.08
2016	AA	ST	6	0.58	42	0.42
2016	OPEN	SD	192	0.05	249	0.14
2016	OPEN	ST	13	0.05	9	0.29
2017	AA	SD	159	0.11	708	0.29
2017	AA	ST	6	0.13	3	0.54
2017	OPEN	SD	200	0.02	216	0.18
2017	OPEN	ST	10	0.08	4	0.45

Table App A4-4: Estimated scallop discards and coeffcient of variation (CV) for Southern New England (SNE) Region

Year	Area Type	Gear	Observed Trips	Discard Rate	Discard (mt)	CV
2010	AA	SD	0			
2010	OPEN	SD	3	0.01	2	1.18
2010	OPEN	ST	0			
2011	AA	SD	0			
2011	AA	ST	0			
2011	OPEN	SD	6	0.08	19	0.44
2011	OPEN	ST	0			
2012	AA	SD	0			
2012	OPEN	SD	3	0	0	0.63
2012	OPEN	ST	0			
2013	AA	SD	0			
2013	AA	ST	0			
2013	OPEN	SD	27	0.02	5	0.33
2013	OPEN	ST	1	0.14	0	
2014	AA	SD	0			
2014	OPEN	SD	39	0.07	20	0.47
2014	OPEN	ST	1	0.04	0	
2015	AA	SD	0			
2015	OPEN	SD	39	0.06	11	0.45
2015	OPEN	ST	2	0.32	0	0.31
2016	AA	SD	0			
2016	OPEN	SD	45	0.05	17	0.25
2016	OPEN	ST	1	0.28	1	
2017	AA	SD	0			
2017	OPEN	SD	32	0.01	2	0.35
2017	OPEN	ST	1	0.53	1	

Table App A4-5: U.S. Landings and Discards (mt meats)

	2010	2011	2012	2013	2014	2015	2016	2017
U.S. Landings								
GB	5,991	8,383	13,736	11,987	5,265	5,067	3,976	6,922
GOM	72	99	215	368	180	184	484	658
MA	19,549	17,841	11,503	5,706	9,180	10,430	13,171	15,150
SNE	181	305	158	318	307	193	350	311
TOTAL	25,793	26,628	25,612	18,379	14,932	15,874	17,981	23,041
U.S. Discards								
GB	715	536	890	357	220	191	98	479
GOM	3	2	0	6	9	2	6	34
MA	730	537	278	157	70	562	2,074	931
SNE	2	19	0	5	20	11	18	3
TOTAL	1,450	1,094	1,168	525	319	766	2,196	1,447

Table App A4-6: Total observation rates, discard rates, discard estimates, and CV by calendar year.

Year	Observer Rate(%)	Discard Rate (%)	Kept All (mt)	Discard Estimate (mt)	CV
2010	0.04	0.08	25,834	1,450	0.12
2011	0.04	0.05	26,759	1,094	0.13
2012	0.05	0.05	25,764	1,169	0.08
2013	0.05	0.03	18,510	526	0.11
2014	0.06	0.03	15,072	319	0.12
2015	0.06	0.05	15,964	765	0.09
2016	0.06	0.11	18,046	2,196	0.07
2017	0.07	0.07	23,094	1,447	0.15

References

S.E. Wigley, P.J. Rago, K.A. Sosebee, and D.L. Palka. The Analytic Component to the Standardized Bycatch Reporting Methodology Omnibus Amendment: Sampling Design and Estimation of Precision and Accuracy (2nd edition). *U.S. Department of Commerce, Northeast Fisheries Science Center*, Reference Document 07-09, 2007.

Benjamin Galuardi, Daniel Linden, and Brant McAfee. discaRd: Cochran bycatch estimation v1.1, 2016. URL https://www.greateratlantic.fisheries.noaa.gov/aps/discard/review/discard_1.1.tar.gz. R package version 1.0.

D.W. Linden, B. Galuardi, and B.M. McAfee. Methods for examining in-season behavior of the cumulative discard estimation in the Greater Atlantic Region. Working Paper #1. Technical report, NOAA/NMFS/GARFO, 2016. URL https://www.greateratlantic.fisheries.noaa.gov/aps/discard/review/index.html.

NEFSC. 59th Northeast Regional Stock Assessment Workshop (59th SAW) Assessment Summary Report. Technical Report 14-07, National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, 2014. URL http://nefsc.noaa.gov/publications/.

W.G. Cochran. *Sampling techniques (3rd edition)*. John Wiley and Sons, New York, 1977. URL http://www.wiley.com/WileyCDA/WileyTitle/productCd-047116240X.html.

APPENDIX A5: Updated selectivity of large camera data from the SMAST drop camera survey using digital still images

N.David Bethoney, Kevin D. E. Stokesbury (University of Massachusetts Dartmouth School of Marine Science and Technology, New Bedford, MA) and Larry D. Jacobson (Northeast Fisheries Science Center, Woods Hole, MA)

Summary

Selectivity curves were estimated for sea scallops in the largest camera view of the SMAST drop camera survey using the Millar's maximum likelihood SELECT model. Digital still camera images nested within largest camera view and provided an order of magnitude higher resolution were used as the control data from 2008-2011. Due to differences in quadrat sizes a split parameter and an adjustment for different measurement probabilities were used to make measurement counts comparable. Selectivity curves were similar in both the Mid-Atlantic and Georges Bank as both regions showed a similar pattern of selectivity when all years were combined or averaged with L50s for the large camera near 55 mm shell height and L95s near 75 mm. These results were similar to 2003-2006 results using a different control camera and combined data from the Mid-Atlantic and Georges Bank to estimated L50 at 48 mm and L95 at 79 mm.

Introduction

The purpose of this analysis was to update the selectivity curve produced as part of the 45th stock assessment workshop that identified the proportions of scallops at different shell heights detected in the largest camera view (large camera) of the SMAST drop camera survey. The selectivity curve converts the relative estimates of scallops at smaller sizes (35 - 80 mm shell height) provided by the large camera to "true" estimates for the size-structured estimation model (CASA) used for estimation of scallop biomass, fishing mortality and recruitment. The previous curve was developed using the Millar's maximum likelihood SELECT model (Millar and Fryer, 1999) to compare paired data from the large and smallest camera views (small camera) from Georges Bank and the Mid-Atlantic Bight area combined during 2003-2006 (Marino et al. 2007). The large and small cameras were the same type, but one was mounted closer to the sea-floor and therefore assumed to be 100% selective at 35+mm shell heights allowing data from the small camera to be used as the control (Figure 1). In this analysis we developed separate SELECT model curves for Georges Bank and the Mid-Atlantic Bight area using digital still images from 2008-2011 as the control. The digital still images represent a better control dataset as they are an order of magnitude higher in resolution, have reduced spherical distortion and overlap more with the large camera view (Figure 1, Table 1, Carey and Stokesbury 2011). Further, the digital still camera used from 2008-2011 provided similar resolution images as the current cameras used as the primary sampling units after 2016 (Table 1). Therefore, these curves can be used to adjust data from the large camera, so that it is comparable to data from digital still images or vise versa. Additionally, this updated analysis quantified and incorporated the difference in measurement bias from utilizing different quadrat sizes.

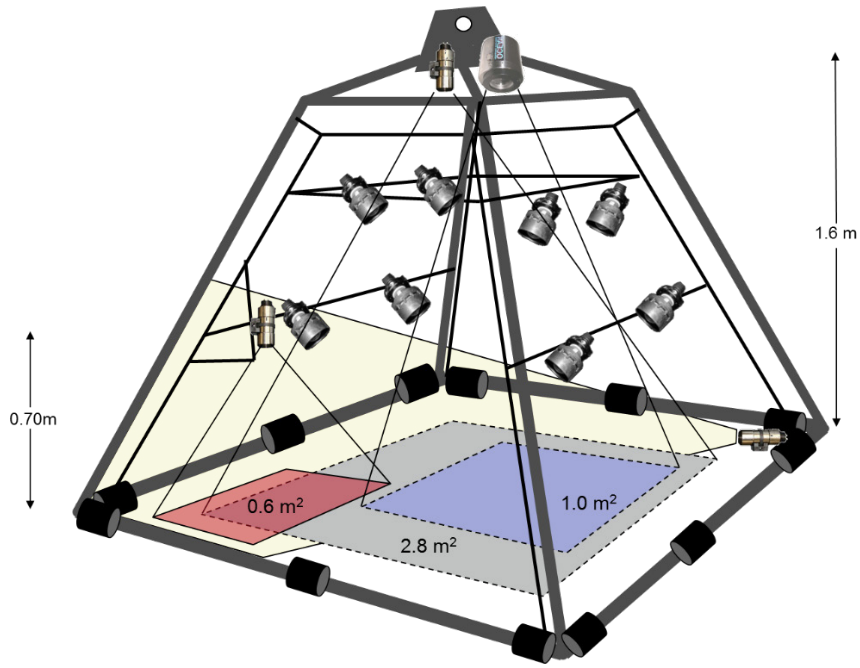


Figure 1. Drop camera pyramid design from 2003 to 2011. Live-feed, analog video cameras provided the largest (grey) and smallest (red) quadrat views, while a digital still camera provided the 1 m^2 quadrat (blue). Eight lights and a third video camera that provided a view parallel to the seafloor are also shown.

Table 1. Time periods and resolution of cameras used on the drop camera survey. The large and small cameras are the same type of video camera (DeepSea MultiSeaCam), but the large is mounted higher on the sampling pyramid resulting in a larger view area and lower resolution. The other three cameras all provided high resolution digital still images.

Camera	View Area (m^2)	Verticle mm/pixel	Hotizontal mm/pixel	Years Used
Large	2.8	2.98	3.10	1999-2016
Small	0.6	1.35	1.43	2000-2016
Nikon	1.0	0.32	0.32	2007-2013
Kongsberg	1.7	0.41	0.41	2015-2017
Imperx	2.4	0.29	0.29	2017-2018

Methods

Selectivity comparisons were based on shell height measurements from the large and Nikon digital still camera images from 2008-2011 (Figure 2). Only stations where measurement data was available for both cameras and all four quadrats were used. Stations in U.S. waters with shell height measurements from both the Kongsberg and large camera were limited and non-existent for the Imperx camera. Similarly, scallops from images taken by the Nikon camera were only counted in 2012 and 2013 and measurements in 2007 were limited to one area of the Mid-Atlantic. In addition to using a new digital image dataset, this analysis incorporated the difference in measurement bias from utilizing different quadrat sizes. To eliminate uncertainties caused by guessing and converting shell widths to shell heights, only clearly visible scallops with their umbo and anti-umbo in the field of view are measured in images processed by SMAST. The larger a scallop the more likely the umbo or anti-umbo is outside the field of view, decreasing its probability to be measured. The effect is relative to quadrat size, decreasing as the quadrat size increases. To quantify this effect, an R code was written to draw the field of view for each camera, drop 1,000 scallops of each size bin into the view area, and quantify the proportion measured. The results of this simulation cannot be used to represent the absolute percentages of scallops measured at each size or as an adjustment for survey results as it does not account for competing biases, such as smaller scallops being more likely be covered, but can be used for relative comparisons between each camera. Therefore for this analysis, scallop measurements in the digital still images were increased by the estimated proportion of scallops measured in the large camera quadrat size divided by the estimated proportion of scallops measured in the Nikon digital still camera quadrat size for each size bin.

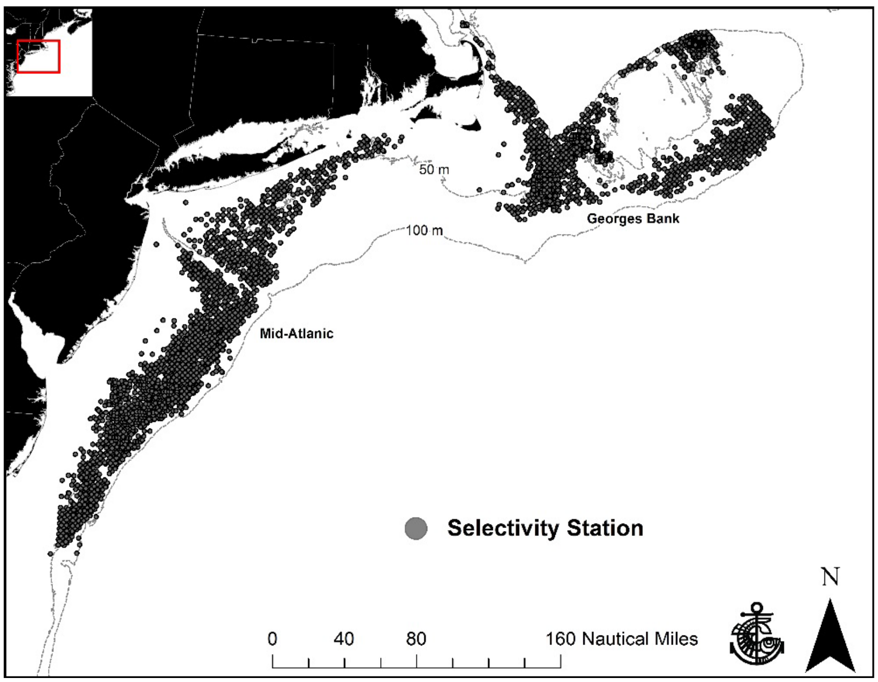


Figure 2. Spatial distribution of SMAST surveys stations used for selectivity analysis of large camera data

Millar's SELECT model was used to fit an increasing logistic shape curve of selectivity for the large camera using the digital still camera as a standard. The model is:

$$s(L) = \frac{p * e^{a+bL}}{(1-p) + e^{a+bL}}$$

where *sL* is selectivity at length and *a* and *b* are parameters fitting the logistic curve (Millar and Fryer, 1999). A third "split" parameter *p* represents relative sampling intensity between the two cameras due to the different view areas (Figure 1) and was initially estimated by taking the average of the ratio of the sample in the large camera to the total sample (large / large + digital still) at each shell height bin. Following Marino et al. 2007, shell height measurements were binned in 10 mm increments to minimize potential effects of imprecise shell height measurements. Increment mid-points were used in all calculations (e.g. 25 mm for the 20-29.99 mm bin). The model was used to estimate the shell heights with selectivity values of 50% (L50), 90% (L90) and 95% (L95) as well as the selectivity range (SR = L75–L25) for each year, the average of each and all years combined in the Mid-Atlantic and Georges Bank.

Results and Discussion

The estimated proportion of with their umbo and anti-umbo within a quadrat view area decreased with scallop size and the effect was greater as quadrat size decreased (Table 2). These results were used to produce an adjustment factor for each scallop size bin used in the selectivity analysis (Table 3). Overall, these adjustment increased the amount of scallop measurements for the digital still images by 5% in the Mid-Atlantic and Georges Bank.

Table 2. The proportion of scallops by size for which the umbo and anti-umbo are within camera view areas based on a simulation for 1,000 scallop drops in each size bin and area.

	Camera and View Area (m^2)				
Size (mm)	**Large (2.8)**	**Imperx (2.4)**	**Kongsberg (1.7)**	**Nikon (1.0)**	**Small (0.6)**
10	0.99	0.99	0.99	0.99	0.99
20	0.98	0.98	0.98	0.97	0.97
30	0.98	0.97	0.97	0.96	0.95
40	0.97	0.97	0.96	0.95	0.93
50	0.96	0.96	0.95	0.93	0.92
60	0.95	0.95	0.94	0.93	0.90
70	0.95	0.94	0.93	0.91	0.88
80	0.94	0.94	0.92	0.90	0.87
90	0.93	0.92	0.91	0.89	0.86
100	0.93	0.92	0.90	0.87	0.84
110	0.92	0.91	0.89	0.86	0.82
120	0.91	0.90	0.88	0.85	0.80
130	0.90	0.89	0.87	0.84	0.79
140	0.89	0.89	0.86	0.83	0.78
150	0.89	0.88	0.86	0.82	0.76
160	0.88	0.86	0.85	0.81	0.74
170	0.87	0.86	0.83	0.79	0.73
180	0.86	0.86	0.83	0.78	0.71
190	0.86	0.84	0.82	0.78	0.70
200	0.85	0.83	0.81	0.76	0.69

Table 3. Multiplier, by size bin, to account for the difference in measurement bias in a 1.0 m^2 quadrat in comparison to a 2.8 m^2 quadrat.

Size (mm)	Multiplier
15	1.00
25	1.01
35	1.02
45	1.02
55	1.03
65	1.03
75	1.04
85	1.04
95	1.05
105	1.06
115	1.07
125	1.07
135	1.07
145	1.08
155	1.09
165	1.09
175	1.10

Selectivity curves were similar in both the Mid-Atlantic and Georges Bank and to previous analysis. Both regions showed a similar pattern of selectivity when all years were combined or averaged with L50s for the large camera near 55 mm shell height and L95s near 75 mm (Table 4). This is very similar to 2003-2006 results using the small camera that combined data from the Mid-Atlantic and Georges Bank and estimated L50 at 48 mm and L95 at 79 mm (Marino et al. 2007). There was some between year variations, but considering shell heights were binned in 10 mm intervals, results were generally consistent. Between year variations was also much lower compared to the previous analysis where L95s ranges from 64 mm to 103 mm. Deviance residuals indicate the best model fit when all years are combined, suggesting smaller sample sizes within years may be contributing to variation.

References

Carey JD, Stokesbury KDE (2011) An assessment of juvenile and adult sea scallop, *Placopecten magellanicus*, distribution in the northwest Atlantic using high-resolution still imagery. J Shellfish Res 30: 569-582.

Marino II MC, O'Keefe CE, Jacobson LD (2007). Selectivity and efficiency of large camera video data from the SMAST video survey during 2003 – 2006. Appendix B7 to NEFSC SAW 45 Northeast Fisheries Science Center, Woods Hole, MA

Millar RB, Fryer RJ (1999) Estimating the size-selection curves of towed gears, traps, nets and hooks. Reviews in Fish Biology and Fisheries,9(1), 89-116.

Table 4. Estimated selectivity parameters *p* (split), *a*, *b*, *L95*, *L90*, *L50* and SR with standard errors and variances from SELECT models fit to large and digital still camera data collected during 2008-2011 in the Mid-Atlantic and on Georges Bank

Year	Split (%)	SE(Split)	L95 (mm)	L90 (mm)	L50 (mm)	SE(L)	SR	SE(SR)	a	SE(a)	b	SE(b)	Large n	DSC n
Mid Atlanic														
2008	71.3	0.011	78	72	51	2.25	21	3.96	-5.49	0.95	0.11	0.01	1,511	724
2009	73.8	0.008	71	67	54	2.15	13	2.54	-9.46	1.70	0.18	0.04	2,441	899
2010	68.3	0.012	84	79	64	2.67	15	3.13	-9.64	1.76	0.15	0.01	1,228	649
2011	69.6	0.013	74	71	60	2.73	11	3.14	-12.19	3.16	0.20	0.06	871	421
Average 2008-2011	70.8	0.011	77	72	57	2.45	15	3.19	-9.20	1.89	0.16	0.03		
2008-2011	71.3	0.005	77	72	55	1.20	17	1.71	-7.26	0.65	0.13	0.01	6,051	2,693
Georges Bank														
Year	Split (%)	SE(Split)	L95 (mm)	L90 (mm)	L50 (mm)	SE(L)	SR	SE(SR)	a	SE(a)	b	SE(b)	Large n	DSC n
2008	60.9	0.012	65	61	49	1.79	12	2.24	-8.84	1.41	0.18	0.03	1,144	861
2009	74.4	0.007	70	65	51	1.46	14	1.99	-7.73	0.95	0.15	0.02	3,276	1,252
2010	77.4	0.009	69	67	57	2.57	9	2.22	-14.17	3.09	0.25	0.06	1,826	569
2011	69.1	0.009	89	80	54	3.04	26	4.37	-4.52	0.61	0.08	0.01	2,479	1,221
Average 2008-2011	70.5	0.009	73	68	53	2.22	15	2.71	-8.82	1.52	0.17	0.03		
2008-2011	71.4	0.004	75	69	53	1.04	17	1.32	-7.01	0.47	0.13	0.01	8,725	3,903

Appendix A6. Technical documentation for the CASA length structured stock assessment model used in the SARC-59 sea scallop stock assessment.

Larry Jacobson and Jui-Han Chang
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.[1] 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.).

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 $n_L = 1 + \text{int}\left[(L_\infty - L_{\min})/L_{bin}\right]$, 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).

[1] 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.

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 ***P₀****(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).[2] 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:

[2] 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 year 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 selectivity[3] 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.[4] 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).[5] 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:

$$M_{L,y} = J\, u_y \alpha_L + A\, v_y (1 - \alpha_L)$$

where J and A are the mean natural mortality parameters for all years (y) and sizes (L) for juveniles and adults, and u and v are the year-specific deviation parameters for juveniles and adults, respectively, and α_L is an estimable descending logistic function based on

---

[3] 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.

[4] 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.

[5] . See the section on per recruit modeling below for formulas used to relate catch, landings and indicental mortality.

size. The α_L are calculated as

$$\alpha_L = 1 - \frac{1}{1 + e^{-b(L-a)}}$$

where b is the slope parameter and a is the L_{50} parameter. The α_L is used to partition juveniles and adults, thus determining whether how the natural mortality changes with size. Even though the logistic partition is descending, it is not necessary that the juvenile natural mortalities will be larger than the adult natural mortalities. If J is smaller than A, the juvenile natural mortalities will be smaller than the adult natural mortalities, and vice versa. If J equals to A, the natural mortality will be the same for all sizes. If the logistic partition is flat at zero, the estimation of natural mortality is the same as previous assessments, which only one mean natural mortality parameter during all years for all sizes, along with one year-specific dev parameter are used (NEFSC 2014).

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.[6] 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.

Recruitment (the sum of new recruits in all length bins) at the beginning of each year after the first is calculated:

$$Ry = e^{\rho + \gamma_y}$$

[6] The model will be modified in future to model discards and landing separately, and to use size composition data for discards.

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 $\neq 1$ 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.[7] 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 length[8]

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:

---

[7] 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.

[8] 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. δ_Y=ln[$D_Y/(1\text{-}D_y)$]) 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(σ) 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.[9] 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.

---

[9] 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.

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 $d_f \geq 30$. 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:

$$L = \ln\left(1 + \frac{t^2}{d_f}\right)\left(1 - \frac{1 - d_f}{2}\right) - \frac{\ln(d_f)}{2}$$

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 $r_Y = (C_Y - \hat{C}_Y)/\sigma_Y$.

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 $(\hat{c}_Y)$ 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:

$$q_k = e^{\frac{\sum_{i=1}^{N_k}\left[\ln\left(\frac{I_{k,i}}{A_{k,i}}\right) \Big/ \sigma_{k,i}^2\right]}{\sum_{j=1}^{N_k}\left(1 \big/ \sigma_{k,j}^2\right)}}$$

where N_v and $\sigma_{k,j}^2$ is the log scale variance corresponding to the assumed CV for the survey observation.[10]

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:

$$A_{k,y} = \sum_{L=1}^{n_L} s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}$$

where $\overline{N}_{y,L}$ is average abundance during the user-specified period of availability and $\tilde{M}_{y,L}$ is the instantaneous rate of natural mortality for the period of availability. Average abundance during the period of availability is:

$$\overline{N}_{y,L} = \frac{\tilde{N}_{y,L}\left(1 - e^{-\tilde{Z}_{y,L}}\right)}{\tilde{Z}_{y,L}}$$

where $\tilde{N}_{y,L} = N_{y,L} e^{-Z\Delta}$ 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 $\tilde{Z}_{y,L} = Z_{y,L}\left(\tau_{k,y} - \nu_k\right)$ and natural $\tilde{M}_{y,L} = M_{y,L}\left(\tau_{k,y} - \nu_k\right)$ mortality are also adjusted to correspond to the period of availability. In using this approach, the user

[10] 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.

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.

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 $\geq L_{bin}$ 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 $\geq L_{min}$ 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 $\leq$ 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 $\leq$ 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:

$$A_{k,y} = \frac{s_{k,L}\widetilde{M}_{y,L}\overline{N}_{y,L}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{Last}} s_{k,L}\widetilde{M}_{y,L}\overline{N}_{y,L}}$$

Considering the possibility of structured measurement errors, the expected length composition $\vec{A}'_{k,y}$ for survey catches is:

$$\vec{A}'_{k,y} = \vec{A}_{k,y} \text{Ⱡ}_k$$

where Ⱡ_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 Ⱡ_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 Ⱡ_k add to one to account for all possible outcomes in assigning individuals to observed length bins. Ⱡ_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 $\left(\gamma_y - \gamma_{y-1} \right) / \sigma_\gamma$ 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.[11] 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

[11] D. Hart, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA, pers. comm.

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 $\sigma_y = CV_y \widehat{L}_y$. Standardized residuals are $r_y = \left(L_y - \hat{L}_y\right)/\sigma_y$.

Per recruit (SBR and YPR) reference points[12]

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

[12] 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.

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 $\mathcal{P}$. 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 models estimated growth matrix $\mathcal{P}$. 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. λ=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:

$$d_t = \left| \begin{array}{l} Ft - \Phi \; if \; Ft > \Phi \\ 0 \; otherwise \end{array} \right.$$

and

$$q_t = \left| \begin{array}{l} \ln(Ft/\Phi) \; if \; Ft > \Phi \\ 0 \; otherwise \end{array} \right.$$

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).

References

J.L. Butler, L.D. Jacobson, J.T. Barnes, and H.G. Moser. 2003. Biology and population dynamics of cowcod (*Sebastes levis*) in the southern California Bight. Fish. Bull. 101: 260-280.

Fournier, D., and Archibald, CP. 1982. General theory for analyzing catch at age data. Can. J. Fish. Aquat. Sci. 39: 1195-1207.

Jacobson, L.D., Cadrin, S.X., and J.R. Weinberg. 2002. Tools for estimating surplus production and F_{MSY} in any stock assessment model. N. Am. J. Fish. Mgmt. 22: 326-338.

Methot, R. D. 2000. Technical description of the stock synthesis assessment program. NOAA Tech. Memo. NMFS-NWFSC-43: 1-46.

Pennington, M., Burmeister, L-M., and Hjellvik, V. 2002. Assessing the precision of frequency distributions estimated from trawl-survey samples. Fish. Bull. 100: 74-80.

Press, W.H., Flannery, B.P., S.A. Teukolsky, and W.T. Vetterling. 1990. Numerical recipes. Cambridge Univ. Press, NY.

Sullivan, P.J., Lai, H.L., and Gallucci, V.F. 1990. A catch-at-length analysis that incorporates a stochastic model of growth. Can. J. Fish. Aquat. Sci. 47: 184-198.

Appendix A7 - Forecasting methodology (SAMS model)

Deborah R. Hart,
Northeast Fisheries Science Center, 166 Water St., Woods Hole, MA 02536

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 five open areas (the Great South Channel, Northern Edge, Southern Flank, Closed Area II Extension, and Nantucket Lightship Extension), three no access portions of the groundfish closed areas (for the years 2017 and before), the access portions of Closed Areas I and II, and three portions of the Nantucket Lightship Closed Area access area: north, south shallow, and south deep ("Peter Pans"). The Mid-Atlantic was subdivided into seven areas: Virginia Beach, the Delmarva, Elephant Trunk Flex, Elephant Trunk Open, and Hudson Canyon South Rotational Areas, New York Bight, Inshore New York Bight, and Long Island, which are all "open" areas (see Figure A12-1).

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:

$$F_i = k{*}f_i{*}L_i$$

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 $F_I = 0.1F_L$. For the Mid-Atlantic (almost all sand), we used $F_I = 0.05F_L$. 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 2001-2017 (see Appendix A1). Recruitment was modeled stochastically, and was assumed to be log-normal in each subarea. As a first step, 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 1979-2017. For the first time in this assessment, recruitment was also scaled to a region-wide Beverton-Holt stock recruitment relationship, making the SAMS model more comparable to the SYM reference points model. Thus, the recruitment as previously estimated is multiplied by $sr(B_r)/sr(B_{mean})$, where sr is the (mean) regional stock recruit relationship (for two years old) used in the SYM model, B_r is the current regional biomass, and B_{mean} is the mean regional biomass. Because of this addition, estimated recruitment for the projections presented here increased by about 20%, due to the high current biomass.

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. Adult natural mortality for these runs was set at 0.2 for the Georges Bank areas, and 0.25 for the Mid-Atlantic. Juvenile natural mortality was calculated as a piecewise linear function of density in that subarea, based on CASA model runs, which suggest that juvenile natural mortality is greater at high densities. For the Mid-Atlantic, juvenile natural mortality was M_J=0.25 except when juvenile density (as measured by dredge units for scallops 40-75 mm) was above 261 scallops per tow (approximately the 90th percentile of observations), above which juvenile natural mortality is calculated as MJ = M_0 + m (D_J - D_0), where M_0 = 0.25, m = 0.0001495, D_J is the juvenile density in the subarea (in dredge survey units), and D0 = 261. These parameters were estimated based on CASA estimated juvenile natural mortality, and observed dredge survey densities in the Elephant Trunk region, where most of the density dependent natural mortality probably occurred. The evidence for density dependent juvenile M on Georges Bank is less clear; it was modeled the same way, but with M_0 = 0.2, m = 0.0001, and D_0 = 500/tow, making the effects of density dependence weaker than in the Mid-Atlantic.

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 *i*th 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 A2 for details. For calculating biomass, the shell height of a size class was taken as its midpoint.

LPUE is estimated using a linear regression relating exploitable biomass to LPUE. In the model, the regression is applied to projected exploitable biomass by area to predict LPUE (Figure A12-2).

Initial conditions for the population vector **p** *(i,t)* were estimated using the 2017 surveys. The 2017 initial conditions were bootstrapped depending on the survey standard errors in each subarea.

Appendix A8: Factors influencing scallop landings per unit effort (LPUE)

Brooke L. Wright[1], Catherine E. O'Keefe[2], Benjamin Galuardi[3], Jonathon Peros[4], Steven X. Cadrin[1]

[1]School for Marine Science and Technology, University of Massachusetts Dartmouth, New Bedford, MA 02744
[2]Massachusetts Division of Marine Fisheries, New Bedford, MA 02744
[3]Greater Atlantic Regional Fisheries Office, National Marine Fisheries Service, Gloucester, MA 01930
[4]New England Fishery Management Council, Newburyport, MA 01950

Abstract

Accurate projections of landings per unit effort (LPUE) are important for management of the sea scallop fishery because Days-at-Sea (DAS) allocations are determined by the projected LPUE. In consultation with fishermen for their insight on potential factors that influence catch and effort, we developed generalized linear models (GLMs) to estimate standardized LPUE as a function of fishing behavior and environmental factors, with a focus on the DAS managed areas. Our models used data from vessel trip reports, dealer, and observer reports from 2007 to 2016 – the period of spatial management in the fishery for which overlapping data from the three sources are available. The selected model of LPUE included significant effects for year (peaking in 2011), month (generally peaking in April), region, price of the 10-20 count market category (generally decreasing with price), permit type, region of sale port, and proportion of the <10 count market category in landings. Performance of the model for forecasting LPUE was based on comparison of retrospective projections to the realized LPUE for 2010 to 2016. Despite some relatively naïve assumptions in the retrospective evaluation (e.g., status quo conditions) and some changes in open area management that would have been anticipated, the average absolute value of prediction error in the LPUE model is comparable to the method that has been used for management, and the projected LPUE was substantially closer to observed LPUE in the most recent two years of comparison, but projected LPUE may be substantially below actual LPUE in 2017. With further development and refinement, this tool could be used with expert judgment to provide an alternate approach for projecting LPUE.

Introduction

Atlantic sea scallops have been managed in the US since 1982 with the establishment of the Scallop Fishery Management Plan (NEFMC 1982). In 1994, Amendment 4 implemented a limited access program to prevent new vessels from entering the fishery (NEFMC 1993). The Limited Access vessels were granted full-time, part-time, or occasional permits, each category with a different DAS allocation. Framework Adjustment 11 granted access for the scallop fleet into portions of the Georges Bank groundfish closed areas (NEFMC 1999), and Amendment 10 instituted formal rotational area management in 2004 (NEFMC 2003). Under this system, there are four types of areas within the scallop resource: 1) "open areas" where scallop fishing can occur using DAS or quota; 2) areas completely closed to scallop fishing year round to reduce impacts on essential fish habitat or groundfish mortality; 3) areas temporarily closed to scallop vessels to protect small scallops until a future date; and 4) areas open to restricted levels of scallop fishing called "access areas" (NEFMC 2014). The DAS allocations are determined by dividing the target open area landings by the LPUE estimate, as constrained by a fishing mortality level decided by the New England Fishery Management Council.

The Scallop Area Management Simulator (SAMS) model used in scallop assessments has been used to estimate LPUE (in number of scallops per day at sea) according to a modified predator-prey model, in which LPUE is modeled according to a saturating function of mean survey exploitable numbers of scallops (NEFSC 2014):

(1)
$$LPUE = \frac{\alpha N}{\sqrt{\beta^2 + N^2}}$$

where N is the exploitable number of scallops and α and β are parameters fit to the relationship between fleet-wide LPUE and exploitable biomass from 1994-2012 (NEFSC 2014). This form is based on the similarity between harvest dynamics and a Holling Type-II predator-prey relationship: at high biomass levels, scallops are caught faster than the crew can process them, much like the time required for a predator to handle and consume its prey limits the consumption rate (NEFSC 2014). In this model, LPUE is influenced only by exploitable abundance and processing time.

In recent years, LPUE was overestimated, especially in 2014 (Figure 1a). The model incorporated information dating back to the 1990s, when fishing behavior was different than the current practices employed since rotational management was introduced. The different behaviors resulted in a non-linearity between estimates of exploitable abundance and LPUE (Figure 1a).

Since the implementation of Amendment 10 in 2004, fishing practices have changed and factors other than processing time are likely influencing LPUE. For this reason, the LPUE model was changed to a more simplistic linear regression in 2016. The linear regression conforms to the more commonly assumed relationship between LPUE and the exploitable stock, and assumes constant catchability (q) in which landings (L) can be expressed as the product of fishing mortality (F) and average exploitable biomass during the period ($\bar{N}$):

(2)
$$L_t = F_t\bar{N}_t = qE_t\bar{N}_t$$

LPUE is typically assumed to be proportional to the exploitable stock:

(3)
$$\frac{L_t}{E_t} = q\bar{N}_t$$

The same relationships can be used to equate catch biomass, LPUE in lb or kg, and exploitable biomass, so that target DAS can also be expressed as the target landings (e.g., Annual Projected Landings, APL) divided by LPUE:

(4)
$$E_{target} = \frac{qE_t\bar{N}_t}{q\bar{N}_t} = \frac{L_{target}}{(L_t/E_t)}$$

Similarly, fishing mortality (F) during a time period (t) is typically considered to be directly proportional to fishing effort (E), assuming constant catchability (q) of a unit of fishing effort:

(5) $$F_t = qE_t$$

Therefore, a DAS allocation can also be derived based on a target F and an estimate of q:

(6) $$E_{target} = \frac{F_{target}}{\hat{q}}$$

These relationships implicitly assume constant catchability (q) of a unit of fishing effort. However, Harley et al. (2001) showed catch per unit effort (CPUE) is not always proportional to abundance. The curvilinear relationship between LPUE and exploitable stock implied by the saturated catch rate assumed in SAMS (Figure 1a) can be accounted for by assuming 'hyperstability':

(7) $$\frac{L_t}{E_t} = q\left(\bar{N}_t^b\right)$$

Where b>1 (Hilborn & Walters 1992).

Only information from 2004 onwards, after full implementation of rotational management, was included in the updated linear regression LPUE model, and the model results show no evidence of non-linearity between LPUE and exploitable biomass (Figure 2). However, other factors can influence catch, and a model that treats them explicitly may be more flexible and accurate, as well as have a more transparent interpretation. Despite the better model fit to data from 2004 through 2015, the 2014 LPUE estimate is still considered an outlier (overestimate), as well as the 2008 estimate (underestimate; Figure 1b).

Changes in the fishing fleet (e.g., vessel characteristics, fishing gear, technology), fishing behavior (location, season), management constraints and market conditions can influence catchability (q). Fishing effort can be statistically standardized by estimating factors of LPUE. LPUE or CPUE are often standardized using GLMs or other methods to account for factors affecting catch rates (e.g. Maunder and Punt 2004). For example, Atlantic blue marlin CPUE is influenced by quarter, area, and gear type (Nakano et al. 1994). Similarly, Glazer and Butterworth (2002) found South African west coast hake CPUE is affected by season, depth, latitude, vessel, and CPUE of bycatch species, while Battaile et al. (2004) found CPUE of walleye Pollock in the Bering Sea varies by individual vessel, vessel speed, percentage of pollock in the catch, and day or night.

LPUE is typically standardized using generalized linear, additive, or mixed models of expected LPUE observations ($LPUE_i$) based on a set of explanatory factors (X to P) in which the year effect is used as a standardized index of relative stock size (Maunder & Punt 2004):

(8) $$\widehat{LPUE_i} = \beta_0 + \beta_x X_i + \cdots + \beta_p P_i + \beta_{Year} Year$$

Scallop LPUE may be influenced by several factors, including rotational harvest management, steam time, fuel costs, crew size, bycatch, and time and area fished. DAS are counted as the time between crossing the VMS demarcation line on the way to and from fishing grounds. Time spent steaming, searching for scallops, and other non-fishing activities beyond the demarcation line are included in the DAS used by each vessel. Considering these factors in addition to exploitable biomass may provide more accurate estimates of LPUE, resulting in reduced management uncertainty associated with DAS specifications.

Methods

We met with fishermen in New Bedford, MA in May 2017 to identify factors that affect catch rates. We began the meeting with a description of the project then asked open-ended questions and encouraged discussion to discover what patterns of behavioral, environmental, market, and management conditions we should consider in building models of LPUE and fishing behavior. We also solicited input from captains from Barnegat Light, NJ via written surveys. Responses were variable, but a number of themes were repeated by multiple participants: price, size/yield, gear, skill/experience of captain and crew, time of year, home port proximity to access areas, and clean catches. The responses from the meeting and surveys largely informed our data request.

Data

We obtained data from the Greater Atlantic Regional Fisheries Office (GARFO) from the Data Matching and Imputation System (DMIS) which combines logbook, electronic Vessel Trip Report (eVTR), and dealer data for all scallop trips from 2007 through 2016 (n = 13,837 trips). Trips that were reported in statistical areas that have little to no overlap with SAMS areas were omitted (Figure 2). Similarly trips with unknown permit type, unknown or unclassified market category, unknown port of sale, missing trip length, or with LPUE > 10,000 pounds were omitted, leaving 12,375 trips retained in the analysis.

LPUE appears to be approximately log-normally distributed (Figure 3), but we attempted several alternative distributional assumptions to evaluate fit. Few trips were longer than 16 days. There were approximately 1,300 trips per year, 77% of which were completed by large vessels with full time permits, 16% by small vessels with full time permits, 4% by trawl vessels with full time permits, 6% by small vessels with part time permits, and <1% by large vessels with part time permits.

Initial exploration of the data indicated that LPUE trends were generally similar across years (Figure 4) and across most vessel permit types (Figure 5). The pattern of LPUE relative to price of the 10-20 count/lb market category (which we used as an indicator of price trends) was similar across vessel size and vessel permit type. LPUE increased monotonically with the percent of landings comprised of 10-20s. However, LPUE peaked when the <10 count/lb market category (U10s) are about 40% of landings then decreases, so we considered a square term of U10 proportion of landings. The decrease in LPUE at high proportions of U10s may result from increased time spent targeting large scallops with less total time spent fishing and processing scallops. Appendix 1 contains figures documenting our data exploration.

Statistical Analyses

We used generalized linear models (GLMs) to develop a standardized series of LPUE based on the factors determined through meetings with fishermen. We evaluated several distributional assumptions, including lognormal, Gamma, negative binomial, and Poisson. The standardization has form $\widehat{U}_{std,t} = f(Year + X_1 + X_2 + \cdots)$ where $U_{std,t}$ is the standardized LPUE in year t and

X_p are explanatory variables such as location, permit type, etc. We built models in statistical software R version 3.3.2 using forward and backward step-wise selection of explanatory variables, which selects an optimal model based on AIC. We modified the selection algorithm when collinear or otherwise repetitive variables were selected (e.g. home port *and* principle port, or proportion of 10-20s *and* proportion of U10s). Among collinear variables, we retained the variable with the lowest AIC. Among similar categorical variables, selection was more subjective. We often retained the most parsimonious variable (e.g. a coarser spatial aggregation) or a variable with a more straightforward explanation (e.g. sale port vs. home port) when the AIC score is higher but there is little change in the percent deviance explained.

Our models considered vessel characteristics (size, gear or permit), month, location (statistical area, approximate SAMS area, region), price indicator, home port and sale port, and percentage of landings comprised of desirable market categories (U10s and 10-20s; Tables 1 and 2). We included month as a categorical variable rather than fitting a cyclic spline because the start of the fishing year in March has notably different fishing patterns than the end of the fishing year in February. We considered several metrics to indicate price: monthly average of 10-20 market category price, trip-based price of 10-20s, trip-based price of 10-20s relative to annual average price of 10-20s, trip based total price, and trip based total price relative to annual average price (to account for the upward shift in price over time, Figure 6).

In addition to model diagnostics, we selected a model based on 10-fold cross validation. The data were systematically divided into ten test sets, and the model was tested by excluding one set at a time from fitting the models then predicting values for the excluded test set. We compared the average mean absolute prediction error (MAPE) and percent deviance explained from the ten runs. For each run, MAPE was the average absolute difference between predicted values for the test set and the actual response values in the test set.

$$MAPE = \frac{\sum_{i=1}^{n} abs(y_i - \hat{y}_i)}{n} \qquad (9)$$

Retrospective LPUE Projections

To test performance of our selected model, we performed retrospective projections for years 2010-2017. For each projection, we consider the “current year” to be the year in which projections are being made for management decisions for the following year. The “projection year” is the year for which projections are being made (i.e. in 2016 the projection year is 2017). “Last year” is the year prior to the current year and the last year for which there is complete fishery and survey data available at the time of management considerations.

We evaluated all possible combinations of factors to use in projections. For the two continuous variables, price and percent U10, we assigned the mean price by month in each year and the mean U10 percent of landings by month and fishing region in each year. For each year we calculated the proportion of trips that were taken in each condition set.

In each current year, we fit the selected model to data from years up to and including last year. We then used the values for last year to predict LPUE in each condition set in the projection year, using the year effect from the last year. Based on a status quo assumption of fishing behavior, we multiplied these predictions by the proportion of trips last year in each condition set and sum across all condition sets to determine the predicted LPUE.

Finally, we multiplied the predicted LPUE based on last year's assumptions by the ratio of exploitable biomass in the projection year to exploitable biomass in the last year. For 2013 – 2018 we used exploitable biomass estimates from the SAMS model as presented to the scallop Plan Development Team in each year, except for 2015, for which an estimate could not be found. In 2015 we used the average value of 2014 and 2016. For years prior to 2013, we used the exploitable biomass estimates from the SMAST drop camera survey.

Results

The selected GLM includes effects of fishing year, vessel permit type, month, region, trip-based price of 10-20s, region of sale, and proportion of landings comprised of U10s (Tables 2 and 3). Predicted LPUE increased from 2007 to 2011 followed by a decrease to 2015 (Figure 7). The best models have a prediction error of about 500 lbs/day (compared to the null model MAPE of about 800 lbs/day) on a trip by trip basis.

Predicted LPUE was greater for large vessels (Figure 8). Trawl vessels with full time permits had slightly lower predicted LPUE. Large vessels with part time permits tended to have slightly lower predicted LPUE, but the effect was non-significant (likely due to relatively few observations from these vessels). Small vessels (both full time and part time permits) had the lowest predicted LPUE. This effect may be particularly useful in improving DAS allocations, because LPUE is currently estimated in full-time equivalents.

Predicted LPUE was greatest early in the fishing year (March through May). LPUE was predicted to decrease through summer and fall, reaching a minimum in November. Predicted LPUE then increased through the end of the fishing year (Figure 9). The high LPUE in spring coincided with the pattern of shell height to meat weight ratio, with growth in the spring producing high meat yield.

Predicted LPUE was greatest in the Great South Channel (Figure 10). The predicted LPUE was not significantly different in the Mid Atlantic than on Georges Bank.

Predicted LPUE decreased with increasing price of 10-20 scallops (Figure 11). This pattern reflected fishermen's willingness to fish in less productive areas and land fewer scallops when the price was high because earnings were high even if landings were not.

The predicted LPUE was greatest for vessels that landed in New England (Figure 12). Although AIC was improved by considering landings by state, some states have a very low sample size. For clarity of interpretation as well as practicality in making projections, we aggregated into two regions, Mid Atlantic and New England. There was a negligible increase in prediction error using region landed instead of state landed (MAPE increased from 493lb/day to 494 lb/day).

The predicted LPUE decreased curvilinearly with an increasing proportion of U10s in the catch, (Figure 13). The more rapid decline at high proportions was presumably related to the value of large scallops: it was worthwhile to spend more time selecting for large scallops relative to time spent fishing, or to fish in areas where the scallops were less aggregated but individuals were larger. A smaller catch of large scallops can be more profitable than a large catch of small scallops.

We compared the parameter estimates for each projection year (Figure 16). The pattern of coefficients is similar across years. However, the magnitude of coefficients is considerably different in some years. The standardized LPUE estimates (i.e. the year effects) are quite consistent from the model for projection years 2011, 2012, 2014, 2017, and 2018. Projection years 2015 and 2016 estimate higher LPUE for the series, whereas projection years 2010 and 2013 estimate lower LPUE for the series. In light of the observed LPUE data, the low values estimated in projection year 2010 result from the model being fitted only to data from 2007-2008, which were both low LPUE years (Table 4). High values in projection years 2015-2016 are based on the dataset 2007 through 2013 and 2014, respectively. These comprise groups with the highest proportion of large observed LPUE. In addition to the patterns of annual aggregate LPUE, the yearly effects should be considered alongside the month effects and other effects (permit type, sale region, fishing regions, price of 10-20s, and percentage of U10s in landings). Month effects are consistent for projection years 2014-2018 with higher estimates in 2010-2013. The high values for month effects are coincident with the low values for year effects during the same projection years. Coefficients for permit types are generally lower in the first two to three projection years and consistent from 2013 onwards. Coefficients for remaining effects (sale region, fishing region, price, and percent U10s) are relatively consistent throughout the time series. These results suggest complicated interactions among factors, and it would be much more difficult to anticipate how the interactions can be considered in projection applications.

We assumed a lognormal distribution (i.e. Gaussian family with log link) when building our models based on the histogram of LPUE (Figure 2), but we also considered a Gamma distribution with inverse link, negative binomial distribution with log link, and Poisson distribution with log link (LPUE was rounded to the nearest whole number for the negative binomial and Poisson distributions). Diagnostic plots suggested the lognormal distribution was best suited to the data (Appendix 2). Histograms of residuals from models of all distributional assumptions showed long tails, but the lognormal model residuals most closely approximated a normal distribution, and the residuals showed less discernible pattern against fitted model values (Appendix 3).

We compared the projected LPUE in each year to the observed aggregate LPUE in each year (i.e. the total landings divided by the total effort reported for the year) and to LPUE estimates that were used in setting regulations (Table 4 and Figure 15). Performance is quite variable from year to year; absolute value of forecast error ranges from 37 to 1141 lbs/day. The average absolute value of forecast error is 473 lbs/day. By comparison, the range of forecast error for the LPUE estimates that have been used in management is 133 to 926 lbs/day with an average of 519 lbs/day (though we were not able to find the value used in management for 2012).

Poor performance of the 2011 projection (1141 lbs/day) may result from changes to the definition of open areas by management actions for that year. Projected LPUE was substantially better than the LPUE that was assumed for management in the last two years of the retrospective evaluation (2015 and 2016).

Discussion

Despite some relatively naïve assumptions in the retrospective performance evaluation of projected LPUE (e.g., status quo conditions) and some changes in open area management that would have been anticipated (e.g., 2011), the projected LPUE model performed comparably to the method that has been used for Days-at-Sea management. The projected LPUE was substantially closer to observed LPUE in the most recent two years in the evaluation (2015 and 2016), possibly resulting from a longer time series of LPUE observations in the model. We expect that, with further refinement, the accuracy of this tool could be improved and used with expert judgment to increase performance of projected LPUE.

Preliminary reports of LPUE in 2017 are extremely high (Figure 14), and this will need to be addressed in the 2018 assessment for effective management of the resource. Understanding drivers of LPUE from the present study may elucidate an explanation for the high observed LPUE. Our selected model incorporates factors that fishermen identified as important drivers of LPUE.

Our model estimates could be improved by using a more consistent exploitable biomass series. Estimates of exploitable biomass from SAMS projections were only available in years 2010, 2013-2014, and 2016-2018. For 2007-2012, we used estimates of exploitable biomass from the SMAST drop camera survey. The lack of consistency in this input may significantly contribute to the variability of our forecast error.

Additionally, standardizing the area of seafloor considered in the exploitable biomass estimates could substantially improve our LPUE projections. Management actions change the amount of seafloor that is considered Open Area fishing grounds in some years, which has ramifications for LPUE estimates because the perceived change in exploitable biomass is reflective of the change in area available to fishing, not only a change in resource abundance or density. For example, in 2011 the Elephant Trunk Access Area was converted to Open Area management, and there is approximately 67% increase in estimated exploitable biomass relative to 2009 (Table 5). Notably, our model's worst performance year is 2011.

Acknowledgements

Funding for this project was provided by the 2017 Scallop Research Set Aside program (17-SCA-24). Dvora Hart provided suggestions and feedback on this work. Sam Asci and Demet Haksever provided helpful insight and support. The Stock Assessment Workshop 65 Scallop Working Group provided useful feedback for the improvement of an earlier draft.

References

Galuardi B. 2017. 2017 Scallop Grades and LPUE Update. Presentation to Scallop Plan Development Team. 29-30 August 2017. Falmouth, MA (available at http://s3.amazonaws.com/nefmc.org/Doc-6.-galuardi_landings_grade_area_2017.pdf).

Glazer JP & Butterworth DS. 2002. GLM-based standardization of the catch per unit effort series for South African west coast hake, focusing on adjustments for targeting other species. South African Journal of Marine Science 24: 323-339.

Harley SJ, Myers RA, Dunn A. 2001. Is catch-per-unit-effort proportional to abundance? Canadian Journal of Fisheries and Aquatic Sciences 58: 1760-1772.

Hart D. 2016. LPUE Modeling. Presentation to the New England Fishery Management Council Scallop Plan Development Team. 30 August 2016, Falmouth, MA (available at http://s3.amazonaws.com/nefmc.org/Doc9.-LPUE-modeling.pdf).

Hilborn R, Walters CJ. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics, and Uncertainty. Chapman and Hall.

Maunder MN & Punt AE. 2004. Standardizing catch and effort data: A review of recent approaches. Fisheries Research 70(2): 141-159.
Nakano H, Uozumi Y, Honma M. 1994. The CPUE trend for Atlantic marlin caught by Japanese longline fishery. Collective Volume of Scientific Papers ICCAT 41: 408-424.
New England Fishery Management Council (NEFMC). 2014. Framework 25 to the Scallop FMP Including a Final Environmental Assessment (EA), an Initial Regulatory Flexibility Analysis and Stock Assessment and Fishery Evaluation (SAFE) Report. Newburyport (MA).
Northeast Fisheries Science Center (NEFSC). 2014. 59th Northeast Regional Stock Assessment Workshop (59th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 14-09; 782 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://nefsc.noaa.gov/publications/
R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

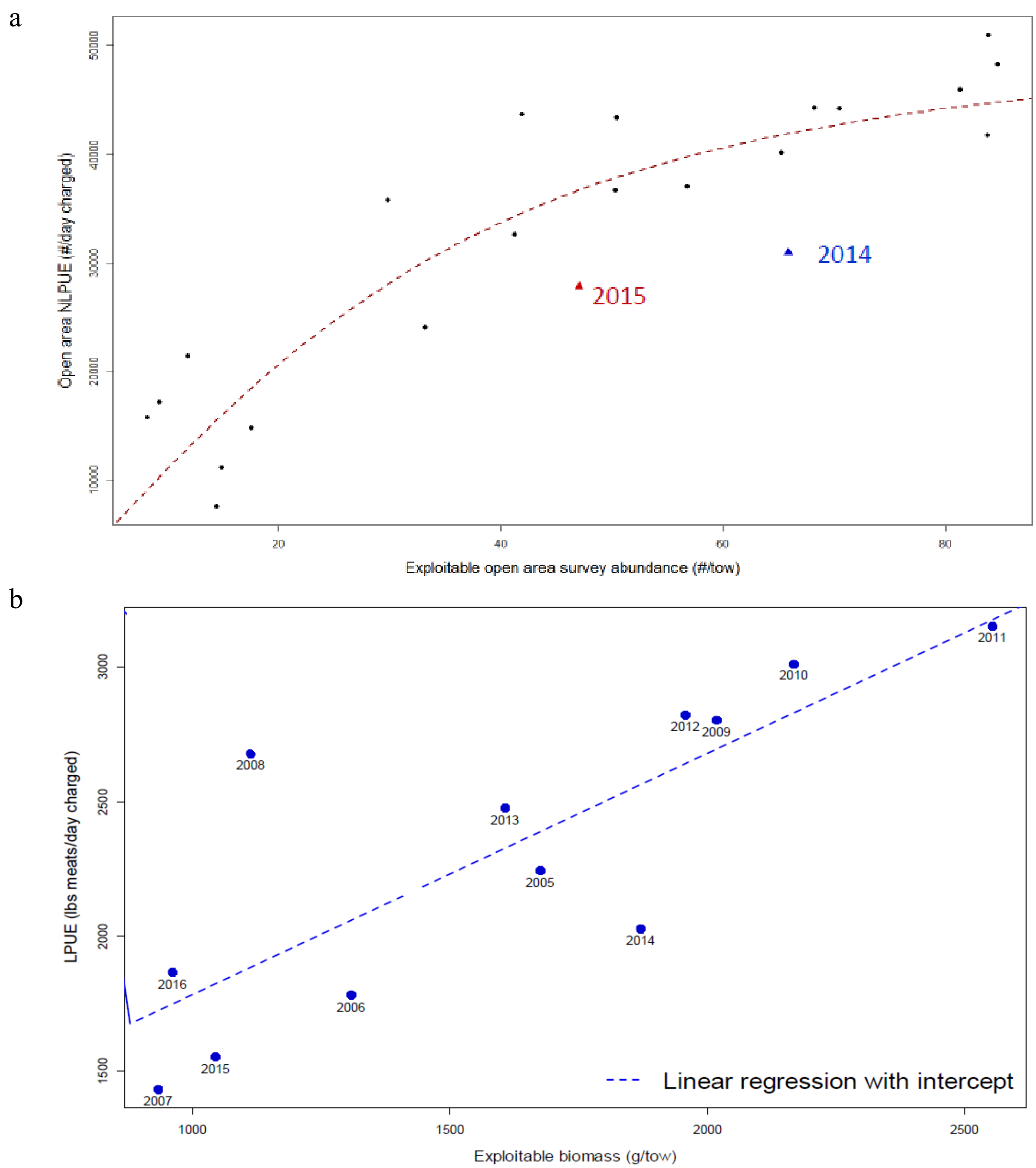


Figure 1. LPUE estimates a) from 1994-2015, using the modified predator-prey model, highlighting the overestimates of LPUE in 2014 and 2015 and b) from 2005-2016, using a simplified linear regression model with an intercept (from Hart 2016).

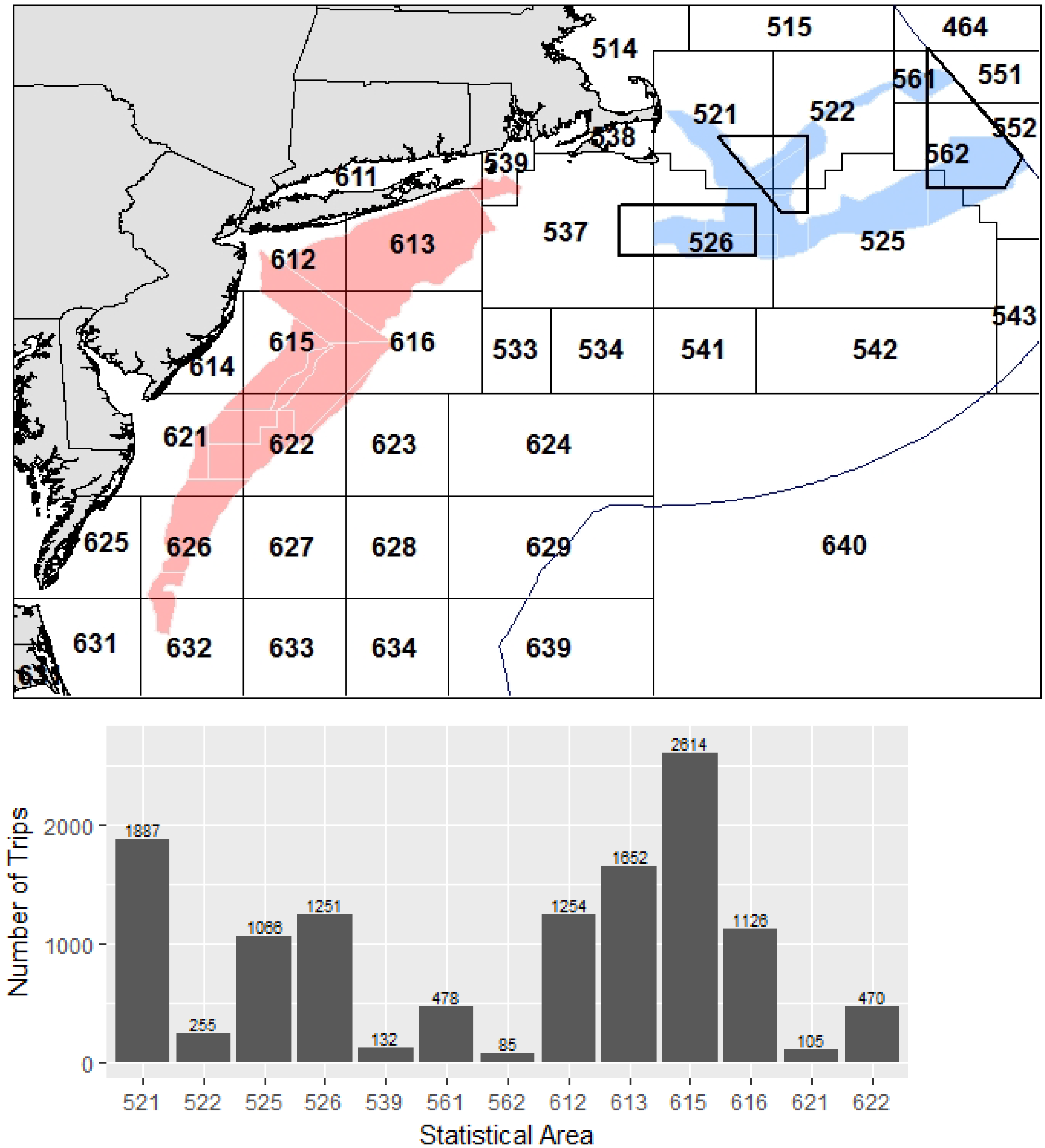


Figure 2. Top: Map of SAMS estimation areas (Georges Bank in blue and Mid Atlantic Bight in red) overlaid on Statistical Areas (numbered boxes). Bottom: Number of scallop trips included in the analysis by Statistical Areas for 2007-2016.

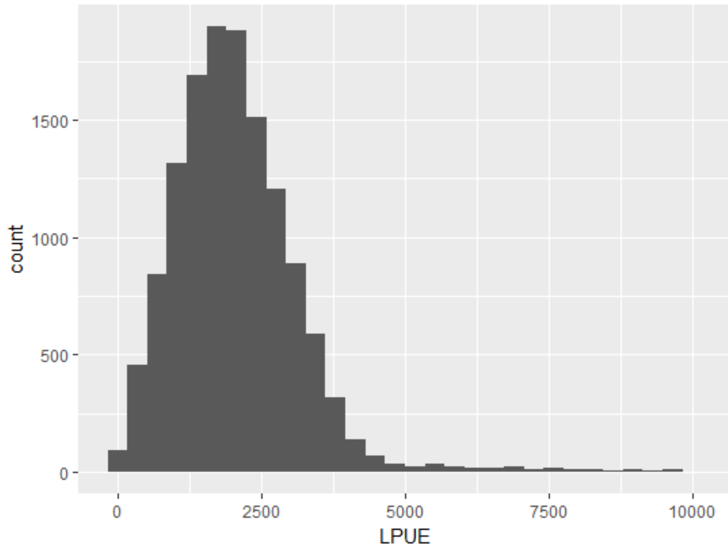


Figure 3. Landings (meat weight in pounds) per unit effort (days fished) for scallop trips 2007-2016.

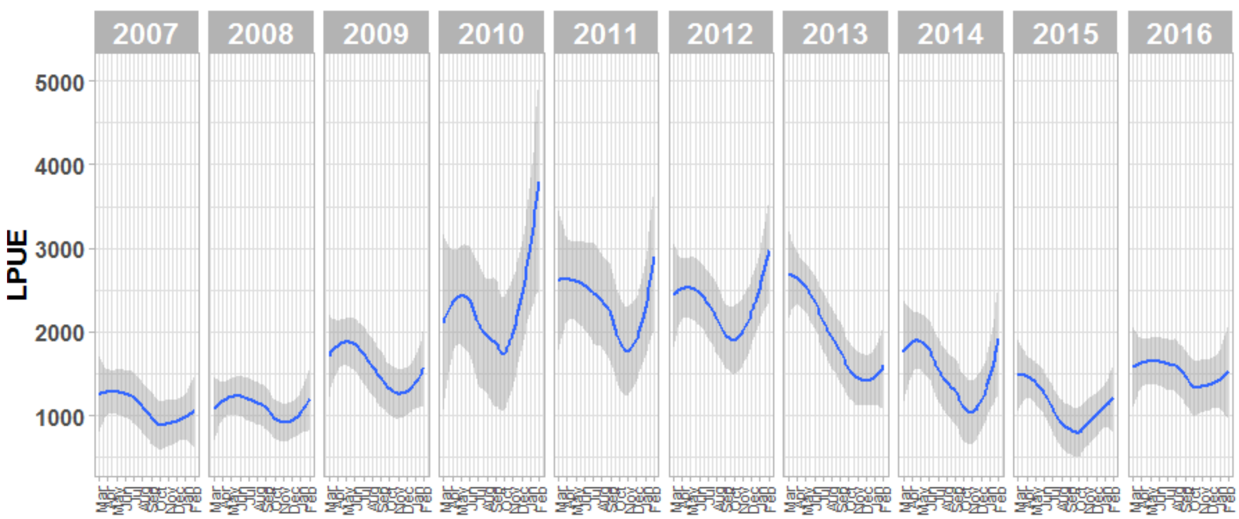


Figure 4. Exploratory plot of LPUE by month and fishing year. Curves are loess smooths of observations. Note the similar pattern (high LPUE in late spring and low in early winter) despite changes in magnitude across years.

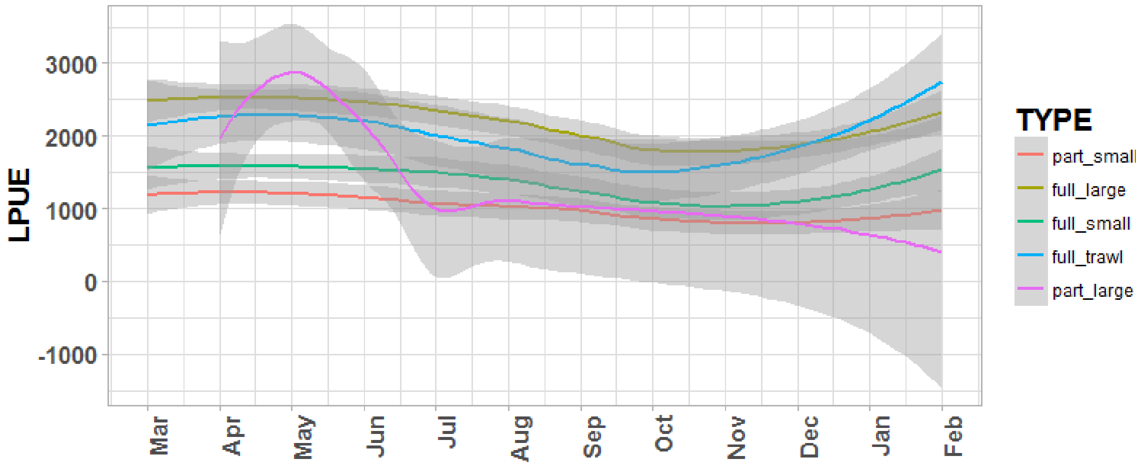


Figure 5. LPUE by month and vessel permit type. Curves are loess smooths of observations to visualize patterns and do not represent smooth functions used in models.

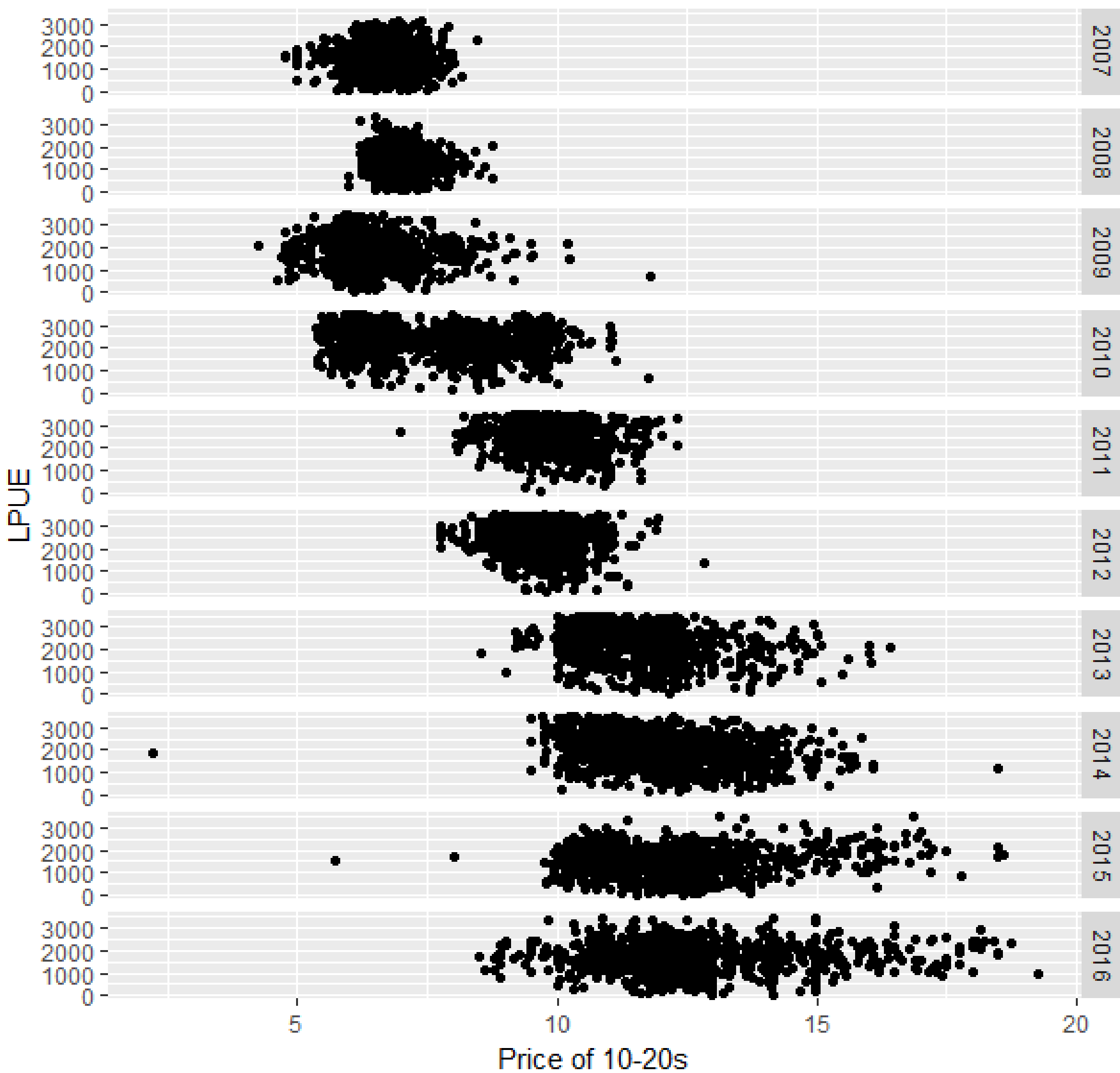


Figure 6. LPUE by price of 10-20 market category each fishing year. Note the gradual increase in price over time as well as distinct price jumps in 2011 and 2013.

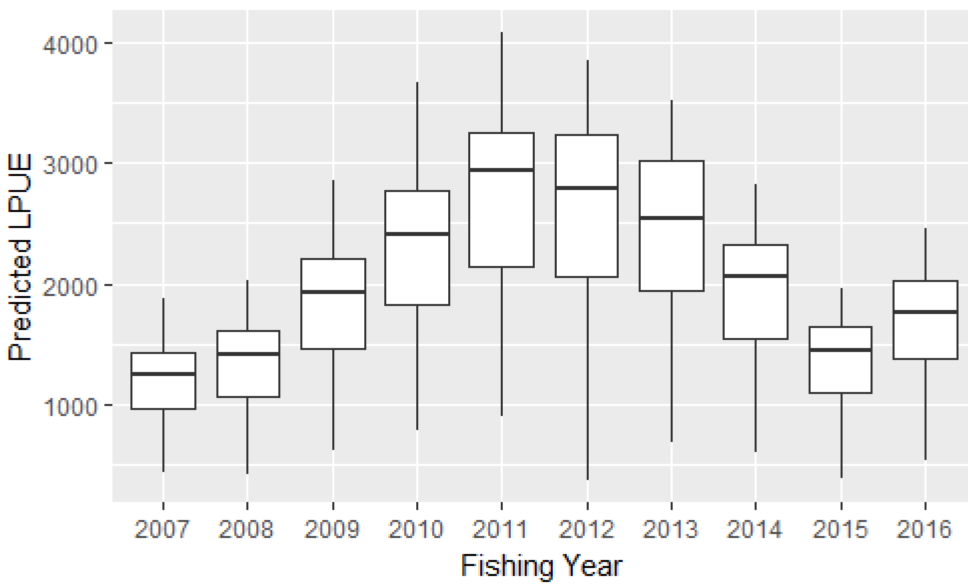


Figure 7. Boxplot of predicted LPUE by fishing year.

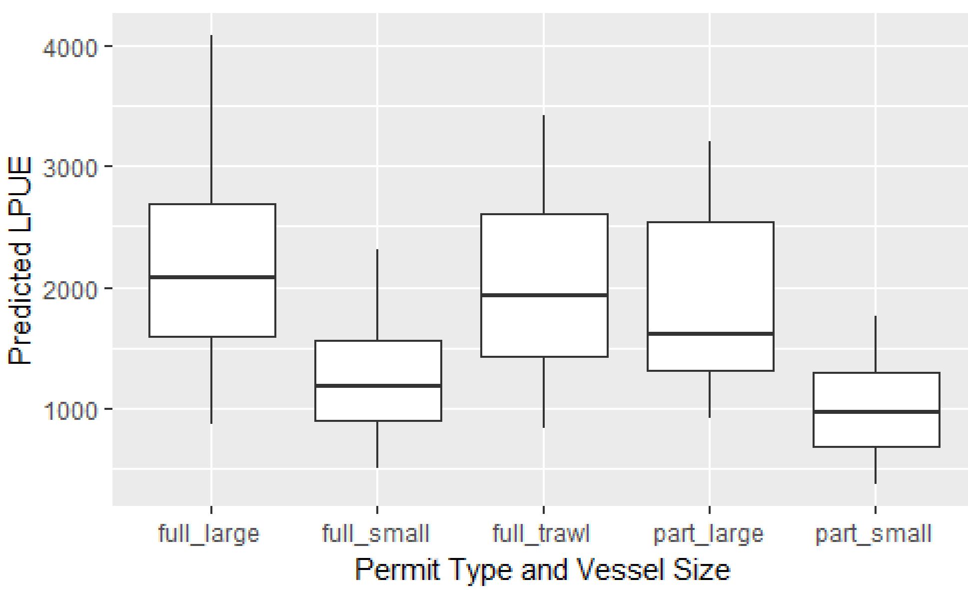


Figure 8. Boxplot of predicted LPUE by vessel permit type.

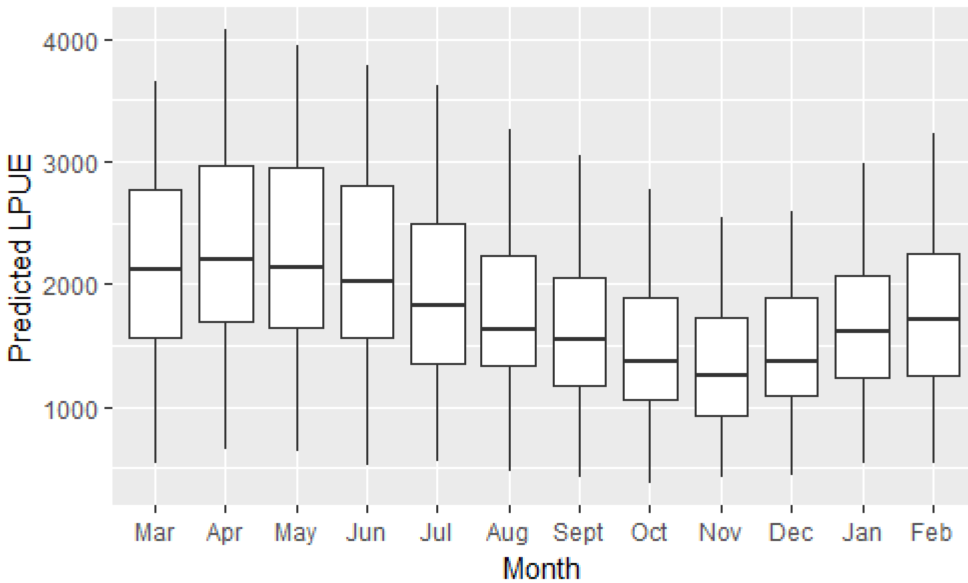


Figure 9. Boxplot of predicted LPUE by month.

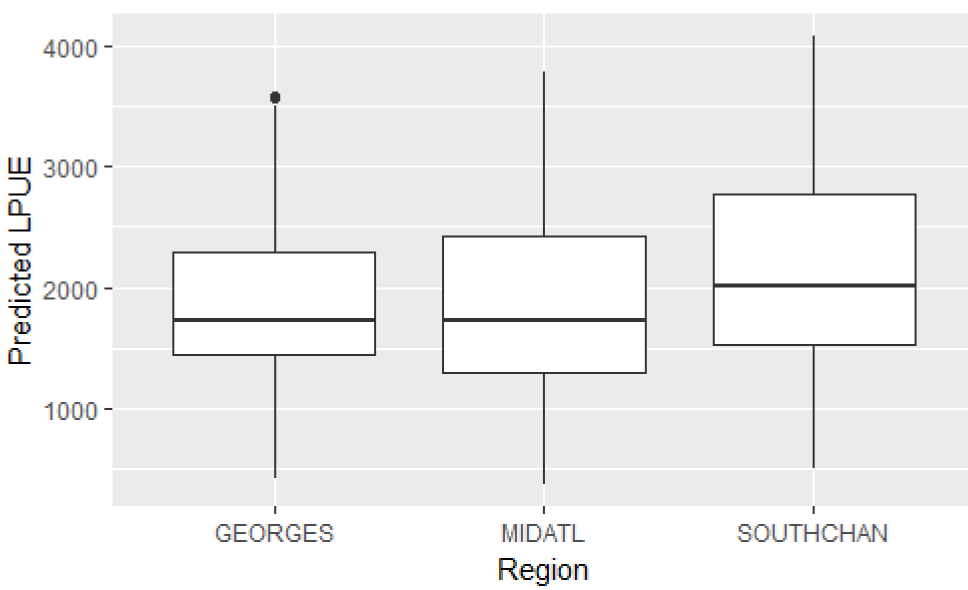


Figure 10. Boxplot of predicted LPUE by region.

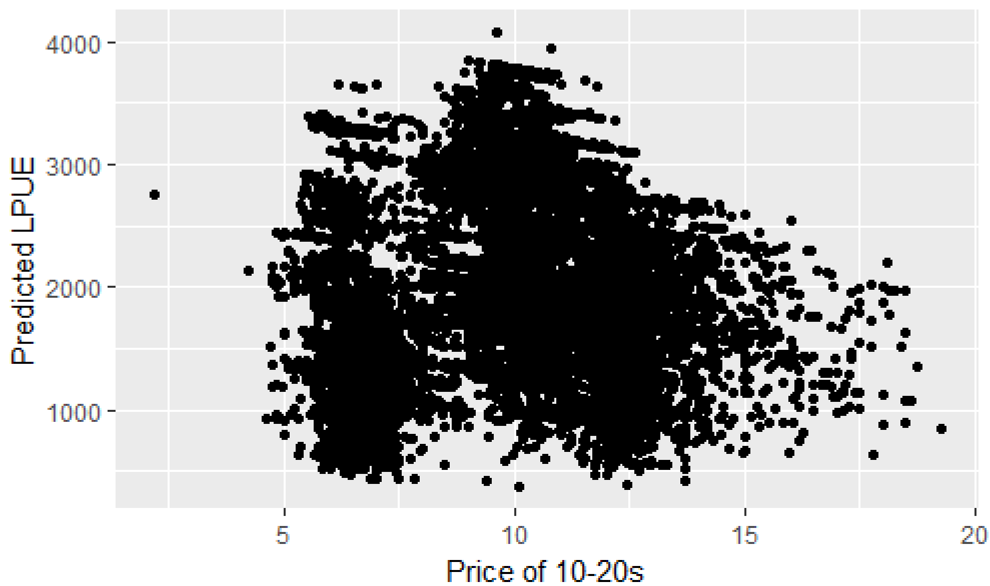


Figure 11. Predicted LPUE by price of 10-20 market category.

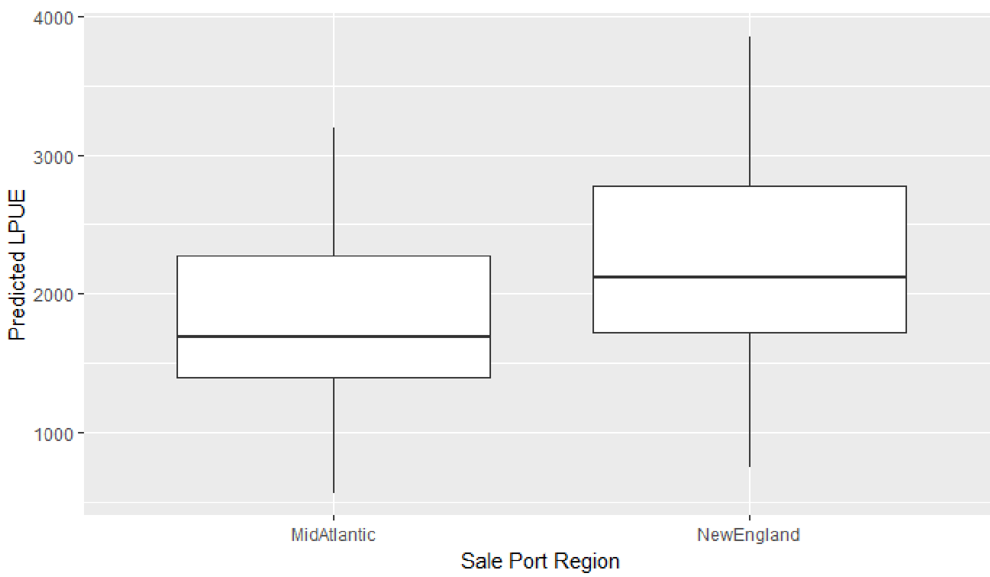


Figure 12. Boxplot of predicted LPUE by sale region.

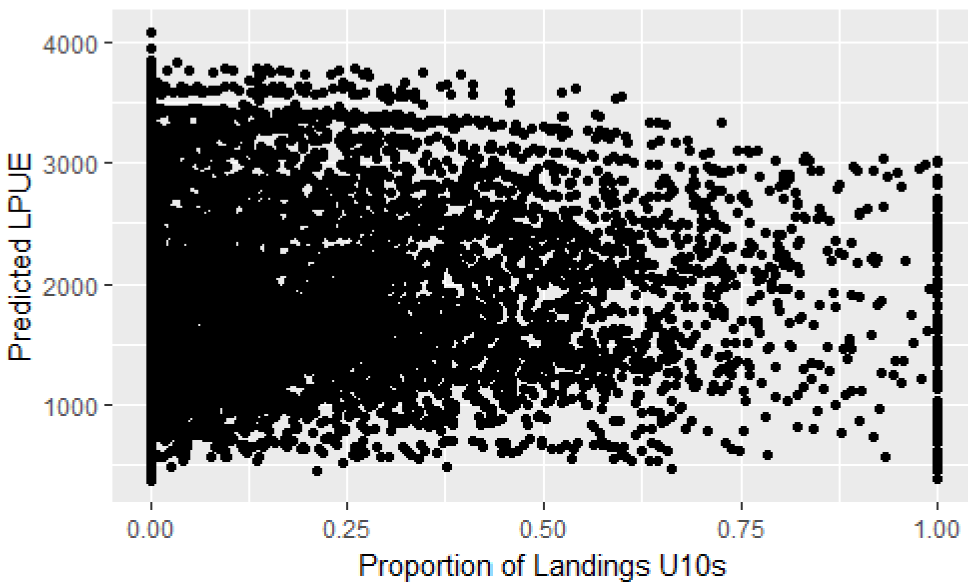


Figure 13. Predicted LPUE by the proportion of landings comprised of U10s.

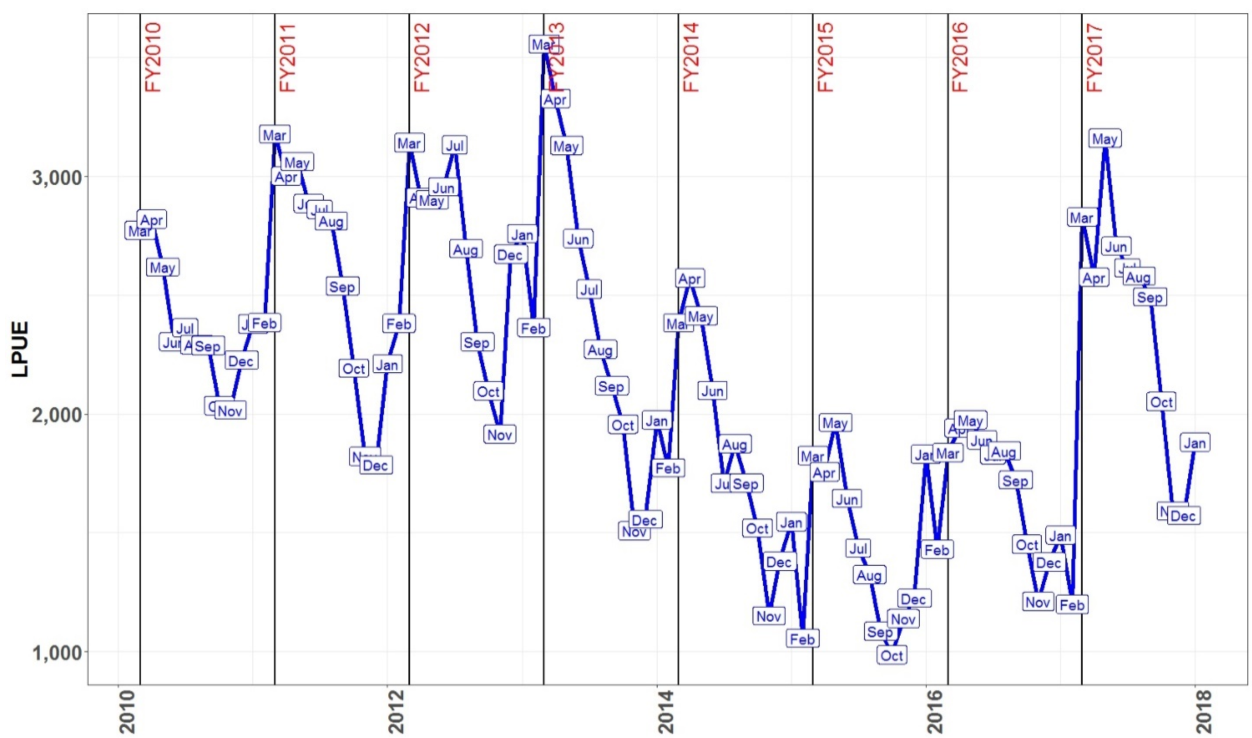


Figure 14. Open area landings per unit effort, by month, for fishing years 2010 - 2017 (from Galuardi 2017).

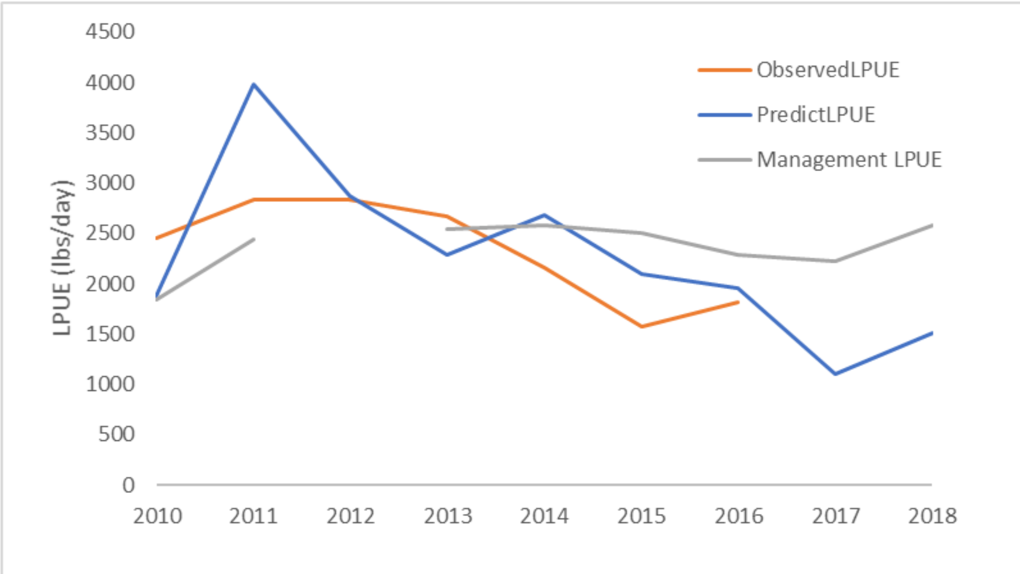


Figure 15. Observed LPUE (orange) series with estimates used in management (gray) and estimates from our model (blue).

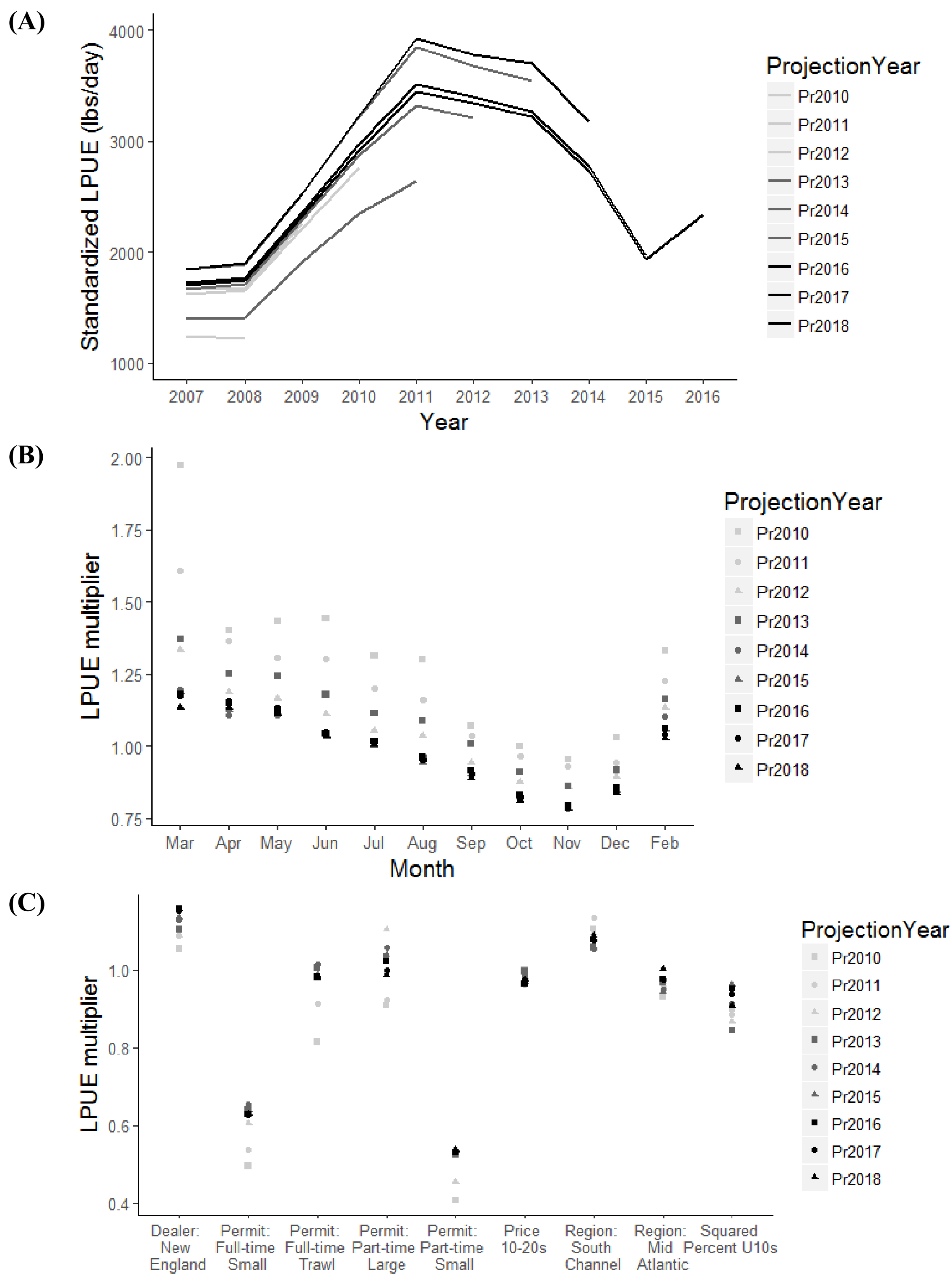


Figure 16. Model coefficients fit to each subset of data (for each projection year, y, the model is fit to data up to year y - 2). (A) Year effects represent the standardized LPUE series, (B) Month effects , and (C) Coefficients for permit types, sale region, fishing region, price, and percent U10s.

Table 1. Data aggregations used in creating alternative factors.

Factor	**Level**	**Aggregate of:**
		Statistical Areas
Region	Georges Bank	522, 525, 561, 562
	Great South Channel	521, 526
	Mid Atlantic	612, 613, 614, 615, 621, 622, 626
SAMS	Great South Channel	521, 526
	Northern flank	522, 561
	Southern flank	525, 562
	Long Island	612, 613
	Hudson Canyon	614, 615
	Elephant Trunk	621, 622
	Delmarva	626
		Vessel permit types
vessel size	small	full time small, part time small
	large	full time large, part time large, full time trawl

Table 2. AIC, deviance explained, and mean absolute prediction error (MAPE) for select candidate models. Our selected model is indicated by boldface type.

Model	Degrees of Freedom	AIC	% Deviance Explained (fit to full data set)	Average from 10-fold cross validation	
				% Deviance Explained	MAPE
fishing year + vessel permit type + month + statistical area + state of sale + trip price of 10-20s + squared U10 proportion of landings + U10 proportion of landings	12327	201754.9	42.3%	45.7%	493
fishing year + vessel permit type + month + statistical area + state of sale + trip price of 10-20s + squared U10 proportion of landings	12328	201755.7	42.3%	45.7%	492
fishing year + vessel permit type + month + SAMS + state of sale + trip price of 10-20s + squared U10 proportion of landings	12334	201761.8	42.2%	45.7%	492
fishing year + vessel permit type + month + statistical area + state of sale + trip price of 10-20s	12329	201762.5	42.3%	45.7%	493
fishing year + vessel permit type + month + statistical area + state of sale + relative trip price of 10-20s + squared U10 proportion of landings	12328	201763.3	42.3%	45.7%	493
fishing year + vessel permit type + month + statistical area + state of sale + average price of 10-20s + squared U10 proportion of landings	12328	201773.5	42.2%	45.7%	492
fishing year + vessel permit type + month + statistical area + state of sale	12330	201800.8	42.1%	45.7%	494
fishing year + vessel permit type + month + region (of fishing) + state of sale + trip price of 10-20s + squared U10 proportion of landings	12338	201809.1	42.0%	45.4%	493
fishing year + vessel permit type + month + region (of fishing) + region of sale + trip price of 10-20s + squared U10 proportion of landings	**12345**	**201826.9**	**41.8%**	**45.6%**	**494**
fishing year + vessel permit type + month + statistical area	12338	201926.3	41.4%	45.3%	499
fishing year + vessel permit type + month	12350	202425.4	38.9%	42.8%	517
fishing year + vessel permit type	12361	203311.0	34.2%	38.0%	554
fishing year	12365	205485.6	21.5%	18.3%	655
null	12374	208470.4	--	-6.2%	799

Table 3. Coefficients of the selected model (fit to all data).

Factor	Level	Coefficients	Std. Error	t value	Pr(>|t|)	
Fishing Year	2007	7.56	0.036	210.5	< 2e-16	***
no standard	2008	7.59	0.038	199.1	< 2e-16	***
	2009	7.88	0.034	231.6	< 2e-16	***
	2010	8.10	0.039	210.4	< 2e-16	***
	2011	8.27	0.048	174.0	< 2e-16	***
	2012	8.24	0.046	179.2	< 2e-16	***
	2013	8.20	0.052	158.1	< 2e-16	***
	2014	8.04	0.056	142.3	< 2e-16	***
	2015	7.70	0.056	136.2	< 2e-16	***
	2016	7.88	0.058	136.0	< 2e-16	***
Vessel Permit Type	full time small	-0.46	0.013	-34.8	< 2e-16	***
standard: full time large	full time trawl	-0.02	0.018	-1.2	0.223	
	part time large	-0.01	0.060	-0.2	0.841	
	part time small	-0.62	0.026	-23.6	< 2e-16	***
Month	April	0.00	0.015	0.0	9.61E-01	
standard: March	May	-0.02	0.014	-1.4	1.57E-01	
	June	-0.09	0.015	-5.8	5.25E-09	***
	July	-0.12	0.017	-7.3	3.24E-13	***
	August	-0.17	0.017	-9.7	< 2e-16	***
	September	-0.24	0.018	-13.1	< 2e-16	***
	October	-0.33	0.021	-15.6	< 2e-16	***
	November	-0.37	0.029	-12.7	< 2e-16	***
	December	-0.30	0.029	-10.4	< 2e-16	***
	January	-0.13	0.024	-5.2	2.63E-07	***
	February	-0.10	0.019	-5.1	3.98E-07	***
Fishing Region	Mid Atlantic	0.00	0.012	0.1	0.909	
standard: Georges Bank	South Channel	0.08	0.012	7.3	3.41E-13	***
Sale Port Region *standard: Mid Atlantic*	New England	0.15	0.010	15.1	< 2e-16	***
Price of 10-20s		-0.03	0.004	-5.8	7.45E-09	***
(Percent of Landings U10)^2		-0.10	0.024	-4.0	5.37E-05	***

Table 4. Observed and predicted LPUE and estimates used in management for fishing years 2010 – 2018.

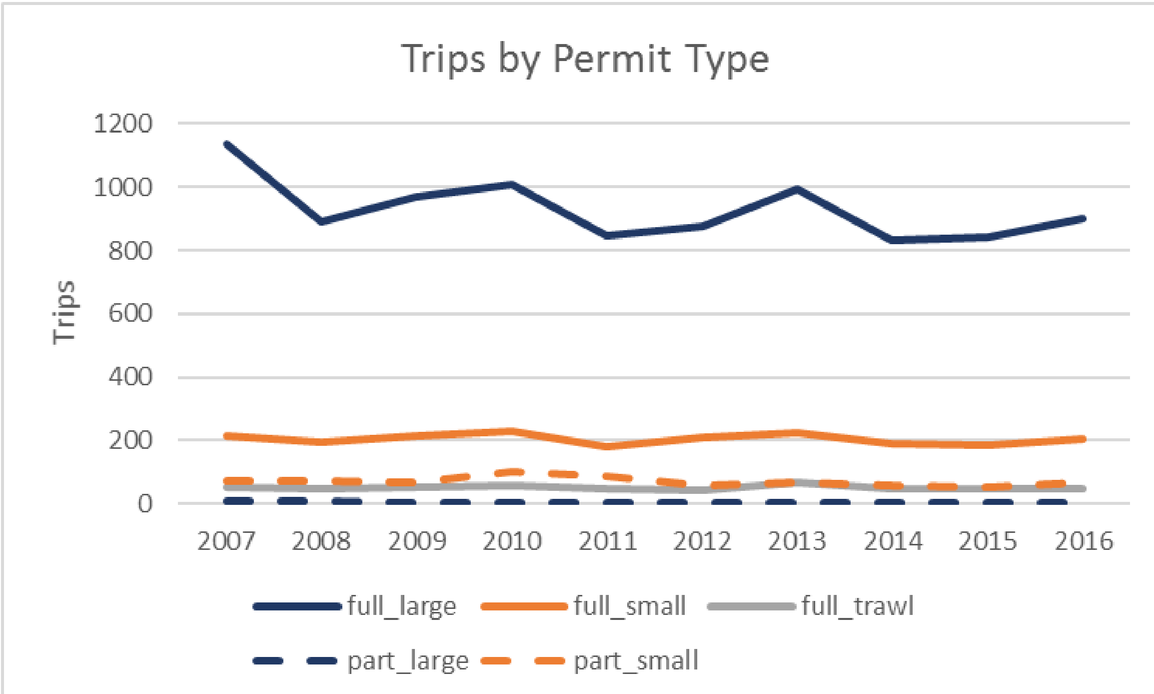
Projection Year	Management LPUE (lbs/day)	Observed LPUE (lbs/day)	Predicted LPUE (lbs/day)	Forecast Error (management)	Forecast Error (this study)
2007	--	1421	--	--	--
2008	--	1542	--	--	--
2009	--	2063	--	--	--
2010	1693	2457	1885	764	572
2011	2442	2839	3980	397	-1141
2012	--	2836	2873	?	-37
2013*	2537	2670	2283	133	387
2014	2581	2159	2682	-422	-523
2015**	2506	1580	2091	-926	-511
2016	2288	1817	1959	-471	-142
2017	2227	--	1103	--	--
2018	2581	--	1507	--	--
			Average Absolute Error	519	473

*Exploitable biomass ratio shifts from SMAST survey to SAMS estimates from information presented to the scallop Plan Development Team.
**Open area exploitable biomass data missing for FY2015. We used the average from FY2014 and FY2016.

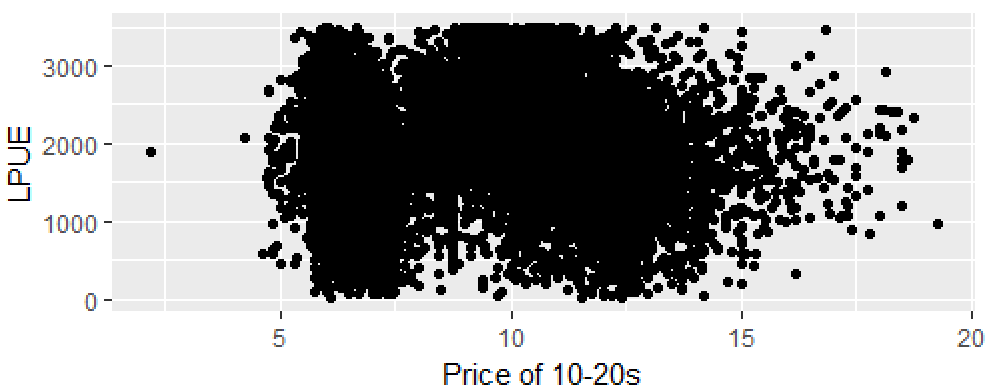
Table 5. Exploitable Biomass estimates used to scale LPUE estimates. The SMAST estimate was used in 2010. The average of 2014 and 2016 was used for 2015.

YEAR	SMAST Open Area Exploitable Biomass Estimate	SAMS Open Area Exploitable Biomass Estimate
2007	23847	--
2008	21523	--
2009	23650	--
2010	27995	27027
2011	48329	--
2012	33982	--
2013	--	39697
2014	--	32693
2015	--	--
2016	--	29972
2017	--	22487
2018	--	25354

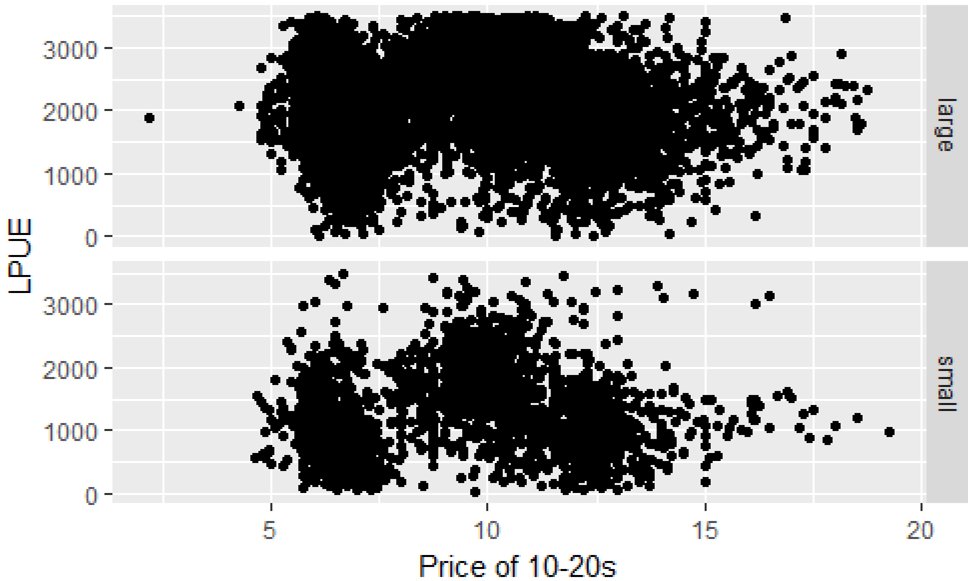
Appendix 1. Data Exploration Figures



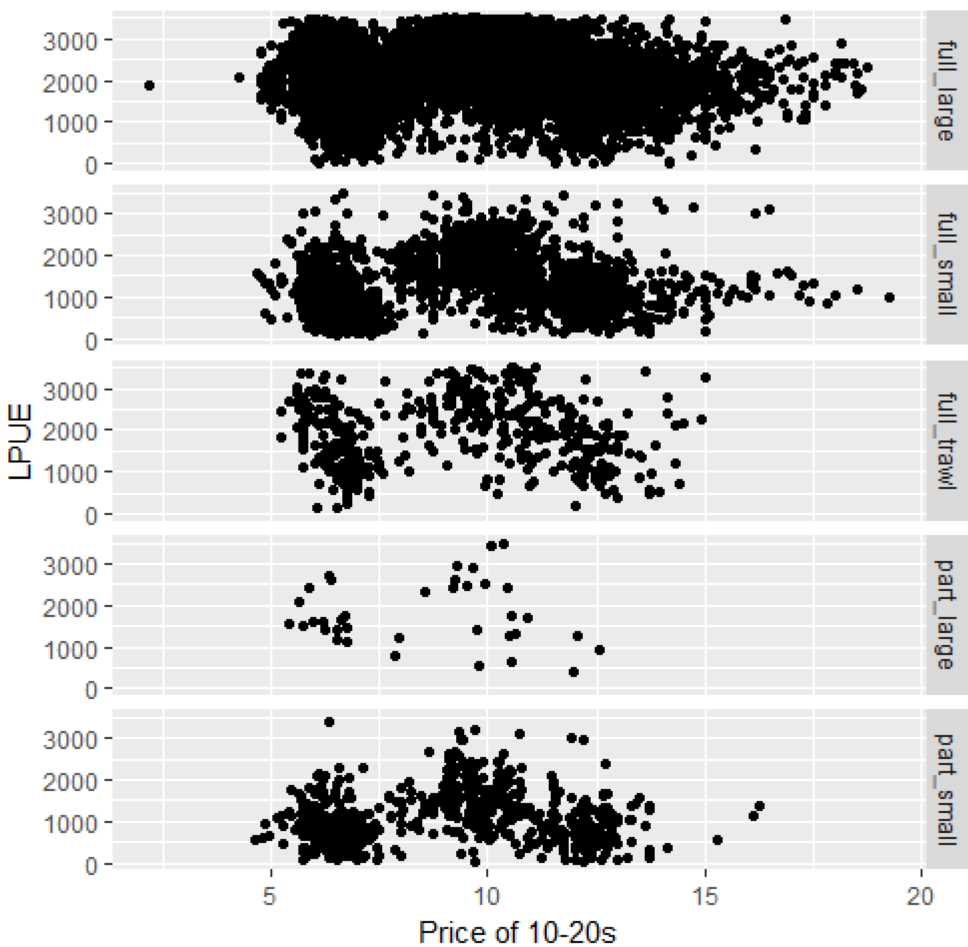
A1. Scallop trips taken by permit type 2007-2016.



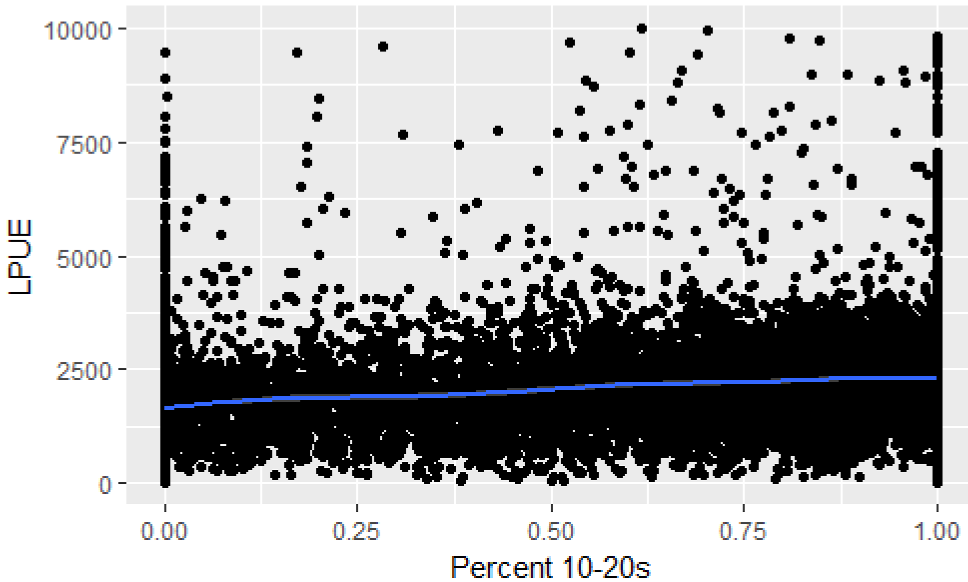
A2. LPUE by price of 10-20 market category.



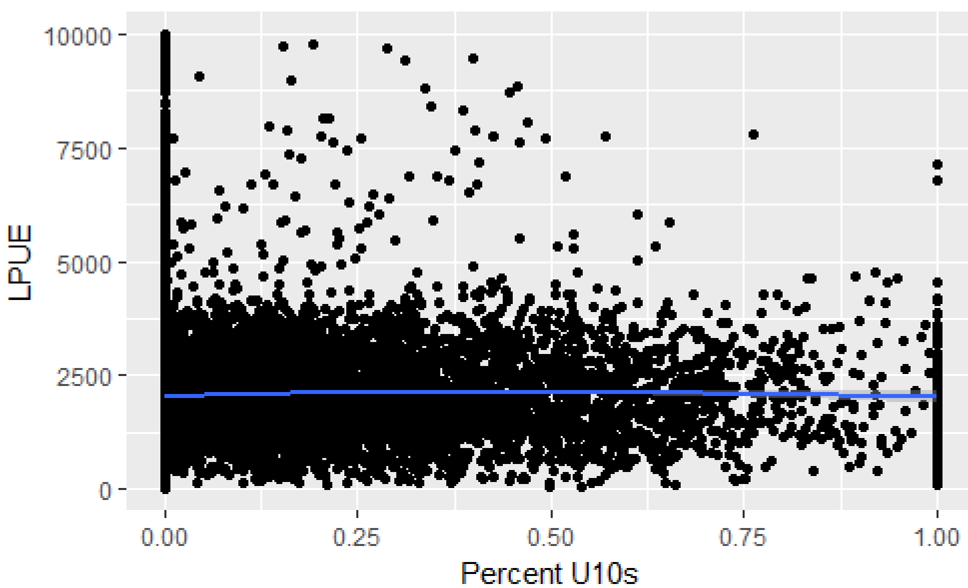
A3. LPUE by price of 10-20 market category by vessel size.



A4. LPUE by price of 10-20 market category by permit type.

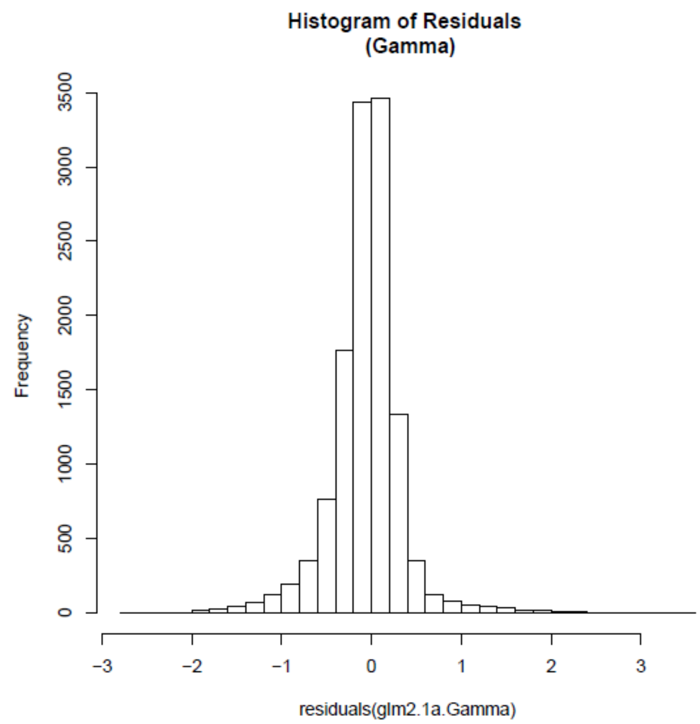


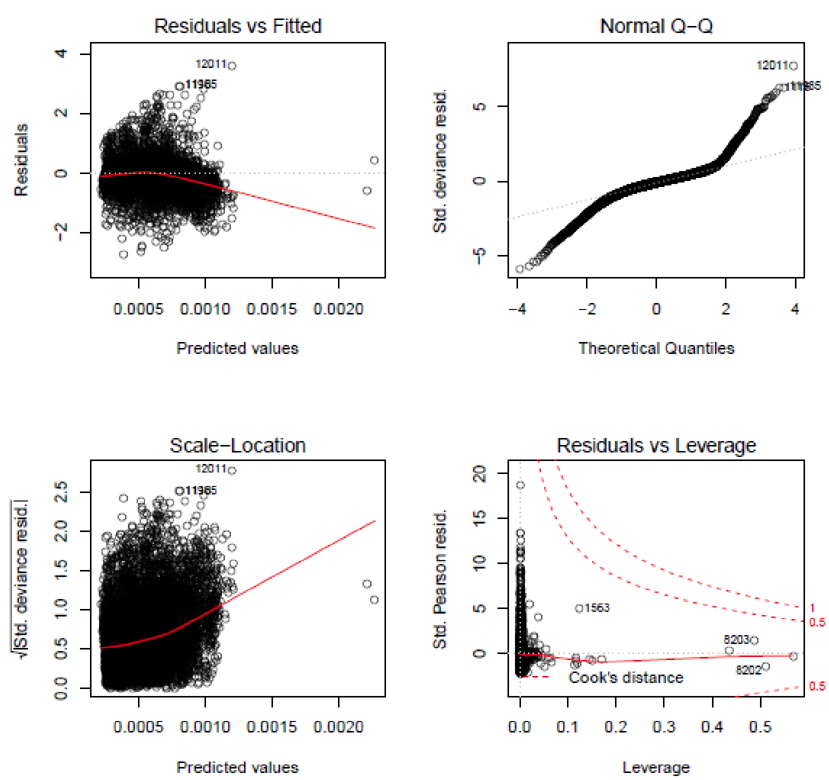
A5. LPUE by percent of landings comprised of 10-20 scallops. Blue curve is exploratory smooth to help visualize trend.

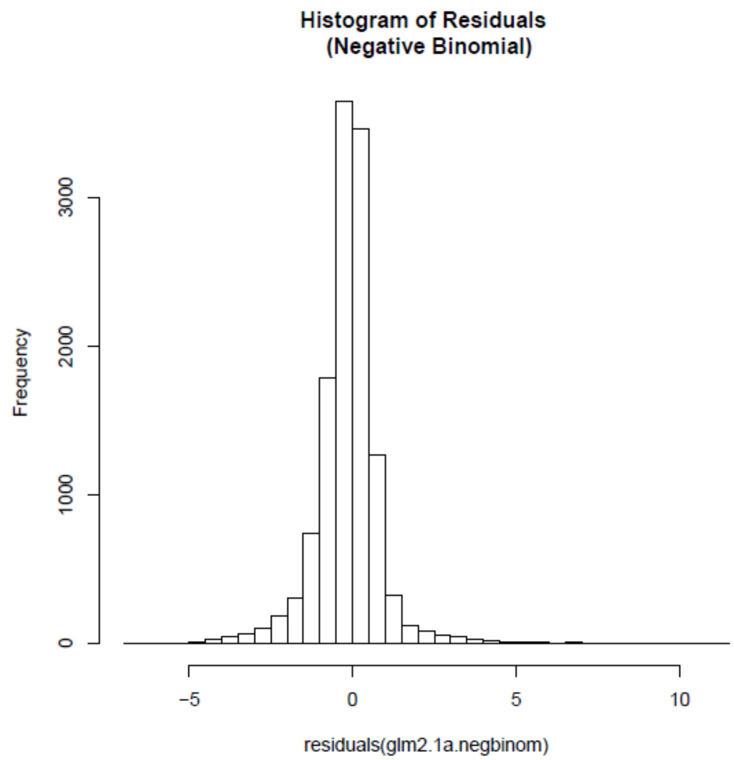


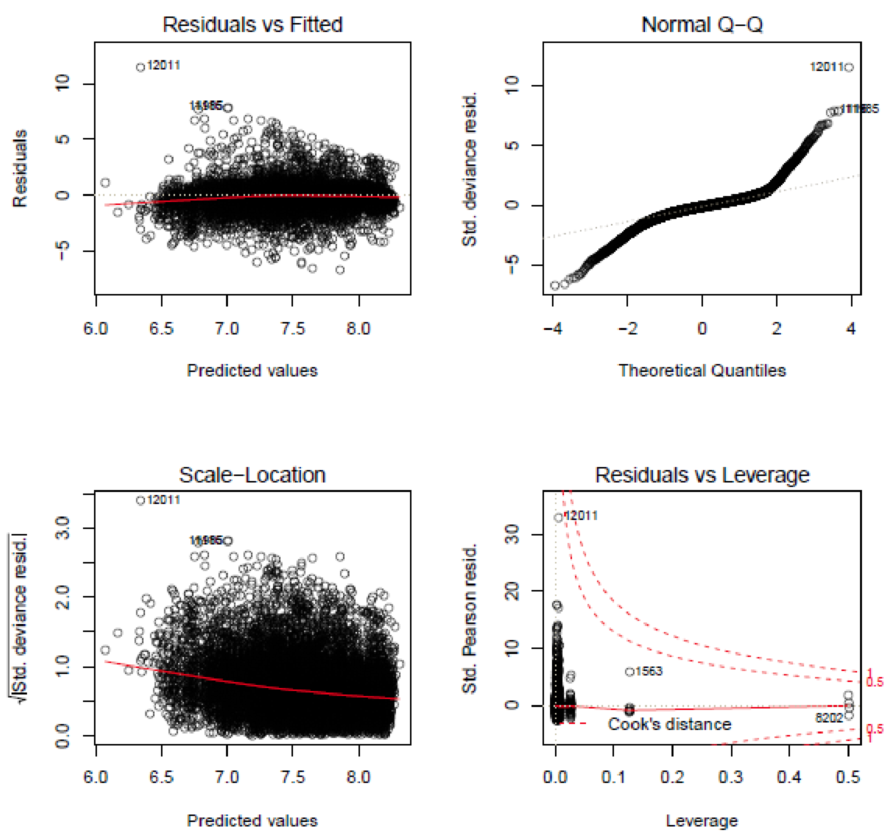
A6. LPUE by percent of landings comprised of U10 scallops. Blue curve is exploratory smooth to help visualize trend.

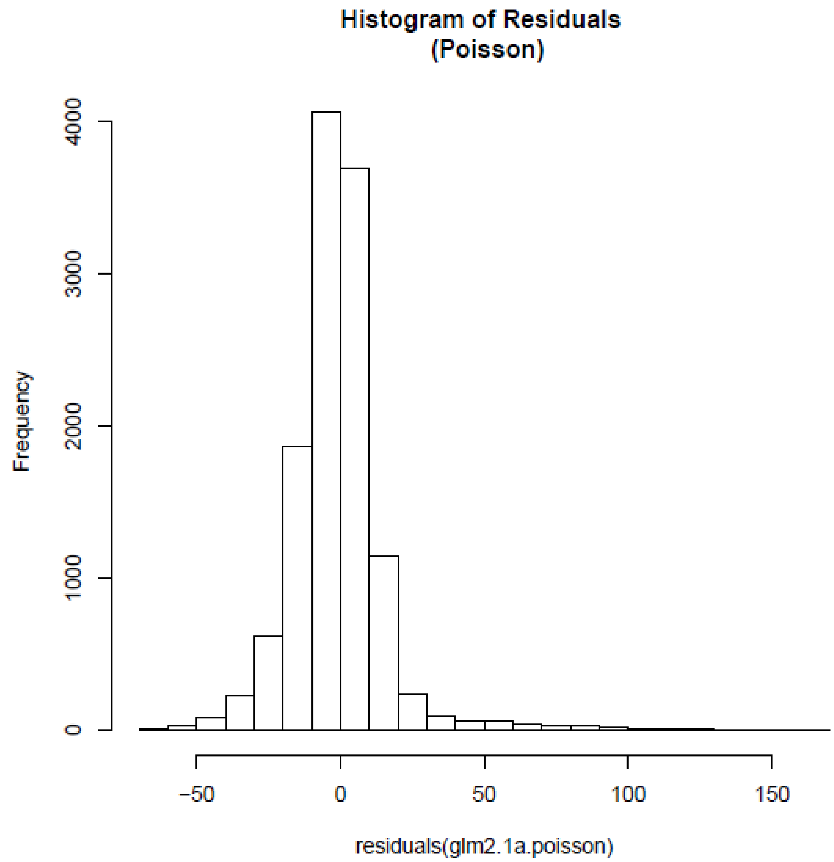
Appendix 2. Diagnostics of Alternative Distributions



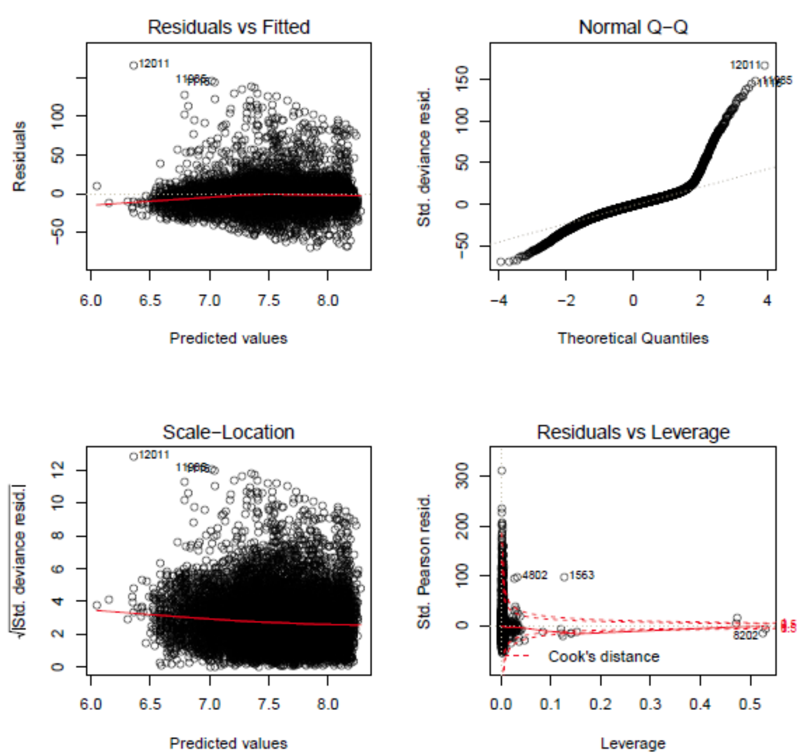



Histogram of Residuals
(Negative Binomial)
Frequency
0
1000
2000
3000
−5
0
5
10
residuals(glm2.1a.negbinom)


Residuals vs Fitted
Residuals
Predicted values
12011
Normal Q−Q
Std. deviance resid.
Theoretical Quantiles
12011
Scale−Location
√|Std. deviance resid.|
Predicted values
12011
Residuals vs Leverage
Std. Pearson resid.
Leverage
12011
1563
8202
Cook's distance

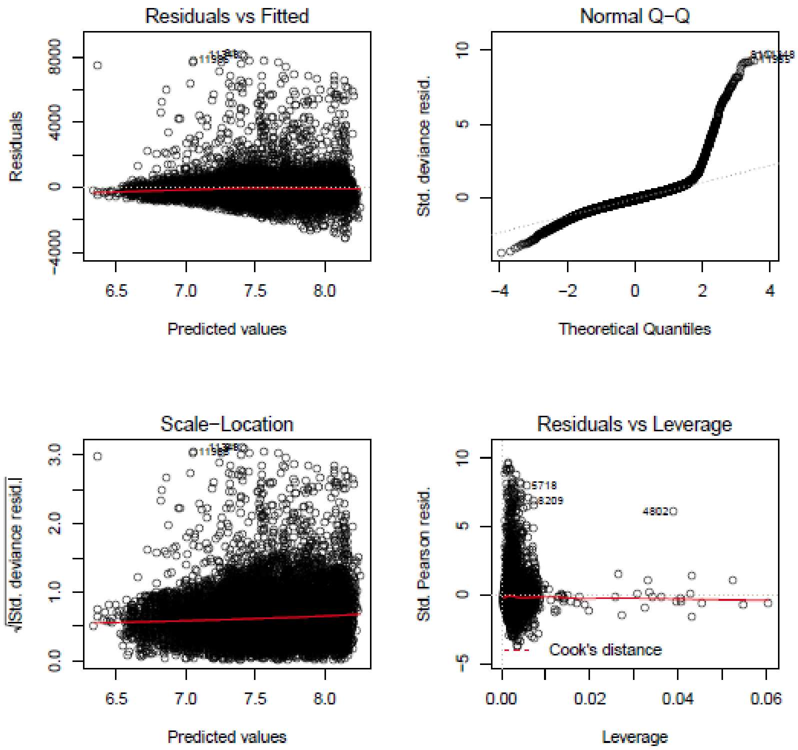


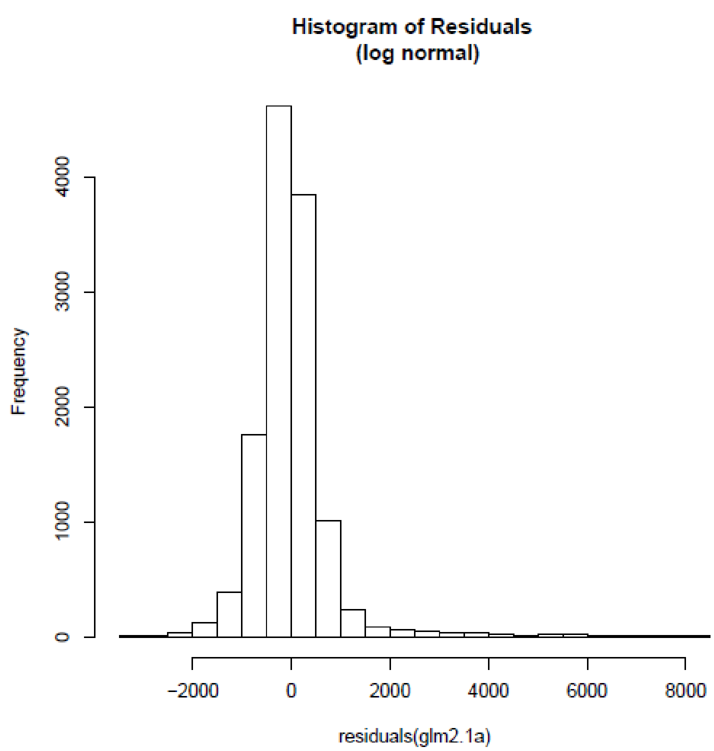
Histogram of Residuals
(Poisson)
Frequency
0
1000
2000
3000
4000
-50
0
50
100
150
residuals(glm2.1a.poisson)

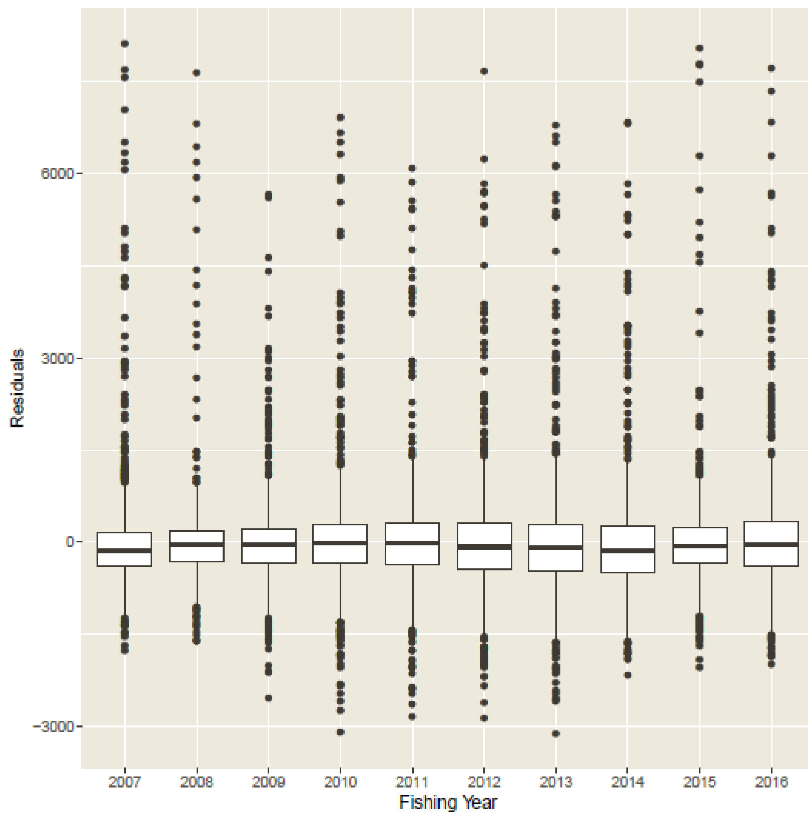


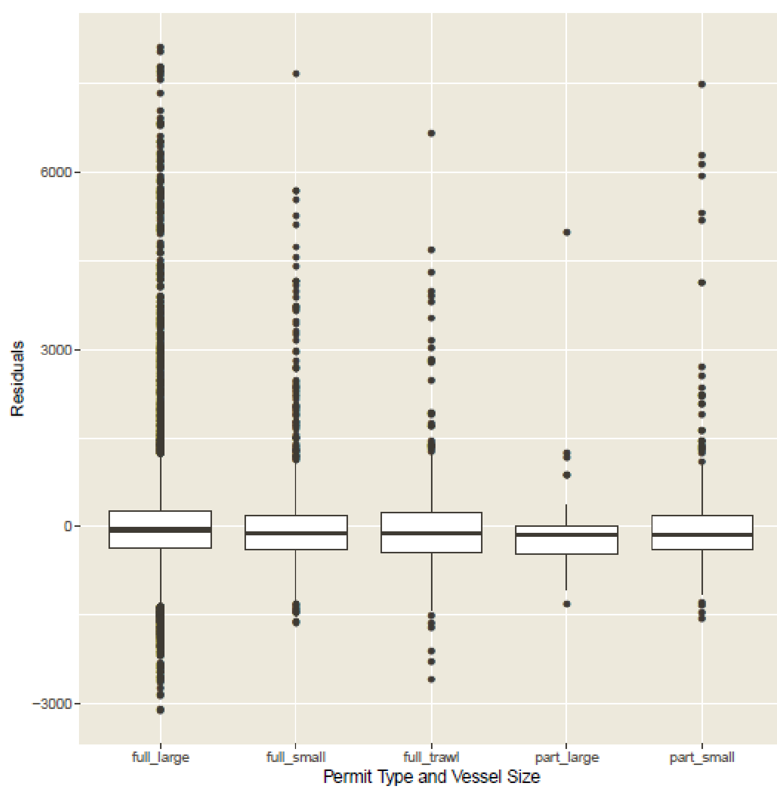
Residuals vs Fitted
Residuals
Predicted values
12011
Normal Q-Q
Std. deviance resid.
Theoretical Quantiles
Scale-Location
√|Std. deviance resid.|
Predicted values
Residuals vs Leverage
Std. Pearson resid.
Leverage
4802
1563
8202
Cook's distance

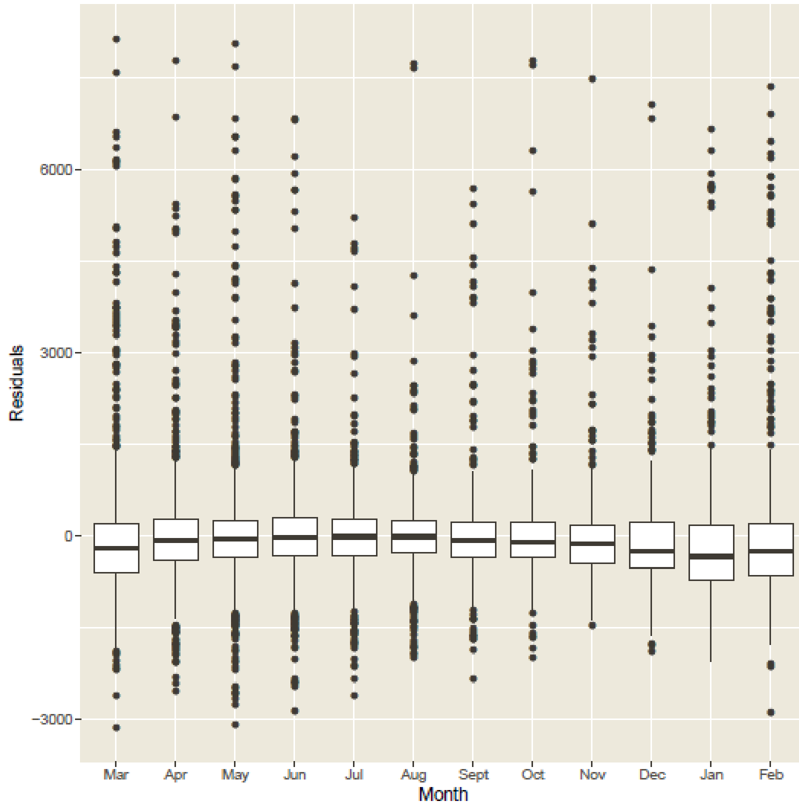
Appendix 3. Model Diagnostics

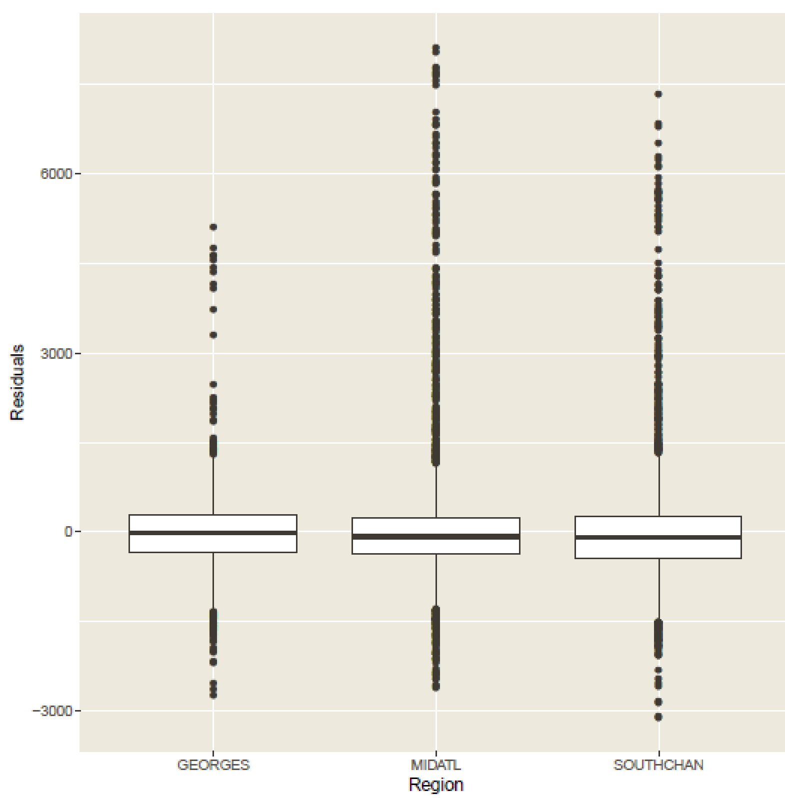


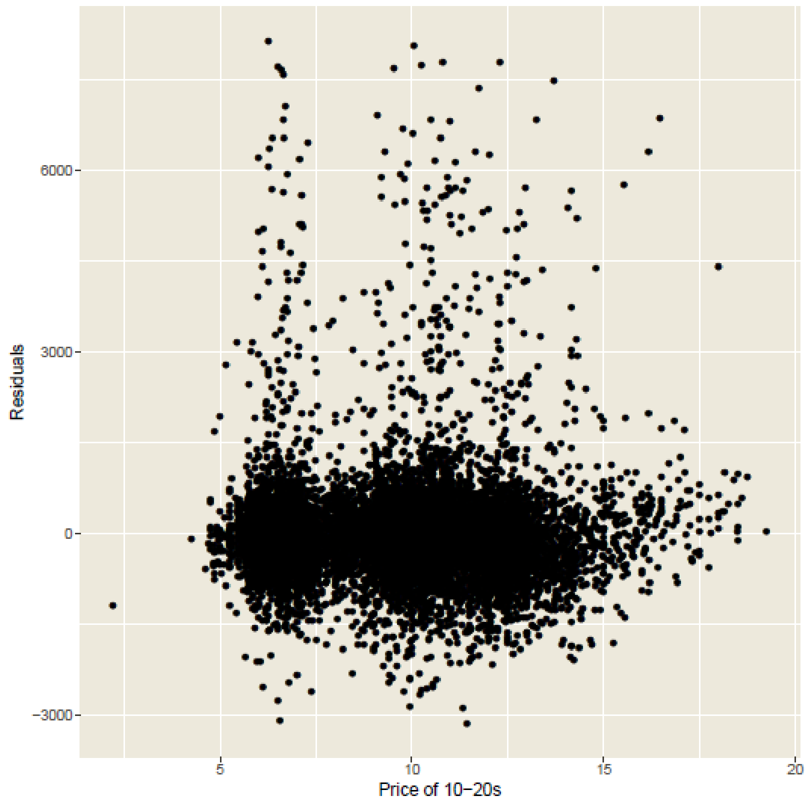



Residuals
6000
3000
0
−3000
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
Fishing Year

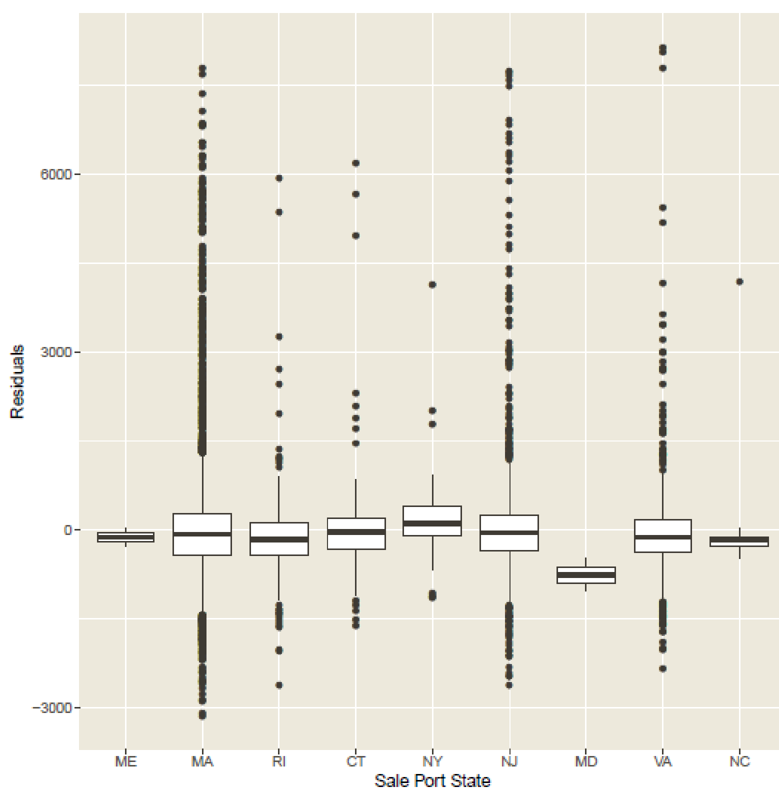

Residuals
6000
3000
0
−3000
full_large
full_small
full_trawl
part_large
part_small
Permit Type and Vessel Size


Residuals
6000
3000
0
−3000
Mar
Apr
May
Jun
Jul
Aug
Sept
Oct
Nov
Dec
Jan
Feb
Month

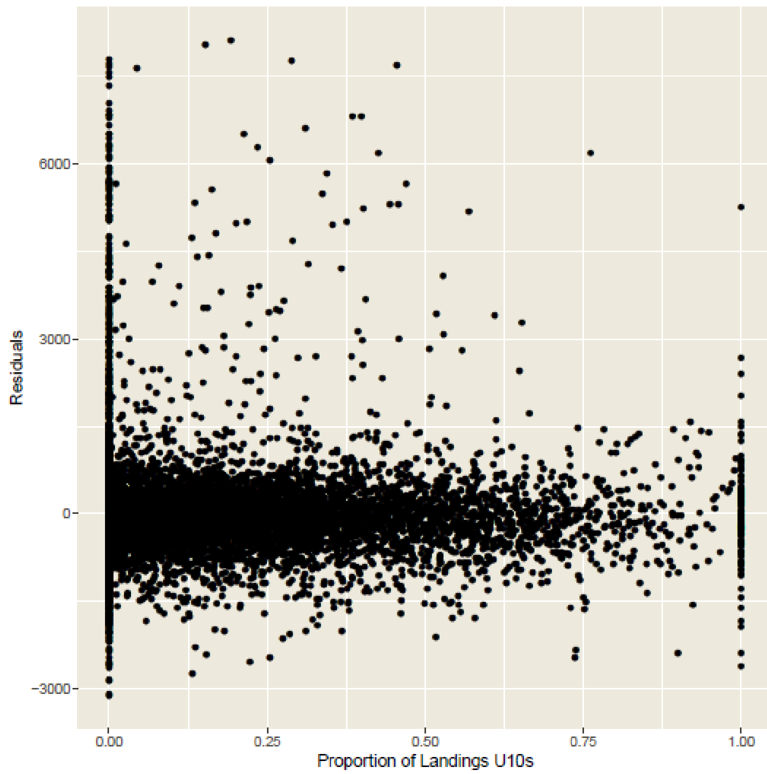

Residuals
6000
3000
0
−3000
GEORGES
MIDATL
SOUTHCHAN
Region

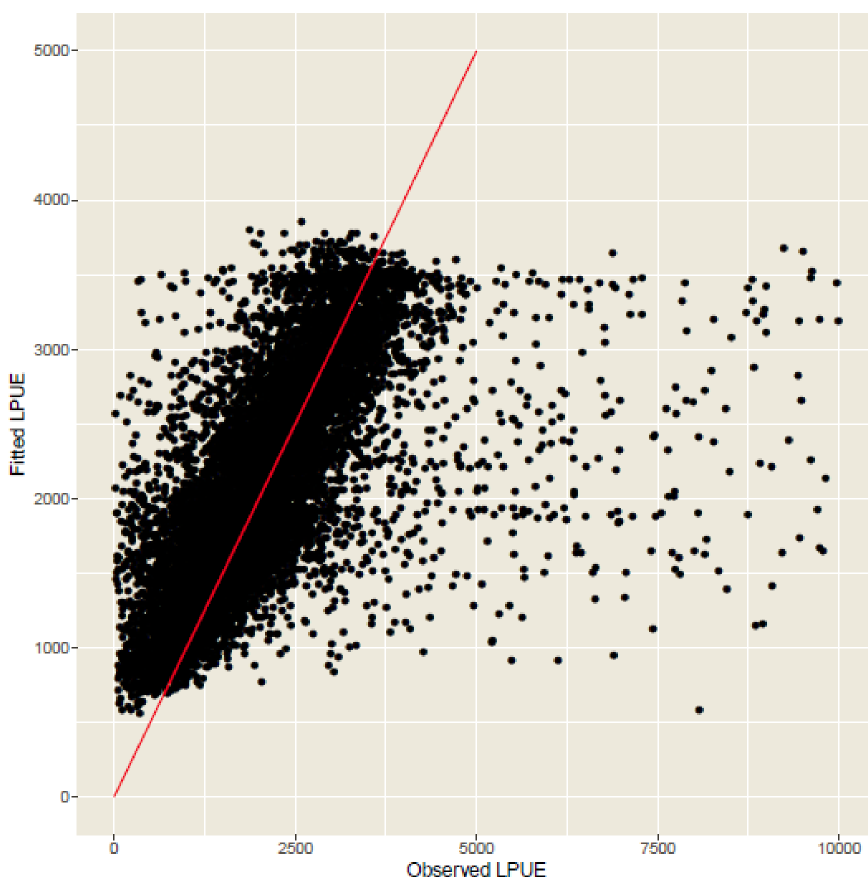


Residuals
6000
3000
0
-3000
5
10
15
20
Price of 10-20s



Residuals
6000
3000
0
-3000
ME
MA
RI
CT
NY
NJ
MD
VA
NC
Sale Port State


Residuals
6000
3000
0
−3000
0.00
0.25
0.50
0.75
1.00
Proportion of Landings U10s


Fitted LPUE
5000
4000
3000
2000
1000
0
0
2500
5000
7500
10000
Observed LPUE

Appendix A9. Survey coverage in comparison to scallop habitat identified using tree models

Kevin Friedland (NEFSC, Narragansett, RI) and Larry Jacobson (NEFSC, Woods Hole, MA)

Summary

Random forest and boosted regressions trees were fit to Northeast Fisheries Science Center (NEFSC) fall and spring bottom trawl survey data from 1992-2016 for a large number of fish and invertebrate species including sea scallops. These models used static and dynamics predictor variables that reflect habitat complexity, physical conditions, and lower trophic levels. Bottom trawls are not designed to target sea scallops, but winter bottom trawl survey data are used in the scallop stock assessment model for the Mid-Atlantic Bight and there is commercial fishing for scallops with bottom trawls in the same region. Appreciable numbers of scallops are taken in NEFSC fall and spring bottom trawl surveys in the Mid-Atlantic and on Georges Bank, particularly by the gear used since 1998, thus it was deemed a useful index for model construction. The bottom trawl survey data have the benefit of providing coverage of the entire shelf, including transition and marginal habitat areas that are not covered by the scallop dredge, Habcam or drop camera surveys that concentrate on sampling in known habitat areas. The analysis showed three relatively large areas of potential scallop habitat that were not sampled by directed scallop surveys during 2017, which was a year with relatively extensive sampling. These areas included the southwest portion of Georges Bank and at the northern and southern ends of the Mid-Atlantic Bight. Results for sea scallops, Atlantic surfclams (NEFSC 2017a) and ocean quahogs (NEFSC 2017b) show that tree model analysis of survey data and the extensive habitat, environmental, plankton and climatological data sets available can be used in stock assessments to track changes in spatial distribution and in survey design (NEFSC 2017a; 2017b; Hennen and Jacobson, in review). Future research could be directed at building tree models tailored to sessile organisms such as sea scallops, incorporating spring and fall data into the same habitat map, and by using habitat, environmental, plankton and climatological data in model-based survey abundance estimation (e.g. Habcam).

Methods

It was decided to utilize habitat predictions for biomass (units: log(catch weight+1)) rather than modeled occupancy probability to keep the habitat units on a similar scale to catch distribution maps. Maps based on fall survey data during 2007-2016 were averaged to form a single habitat map that illustrated “recent conditions”. Maps based on spring data were not used because tree model results for spring and fall showed similar spatial patterns; however, catches were higher in fall. Survey data for 2017 were plotted on the average habitat map to show survey coverage because 2017 was a year of relatively complete and extensive survey sampling. We also overlaid 2017 fishing effort data from VMS (showing squares with at least 10 pings) to see if the predicted habitat areas included current fishing grounds.

Results

Observed fishing grounds were all on areas of scallop habitat predicted by the tree models. Areas of predicted habitat on the US side of Georges Bank were sampled completely during the Habcam survey during 2017, but not during the dredge or drop-camera surveys (Figures 1). Areas of predicted habitat at the northern and southern end of the Mid-Atlantic were not sampled during 2017 by any of the surveys (Figures 2).

Discussion

Working Group members noted that habitat based on tree model results in the Great South Channel appears less suitable than along the southeast flank of Georges Bank, even though other surveys show higher density and fishing is more intense in the Great South Channel. Members suggested that the unexpected pattern might be due to reduced efficiency of the survey bottom trawl on rocky bottom in the Great South Channel, relative to efficiency on relatively smooth bottom elsewhere on Georges Bank. It may be useful in future to combine multiple surveys in the same tree model to counteract spatial variability among gear capture efficiency or to use regional designations to allow interactions between region and variables describing roughness of the bottom.

Working group members noted that depth was the most important predictor in all of the tree models and that the effects of depth were assumed constant along the coast, up to the interaction between predictors in the model despite obvious latitudinal shoaling for scallops. In future, it may be desirable to include regional designations or some measurement of position along the coast in the tree models so that the effects of depth can depend on location.

References

Hennen, D., and Jacobson, L. In prep. Improving the NEFSC clam¬ survey for Atlantic surfclams and ocean quahogs. Northeast Fish Sci Cent Ref Doc.

Northeast Fisheries Science Center. 2017a. 61st Northeast Regional Stock Assessment Workshop (61st SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 17-05; 466 p. (doi:10.7289/V5/RD-NEFSC-17-05) Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://nefsc.noaa.gov/publications/

Northeast Fisheries Science Center. 2017b. 63rd Northeast Regional Stock Assessment Workshop (63rd SAW) Assessment Summary Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 17-09; 28 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://nefsc.noaa.gov/publications/

Figure 1. Habcam, drop camera, and dredge sampling locations and VMS ping locations plotted with biomass habitat gradients on Georges Bank (purple shading).

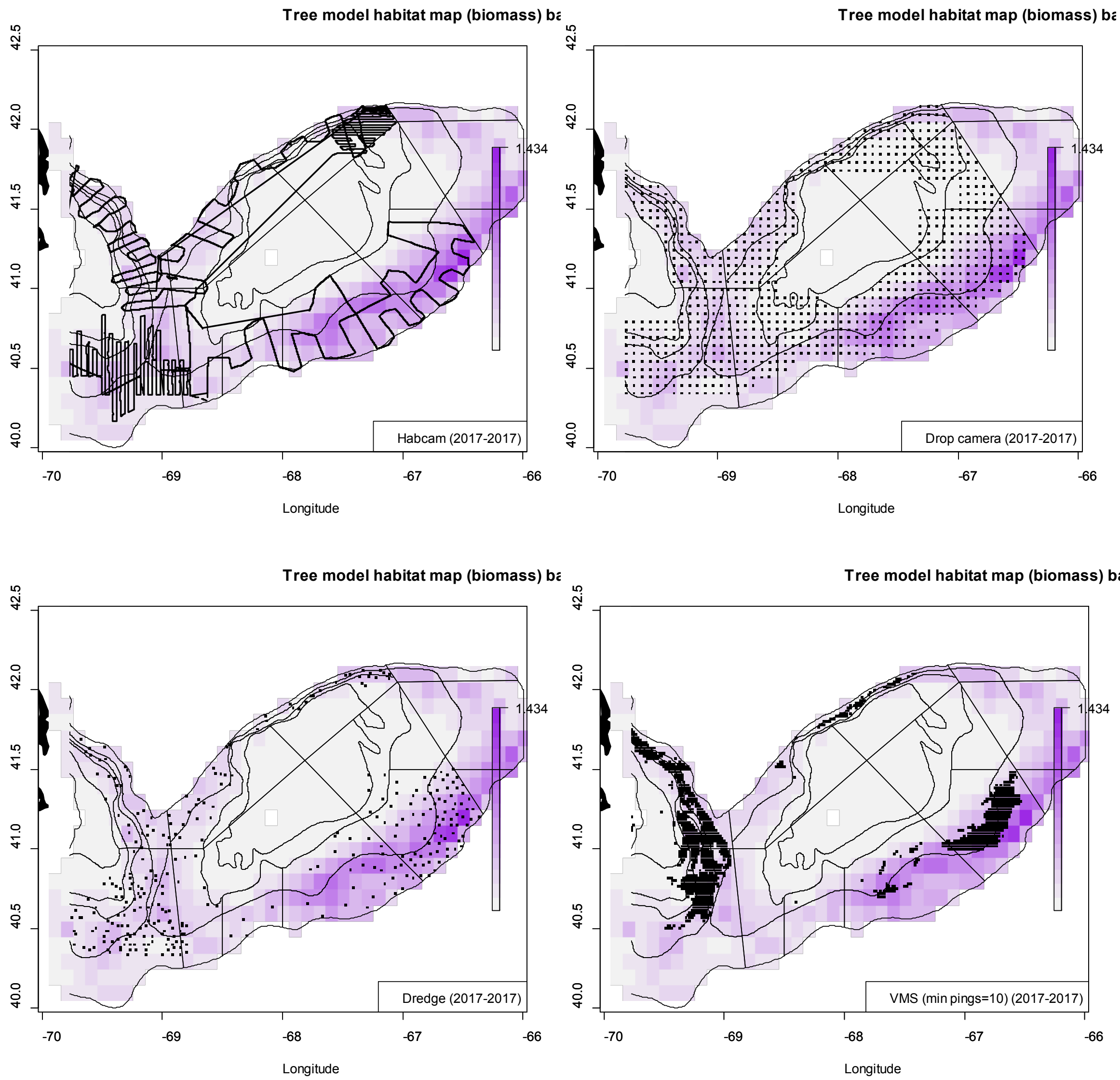
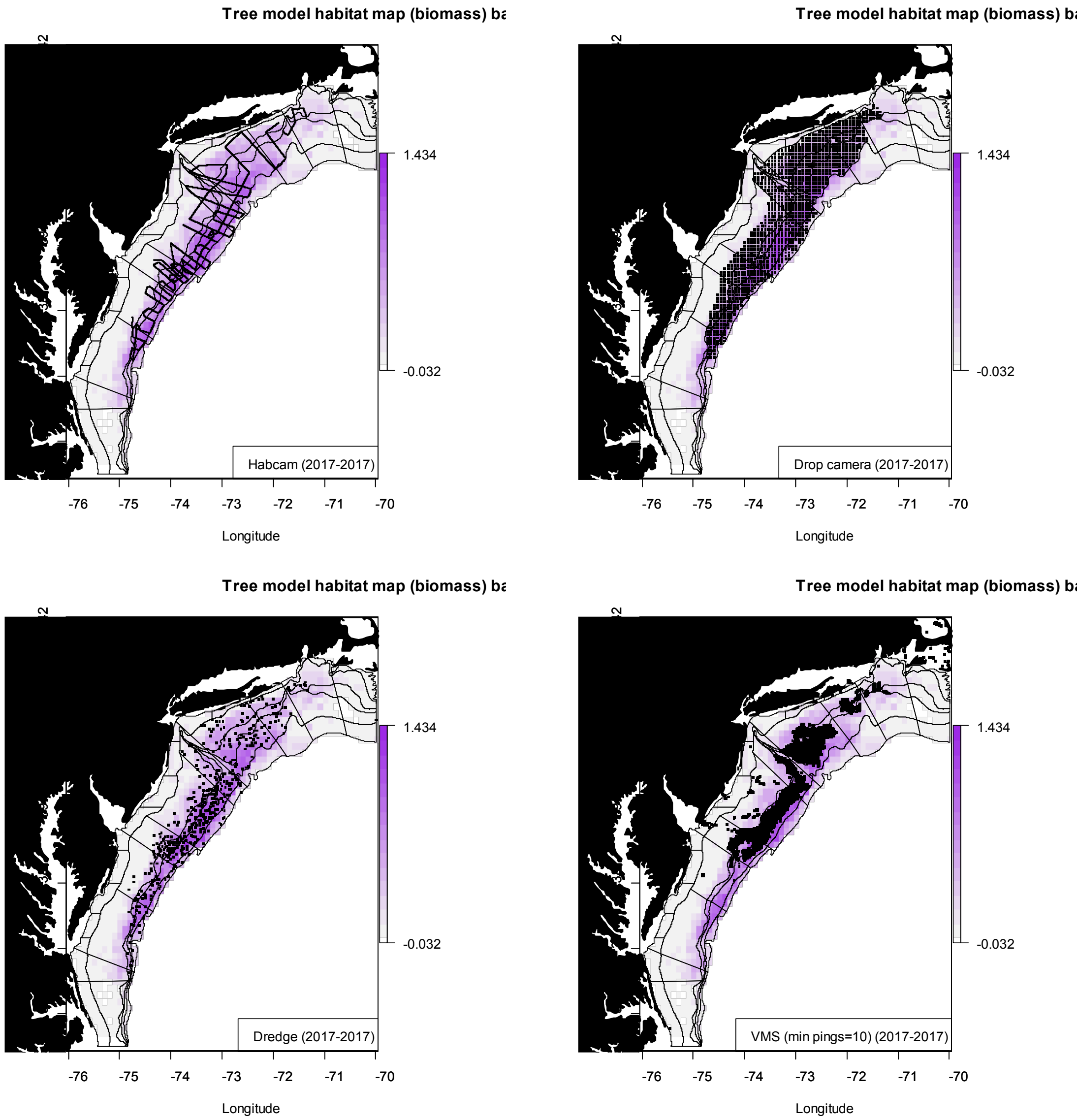


Figure 2. Habcam, drop camera, and dredge sampling locations and VMS ping locations plotted with biomass habitat gradients on Middle Atlantic Bight (purple shading).



Appendix A10 - Reference Points Based on Gonad Weights

Deborah R. Hart
Northeast Fisheries Science Center, 166 Water St., Woods Hole MA 02543

Introduction

This section calculates reference points using the SYM model, as discussed in section 9 (TOR-6) of the main document, except that gonad weight (rather than meat weight) was used to represent spawning stock biomass (SSB). Conversion of shell height to gonad weight was based on Table A4.3 of the main document, from Hennen and Hart (2012).

Results

Similar to the results using meat weights for SSB, both regions show evidence for higher recruitment at higher SSB (Figure App A10-1). Estimated stock recruit curves were more stable on Georges Bank, where it appears that most of the uncertainty is in the asymptote s. In the Mid-Atlantic, the stock-recruit relationship in some iterations continues to increase well beyond the observed SSB. Although Y_{MAX} and B_{MAX} values were generally well defined, F_{MAX} was highly uncertain in both regions, and hit the $F = 2$ bound in a majority of the simulations in the Mid-Atlantic and in about 15% of the cases on Georges Bank (Figures App A10-2). MSY-based reference points are somewhat better defined, as the stock-recruit relationships tend to constrain F_{MSY} (Figure App A10-4).

Estimates for the combined (Georges Bank and Mid-Atlantic) MSY range up to about 150,000 mt meats, with outliers over twice that (Figure App A10-5, App A10-6), and B_{MSY} also ranged up to 150,000 mt gonads. F_{MSY} values for the combined stock are highly uncertain. Median mean yield curves have a maximum at $F = 0.49$ on Georges Bank, and $F = 0.61$ in the Mid-Atlantic, with corresponding MSY values of 19,974 and 25,754 mt meats, respectively (Table App A10-1). One complication with using median curves is that the sum of the median curves for Mid-Atlantic and Georges Bank is not equal to the median curve of the sum of the Georges Bank and Mid-Atlantic SYM runs (Figure App A10-7). In Table App A10-1, Med. Yield$_{0.58}$ and Med. SSB$_{0.58}$ for the combined stock represent the value of the combined median yield curve at $F_{\mathrm{MSY}} = 0.58$, which is different than the sum of the corresponding values for each region. The Working Group chose to use the sum of the regional yields and SSBs at $F_{\mathrm{MSY}} = 0.58$ as the value of the combined MSY and B_{MSY}.

The probabilistic approach to reference points employed here can be used to examine the tradeoff between the risk of overfishing and the expected loss of yield (Hart 2013). At a fixed fishing mortality F, the probability that this F exceeds the true but unknown F_{MSY} is simply one minus the cdf of the distribution of the F_{MSY}s (Figure App A10-8, red dashed line). On the other hand, fishing at an F other than the estimated F_{MSY} will result in a loss of expected median yield (Figure App A10-8, blue solid line). According to Amendment 15 to the Sea Scallop Fishery Management Plan (NEFMC 2011), the Allowable Biological Catch and Annual Catch Limit (ABC and ACL) is estimated using the fishing mortality corresponding to the 25th percentile of the distribution of the combined F_{MSY}, which in this case is $F_{\text{ACL}} = 0.44$. Fishing at this rate considerably reduces the risk of overfishing (i.e., fishing greater than the true F_{MSY}), with little cost in terms of loss of median expected yield (Figure App A10-8).

Discussion

The estimated overall F_{MSY} of 0.58 calculated using gonad weights as a surrogate for SSB is somewhat lower than the corresponding estimate using meat weights of $F_{\text{MSY}} = 0.64$. The exponent on the shell height to gonad weight relationships is slightly greater than the corresponding exponent for shell height to meat weight. This reflects the increasing investment by scallops in reproduction as they get larger and older. The higher exponent means that larger and older scallops contribute more to SSB as measured by gonad weight compared to using meat weight. For this reason, increasing fishing mortality causes a larger decrease in SSB using gonads than with meats, which in turn causes the estimated F_{MSY} to be lower using gonad weights.

Table App A10-1. Summary of reference points for Georges Bank, Mid-Atlantic and combined. Med. $\text{Yield}_{0.58}$ and Med. $\text{SSB}_{0.58}$ are the yield and SSB from the median curves at the combined $F_{\text{MSY}} = 0.58$.

Region	MSY	F_{MSY}	B_{MSY}	Med. $\text{Yield}_{0.58}$	Med. $\text{SSB}_{0.58}$
GB	19974	0.49	34211	19813	29487
MA	25754	0.61	20243	25748	21093
Combined	45561	0.58	50580	49433	55895

Figures

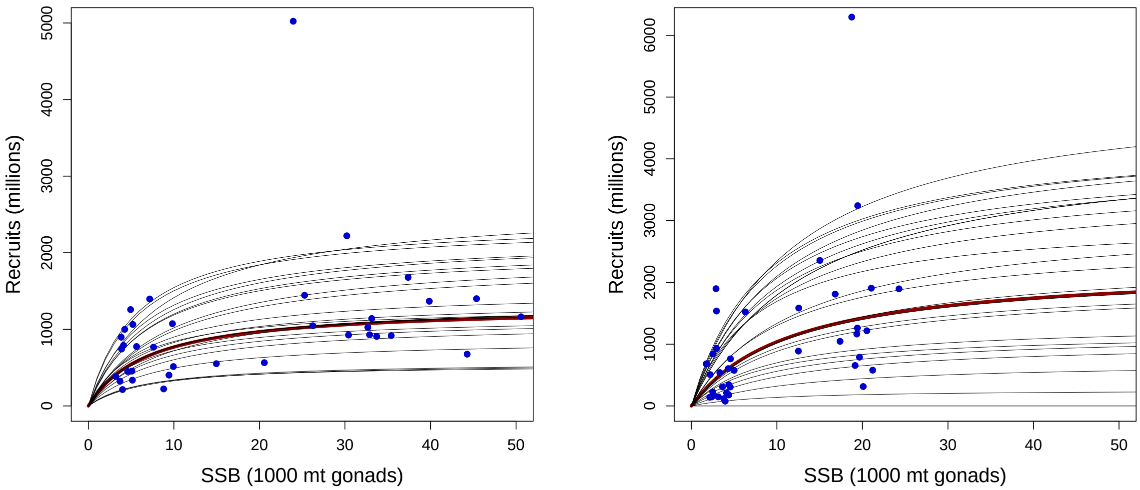


Figure App A10-1. Stock-recruit plots for Georges Bank (left), and Mid-Atlantic (right) (blue dots), with best fits to the data (think red line), and 25 example fits (thin black lines) from the SYM model.

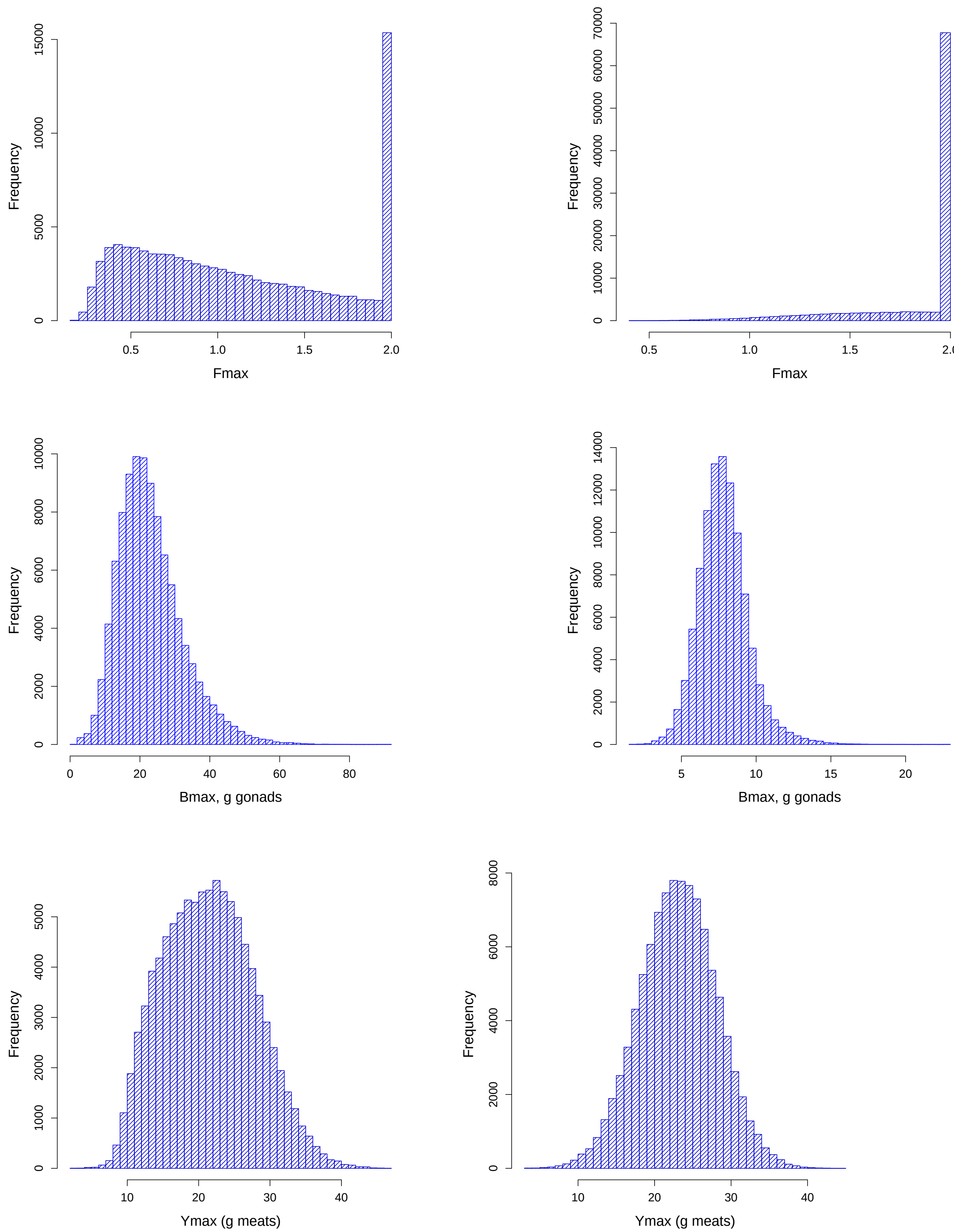


Figure App A10-2. Distribution of the yield per recruit reference point F_{MAX}, B_{MAX} and Y_{MAX} for Georges Bank (left), and the Mid-Atlantic (right).

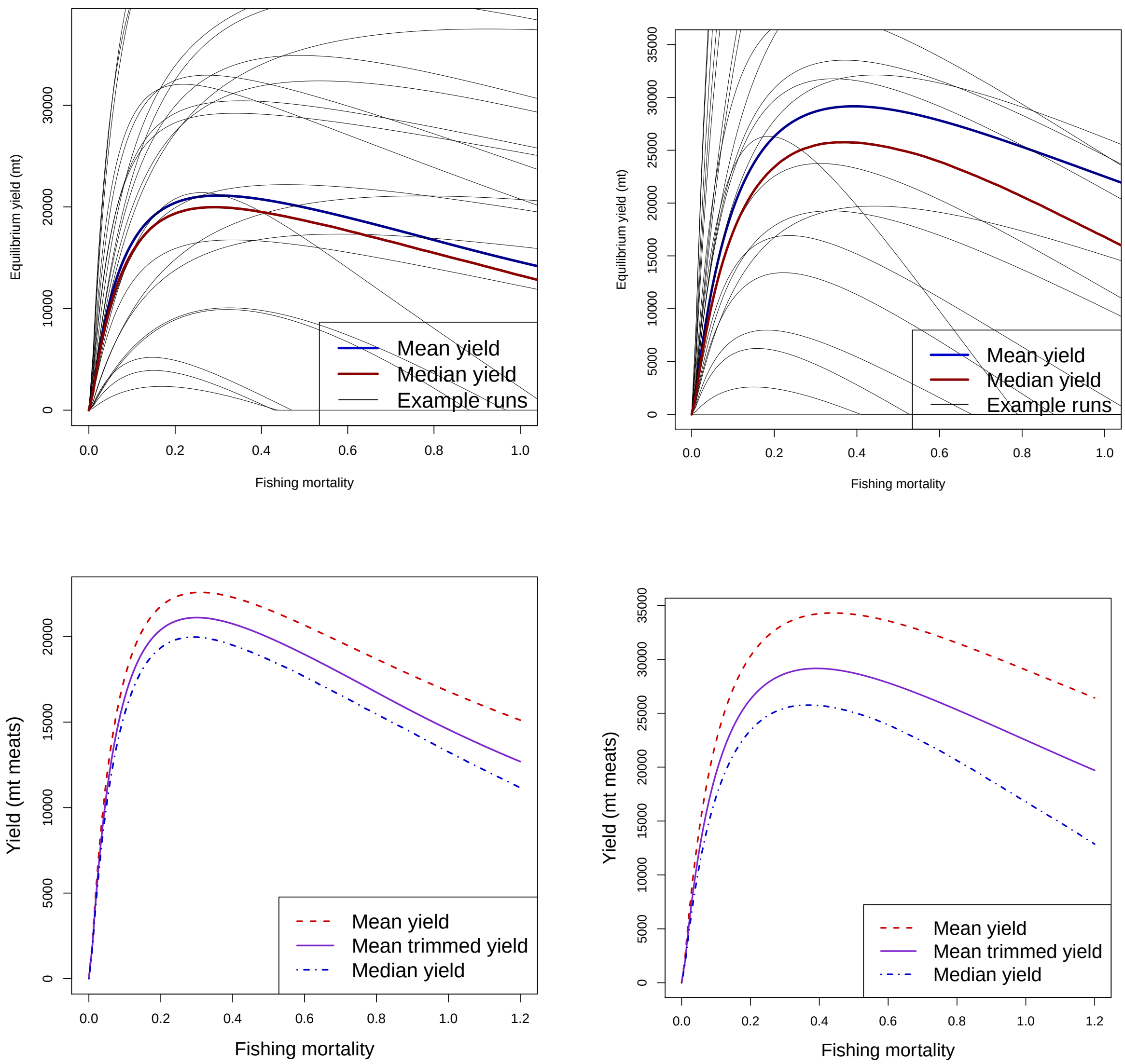


Figure App A10-3. Estimated mean and median yield curves with example runs (left), mean, trimmed mean, and median yield curves (center), and yield box plot for Georges Bank (above) and Mid-Atlantic (below).

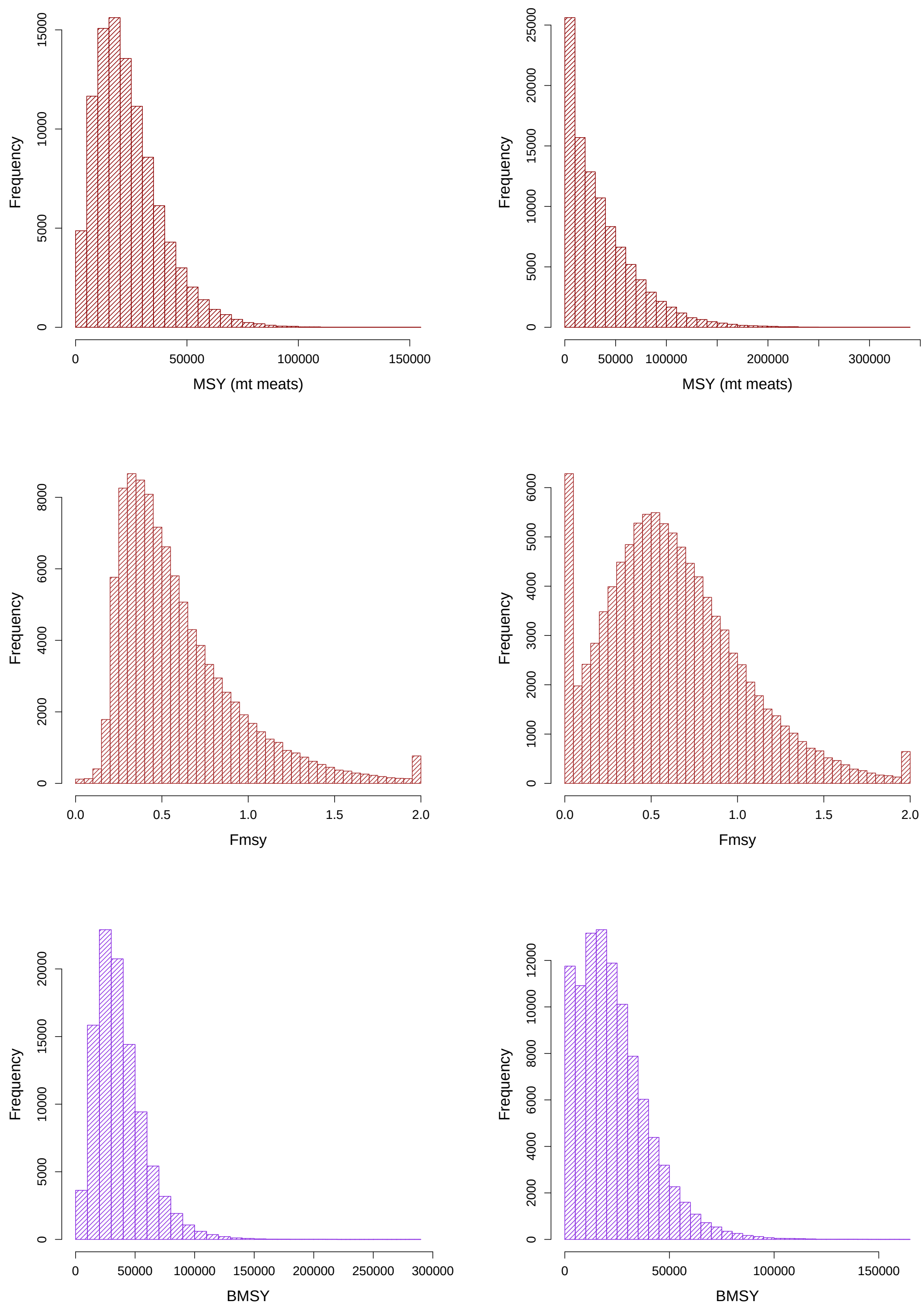


Figure App A10-4. Distributions of MSY, F_{MSY} and B_{MSY} for Georges Bank (left) and Mid-Atlantic (right).

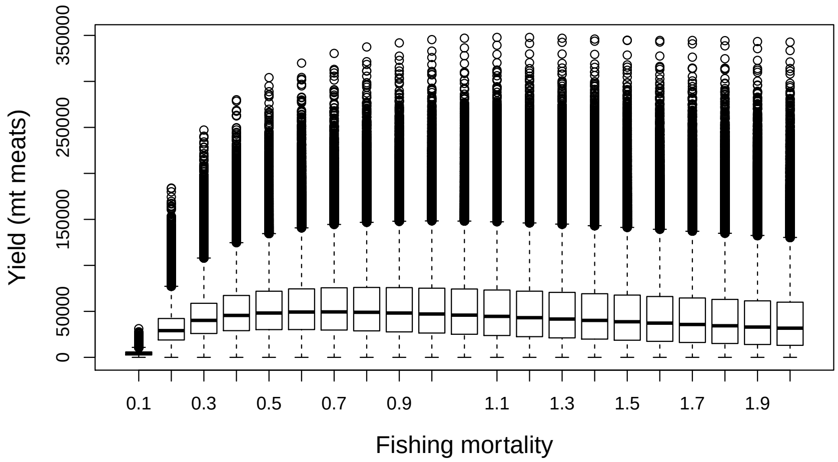


Figure App A10-5. Boxplots of SYM yields for Georges Bank and Mid-Atlantic combined.

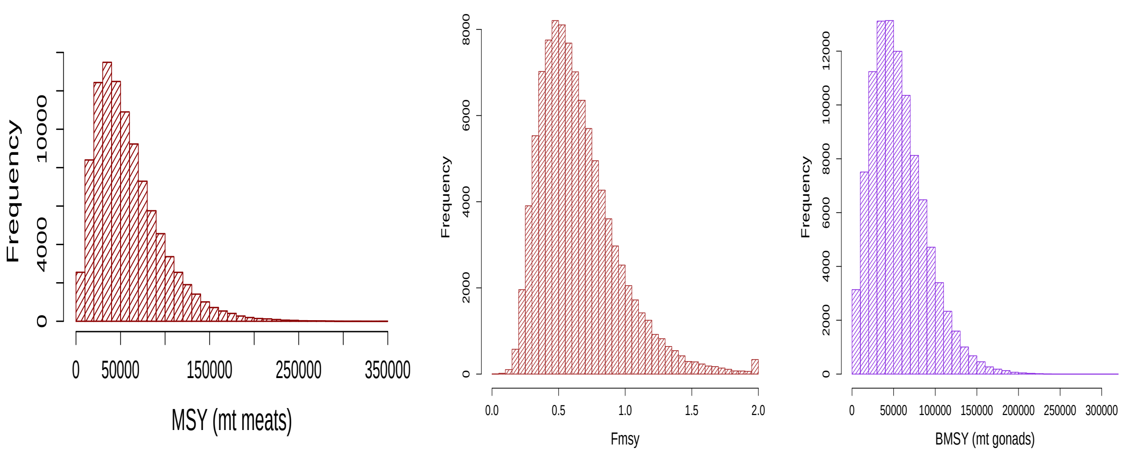


Figure App A10-6. Distributions of MSY, F_{MSY} and B_{MSY} for Mid-Atlantic and Georges Bank combined.

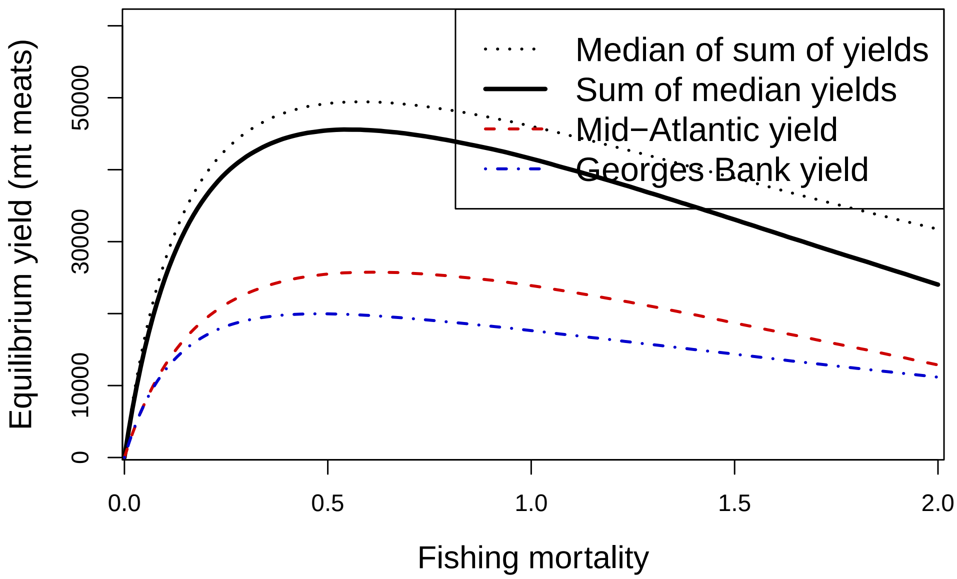


Figure App A10-7. Comparison between the sum of the regional median yields (solid line) with the median of the sum of the Georges Bank and Mid-Atlantic SYM runs (dotted line).

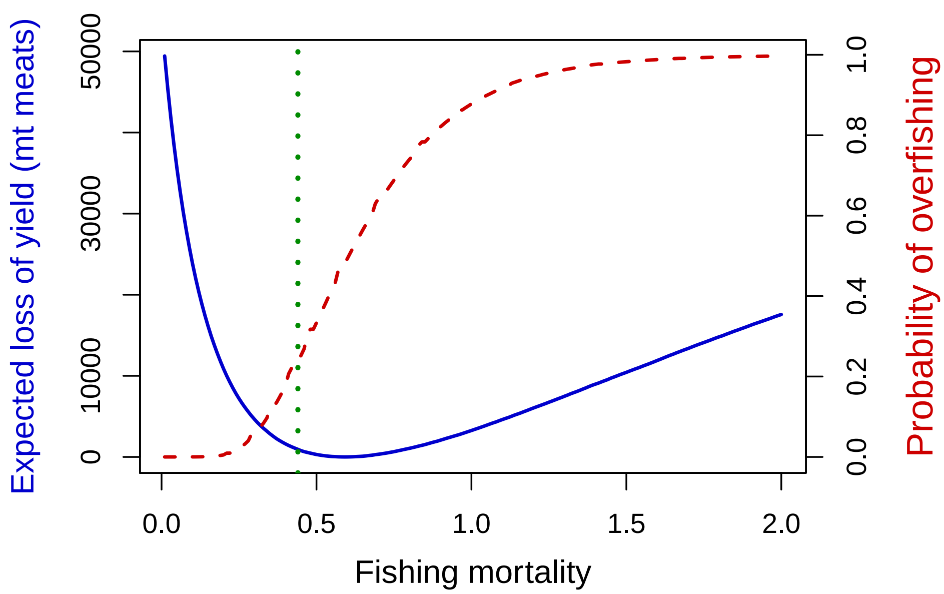


Figure App A10-8. The tradeoff between overfishing risk (red dashed lines) and loss of expected yield (solid blue line). The vertical green line marks the fishing mortality where the risk of overfishing is 0.25, which is the fishing mortality corresponding to the ACL/ABC.

B. Stock Assessment of Atlantic herring for 2018

Herring Working Group

The SARC 65 Atlantic Herring Working Group conducted a Data meeting (February 6-7, 2018) and a Model meeting (May 2-4, 2018) in the development of this assessment. The SAW/SARC Herring Working Group members are:

Jon Deroba – NEFSC Population Dynamics (Assessment lead)

John Manderson – NEFSC Coop Research

Chris Legault- NEFSC Population Dynamics

Deirdre Boelke – New England Fishery Management Council

Sarah Gaichas – NEFSC Ecosystem Dynamics and Assessment

Matt Cieri –ME DMR

Ashleen Benson – Landmark Fisheries

Gary Shepherd – NEFSC Population Dynamics (WG-Chair)

Executive Summary by Term of Reference (TOR)

TOR B1. *Estimate catch from all sources including landings and discards. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize uncertainty in these sources of data. Comment on other data sources that were considered but were not included.*

US catches were developed for the years 1965-2017 and were a sum of landings and self-reported discards. Discards have only been available since 1996, but were generally less than 1% of landings. Consequently, discards do not represent a significant source of mortality and a lack of historical discards is not considered problematic for the assessment. US catches were developed separately for fixed and mobile gear types. Catches from the New Brunswick, Canada, weir fishery were provided for the years 1965-2017 and were added to the US fixed gear catches for the purposes of assessment.

Total catches during 1964-2017 ranged from 44,613 mt in 1983 to 477,767 mt in 1968. Total catches during the past five years ranged from 50,250 mt in 2017 to 101,622 mt in 2013 and averaged 79,206 mt. Mobile gear catches have been the dominant gear type since about 1995.

TOR B2. *Present the survey data being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, age-length data, food habits, etc.). Characterize the uncertainty and any bias in these sources of data.*

Abundances (i.e., arithmetic mean numbers per tow) from the NMFS spring, fall, and summer shrimp bottom trawl surveys were used in the assessment model along with annual coefficients of variation and age composition when they were available. The trawl door used on the spring and fall surveys changed in 1985 and likely altered the catchability of the survey gear. Consequently, the spring and fall surveys were split into two time series between 1984 and 1985, and these were treated as separate indices in assessment models. The spring and fall surveys also used a different vessel (i.e., the Bigelow) beginning in 2009, and so these surveys were split again to account for this vessel change. Ultimately, the spring and fall surveys had three time stanzas: 1965-1984, 1985-2008, 2009-2017.

An acoustic index collected during the NMFS fall bottom trawl survey was also used as an index of herring abundance. This survey has no age composition data and so selectivity was knife-edged at age-3.

Several other indices of abundance were considered, but not used in the final assessment model. These indices included: NMFS winter survey, Massachusetts state surveys (spring and fall), joint Maine/New Hampshire state surveys (spring and fall), and an index based on food habits data.

TOR B3*. Estimate consumption of herring, at various life stages. Characterize the uncertainty of the consumption estimates. Address whether herring distribution has been affected by environmental changes.*

Fish food habits data from NEFSC bottom trawl surveys were evaluated for 12 herring predators. From these data, diet composition of herring, per capita consumption, and the amount of herring removed by the 12 predators were calculated. Combined with abundance estimates of these predators, herring consumption was summed across all predators as total herring consumption. Annual removal of herring amounted to 10s to 100s of thousands of mt by these predators. Annual removal ranged from 32,700mt in 1983 to 390,000mt in 2008. Amount of deaths due to input natural mortality in the stock assessment were compared to the estimates of predatory consumption as a general check of scale.

TOR B4*. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and estimate their uncertainty. Incorporate ecosystem information from TOR B3 into the assessment model, as appropriate. Include retrospective analyses (both historical and within -model) to allow a comparison with previous assessment results and projections, and to examine model fit.*

The base ASAP model made structural changes to the previous assessment (e.g., M, selectivity), included new index time series, and re-evaluated some other relatively minor issues (e.g., weak likelihood penalties). Of particular importance, however, was a change to M. Natural mortality in recent assessments varied by time and age, with values based on a combination of the Hoenig and Lorenzen methods (Hoenig 1983; Lorenzen 1996). In 2012, the natural mortality rates during 1996-2011 were increased from these base rates by 50% to resolve a retrospective pattern and to ensure that the amount of herring deaths due to input M were consistent with observed increases in estimated consumption of herring. In 2015, a retrospective pattern re-emerged and implied levels of consumption were no longer consistent with estimated consumption. Thus, assumptions about time- and age-varying M were reevaluated as part of this assessment. Ultimately, M equaled 0.35 for all years and ages in this assessment.

The base ASAP model estimated SSB in 2017 to be 141,473 mt, with SSB ranging from a minimum of 53,084 mt (1982) to a maximum of 1,352,700 mt (1967) over the entire time series. The base ASAP model estimated total January 1 biomass in 2017 to be 239,470 mt, ranging from a minimum of 169,860 mt (1982) to a maximum of 2,035,800 mt (1967) over the entire time series.

No common age is fully selected in both the mobile and fixed gear fishery. Consequently, the average F between ages 7 and 8 was used for reporting results related to fishing mortality ($F_{7\text{-}8}$), and this includes reference points. These ages are fully selected by the mobile gear fishery, which has accounted for most of the landings in recent years. $F_{7\text{-}8}$ in 2017 equaled 0.45. The all-time low of 0.13 occurred in 1965. The all-time high of 1.04 occurred in 1975.

Age-1 recruitment has been below average since 2013. The all-time high of 1.4 billion fish occurred in 1971. The estimates in 2009 and 2012 are still estimated to be relatively strong cohorts, as in previous assessments. The all-time low of 1.7 million fish occurred in 2016, and the second lowest of 3.9 million fish occurred in 2017. Four of the six lowest recruitment estimates have occurred since 2013 (2013, 2015, 2016, 2017).

The internal relative retrospective pattern suggested consistent overestimation of SSB with Mohn's Rho = 0.15, and underestimation of $F_{7\text{-}8}$ with Mohn's Rho = -0.11. The retrospective pattern for recruitment at age 1 was characterized by both positive and negative peels. The presence of the retrospective pattern was sensitive to the indices of abundance used in the model. The retrospective pattern was not severe enough, however, to warrant an adjustment for stock status determination or projections. Estimating catchability separately for the Bigelow years in 2009-2017 may also be aliasing other causes of the retrospective pattern, and so future herring assessments may have worsening retrospective patterns.

TOR B5. *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 were based on the fit of a Beverton-Holt stock-recruitment relationship, estimated internally to the ASAP model, and inputs (e.g., weights-at-age, natural mortality) from the terminal year of the assessment (i.e., 2014). Point estimates of the MSY BRPs equaled: MSY = 77,247 mt, F_{MSY} = 0.24, and SSB_{MSY} = 311,145 mt.

No stock-recruit relationship was able to be estimated in this assessment, therefore $F_{40\%}$ was used as a proxy for F_{MSY} and long-term projections were used to derive other MSY BRP proxies. F_{MSY} proxy = 0.51, SSB_{MSY} proxy = 189,000 mt (½ SSB_{MSY} = 94,500 mt), and MSY proxy = 112,000 mt.

The existing MSY reference points were based on estimates of a Beverton-Holt stock-recruit curve fit internally to the ASAP model. The ability to estimate the stock-recruit curve seems to have deteriorated in this assessment, but the ability of previous models to estimate a stock-recruit curve has also been noted as tenuous. The newly proposed reference points no longer rely on a poorly estimated stock-recruit relationship.

***TOR B6**. Make a recommendation about what stock status appears to be based on the existing model (from previous peer reviewed accepted assessment) and based on a new model or model formulation developed for this peer review.*

a. Update the existing model with new data and evaluate stock status (over fished and overfishing) with respect to the existing BRP estimates.

Given the Working Group's conclusion that MSY reference points based on the estimation of a stock-recruit curve were unjustified, and were likely unjustified in previous assessments, the existing BRPs are not meaningful. Similarly, evaluating stock status of the existing model with updated data to the existing MSY BRPs is not informative.

b. Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR B5).

The base ASAP model estimated $F_{7\text{-}8}$ in 2017 to be 0.45 and SSB in 2017 was 141,473 mt. Since the retrospective adjusted values do not fall outside of the confidence intervals of the base model estimates, no retrospective adjustment was warranted. A comparison of the base model values to the new MSY proxy reference points suggest that overfishing is not occurring and that the stock is not overfished. The error bars for $F_{7\text{-}8}$, however, included overfishing.

c. Include descriptions of stock status based on simple indicators/metrics.

The estimated numbers at age in 2017 indicate that the population is characterized by more age 6 fish than age 1 and age 2 combined. This result suggests a reliance on the ageing 2011 cohort (age 6 in 2017). If the estimated record low recruitments in recent years hold true, then the SSB is likely to remain relatively low and put the stock at relatively high risk of becoming overfished. Without improved recruitment, the probability of overfishing under recent catch levels is also likely relatively high.

TOR B7*. Develop approaches and apply them to conduct stock projections.*

a. Provide numerical annual projections (through 2021) and the statistical distribution (i.e., probability density function) of the catch at F_{MSY} or an F_{MSY} proxy (i.e. the overfishing level, OFL) (see Appendix to the SAW TORs). 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).

Short-term projections of future stock status were conducted based on the results of the base ASAP model. The projections did not account for any retrospective pattern because the Mohn's Rho adjusted values for stock status were within the 80% probability intervals of the 2017 point estimates of $F_{7\text{-}8}$ and SSB. If the Allowable Biological Catch (ABC) is fully utilized in 2018 (i.e., 111,000mt), then catch at F_{MSY} proxy in 2019=13,700mt, 2020=31,000mt, and 2021=55,700mt. If only half the ABC is utilized in 2018 (i.e., 55,000mt), then catch at F_{MSY} proxy in 2019=28,900mt, 2020=38,000mt, and 2021=59,400mt. As with the catches, future short-term stock status was also sensitive to the catch specified in 2018.

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. Identify reasonable projection parameters (recruitment, weight-at-age, retrospective adjustments, etc.) to use when setting specifications.

The Working Group agreed that the 2018 ABC of 111,000mt is unlikely to be fully utilized and that some lower value was more realistic, but that value is likely best determined by a technical group of the New England Fishery Management Council. The projections assumed that future recruitment will approach the mean for the time series (1965-2015). If recruitment continues to be below average, the projected catch increases may be overly optimistic.

c. Describe the stock's vulnerability (see "Appendix to the SARC TORs") to becoming overfished, and how this could affect the choice of ABC (or DEF, possibly even GH&I).

The unknown contributions of the Scotian Shelf (4WX), Gulf of Maine, and Georges Bank stocks can affect the stocks vulnerability to becoming overfished. The vulnerability of the stock has been demonstrated by the historical collapse of the Georges Bank component in the 1980s, which also demonstrated that the multiple spawning groups can be differentially impacted by fishing. Varying contributions from the Scotian Shelf (4WX) stock may also contribute to a retrospective pattern (see below).

In the short-term, the relatively poor recruitments in 2013-2017 will increase the vulnerability of the stock to becoming overfished. The 2016 and 2017 cohorts were imprecisely estimated and so estimates of these cohorts may change significantly in either direction in future assessments, and decisions should likely consider this uncertainty. Growth (i.e., weight at age) also continues to be relatively low when compared to the 1990s, and this seems to be a longer-term feature of the stock that also reduces production. The stock, however, seems to be capable of producing relatively large and small year classes regardless of growth, and so recruitment is likely the more significant driver of short-term vulnerability.

While this assessment had a retrospective pattern that did not warrant adjustments (i.e., via Mohn's Rho), the history of the Atlantic herring stock assessment suggests that resolutions to retrospective patterns are ephemeral, and so future herring assessments may have worsening retrospective patterns. Retrospective patterns are indicative of model misspecification, and this would increase the vulnerability of the stock to becoming overfished.

TOR B8*. If possible, make a recommendation about whether there is a need to modify the current stock definition for future assessments.*

Previous assessments concluded that there is likely sub-stock structure unaccounted for in the assessment, but that there is no ability to distinguish mixed survey and fishery catches to stock of origin. This lack of information on stock of origin precludes accounting for the sub-stock structure. An attempt was made to use an assessment model (Stock Synthesis) that accounted for stock structure on a coarse level (i.e., inside and outside of Gulf of Maine), but estimating area-specific recruitment and movement rates required unrealistic assumptions and the model generally performed poorly (e.g., poor convergence). The consequences of not accounting for stock structure are unclear, and therefore the need to modify the stock definition is also unclear. More certain, however, is that changing the stock definition and accounting for stock structure in the assessment is currently not possible. Continued research on the topic is warranted.

TOR B9. *For any research recommendations listed in SARC and other recent peer reviewed assessment and review panel reports, review, evaluate and report on the status of those research recommendations. Identify new research recommendations.*

Research recommendations from previous assessments were reviewed and progress on each updated and documented. Several new research recommendations were developed.

Herring Management Summary and History

Fisheries Management

The Atlantic herring fishery in the Northeastern U.S. operates from Maine to Cape Hatteras, North Carolina and from inshore to offshore waters on the edge of the continental shelf. The herring fishery uses predominantly single and paired mid-water trawl, bottom trawl, purse seine, and to a lesser extent, gillnet gear throughout the entire range. Herring is used primarily in the U.S. as bait for the American lobster and tuna fisheries, but is also frozen whole and canned for human consumption. Herring is managed in federal waters by the New England Fishery Management Council (NEFMC), and in state waters by the Atlantic States Marine Fisheries Commission (ASMFC). Individual states may set different regulations, such as possession/landing restrictions or spawning area closures. If state regulations differ from Federal regulations, herring permit holders must adhere to the more restrictive regulations.

Atlantic herring stocks were first managed in 1972 through the International Commission for the Northwest Atlantic Fisheries (ICNAF). ICNAF regulated the international fishery until the United States withdrew from the organization in 1976 with the passage of the Magnuson Fishery Conservation and Management Act (MSA). The Atlantic Herring Fishery Management Plan (FMP) was one of the first plans developed by the NEFMC, approved in 1978. In 1982, NMFS withdrew the Federal Herring FMP because of conflicts between state and federal regulations, and catch quotas for adult herring in the Gulf of Maine were not enforced in state waters. In the absence of a Federal FMP, Atlantic herring was placed on the prohibited species list, thereby eliminating directed fisheries by foreign nationals or joint ventures in the EEZ and requiring any herring bycatch by such vessels to be discarded.

While directed fishing for Atlantic herring was prohibited in Federal waters in 1983, the herring fishery in State waters was managed through an agreement among the States of Maine, New Hampshire, Massachusetts, and Rhode Island. The final draft of the "Interstate Herring Management Plan of Maine, New Hampshire, Massachusetts, and Rhode Island" was adopted in late 1983 and formally recognized by the ASMFC in 1987. The primary management tool was spawning closures, but as the size of the resource and fishery grew, this measure was not sufficient. The ASMFC developed the Atlantic Herring Fishery Management Plan in 1993 to address the growth of the herring resource, formalize the allocation process, and lay the foundation for a joint ASMFC-NEFMC management plan.

The New England Council's Herring FMP became effective on January 10, 2001 and included administrative and management measures to ensure effective and sustainable management of the herring resource. The FMP establishes Total Allowable Catches (TACs, now referred to as sub-ACLs, or annual catch limits) for each of four management areas as the primary control on fishing mortality (see Figure B1- 12 for current herring management areas). ASMFC adopted Amendment 2 to complement the federal Amendment 1 measures.

The federal FMP has been improved by several subsequent Amendment and Framework actions over the years (Amendments 1-7 and Frameworks 1-4). These actions are described briefly in the bullets below.

- **Framework Adjustment 1 (effective 2002)** set measures for fishing year 2002 and split the TAC for Area 1A into two seasonal components to prevent an early closure of the fishery in 1A.
- **Amendment 1 (effective 200**7) was developed to improve resource conservation, address new scientific information to the extent possible, minimize the potential for excess harvesting capacity in the fishery, and provide a platform to promote long-term economic stability for harvesters, processors, and fishing communities. A limited access program was implemented, management boundaries were adjusted, a seasonal purse seine/fixed gear only area was established for all of Area 1A from June-September, a three-year specifications process was developed, as well as several other adjustments to the management program.
- **Amendment 2 (effective 2008)** was part of an omnibus amendment developed by NMFS to ensure that all FMPs of the Northeast Region comply with the Standardized Bycatch Reporting Methodology (SBRM) requirements of the MSA.
- **Amendment 4 (effective in 2011)** implemented a process for establishing annual catch limits (ACLs) and accountability measures (AMs) in the herring fishery and brought the Herring FMP into compliance with the reauthorized MSA.
- **Framework 2 (effective 2014)** Framework 2 set catch specifications for the herring fishery for the 2013–2015 fishing years and established seasonal splits for management

areas 1A and 1B as recommended to NMFS by the Council, and other measures related to specifications.

- **Framework 3 (effective 2014**) to establish a process for setting river herring (alewife and blueback) and shad (American and hickory) catch caps for the herring fishery, including allocations for 2014 and 2015 fishing years.
- **Amendment 5 (effective 2014) to:** Improve the collection of real-time, accurate catch information; enhance the monitoring and sampling of catch at-sea; and address bycatch issues through responsible management by revising several program provisions, expanding vessel requirements to maximize observers' ability to sample catch-at-sea, minimize discarding of unsampled catch, addressing incidental catch of RH/S and revising criteria for MWT vessels in groundfish closed areas.
- **Framework 4 (effective 2016)** to further enhance catch monitoring and address discarding in the herring fishery by establishing requirements for herring dealers and restrictions on vessels when they release catch before it can be sampled by at-sea-observers (known as slippage).
- **Amendment 6 (effective 2016)** was part of an omnibus amendment to establish standards of precision for bycatch estimation for all Northeast fisheries (SBRM Amendment).
- **Amendment 3 (effective 2018)** was part of an omnibus amendment to all New England Council FMPs to address Essential Fish Habitat (EFH) consistent with the MSA.
- **Amendment 7 (scheduled 2018)** to allow the Councils to implement industry-funded monitoring above levels required by SBRM Amendment, including specific measures for an industry funded monitoring program for the herring fishery.
- **Amendment 8 (scheduled 2019)** to implement an ABC control rule and consider measures to address potential localized depletion and user conflicts in the herring fishery.

In general, the herring fishery is managed by a stock-wide annual catch limit (ACL) that is allocated to four distinct management areas (sub-ACLs, also known as management area quotas). The fishery allocations or specifications stem from the sub-ACLs and are currently set every three years. Due to the spatial structure of the Atlantic herring stock complex (multiple stock components that separate to spawn and mix during other times of the year), the total annual catch limit for Atlantic herring (stock-wide ACL/OY) is divided and assigned as sub-ACLs to four management areas (Figure B1- 12; Figure B1- 13). The best available information is used about the proportion of each spawning component of the Atlantic herring stock complex in each area/season and minimizing the risk of overfishing an individual spawning component to the extent practicable. Atlantic herring catch has been variable in recent years, but on average about 90,000 mt for the last decade or so, and lower in more recent year (2016-2017) (Table B1- **1**).

Other species are caught incidentally in the directed herring fishery. The species composition varies based on gear type, year, season, and area, but some of the species caught include: Atlantic mackerel, haddock, river herring (alewife and blueback herring), shad (American shad and hickory shad), whiting, and spiny dogfish. Due to the high-volume nature of the Atlantic herring fishery, non-target species are often retained once the fish are brought on board and sometimes sold as part of the overall catch if they are not separated. The herring fishery has been allocated a sub-ACL of Georges Bank haddock, and there are also bycatch caps for river herring/shad. The herring fishery is subject to accountability measures for both caps and directed herring fishing is prohibited in specific areas for the remainder of the fishing year when 95% of a bycatch cap is estimated to be caught.

TOR B1: *Estimate catch from all sources including landings and discards. Describe the spatial and temporal distribution of landings, discards and fishing effort. Characterize uncertainty in these sources of data. Comment on other data sources that were considered but were not included.*

Data from the United States

The catch data used to develop the US herring catch at age for 1965 to 2017 comes from a combination of NMFS Dealer reports and Vessel Trip Reports (VTR), NAFO reports, DFO Canada, Maine DMR, and other state landings reports. The reported catch is a sum of landings and self-reported discards, but discard estimates were not available in all years (Table B1- 1; Table B1- 2). Observed discards, however, were generally less than 1% of landings and do not represent a significant source of mortality (Table B1- 2; Wigley et al. 2011). Consequently, a lack of historical estimates of discards is not considered problematic for stock assessments. When data availability permitted, all the calculations used to produce the catch at age data below were done at the level of year, quarter, and gear type. Gear type was defined as either fixed or mobile gear. All trawl gears and purse seines were considered mobile, while all other gears (weirs, fyke nets, pound nets, etc.) were classified as fixed. These two aggregate gear types were used because biological data (e.g., lengths, ages, weights) were insufficient to do calculations on specific gear types. Weight-length relationships were similar between fixed and mobile gears, so data were combined for the gear types to estimate the parameters of this relationship. When no weight-length or length frequency data existed for a unique combination of year, quarter, and gear type, the calculations were then done at the level of year, semester (January-June or July-December), and gear type. Similarly, when no weight-length or length frequency data existed for a unique combination of year, semester, and gear type, the calculations were done at the level of year and gear type. Aggregations to the level of year and gear type were only necessary for 7 years for the fixed gear type (none for mobile gear). For the fixed gear type, no biological data were available in 15 years (1995, 1997, 2000, 2002-2005, 2008-2013, 2016-2017). US catch at age for the fixed gear type was consequently not developed in these years. Age-length keys were developed at the level of year, semester, and gear type. When an observed length had no corresponding age data, age samples for that length from the alternative gear type were used. Any remaining lengths with no corresponding age data were imputed based on a multinomial logistic model fit to the age observations at that length for the given year, semester, and gear type combination (Gerritsen et al., 2006). Data on sampling intensity is provided in Table B1- 3,

Table B1- 4, and Table B1- 5.

The Working Group had concerns that the purse seine gear had a selectivity distinct from trawl gears. More specifically, that the purse seine length frequencies were sometimes bi-modal and generally caught some smaller fish than trawl gears (Figure B1- 1; Figure B1- 2). Combining purse seines and trawl gears into the aggregate mobile gear and not accounting for these selectivity differences in an assessment model may induce diagnostic issues (e.g., residual patterns, retrospective patterns), especially since there have been temporal changes in the composition of the catch coming from each gear type (Figure B1- 3). One way to address this concern would be to develop separate catch at age matrices for purse seines and trawls, but a purse seine specific catch at age matrix could not be developed in time for this assessment and it is not clear whether biological data would support such efforts. Consequently, the working group considered some assessment models with time-varying selectivity for the mobile gear fleet as a way to evaluate the necessity of distinguishing between purse seines and trawl gears. The models with time-varying selectivity suggested that modeling purse seines separately from trawls was not supported (see TOR B4 for details).

US catch at age calculations did not include any spatial element because adding this to the stratification scheme resulted in a large number of combinations with little or no biological data (Table B1- 4; Table B1- 5). The gear types are also confounded in space, with nearly all the fixed gear catch coming from the Gulf of Maine (Figure B1- 4). Furthermore, the length frequencies of catches from different gears in the same area are clearly different, while length frequencies from the same gear in different areas are similar (Figure B1- 5; Figure B1- 6); suggesting that accounting for gear type was necessary while spatial differences were relatively inconsequential.

Data from New Brunswick, Canada

Department of Fisheries and Oceans, Canada, personnel (Rabindra Singh) provided catch at age data for the New Brunswick (NB), Canada, weir fishery during 1965-2017 (Table B1- 6). The NB weir fishery uses the same gears as the US fixed gear fishery and have similar age compositions (NEFSC 2012). Furthermore, some US weir operations are located in close geographic proximity to the NB weir fishery. Consequently, the working group agreed that data from the NB weir fishery and the US fixed gear fishery should be combined for the assessment.

Data summaries and assessment inputs

Catch in the mobile gear fishery peaked in the late 1960s and early 70s, largely due to efforts from foreign fleets (Figure B1- 7). Catch in this fishery was relatively stable during the 2000s, and has accounted for most of the Atlantic herring catches in recent years although the contribution has declined for the last four years. Catch in the fixed gear fishery has been variable, but has been relatively low since the mid-1980s (Figure B1- 7).

The US mobile gear fishery catches a relatively broad range of ages and some strong cohorts can be seen for several years (Figure B1- 8; Table B1- 7). In contrast, the fixed gear fishery harvests almost exclusively age 2 herring (Figure B1- 9; Table B1- 8).

A single matrix of catch weights at age was estimated as the catch weighted mean weights at age among the strata used to develop the US catch at age matrices and ultimately among the mobile and fixed gear fisheries (Table B1- 9). Weights at age for spawning stock biomass were estimated as the mean weights at age from the mobile gear fishery in quarter three (i.e., July-September; Table B1- 10). This data was used because the mobile gear fishery is relatively well sampled in all years and quarter three is when herring typically begin spawning. January 1 weights at age were estimated by using a Rivard calculation (Rivard 1982) of the SSB weights at age (Table B1- 11). Any missing weights at age in each matrix were replaced by a time series average from one of three time stanzas: 1965-1985, 1986-1994, or 1995-2017. These three time stanzas were used to accommodate the temporal changes in herring growth, mostly evident for older aged herring (e.g., Figure B1- 10). Since herring beyond age 8 experience relatively little growth, weight at age 8 was used to characterize fish in the plus group (age 8+) in the model.

Maturity at age was developed using samples from commercial catches during quarter three (July to September). Fish caught during this time of year were used because they reflect the maturity condition of herring just prior to or during spawning, and therefore are best for calculations related to spawning stock biomass. Fish of both sexes were included. Fish of unknown maturity were removed from the analysis (codes 0 and 9 in the dataset). Immature fish were defined as those classified as immature I or immature II (codes 1 and 2, respectively in dataset) while all other fish were considered mature (3=ripe, 4=eyed, 5=ripe and running, 6=spent, 7=resting). The observed proportions mature at age from quarter three of each year were input to assessments and used in the calculation of SSB (Table B1- 12). Using predicted proportions at age from a generalized additive model fit to the annual observations was considered (NEFSC 2012), but sample sizes were generally considered large enough that such modeling to reduce the effect of measurement uncertainty was deemed unnecessary (Table B1-13). Microscopic verifications of the maturity classifications was conducted, as was an exploration of the consequences of possible spring or skipped spawning (Appendix B7).

Spatial distribution of fishing effort

The fishery tends to operate as expected given what is known about Atlantic herring migration patterns. In the winter, fishery landings tend to be more southerly than other times of year. As warming occurs through the spring and summer and herring migrate to the north, fishery landings occur more frequently throughout the Gulf of Maine. As fish separate into components to spawn in the fall, fishery landings span the Gulf of Maine and Georges Bank (Figure B1- 11). Also see:

http://noaa.maps.arcgis.com/apps/webappviewer/index.html?id=5d3a684fe2844eedb6beacf1169ca854

Other data sources discussed

The Northeast Fisheries Science Centers (NEFSC) Cooperative Research Branch's Study Fleet pilot program began field-testing data collection with electronic systems in late 2001. The Goals were to (1) to assemble a group of commercial fishermen to collect high resolution (haul-by-haul) self-reported data on catch, effort and environmental conditions during usual fishing operations, (2) develop and implement an electronic data collection system. The program was intended to ultimately provide stock assessment scientists with more precise and accurate fishery-dependent data and to improve the understanding of catch rates and species assemblages through examination of variables such as time of day, temperature, depth, tidal strength, and sediment type (Palmer et al 2007). The Fisheries Logbook and Data Recording Software (FLDRS) was established in 2006 as a product of this pilot work. FLDRS collects information at both the trip and haul level including detailed information of effort, catch and apportionment.

From 2006-2013 the number of vessels using FLDRS while participating in the commercial Atlantic herring fishery varied from 1-7. Most of these vessel participated in the small-mesh bottom trawl fishery off Rhode Island. In late 2014, through collaboration with the Pacific States Marine Fisheries Commission and with cooperation with the Massachusetts Division of Marine, Fisheries Cooperative Research staff deployed FLDRS on the midwater and paired midwater Atlantic herring fleet. This greatly increased the number vessels using FLDRS, the amount of data collected and expanded the spatial extent of coverage. In 2016, vessels reporting haul-by-haul using FLDRS accounted for >40% of the total landings. This and future information should be able address specific research and management questions.

A more detailed description of the program is an Appendix B6.

Management measures that may affect catch

The quota allocated to the fishery (stock wide ACL) has decreased over time, over 200,000 mt in the 1990s, and between 100-150,000mt over the last decade. As the total allocation has decreased, the Atlantic herring fishery has become more fully used in recent years, with some exceptions. These exceptions could be related to resource abundance, but there are a variety of factors that have likely caused under harvests of catch limits, including management measures in the plan.

For example, in 2015 the fishery in Area 3 became constrained by the Georges Bank Haddock catch cap accountability measure. Area 3 closed to midwater trawl (MWT) gear during the season, and under 75% of the herring quota was harvested in that area before the haddock cap was reached and directed fishing with MWT gear was prohibited. This closure also had impacts on 2016 catch in Area 3 because the restriction is based on the multispecies fishing year, which is May 1 through April 30. Therefore, directed herring fishing in Area 3 was also prohibited in January 1 – April 30 in 2016, making it more difficult for the fishery to harvest the full allocation in Area 3 in the remaining months of the year. Therefore, the utilization of Area 3 herring quota was potentially impacted by the haddock catch cap in both 2015 and 2016.

In addition, there are other measures in place that have the potential to limit herring landings, especially when they are combined, potentially having cumulative impacts that limit flexibility and reduce the ability for the fishery to harvest the full TAC in each area. For example, there are various seasonal restrictions that limit when vessels can fish in certain areas. Table B1- 14 summarizes some of these restrictions by month. Despite these restrictions, the sub-ACL for Area 1A and 1B have been fully harvested in most years. More recently, ASMFC has also placed restrictions on Area 1A that has further reduced flexibility and impacted fishing behavior in that area. In 2018 Addendum 1 to Amendment 3 of the ASMFC plan implemented weekly catch limits and restrictions on carrier vessels in Area 1A, in addition to the days-out measures that control the number of potential harvesting days per week. In 2018, the full Area 1A sub-ACL was not harvested, in part potentially due to the weekly catch limits, as well as implementation of spawning closures that prevent herring fishing by any gear type in different areas within Area 1A.

While bycatch caps have not been reached in many cases, there have certainly been a number of years that the fleet has approached them, and adjusted fishing behavior mid-season to avoid closures. As the fleet approaches the cap, avoidance behaviors have been observed such as moving to new areas and that can impact full utilization of the herring sub-ACLs. In addition to the example explained above for the haddock cap in 2015 and 2016, fishing behavior was impacted in 2017 around Cape Cod when the RH/S cap reached about 80%, vessels voluntarily avoided that area for the remainder of the fishing year to avoid exceeding the cap. Furthermore, fishing behavior and ability to harvest sub-ACLs was definitely impacted in 2018 when the RH/S cap was reached in March closing the MWT herring fishery in all SNE/MA waters. At that time only 20% of the Area 2 herring sub-ACL has been harvested.

In addition to bycatch measures that can impose in-season restrictions on directed herring fishing, the federal herring plan also includes several measures that impose seasonal restrictions for other purposes. For example, Area 1B is closed every year from January 1 through April 30, primarily to provide more herring landings when it is needed most for the bait market, late spring through summer. This quota is a small fraction of the overall herring catch, under 5%, but seasonal closures can limit flexibility and if the fish are not in that area during other months when the area is open, it can potentially impact the ability to harvest the sub-ACL for that area. Similarly, Area1A is closed to all herring fishing in January 1 through May 30, and only open to purse seine gear June 1 – September 30. While the Area 1A sub-ACL is usually fully harvested, these seasonal restrictions, especially when combined with spawning closures imposed by ASMFC, can limit flexibility and potentially impact the ability of the fishery to harvest the full sub-ACL in that area.

Another measure that may also make it more difficult to fully harvest sub-ACLs is the requirement to carry an at-sea observer if a vessel wants to fish in a groundfish closed area. Amendment 5 to the Herring FMP allowed midwater trawl vessels to fish in Closed Areas is a fishery observer is onboard. If observers are not available, herring vessels are prohibited from fishing in those areas (Closed Areas 1 and II). If herring is more concentrated in groundfish closed areas in a particular year or season, and vessels are unable to get observers, it may be more difficult to harvest the Area 3 sub-ACL since those areas cover a relatively large portion of Area 3 where herring are typically found. While some of these restrictions and closed areas have recently changed under the Omnibus Habitat Amendment 2, many of the requirements for herring vessels to carry observers in groundfish closed areas remain the same.

Finally, many herring vessels are active in other fisheries so in some cases, effort in other fisheries can impact when and how much herring fishing occurs during a fishing year. For example, if squid fishing or mackerel fishing is productive, some vessels that have permits in those fisheries will decide to prosecute those fisheries that often have higher revenues and prices. Conditions change every year, and if a herring sub-ACL is not harvested in a particular year, that may not be related to herring resource conditions or herring management restrictions; it is possible that availability or market conditions in other fisheries drives herring fishing activity, at least partially. If herring vessels are focused in other fisheries, i.e. mackerel or squid, herring fishing patterns can be impacted. In summary, herring management is complex, and trends in catch alone may not be reflective of resource conditions. If sub-ACLs are not fully harvested it can be related to resource availability, but there are a web of management measures in place that can inhibit herring fishing activity and full utilization of sub-ACLs.

Table B1- 1 Atlantic herring landings (mt)

Year	US Fixed	New Brunswick Weir	US Mobile	US Fixed + NB Weir (mt)	Total
1965	36440	31682	58161	68122	126282
1966	23178	35602	162022	58780	220802
1967	17458	29928	258306	47386	305692
1968	24565	32111	421091	56676	477767
1969	9007	25643	362148	34650	396798
1970	4316	15070	302107	19386	321493
1971	5712	12136	327980	17848	345828
1972	22800	31893	225726	54693	280419
1973	7475	19053	247025	26528	273553
1974	7040	19020	203462	26060	229522
1975	11954	30816	190689	42770	233459
1976	35606	29207	79732	64813	144545
1977	26947	19973	56665	46920	103585
1978	20309	38842	52423	59151	111574
1979	47292	37828	33756	85120	118876
1980	42325	13526	57120	55851	112971
1981	58739	19080	26883	77819	104702
1982	15113	25963	29334	41076	70411
1983	3861	11383	29369	15244	44613
1984	471	8698	46189	9169	55358
1985	6036	27864	27316	33900	61216
1986	2120	27885	38100	30005	68104
1987	1986	27320	47971	29306	77277
1988	2598	33421	51019	36019	87038
1989	1761	44112	54082	45873	99954
1990	670	38778	54737	39448	94184
1991	2133	24574	78032	26707	104739
1992	3839	31968	88910	35807	124717
1993	2288	31572	74593	33860	108452
1994	539	22242	63161	22781	85943
1995	6	18248	106179	18254	124433
1996	631	15913	116788	16544	133332
1997	275	20551	123824	20826	144651
1998	4889	20092	103734	24981	128715
1999	653	18644	110200	19298	129497
2000	54	16830	109087	16884	125971
2001	27	20210	120548	20237	140785
2002	46	11874	93176	11920	105096
2003	152	9008	102320	9160	111480
2004	96	20685	94628	20781	115409
2005	68	13055	93670	13123	106793
2006	1007	12863	102994	13870	116864
2007	403	30944	81116	31347	112462
2008	31	6448	84650	6479	91129
2009	98	4031	103458	4129	107587
2010	1263	10958	67191	12221	79413
2011	422	3711	82022	4133	86155
2012	9	504	87162	513	87675
2013	9	6431	95182	6440	101622
2014	518	2149	92566	2667	95233
2015	738	146	80465	884	81350
2016	1208	4060	62307	5267	67574
2017	258	2103	47889	2361	50250

Table B1- 2 Atlantic herring discards (mt), landings (mt), and the ratio of the two quantities for the fixed and mobile fleets

Year	Discards (mt)		Landings (mt)		D/L	
	Fixed	**Mobile**	**Fixed**	**Mobile**	**Fixed**	**Mobile**
1996	13	131	666	116609	0.02	0.00
1997	29	225	342	123504	0.08	0.00
1998	7	188	4925	103503	0.00	0.00
1999	5	48	704	110096	0.01	0.00
2000	6	317	62	108756	0.10	0.00
2001	11	539	54	119971	0.21	0.00
2002	3	38	52	93129	0.07	0.00
2003	8	22	159	102284	0.05	0.00
2004	9	477	103	94136	0.08	0.01
2005	3	299	76	93359	0.03	0.00
2006	1	199	1029	102772	0.00	0.00
2007	3	52	418	81045	0.01	0.00
2008	3	526	41	84111	0.07	0.01
2009	2	460	158	102928	0.01	0.00
2010	33	230	1511	66673	0.02	0.00
2011	5	174	582	81683	0.01	0.00
2012	7	145	176	86843	0.04	0.00
2013	3	166	78	94944	0.04	0.00
2014	1	292	533	92259	0.00	0.00
2015	1	83	757	80363	0.00	0.00
2016	2	122	1253	62137	0.00	0.00
2017	0	74	274	47798	0.00	0.00

Table B1- 3 Number of commercial trips sampled for Atlantic herring biological data

	Number of Trips		
Year	Fixed	Mobile	Total
1965	353	13	366
1966	221	29	250
1967	241	66	307
1968	308	14	322
1969	300	25	325
1970	117	40	157
1971	103	91	194
1972	120	103	223
1973	95	69	164
1974	144	146	290
1975	154	131	285
1976	238	150	388
1977	248	106	354
1978	232	276	508
1979	559	121	680
1980	192	268	460
1981	352	100	452
1982	127	105	232
1983	62	134	196
1984	10	161	171
1985	54	88	142
1986	18	56	74
1987	21	79	100
1988	24	77	101
1989	29	68	97
1990	37	107	144
1991	24	99	123
1992	38	126	164
1993	32	125	157
1994	15	75	90
1995	0	124	124
1996	6	137	143
1997	0	213	213
1998	10	173	183
1999	3	206	209
2000	0	195	195
2001	2	214	216
2002	0	200	200
2003	0	155	155
2004	0	141	141
2005	0	186	186
2006	1	211	212
2007	1	147	148
2008	0	125	125
2009	0	123	123
2010	0	119	119
2011	0	119	119
2012	0	120	120
2013	0	132	132
2014	1	142	143
2015	2	119	121
2016	0	93	93
2017	0	103	103

Table B1- 4 Number of Atlantic herring length samples by fleet and spatial area

Year	# Length Samples		Total	# Length Samples		Total
	Fixed	Mobile		Gulf of Maine	Other	
1965	20671	715	21386	21386	0	21386
1966	11123	1401	12524	36766	19888	56654
1967	11410	12263	23673	27583	22156	49739
1968	16521	698	17219	36167	18944	55111
1969	14502	2910	17412	50050	30086	80136
1970	4171	20099	24270	34914	26580	61494
1971	7879	41157	49036	21537	44213	65750
1972	12945	33970	46915	35384	23685	59069
1973	4682	33633	38315	26913	27120	54033
1974	13340	45394	58734	37424	29368	66792
1975	14816	35026	49842	32797	31181	63978
1976	21267	31556	52823	43546	21457	65003
1977	23336	20257	43593	45443	11316	56759
1978	11574	15154	26728	44045	863	44908
1979	28815	8479	37294	37108	186	37294
1980	8867	19448	28315	28115	200	28315
1981	17433	6095	23528	23428	100	23528
1982	6327	6369	12696	12496	200	12696
1983	3100	7915	11015	11015	0	11015
1984	500	9595	10095	10095	0	10095
1985	2700	6288	8988	8888	100	8988
1986	896	3850	4746	4746	0	4746
1987	1050	5344	6394	6394	0	6394
1988	1200	5340	6540	6440	100	6540
1989	1450	4850	6300	6300	0	6300
1990	1847	6727	8574	8574	0	8574
1991	1200	6963	8163	8113	50	8163
1992	1900	9643	11543	11543	0	11543
1993	1671	6265	7936	7879	57	7936
1994	755	3717	4472	4072	400	4472
1995	0	6183	6183	5895	288	6183
1996	300	7181	7481	6483	998	7481
1997	0	10905	10905	8855	2050	10905
1998	500	8656	9156	5517	3639	9156
1999	150	10296	10446	9095	1351	10446
2000	0	9159	9159	6852	2307	9159
2001	100	10078	10178	6252	3926	10178
2002	0	9640	9640	7569	2071	9640
2003	0	7712	7712	4656	3056	7712
2004	0	7099	7099	4658	2441	7099
2005	0	9280	9280	5683	3597	9280
2006	50	11005	11055	5869	5186	11055
2007	45	7730	7775	4984	2791	7775
2008	0	6359	6359	3744	2615	6359
2009	0	6157	6157	3426	2731	6157
2010	0	6127	6127	2737	3390	6127
2011	0	6248	6248	3579	2669	6248
2012	0	6307	6307	2655	3652	6307
2013	0	6676	6676	2255	4421	6676
2014	50	7160	7210	3584	3626	7210
2015	89	5824	5913	3032	2881	5913
2016	0	4868	4868	2850	2018	4868
2017	0	5311	5311	3893	1418	5311

Table B1- 5 Number of Atlantic herring age samples by fleet and spatial area

Year	# Age Samples		Total	# Age Samples		Total
	Fixed	Mobile		Gulf of Maine	Other	
1965	2794	309	3103	3103	0	3103
1966	2337	481	2818	3862	1032	4894
1967	2250	1079	3329	3733	1190	4923
1968	2431	208	2639	3649	976	4625
1969	2149	392	2541	4185	1566	5751
1970	1173	1582	2755	3063	1444	4507
1971	1654	3248	4902	2982	2854	5836
1972	1521	2904	4425	3611	1516	5127
1973	940	2270	3210	2562	1396	3958
1974	1366	3251	4617	3329	1692	5021
1975	1848	2799	4647	3506	1973	5479
1976	1985	2632	4617	4135	1298	5433
1977	2070	2064	4134	4069	785	4854
1978	1272	2584	3856	5013	263	5276
1979	2178	1360	3538	3460	78	3538
1980	1285	2197	3482	3434	48	3482
1981	1370	1166	2536	2488	48	2536
1982	868	1339	2207	2105	102	2207
1983	385	1372	1757	1757	0	1757
1984	102	1971	2073	2073	0	2073
1985	344	1342	1686	1665	21	1686
1986	177	981	1158	1158	0	1158
1987	208	1384	1592	1592	0	1592
1988	202	1260	1462	1418	44	1462
1989	200	983	1183	1183	0	1183
1990	215	1433	1648	1648	0	1648
1991	197	1394	1591	1571	20	1591
1992	284	1887	2171	2171	0	2171
1993	257	1954	2211	2159	52	2211
1994	127	1268	1395	1226	169	1395
1995	0	1582	1582	1474	108	1582
1996	67	1735	1802	1440	362	1802
1997	0	2425	2425	1777	648	2425
1998	112	2059	2171	1281	890	2171
1999	37	2023	2060	1692	368	2060
2000	0	2023	2023	1380	643	2023
2001	41	2394	2435	1410	1025	2435
2002	0	2521	2521	1987	534	2521
2003	0	2146	2146	1206	940	2146
2004	0	1920	1920	1180	740	1920
2005	0	2417	2417	1384	1033	2417
2006	12	2427	2439	1246	1193	2439
2007	11	1829	1840	1085	755	1840
2008	0	1973	1973	1135	838	1973
2009	0	1950	1950	906	1044	1950
2010	0	2115	2115	815	1300	2115
2011	0	1634	1634	861	773	1634
2012	0	1529	1529	660	869	1529
2013	0	1979	1979	518	1461	1979
2014	14	2085	2099	861	1238	2099
2015	25	1547	1572	670	902	1572
2016	0	1332	1332	610	722	1332
2017	0	947	947	511	436	947

Table B1- 6 New Brunswick, Canada, Atlantic herring weir catches (numbers)

	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11+
1965	992000	852368000	65449000	53194000	6897000	240000	116000	77000	0	0	0
1966	3899000	151087000	432061000	49134000	30162000	1182000	28000	13000	22000	29000	0
1967	127374000	194566000	57421000	111164000	12573000	4326000	1170000	119000	3000	0	0
1968	2409000	758766000	51933000	25098000	31655000	3957000	3141000	757000	77000	10000	0
1969	71191000	375586000	101361000	5067000	9845000	7692000	6449000	2025000	300000	3000	0
1970	3553000	348916000	9924000	12598000	6034000	3788000	2356000	893000	61000	10000	0
1971	92253000	183690000	37348000	7925000	3912000	2078000	3068000	1195000	332000	52000	62000
1972	8102000	660547000	6446000	10817000	4226000	2005000	1029000	1161000	354000	34000	11000
1973	31803000	149051000	125965000	14773000	1038000	529000	57000	121000	56000	4000	22000
1974	3259000	246044000	43483000	31147000	1227000	48000	54000	35000	38000	27000	37000
1975	16880000	462977000	57228000	9555000	16380000	2183000	1111000	916000	294000	158000	174000
1976	51791000	199268000	104624000	19989000	14911000	10128000	1601000	366000	457000	193000	112000
1977	459109000	122921000	10305000	20941000	7237000	7050000	4674000	230000	5000	0	1000
1978	213778000	894372000	52125000	3665000	810000	1064000	280000	132000	0	0	0
1979	2396000	423731000	247356000	12236000	822000	841000	479000	1005000	190000	0	0
1980	257995000	5325000	62087000	21615000	924000	125000	124000	67000	57000	63000	0
1981	53336000	294720000	18781000	10199000	5368000	306000	46000	34000	27000	0	0
1982	30210000	395416000	73197000	3199000	1795000	1596000	196000	42000	68000	0	0
1983	2532000	135283000	21684000	7526000	444000	398000	189000	0	0	0	0
1984	14353000	82920000	17292000	5658000	4332000	611000	251000	15000	85000	0	0
1985	20295000	385381000	45879000	17936000	7411000	3507000	304000	71000	73000	0	0
1986	3210000	136292000	119736000	24061000	10636000	4644000	2272000	335000	94000	66000	9000
1987	35677000	129348000	47981000	53150000	22941000	7097000	2472000	606000	173000	96000	0
1988	76053000	347765000	45078000	22366000	38843000	14212000	1680000	101000	247000	1000	9000
1989	26855000	331014000	81410000	21442000	22723000	43020000	11532000	3095000	810000	121000	249000
1990	12576000	454802000	69004000	30689000	6358000	7230000	15031000	3420000	2520000	620000	310000
1991	5530000	338263000	44450000	23618000	9532000	3154000	2620000	3436000	1461000	267000	150000
1992	799000	375772000	97678000	36438000	10378000	3992000	1613000	1360000	558000	245000	44000
1993	1718000	244079000	106099000	37186000	23218000	12260000	4915000	1120000	1101000	864000	175000
1994	1986000	291956000	63902000	9972000	16258000	9332000	3893000	1479000	1080000	544000	334000
1995	57844000	259741000	40122000	14803000	1822000	1567000	1549000	30000	0	0	0
1996	5351000	269431000	22390000	9342000	4302000	1147000	1273000	426000	38000	9000	2000
1997	9309000	216159000	113197000	11333000	3597000	523000	206000	95000	11000	0	0
1998	440000	387723000	36062000	9595000	3404000	1842000	297000	69000	25000	1000	0
1999	167679	106127770	100722414	11903080	9057476	3968746	1365910	154714	3950	3909	8434
2000	1665260	256784705	8082353	7871514	5376908	1416883	521421	101422	190	0	0
2001	1320542	113200008	119194370	8018810	5712883	1823813	588419	95017	101838	2081	0
2002	31858563	180051484	16260128	11528872	3020062	432017	101972	48714	18817	19556	11509
2003	11470685	162210672	15488021	2912807	1987414	456774	128273	27994	27934	13587	12487
2004	6711148	184123131	103911073	18753448	2537258	1751082	305572	358008	92686	31016	45060
2005	1152478	102401310	73912834	19379433	4269372	533907	268965	109207	13692	450	2466
2006	201206756	139578332	25001134	3786465	3705592	1275745	684331	138912	6539	842	1725
2007	6322626	571186007	31093039	2644604	812012	1274805	419924	63163	13985	1667	220
2008	27894408	122185141	19783355	203318	82469	105017	120277	45529	17154	1270	76
2009	12987445	99615384	3302958	141258	3842	1285	832	237	79	0	0
2010	7224	371400620	16967663	522825	463391	29356	21701	28636	16157	5620	612
2011	14254158	44743409	21030320	2153126	262891	61326	3942	0	0	0	0
2012	23399306	4309339	467710	611200	232280	62349	16952	3094	1028	543	287
2013	35483478	126916853	10474516	642836	435504	216325	52156	13511	993	0	253
2014	21037481	38784963	1422384	711520	288369	218518	75676	30661	8797	355	1892
2015	429076	5944638	49852	6985	3867	1622	748	0	748	0	0
2016	832028	61493618	9108761	1707005	657193	253407	145416	180769	15169	5202	0
2017	2427711	13588301	2360908	5096051	1860612	1233993	583536	284588	82045	22132	0

Table B1- 7 Mobile fleet age composition proportions at age

	1	2	3	4	5	6	7	8+
1965	0.001	0.965	0.026	0.008	0.000	0.000	0.000	0.000
1966	0.000	0.416	0.529	0.017	0.027	0.009	0.002	0.000
1967	0.000	0.048	0.213	0.168	0.094	0.138	0.265	0.075
1968	0.011	0.716	0.210	0.039	0.024	0.000	0.000	0.000
1969	0.096	0.257	0.486	0.062	0.013	0.009	0.019	0.058
1970	0.075	0.250	0.111	0.201	0.143	0.074	0.063	0.082
1971	0.053	0.028	0.209	0.182	0.184	0.125	0.084	0.135
1972	0.017	0.234	0.087	0.141	0.182	0.164	0.098	0.077
1973	0.017	0.153	0.524	0.139	0.052	0.038	0.043	0.034
1974	0.008	0.103	0.126	0.629	0.070	0.023	0.021	0.020
1975	0.007	0.025	0.066	0.140	0.635	0.061	0.029	0.037
1976	0.000	0.007	0.176	0.089	0.114	0.545	0.040	0.030
1977	0.013	0.174	0.078	0.264	0.068	0.076	0.293	0.033
1978	0.008	0.201	0.263	0.119	0.191	0.026	0.037	0.155
1979	0.000	0.209	0.332	0.225	0.075	0.092	0.014	0.053
1980	0.001	0.106	0.425	0.363	0.053	0.015	0.022	0.014
1981	0.000	0.107	0.039	0.495	0.299	0.033	0.010	0.017
1982	0.002	0.233	0.200	0.040	0.297	0.186	0.020	0.023
1983	0.033	0.369	0.243	0.191	0.011	0.079	0.062	0.013
1984	0.000	0.222	0.464	0.160	0.107	0.007	0.026	0.013
1985	0.001	0.178	0.157	0.401	0.154	0.086	0.004	0.019
1986	0.005	0.291	0.431	0.105	0.101	0.043	0.019	0.005
1987	0.001	0.155	0.306	0.428	0.057	0.038	0.010	0.005
1988	0.001	0.124	0.230	0.210	0.321	0.077	0.026	0.011
1989	0.000	0.322	0.259	0.106	0.093	0.160	0.041	0.019
1990	0.000	0.172	0.333	0.135	0.066	0.073	0.131	0.089
1991	0.000	0.145	0.281	0.182	0.146	0.074	0.062	0.110
1992	0.000	0.093	0.278	0.166	0.182	0.120	0.066	0.096
1993	0.000	0.128	0.245	0.193	0.181	0.108	0.081	0.064
1994	0.000	0.152	0.236	0.134	0.169	0.159	0.087	0.064
1995	0.003	0.198	0.128	0.069	0.073	0.164	0.198	0.168
1996	0.001	0.267	0.159	0.081	0.103	0.202	0.134	0.052
1997	0.000	0.080	0.551	0.094	0.067	0.084	0.093	0.031
1998	0.000	0.162	0.178	0.425	0.100	0.048	0.050	0.037
1999	0.001	0.148	0.346	0.117	0.228	0.093	0.040	0.028
2000	0.000	0.272	0.070	0.153	0.189	0.231	0.058	0.027
2001	0.000	0.078	0.422	0.065	0.111	0.142	0.141	0.040
2002	0.009	0.091	0.169	0.341	0.131	0.099	0.101	0.059
2003	0.002	0.287	0.193	0.083	0.228	0.079	0.075	0.054
2004	0.001	0.199	0.463	0.112	0.080	0.093	0.041	0.011
2005	0.000	0.064	0.443	0.276	0.080	0.074	0.052	0.011
2006	0.000	0.076	0.292	0.384	0.149	0.044	0.031	0.024
2007	0.000	0.241	0.216	0.201	0.196	0.101	0.029	0.017
2008	0.000	0.020	0.434	0.140	0.121	0.153	0.082	0.049
2009	0.000	0.107	0.135	0.413	0.101	0.096	0.104	0.045
2010	0.000	0.420	0.218	0.089	0.177	0.043	0.034	0.019
2011	0.000	0.049	0.803	0.104	0.022	0.017	0.003	0.003
2012	0.002	0.127	0.049	0.652	0.111	0.024	0.027	0.010
2013	0.000	0.156	0.154	0.085	0.499	0.089	0.012	0.005
2014	0.000	0.073	0.515	0.100	0.049	0.221	0.038	0.004
2015	0.000	0.133	0.100	0.488	0.065	0.063	0.134	0.018
2016	0.000	0.015	0.194	0.158	0.368	0.096	0.062	0.107
2017	0.000	0.014	0.184	0.328	0.118	0.267	0.033	0.057

Table B1- 8 Fixed fleet age composition proportions at age

	1	2	3	4	5	6	7	8+
1965	0.027	0.865	0.066	0.025	0.004	0.000	0.004	0.009
1966	0.032	0.368	0.523	0.042	0.025	0.001	0.003	0.006
1967	0.159	0.487	0.162	0.153	0.022	0.008	0.002	0.008
1968	0.069	0.801	0.085	0.017	0.022	0.002	0.002	0.001
1969	0.120	0.619	0.219	0.009	0.013	0.010	0.008	0.003
1970	0.057	0.848	0.036	0.030	0.013	0.008	0.005	0.002
1971	0.320	0.473	0.123	0.029	0.017	0.014	0.012	0.012
1972	0.008	0.930	0.012	0.013	0.015	0.010	0.008	0.004
1973	0.100	0.460	0.387	0.044	0.005	0.002	0.001	0.001
1974	0.056	0.741	0.126	0.073	0.004	0.000	0.000	0.000
1975	0.055	0.791	0.104	0.017	0.027	0.003	0.001	0.002
1976	0.083	0.635	0.227	0.023	0.017	0.013	0.002	0.001
1977	0.436	0.452	0.060	0.028	0.008	0.008	0.008	0.000
1978	0.154	0.780	0.059	0.003	0.002	0.001	0.000	0.001
1979	0.004	0.764	0.219	0.010	0.001	0.001	0.001	0.001
1980	0.349	0.293	0.290	0.064	0.003	0.000	0.000	0.001
1981	0.042	0.903	0.026	0.016	0.012	0.001	0.000	0.000
1982	0.071	0.809	0.111	0.004	0.003	0.002	0.000	0.000
1983	0.126	0.769	0.077	0.025	0.001	0.001	0.001	0.000
1984	0.152	0.654	0.119	0.039	0.030	0.004	0.002	0.001
1985	0.060	0.823	0.072	0.027	0.011	0.005	0.000	0.000
1986	0.074	0.438	0.364	0.072	0.030	0.013	0.006	0.001
1987	0.187	0.454	0.131	0.140	0.060	0.019	0.007	0.002
1988	0.119	0.688	0.071	0.035	0.061	0.022	0.003	0.001
1989	0.044	0.645	0.141	0.036	0.037	0.071	0.019	0.007
1990	0.020	0.762	0.113	0.049	0.010	0.012	0.024	0.011
1991	0.011	0.806	0.094	0.047	0.019	0.006	0.005	0.011
1992	0.001	0.749	0.164	0.057	0.016	0.006	0.003	0.003
1993	0.003	0.616	0.221	0.073	0.046	0.024	0.010	0.007
1994	0.005	0.741	0.153	0.024	0.039	0.022	0.009	0.008
1995	0.153	0.688	0.106	0.039	0.005	0.004	0.004	0.000
1996	0.018	0.859	0.070	0.029	0.013	0.004	0.004	0.002
1997	0.026	0.610	0.319	0.032	0.010	0.001	0.001	0.000
1998	0.001	0.843	0.082	0.048	0.012	0.007	0.005	0.002
1999	0.001	0.464	0.418	0.052	0.038	0.018	0.006	0.004
2000	0.006	0.911	0.029	0.028	0.019	0.005	0.002	0.000
2001	0.005	0.453	0.477	0.032	0.023	0.007	0.002	0.001
2002	0.131	0.740	0.067	0.047	0.012	0.002	0.000	0.000
2003	0.059	0.833	0.080	0.015	0.010	0.002	0.001	0.000
2004	0.021	0.578	0.326	0.059	0.008	0.005	0.001	0.002
2005	0.006	0.507	0.366	0.096	0.021	0.003	0.001	0.001
2006	0.521	0.363	0.086	0.015	0.010	0.003	0.002	0.000
2007	0.010	0.925	0.056	0.005	0.001	0.002	0.001	0.000
2008	0.164	0.717	0.116	0.001	0.000	0.001	0.001	0.000
2009	0.112	0.858	0.028	0.001	0.000	0.000	0.000	0.000
2010	0.000	0.954	0.044	0.001	0.001	0.000	0.000	0.000
2011	0.173	0.542	0.255	0.026	0.003	0.001	0.000	0.000
2012	0.804	0.148	0.016	0.021	0.008	0.002	0.001	0.000
2013	0.204	0.728	0.060	0.004	0.002	0.001	0.000	0.000
2014	0.297	0.627	0.055	0.011	0.004	0.004	0.001	0.001
2015	0.037	0.514	0.007	0.184	0.038	0.063	0.141	0.015
2016	0.011	0.827	0.122	0.023	0.009	0.003	0.002	0.003
2017	0.088	0.493	0.086	0.185	0.068	0.045	0.021	0.014

Table B1- 9 Catch weights at age (kg)

	1	2	3	4	5	6	7	8+
1965	0.009	0.024	0.055	0.112	0.134	0.272	0.189	0.189
1966	0.011	0.027	0.068	0.142	0.219	0.272	0.189	0.189
1967	0.009	0.028	0.062	0.114	0.170	0.210	0.238	0.351
1968	0.058	0.034	0.068	0.143	0.186	0.239	0.276	0.276
1969	0.010	0.035	0.100	0.137	0.210	0.240	0.288	0.288
1970	0.010	0.044	0.121	0.159	0.186	0.232	0.269	0.413
1971	0.012	0.044	0.129	0.168	0.199	0.242	0.289	0.346
1972	0.026	0.039	0.113	0.175	0.212	0.260	0.292	0.361
1973	0.010	0.044	0.110	0.137	0.219	0.280	0.331	0.370
1974	0.010	0.038	0.103	0.167	0.203	0.271	0.293	0.293
1975	0.016	0.044	0.107	0.177	0.206	0.244	0.288	0.375
1976	0.014	0.036	0.106	0.174	0.205	0.229	0.263	0.333
1977	0.012	0.037	0.094	0.153	0.196	0.227	0.236	0.305
1978	0.011	0.037	0.096	0.158	0.196	0.220	0.239	0.318
1979	0.006	0.031	0.082	0.169	0.216	0.243	0.265	0.294
1980	0.012	0.041	0.097	0.150	0.229	0.265	0.291	0.332
1981	0.010	0.041	0.098	0.177	0.213	0.281	0.310	0.356
1982	0.019	0.042	0.104	0.204	0.229	0.253	0.305	0.367
1983	0.018	0.041	0.124	0.199	0.219	0.283	0.319	0.410
1984	0.014	0.041	0.117	0.154	0.195	0.209	0.291	0.305
1985	0.017	0.036	0.096	0.148	0.162	0.188	0.198	0.220
1986	0.018	0.042	0.101	0.159	0.210	0.236	0.247	0.266
1987	0.011	0.041	0.092	0.137	0.088	0.147	0.145	0.160
1988	0.007	0.031	0.091	0.106	0.123	0.132	0.190	0.208
1989	0.009	0.031	0.066	0.104	0.116	0.133	0.157	0.157
1990	0.004	0.029	0.080	0.138	0.172	0.169	0.179	0.235
1991	0.004	0.036	0.074	0.124	0.150	0.184	0.200	0.244
1992	0.009	0.035	0.073	0.124	0.139	0.164	0.191	0.249
1993	0.003	0.032	0.078	0.119	0.125	0.148	0.183	0.265
1994	0.008	0.029	0.070	0.118	0.134	0.152	0.162	0.166
1995	0.014	0.046	0.090	0.118	0.134	0.149	0.160	0.259
1996	0.024	0.043	0.083	0.120	0.146	0.164	0.179	0.280
1997	0.017	0.045	0.085	0.118	0.146	0.167	0.182	0.182
1998	0.021	0.037	0.080	0.112	0.133	0.158	0.178	0.222
1999	0.026	0.048	0.087	0.116	0.132	0.149	0.176	0.216
2000	0.018	0.060	0.101	0.127	0.147	0.159	0.182	0.244
2001	0.005	0.047	0.089	0.127	0.147	0.161	0.175	0.240
2002	0.020	0.045	0.093	0.121	0.138	0.158	0.169	0.200
2003	0.015	0.052	0.090	0.130	0.149	0.166	0.184	0.207
2004	0.011	0.043	0.092	0.125	0.152	0.166	0.186	0.209
2005	0.019	0.042	0.083	0.123	0.149	0.170	0.188	0.252
2006	0.019	0.066	0.085	0.120	0.147	0.172	0.188	0.198
2007	0.016	0.047	0.085	0.118	0.141	0.161	0.185	0.199
2008	0.016	0.041	0.100	0.131	0.152	0.169	0.180	0.221
2009	0.004	0.047	0.090	0.133	0.156	0.172	0.184	0.206
2010	0.028	0.036	0.072	0.113	0.142	0.162	0.174	0.174
2011	0.019	0.044	0.069	0.100	0.138	0.160	0.189	0.183
2012	0.013	0.049	0.085	0.096	0.109	0.145	0.160	0.184
2013	0.012	0.050	0.070	0.107	0.118	0.129	0.155	0.204
2014	0.012	0.060	0.096	0.106	0.144	0.146	0.150	0.165
2015	0.025	0.043	0.087	0.126	0.136	0.158	0.158	0.206
2016	0.025	0.047	0.068	0.107	0.143	0.151	0.166	0.183
2017	0.014	0.044	0.085	0.114	0.140	0.158	0.167	0.167

Table B1- 10 Spawning stock biomass weights at age (kg)

	1	2	3	4	5	6	7	8+
1965	0.013	0.038	0.095	0.113	0.202	0.265	0.298	0.355
1966	0.016	0.047	0.096	0.170	0.224	0.279	0.302	0.355
1967	0.016	0.043	0.107	0.172	0.206	0.227	0.242	0.371
1968	0.011	0.038	0.069	0.178	0.223	0.265	0.298	0.355
1969	0.011	0.041	0.102	0.134	0.222	0.265	0.298	0.311
1970	0.011	0.061	0.126	0.163	0.191	0.239	0.276	0.419
1971	0.014	0.068	0.144	0.170	0.202	0.248	0.296	0.353
1972	0.031	0.069	0.154	0.197	0.235	0.268	0.289	0.344
1973	0.011	0.051	0.133	0.170	0.238	0.295	0.352	0.379
1974	0.008	0.045	0.124	0.169	0.196	0.270	0.290	0.352
1975	0.015	0.055	0.133	0.188	0.211	0.248	0.295	0.362
1976	0.015	0.088	0.132	0.184	0.210	0.236	0.278	0.371
1977	0.013	0.045	0.131	0.175	0.215	0.243	0.249	0.342
1978	0.032	0.051	0.119	0.178	0.208	0.239	0.252	0.321
1979	0.015	0.073	0.133	0.187	0.229	0.253	0.302	0.389
1980	0.007	0.054	0.104	0.185	0.250	0.294	0.319	0.366
1981	0.015	0.039	0.135	0.192	0.236	0.301	0.339	0.379
1982	0.017	0.050	0.139	0.200	0.240	0.272	0.328	0.368
1983	0.024	0.069	0.144	0.214	0.265	0.297	0.332	0.413
1984	0.007	0.064	0.140	0.193	0.239	0.286	0.313	0.379
1985	0.006	0.047	0.146	0.208	0.237	0.268	0.318	0.269
1986	0.032	0.057	0.116	0.176	0.227	0.252	0.271	0.319
1987	0.010	0.068	0.108	0.159	0.202	0.238	0.256	0.315
1988	0.027	0.066	0.117	0.154	0.192	0.229	0.264	0.316
1989	0.027	0.068	0.116	0.172	0.201	0.234	0.260	0.329
1990	0.024	0.062	0.106	0.156	0.189	0.216	0.233	0.312
1991	0.024	0.063	0.096	0.142	0.171	0.205	0.225	0.306
1992	0.024	0.060	0.102	0.135	0.164	0.190	0.220	0.305
1993	0.024	0.047	0.096	0.137	0.156	0.180	0.209	0.309
1994	0.024	0.054	0.086	0.120	0.138	0.159	0.180	0.307
1995	0.027	0.051	0.095	0.123	0.145	0.162	0.175	0.275
1996	0.028	0.055	0.088	0.125	0.150	0.171	0.188	0.228
1997	0.014	0.059	0.091	0.124	0.150	0.174	0.194	0.222
1998	0.027	0.052	0.092	0.117	0.138	0.164	0.187	0.216
1999	0.026	0.060	0.091	0.123	0.140	0.157	0.186	0.205
2000	0.027	0.065	0.111	0.137	0.156	0.172	0.198	0.221
2001	0.033	0.056	0.099	0.134	0.153	0.166	0.181	0.201
2002	0.030	0.059	0.099	0.126	0.143	0.167	0.183	0.195
2003	0.027	0.059	0.099	0.137	0.153	0.171	0.192	0.198
2004	0.027	0.047	0.091	0.129	0.155	0.173	0.194	0.203
2005	0.027	0.054	0.087	0.131	0.159	0.183	0.199	0.198
2006	0.027	0.062	0.089	0.133	0.163	0.184	0.203	0.204
2007	0.027	0.064	0.106	0.140	0.164	0.184	0.203	0.207
2008	0.027	0.068	0.106	0.135	0.162	0.175	0.188	0.201
2009	0.027	0.057	0.095	0.138	0.159	0.179	0.191	0.209
2010	0.027	0.043	0.089	0.121	0.146	0.169	0.183	0.203
2011	0.027	0.049	0.076	0.110	0.141	0.168	0.183	0.198
2012	0.032	0.049	0.090	0.107	0.123	0.155	0.188	0.198
2013	0.027	0.061	0.090	0.124	0.132	0.144	0.180	0.199
2014	0.027	0.066	0.106	0.119	0.155	0.158	0.165	0.196
2015	0.027	0.057	0.103	0.136	0.148	0.169	0.170	0.195
2016	0.027	0.065	0.080	0.114	0.151	0.158	0.171	0.190
2017	0.027	0.058	0.093	0.121	0.148	0.169	0.186	0.185

Table B1- 11 Jan. 1 Weights at age (kg)

	1	2	3	4	5	6	7	8+
1965	0.007	0.024	0.071	0.080	0.172	0.248	0.287	0.356
1966	0.010	0.025	0.060	0.127	0.159	0.237	0.283	0.352
1967	0.010	0.026	0.071	0.129	0.187	0.226	0.260	0.360
1968	0.006	0.025	0.055	0.138	0.196	0.234	0.260	0.354
1969	0.005	0.021	0.062	0.096	0.199	0.243	0.281	0.326
1970	0.004	0.026	0.072	0.129	0.160	0.230	0.270	0.364
1971	0.006	0.027	0.094	0.146	0.182	0.218	0.266	0.360
1972	0.024	0.031	0.102	0.168	0.200	0.233	0.268	0.345
1973	0.005	0.040	0.096	0.162	0.217	0.263	0.307	0.356
1974	0.003	0.022	0.080	0.150	0.183	0.254	0.293	0.366
1975	0.006	0.021	0.077	0.153	0.189	0.221	0.282	0.353
1976	0.009	0.036	0.085	0.156	0.199	0.223	0.263	0.363
1977	0.007	0.026	0.107	0.152	0.199	0.226	0.242	0.351
1978	0.021	0.026	0.073	0.153	0.191	0.227	0.248	0.324
1979	0.008	0.048	0.082	0.149	0.202	0.229	0.269	0.341
1980	0.003	0.029	0.087	0.157	0.216	0.260	0.284	0.365
1981	0.008	0.017	0.085	0.141	0.209	0.274	0.316	0.370
1982	0.008	0.027	0.074	0.164	0.215	0.253	0.314	0.373
1983	0.015	0.034	0.085	0.173	0.230	0.267	0.301	0.389
1984	0.003	0.039	0.098	0.167	0.226	0.275	0.305	0.395
1985	0.002	0.018	0.097	0.171	0.214	0.253	0.302	0.268
1986	0.022	0.019	0.074	0.160	0.217	0.244	0.270	0.252
1987	0.004	0.047	0.079	0.136	0.189	0.232	0.254	0.312
1988	0.017	0.026	0.089	0.129	0.175	0.215	0.251	0.309
1989	0.018	0.043	0.088	0.142	0.176	0.212	0.244	0.317
1990	0.015	0.041	0.085	0.135	0.180	0.208	0.234	0.314
1991	0.015	0.039	0.077	0.123	0.163	0.197	0.221	0.301
1992	0.017	0.038	0.080	0.114	0.153	0.180	0.212	0.298
1993	0.016	0.034	0.076	0.118	0.145	0.172	0.199	0.298
1994	0.017	0.036	0.064	0.107	0.138	0.158	0.180	0.298
1995	0.019	0.035	0.072	0.103	0.132	0.150	0.167	0.280
1996	0.019	0.039	0.067	0.109	0.136	0.158	0.175	0.237
1997	0.007	0.041	0.071	0.105	0.137	0.162	0.182	0.193
1998	0.018	0.027	0.074	0.103	0.131	0.157	0.180	0.148
1999	0.016	0.040	0.069	0.106	0.128	0.147	0.175	0.211
2000	0.019	0.041	0.082	0.112	0.139	0.155	0.176	0.211
2001	0.025	0.039	0.080	0.122	0.145	0.161	0.176	0.210
2002	0.021	0.044	0.075	0.112	0.138	0.160	0.174	0.198
2003	0.021	0.042	0.076	0.117	0.139	0.156	0.179	0.197
2004	0.019	0.036	0.073	0.113	0.146	0.163	0.182	0.201
2005	0.018	0.038	0.064	0.109	0.143	0.168	0.186	0.202
2006	0.018	0.041	0.069	0.108	0.146	0.171	0.193	0.203
2007	0.017	0.042	0.081	0.112	0.148	0.173	0.193	0.207
2008	0.019	0.043	0.082	0.120	0.151	0.169	0.186	0.205
2009	0.021	0.039	0.080	0.121	0.147	0.170	0.183	0.205
2010	0.020	0.034	0.071	0.107	0.142	0.164	0.181	0.202
2011	0.020	0.036	0.057	0.099	0.131	0.157	0.176	0.202
2012	0.023	0.036	0.066	0.090	0.116	0.148	0.178	0.198
2013	0.017	0.044	0.066	0.106	0.119	0.133	0.167	0.199
2014	0.019	0.042	0.080	0.104	0.139	0.144	0.154	0.198
2015	0.017	0.039	0.082	0.120	0.133	0.162	0.164	0.195
2016	0.018	0.042	0.068	0.108	0.143	0.153	0.170	0.192
2017	0.018	0.040	0.078	0.098	0.130	0.160	0.171	0.187

Table B1- 12 Proportion mature at age

	1	2	3	4	5	6	7	8+
1965	0.0000	0.0529	0.2143	0.8000	1.0000	1.0000	1.0000	1.0000
1966	0.0000	0.0264	0.3082	0.8304	0.9979	0.9993	1.0000	1.0000
1967	0.0000	0.0264	0.3082	0.8304	0.9979	0.9993	1.0000	1.0000
1968	0.0000	0.0264	0.3082	0.8304	0.9979	0.9993	1.0000	1.0000
1969	0.0000	0.0264	0.3082	0.8304	0.9979	0.9993	1.0000	1.0000
1970	0.0000	0.0264	0.3082	0.8304	0.9979	0.9993	1.0000	1.0000
1971	0.0000	0.0000	0.4021	0.8608	0.9959	0.9986	1.0000	1.0000
1972	0.0000	0.0264	0.6241	0.9304	0.9979	0.9993	1.0000	1.0000
1973	0.0000	0.0529	0.8462	1.0000	1.0000	1.0000	1.0000	1.0000
1974	0.0000	0.0264	0.5514	0.9828	1.0000	1.0000	1.0000	1.0000
1975	0.0000	0.0264	0.5514	0.9828	1.0000	1.0000	1.0000	1.0000
1976	0.0000	0.0264	0.5514	0.9828	1.0000	1.0000	1.0000	1.0000
1977	0.0000	0.0000	0.2566	0.9655	1.0000	1.0000	1.0000	1.0000
1978	0.0000	0.0000	0.2722	0.9782	1.0000	0.9762	1.0000	1.0000
1979	0.0000	0.0000	0.4303	0.9944	1.0000	1.0000	1.0000	1.0000
1980	0.0000	0.0529	0.1641	0.9680	1.0000	1.0000	1.0000	1.0000
1981	0.0000	0.0000	0.1485	0.9711	0.9972	1.0000	1.0000	1.0000
1982	0.0000	0.0000	0.6276	1.0000	1.0000	1.0000	1.0000	1.0000
1983	0.0000	0.0000	0.5831	0.9938	1.0000	1.0000	1.0000	1.0000
1984	0.0000	0.0000	0.6102	1.0000	1.0000	1.0000	1.0000	1.0000
1985	0.0000	0.0833	0.7166	0.9947	1.0000	1.0000	1.0000	1.0000
1986	0.0000	0.0000	0.5039	0.9744	1.0000	1.0000	1.0000	1.0000
1987	0.0000	0.2000	0.2986	0.9517	1.0000	1.0000	1.0000	1.0000
1988	0.0000	0.0000	0.2966	0.9769	1.0000	1.0000	1.0000	1.0000
1989	0.0000	0.0000	0.4046	0.9837	1.0000	1.0000	0.9762	1.0000
1990	0.0000	0.0000	0.2378	0.9646	1.0000	1.0000	1.0000	1.0000
1991	0.0000	0.0000	0.2297	0.9701	1.0000	1.0000	1.0000	1.0000
1992	0.0000	0.0529	0.3982	0.9632	1.0000	1.0000	1.0000	1.0000
1993	0.0000	0.0529	0.3186	0.9845	0.9954	1.0000	1.0000	1.0000
1994	0.0000	0.0529	0.1646	0.9082	1.0000	1.0000	1.0000	1.0000
1995	0.0000	0.0529	0.3370	0.8939	1.0000	1.0000	1.0000	1.0000
1996	0.0000	0.0529	0.4500	0.9467	1.0000	1.0000	1.0000	1.0000
1997	0.0000	0.6667	0.8523	1.0000	1.0000	1.0000	1.0000	0.9756
1998	0.0000	0.0529	0.6117	0.9891	1.0000	0.9804	1.0000	1.0000
1999	0.0000	0.0000	0.3548	0.9184	0.9926	1.0000	0.9677	1.0000
2000	0.0000	0.0000	0.6535	0.9919	1.0000	1.0000	1.0000	0.9412
2001	0.0000	0.0000	0.8438	1.0000	1.0000	0.9919	0.9913	1.0000
2002	0.0000	0.0000	0.5252	0.9802	1.0000	1.0000	1.0000	1.0000
2003	0.0000	0.0400	0.5924	0.9552	1.0000	1.0000	1.0000	0.9500
2004	0.0000	0.3333	0.6257	1.0000	1.0000	1.0000	1.0000	1.0000
2005	0.0000	0.5000	0.5662	1.0000	1.0000	1.0000	1.0000	1.0000
2006	0.0000	0.0000	0.3370	0.9927	1.0000	1.0000	1.0000	1.0000
2007	0.0000	0.0063	0.7798	0.9921	1.0000	1.0000	1.0000	1.0000
2008	0.0000	0.0000	0.7890	0.9899	1.0000	1.0000	1.0000	1.0000
2009	0.0000	0.0000	0.7317	1.0000	1.0000	1.0000	1.0000	1.0000
2010	0.0000	0.0087	0.7324	0.9917	1.0000	0.9800	1.0000	1.0000
2011	0.0000	0.0000	0.4842	0.9830	1.0000	1.0000	1.0000	1.0000
2012	0.0000	0.0000	0.6230	0.9906	1.0000	1.0000	1.0000	1.0000
2013	0.0000	0.0660	0.5556	0.9242	0.9973	1.0000	1.0000	1.0000
2014	0.0000	0.0000	0.8817	1.0000	1.0000	1.0000	1.0000	1.0000
2015	0.0000	0.0000	0.6543	0.9965	1.0000	1.0000	1.0000	1.0000
2016	0.0000	0.0000	0.5306	0.7778	1.0000	1.0000	1.0000	1.0000
2017	0.0000	0.0000	0.7765	0.9110	1.0000	1.0000	1.0000	1.0000

Table B1- 13 Number of samples used for maturity at age each year

Year	Maturity Samples
1965	21
1966	0
1967	0
1968	0
1969	0
1970	0
1971	3692
1972	0
1973	84
1974	0
1975	0
1976	0
1977	366
1978	1504
1979	1307
1980	1604
1981	1072
1982	751
1983	993
1984	1107
1985	1037
1986	440
1987	710
1988	468
1989	581
1990	486
1991	674
1992	842
1993	1033
1994	502
1995	804
1996	567
1997	1166
1998	583
1999	640
2000	672
2001	902
2002	998
2003	594
2004	289
2005	959
2006	985
2007	716
2008	744
2009	804
2010	923
2011	1093
2012	851
2013	775
2014	915
2015	602
2016	352
2017	449

Table B1- 14 – Summary of spatial and seasonal restrictions that are in place in the Atlantic herring fishery (both NEFMC and ASMFC actions) (Source: Manderson and Sarro (in prep.) Fishing industry perspectives on socio-ecological factors driving Atlantic Mackerel landings in US waters).

x = represents no herring fishing

y = represents no midwater trawl gear permitted

z = possible spawning closures, restricts all herring fishing, all gear types

		Sub-Area			
		1A	**1B**	**2**	**3**
Month	**1**	**x**	**x**		
	2	**x**	**x**		
	3	**x**	**x**		
	4	**x**	**x**		
	5	**x**			
	6	**y**			
	7	**y**			
	8	**y, z**			
	9	**y, z**			
	10	**z**			
	11				
	12				

Figure B1- 1 Length (cm) composition of Atlantic herring caught by purse seine, midwater trawl, or paired midwater trawl during 2007-2016

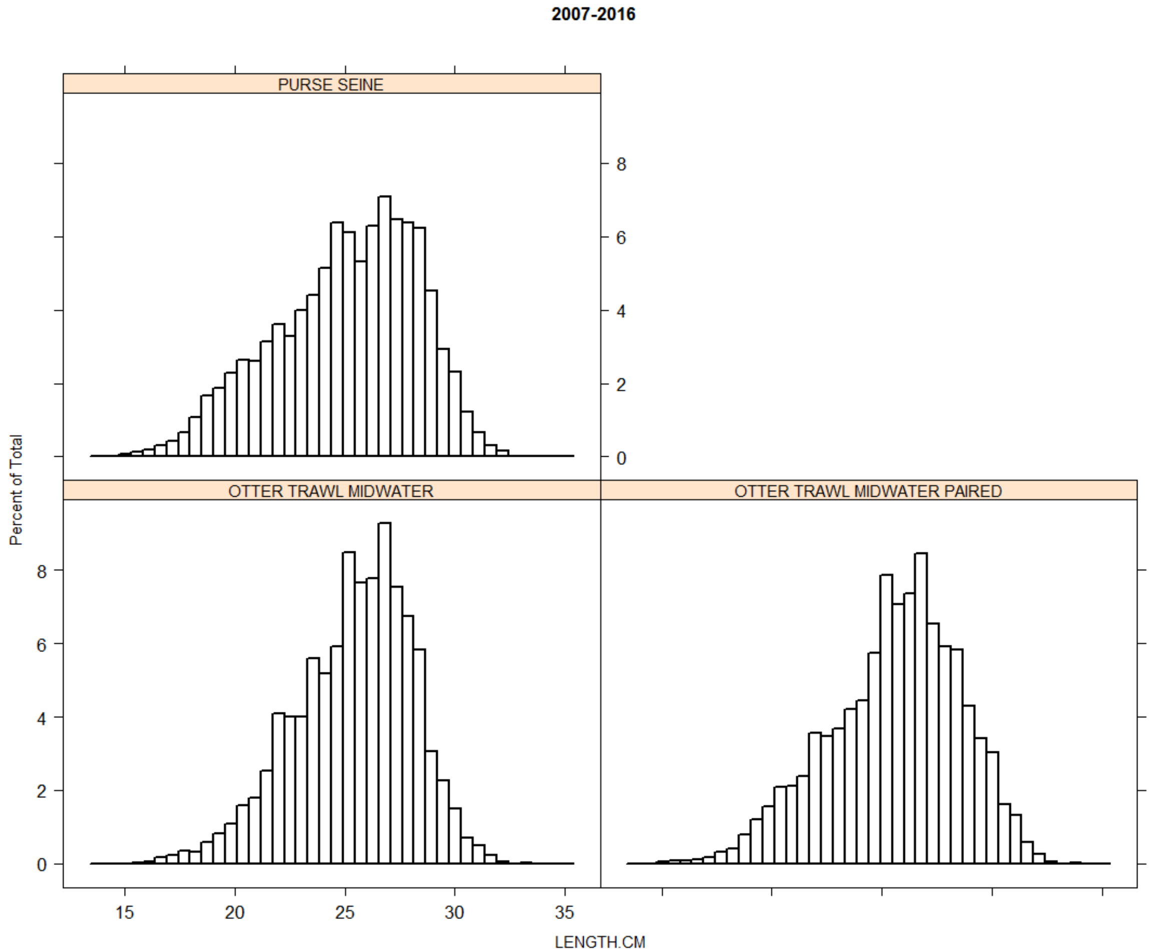


Figure B1- 2 Example of a Atlantic herring bimodal length frequency (cm) observed for the purse seine gear but not midwater trawls (data from 2012)

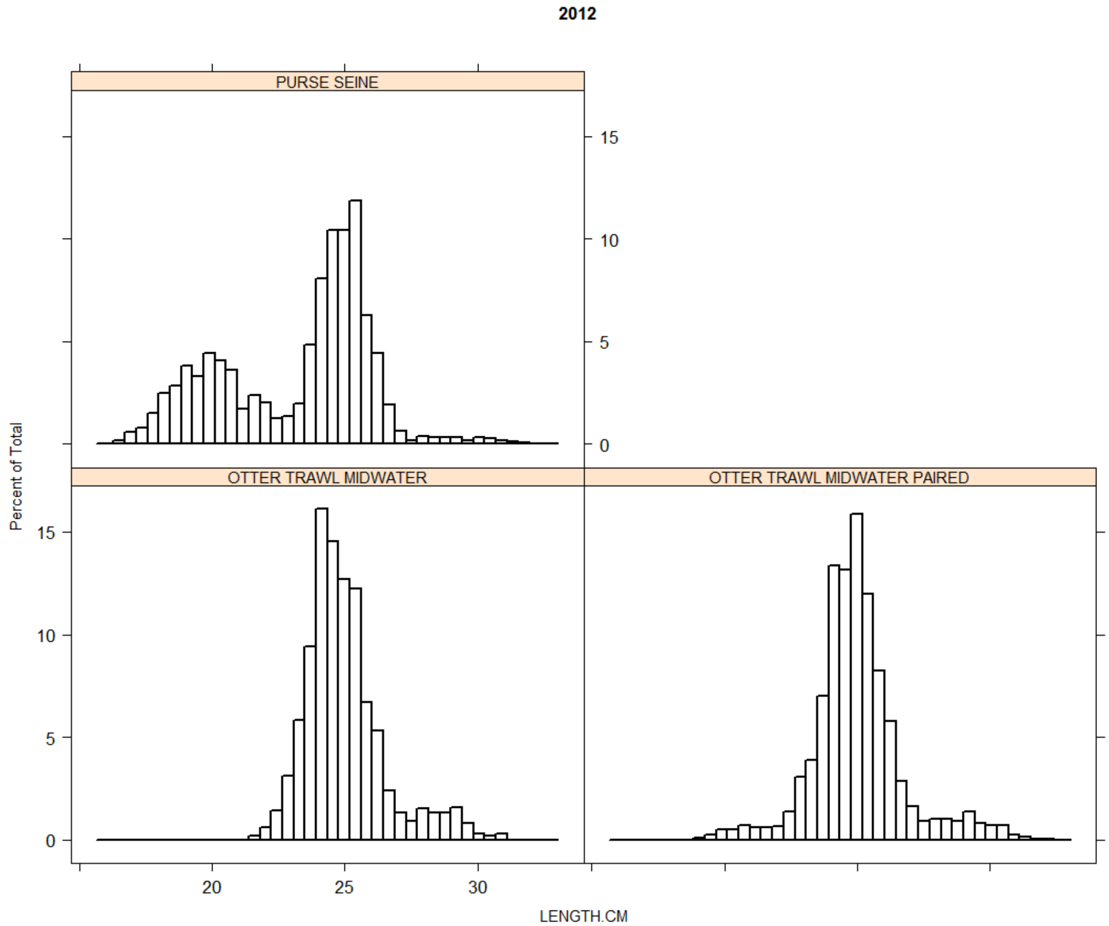


Figure B1- 3 Atlantic herring catch (mt) by purse seine, midwater trawl, and paired midwater trawl

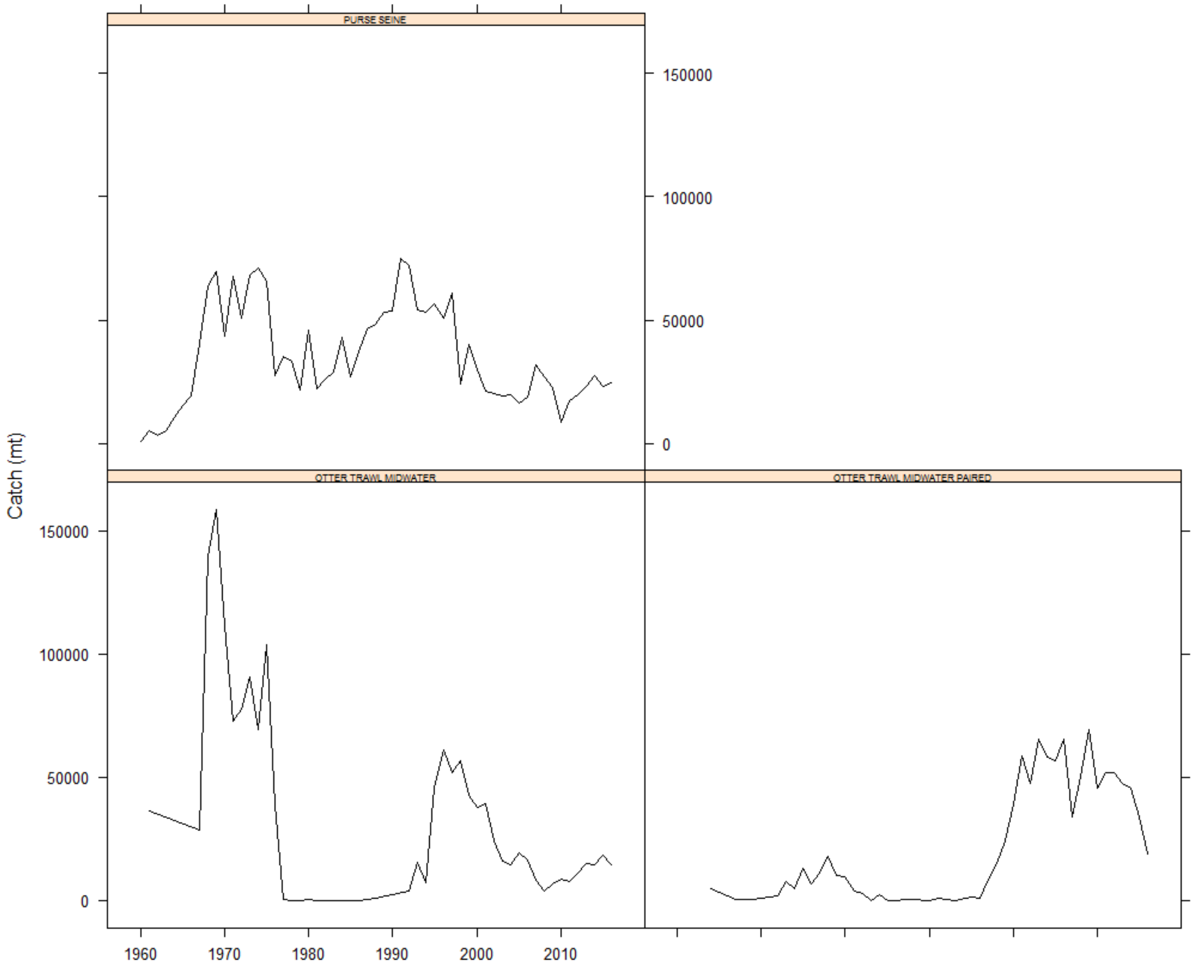


Figure B1- 4 Atlantic herring catch (mt) by mobile and fixed fleets in the Gulf of Maine (GOFM) and outside the GOFM (OTHER)

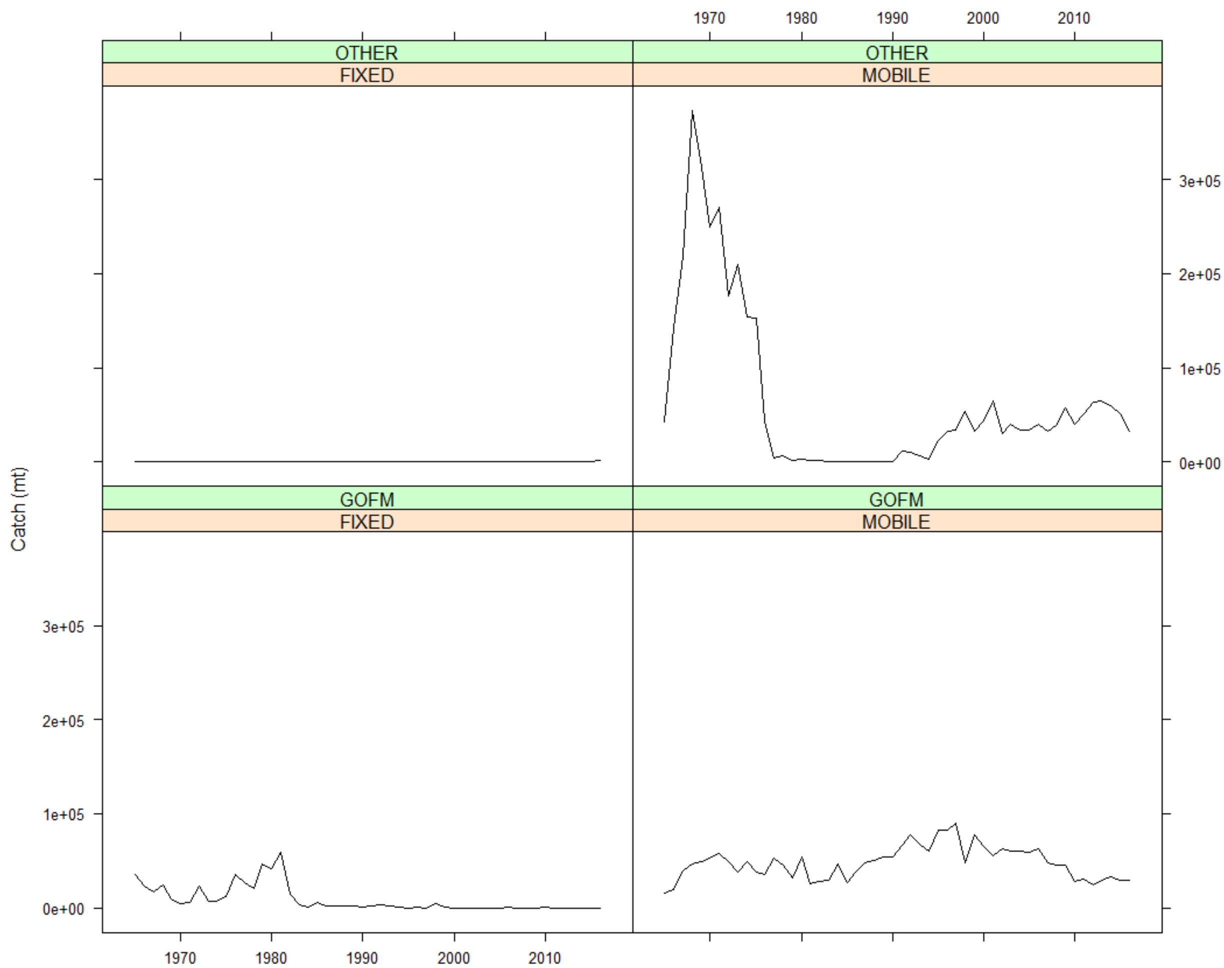


Figure B1- 5 Atlantic herring length composition (cm) of the mobile fleet during 1964-2011 in the Gulf of Maine (GOFM) and all other areas (OTHER)

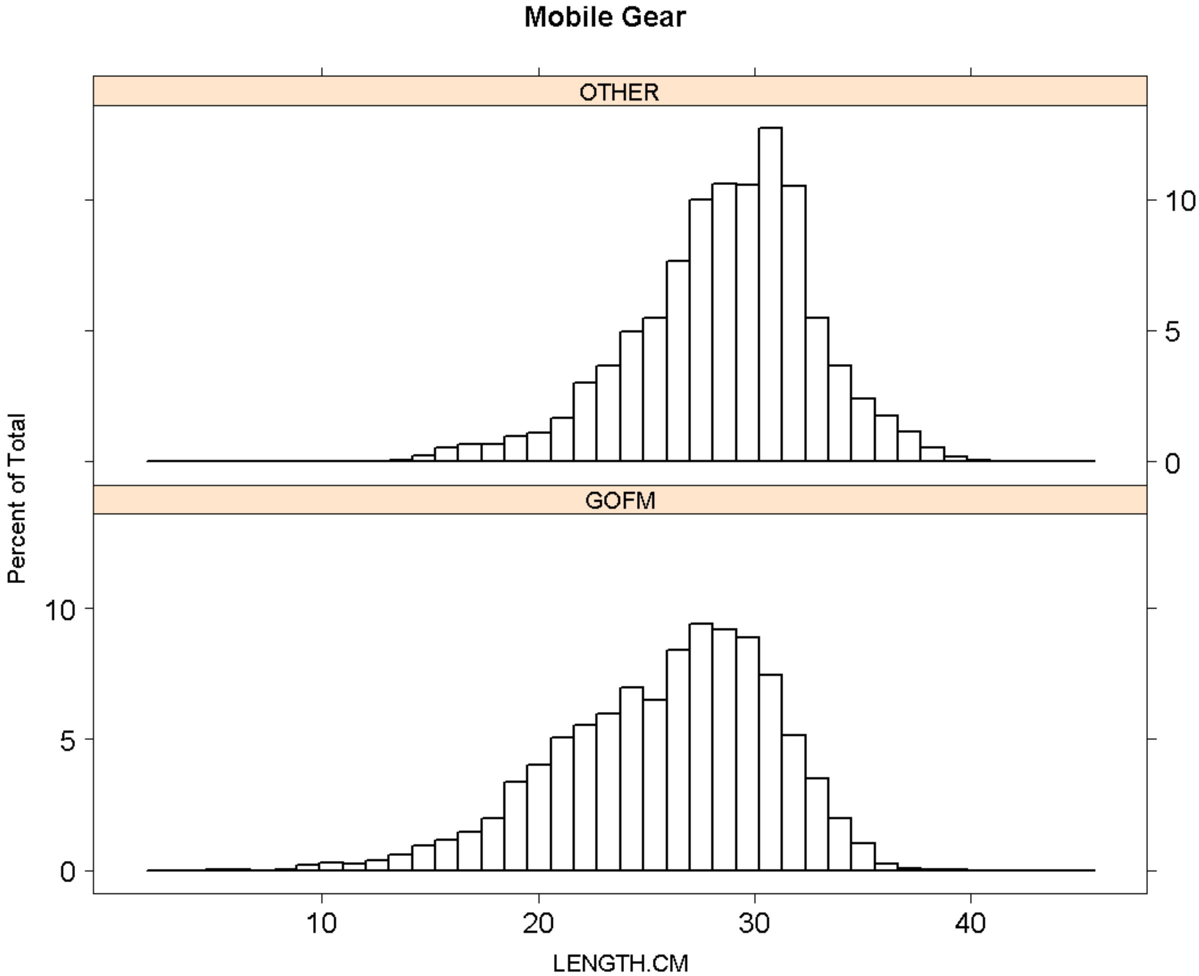


Figure B1- 6 Atlantic herring length composition of the mobile and fixed fleets during 1965-2011 in the Gulf of Maine (GOFM)

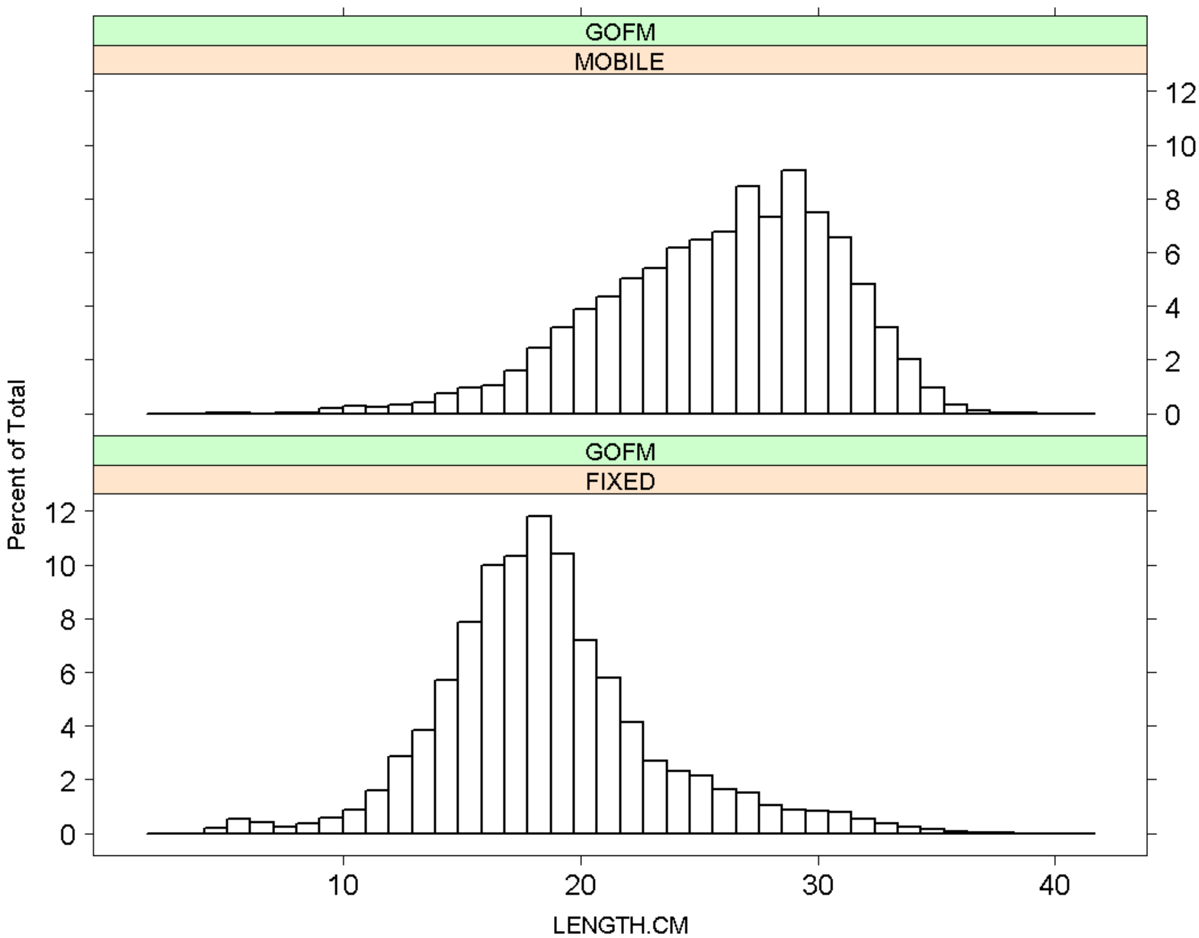


Figure B1- 7 Atlantic herring catch (mt) by the mobile and fixed fleets

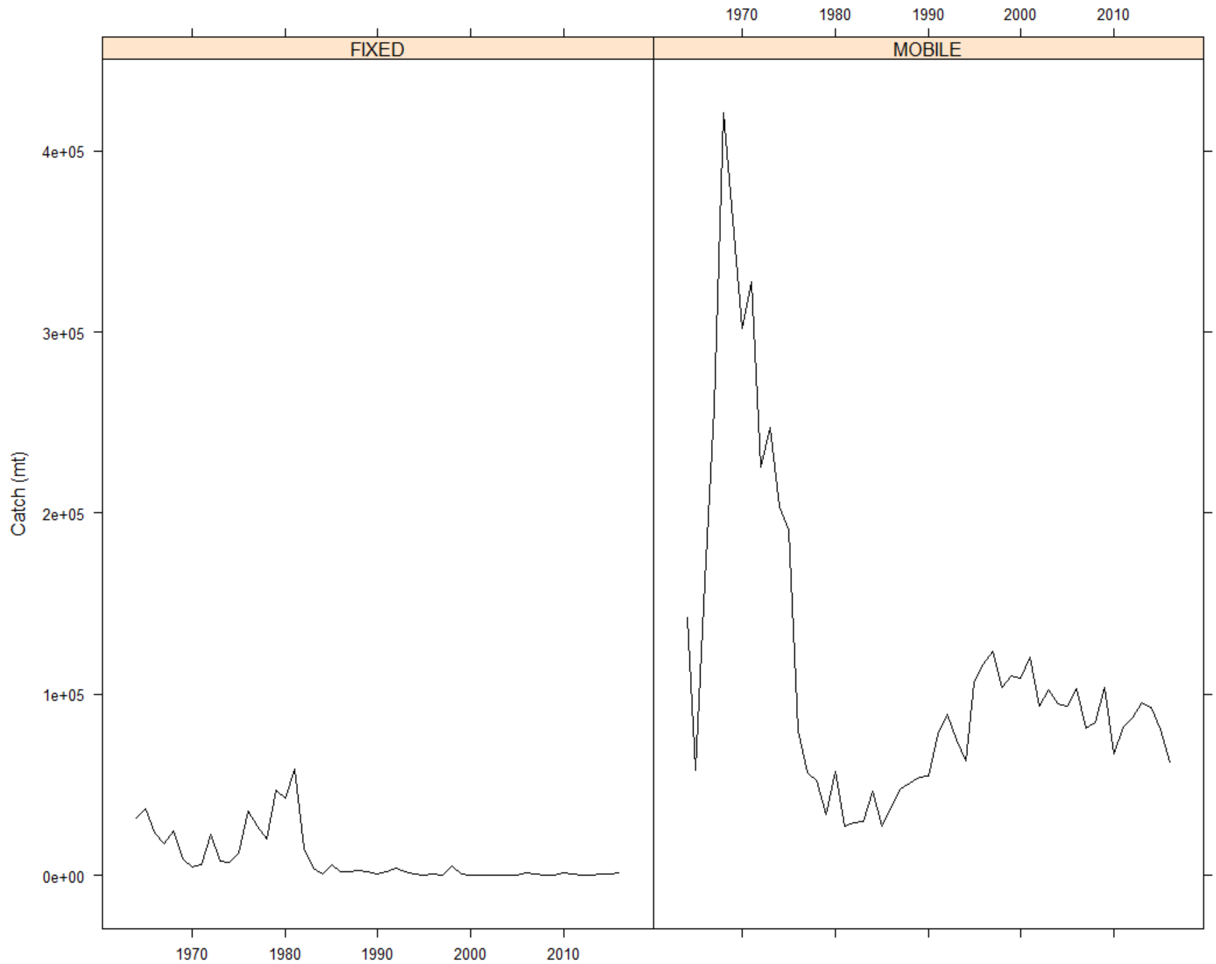


Figure B1- 8 Atlantic herring proportions at age for the mobile fleet

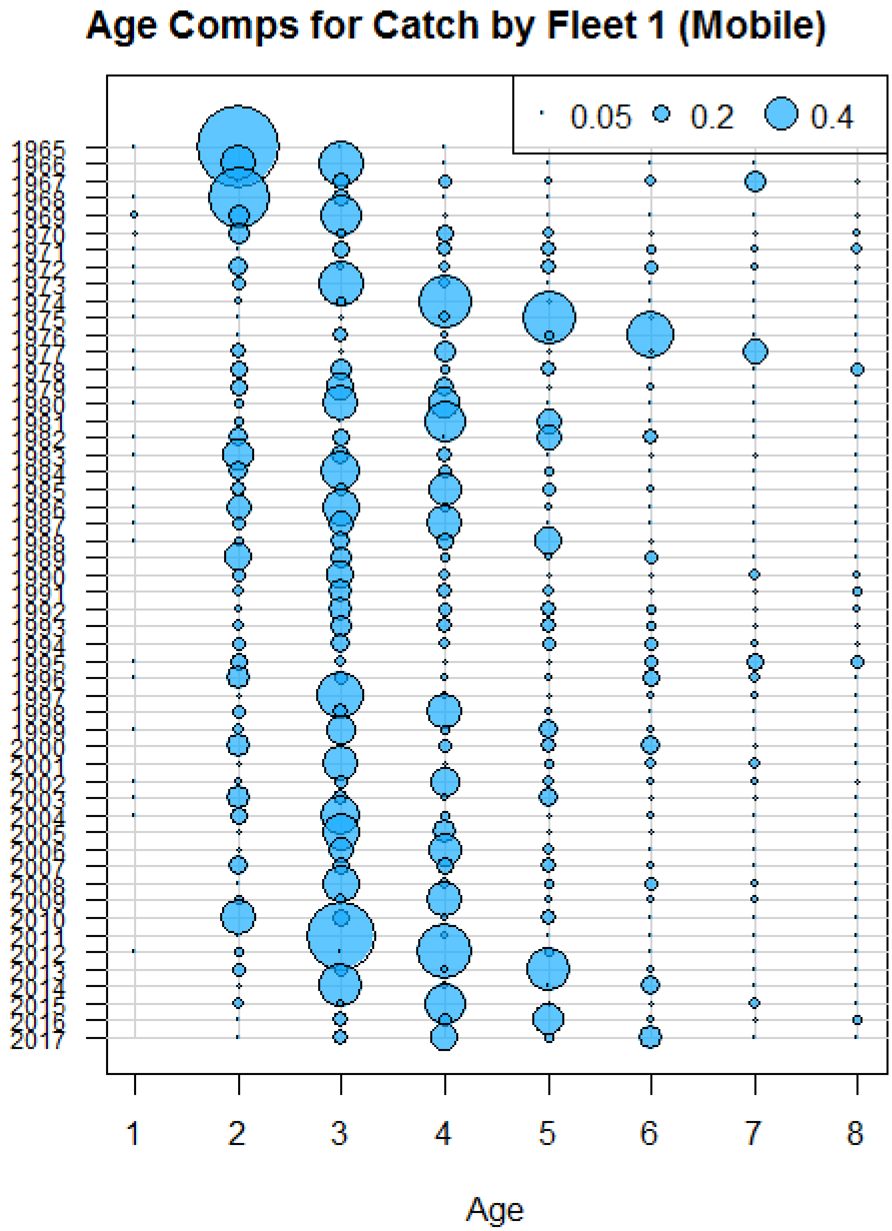


Figure B1- 9 Atlantic herring proportions at age for the fixed fleet

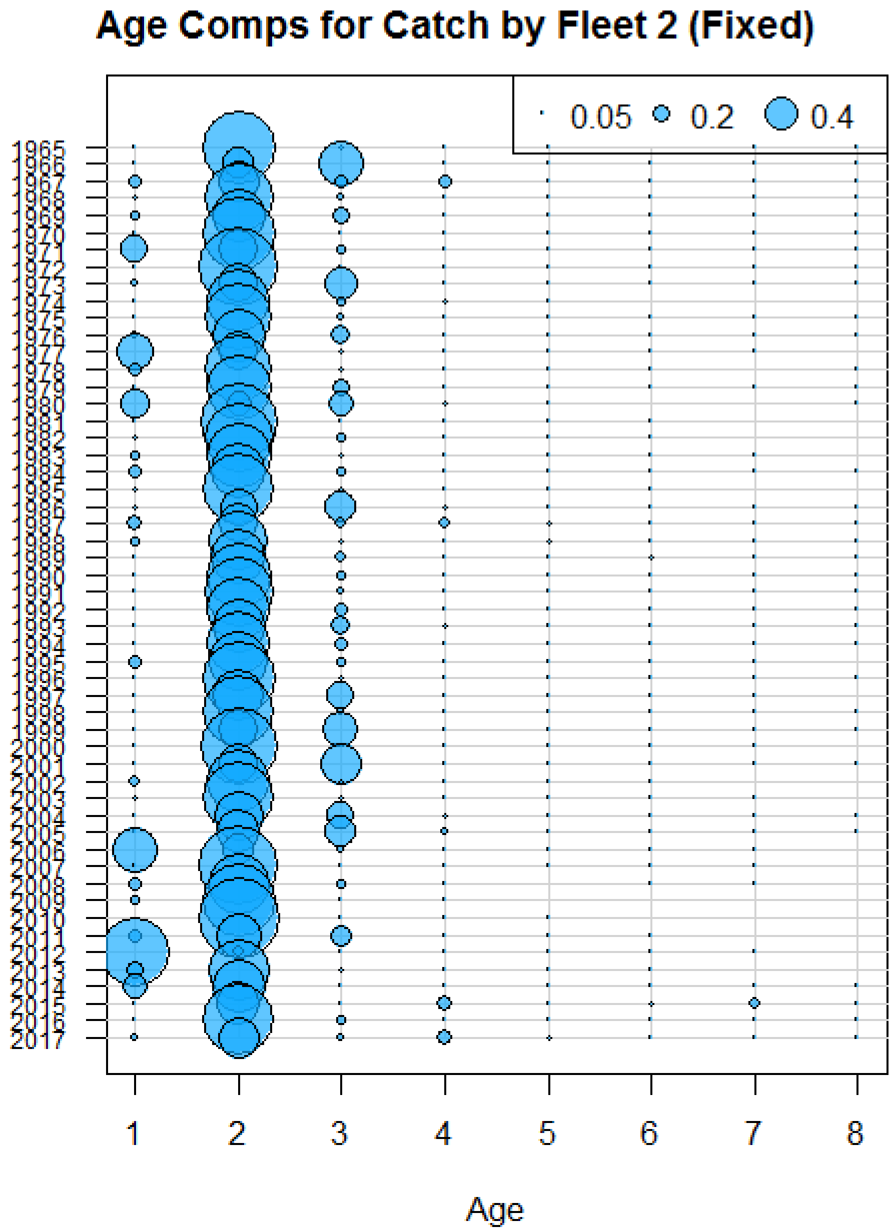


Figure B1- 10 Atlantic herring spawning stock biomass weights (kg) at age

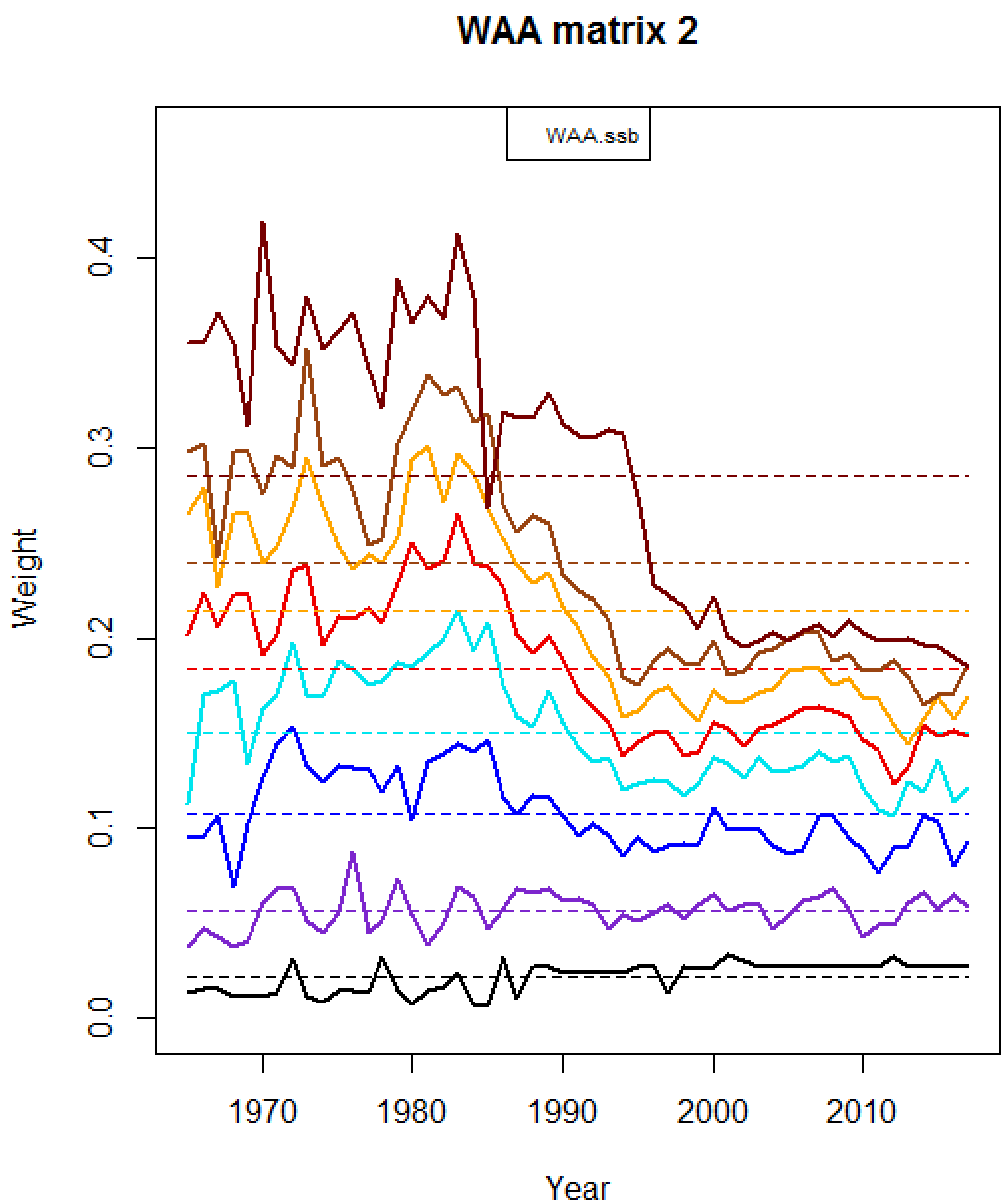
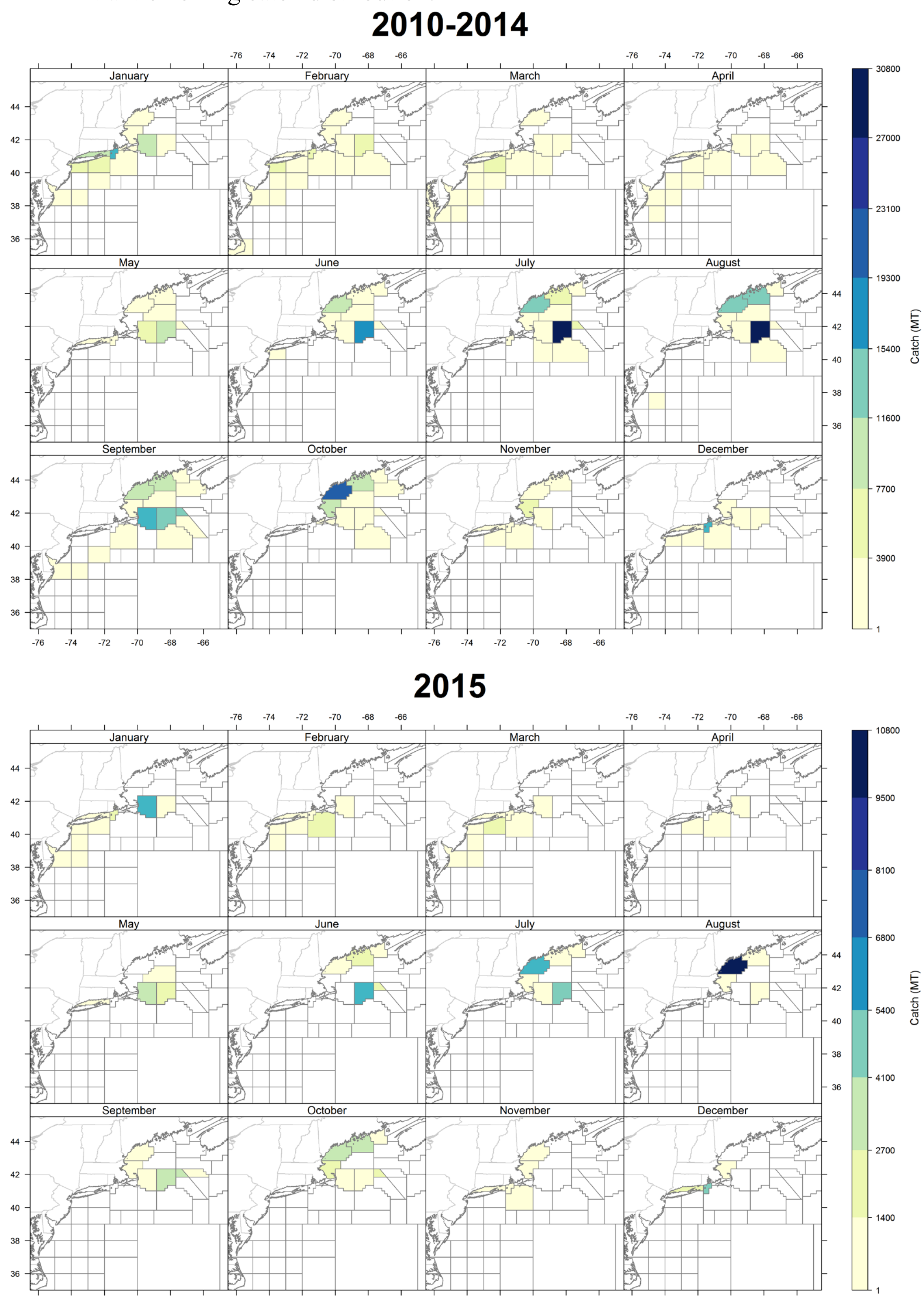
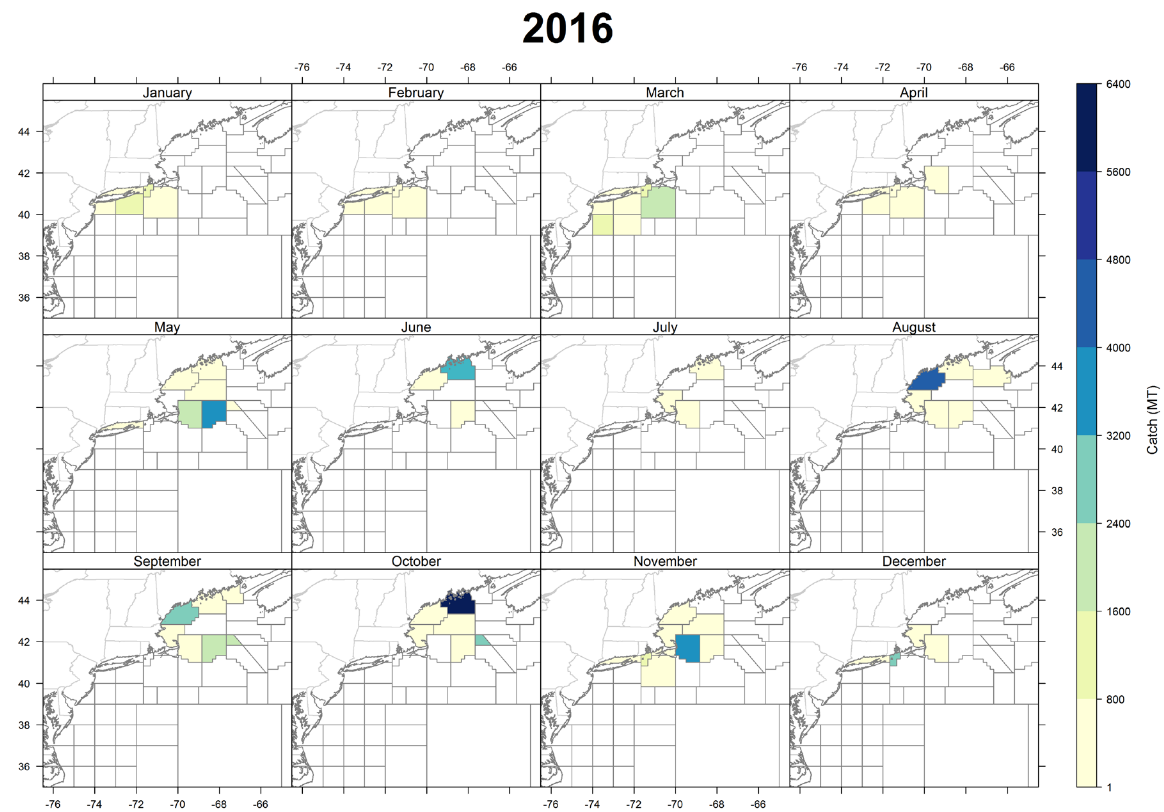


Figure B1- 11 Atlantic herring catch distribution.





2016
January
February
March
April
May
June
July
August
September
October
November
December
Catch (MT)
6400
5600
4800
4000
3200
2400
1600
800
1

Figure B1- 12 Atlantic Herring Fishery Management Areas

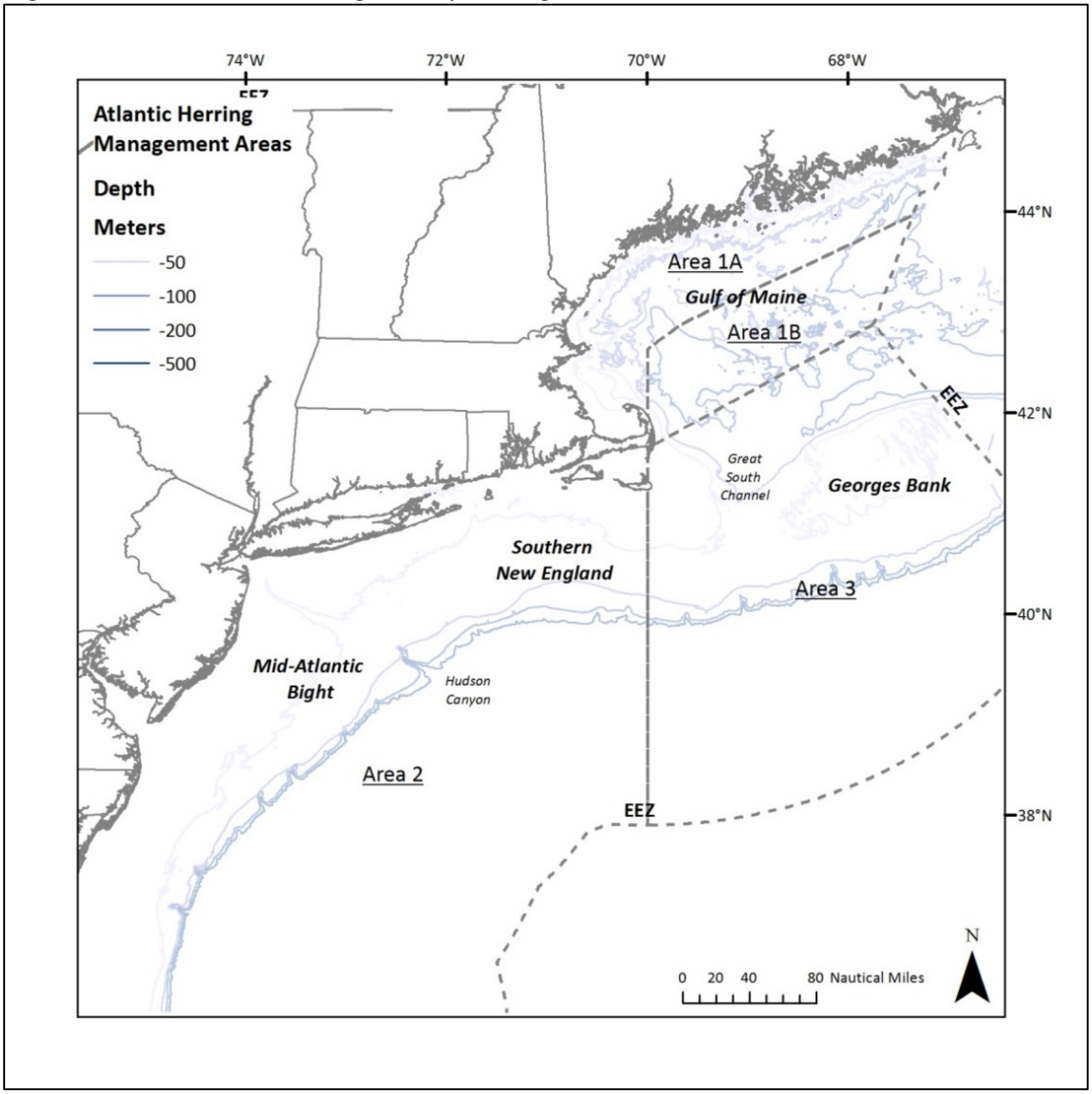
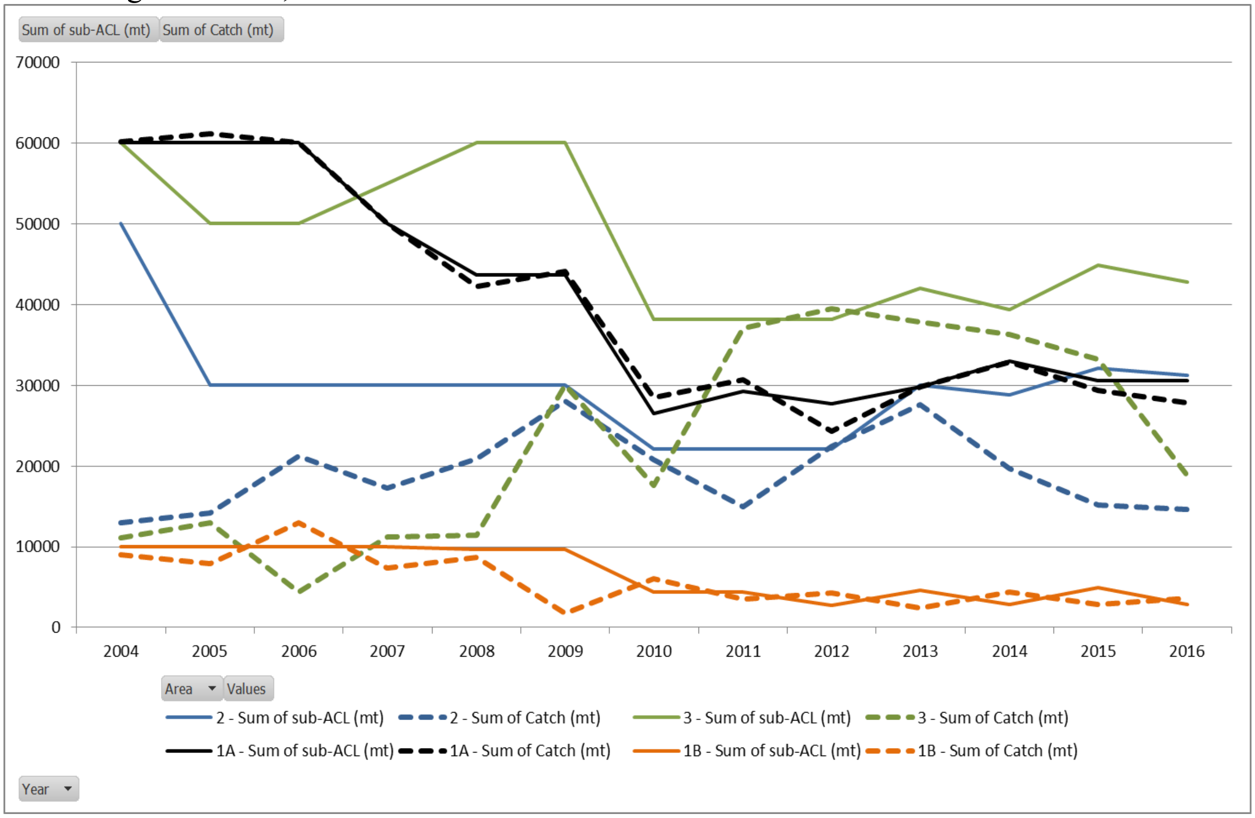


Figure B1- 13 Atlantic herring sub-ACLs (solid lines) and estimated catch (dashed lines) by year and management area, 2004-2016.



TOR B2: *Present the survey data used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, age-length data, food habits, etc.). Characterize the uncertainty and any bias in these sources of data.*

NMFS bottom trawl surveys

NMFS spring and fall bottom trawl surveys began in 1968 and 1963, respectively, and have continued through 2017. All survey tows in the spring and fall were conducted using the FRV Delaware II, FRV Albatross IV, or FSV Henry B. Bigelow. The Albatross IV was used for most tows in most years prior to 2009. In the spring, however, the Delaware II was responsible for most or all catches in 1973, 1979-1982, 1989-1991, 1994, and 2003. In the fall, the Delaware II was responsible for most or all of the catches in 1977-1978, 1980-1981, 1989-1991, and 1993. The Bigelow has been used exclusively since 2009. To ensure that changes in the indices were more reflective of changes in herring abundance and not due to differences in vessel catchability, Delaware II catches were calibrated to Albatross IV equivalents. Calibration coefficients were based on paired tow experiments (Byrne et al., 1991). Catch numbers from the Delaware II were multiplied by 0.59, and this value was constant among seasons and lengths (Byrne et al. 1991). A range of models used to develop the calibration coefficients for converting Bigelow catches to Albatross IV catches were previously explored and applied in assessment models (Miller et al. 2010; NEFSC 2012). Rather than convert Bigelow catches to Albatross IV equivalents in this assessment, however, the bottom trawl survey index during 2009-2017 (when the Bigelow was used) was treated as a separate survey time series with catchability and selectivity estimated separately from the Albatross IV years. This decision was made because the switch to the Bigelow represents a long-term shift in the survey vessel with known catchability and selectivity differences from previous years. The number of years available for the Bigelow is also now sufficient to estimate relatively precise catchability and selectivity parameters. Treating 2009-2017 as a separate time series was preferred over continued use of the calibration coefficients (Miller et al. 2010; NEFSC 2012) because the calibration coefficients were estimated based on a single year of paired tow experiments and subject to measurement and estimation uncertainties (NEFSC 2012) that are difficult to carry forward into assessment model estimation. Conversely, treating 2009-2017 as a separate time series may allow the estimated difference in catchability to alias other model misspecifications (e.g., the estimated changes in catchability among years may be due to something other than catchability; NEFSC 2008). Thus, while treating 2009-2017 as a separate time series was preferred, assessment models were also run having converted Bigelow catches to Albatross IV equivalents, and these two alternatives were compared and contrasted (see TOR B4). The fall 2017 survey did not cover some survey strata in the mid-Atlantic region (strata 5-12; Figure B2- 1). To account for this inconsistent spatial coverage, a linear regression was fit to the aggregate fall survey indices (arithmetic mean numbers per tow) from 2009-2016 estimated with (dependent variable) and without (independent variable) these strata. This regression was used to calibrate the fall 2017 survey observation (aggregate and at age) to a value assumed equivalent to having sampled the entire survey area. The Working Group noted that the regressions fit to the aggregate indices and indices at age were similar, and that the difference between the uncalibrated and calibrated values (100.9 uncalibrated to 78.6 calibrated) were within the 90%CI of the uncalibrated index. Consequently, this issue was considered relatively inconsequential.

Herring age samples in the spring and fall surveys were collected beginning in 1987. In previous assessments for years prior to 1987, age specific indices were estimated by using age-length keys developed mostly from commercial catch data. Previous assessments, however, have found significant and inexplicable differences in age-length keys from survey and commercial sources and so this practice was abandoned (NEFSC 2012; Appendix B1). Arithmetic mean numbers per tow and associated coefficients of variation in each year were used as indices of Atlantic herring abundance, and age composition since 1987 data was used in assessments Figure B2- 2; Figure B2- 3; Figure B2- 4). As in previous assessments, age-1 survey observations were excluded from the indices because age-1 fish are not selected by the trawl gear, and most observations are thought to be measurement uncertainty as opposed to reflective of changes in herring abundance. Length frequencies were also provided (Figure B2-5).

The trawl doors used on the NMFS spring and fall bottom trawl surveys changed in 1985. Previous assessments have split the spring and fall surveys into separate time series to account for the associated catchability difference caused by the change in trawl doors. This decision was also supported by residual patterns in assessment fit. This practice was continued for this assessment. Ultimately, the spring and fall surveys were each split into three separate series to account for the door change in 1985 and the change to the Bigelow vessel in 2009 (spring: 1968-

1984, 1985-2008, 2009-2017; fall: 1963-1984, 1985-2008, 2009-2017).

The NMFS winter survey was conducted during 1992-2007. As in previous assessments, the winter survey was eliminated from consideration as an index of abundance because of concerns over inconsistent spatial coverage among years and lack of fit in previous assessments.

A NMFS summer survey directed at shrimp began in 1983 and has continued through 2017, with the exception of 1984. The spatial extent of this survey is limited to the Gulf of Maine (Figure B2- 6). The working group agreed, however, that fish from the entire complex are mixed in the Gulf of Maine during the summer, and so this survey would be a valid index of the entire stock complex. Age data for Atlantic herring have never been collected on this survey. This survey occurs approximately half way between the spring and fall bottom trawl surveys, however, and so the average of the age-length keys from the spring and fall surveys were used to develop indices at age for the summer survey. Arithmetic mean numbers per tow and associated coefficients of variation in each year were proposed as indices of Atlantic herring abundance (Figure B2- 7; Figure B2- 8). Length frequencies were also provided (Figure B2- 5).

State surveys

Massachusetts Division of Marine Fisheries (MA DMF) spring and fall bottom trawl surveys began in 1977 and have continued uninterrupted through 2017. Joint Maine and New Hampshire spring and fall bottom trawl surveys began in 2001 and 2000, respectively, and have continued uninterrupted through 2017. These surveys cover state waters ≤ 3 nm from shore, and cover a relatively small proportion of the stock, in terms of both spatial coverage and size/age composition. Consequently, the working group agreed that they should not be used for the assessment.

An index from food habits data

An index of herring abundance was developed from stomach contents data collected on the NMFS spring and fall bottom trawl surveys (see TOR B3 for details about stomach contents data collection). The methods were identical to Deroba (2018) and only a brief update and overview were provided here. Data were identical to that in Deroba (2018) except the time series extended through 2016 and some additional observations were added to the years 2012-2014 that had not been previously analyzed. Each stomach observation was essentially treated as a catch-per-effort observation, and a delta approach (hurdle model) was used to develop the index of herring abundance. Separate generalized additive mixed models (GAMMs) were fit to:

(i) the amount of herring observed in predator stomachs using only those stomachs in which herring were identified, and (ii) a model of the probability of a stomach containing herring using data from all sampled stomachs. After using a AIC for model selection, the overall best GAMM model for the amount of herring in stomachs with positive herring occurrence included a fixed effect for the product factor of area and season α_{as}, a smooth for predator length $f(l_i)$, and random intercepts for year b_y, predator species m_r, the interaction of year and the product factor of area and season $d_{y,as}$, and the interaction of year, predator species, and the product factor of area and season $g_{y,r,as}$:

$$ln(h_i) = \mu + \alpha_{as} + f(l_i) + b_y + m_r + d_{y,as} + g_{y,r,as} + \varepsilon_i \ .$$

The overall best GAMM model for the probability of a positive herring occurrence included fixed effects for year β_y and the product factor of area and season, smooths for predator length and the amount of herring catch in the tow from which a stomach was sampled $f(c_i)$, and random effects for predator species, and the interaction of predator species and the product factor of area and season $n_{r,as}$:

$$ln\left(\frac{p_i}{(1-p_i)}\right) = \mu + \beta_y + \alpha_{as} + \ f(l_i) + \ f(c_i) + m_r + n_{r,as} \ .$$

An annual index of herring abundance I_y was developed using the year effect coefficients from the GAMM for the amount of herring in stomachs b_y, and the probability of a stomach containing a herring β_y:

$$\hat{h}_y = e^{\mu + b_y};$$

$$\hat{p}_y = \frac{e^{\mu+\beta_y}}{\left(1+e^{\mu+\beta_y}\right)};$$

$$I_y = \hat{h}_y \times \hat{p}_y$$

where μ was the overall model intercept from one of the GAMMs. Estimating measures of uncertainty for this index is not straightforward because methods for combining uncertainty measures from the multistage sampling of the stomachs within the bottom-trawl survey and those from the separate GAMMs have not been developed. Approximate CVs were estimated, however, by summing the year effect variance parameters from each model, and then converting this aggregate variance to a CV for the annual indices of abundance.

The index of abundance was relatively imprecise (Figure B2- 9). The index of abundance was also sensitive to the data used in the GAMM models. Updating the time series through 2016 caused a decrease in the index, mostly in recent years (Figure B2- 10). Eliminating spiny dogfish stomach observations, the most common herring predator in the food habits database, caused a similar change (Figure B2- 11). Removing spiny dogfish had different effects on each of the GAMMs, with the scale of the probability of observing a herring decreasing with the removal of spiny dogfish and the variance among years in the amount of herring in stomachs reducing to near zero (Figure B2- 11). A retrospective analysis of the index of abundance, where one year of data is sequentially dropped from each of the models, was relatively stable (Figure B2- 12). Thus, the models used to derive the index of abundance were insensitive the number of years of data, but relatively sensitive the amount of data contained within each year and throughout the time series. This instability led the Working Group to eliminate the food habits index from consideration in assessment modeling, but assessment sensitivity runs were conducted and further research on this topic was encouraged.

Acoustic index

Water-column acoustic data were collected from 1998 to 2017 during the NEFSC's autumn stratified-random survey along the continental shelf from Cape Hatteras, North Carolina to Canadian waters in the Gulf of Maine (Figure B2- 13). Details of acoustic data acquisition, processing, and post-processing are detailed in Jech and Michaels (2006), Jech and Stroman (2012), Jech (2014), and Jech and Sullivan (2014) but a brief description is provided here.

All echosounders and frequencies were calibrated prior to each survey, and usually near the completion of the cruise using the standard target method (Foote et al., 1987). Transducers were calibrated using either copper (Cu) or tungsten carbide with 6% cobalt binder (WC) spheres, depending on year and conditions. For Cu spheres, a 64-mm diameter Cu sphere was used to calibrate the 18-kHz echosounders, a 60-mm Cu sphere was used to calibrate the 38-kHz echosounders, and a 23-mm Cu sphere was used to calibrate the 120-kHz echosounders. The 38.1-mm diameter WC was used to calibrate the 18, 38, 70 and 200-kHz echosounders.

Water-column acoustic data during the stratified-random bottom survey were collected continuously as the vessel transited between randomly-located trawl-haul sites and during all deployments (Figure B2- 13). Trawl locations were selected randomly within bathymetrically-defined strata for each cruise (Azarovitz et al., 1997). The sampling order was selected by minimizing travel time among trawl locations, thus while locations were random, the order was not. Data from 1998-2005 were collected on the NOAA ship *Albatross IV* (hereafter *Albatross*

IV). Data collected from 2009-2012 were collected on the NOAA ship *Henry B. Bigelow* (hereafter *Bigelow*). Data collected during 2007-2008 were collected on both vessels as part of inter-ship comparison surveys (Miller et al., 2010). No data were collected in 2006 and data in 2010 were collected only to 50 m, thus were not used for analysis. An EK500 echosounder collected 12, 38, and 120-kHz data on the *Albatross IV* from 1998-2002. In 2003, the EK500 was replaced with 18-, 38-, and 120-kHz EK60 echosounders. The 12-kHz single-beam, and 18, 38, and 120-kHz split-beam transducers were located downward-looking on the keel. The *Bigelow* collected acoustic data from EK60 echosounders operating at 18, 38, 120, and 200 kHz from 2007-2012 and a 70 kHz EK60 echosounder was added in 2009. Beam angles were 16^{o} for the 12 kHz, 11^{o} for the 18 kHz, and 7^{o} for all other frequencies. The *Albatross IV*'s EK500 was calibrated in 1996, March 2001, and April 2002. The *Albatross IV*'s EK60 was calibrated in 2008, just before decommissioning. Gain settings for years without calibrations were applied from years with calibrations (Jech, 2014). The *Bigelow*'s EK60s were calibrated in spring 2007, and then immediately prior to each survey from 2008-2012. All calibrations followed protocols set from the systematic survey. *Bigelow* 38-kHz gain settings were very stable with ±0.1 dB variation over the calibrations.

Multi-frequency volume backscatter (S_v, dB·re 1 m^{-1}) data were post-processed and classified as described in Jech and Michaels (2006) and Jech (2014) using Myrix Echoview software (v8+; GPO Box 1387 Hobart, Tasmania, Australia, www.echoview.com). Briefly, echograms were scrutinized to remove acoustic and electrical noise, erroneous seafloor detections, data shallower than 10 m, and data deeper than 0.5 m above the sea floor. When 12 or 18, 38, and 120-kHz data were available, the indices of the echogram pixels that contained S_v values greater than -66 dB in all three frequencies were mapped to the 38-kHz echogram and that echogram was used to visually classify Atlantic herring. In cases where only one or two frequency data were available, a modified version of the methods described in Jech and Michaels (2006) was applied (Jech, 2014).

Visual scrutiny of the acoustic data from the stratified-random survey sometimes suggested the presence of Atlantic herring in the water column, but the species composition of the bottom trawl catch co-located or in the immediate vicinity of the acoustic data did not support apportioning acoustic backscatter to Atlantic herring (e.g., Figure B2- 14). In these cases, these aggregations were scrutinized as "unverified" Atlantic herring and used to evaluate the level of uncertainty in examining acoustic data collected during the stratified-random surveys.

After the S_v data were scrutinized for Atlantic herring, area backscattering, also known as nautical area scattering coefficient (NASC, m^2 nmi^{-2}; MacLennan et al. 2002), attributable to Atlantic herring, was generated by vertically integrating throughout the water column and horizontally averaging into 0.5 nmi elementary distance sampling units (EDSU). Geographical location, date, and time were associated with each s_A value. The final water-column data were 38-kHz s_A data classified as Atlantic herring s_A in 0.5 nmi EDSU. Data analyses were done in QGIS (QGIS Development Team, 2018), R statistical package (R Core Team, 2015), and PBS Mapping (Schnute et al., 2004).

The mean s_A ($\overline{s_A}(S_f, y)$) and standard deviation ($SD(S_f, y)$) were calculated annually for each finfish stratum (S_f) for a subset of offshore finfish strata (OS_f) where only offshore strata that had at least one occurrence of acoustic backscatter classified as Atlantic herring among the years were used (Figure B2- 15; Figure B2- 16; Figure B2- 17):

$$\overline{s_A}(S_f, y) = \frac{1}{N(S_f, y)} \sum_{i=1}^{N(S_f,y)} s_A(i) \qquad (1),$$

where the number of s_A values within each stratum were different among stratum and among years (y), and all s_A values were used regardless of activity, i.e., data during steaming and trawls were included. Those mean s_A values for each stratum and year were used to calculate a stratum-area (A_{S_f}) weighted mean ($\overline{s_A}(y)$) and variance ($Var(y)$) for each year:

$$\overline{s_A}(y) = \sum_{j=1}^{M} \frac{A_{S_f}(j)}{A_{OS_f}} \overline{s_A}(j, y) \qquad (2),$$

$$Var(y) = \sum_{j=1}^{M} \left[\left(\frac{A_{S_f}(j)}{A_{OS_f}} \right)^2 \frac{SD_j(S_f, y)^2}{N_j(S_f, y)} \right] \qquad (3),$$

where there were $M = 49$ offshore strata used in this analysis, j indexes strata, and A_{OS_f} is the total area (nmi^2) of all 49 offshore strata. Table B2- 1 provides the mean and variance estimates for the offshore strata from 1998 to 2017.

Table B2- 1 Stratum-area weighted mean ($\overline{s_A}(y)$) and variance ($Var(y)$) estimates for the offshore strata where acoustic backscatter was classified as Atlantic herring for each year.

Year	Mean	Var
1998	114.85	344.14
1999	78.04	23.84
2000	191.80	2726.22
2001	112.21	120.15
2002	113.92	123.23
2003	33.83	33.05
2004	117.57	1048.22
2005	33.76	11.56
2007	33.08	71.00
2007	32.55	25.76
2008	4.54	0.27
2008	40.74	17.41
2009	52.74	22.92
2011	41.50	76.18
2012	64.65	38.43
2013	51.76	13.61
2014	93.05	68.06
2015	44.15	8.42
2016	40.48	4.54
2017	37.68	19.90

Figure B2- 1 NMFS offshore bottom trawl survey strata

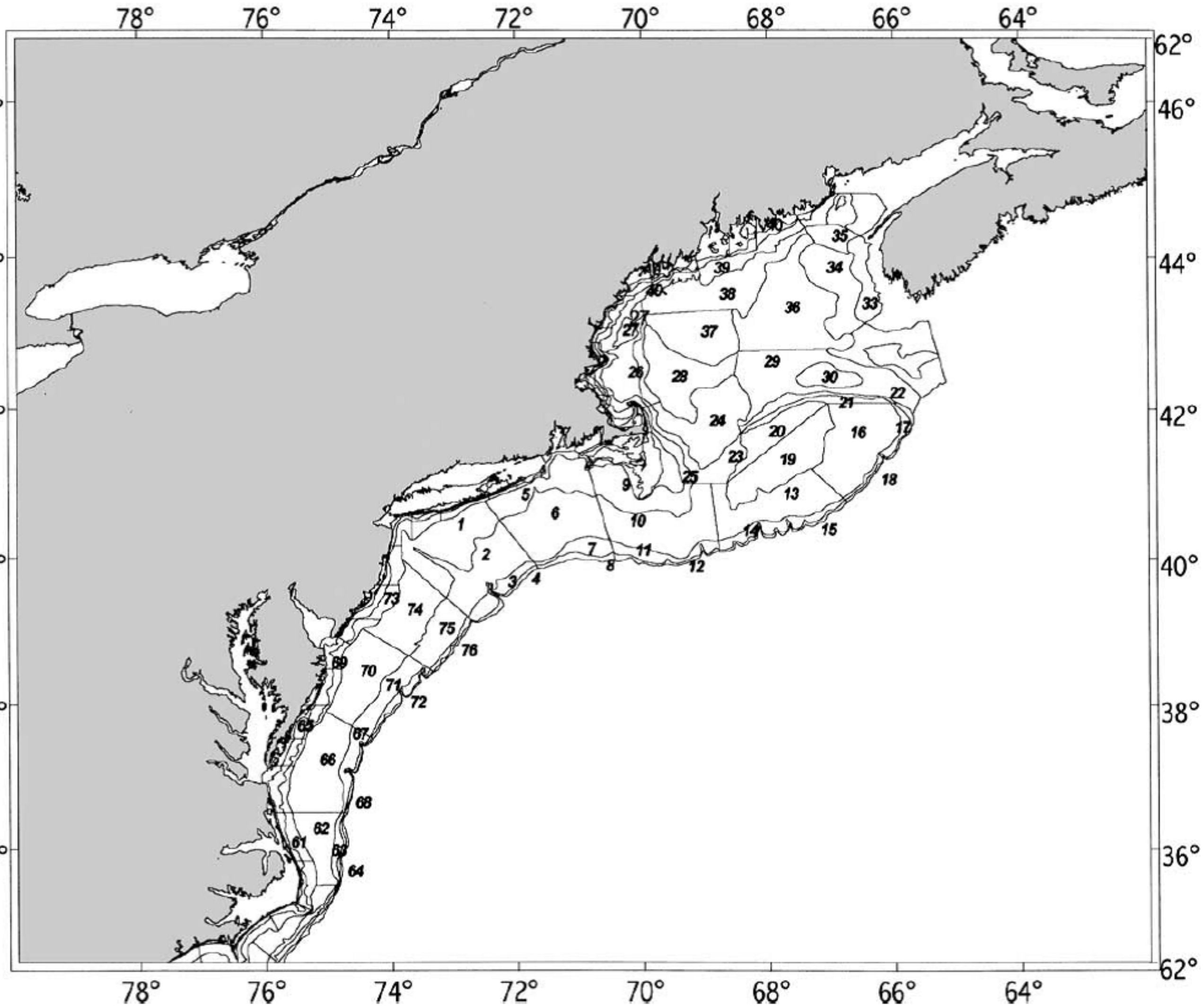
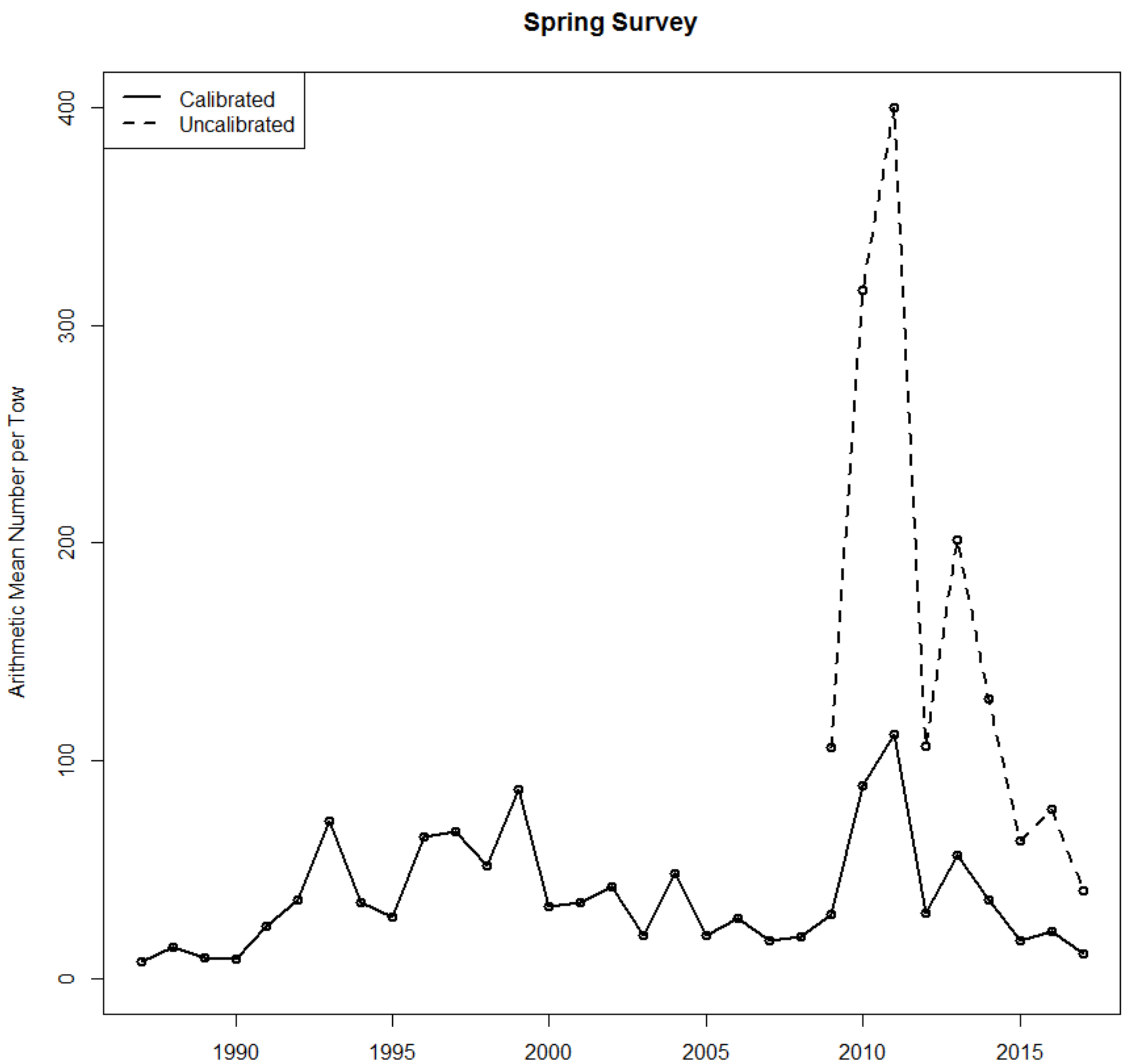


Figure B2- 2 Time series of NMFS spring bottom trawl survey Atlantic herring abundance indices with and without 90%CI.



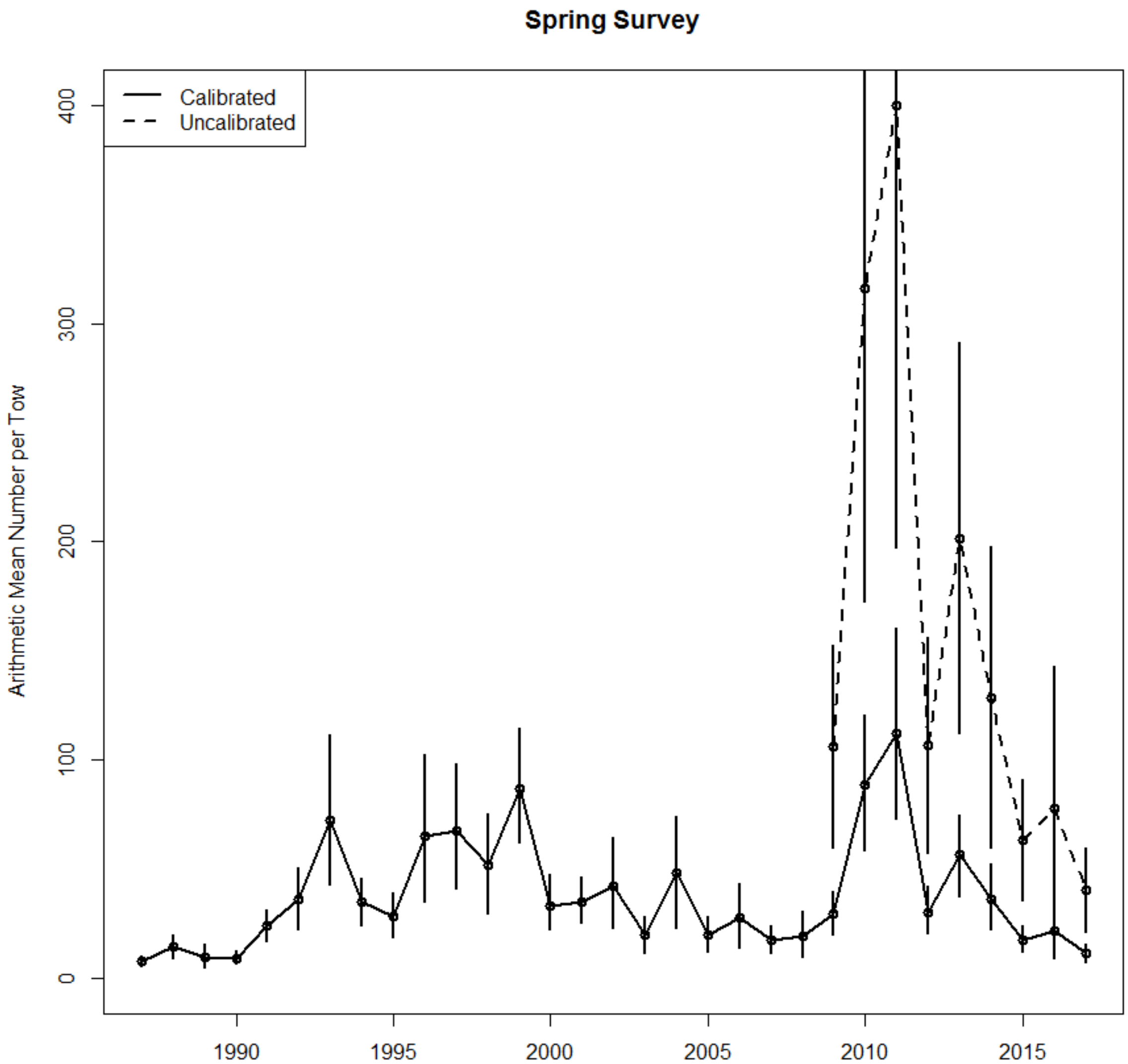
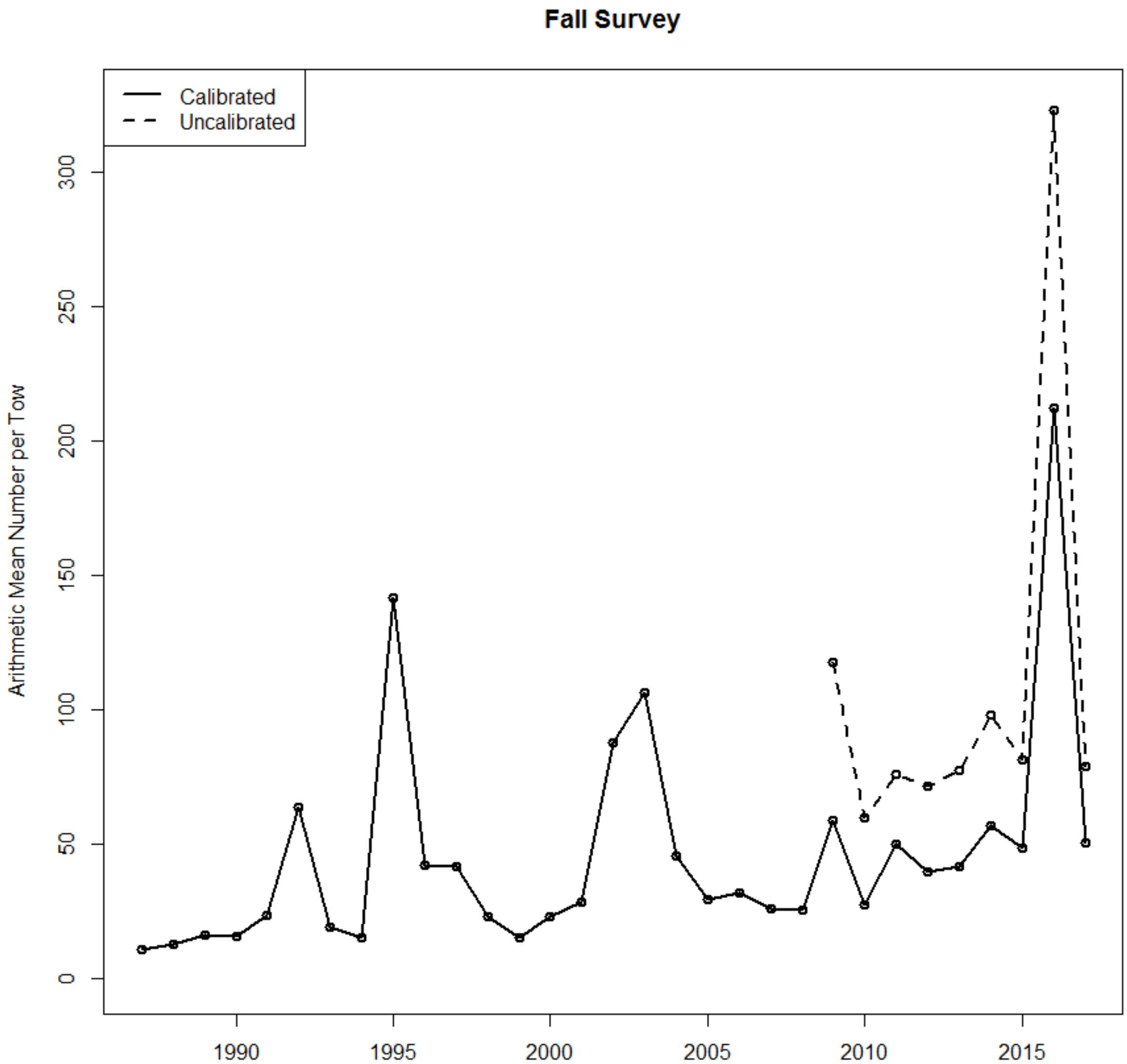

Spring Survey
Calibrated
Uncalibrated
Arithmetic Mean Number per Tow
0
100
200
300
400
1990
1995
2000
2005
2010
2015

Figure B2- 3 Time series of NMFS Fall bottom trawl survey Atlantic herring abundance indices with and without 90%CI



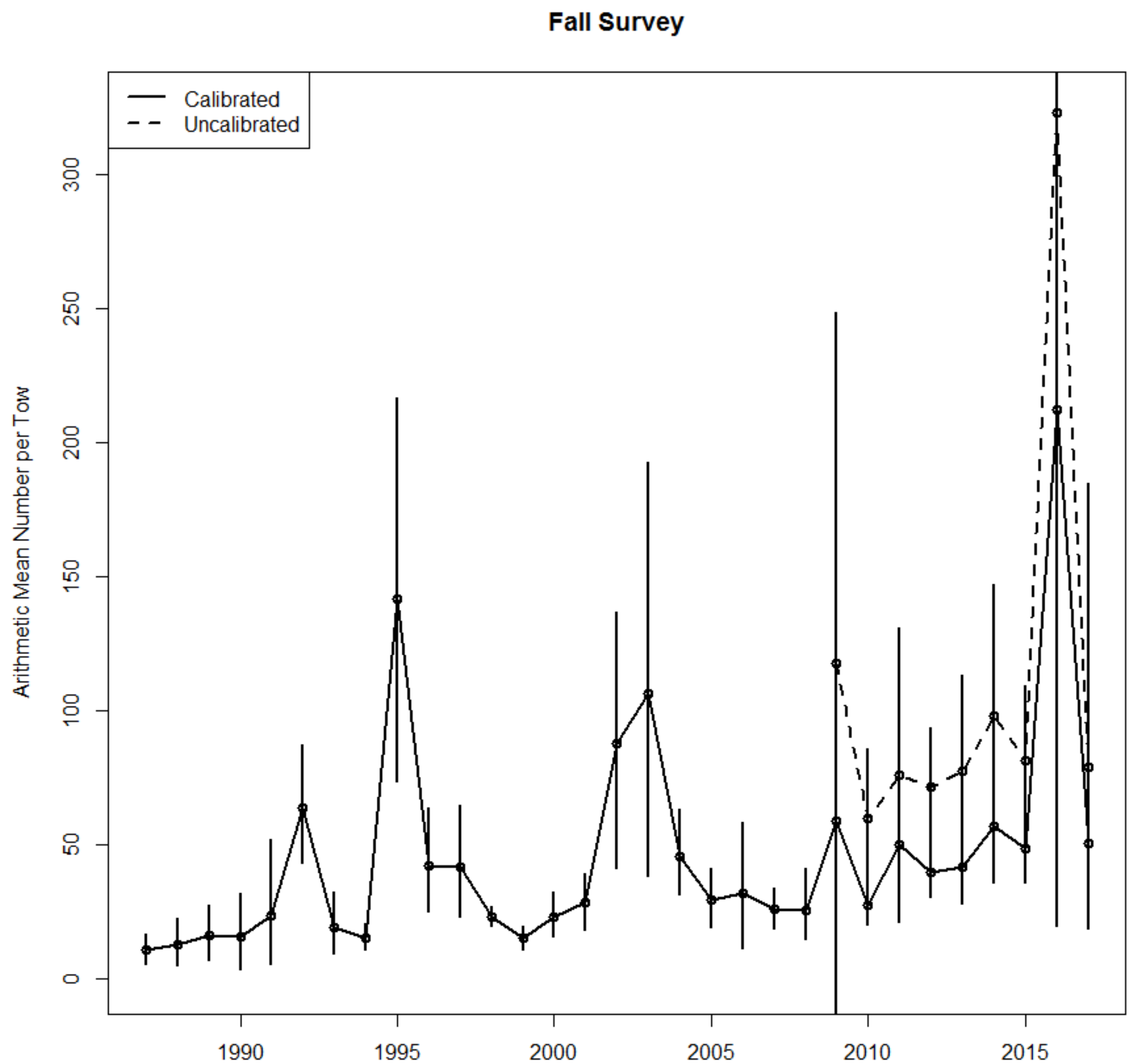
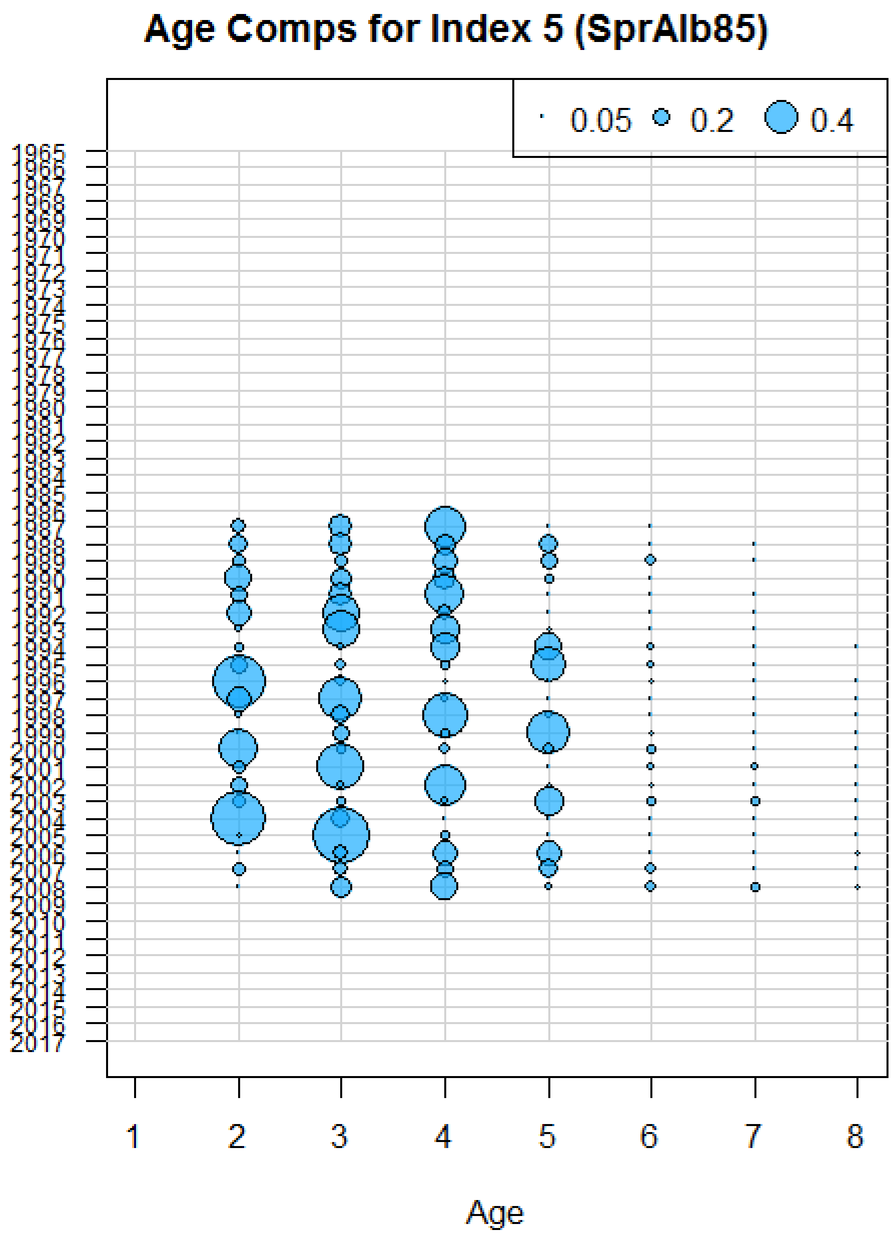
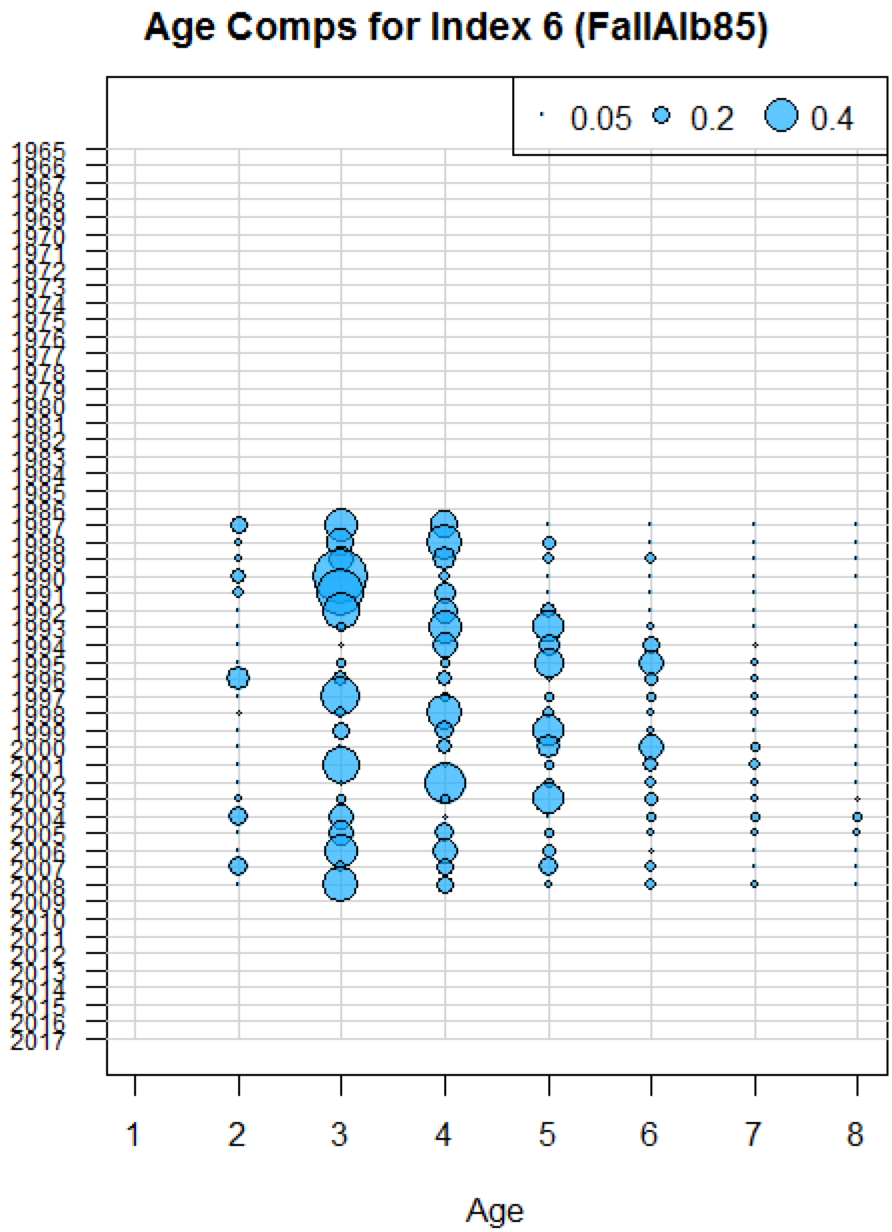
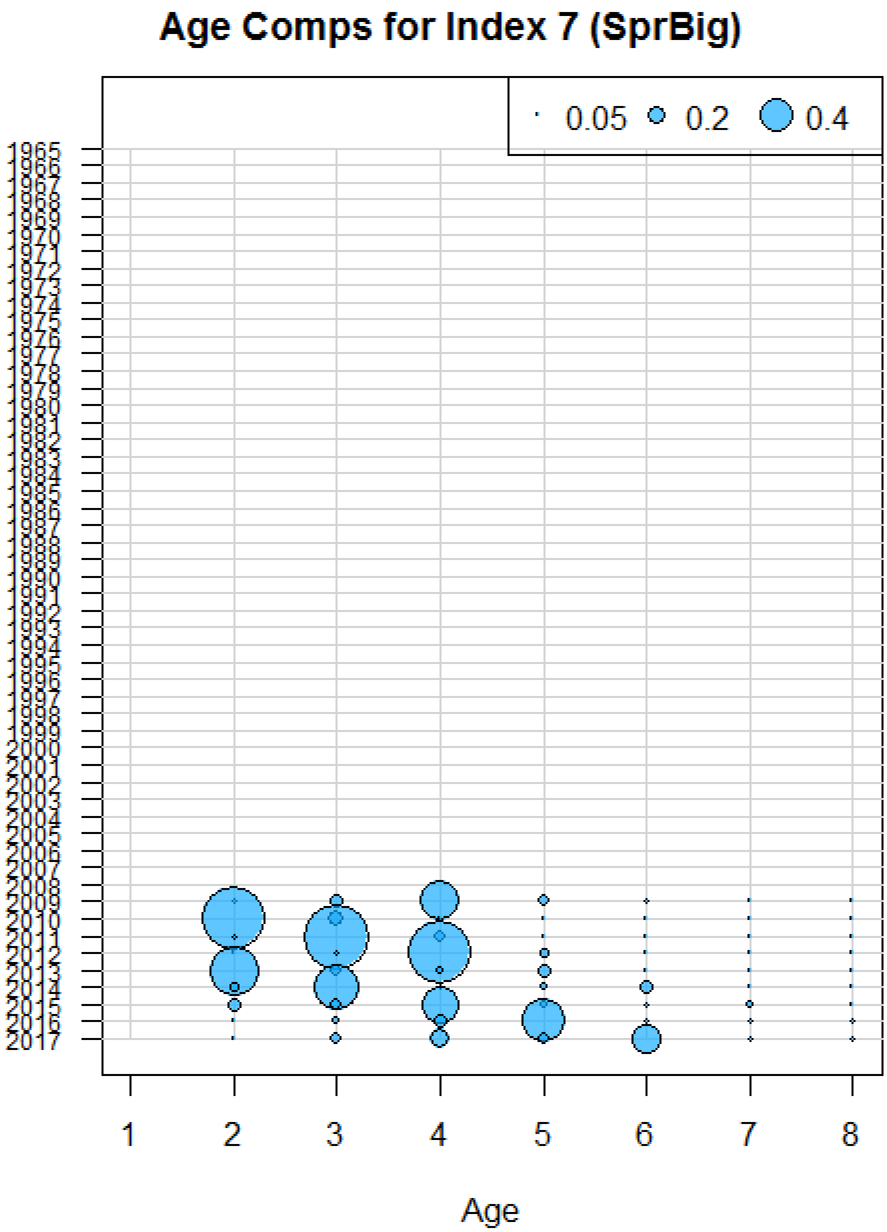

Fall Survey
Calibrated
Uncalibrated
Arithmetic Mean Number per Tow
0
50
100
150
200
250
300
1990
1995
2000
2005
2010
2015

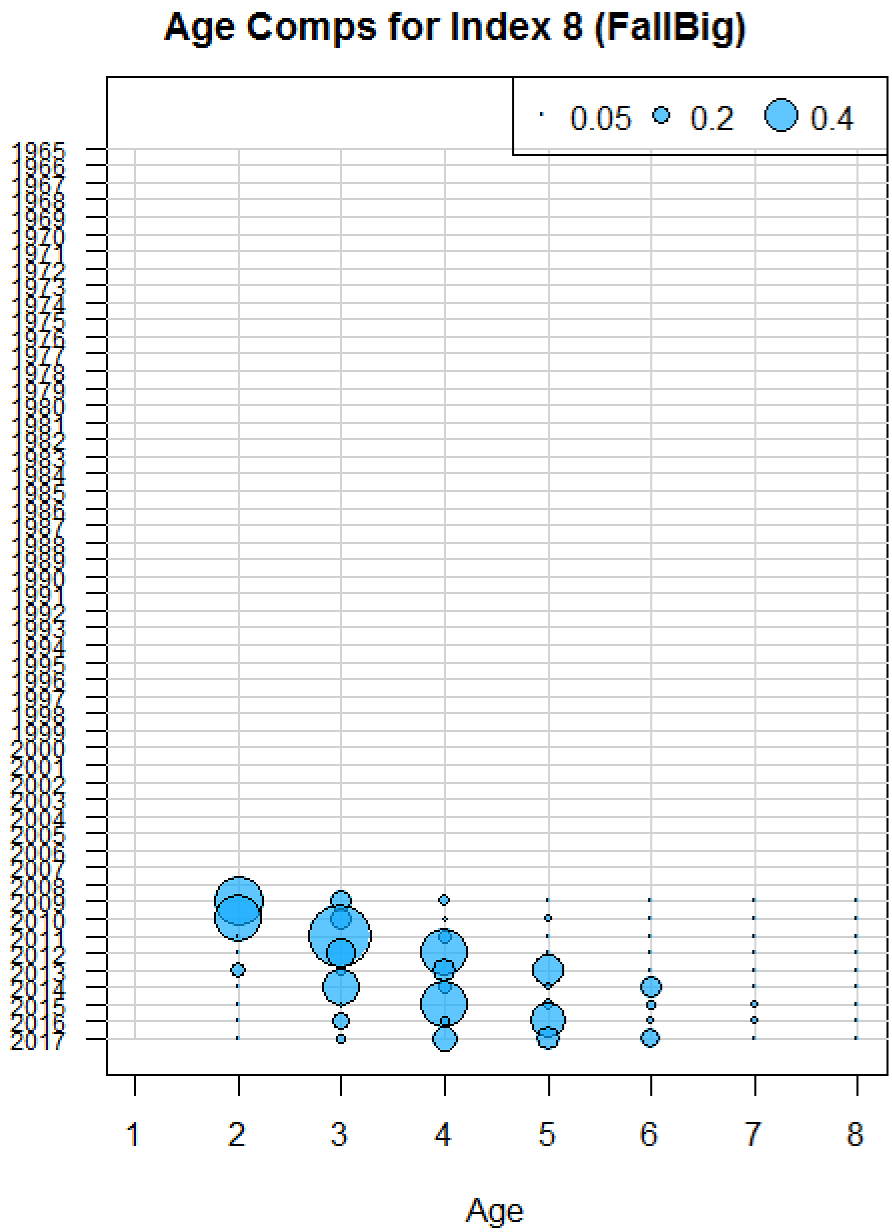
Figure B2- 4 Atlantic herring proportions at age for the spring Albatross years (SprAlb85), spring Bigelow years (SprBig), fall albatross years (FallAlb85), and fall Bigelow years (FallBig)




Age Comps for Index 6 (FallAlb85)
0.05
0.2
0.4
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
1
2
3
4
5
6
7
8
Age

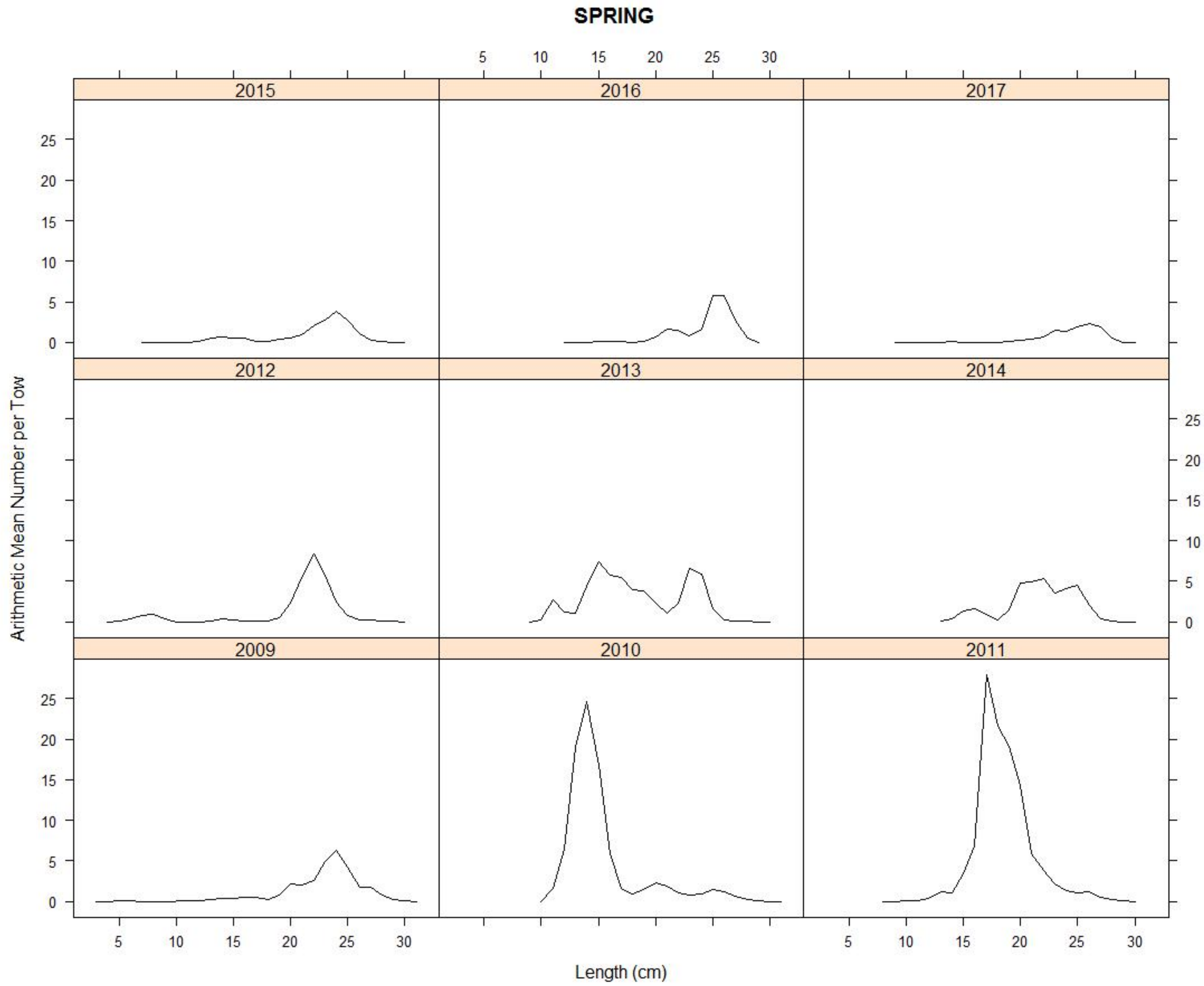


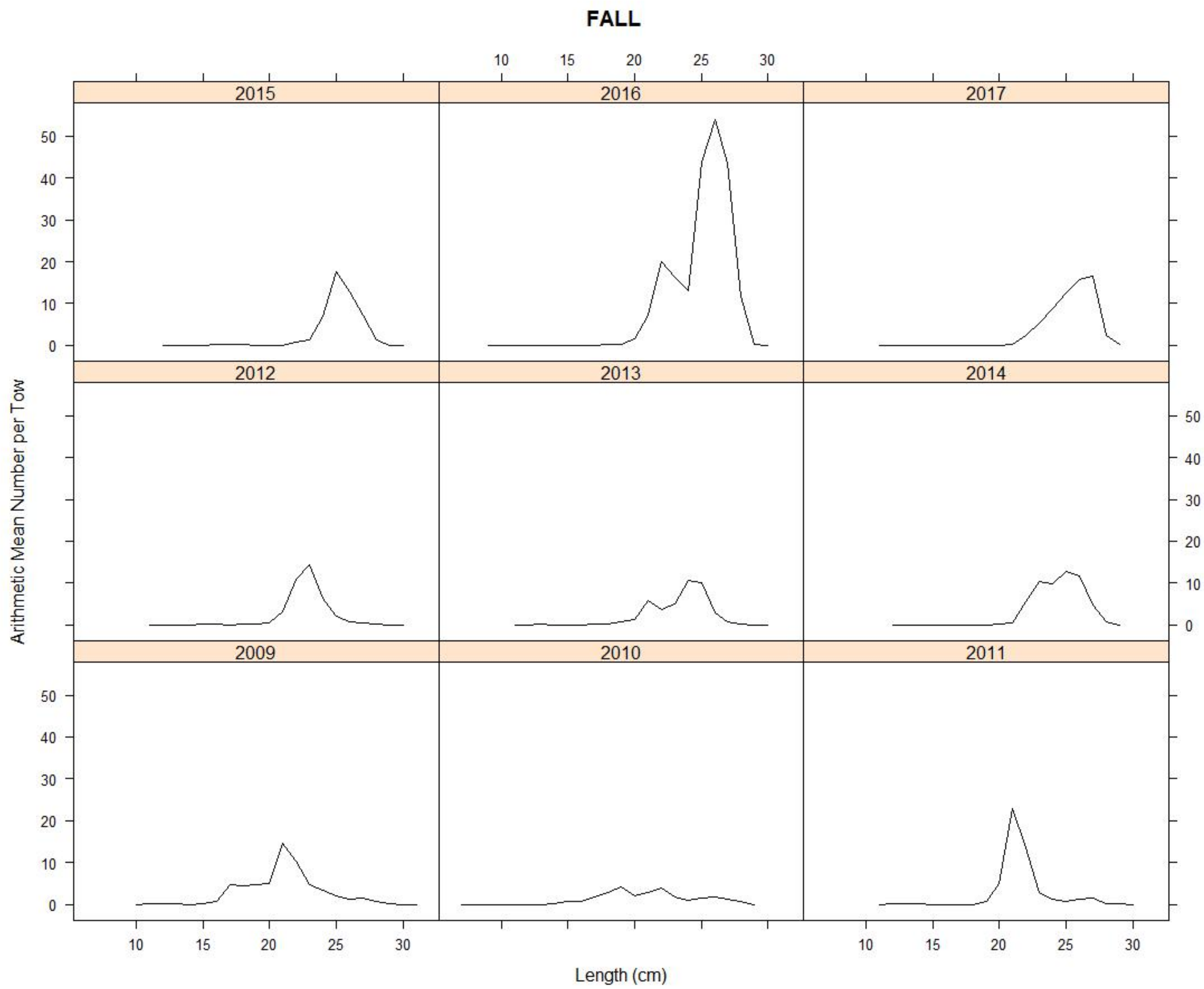
Age Comps for Index 7 (SprBig)
0.05
0.2
0.4
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
1
2
3
4
5
6
7
8
Age

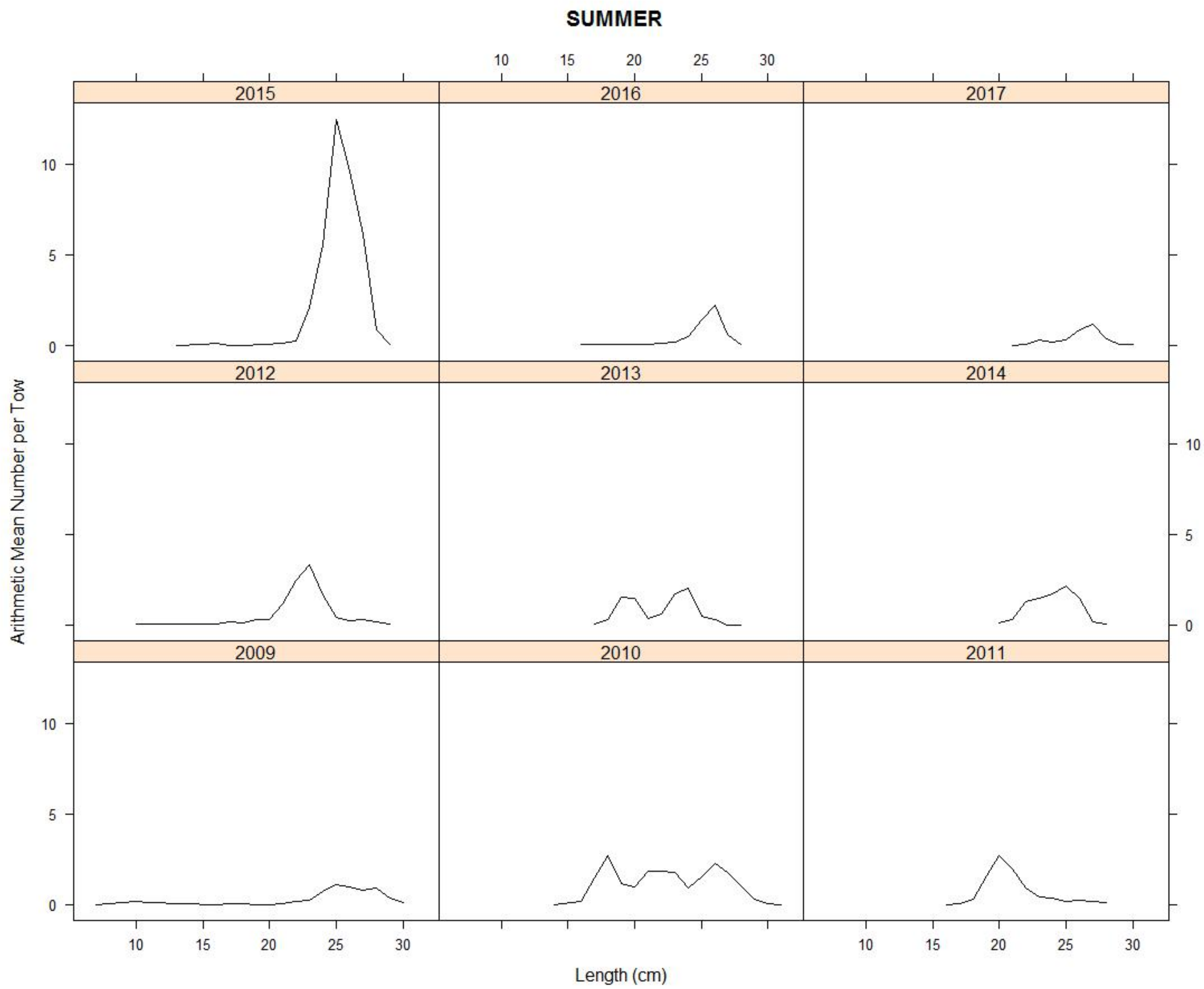


Age Comps for Index 8 (FallBig)
0.05
0.2
0.4
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
1
2
3
4
5
6
7
8
Age

Figure B2- 5 Atlantic herring length frequency from NMFS spring, fall, and summer (shrimp) bottom trawl surveys




FALL
2015
2016
2017
2012
2013
2014
2009
2010
2011
Arithmetic Mean Number per Tow
Length (cm)
0
10
20
30
40
50
10
15
20
25
30



SUMMER
2015
2016
2017
2012
2013
2014
2009
2010
2011
Arithmetic Mean Number per Tow
Length (cm)

Figure B2- 6 Location of tows taken during the NMFS summer (shrimp) survey 1983-2017.

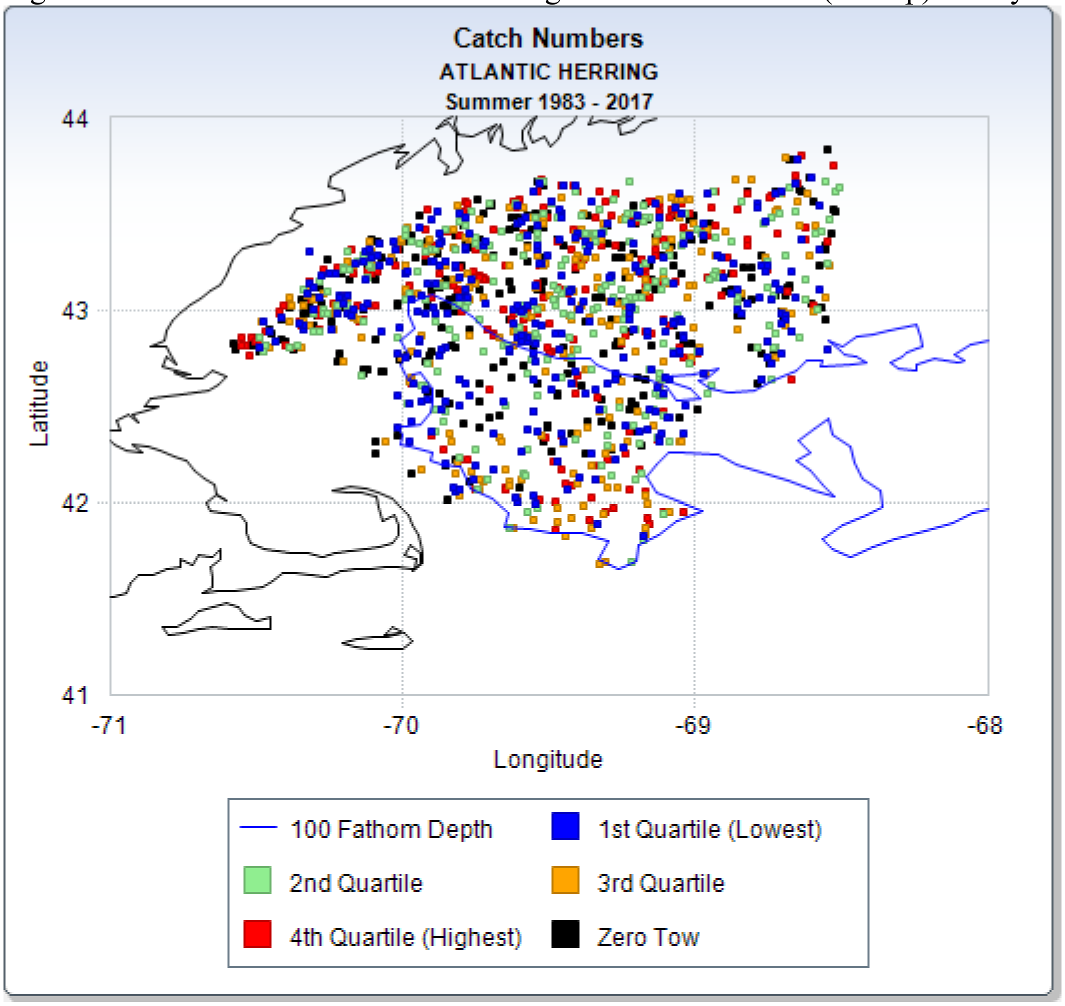


Figure B2- 7 Summer NMFS bottom trawl survey abundance index time series for Atlantic herring with 90%CI

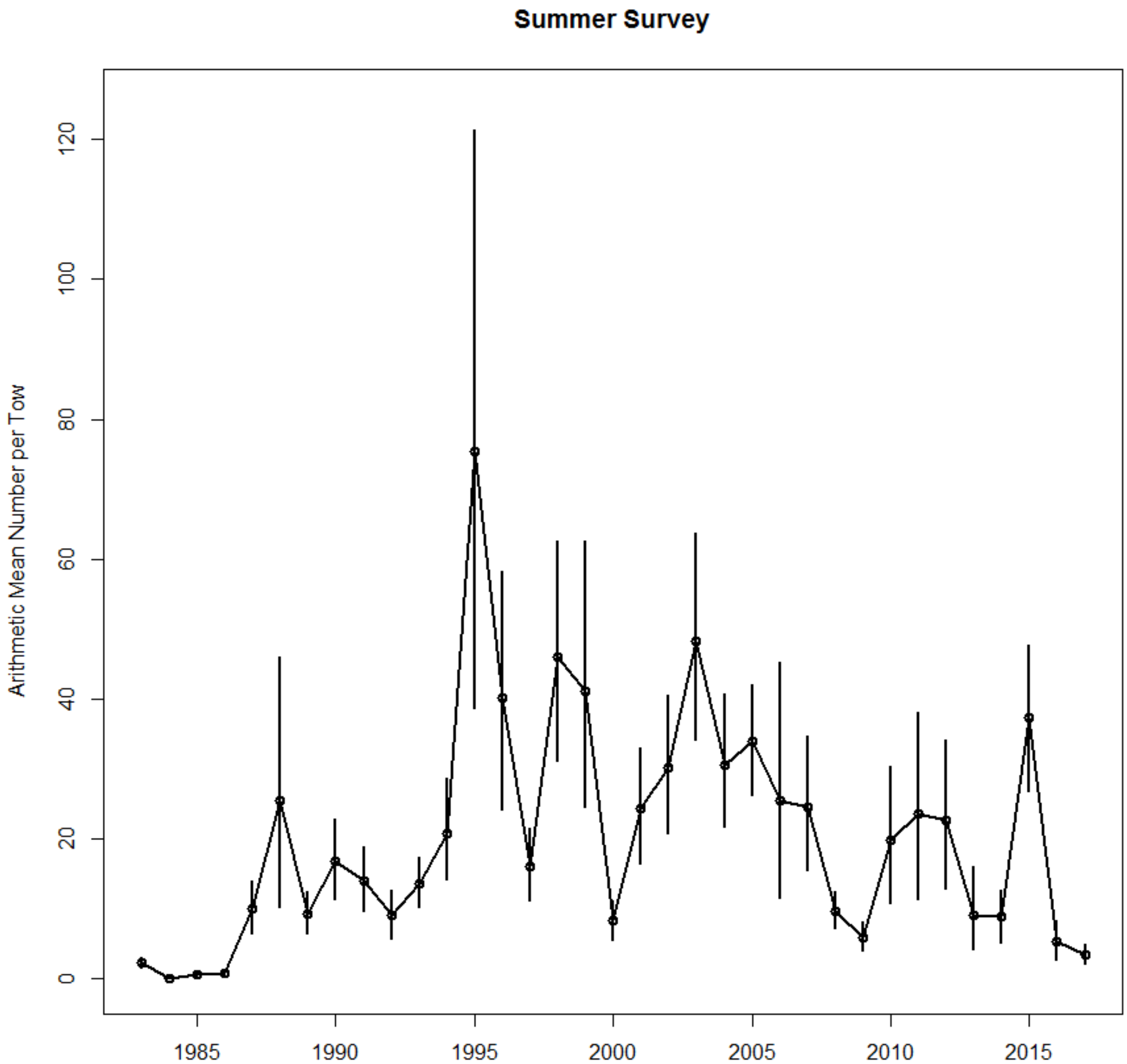


Figure B2- 8 Atlantic herring proportions at age from the NMFS summer (shrimp) survey.

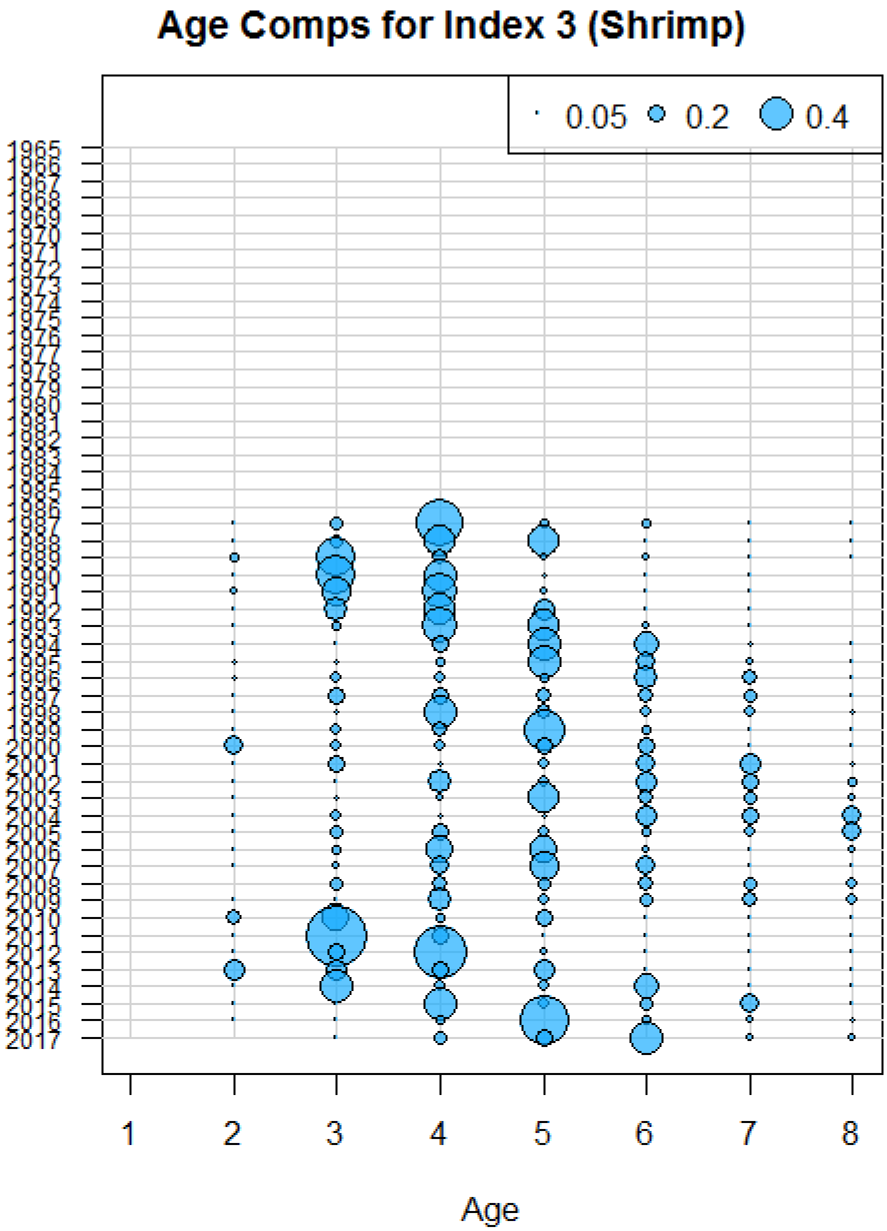


Figure B2- 9 Index of herring abundance derived from NEFSC stomach contents data +/- 2SD

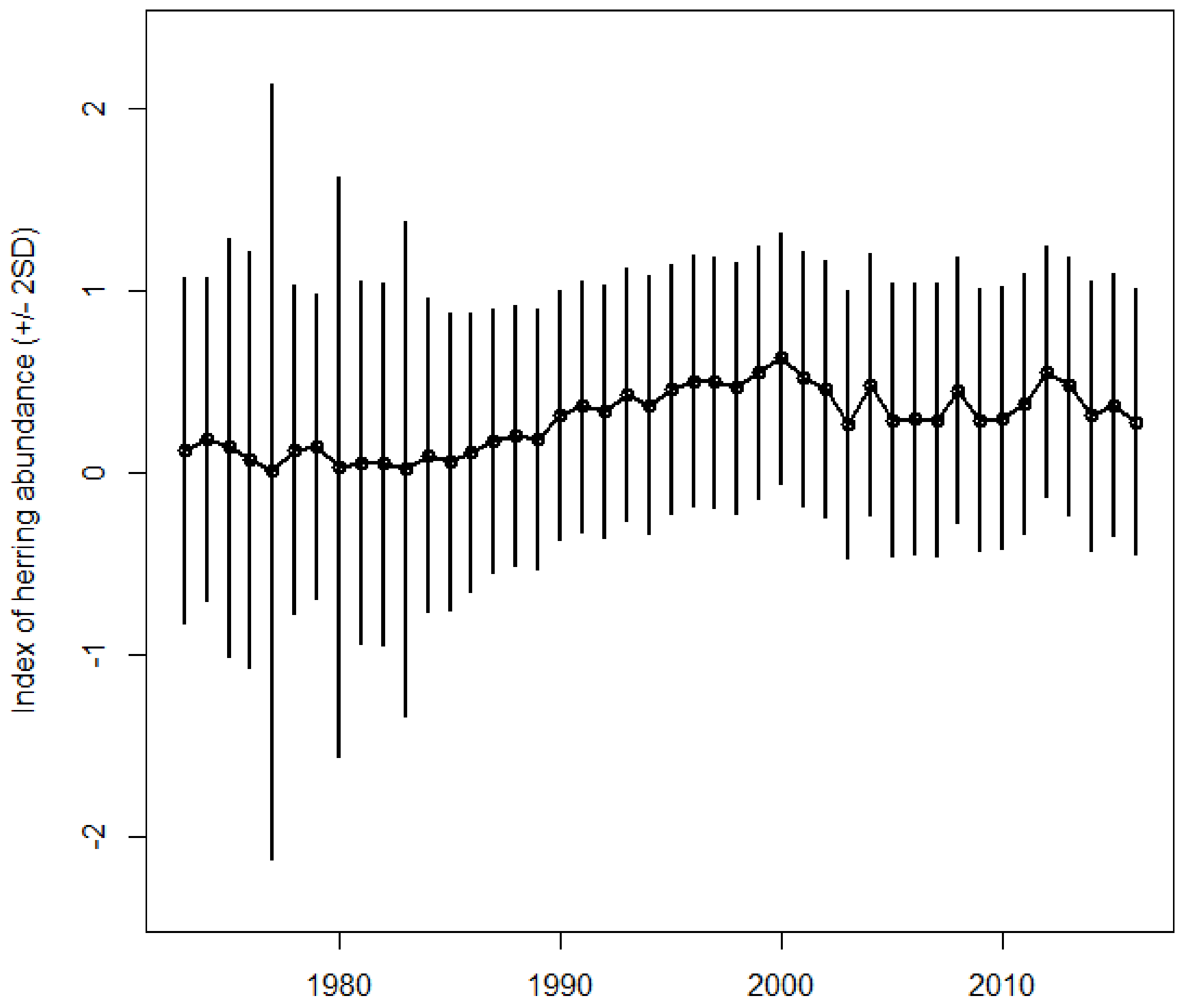


Figure B2- 10 Index of herring abundance derived from stomach contents data using data through 2014 (red) and with revisions to data during 2012-2014 and updated through 2016 (black)

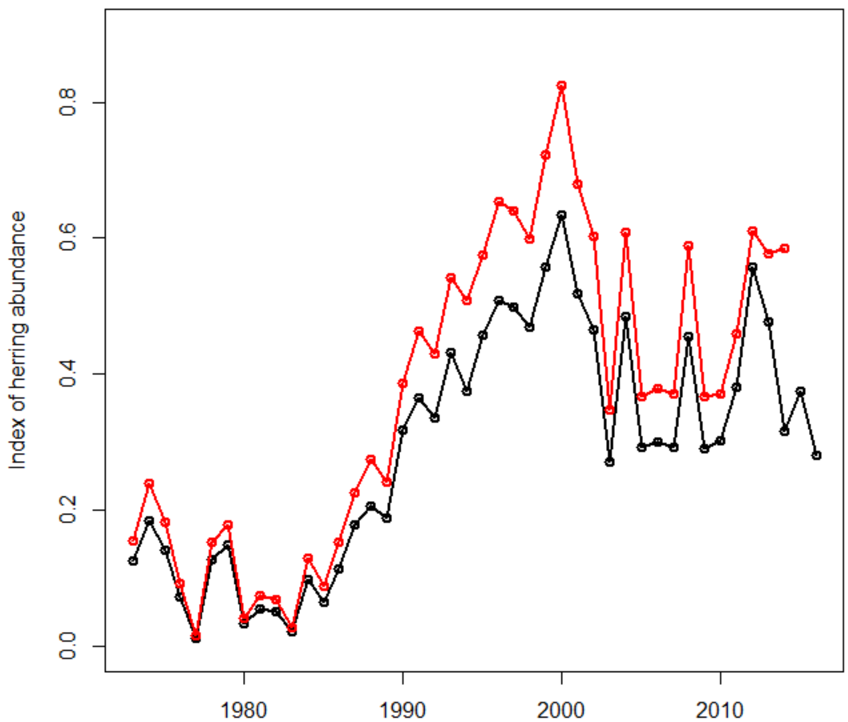
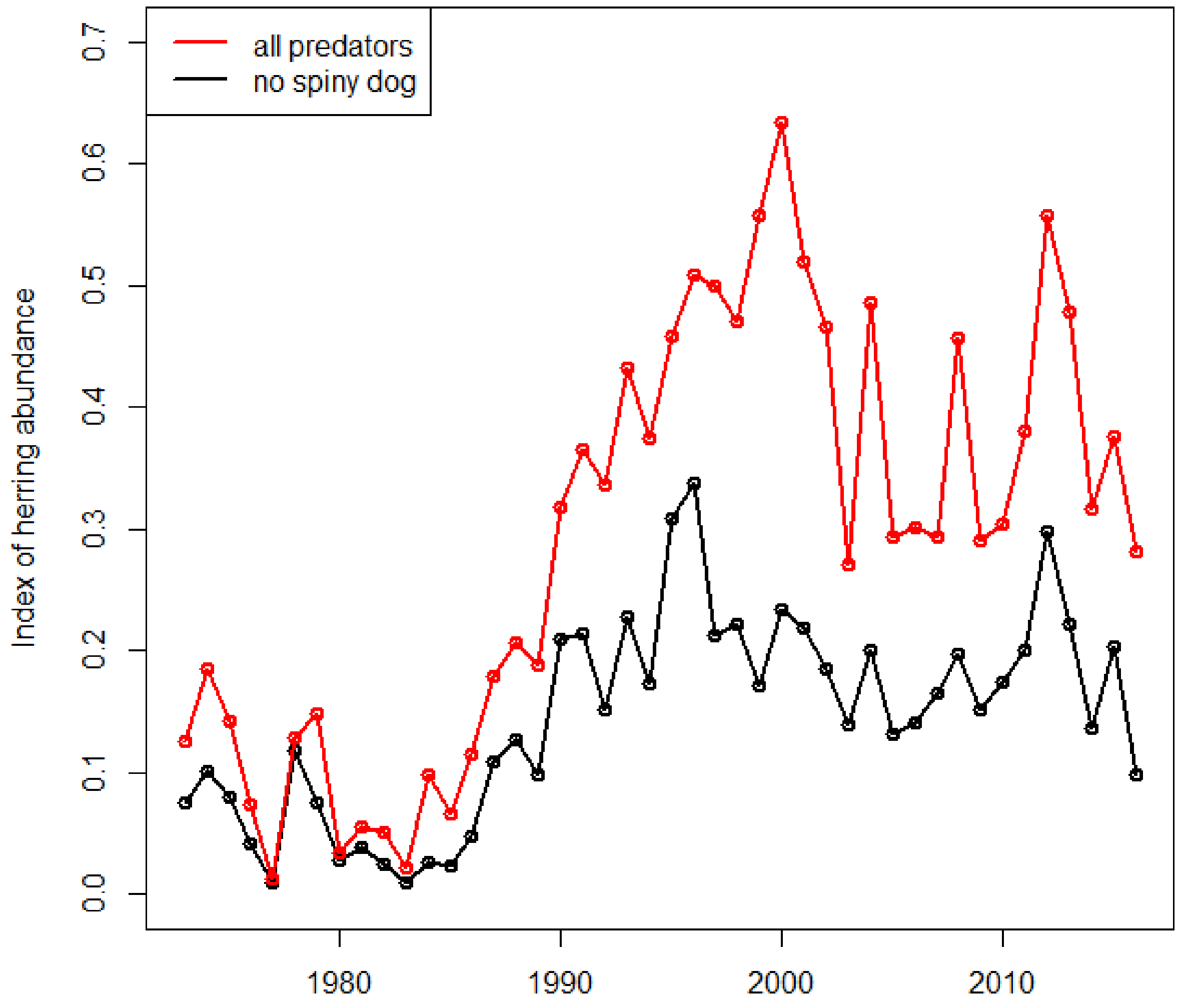
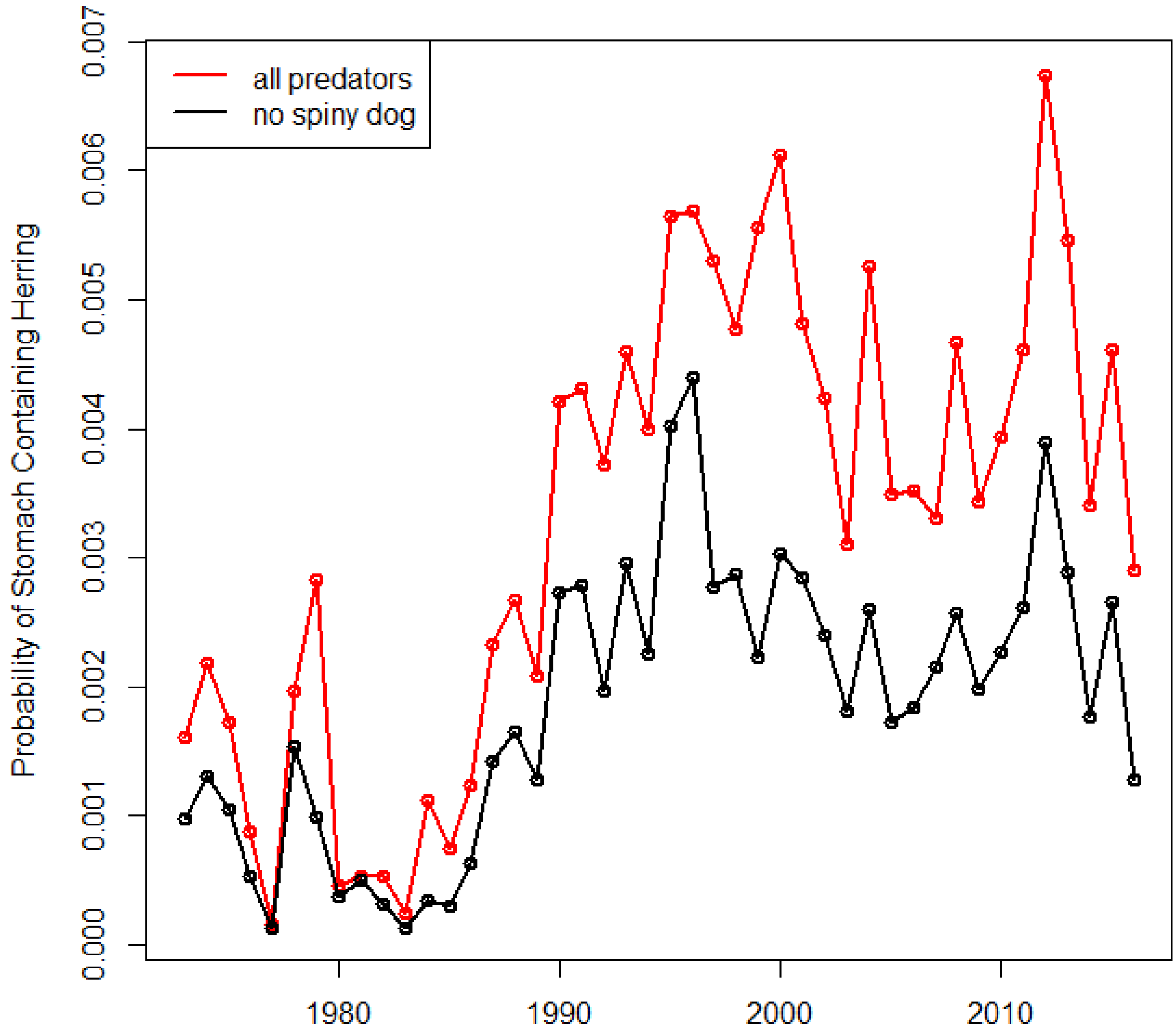


Figure B2- 11 Effect of removing spiny dogfish observations from the index of herring abundance derived from stomach contents data, and the effect on each element of the hurdle model used to create the index




all predators
no spiny dog
Probability of Stomach Containing Herring
0.000
0.001
0.002
0.003
0.004
0.005
0.006
0.007
1980
1990
2000
2010

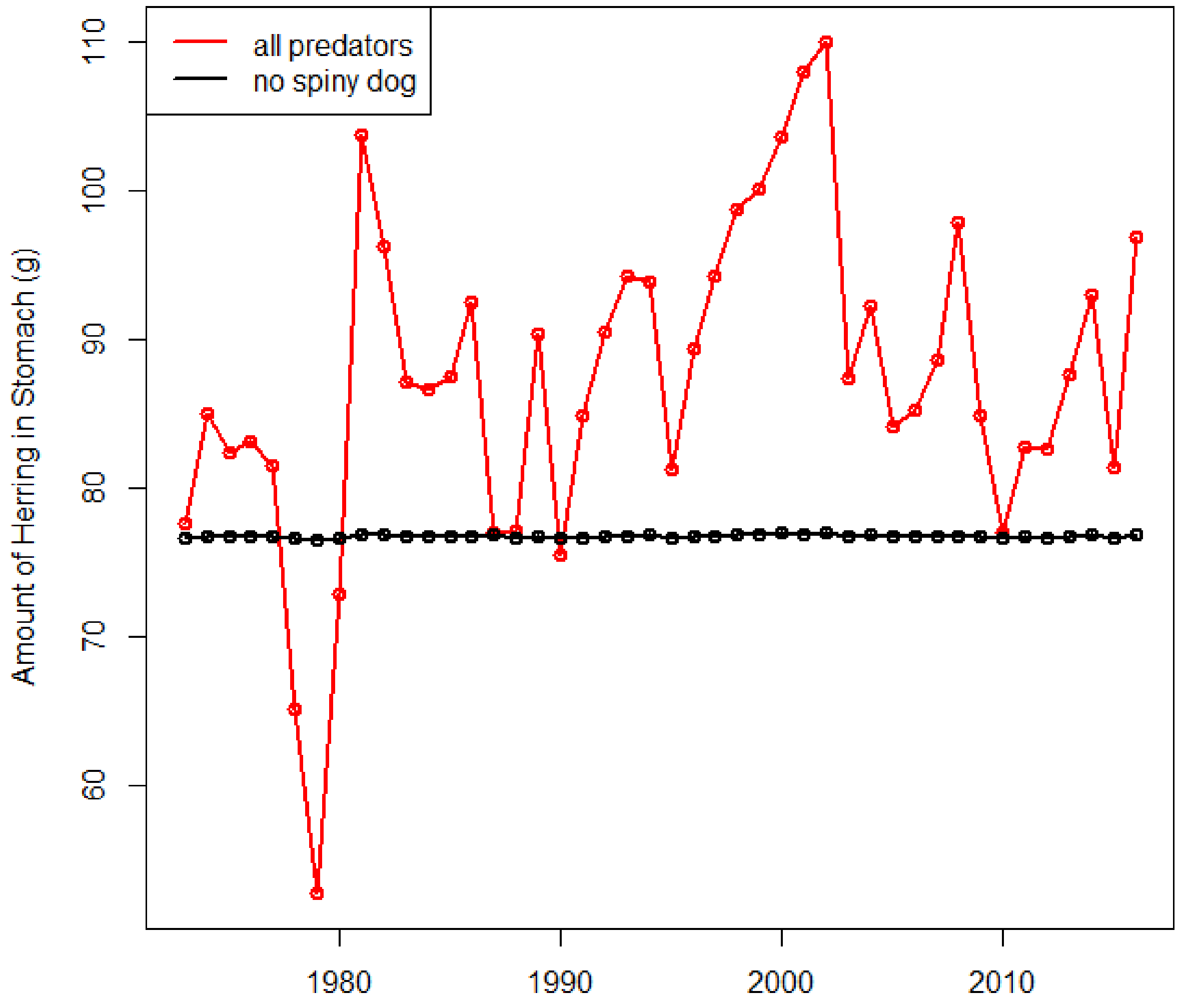

all predators
no spiny dog
Amount of Herring in Stomach (g)
110
100
90
80
70
60
1980
1990
2000
2010

Figure B2- 12 Relative retrospective pattern for the index of herring abundance derived from stomach contents data

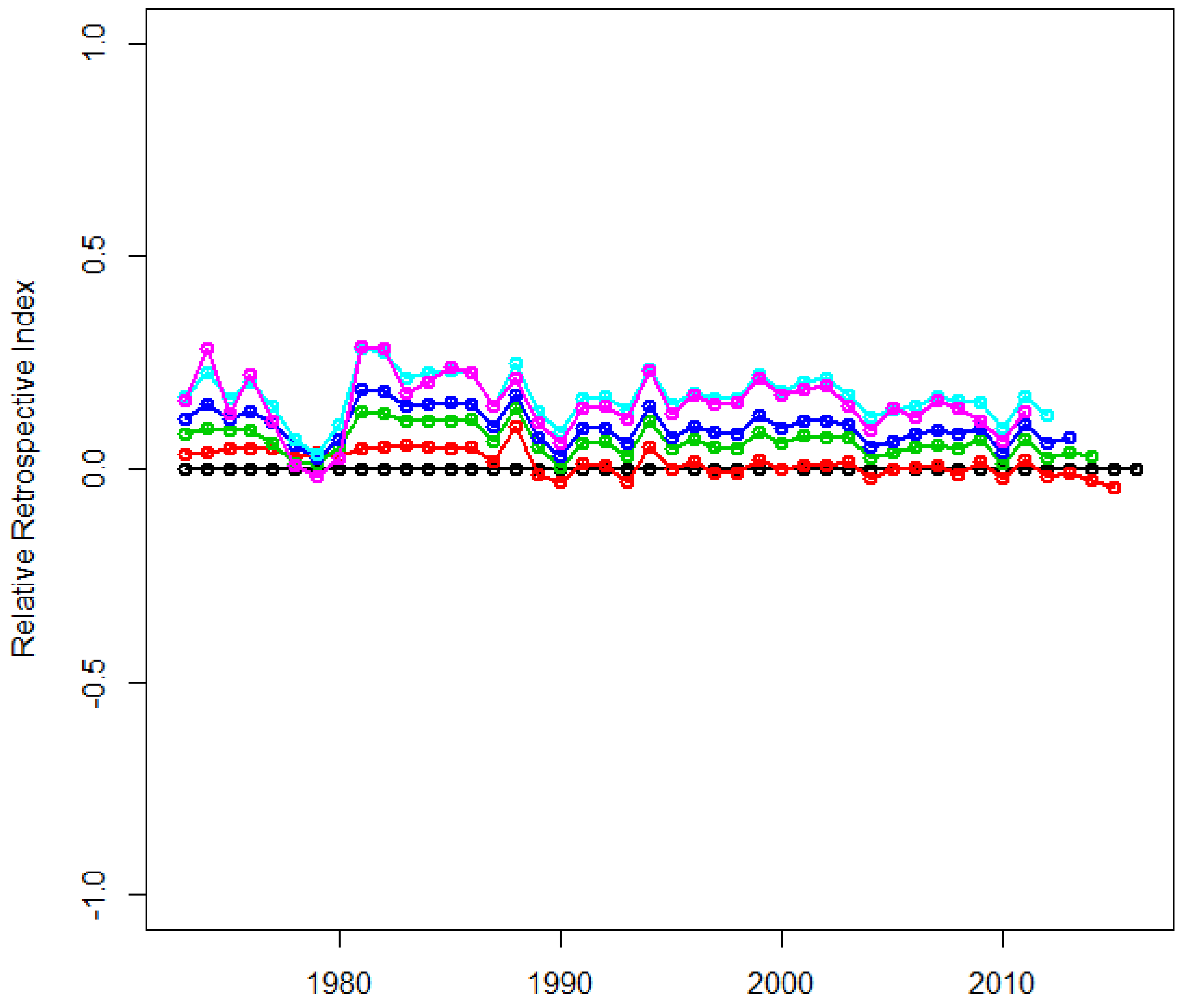


Figure B2- 13 Distribution of acoustic backscatter classified as Atlantic herring along the cruise track during the fall bottom trawl survey in 2016 on the HB Bigelow. Symbol size and color is related to areal acoustic backscatter, s_A (m^2 nmi^{-2}).

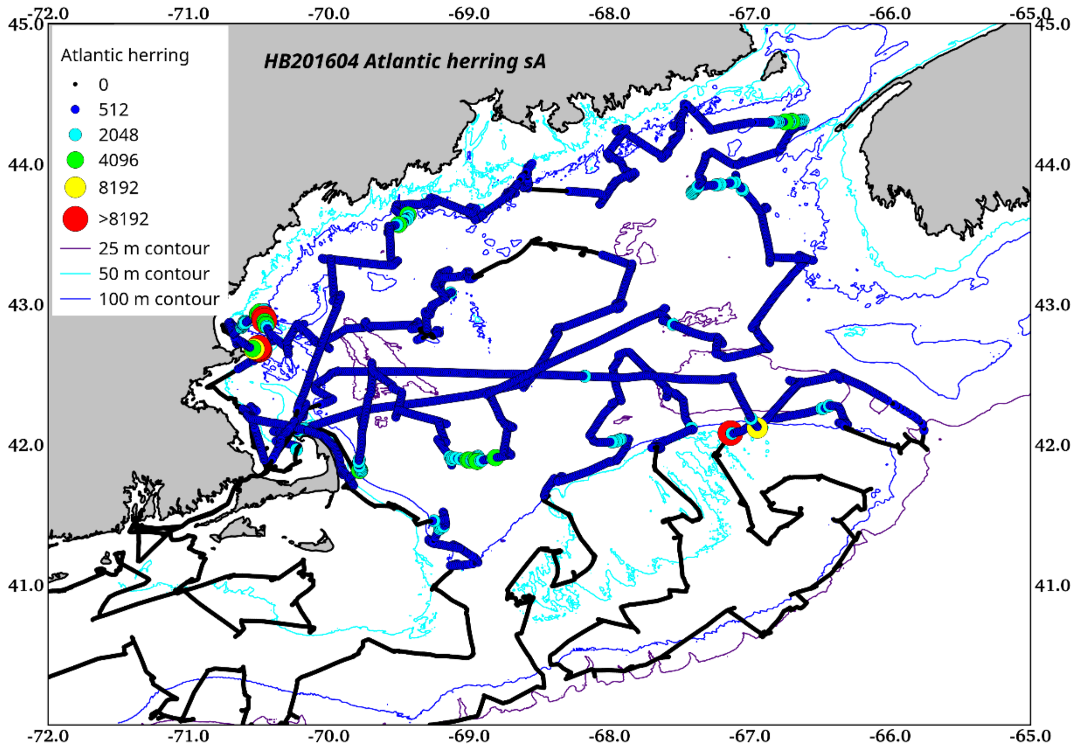


Figure B2- 14 Echogram of acoustic backscatter in the upper water column where the species composition can not be verified because the bottom trawl did not adequately sample these aggregations.

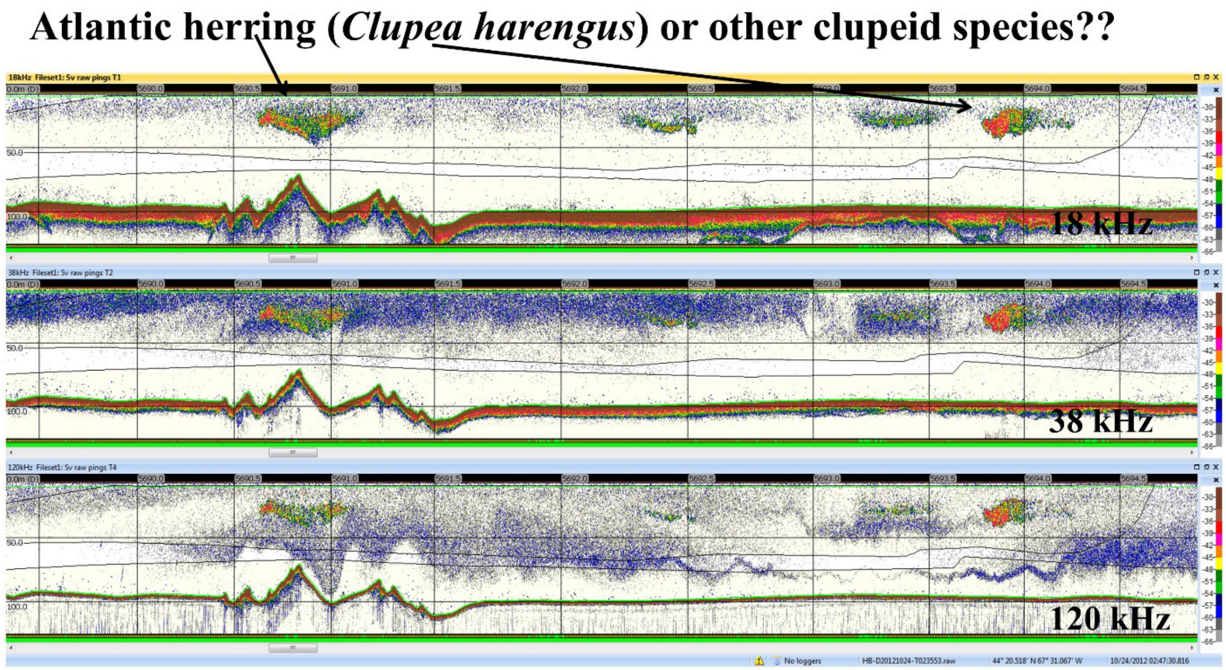


Figure B2- 15 The finfish strata are shown in green. The “acoustic area” encompasses all strata where used to aggregate acoustic backscatter classified as Atlantic herring throughout the years from 1998-2017.

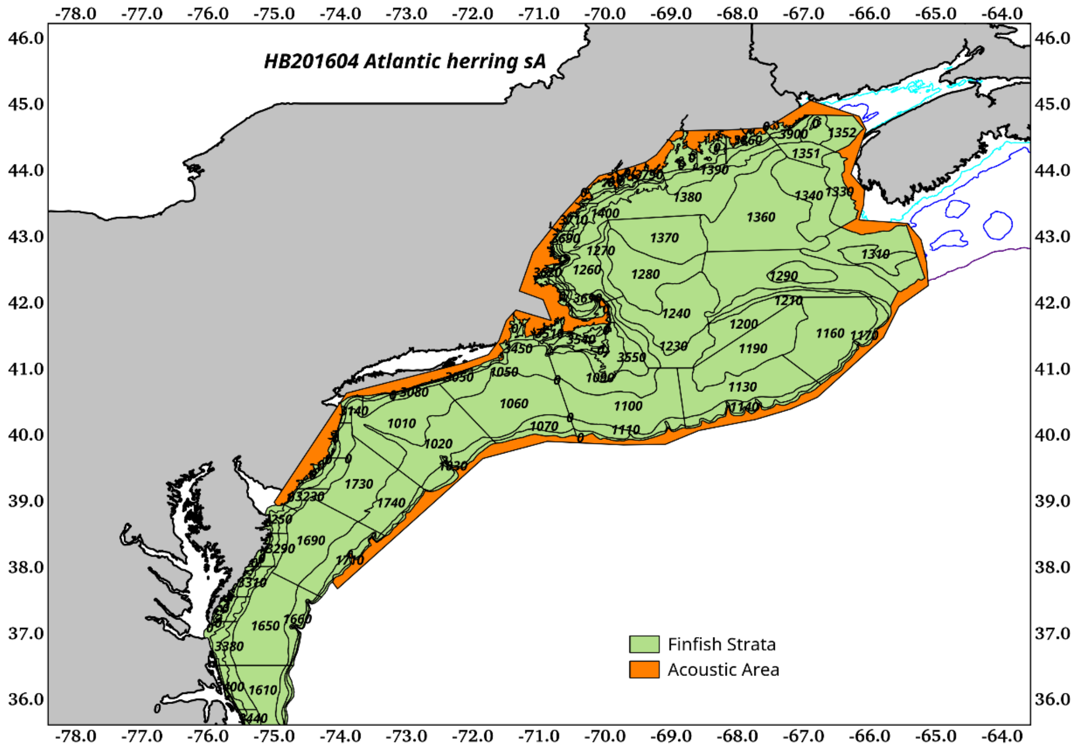


Figure B2- 16 Acoustic backscatter classified as Atlantic herring during the 2016 fall bottom trawl survey on the HB Bigelow overlaid on the finfish strata.

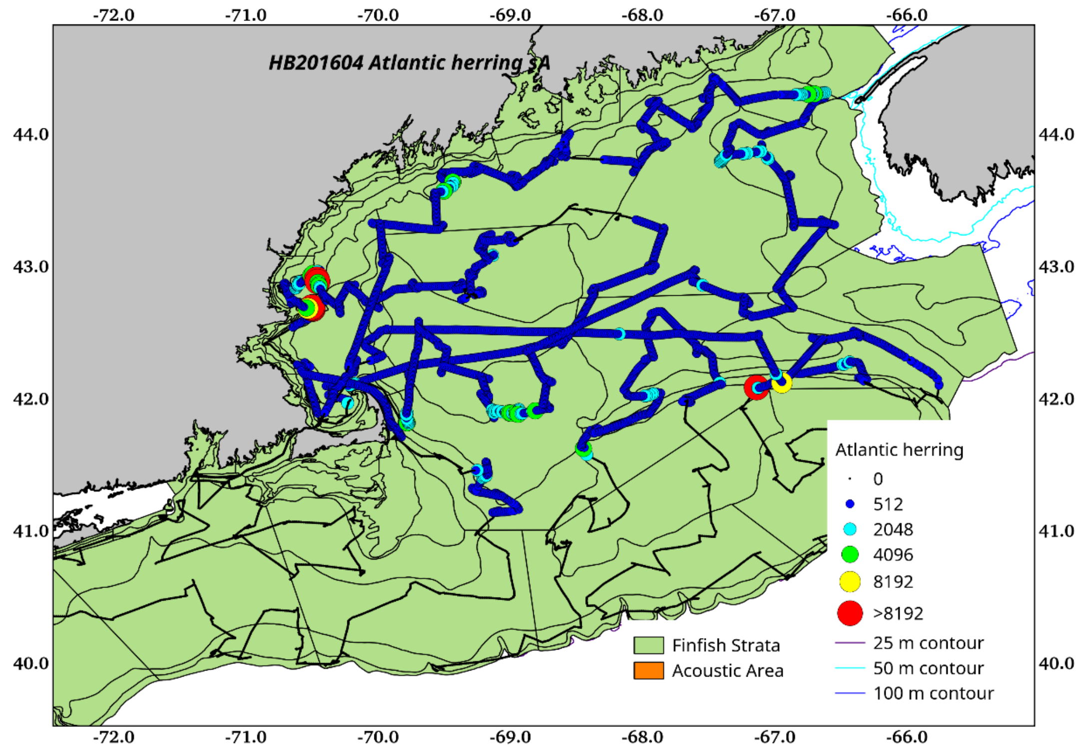
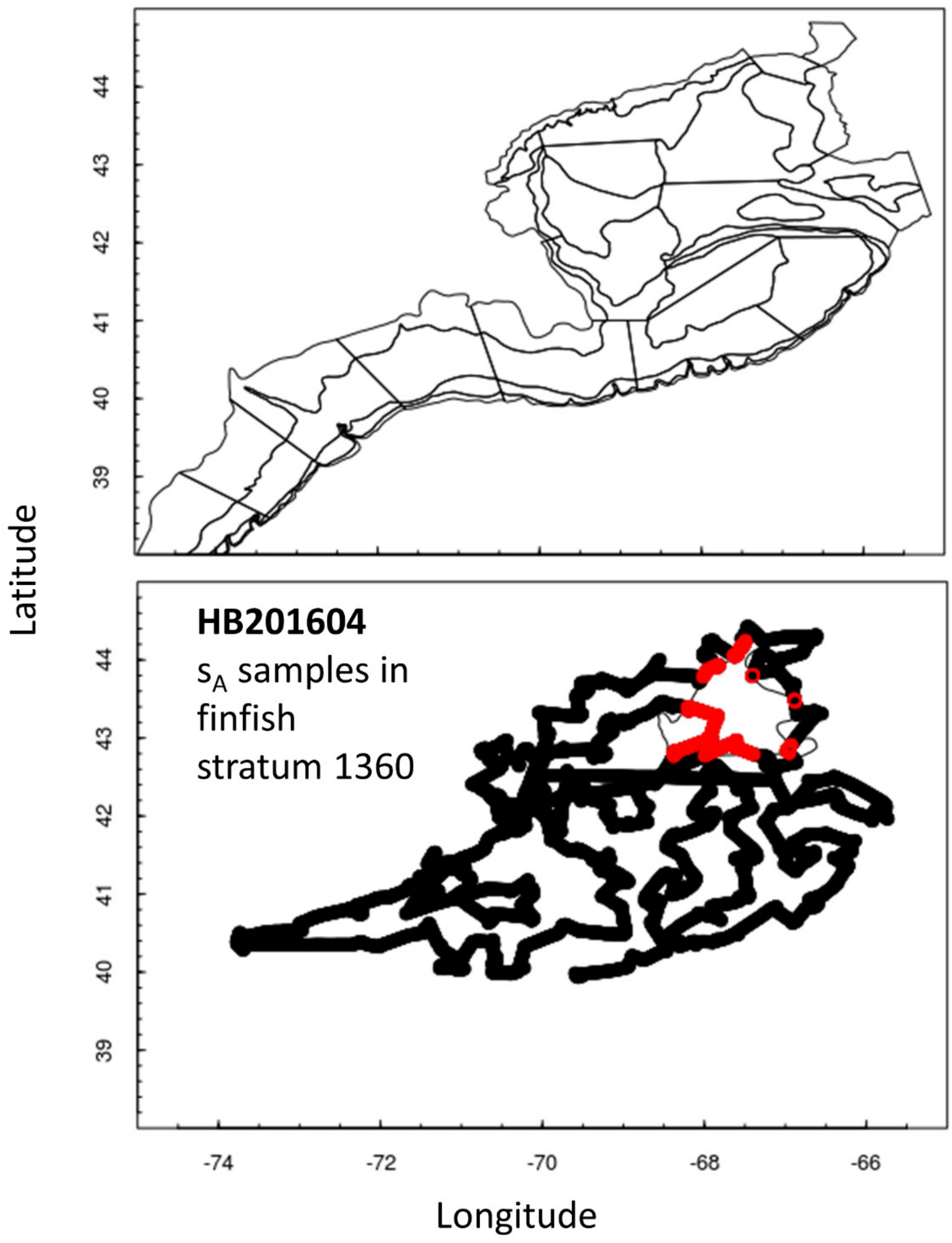


Figure B2- 17 The finfish strata included in the analyses (upper panel) and the entire set of s_A samples (black symbols) and s_A samples within stratum 1360 (red symbols) during the 2016 bottom trawl survey on the HB Bigelow



TOR B3: *Estimate consumption of herring, at various life stages. Characterize the uncertainty of the consumption estimates. Address whether herring distribution has been affected by environmental changes.*

Estimate consumption of herring, at various life stages. Characterize the uncertainty of the consumption estimates.

Summary

A time series of Atlantic herring consumption was estimated with an evacuation rate model for 12 fish predators of the NE US continental shelf, 1968-2016. Annual removal of herring amounted to 10s to 100s of thousands of MT by these predators.

Herring prey length data indicated adult herring (200+ mm) were primarily targeted, but this may be a result of limited inshore sampling coupled with sporadic inter-annual prey length sampling. Relative to January 1 biomass of herring available in the environment, an annual natural mortality proxy was produced using estimates of herring consumption. The time series average natural mortality was 0.12, reflecting predation of primarily adult herring by these predators.

Introduction

Fish food habits data from NEFSC bottom trawl surveys were evaluated for 12 herring predators (Table B3- 1). From these data, diet composition of herring, per capita consumption, and the amount of herring removed by the 12 predators were calculated. Combined with abundance estimates of these predators, herring consumption was summed across all predators as total herring consumption.

Methods

Every predator that contained Atlantic herring (*Clupea harengus*, and unidentified clupeid remains) was identified. From that original list, a subset of the top 12 predators comprising 94% of the occurrences of all herring predation and were regularly encountered by the sampling survey were included for estimating total herring consumption. Minimum sizes for herring predation were derived from the NEFSC Food Habits Database for each predator (Table B3- 1). Diet data were not restricted by geographic area and were evaluated over the entire northeast U.S. shelf as one geographic unit to match the assessed herring stock structure (see above).

Estimates were calculated on a seasonal basis (two 6 month periods) for each predator and summed per annum. Although food habits data collections for these predators started quantitatively in 1973 (Order Gadiformes only) and extends to the present (through 2016), not all herring predators were sampled during the full extent of this sampling program. Stomach sampling for the non-Gadiformes considered here began in 1977 and extends through 2016. For more details on the food habits sampling protocols and approaches, see Link and Almeida (2000) and Smith and Link (2010). This sampling program was part of the NEFSC bottom trawl survey program; further details of the survey program can be found in Azarovitz (1981), NEFC (1981), and Reid et al. (1999).

Basic Diet Data

Mean amounts of herring eaten ($D_{i,t}$; as observed from diet sampling) for each predator (i) and temporal scheme (t, fall or spring; year) were weighted by the number of fish at length per tow and the total number of fish per tow as part of a two-stage cluster design (See Link and Almeida 2000; Latour et al. 2007). These means included empty stomachs, and units for these estimates are in grams (g).

Numbers of Stomachs

The adequacy of stomach sample sizes were assessed with trophic diversity curves by estimating the mean cumulative Shannon-Wiener diversity of stomach contents plotted as a function of stomach number. The order of stomachs sampled was randomized 100 times, and cumulative diversity curves were constructed for each species focusing on the early 1980s when stomach sampling effort was generally lowest for the entire time series. The criteria for asymptotic diversity was met when the slope of the three proceeding mean cumulative values was ≤ 0.1 which was similar to previous fish trophic studies (e.g. Koen Alonso et al. 2002; Belleggia et al. 2008; Braccini 2008). A minimum sample size approximately equal to 20 stomachs for each predator per year-season emerged as the general cutoff for these asymptotes. Annual estimates of diet compositions of herring were estimated for each predator and season. For all predators, mean amounts of herring consumed ($D_{i,t}$) were not averaged between years with zero stomachs containing herring.

Consumption Rates

To estimate per capita consumption, the gastric evacuation rate method was used (Eggers 1977; Elliott and Persson 1978). There are several approaches for estimating consumption, but this approach was chosen as it was not overly simplistic (as compared to % body weight; Bajkov 1935) or overly complex (as compared to highly parameterized bioenergetics models; Kitchell et al. 1977). Additionally, there has been extensive use of these models (Durbin et al. 1983; Ursin et al 1985; Pennington 1985; Overholtz et al. 1999, 2000; Tsou and Collie 2001a, 2001b; Link and Garrison 2002; Link et al. 2002; Overholtz and Link 2007). Units are in g year^{-1}.

Using the evacuation rate model to calculate consumption requires two variables and two parameters. The daily per capita consumption rate of herring, $C_{i,t}$ is calculated as:

$$C_{i,t} = 24 \cdot E_{i,t} \cdot D_{i,t} \quad ,$$

where 24 is the number of hours in a day. The evacuation rate $E_{i,t}$ is:

$$E_{i,t} = \alpha e^{\beta T_{i,t}} \quad ,$$

and is formulated such that estimates of mean herring eaten ($D_{i,t}$) and ambient temperature ($T_{i,t}$) as stratified mean bottom temperature associated with the presence of each predator from the NEFSC bottom trawl surveys (Taylor and Bascuñán 2000; Taylor et al. 2005) are the only data required. The parameters α and β were set as 0.002 and 0.115 for the elasmobranch predators respectively and 0.004 and 0.115 for the teleost predators respectively (Tsou and Collie 2001a, 2001b, Overholtz et al. 1999, 2000).

To evaluate the performance of the evacuation rate method for calculating consumption, a simple sensitivity analysis had been previously executed (NEFSC 2007). The ranges of α and β within those reported for the literature do not appreciably impact consumption estimates (< half an order of magnitude), nor do ranges of *T* which were well within observed values (<< quarter an order of magnitude). An order of magnitude change in the amount of food eaten linearly results in an order of magnitude change in per capita consumption. Variance about any particular species of predator stomach contents has a CV of ~50%. Estimates of abundance, and changes in estimates thereof, are likely going to dominate the scaling of total consumption by a broader range of magnitudes than the parameters and variables requisite for an evacuation method of estimating consumption.

Fish Predator Abundance Estimation

The scaling of total consumption requires information on predator population abundance of sizes actively preying on herring (Table B3- 1). Where age information was available, minimum size was converted to age using the average age at length from Table B3- 1. Abundance estimates were either from assessment models or swept area abundance for each predator (Table B3- 2). Predators with a short time series (data not available 1968 -2016) were extrapolated back using survey indices and their relationship with abundance estimates (Atlantic cod, pollock, summer flounder, and goosefish) or landings using the relationship between landings and abundance (bluefish). Species estimated using swept area abundance (winter and thorny skate, silver and red hake, and sea raven) used an assumed q= 1.0. For Georges Bank cod and goosefish, the most recent assessment model (cod) was not accepted (NEFSC 2015) or ageing method invalidated (goosefish; Richards 2016); thus, abundance data from previously accepted assessments were used and the time series expanded based on the relationship with survey indices.

Scaling Consumption

Following the estimation of consumption rates for each predator and temporal (*t*) scheme they were scaled up to a seasonal estimate ($C'_{i,t}$) by multiplying the number of days in each half year:

$$C'_{i,t} = C_{i,t} \cdot 182.5 \quad .$$

These were then summed to provide an annual estimate, $C'_{i,\,year}$:

$$C'_{i,year} = C_{i,fall} + C_{i,spring} \quad .$$

and were then scaled by the annual abundance to estimate a total annual amount of herring removed by predator, $C_{i,\,year}$:

$$C_{i,year} = C'_{i,year} \cdot N_{i,year} \quad .$$

To complement the herring assessment time series prior to 1973, 5-yr averages of annual per capita consumption of herring ($C'_{i,\,year}$) for the gadiform predators (1973-1977) and non-gadiform predators (1977-1981) were estimated and scaled for each predator by the available abundance data from 1968-1976. The final herring consumption time series was 1968-2016. The total amount of herring removed ($C_{i,\,year}$) were then summed across all *i* predators to estimate a total amount of herring removed, C_{year}:

$$C_{year} = \sum_i C_{i,year} \quad .$$

The total consumption of herring per predator and total amount of herring removed by all predators are presented as thousands of metric tons year^{-1}.

Prey Lengths of Herring

Prey length data were available for herring consumed by the 12 fish predators considered here. In total, 2,916 length records were collected from 1973-2016. Not all observed herring prey had length data available due to digestion or other sampling constraints; thus, sampling was sporadic year to year. The data were aggregated by decade and kernel density plots produced for each season.

Results and Conclusions

Total consumption of herring by fish predators was variable throughout 1968-2016 with the amount of herring removed equal to 32 MT year^{-1} (minimum) and 390,233 MT year^{-1} (maximum; Figure B3- 1). Years with lesser total amounts of herring predation were earlier in the time series (1968-1987; averaging 61,924 MT year^{-1} compared to later in the time series (1987-2016; averaging 137,051 MT year^{-1}).

Prey length data revealed much of the predation from fishes collected on the bottom trawl survey center on herring around 200 mm or greater for the fall and spring by decade (Figure B3-2). We suspect some of this is due to the bottom trawl survey design focusing on offshore waters and sporadic sampling of prey-lengths per year. It is believed similar or even greater amounts of predation on juvenile herring is likely occurring on this shelf primarily inshore, and in addition to fish predators, by other predators such as birds or marine mammals.

As a proxy for natural mortality due to predation, the proportion of total herring consumption to January 1 biomass of herring from the most recent herring benchmark assessment (NEFSC 2012) was estimated (Figure B3- 3). Here, predation by the 12 predators accounted for approximate proportions of 0.0002 (minimum) and 0.64 (maximum) of the population from 1968-2011. The time series mean of this proxy equaled 0.12. Considering that these estimates largely reflect predation on adult herring, additional work assessing consumption of herring less than 200 mm is warranted, particularly for the inshore waters of this shelf.

Address whether herring distribution has been affected by environmental changes.

Herring distribution at the shelfwide scale has been fairly stable from the 1970's to the present (based on observations from the NEFSC bottom trawl survey). This is in contrast to many New England species, which show significant along-shelf (northeastward) trends in their centers of distribution (see https://www.nefsc.noaa.gov/ecosys/current-conditions/species-dist.html). However, there is evidence that herring are found in deeper survey strata in recent years.

We compared NEFSC trawl survey information to determine whether herring distribution has changed. Comparisons of spring and fall kernel density maps from the 1970's (blue) and the most recent years (red, 2014-2017) shows no substantial change in herring distribution (Figure B3- 4). Further, a time series of the mean along shelf distance from both spring and fall surveys shows no trend over time, indicating that the center of the herring population has remained the same (Figure B3- 5). However, there is a significant long term trend in the mean depth of stations where herring are caught on the survey (Figure B3- 5), which may reflect less herring biomass over shallower Georges Bank and more over deeper Gulf of Maine now than in the past (supported by the kernel density maps).

Atlantic herring's overall climate vulnerability ranking was low in a recent assessment applied to many Northeast U.S. shelf species (Hare et al. 2016). Climate exposure of all Northeast U.S. species including herring was considered high, but Atlantic herring had low biological sensitivity. While the assessment ranked Atlantic herring as having a high potential for distribution shifts due to their low habitat specialization, highly mobile adult stage, and long larval duration with potentially broad dispersal, observations from the NEFSC surveys indicate that a shift has not yet happened.

Table B3- 1 Top 12 predators of Atlantic herring (*Clupea harengus* and unidentified clupeid remains) along with minimum sizes for herring predation from the NEFSC Food Habits Database and average age (where available).

Common Name	Scientific Name	Minimum Size (cm)	Avg. Age (years)
Spiny dogfish	*Squalus acanthias*	29	
Winter skate	*Leucoraja ocellata*	39	
Thorny skate	*Amblyraja radiata*	41	
Silver hake	*Merluccius bilinearis*	13	0.8
Atlantic cod	*Gadus morhua*	16	1.1
Pollock	*Pollachius virens*	19	1.4
White hake	*Urophycis tenuis*	21	0.4
Red hake	*Urophycis chuss*	24	1.3
Summer flounder	*Paralichthys dentatus*	23	0.9
Bluefish	*Pomatomus saltatrix*	17	0.0
Sea raven	*Hemitripterus americanus*	13	
Goosefish	*Lophius americanus*	12	1.2

Table B3- 2 Summary of methods used for determining predator abundances.

Common Name	Method
Spiny dogfish	Model-based estimate
Winter skate	Swept area biomass, fall offshore
Thorny skate	Swept area biomass, fall offshore
Silver hake	Swept area biomass, fall offshore
Atlantic cod	ASAP model, two stocks combined, linear extrapolation
	GB data from previously accepted model used
Pollock	ASAP model and ln curve extrapolation
White hake	Model-based estimate with fall q 2012-13 (last benchmark)
Red hake	Swept area biomass, fall offshore
Summer flounder	ASAP model and ln curve extrapolation
Bluefish	ASAP model and linear extrapolation
Sea raven	Swept area biomass, fall offshore
Goosefish	SCALE model and linear extrapolation
	Ageing method invalidated in 2015, but data from previously accepted model used

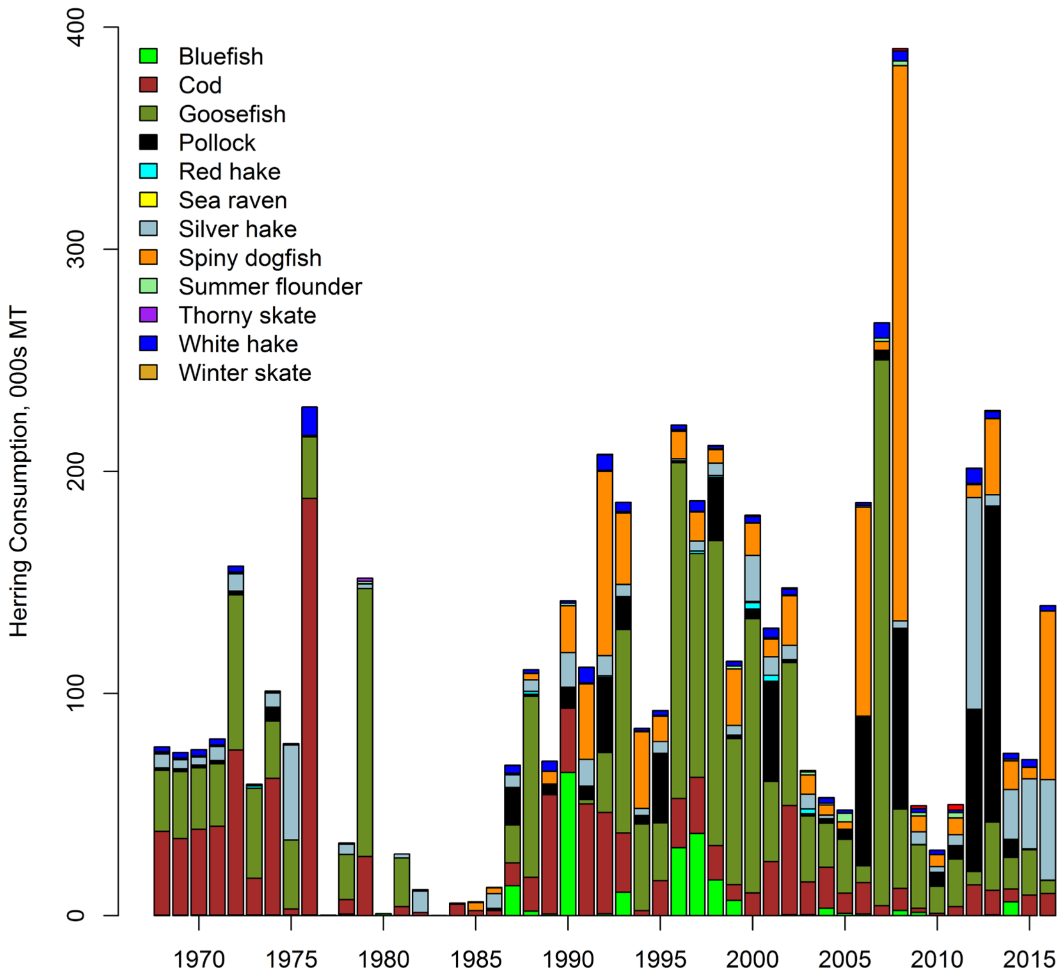


Figure B3- 1 Time series of herring consumption (000s MT) by 12 fish predators.

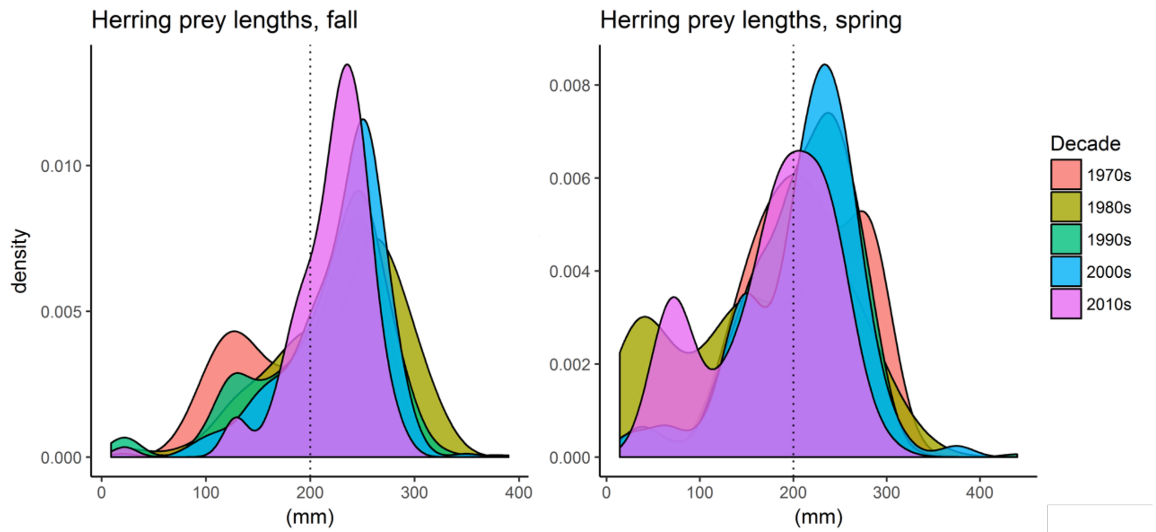


Figure B3- 2 Gaussian kernel density plots of herring prey lengths by decade for the spring and fall, 1973-2016.

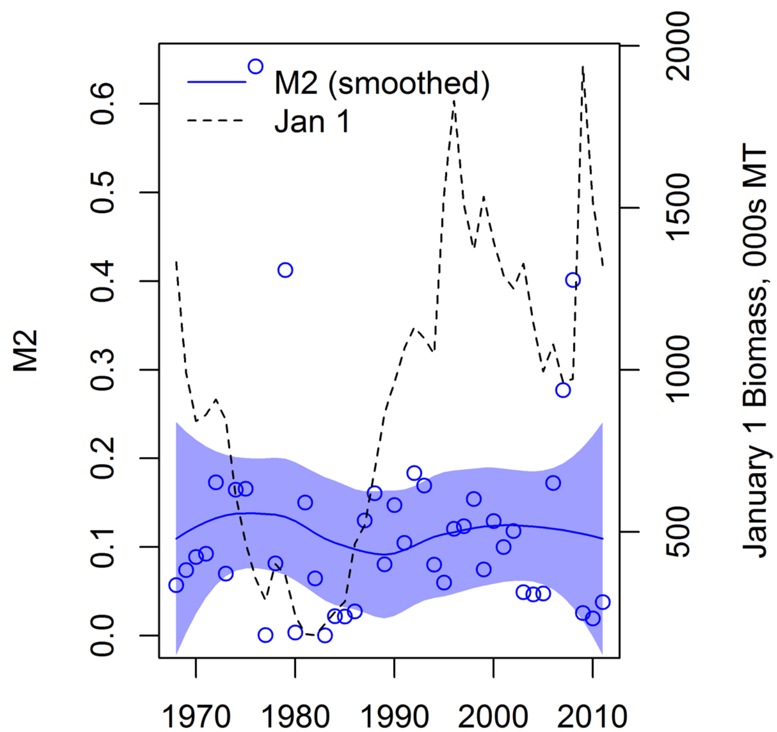


Figure B3- 3 Proxy estimate of natural mortality due to predation (M2) and January 1 biomass of herring, 1968-2011. M2 smoother is loess with span = 0.8 and 95% ci.

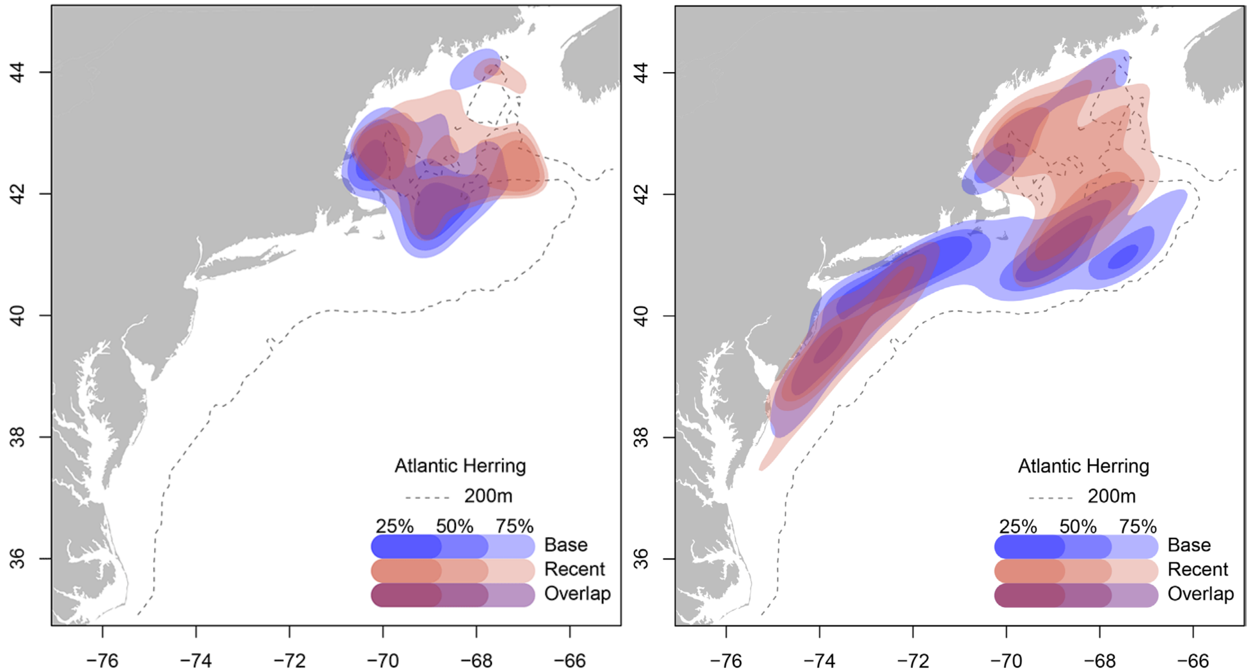
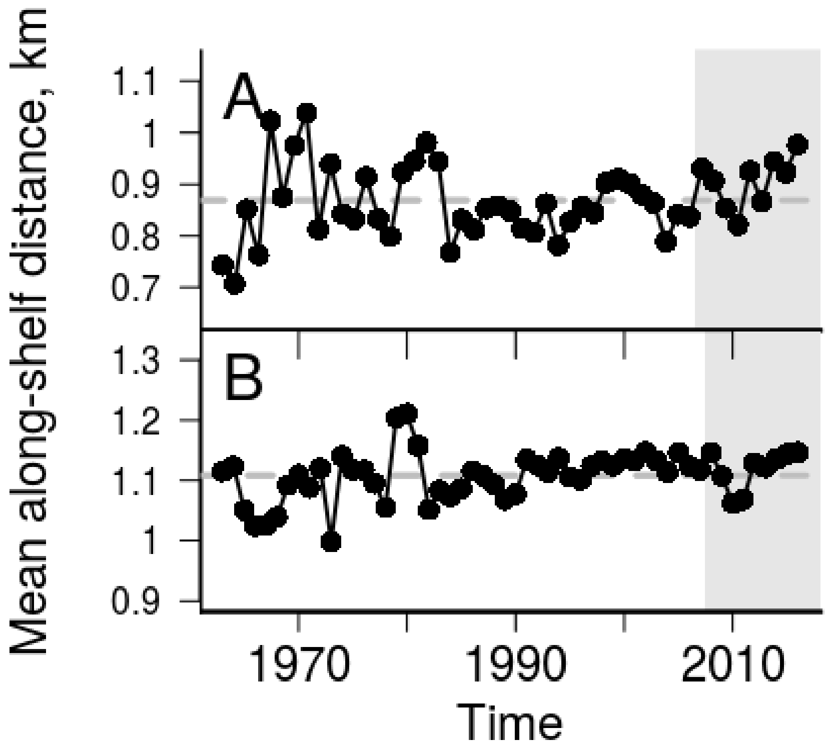


Figure B3- 4 Atlantic herring historical (1970s; blue) and current (2014-2017; red) distribution in the fall (left) and spring (right)



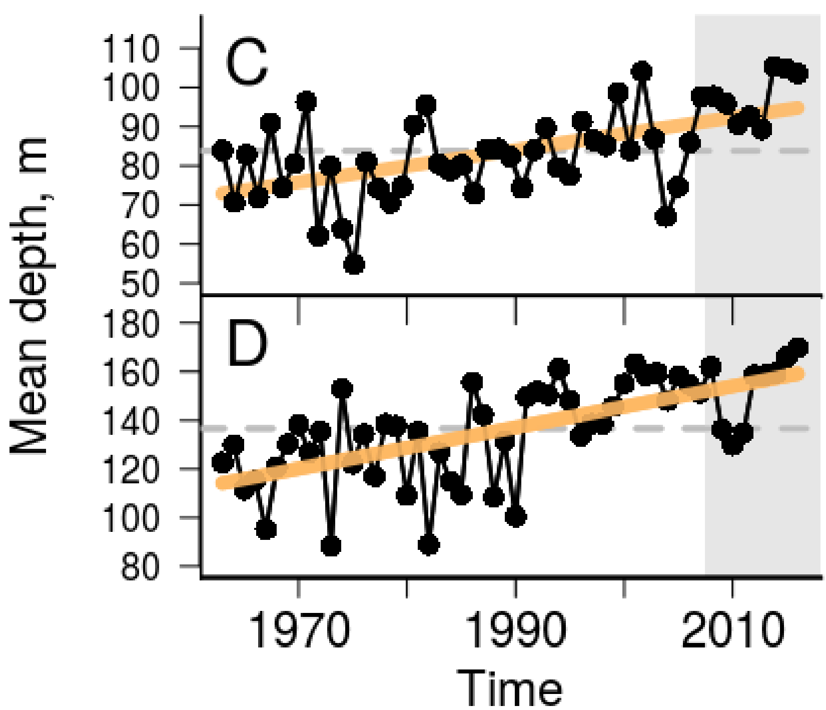


Figure B3- 5 Atlantic herring distribution trends (Along shelf distance in A. Spring, B. Fall, and mean depth in C. Spring, D. Fall)

TOR B4: *Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and estimate their uncertainty. Incorporate ecosystem information from TOR B3 into the assessment model, as appropriate. Include retrospective analyses (both historical and within-model) to allow a comparison with previous assessment results and projections, and to examine model fit.*

Update the 2015 ASAP model

The ASAP model formulation (Age Structured Assessment Program, Legault and Restrepo 1998) used in the 2012 (NEFSC 2012) and 2015 (Deroba 2015) stock assessments was updated using data through 2017. A brief description of this model formulation is provided here. The models used two fleets (mobile and fixed) as described above (see TOR B1). Indices of abundance included spring, fall, and summer NMFS bottom trawl surveys. The indices of abundance collected with the Biglow from 2009 to the terminal year of each assessment were calibrated to Albatross IV equivalent catches. Natural mortality was based on a combination of the Hoenig and Lorenzen methods, with the Hoenig method providing the scale of natural mortality and the Lorenzen method defining how natural mortality declined with age (Hoenig 1983; Lorenzen 1996; Brodziak et al. 2011). The natural mortality rates during 1996 to the terminal year of each assessment were increased by 50% from these base rates. In 2012, predatory consumption estimates of Atlantic herring were used in justifying time varying M (i.e., the 50% increase from base rates) that also resolved a retrospective pattern (NEFSC 2012). In the 2015 operational assessment, however, a retrospective pattern re-emerged and predatory consumption estimates no longer supported the time varying M (Deroba 2015). Reconsideration of time varying M is not permissible in an operational assessment, and so this feature was retained in the model, but an adjustment for the retrospective pattern was made for determining stock status and in short-term projections that informed catch specifications. In updating this model formulation through 2017, all model specifications (e.g., selectivity, data weighting, likelihood penalties) were identical to the previous assessments (NEFSC 2012; Deroba 2015), with the exception of a correction to input data. In the course of this assessment, the Working Group discovered that the age 8+ fall NMFS bottom trawl survey data were incorrectly calculated as an age 7+ value. This error was corrected.

Fits to catch, survey trends, and age compositions in the form of residual diagnostics were generally similar between the updated model and the 2015 assessment (results not shown). The updated model also exhibited a retrospective pattern, similar in severity to that of the 2015 assessment (Figure B4- 1).

A comparison of time series trends between the updated model and the 2015 assessment (with the plus group corrected in both models) showed a decrease in scale in the updated model, with the retrospectively adjusted SSB value from the 2015 assessment being similar to the estimate from the updated model (Figure B4- 2).

Review of models considered for this assessment

Three modeling platforms with different data inputs and different model structures were considered to varying degrees during this assessment. Building from previous assessments, the Working Group spent the most time evaluating the ASAP model, which ultimately was used for the base assessment (Legault and Restrepo 1998). A state-space assessment model (SAM; Nielsen and Berg 2014) was also developed. The Working Group was not as familiar with the SAM model as ASAP, and so SAM was ultimately used a point of comparison for ASAP fits. The details of the SAM configuration are in Appendix B2. An attempt was also made at model averaging the ASAP and SAM models (Appendix B3). Largely in response to research recommendations from previous assessments, a Stock Synthesis (SS) model was briefly reviewed (Methot and Wetzel 2013). The research recommendations that the SS model was primarily intended to address were the ability to fit to length composition data (ASAP and SAM cannot) and consideration of stock structure. So, a two area SS model was developed that fit to a broad range of data types, including length and conditional age-at-length composition data. The SS model reviewed by the Working Group during the Model Meeting, May2-4, 2018, had unresolved residual patterns in the composition data. Furthermore, in order to consider the estimation of movement among areas, the SS model assumed that 100% of Atlantic herring from each spatial area returned to their natal location to spawn. The Working Group felt that this assumption was unjustified and likely invalid. Given these concerns, the Working Group did not consider the SS model viable at this time. The Working Group agreed that the consideration of stock structure in the herring assessment may not be reasonable until more information is available on movement rates and the relative size of each sub-stock, which might come from morphometrics, tagging, or some other source. The Working Group recommended the continued consideration of using length composition data, whether through SS or some other model platform. The details of the SS model are in Appendix B4.

Base ASAP model development

The base ASAP model made structural changes to the previous assessment (e.g., M, selectivity), included new index time series, and re-evaluated some other relatively minor issues (e.g., weak likelihood penalties). The reasoning behind some of these modeling choices was described below. Some consequences of the changes to model structure and data inputs were documented in more detail in the *Sensitivities to the base ASAP model* section below.

The base model considered age 1 to an age 8 plus group and covered the time period 1965-2017. The age 8 plus group was based on the difficulties that ASAP had estimating the abundance of age 9 and older herring in the first year (i.e., 1965) and concerns about the reliability of age data for older ages in previous assessments (NEFSC 2012). The model was started in 1965 when catch data from all sources (i.e., US and Canadian weir) was first available.

Estimates of abundance at age in the first year (i.e., 1965) in previous assessments were imprecisely estimated and sometimes caused issues of model non-convergence (NEFSC 2012). To reduce imprecision and help with convergence, these estimates were previously given a relatively weak likelihood penalty for deviating from initial starting guesses. This penalty was removed in the base model, and initial abundances at age were estimated as deviations from an equilibrium age structure (Legault and Restrepo 1998). While these initial abundance estimates were still relatively imprecise (CVs ranging from 0.37-3.09), the imprecision was not considered problematic and the model consistently converged. A model with no likelihood penalty was also considered more parsimonious.

The base ASAP model used age- and time- invariant M = 0.35, which was a value based on the longevity methods of Hoenig (1983). The method assumed a maximum age equal to 14, which was the oldest age ever observed in commercial or survey gear catches and was consistent with maximum ages reported elsewhere (Collette and Klein-MacPhee 2002). Implied amounts of mortality based on the constant M were generally higher or similar to estimates of predatory consumption from stomach contents data (Figure B4- 3). The estimates of predatory consumption from stomach contents are likely underestimates, and so the Working Group was comfortable with implied amounts of mortality from the assessment being higher. The estimates of predatory consumption from stomach contents are also highly imprecise (although largely unquantified), and so the Working Group was satisfied with the general similarity of this comparison, and felt considering changes to M based on this comparison to be unjustified. This constant M was a departure from previous assessments that included age- and time-varying M (NEFSC 2012; Deroba 2015). The 50% increase in M beginning in 1996 was no longer justified given that that this increase in M no longer resolved retrospective patterns in the previous operational assessment (Deroba 2015) and was not needed to create general agreement between estimates of predatory consumption from stomach contents data and the amount of herring implied by the input M. Time-invariant M was also generally supported by a predation pressure index of M (Richards and Jacobson 2016; Appendix B5). The age-variant M based on a combination of the Hoenig and Lorenzen methods (Hoenig 1983; Lorenzen 1996; Brodziak et al 2011) provided a nearly identical fit to using a constant M (Neg. LL = 3773 for constant M and 3774 for age-variant), and so the Working Group agreed to use the more parsimonious constant M. A likelihood profile over time- and age-invariant M values found a minimum at 0.45 (Figure B4- 4).

For the mobile gear fishery, selectivity at age was freely estimated for ages 1-6, while selectivity at ages 7-8 was fixed at 1.0. Preliminary assessment fits were attempted that also estimated selectivity at ages 7 and 8, but estimates were at or near 1.0. The working group agreed that the mobile gear fishery, which is characterized by mostly large scale trawlers and purse seine operations, should have a flat-topped selectivity curve. Previous assessments (NEFSC 2012; Deroba 2015) also fixed the selectivity at ages 5 and 6 to 1.0. Estimating selectivity for these ages, however, improved model fit (Neg. LL improved by 7 units) and reduced some age composition residual patterns (Figure B4- 5).

The fixed gear fishery almost exclusively harvests age 2 fish, while other ages are caught in relatively small proportions (see TOR B1). Consequently, selectivity at age 2 was fixed at 1.0, while selectivity for all other ages was estimated. Previous assessments (NEFSC 2012; Deroba 2015) included a relatively weak likelihood penalty for deviations from initial guesses for each estimated selectivity at age parameter. These penalties were to help with precision and convergence, but were unnecessary for the base model here and so eliminated.

Selectivity at age on the NMFS spring survey during 1968-1984 was fixed and equaled 0.0 at ages 1 and 2, 0.5 at age 3, and 1.0 at ages 4-8. Selectivity-at-age on the NMFS fall survey during 1965-1984 was fixed and equaled 0.0 at ages 1-3, 0.5 at age 4, and 1.0 at ages 5-8. The selectivities for these surveys were fixed because no age composition data was available. The values input for the selectivities were justified in previous assessments by examining length compositions for each survey (see TOR B2). Sensitivity runs excluding these two surveys suggest that the base model is robust to their inclusion/exclusion and selectivity pattern, but that they provide some information for the estimation of initial abundance at age (Figure B4- 6), and so the Working Group agreed that they should be retained.

The NMFS spring survey during 1985-2017 (Albatross and Bigelow vessels) rarely caught any age 1 herring, while the fall frequently caught low proportions of age 1 herring (see TOR B2). In some years, however, a relatively large proportion of age 1 herring were caught.Previous assessments (NEFSC 2012) have found that assessment models would "chase" these signals about year class strength and estimate a relatively high recruitment in those years with high age 1 catches, which created retrospective patterns as more years of data about the given year class revealed a much weaker signal As in previous assessments, this Working Group agreed that the age-1 catches from these surveys were driven more by measurement uncertainty than by true measures of cohort strength. Consequently, age 1 catches from these surveys were discarded from the base ASAP model and selectivity at age 1 fixed to 0.0. For the NMFS spring survey during 1985-2008 (Albatross) and 2009-2017 (Bigelow), selectivity-at-age was freely estimated for ages 2-3 and was fixed and equaled 1.0 for ages 5-8. Age 4 selectivity was initially estimated, but kept hitting the bound of 1.0, which can cause convergence problems, and so this age was fixed at 1.0. For the NMFS fall survey during 1985-2008 (Albatross) and 2009-2017 (Bigelow), selectivity was logistic. Using age based selectivity in the spring resolved age composition residual patterns that were not present in the fall survey, making the more flexible age based alternative unnecessary in the fall (NEFSC 2012). As the NMFS summer survey used an average of the spring and fall NMFS survey age length keys, selectivity at age 1 was also assumed 0.0 in this survey. Otherwise, selectivity followed a logistic pattern.

No age composition data is available to inform selectivity estimation for the acoustic survey (as collected during the fall bottom trawl survey; see TOR B2). While all ages should theoretically be detected by the acoustic survey, some younger ages may be unavailable to the survey if they are not present at the time of sampling, which may be especially true during the fall when spawning occurs. A model with knife-edged selectivity at age 3, informed by the maturity data, provided a better fit than a model with full selection at all ages (Neg. LL better by 7 units). Consequently, the base model assumed knife-edged selectivity at age 3 for the acoustic index.

Input annual effective sample sizes (ESS) for the mobile and fixed gear fishery age composition data were initially set equal to the number of trips sampled for age in each year for each fishery, with a minimum of 5 and maximum of 150. In years for which no age samples were taken from the US fixed gear fishery and the age composition for the fleet relied solely on Canadian data, the ESS was set equal to 5 (the number of Canadian samples was unavailable; NEFSC 2012). Survey input annual ESS were initially set equal to the number of positive survey stations (i.e., stations that captured at least one herring) for each year and survey. All of these ESS were then iteratively reweighted as described for the multinomial distribution in Francis (2011).

The CVs on each survey data point were initially set equal to the CV estimated for a given survey in each year (see TOR B2). These CVs were then adjusted in an iterative fashion until the root mean square error (RMSE) of the standardized residuals for each survey was approximately within the 95% confidence intervals of the RMSE expected at the given sample size (i.e., number of years) for each survey (Figure B4- 7; Table B4- 1). The RMSE in this context was used as a measure of the consistency between the input precision of the survey values (i.e., CVs) and the uncertainty in the fits to a given survey index (i.e., variance of the standardized residuals). An RMSE equal to 1.0 suggests that the input CVs exactly match the uncertainty in the model fit. An RMSE greater than 1.0 suggests that the CVs need to be increased and the opposite for an RMSE less than 1.0. In this assessment, when the RMSE was outside of the 95% confidence intervals of the RMSE expected at the given sample size for a survey, each input CV for that survey was multiplied by the RMSE and the model was refit. For example, if the RMSE equaled 1.5, each CV was multiplied by 1.5 (increasing the CVs by 50%) and the model was refit. This process was repeated until the RMSE agreed with expectations, which usually only required one iteration. CVs were not allowed to exceed 0.95 during this process.

An annual CV of 0.1 was assumed in all years for the catch from both fisheries. Although ad hoc, this value admits some uncertainty in the catches and does not force an exact fit.

Unconstrained annual recruitment deviations were estimated without any penalty for deviating from some underlying mean stock-recruit relationship. Previous assessments have estimated the parameters of a Beverton-Holt stock-recruit relationship, and penalized recruitment for deviating from this underlying curve (NEFSC 2012; Deroba 2015). This practice was not used here because a likelihood profile of steepness revealed that the data provided nearly no information about the correct value of steepness, and the model's ability to estimate steepness seemed to rely solely on a relatively high degree of negative correlation between steepness and unexploited SSB (correlation = -0.96; Figure B4- 8).

Catchability for all surveys was freely estimated.

ASAP base model diagnostics and results

The ASAP base model fit to the fishery catches closely with the scale of residuals being relatively small (Figure B4- 9). The residuals for both fleets, however, were characterized by sequences of positive or negative residuals that were unlikely to have occurred by random chance (Figure B4- 9). The iteratively reweighted ESS for both fisheries led to estimated mean ages in each year that were generally within the 95% confidence intervals of the observed mean ages (Figure B4- 10). Exceptions to this occurred early in the time series for the mobile fleet and in more recent years for the fixed fleet, most often in years with relatively low ESS (Figure B4- 10). Fits to the mobile gear age composition exhibited only a few sequences of patterned residuals (e.g., age 4 from 1989-2002) and had no obvious year class effects (Figure B4- 11). Fits to the fixed gear age composition generally did not exhibit any obvious runs of residuals except for some relatively large residuals for ages $\geq$ 4 during 1986-1991 (Figure B4- 11). The fixed gear fishery caught more fish at these ages during those years than is typical, although still a relatively small amount (TOR B1). Thus, these relatively large residuals are likely not problematic. The mobile gear fishery selectivity increased in a near linear fashion to age-7, when full selection began (Figure B4- 12). The fixed gear fishery selectivity increased from near 0.0 at age 1 to full selection at age 2 and then quickly declined at older ages (Figure B4- 12). Average selectivity was generally less than average maturity at age, with herring maturing prior to full selection (Figure B4- 13).

The ASAP base model fit the survey trends relatively well. With few exceptions, residuals for fits to the survey trends did not exhibit long runs of residuals and residuals were generally centered on zero (Figure B4- 14). The estimated log scale survey indices also generally fell within the 95% confidence intervals of the log scale observations (Figure B4- 14). With rare exception, the iteratively reweighted ESS for the surveys led to estimated mean ages in each year that were generally within the 95% confidence intervals of the observed mean ages (Figure B4- 15). Fits to the survey age compositions also generally did not exhibit patterns, with exceptions being some age effects (e.g., age 8 in the shrimp survey and spring 1985-2008; Figure B4- 16).

The CVs on estimates of catchability (q) among all the surveys ranged from 32% to 55%. The q for the NMFS spring survey between the 1968-1984 period and the 1985-2008 period increased by a factor of 3.8 (0.0000017 to 0.0000064; Figure B4- 17). The q for the NMFS spring survey between the 1985-2008 period and the 2009-2017 period increased by a factor of 5.7 (0.0000064 to 0.000037; Figure B4- 17). The q for the NMFS fall survey between the 1965-1984 period and the 1985-2008 period increased by a factor of 29 (0.00000035 to 0.0000101; Figure B4- 17). The q for the NMFS fall survey between the 1985-2008 period and the 2009-2017 period increased by a factor of 3.43 (0.0000101 to 0.000035; Figure B4- 17). The NMFS shrimp survey q equaled 0.0000099 and the q for the acoustic index equaled 0.000024 (Figure B4- 17). Whether the catchability changes estimated by the base ASAP model in the NMFS spring and fall surveys between the 1985-2008 period and the 2009-2017 period (i.e., Albatross to Bigelow time periods) are aliasing some other factors is unclear. But, a retrospective analysis of the base ASAP model using 17 peels showed the scale of the relative differences in SSB increasing as fewer years of data were used, which includes a general increase in the scale of the relative differences beginning in ~2009 (Figure B4- 18). This result may suggest that some other model mis-specification exists and could be aliased by the modeled changes in catchability. The retrospective pattern is likely to worsen as additional years of data are added to the base ASAP model structure.

No two parameters of the ASAP base model had correlations greater than 0.9 or less than -0.9. Log unexploited SSB was estimated to be 13.2 with a CV of 25%. Time series estimates of SSB, F (averaged over ages 7 and 8), and recruitment were estimated relatively precisely, with the exception of recruitment in 2016 and 2017 that had CVs of 100% and 252%, respectively (Figure B4- 19).

The base ASAP model estimated SSB in 2017 to be 141,473 mt, with SSB ranging from a minimum of 53,084 mt (1982) to a maximum of 1,352,700 mt (1967) over the entire time series (Figure B4- 20; Figure B4- 23; Table B4- 2). The base ASAP model estimated total January 1 biomass in 2017 to be 239,470 mt, ranging from a minimum of 169,860 mt (1982) to a maximum of 2,035,800 mt (1967) over the entire time series (Figure B4- 20; Table B4- 2).

No common age is fully selected in both the mobile and fixed gear fishery. Consequently, the average F between ages 7 and 8 was used for reporting results related to fishing mortality (F_{7-8}), and this includes reference points (see TOR B5). These ages are fully selected by the mobile gear fishery, which has accounted for most of the landings in recent years (TOR B1). F_{7-8} in 2017 equaled 0.45. The all-time low of 0.13 occurred in 1965 (Figure B4- 23; Table B4- 2). The maximum F_{7-8} over the time series equaled 1.04 (1975).

Age-1 recruitment has been below average since 2013 (Figure B4- 21; Figure B4- 22; Table B4- 2). The all-time high of 1.4 billion fish occurred in 1971. The estimates in 2009 and 2012 are still estimated to be relatively strong cohorts, as in previous assessments. The all-time low of 1.7 million fish occurred in 2016, and the second lowest of 3.9 million fish occurred in 2017. Four of the six lowest recruitment estimates have occurred since 2013 (2013, 2015, 2016, 2017).

Markov chain Monte Carlo (MCMC) simulation was performed to obtain posterior distributions of SSB and F_{7-8} time series. An MCMC chain of length 6,000,000 was simulated with every 6000th value saved to create an MCMC chain with length 1,000 for defining the posterior densities. Traces and lag correlation plots for SSB and F_{7-8} in 1965 and 2017 had no obvious irregularities and chains are presumed to have converged (Figure B4- 24; Figure B4- 25). The posteriors for SSB and F_{7-8} in 1965 and 2017 are also provided as examples (Figure B4- 27). Time series plots of the 90% probability intervals are in Figure B4- 26while ASAP point estimates and the 80% probability intervals for SSB and F_{7-8} in 2017 are below:

Metric	**ASAP point estimate**	**80% probability interval**
2017 SSB (mt)	141,473	114,281 - 182,138
2017 F7-8	0.45	0.32 - 0.57

Internal retrospective patterns were characterized by using 5 “peels” rather than the 7 peels that is more common because of the relatively few numbers of years available for the NMFS spring and fall bottom trawl surveys during years when only the Bigelow vessel was used (2009-2017). Using 7 peels would require estimating q parameters for these surveys based on 2-3 years of data for the last 2 peels, and this has caused large imprecision and non-convergence in other assessments (Atlantic mackerel; NEFSC 2018). The internal relative retrospective pattern suggested consistent overestimation of SSB with Mohn’s Rho = 0.15, and underestimation of F_{7-}

$_{8}$ with Mohn's Rho = -0.11 (Figure B4- 28). The retrospective pattern for recruitment at age 1 was characterized by both positive and negative peels, with all of the positive peels greater than 4 (Figure B4- 28). The presence of the retrospective pattern is sensitive to the indices used in the ASAP base model (see sensitivity below with no acoustic index).

Estimates of SSB and fishing mortality among assessments from 1995, 2005, 2009, 2012, 2015, and the current ASAP base model were compared. Exact values from an assessment in 1998 were unavailable, but graphical representations of that assessment were similar in trend and scale as the 1995 assessment. The range of ages over which fishing mortality was calculated differed among assessments, as did selectivity, and therefore F values are not directly comparable, but were still useful for examining temporal trends. Estimates of SSB diverged among assessments more so at the beginning and end of the time series, with more similarity in intermediate years (~1970-1988; Figure B4- 29). Assessments in 1995 and 1998, however, estimated SSB to be about four times higher in the mid-1990s than assessments in 2005-2018 (Figure B4- 29). This contrast can be explained by a switch from a VPA model in 1995 and 1998 to an ASAP model for the other assessments. Estimates of SSB since about 2000 have generally decreased in each subsequent assessment (Figure B4- 29). Estimates of F from all the assessments were similar to that of SSB, except with differences occurring in the opposite direction; F generally increasing since 2000 in each subsequent assessment (Figure B4- 29). Changes in input data (e.g., acoustic index, time varying maturity) and model structure (e.g., M, selectivity) have occurred among assessments, and so the results for SSB and F are not entirely comparable.

ASAP base model sensitivity runs

In each of the sensitivity runs described below, all of the data and settings in the base model were the same as described above, except for the changes required for the given sensitivity run. Results focused on SSB because changes induced by the sensitivity runs were similar for F except in the opposite directions. Results also focused on retrospective patterns, and when appropriate, likelihood values.

ASAP base model sensitivity – M

Amending the ASAP base model to have age- and time-varying M as in previous assessments (NEFSC 2012; Deroba 2015) using the combination of Hoenig and Lorenzen methods (Hoenig 1983; Lorenzen 1996; Brodziak et al 2011) and a 50% increase in those values during 1996-2017, increased the scale of SSB and recruitment (Figure B4- 30). The retrospective pattern was similar between this sensitivity and the base model (Figure B4- 28; Figure B4- 31). The fit of the model was 5 likelihood units better than the base.

Eliminating the 50% increase in M during 1996-2017 and basing M only on the combination of Hoenig and Lorenzen methods, reduced the scale of SSB relative to the base (Figure B4- 32). The retrospective pattern was similar between this sensitivity and the base model (Figure B4- 28; Figure B4- 33). The fit of the model was similar (1likelihood unit worse) to the base.

ASAP base model sensitivity – calibrate Bigelow to Albatross

Calibrating the spring and fall NMFS bottom trawl surveys collected with the Bigelow vessel (2009-2017) to Albatross vessel equivalents using results from the paired tow experiments (Miller et al. 2010; NEFSC 2012) increased the scale of SSB relative to the base (Figure B4- 34). The retrospective pattern was also worse relative to the base, with Mohns's Rho equal to 0.34 for SSB and -0.24 for $F_{7\text{-}8}$ (Figure B4- 28Figure B4- 35). The base ASAP model estimated a 5.7 fold increase in catchability in the spring between the Bigelow and the Albatross, and a 3.4 fold increase in the fall (Figure B4- 36). These changes are 61% larger than the changes in catchability estimated by the paired tow experiments for the spring, and 73% larger for the fall (Figure B4- 36), and this explains the scale shift between the base model and using the paired tow calibrations.

ASAP base model sensitivity –time varying mobile fleet selectivity

The Working Group had concerns that the purse seine gear had a distinct selectivity from trawl gears, but these gears were combined in the mobile gear fleet (see TOR B1). To address this concern, time varying selectivity was added to the mobile gear fleet in the form of separate selectivity blocks for 1965-1990 and 1991-2017, where the break occurs in a year when purse seine catches decreased relative to the trawl gears and remained so. Selectivity at age in both blocks was freely estimated for ages 1-6, but fixed at 1.0 for ages 7-8. The model with 2 selectivity blocks improved model fit by 4 likelihood units over the base model, but also estimated 6 more parameters than the base (AIC would not support the 2 blocks). The model with 2 selectivity blocks also had qualitatively similar residuals as the base, nearly indistinguishable estimates of SSB, and the retrospective patterns were also similar (Figure B4- 28; Figure B4- 37; Figure B4- 38).

ASAP base model sensitivity–drop surveys ("leave one out")

The base ASAP model was re-run with each of the surveys removed from the model. The point estimates of SSB from each of the surveys remained within the 95% confidence intervals of the base run (Figure B4- 39). In more recent years, the base model was most sensitive to the exclusion of the acoustic index, with removing the acoustic index reducing the scale of SSB relative to the base (Figure B4- 39). Exclusion of the acoustic survey also eliminated the retrospective pattern, with peels for SSB and $F_{7\text{-}8}$ being both positive and negative (Figure B4- 40). The model was also less precise without the acoustic index, as evidenced by wider confidence intervals on stock status when compared to the base (Figure B4- 41Figure B6- 1). Stock status would also change in a model without the acoustic index, with overfishing occurring (Figure B4- 41).

ASAP base model sensitivity–fit with food habits index of abundance

The base model fit with the addition of the food habits index of abundance provided similar time series estimates of SSB (and F and recruitment) as the base model (Figure B4- 42). The fit to the food habits index was characterized by mostly negative residuals before ~1995 and mostly positive residuals after ~1995, although the estimated indices were generally within the 95% confidence intervals of the log scale observations (Figure B4- 43). The retrospective pattern was similar to the base (not shown).

ASAP base model sensitivity runs–explaining the scale difference from 2015

These sensitivities demonstrate that the shift in scale from the 2015 operational assessment (Deroba 2015) is a combination of: 1) basic data updates, with the retrospectively adjusted SSB value from 2015 being similar to that of the 2015 assessment updated through 2017, 2) treating the NMFS spring and fall bottom trawl surveys in years sampled by the Bigelow (2009-2017) as a separate index time series, 3) using a constant M as opposed to an age- and time-varying M, and 4) to a lesser extent than the other model changes, new data sources such as the acoustic index.

Table B4- 1 Root mean squared error table for the base Atlantic herring ASAP model, after iteratively reweighting

ot Mean Square Error computed from Standardized Residu

Component	# resids	RMSE
catch.fleet1	53	0.141
catch.fleet2	53	0.066
catch.tot	106	0.11
discard.fleet1	0	0
discard.fleet2	0	0
discard.tot	0	0
ind01	17	1.11
ind02	20	1.62
ind03	34	1.27
ind04	18	1.37
ind05	24	1.04
ind06	24	1.2
ind07	9	0.966
ind08	9	1.08
ind.total	155	1.25
N.year1	0	0
Fmult.year1	0	0
Fmult.devs.fleet1	0	0
Fmult.devs.fleet2	0	0
Fmult.devs.total	0	0
recruit.devs	0	0
fleet.sel.params	0	0
index.sel.params	0	0
q.year1	0	0
q.devs	0	0
SR.steepness	0	0
SR.scaler	0	0

Table B4- 2 Time series estimates of Atlantic herring from the base ASAP model

Year	SSB (mt)	Jan.1 Biomass (mt)	Age-1 Recruitment (000s)	$F_{7\text{-}8}$
1965	822530	1684170	5455740	0.13
1966	1158280	1908910	4582210	0.22
1967	1352730	2035820	9893020	0.36
1968	879319	1757780	4584770	0.64
1969	558945	1252230	5314820	0.66
1970	495252	990597	2726970	0.65
1971	309278	939626	14034800	0.97
1972	256642	941744	2487340	0.92
1973	421291	933513	2480520	0.89
1974	358470	694550	3080770	0.84
1975	234402	520342	1870600	1.04
1976	179914	390076	1889890	0.69
1977	107066	290770	5610910	0.70
1978	78307	348807	5312970	0.83
1979	72862	369009	760023	0.60
1980	86845	252059	3951690	0.99
1981	75400	183556	2123520	0.60
1982	53084	169857	1877800	0.64
1983	70978	183135	1371520	0.53
1984	64660	205018	4522510	0.89
1985	72605	212830	3327060	0.57
1986	103420	326508	3045410	0.46
1987	150558	389461	4230370	0.56
1988	253638	481774	6570600	0.61
1989	189046	653702	7616190	0.66
1990	182758	730272	8262200	0.42
1991	302782	834362	6996250	0.43
1992	452094	900893	3420850	0.40
1993	442046	851067	3432600	0.31
1994	389308	751971	4041110	0.26
1995	366549	848662	11221100	0.41
1996	433942	885059	5024520	0.44
1997	310950	861261	4848000	0.44
1998	447860	805422	3131950	0.38
1999	414034	840145	7791940	0.41
2000	394747	833338	2062680	0.37
2001	411161	796936	2067110	0.42
2002	332243	698200	4638660	0.38
2003	264895	684761	5505720	0.47
2004	240243	625565	2896810	0.48
2005	307228	560570	2036360	0.47
2006	260012	557285	4272270	0.58
2007	196392	503615	1229030	0.56
2008	207711	444931	2712310	0.58
2009	139353	577250	10579800	0.94
2010	121661	519530	2364220	0.72
2011	185013	500048	2110360	0.61
2012	243767	602132	6941730	0.60
2013	210106	580801	1370270	0.65
2014	330492	547060	1608170	0.51
2015	264982	471603	776348	0.47
2016	175698	347230	174758	0.47
2017	141473	239472	392286	0.45

Figure B4- 1 Retrospective pattern for the 2015 ASAP operational herring assessment updated using data through 2017

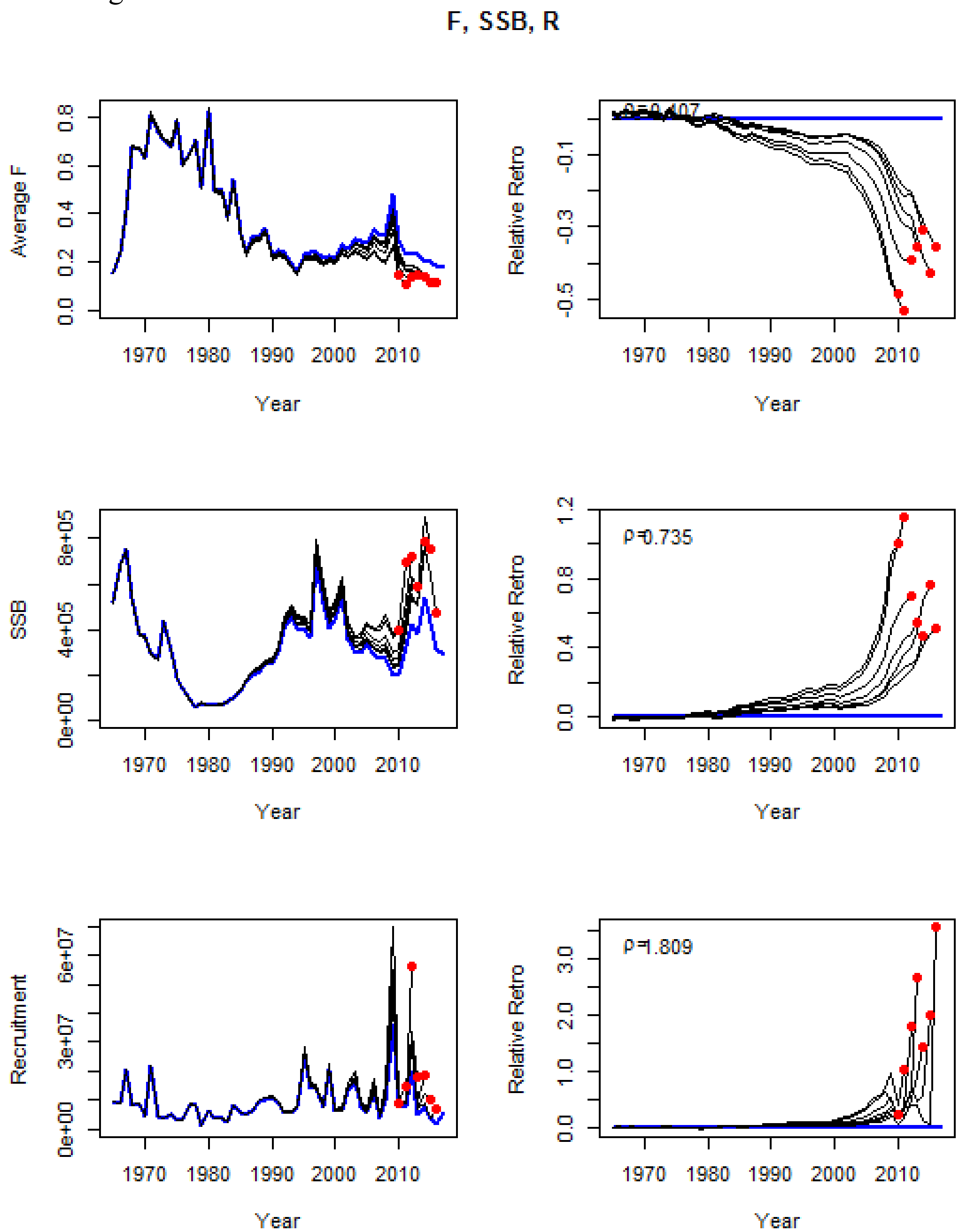


Figure B4- 2 Results of updating the 2015 ASAP operational herring assessment (2015FixFall) with data through 2017 (Run1_2017). The black diamond is the retrospectively adjusted SSB value from 2015.

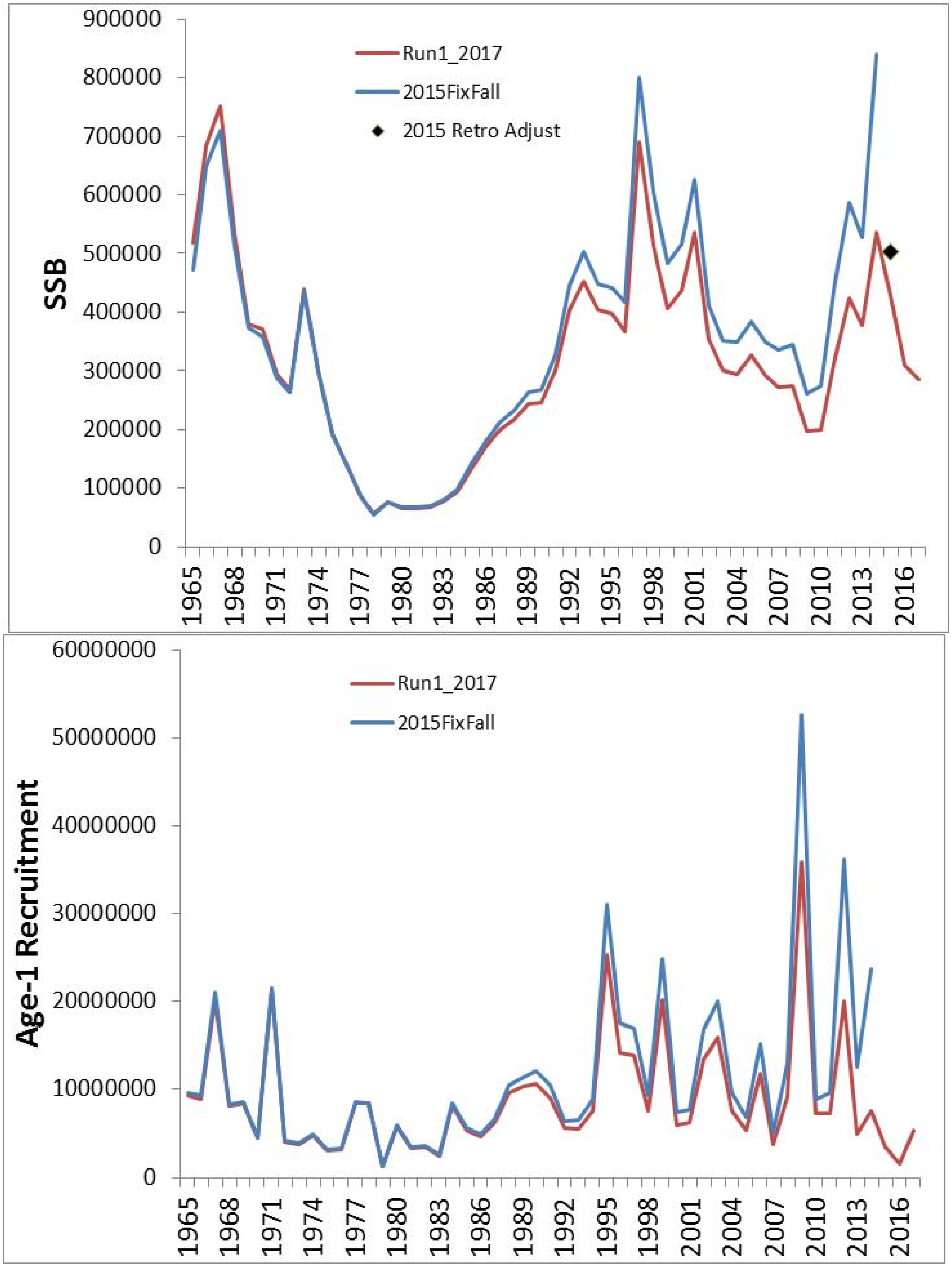


Figure B4- 3 Consumption of Atlantic herring by piscivorous predators as estimated using food habits data (Food Habits), and the amount of herring dying to due natural mortality in the ASAP base model (ASAP)

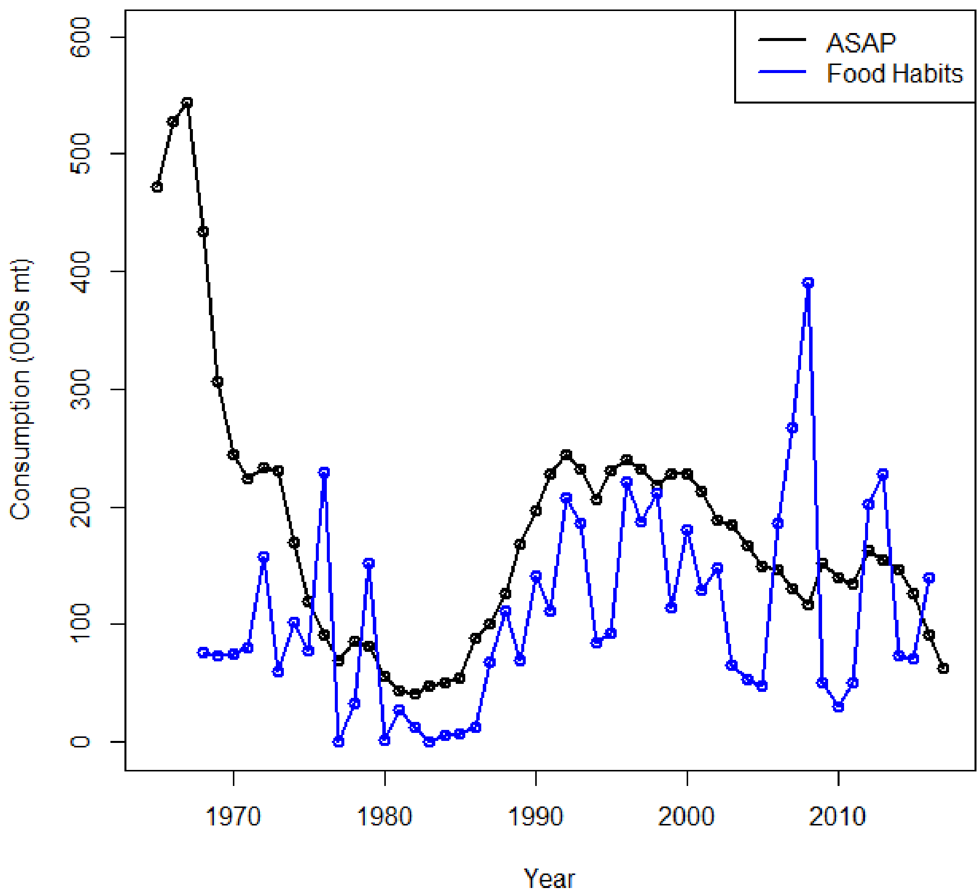


Figure B4- 4 Likelihood profile over time- and age-invariant natural mortality values for the base Atlantic herring ASAP model

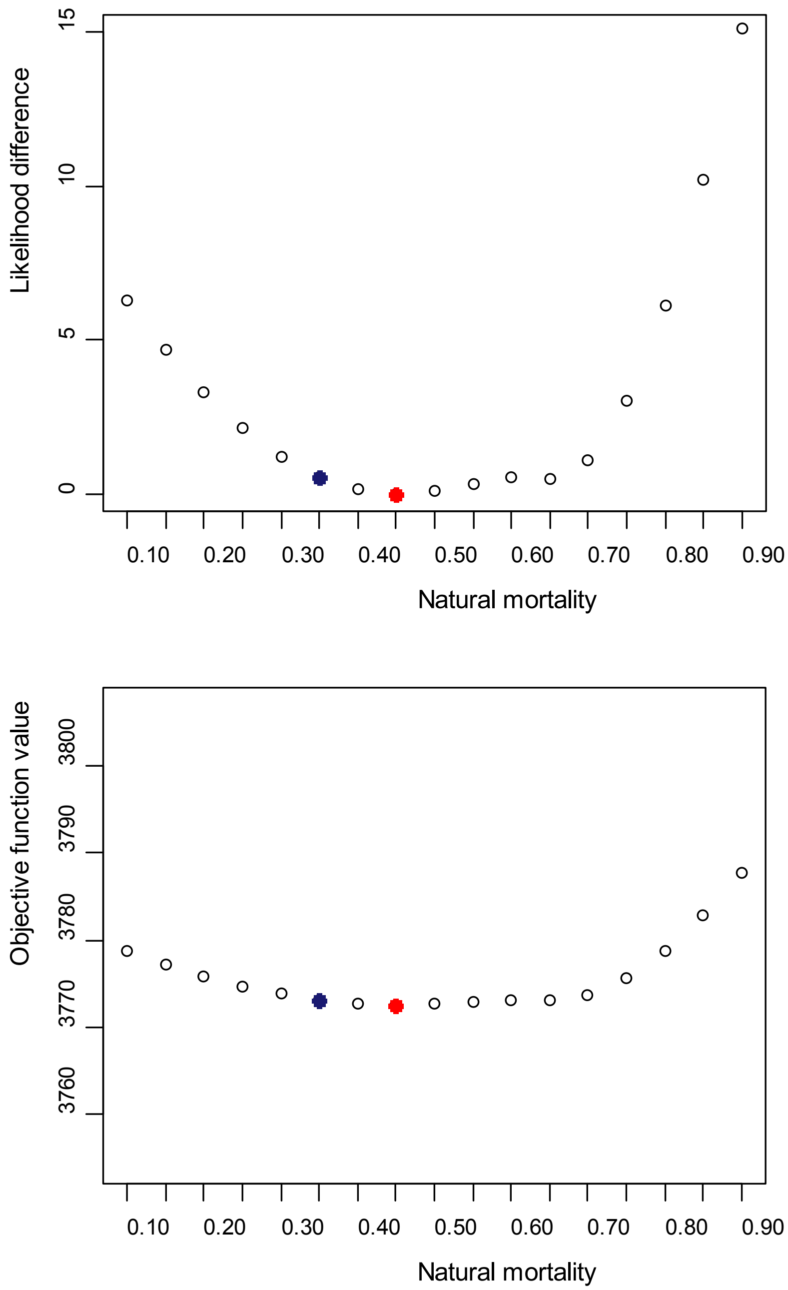
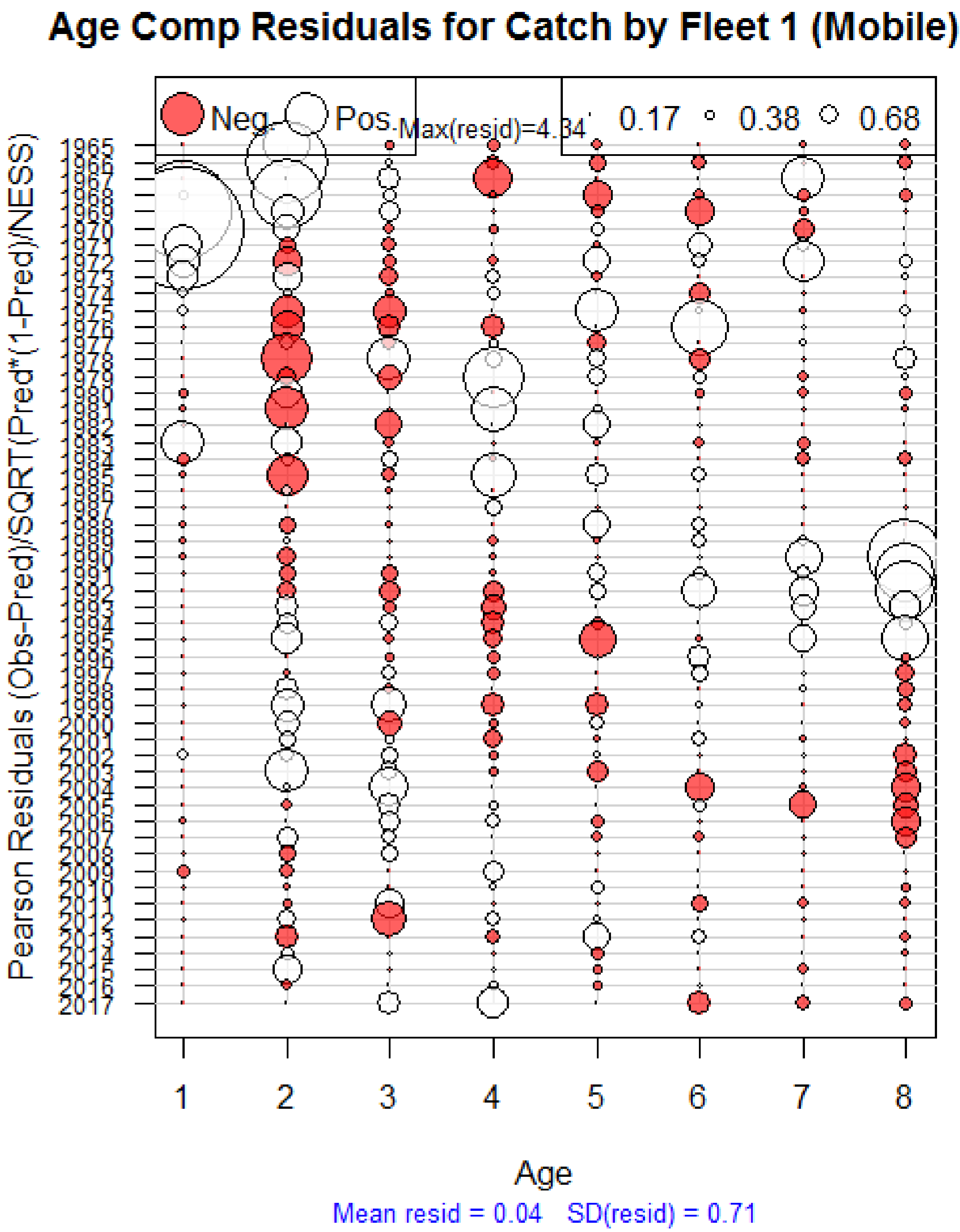
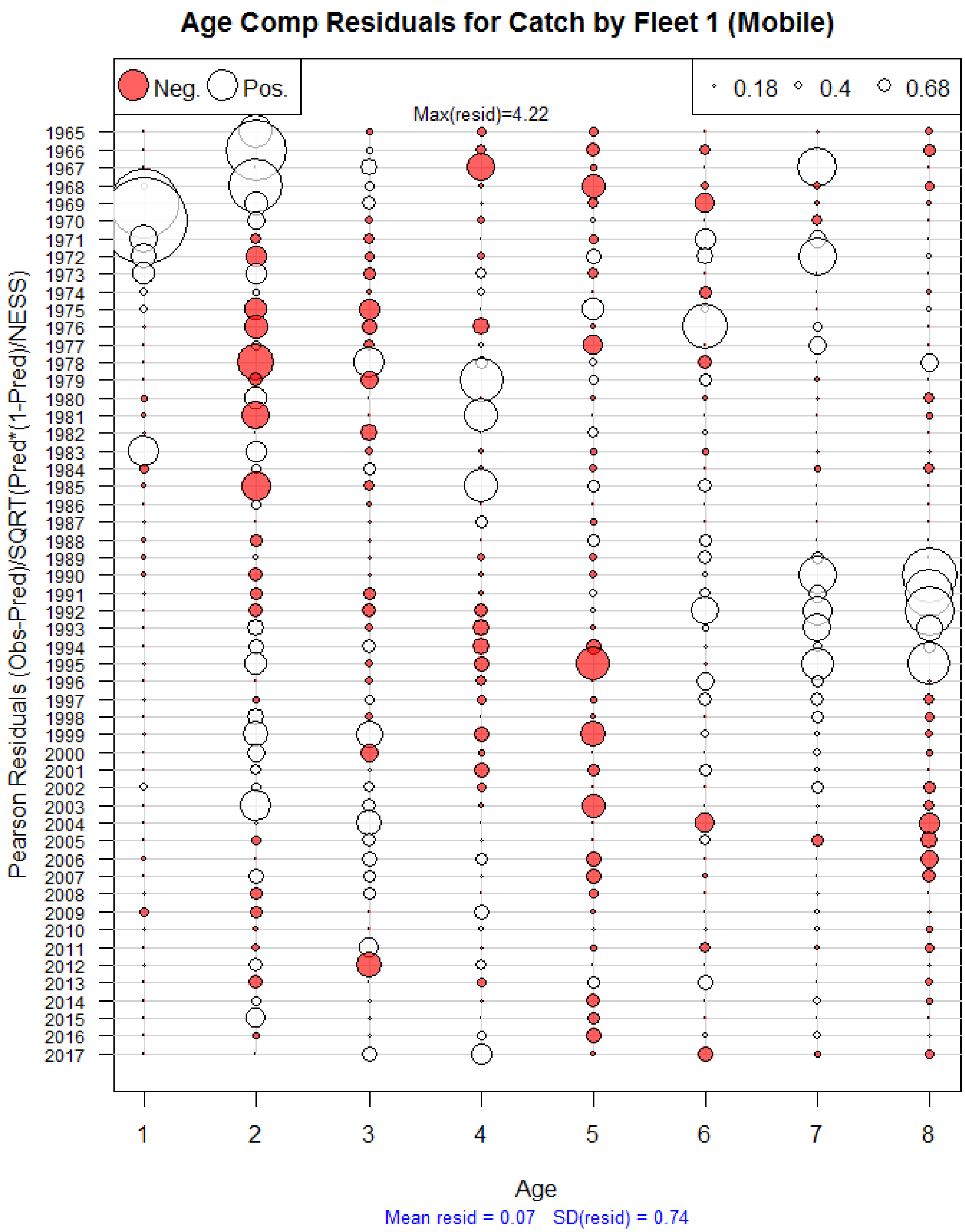


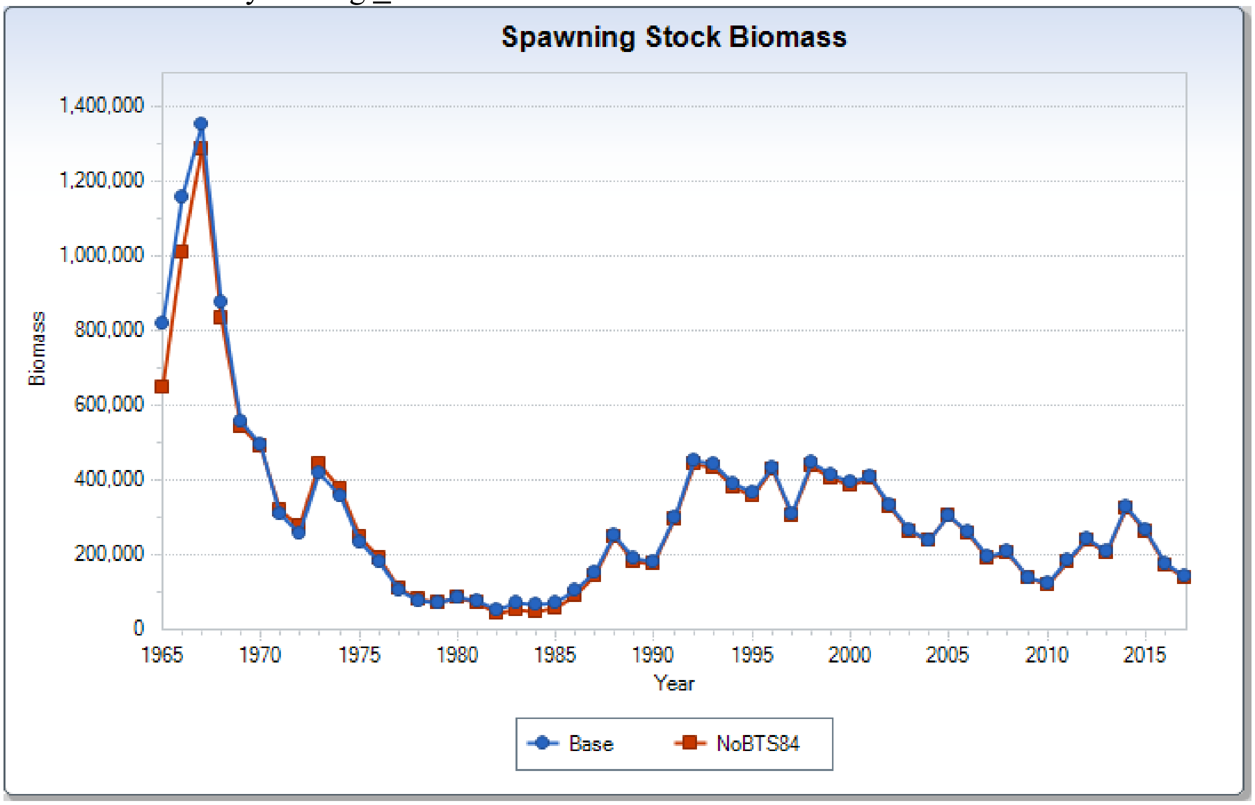
Figure B4- 5 Fits to Atlantic herring age composition for the mobile fleet from the base ASAP model (top panel) and from a fit with the mobile fleet selectivity at ages 5 and 6 fixed at 1.0 (bottom panel)





Age Comp Residuals for Catch by Fleet 1 (Mobile)
Neg.
Pos.
0.18
0.4
0.68
Max(resid)=4.22
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
1
2
3
4
5
6
7
8
Age
Mean resid = 0.07   SD(resid) = 0.74

Figure B4- 6 Effect of inclusion (Base) or exclusion (NoBTS84) of the NMFS spring and fall bottom trawl surveys during ≤1984



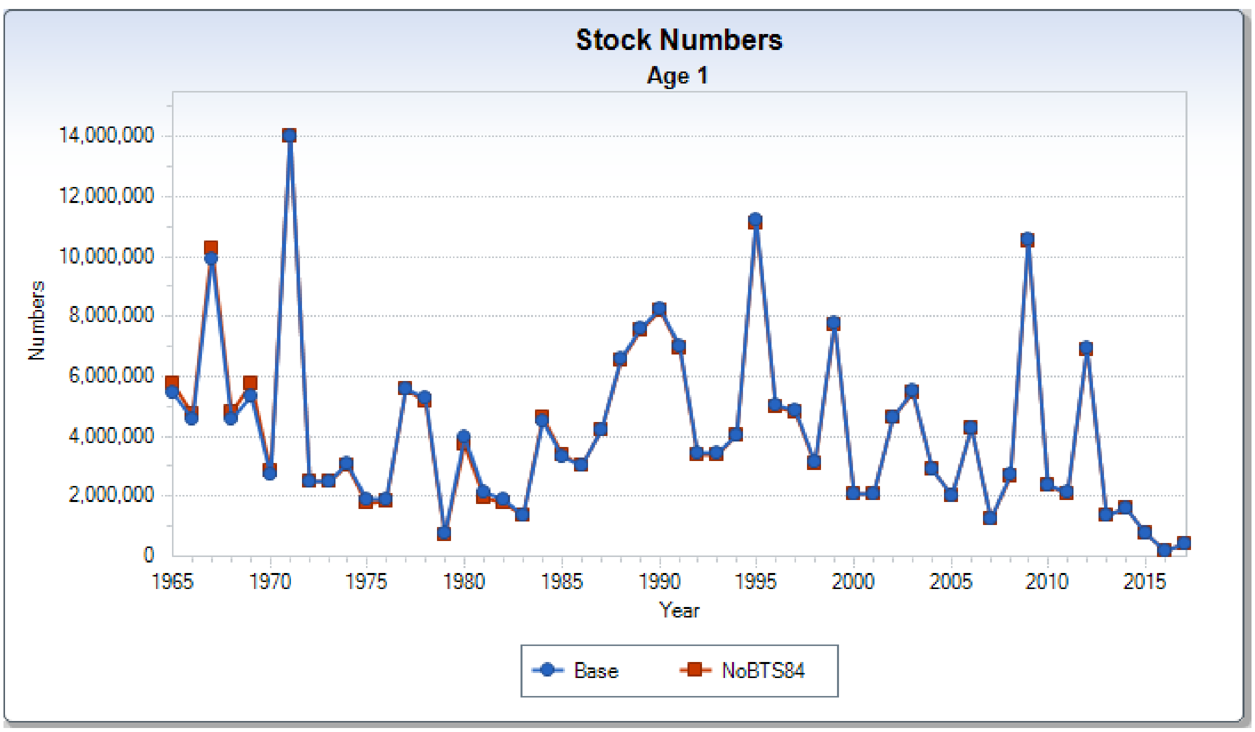


Figure B4- 7 RMSE of the indices after iteratively reweighting in the base ASAP model

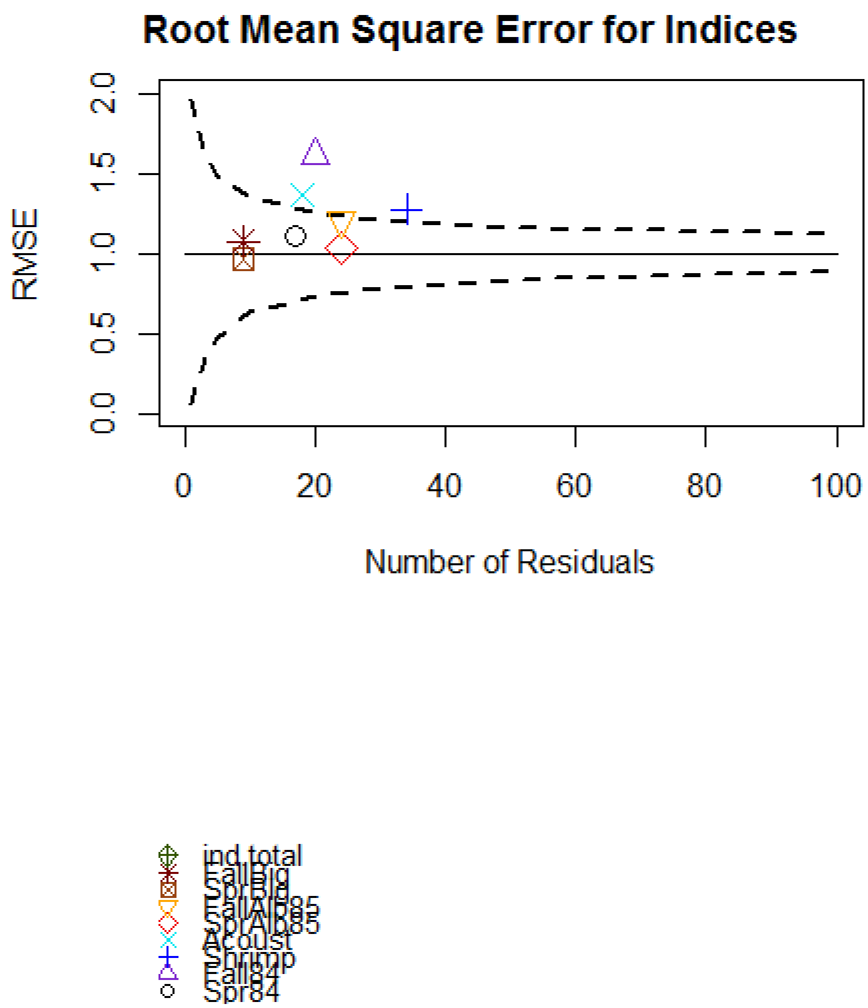


Figure B4- 8 Likelihood profile over steepness for the base ASAP model.

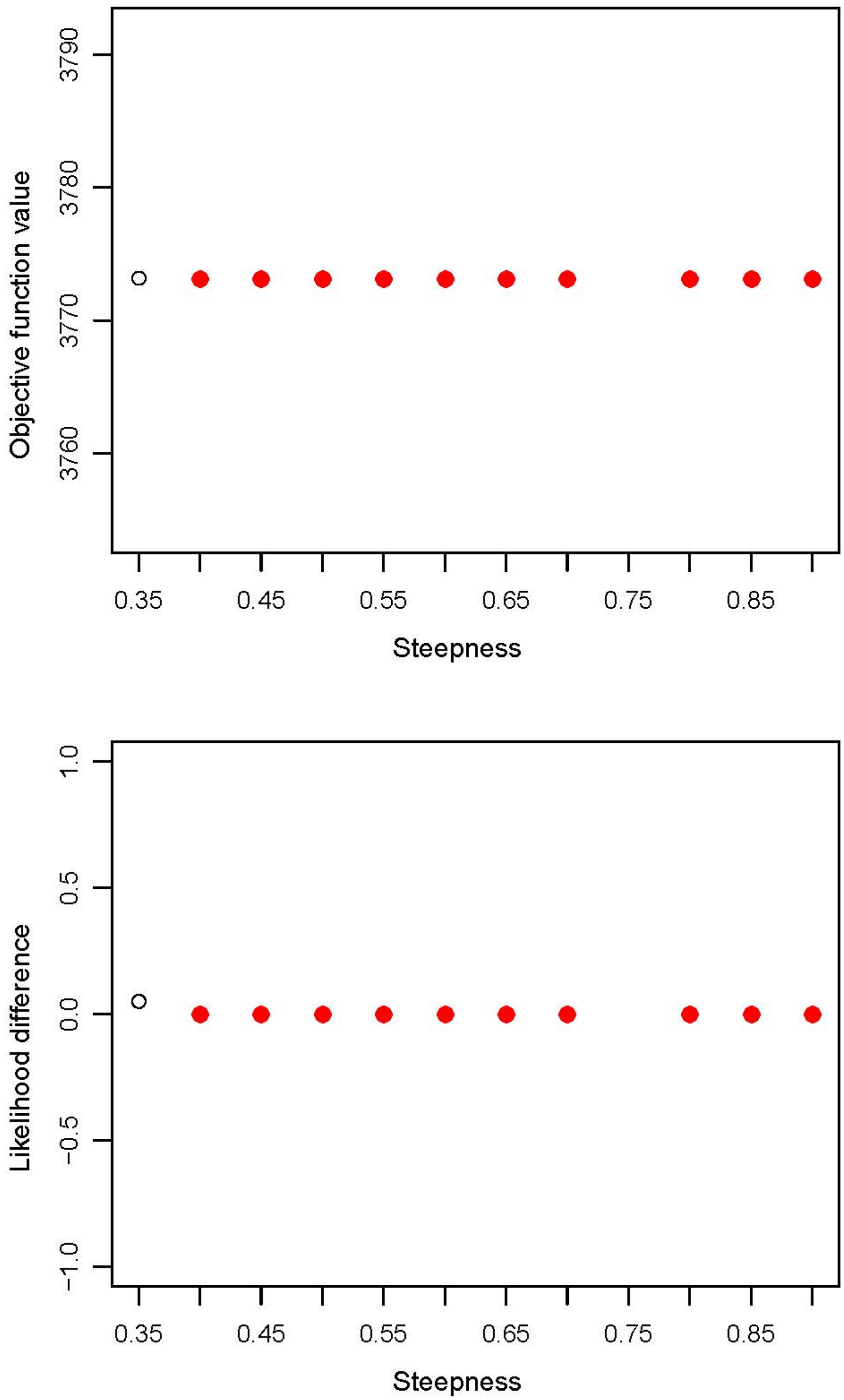
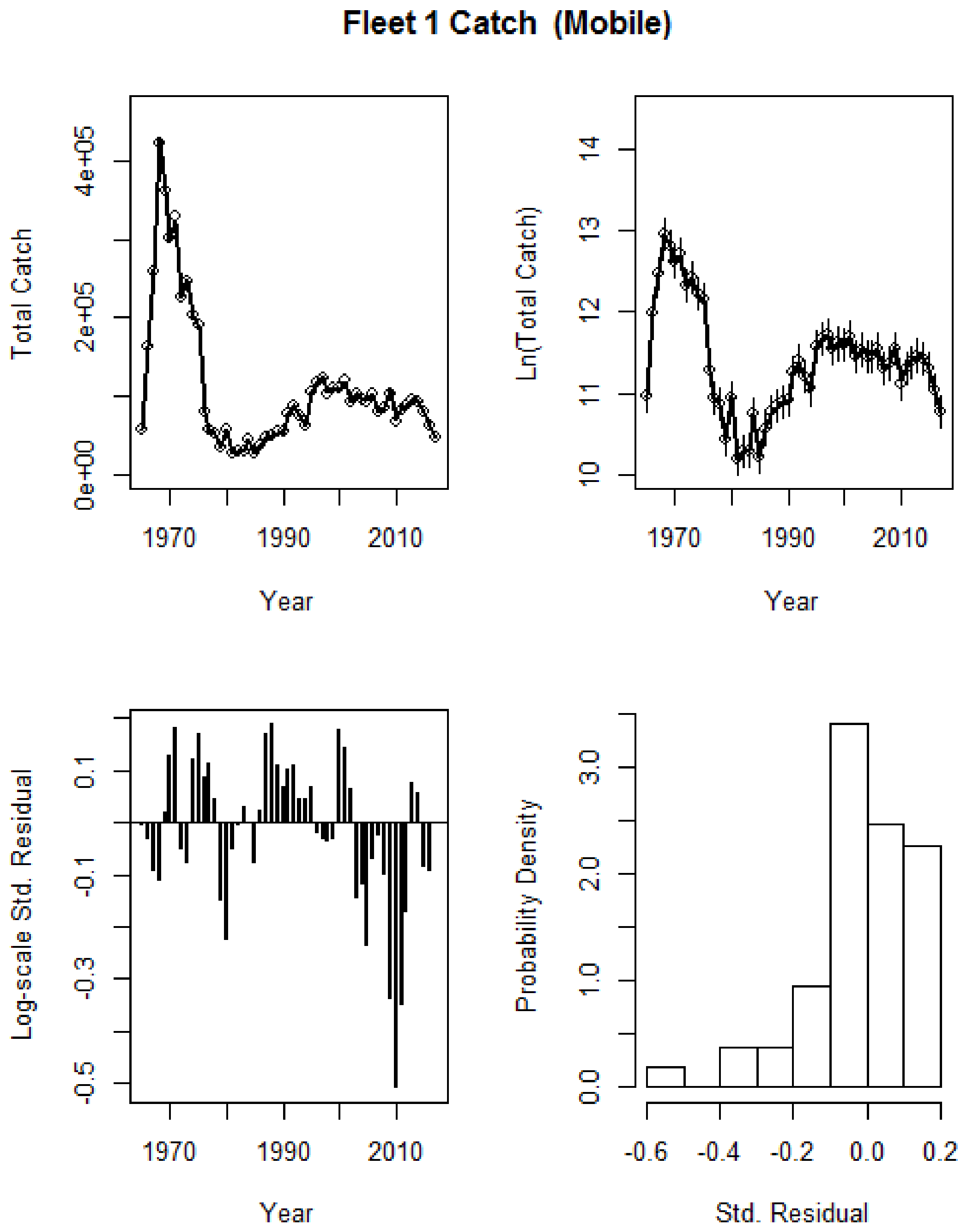


Figure B4- 9 Fits to Atlantic herring catch (mt) for the mobile (top panel) and fixed (bottom panel) fleets from the fit of the base ASAP model



Fleet 2 Catch (Fixed)

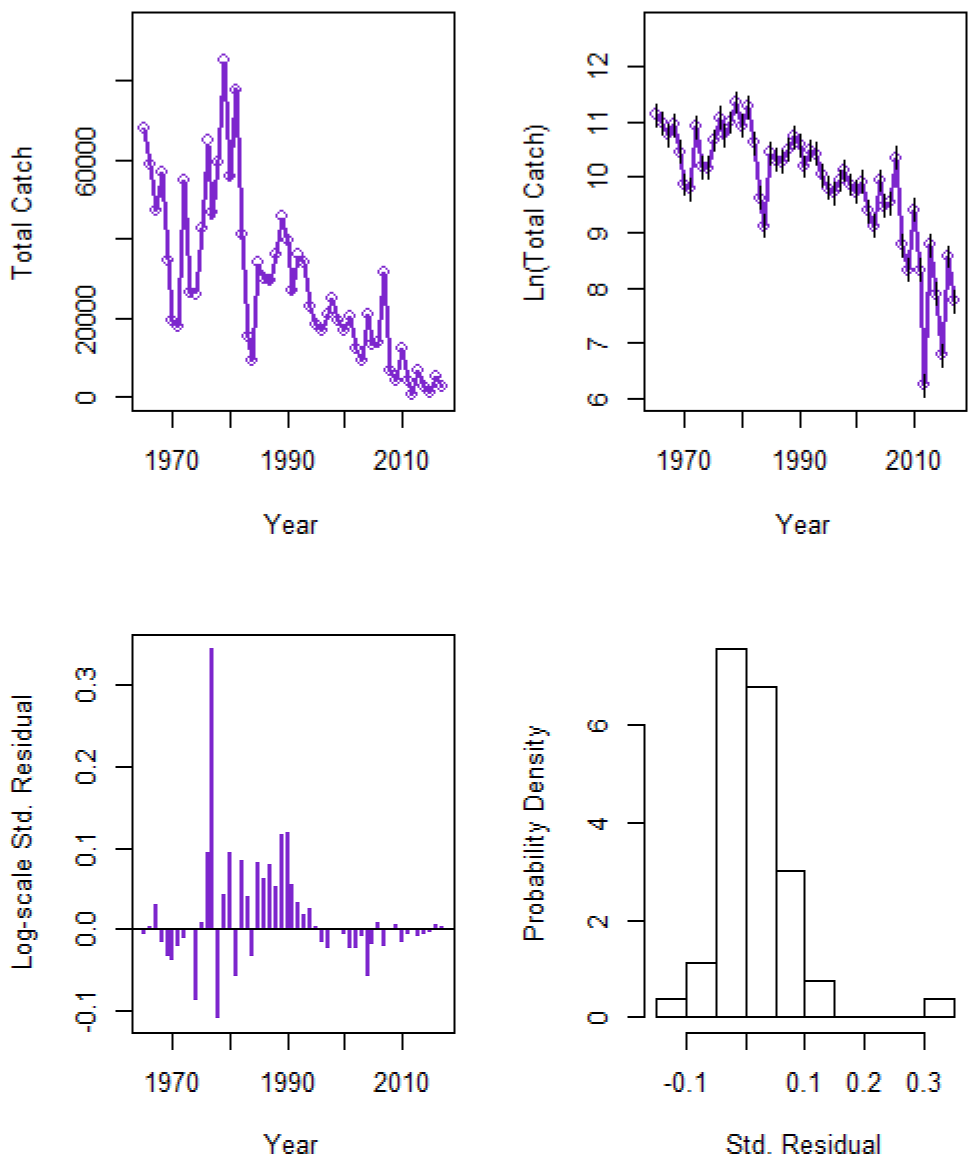
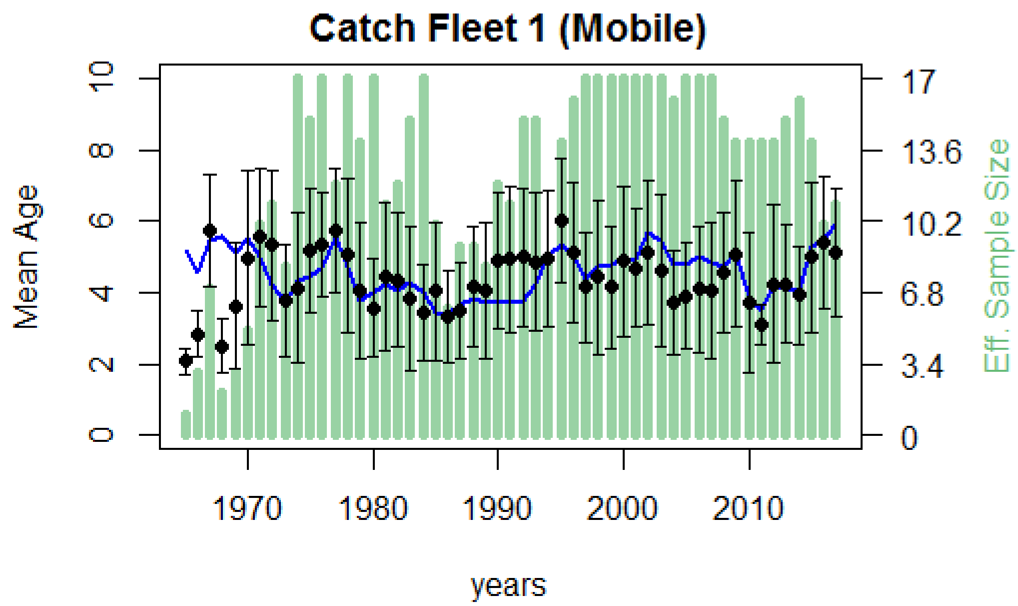
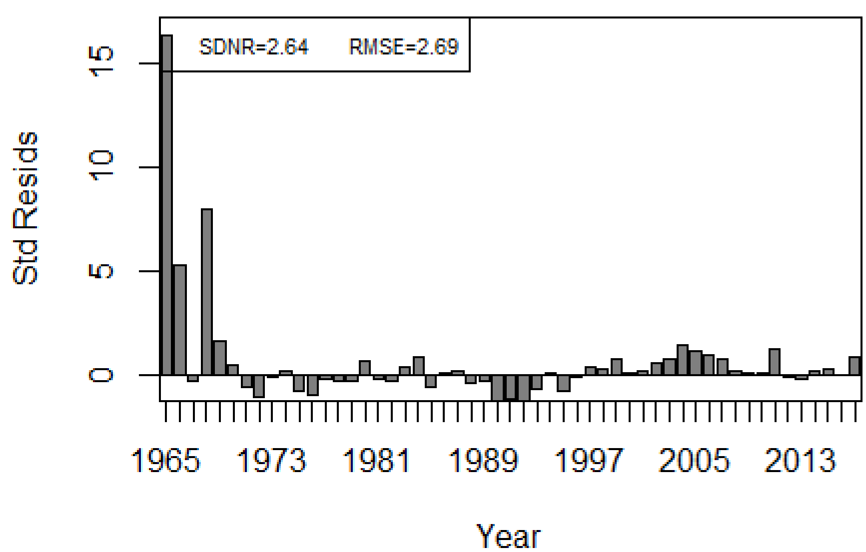


Figure B4- 10 Fits to Atlantic herring mean age and standardized residuals for the mobile (top panel) and fixed (bottom panel) fleets from the fit of the base ASAP model





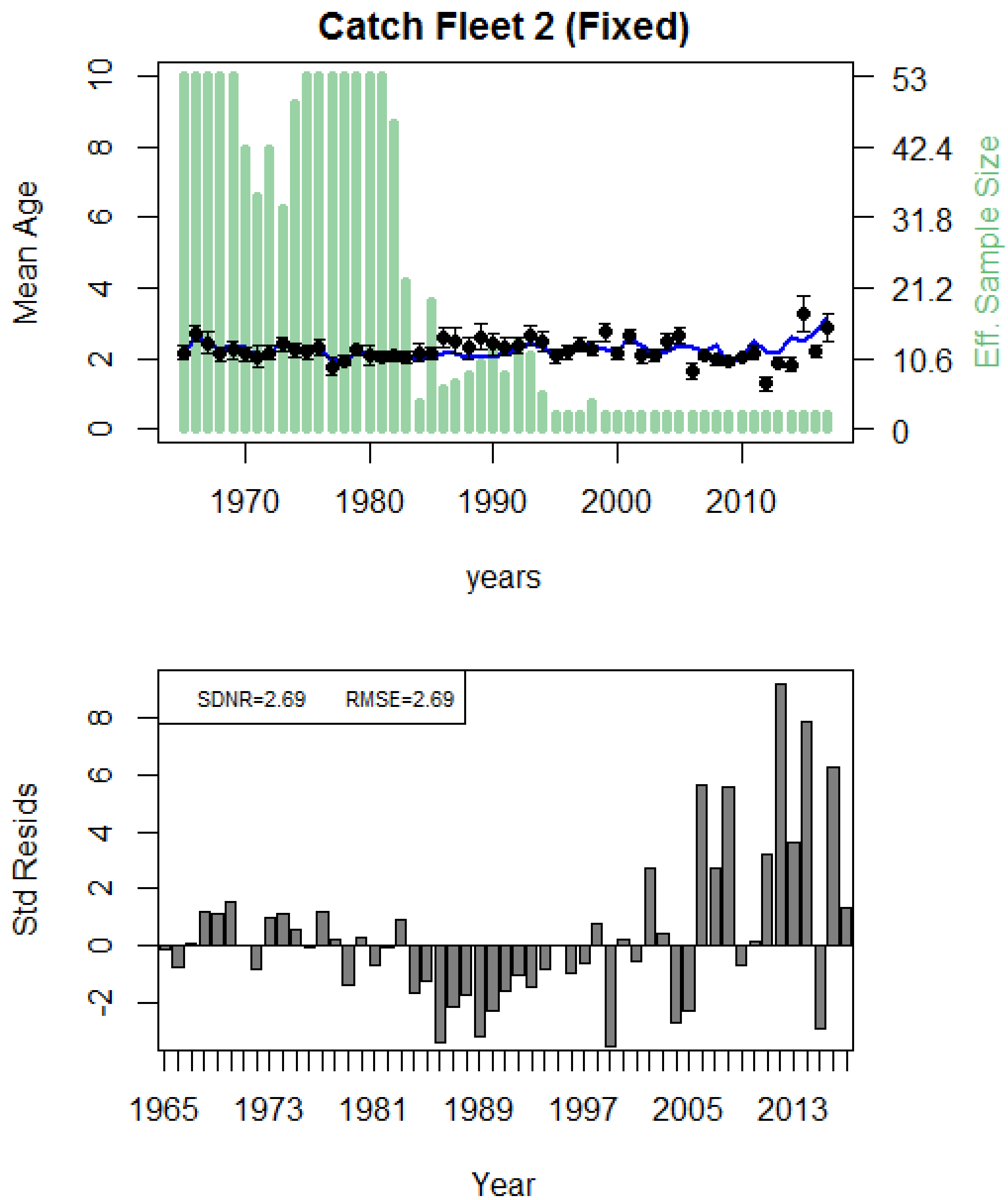
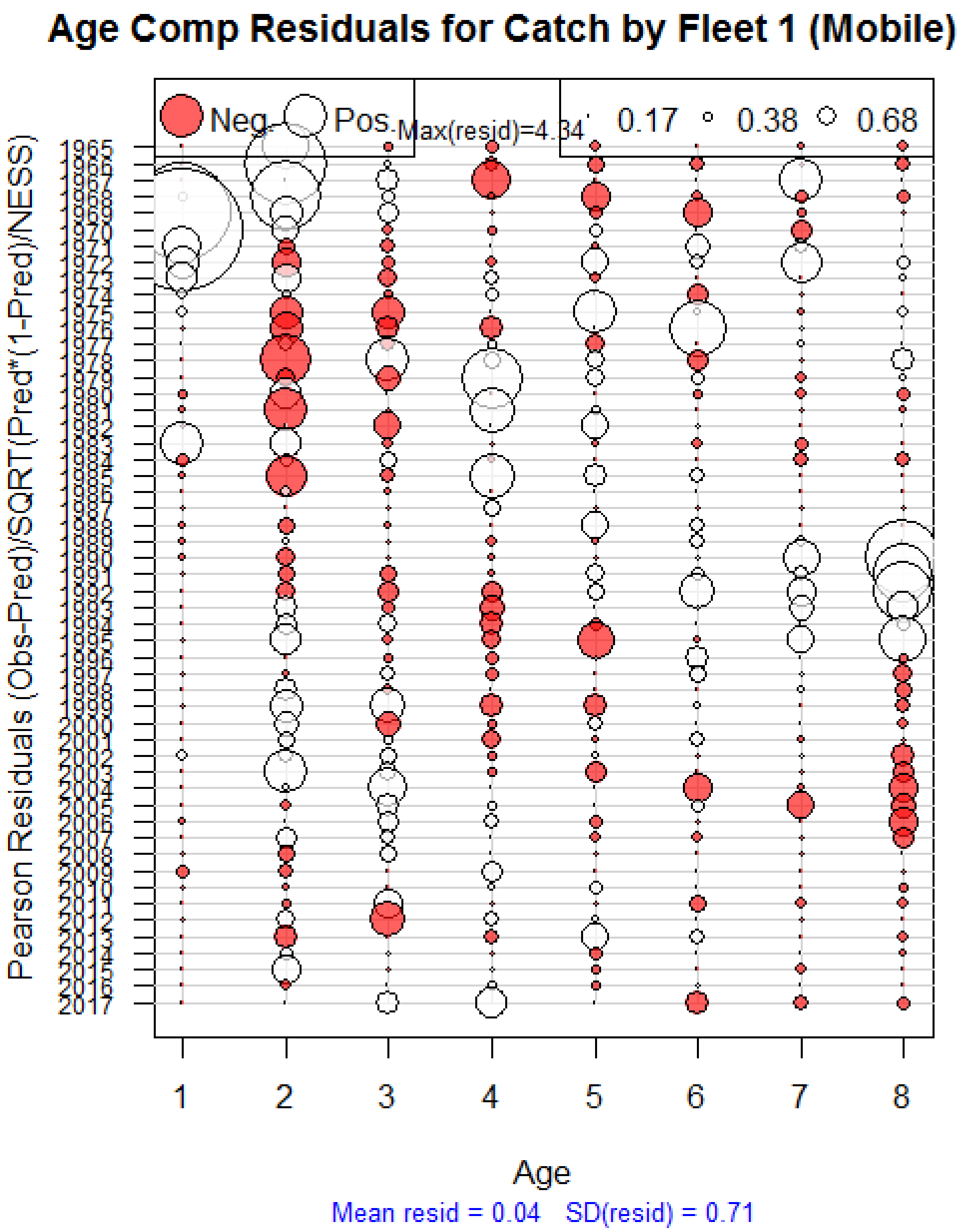

Catch Fleet 2 (Fixed)
Mean Age
Eff. Sample Size
years
SDNR=2.69
RMSE=2.69
Std Resids
Year

Figure B4- 11 Fits to the Atlantic herring age compositions for the mobile (top panel) and fixed (bottom panel) fleets from the base ASAP model



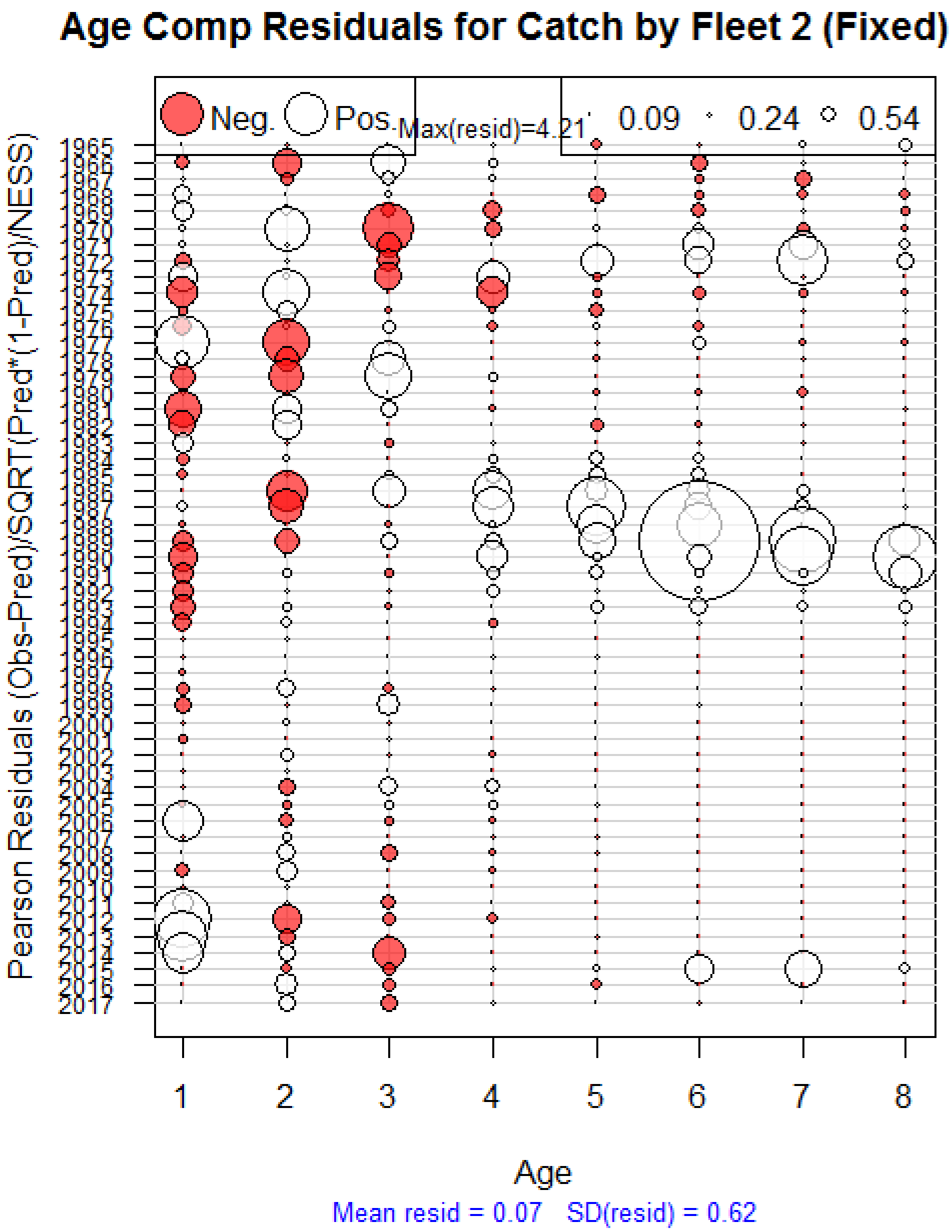
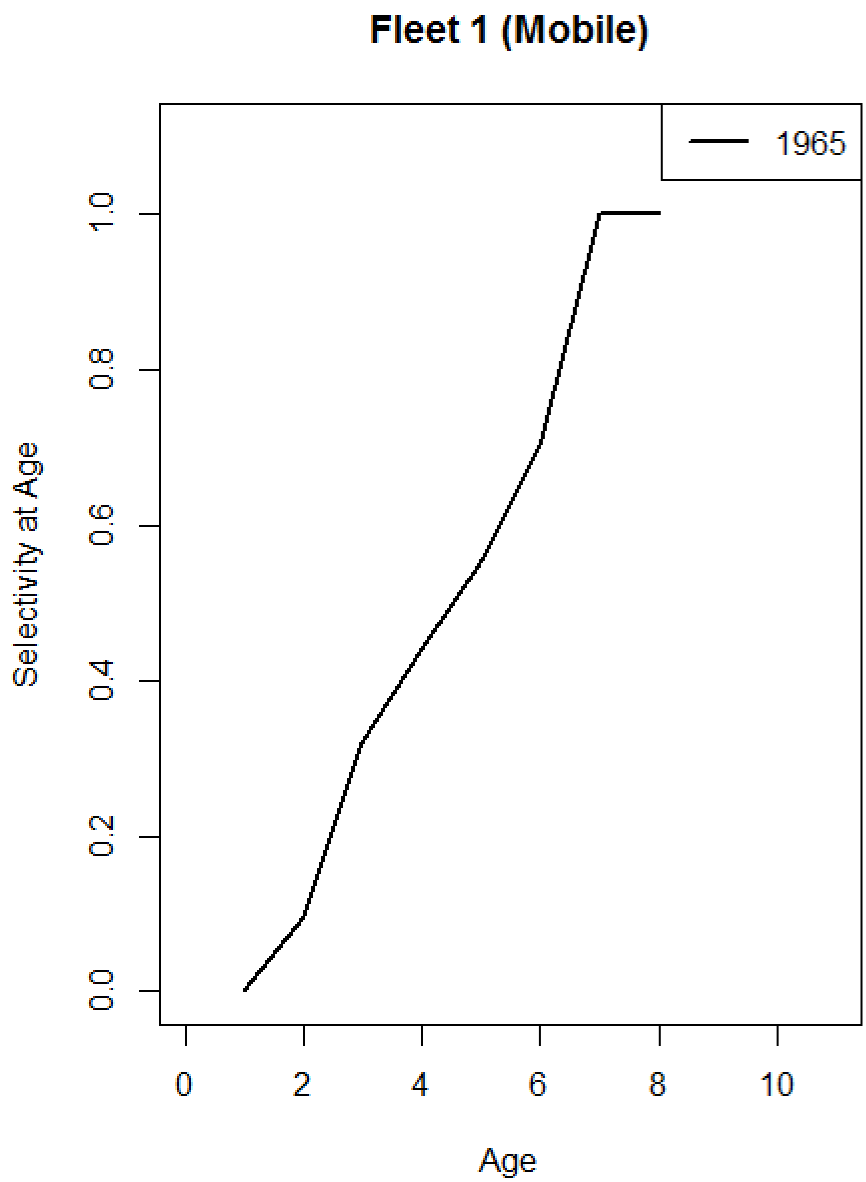
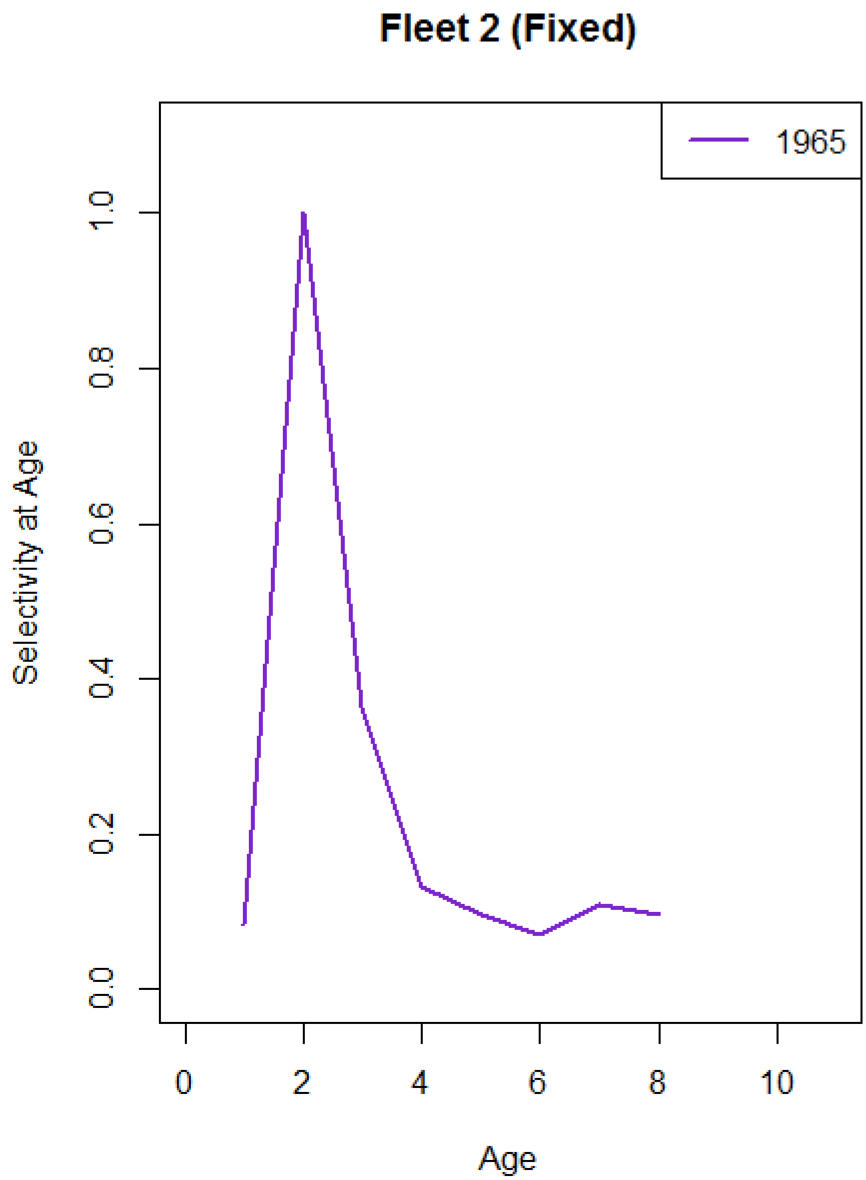

Age Comp Residuals for Catch by Fleet 2 (Fixed)
Neg.
Pos.
Max(resid)=4.21
0.09
0.24
0.54
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
Age
Mean resid = 0.07 SD(resid) = 0.62

Figure B4- 12 Selectivity for the mobile (top panel) and fixed (bottom panel) fleets from the base ASAP model





Fleet 2 (Fixed)
1965
Selectivity at Age
0.0
0.2
0.4
0.6
0.8
1.0
0
2
4
6
8
10
Age

Figure B4- 13 Average selectivity and terminal year maturity at age from the base ASAP model

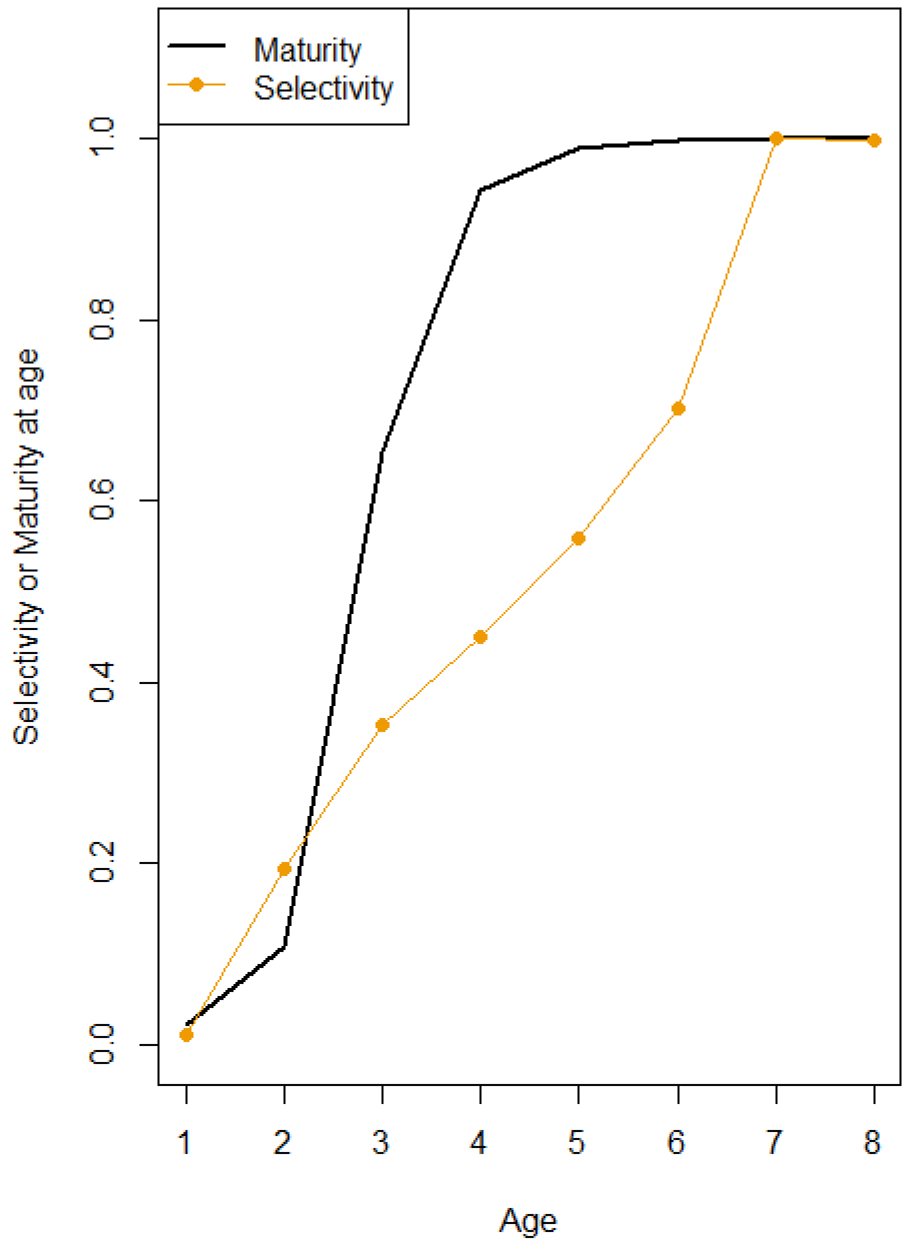
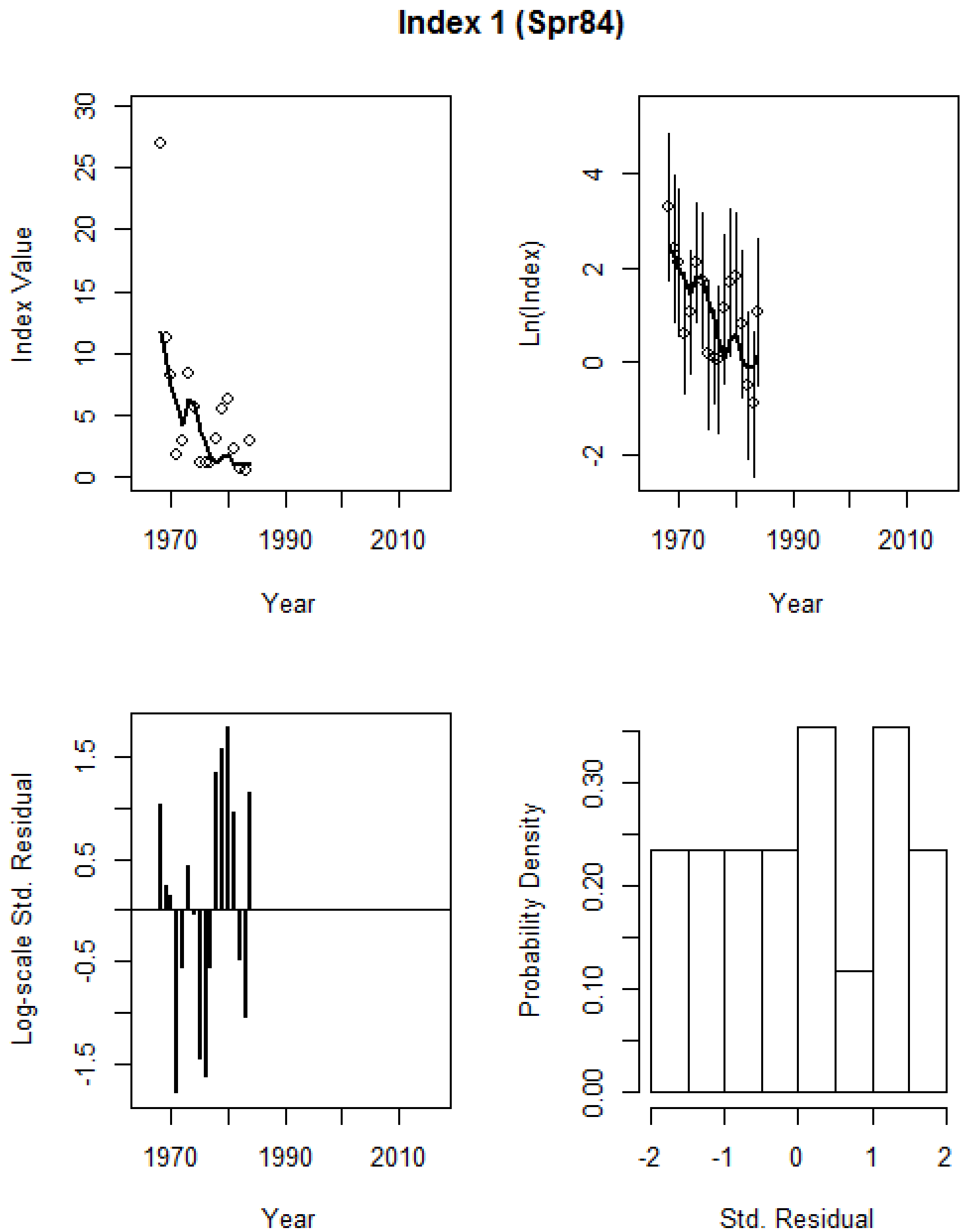
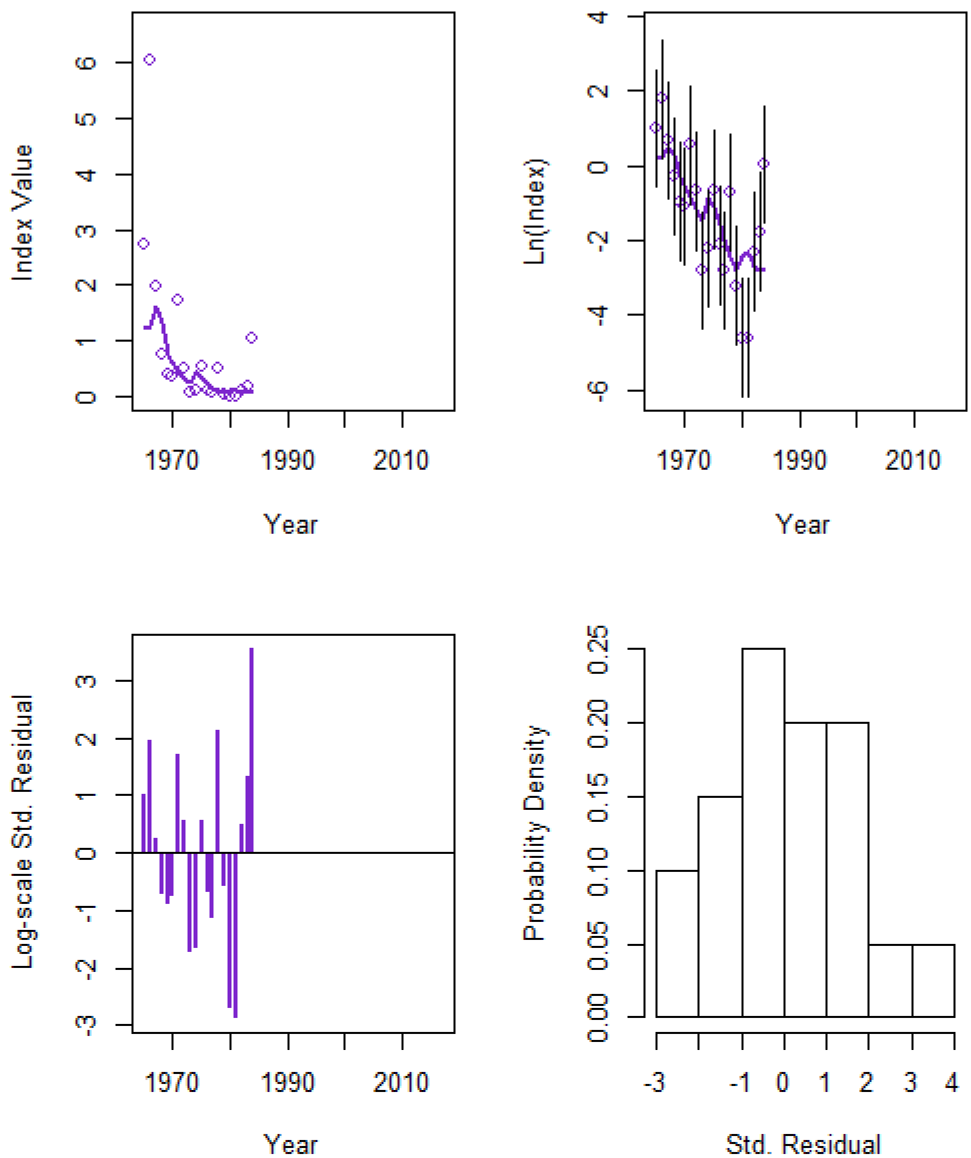
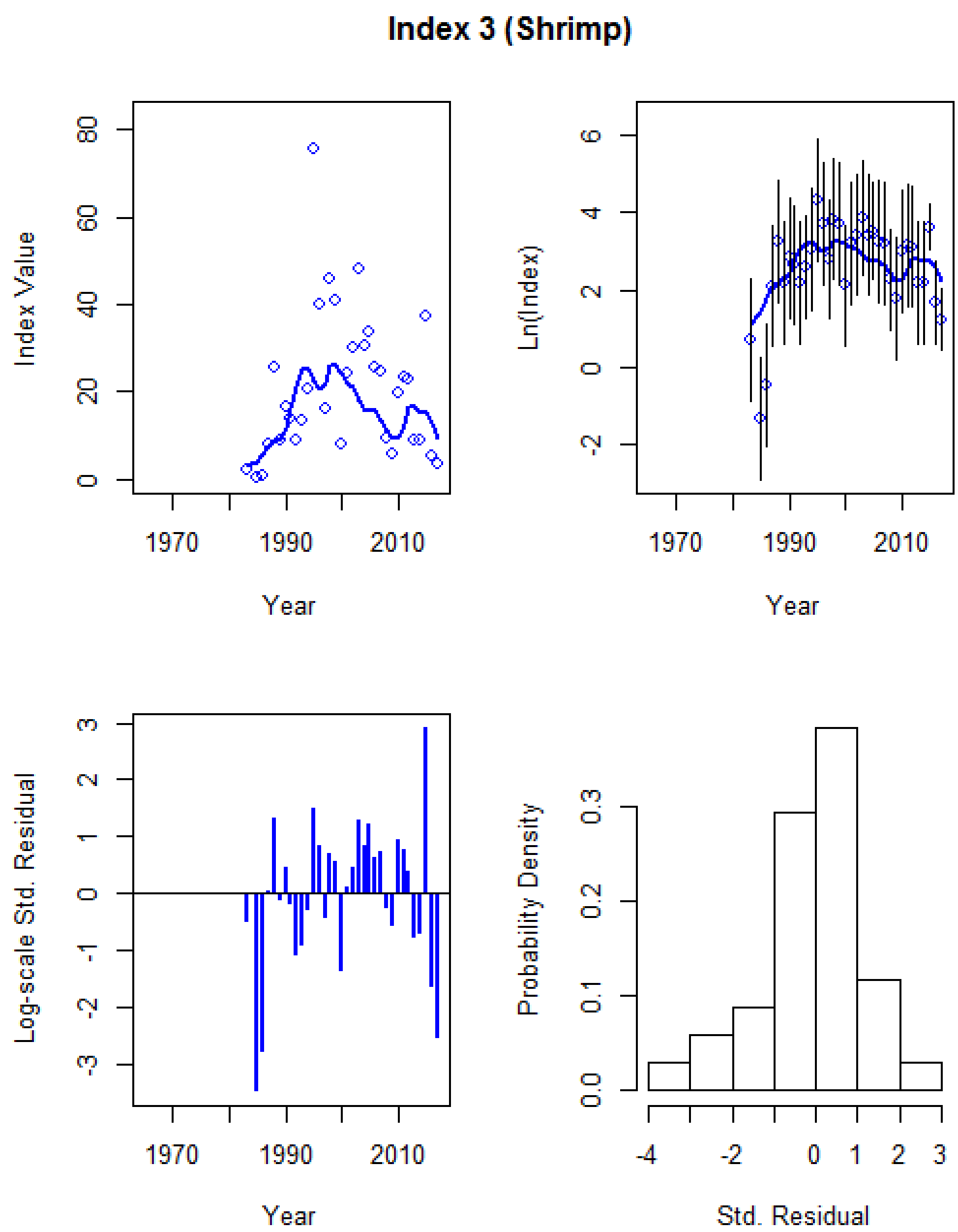


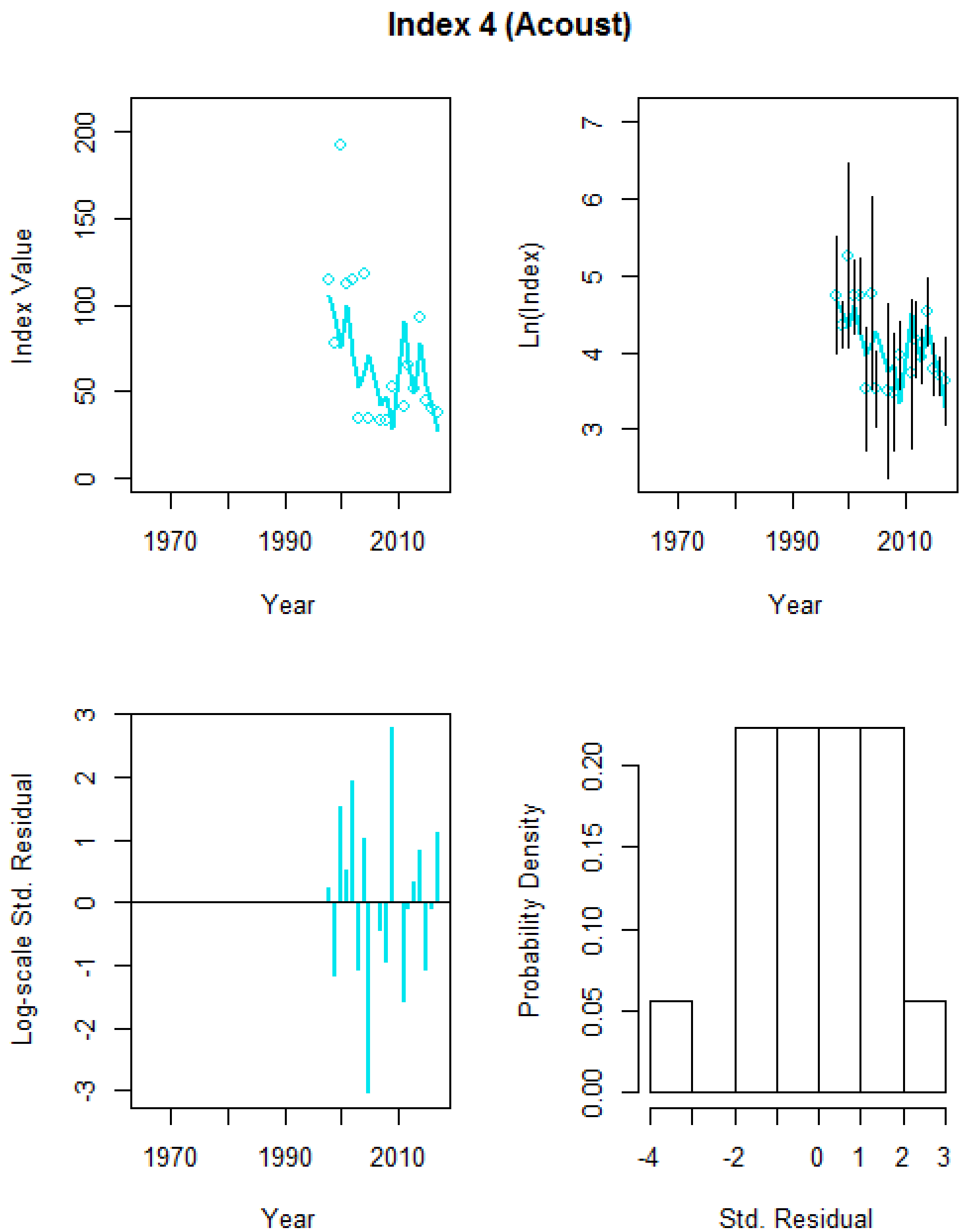
Figure B4- 14 Fits to indices for the base ASAP model (Spr84: Albatross 1965-1984; Fall84: Albatross 1965-1984; Shrimp: NMFS summer/shrimp; Acoust: NMFS acoustic index; Spr85: Albatross 1985-2008; Fall85: Albatross 1985-2008; SprBig: Bigelow 2009-2017; FallBig: Bigelow 2009-2017)



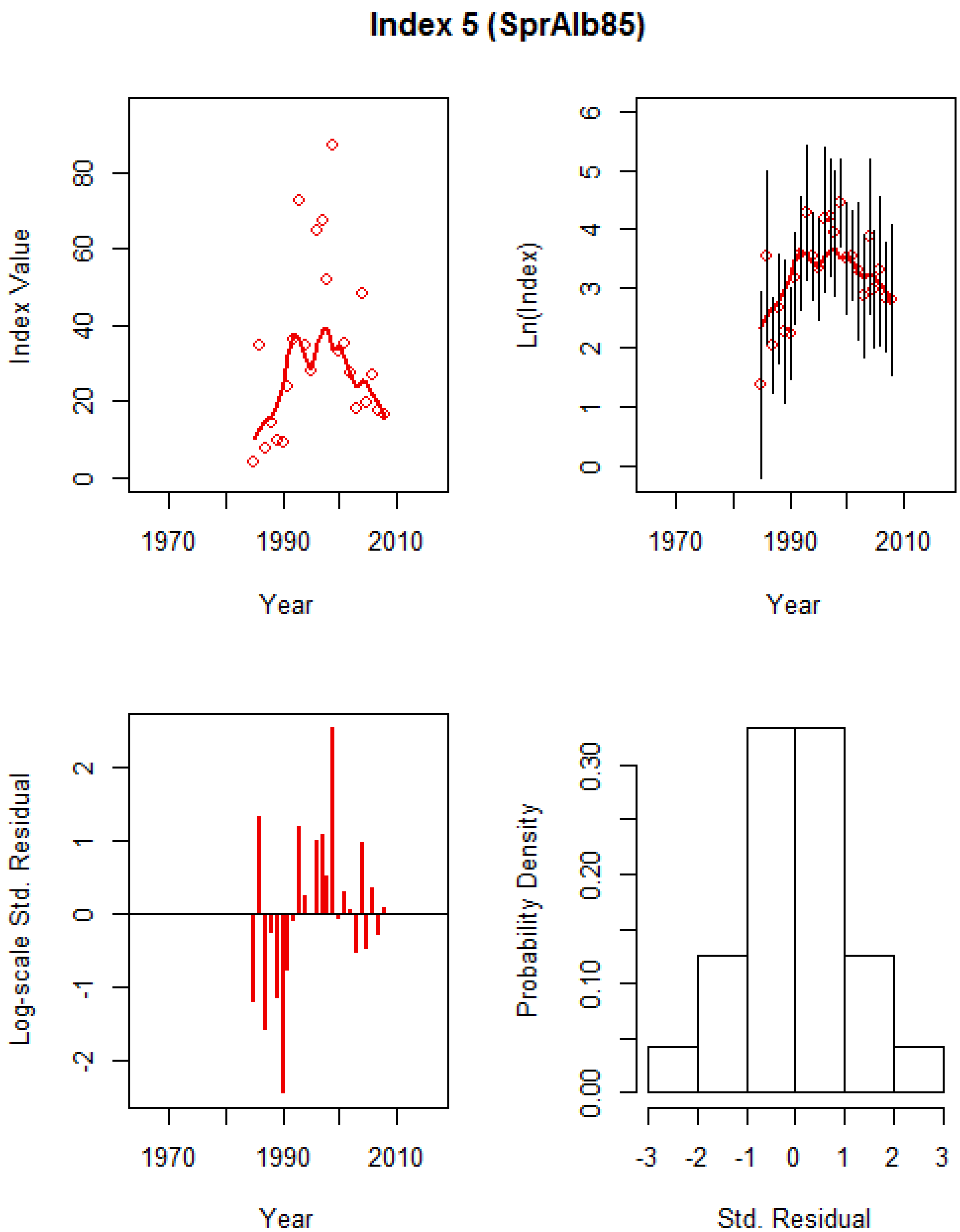
Index 2 (Fall84)

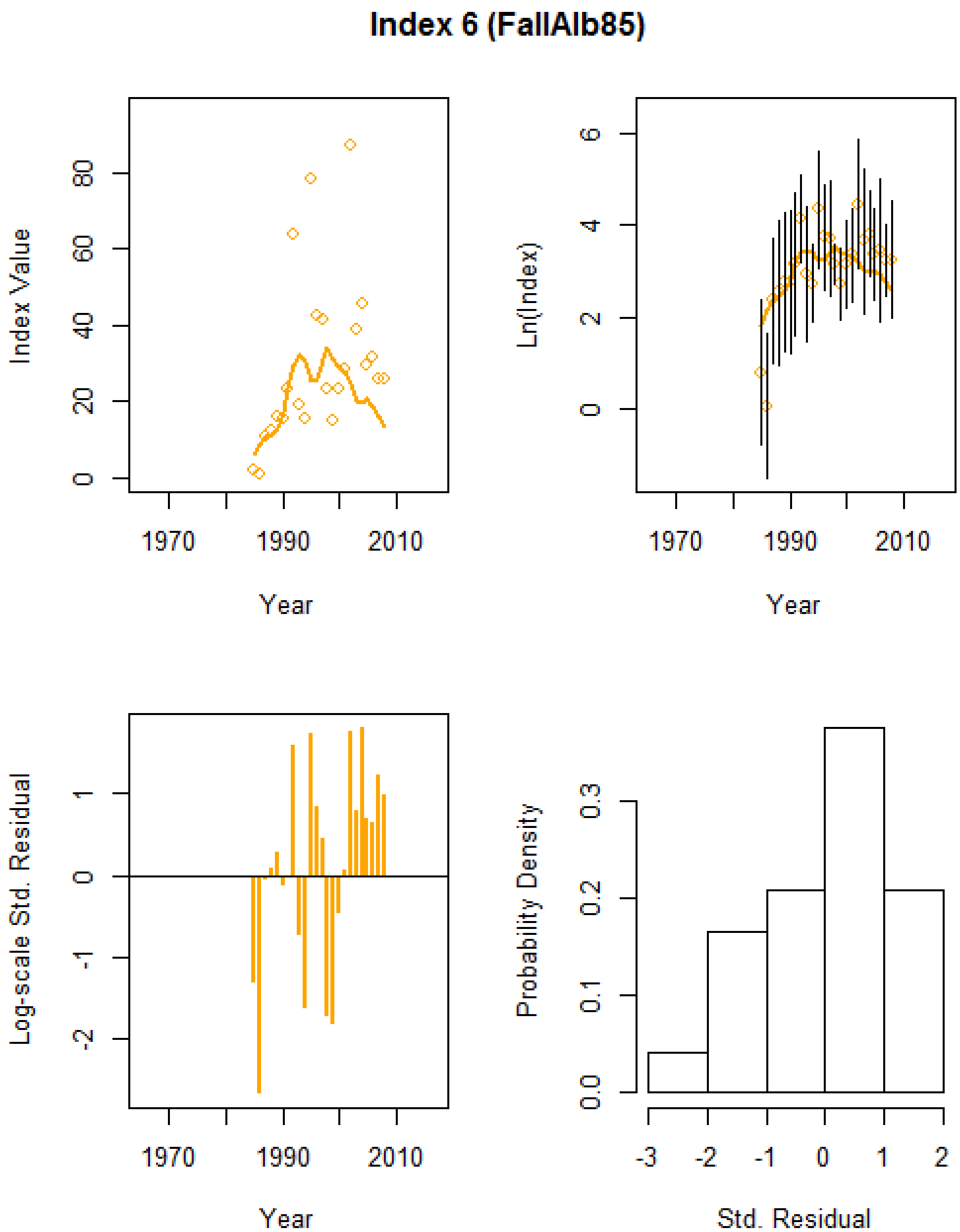


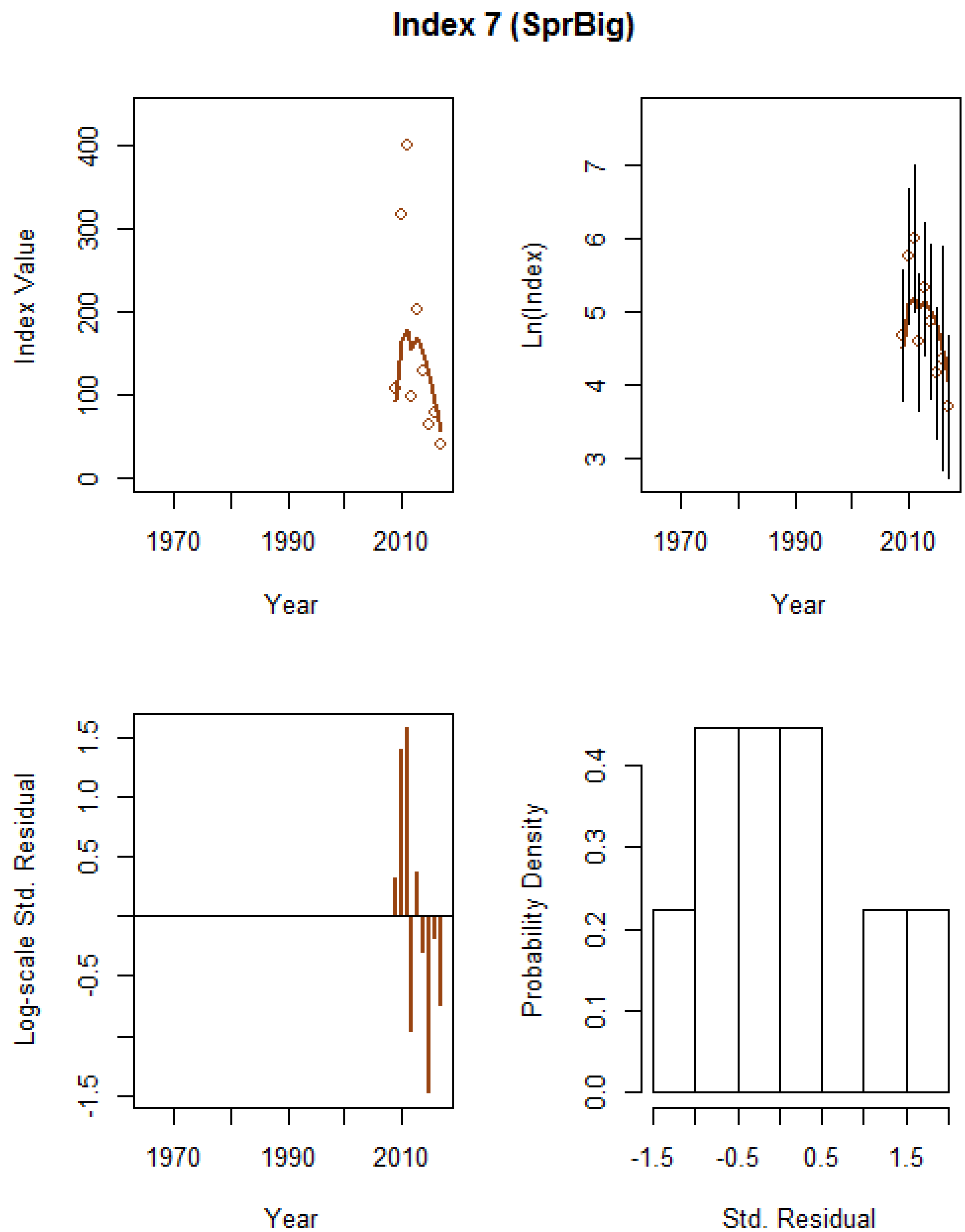

Index 3 (Shrimp)
Index Value
Year
Ln(Index)
Year
Log-scale Std. Residual
Year
Probability Density
Std. Residual



Index 4 (Acoust)
Index Value
0
50
100
150
200
1970
1990
2010
Year
Ln(Index)
3
4
5
6
7
1970
1990
2010
Year
Log-scale Std. Residual
-3
-2
-1
0
1
2
3
1970
1990
2010
Year
Probability Density
0.00
0.05
0.10
0.15
0.20
-4
-2
0
1
2
3
Std. Residual


Index 5 (SprAlb85)
Index Value
Year
Ln(Index)
Year
Log-scale Std. Residual
Year
Probability Density
Std. Residual


Index 6 (FallAlb85)
Index Value
Year
Ln(Index)
Year
Log-scale Std. Residual
Year
Probability Density
Std. Residual



Index 7 (SprBig)
Index Value
0
100
200
300
400
Ln(Index)
3
4
5
6
7
1970
1990
2010
Year
Log-scale Std. Residual
-1.5
-0.5
0.5
1.0
1.5
Probability Density
0.0
0.1
0.2
0.3
0.4
-1.5
-0.5
0.5
1.5
Std. Residual

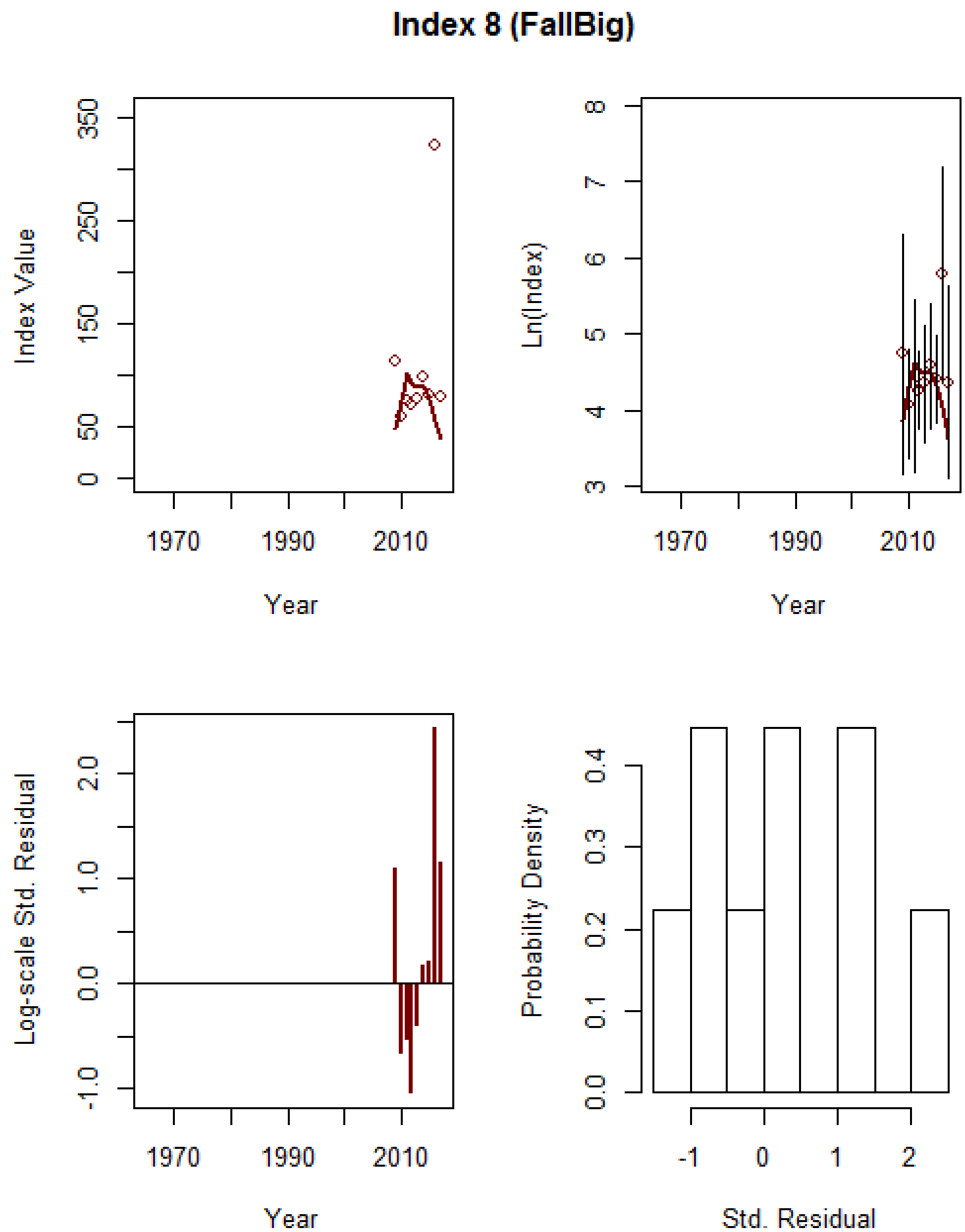
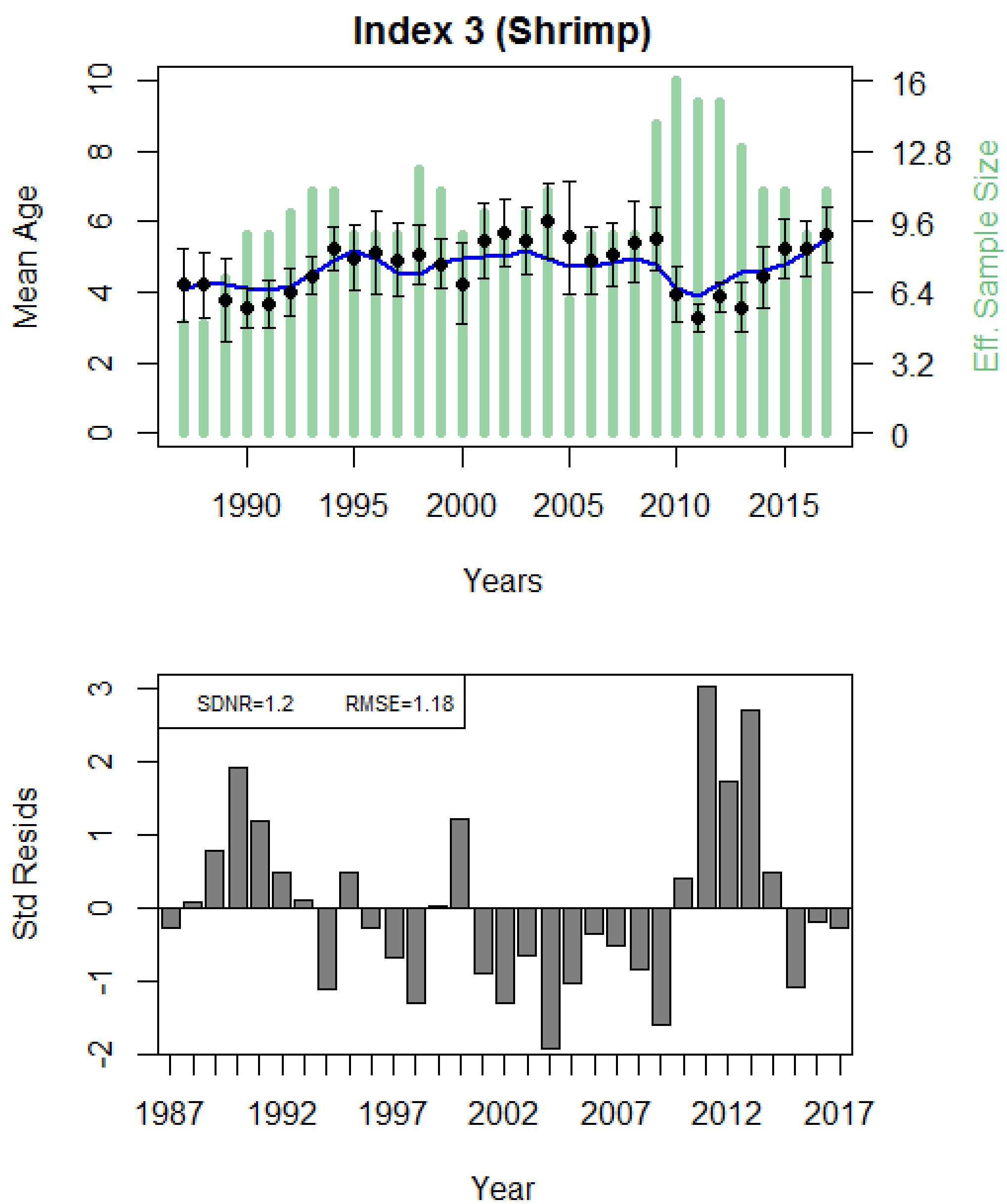
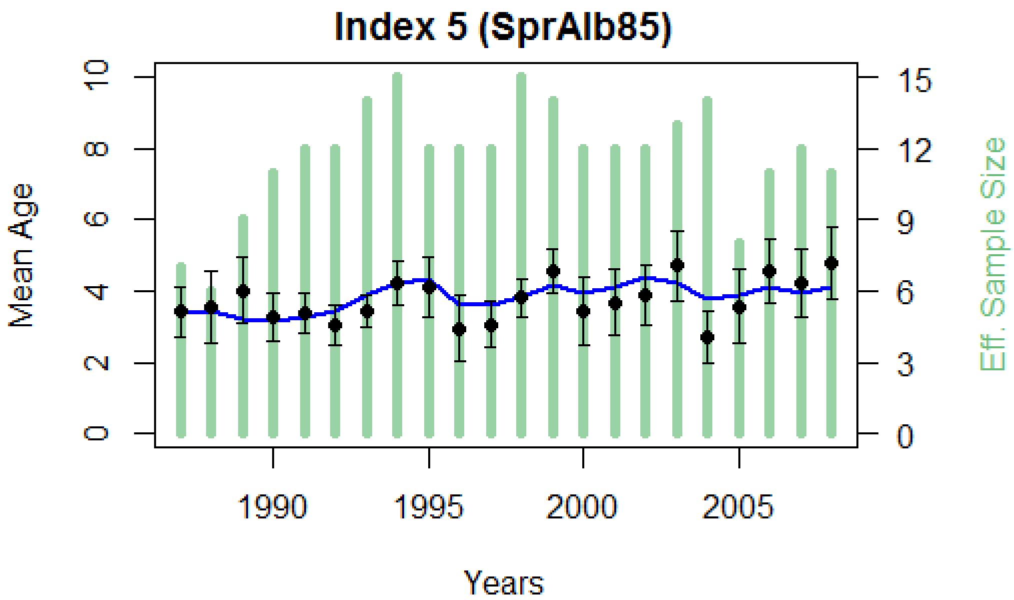
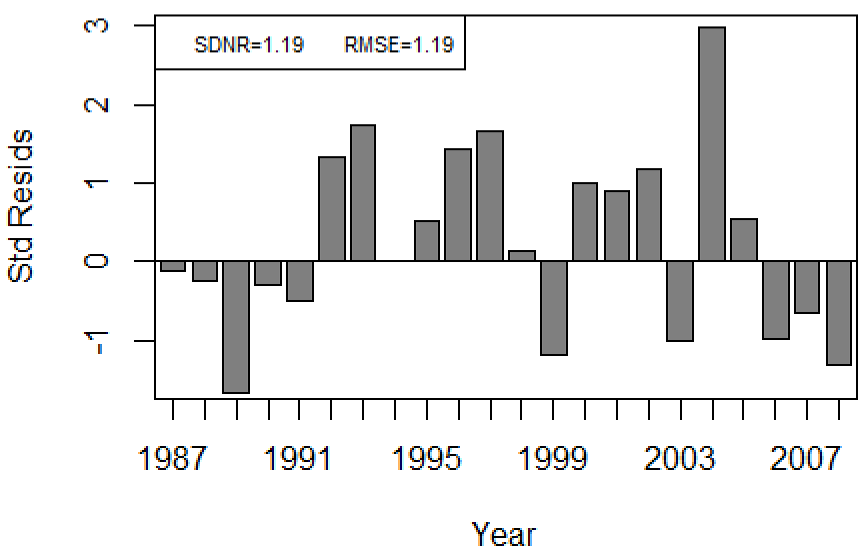

Index 8 (FallBig)
Index Value
Year
Ln(Index)
Year
Log-scale Std. Residual
Year
Probability Density
Std. Residual

Figure B4- 15 Fits to Atlantic herring mean age for the base ASAP model (Shrimp: NMFS summer/shrimp; Spr85: Albatross 1985-2008; Fall85: Albatross 1985-2008; SprBig: Bigelow 2009-2017; FallBig: Bigelow 2009-2017)

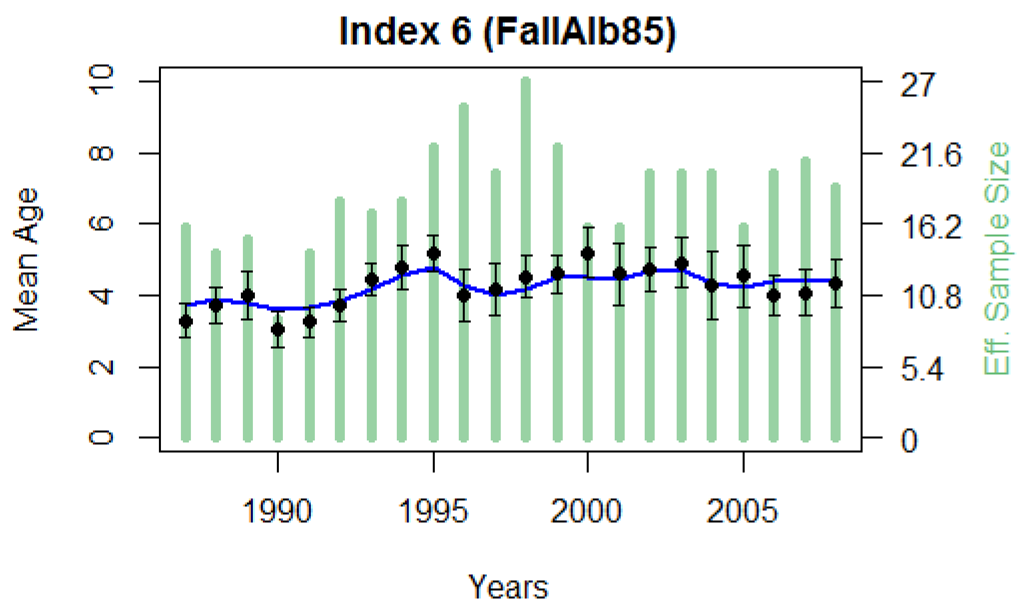


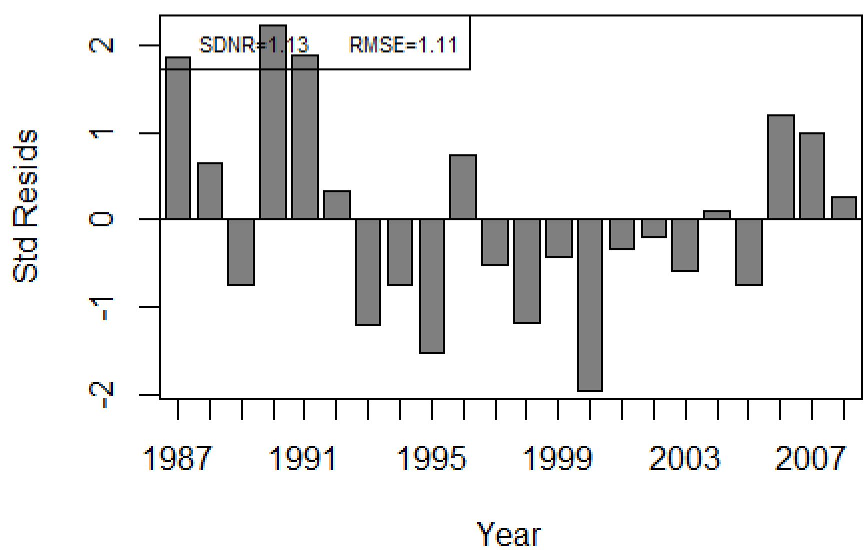


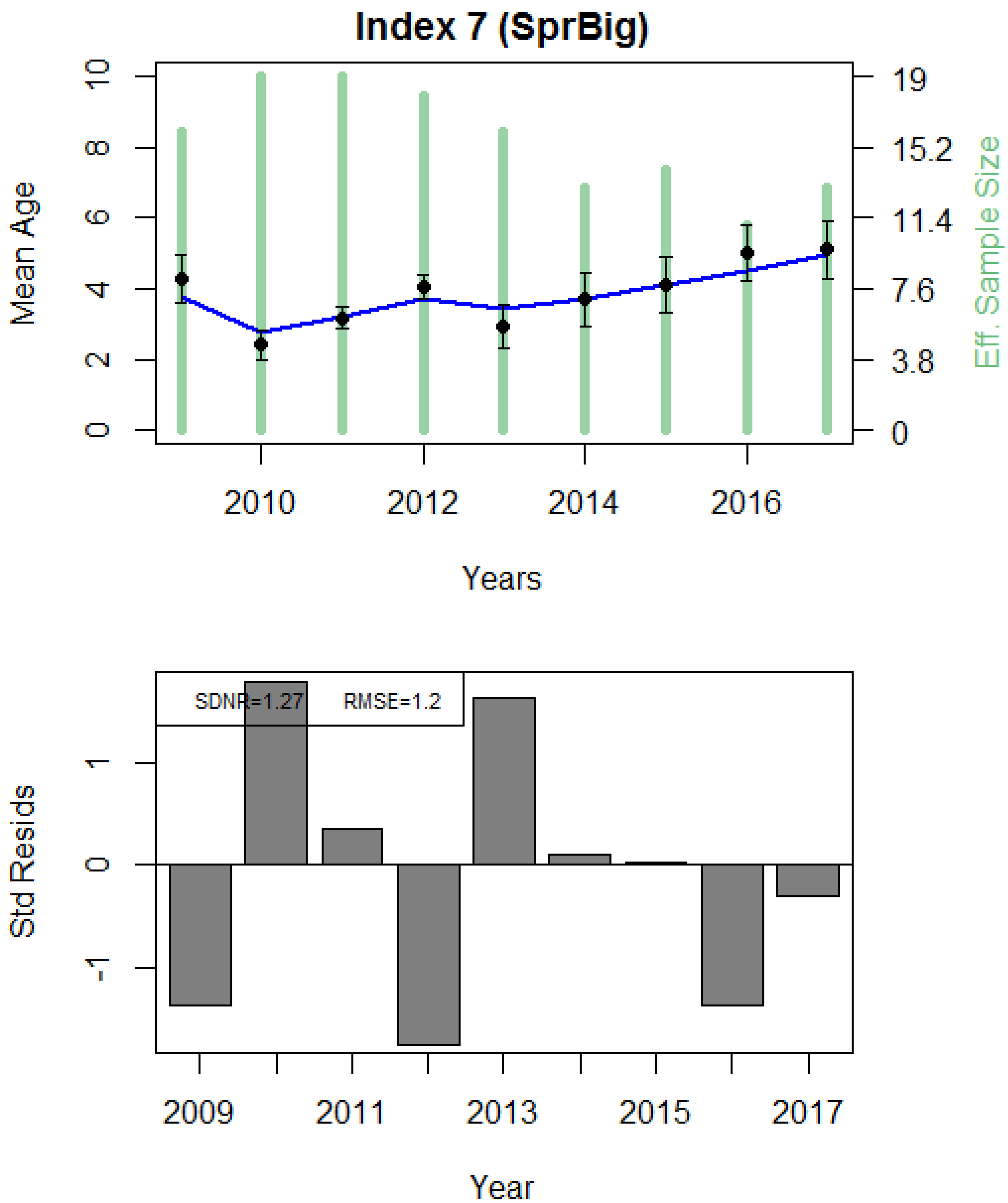
Index 5 (SprAlb85)
Mean Age
Eff. Sample Size
Years
0
2
4
6
8
10
0
3
6
9
12
15
1990
1995
2000
2005



SDNR=1.19
RMSE=1.19
Std Resids
Year
-1
0
1
2
3
1987
1991
1995
1999
2003
2007


Index 6 (FallAlb85)
Mean Age
Eff. Sample Size
Years
0
2
4
6
8
10
0
5.4
10.8
16.2
21.6
27
1990
1995
2000
2005


SDNR=1.13
RMSE=1.11
Std Resids
Year
-2
-1
0
1
2
1987
1991
1995
1999
2003
2007


Index 7 (SprBig)
Mean Age
0
2
4
6
8
10
19
15.2
11.4
7.6
3.8
0
Eff. Sample Size
2010
2012
2014
2016
Years
SDNR=1.27
RMSE=1.2
Std Resids
-1
0
1
2009
2011
2013
2015
2017
Year

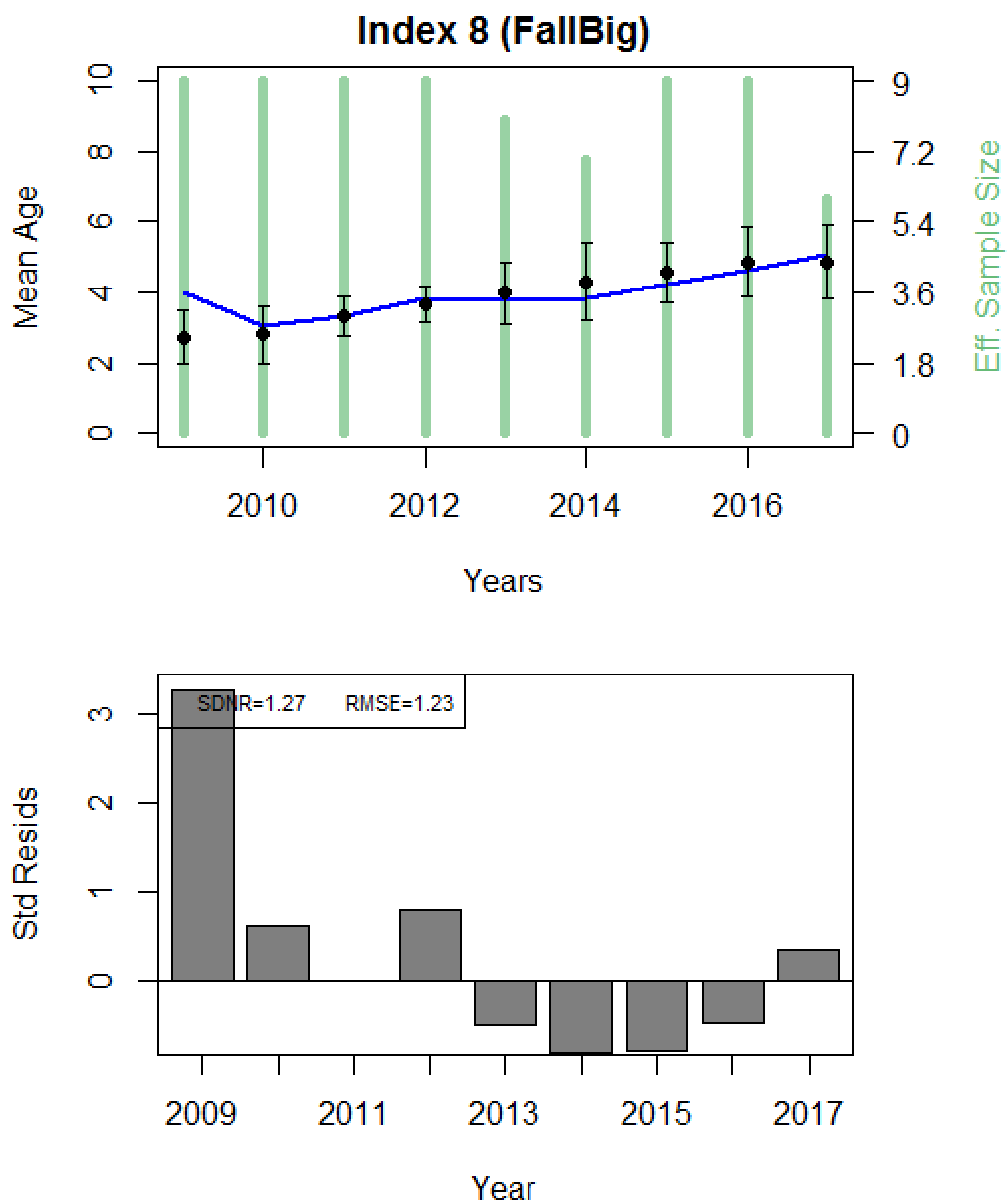
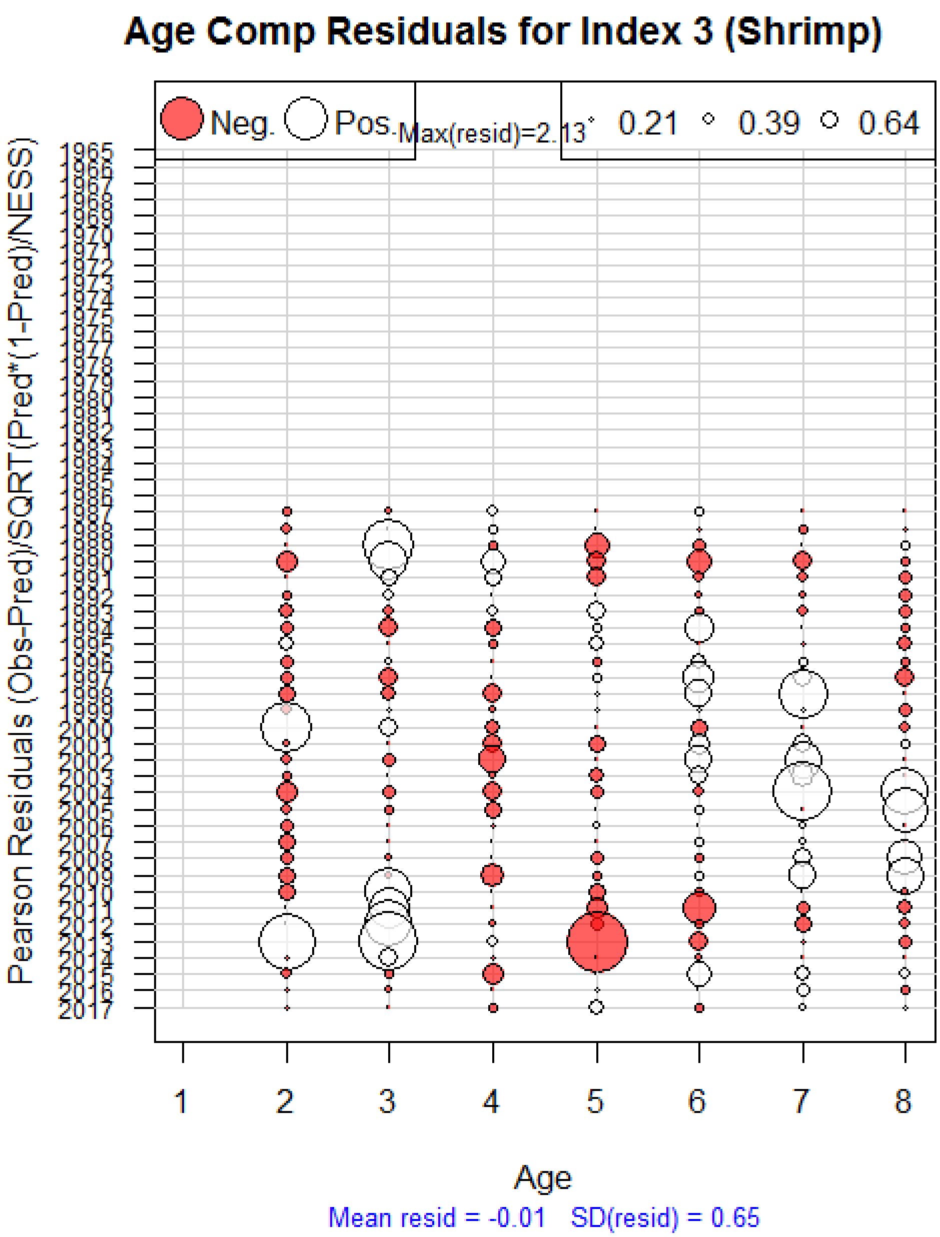
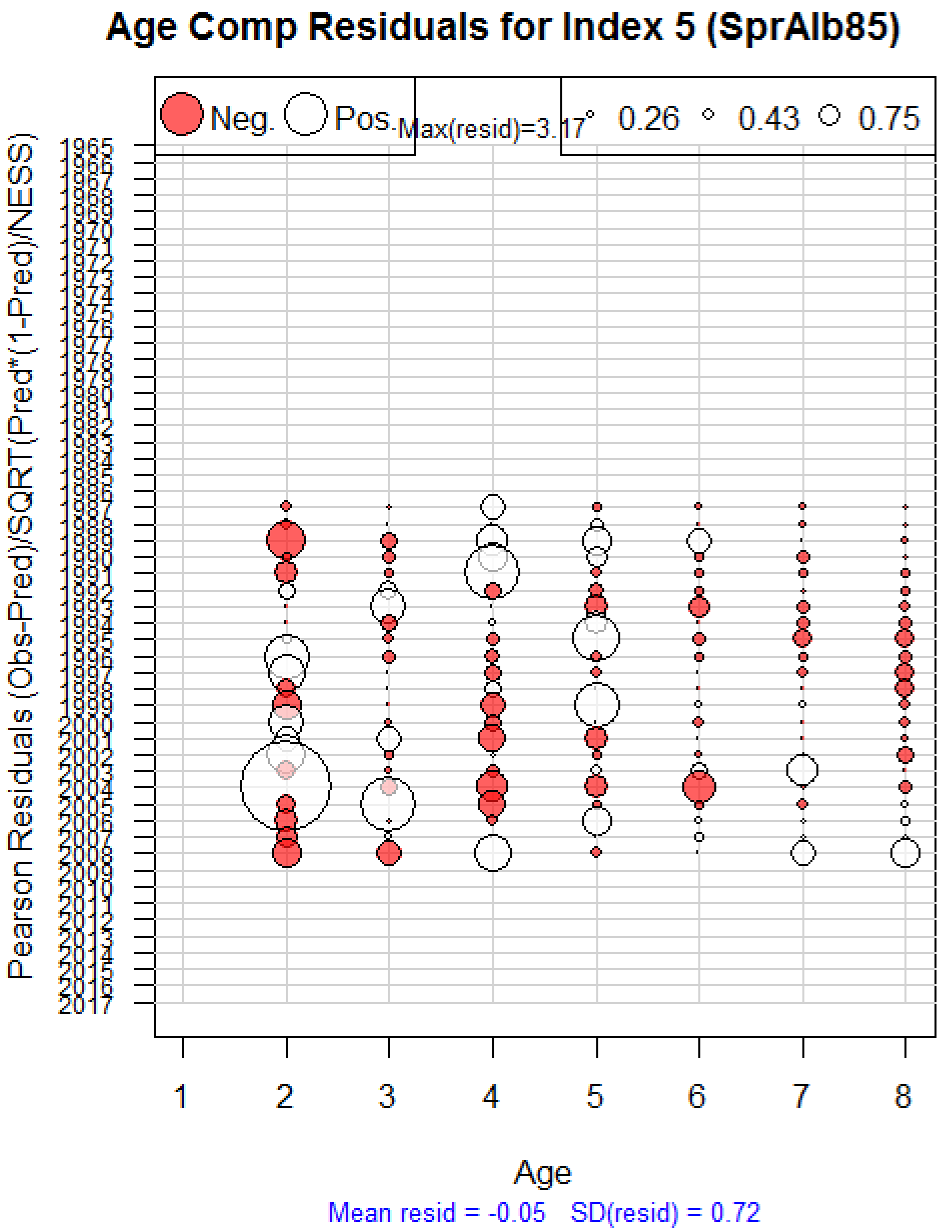
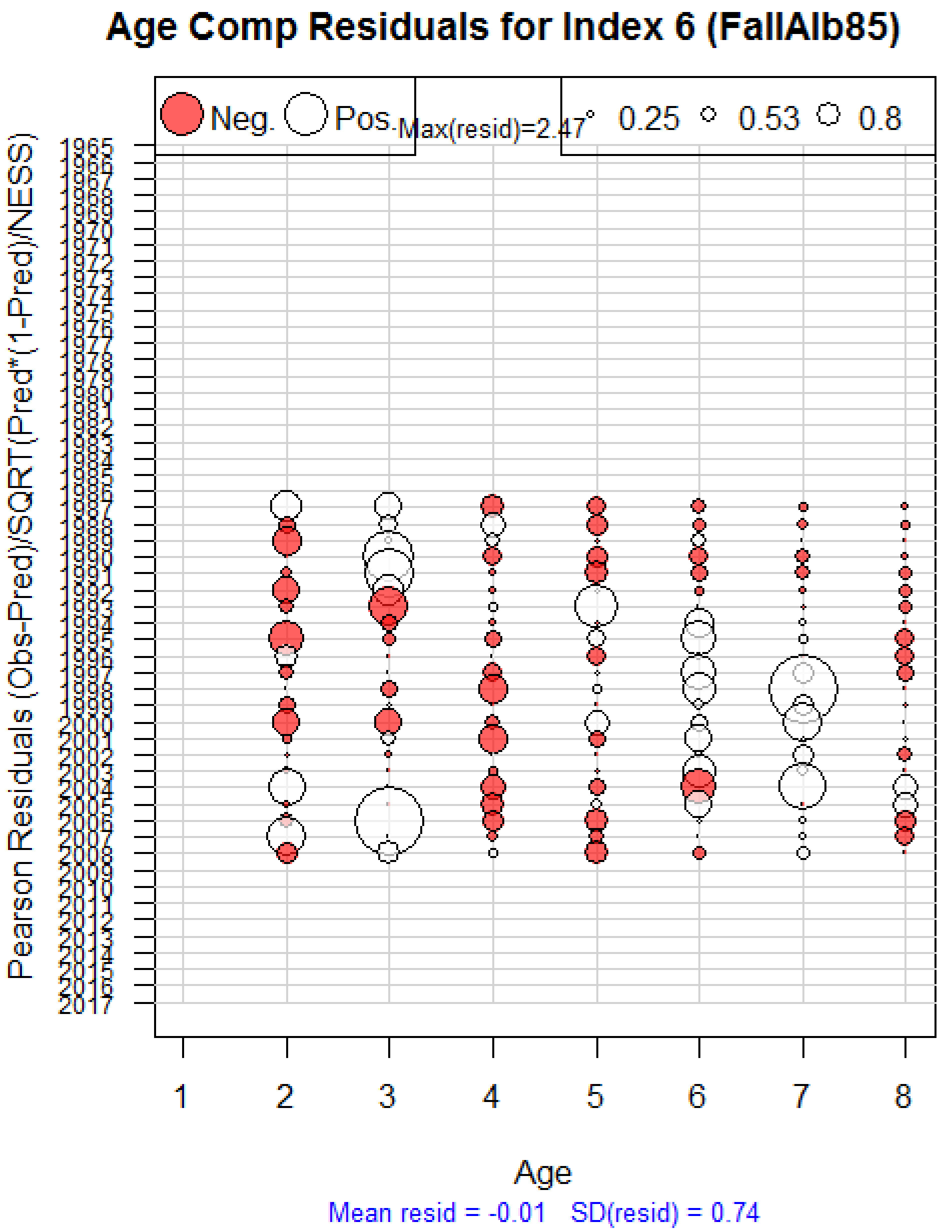

Index 8 (FallBig)
Mean Age
Eff. Sample Size
Years
SDNR=1.27 RMSE=1.23
Std Resids
Year

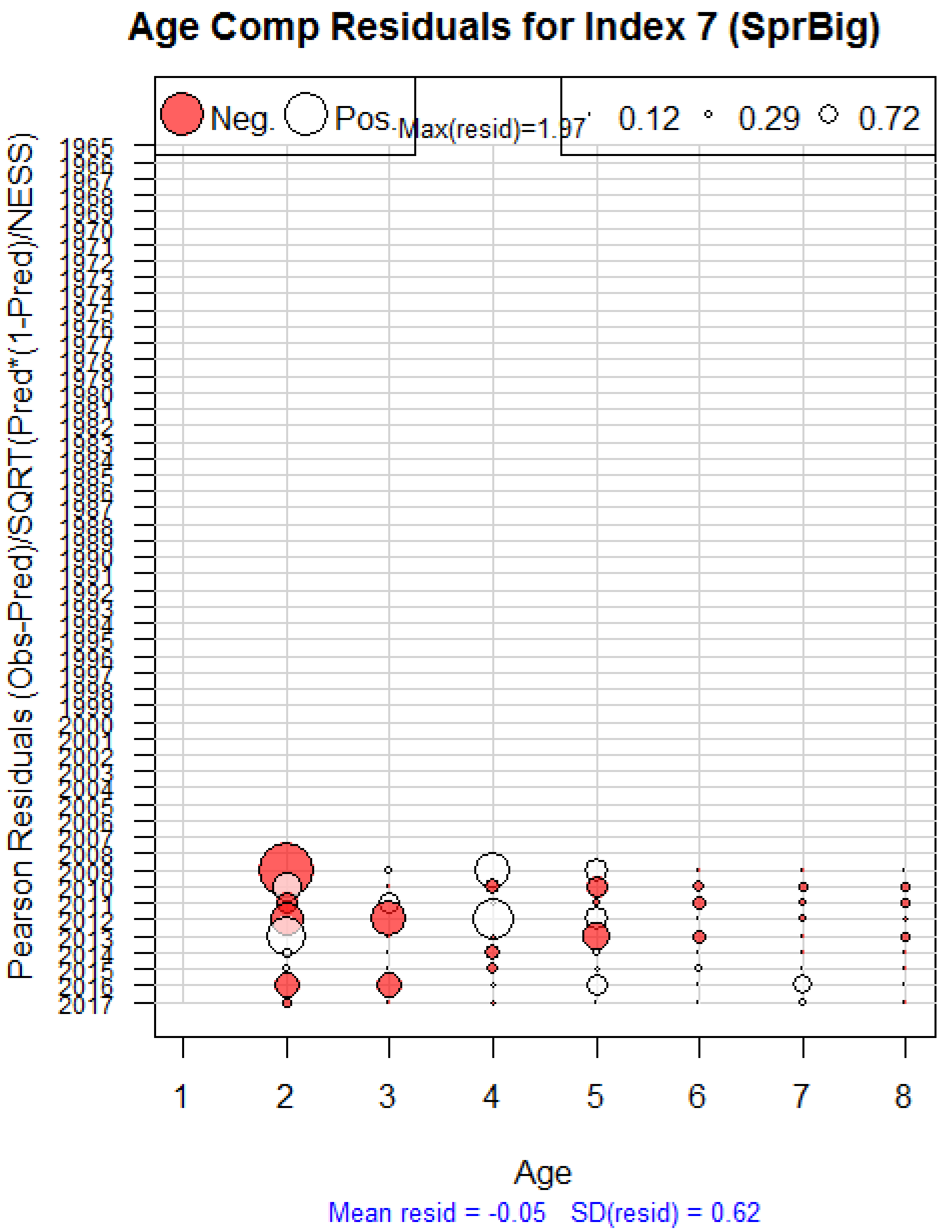
Figure B4- 16 Fits to Atlantic herring age compositions for the base ASAP model (Shrimp: NMFS summer/shrimp; Spr85: Albatross 1985-2008; Fall85: Albatross 1985-2008; SprBig: Bigelow 2009-2017; FallBig: Bigelow 2009-2017)





Age Comp Residuals for Index 5 (SprAlb85)
Neg.
Pos.
Max(resid)=3.17
0.26
0.43
0.75
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
1965
2017
1
2
3
4
5
6
7
8
Age
Mean resid = -0.05 SD(resid) = 0.72


Age Comp Residuals for Index 6 (FallAlb85)
Neg.
Pos.
Max(resid)=2.47
0.25
0.53
0.8
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
Age
Mean resid = -0.01 SD(resid) = 0.74



Age Comp Residuals for Index 7 (SprBig)
Neg.
Pos.
Max(resid)=1.97
0.12
0.29
0.72
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
Age
Mean resid = -0.05 SD(resid) = 0.62

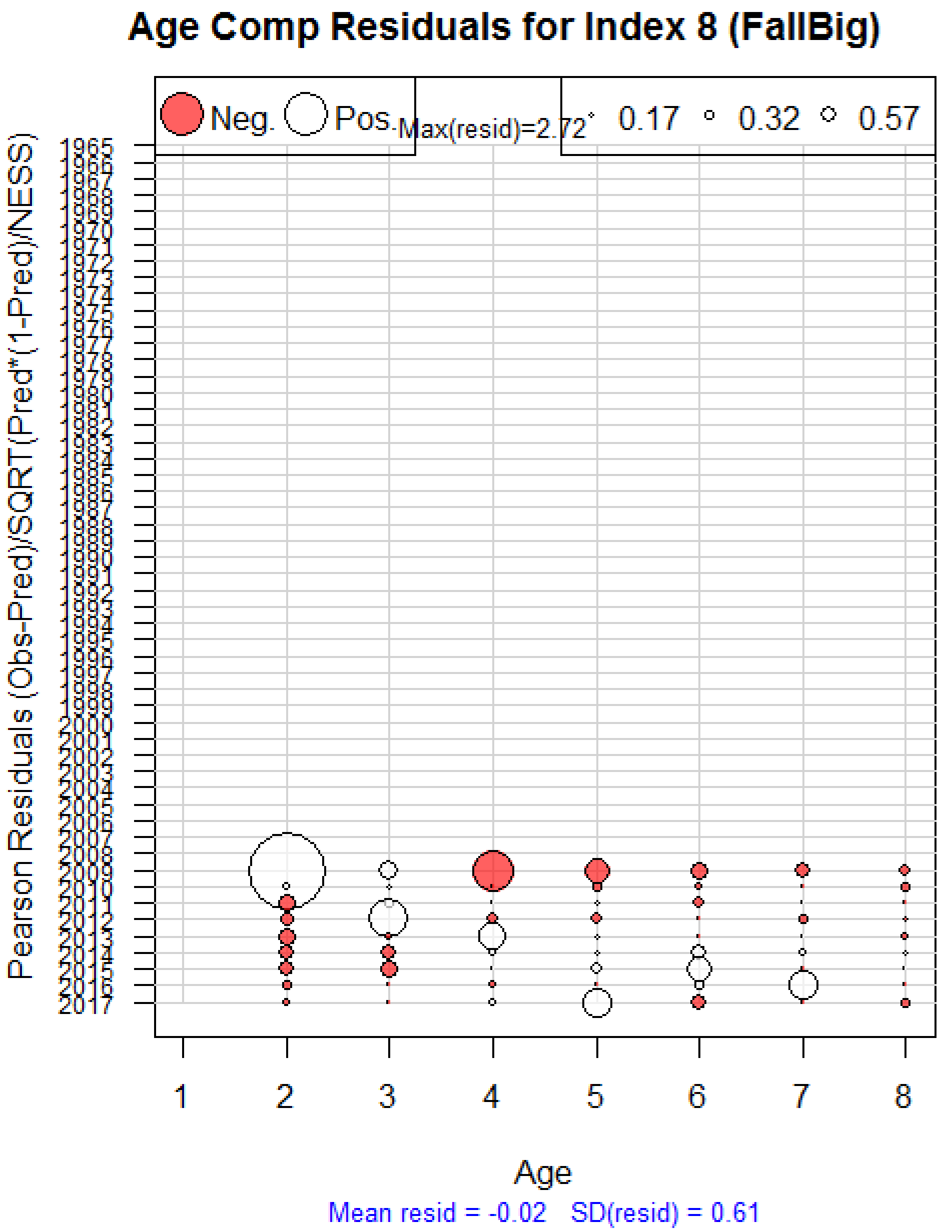

Age Comp Residuals for Index 8 (FallBig)
Neg.
Pos.
Max(resid)=2.72
0.17
0.32
0.57
Pearson Residuals (Obs-Pred)/SQRT(Pred*(1-Pred)/NESS)
Age
Mean resid = -0.02   SD(resid) = 0.61

Figure B4- 17 Atlantic herring catchability estimates for the indices from the base ASAP model

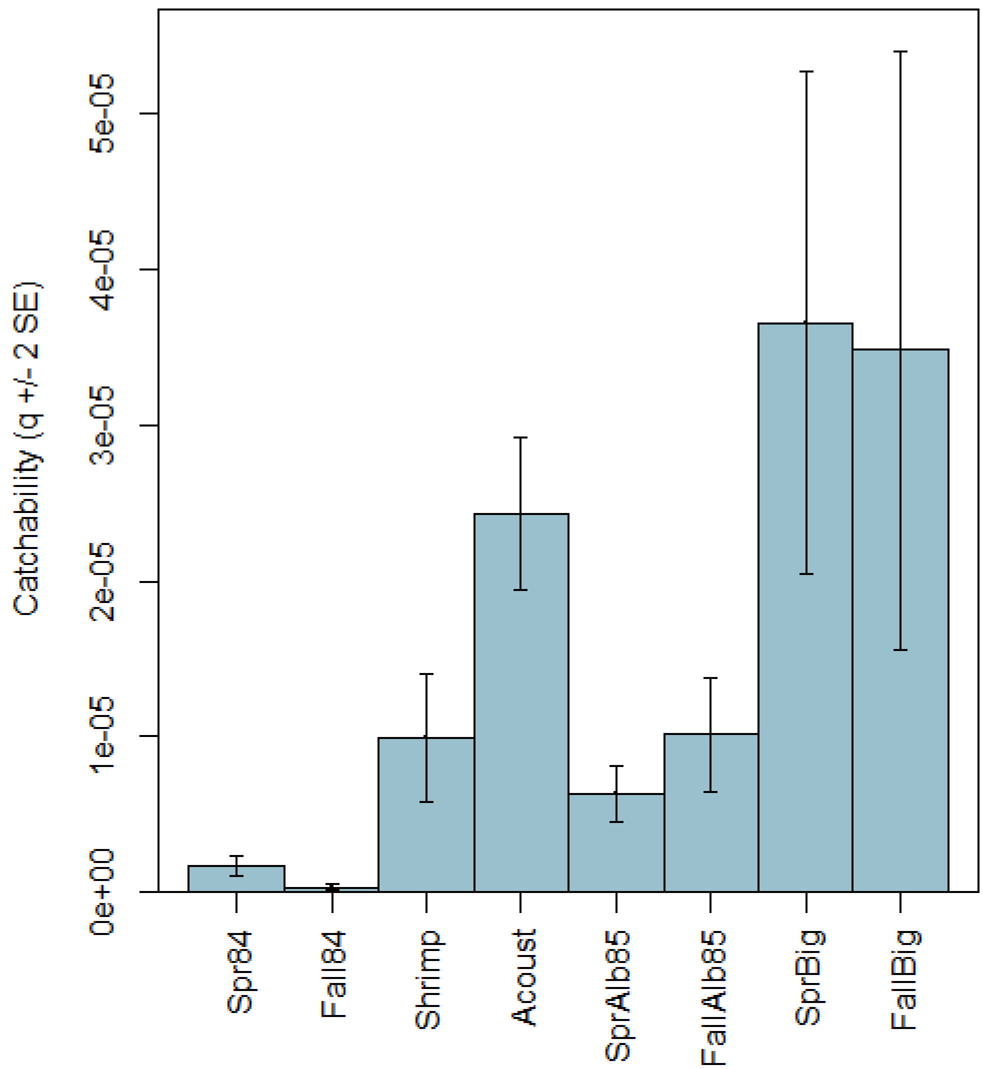


Figure B4- 18 Relative retrospective pattern for Atlantic herring SSB using a 17-year peel for the base ASAP model

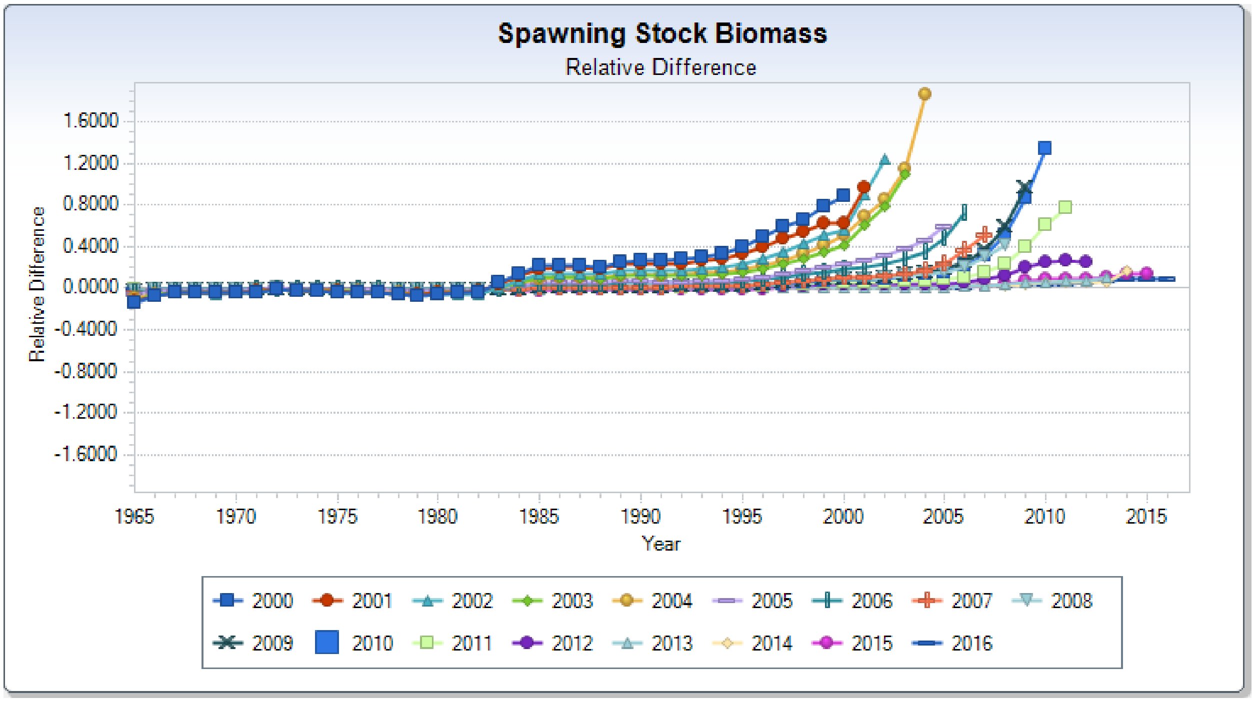


Figure B4- 19 Coefficients of variation of the time series estimates of Atlantic herring recruitment (Recr.), SSB, and F (average F over ages 7 and8) from the base ASAP model

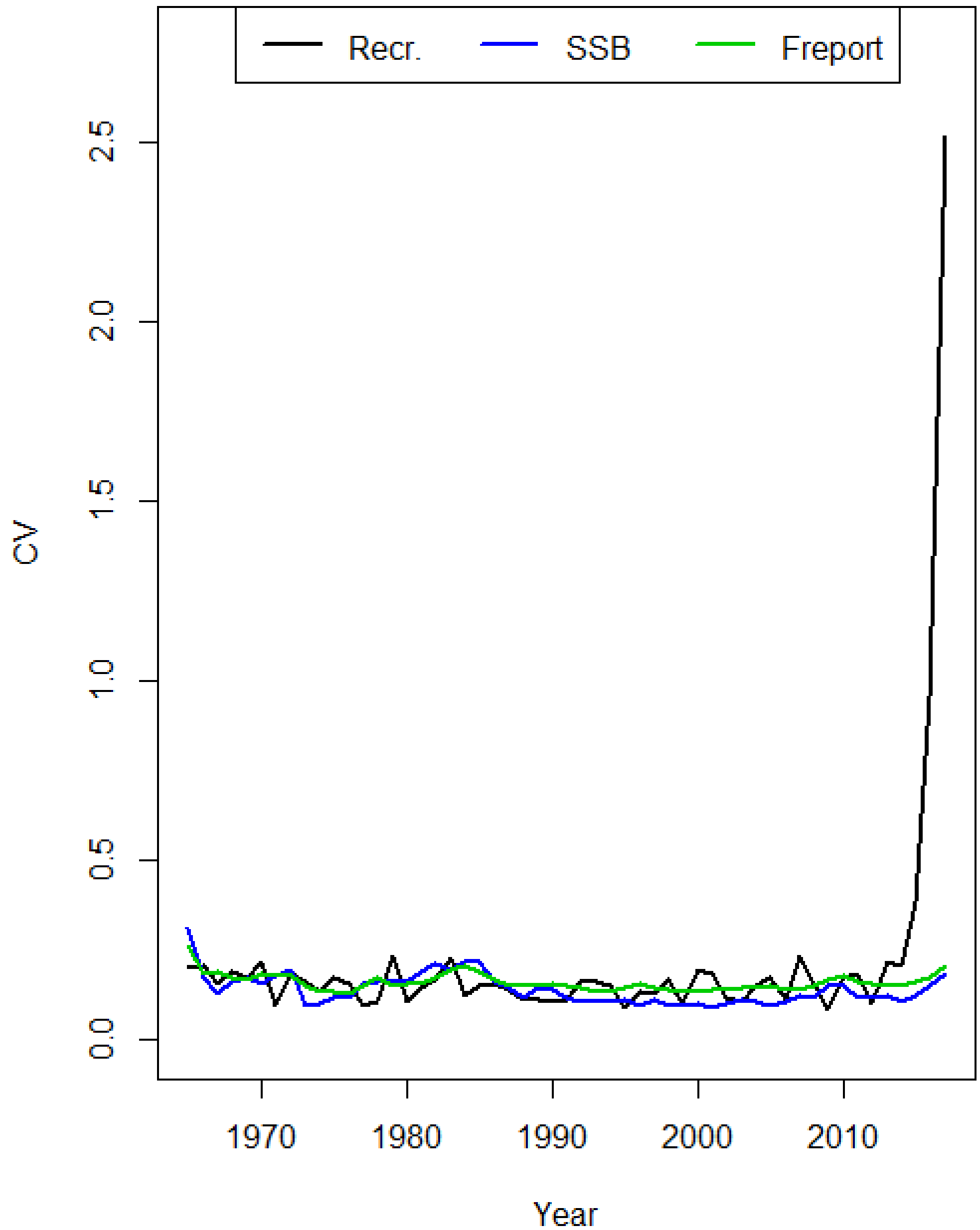


Figure B4- 20 Atlantic herring biomass time series from base ASAP model

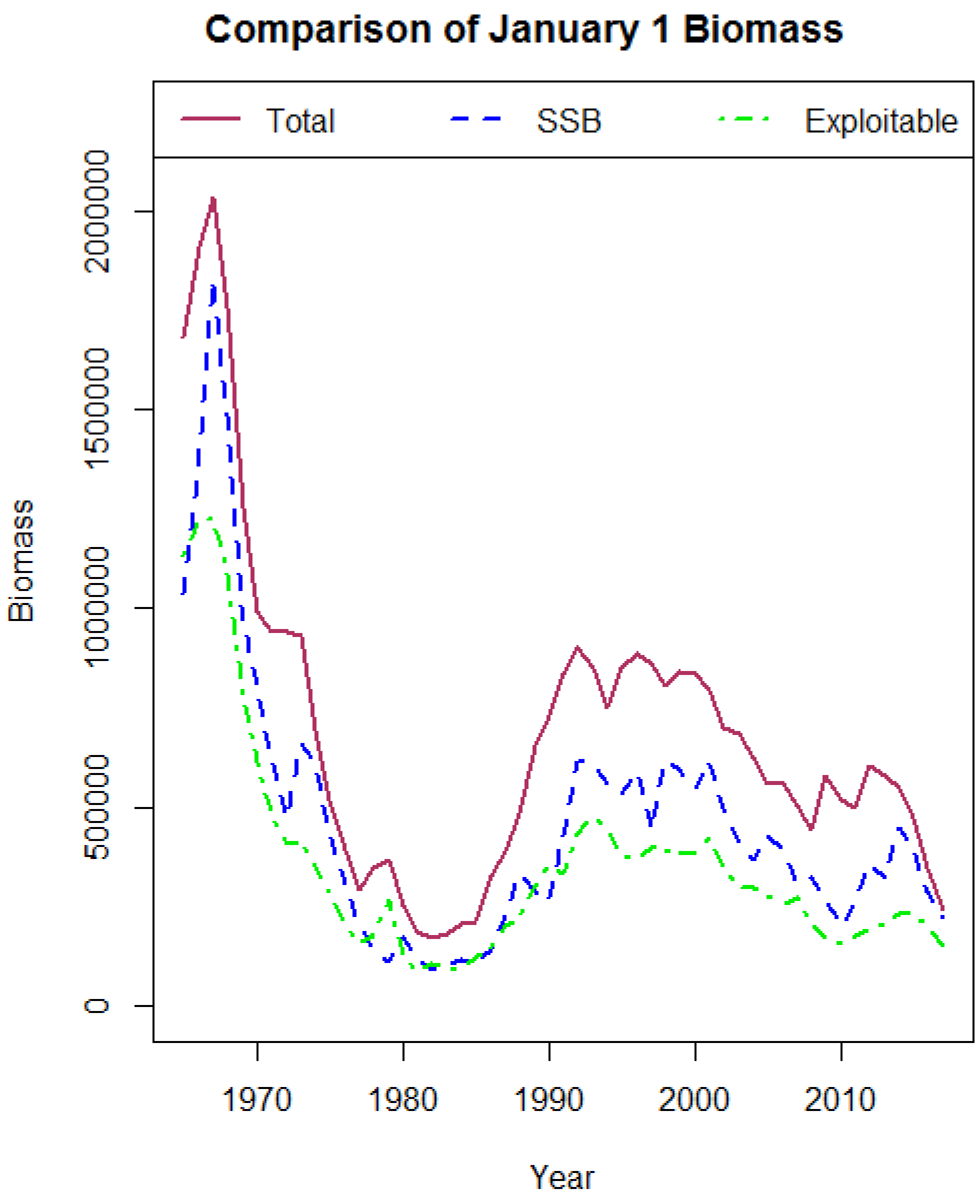


Figure B4- 21 Atlantic herring recruit time series and log-scale deviations from the base ASAP model

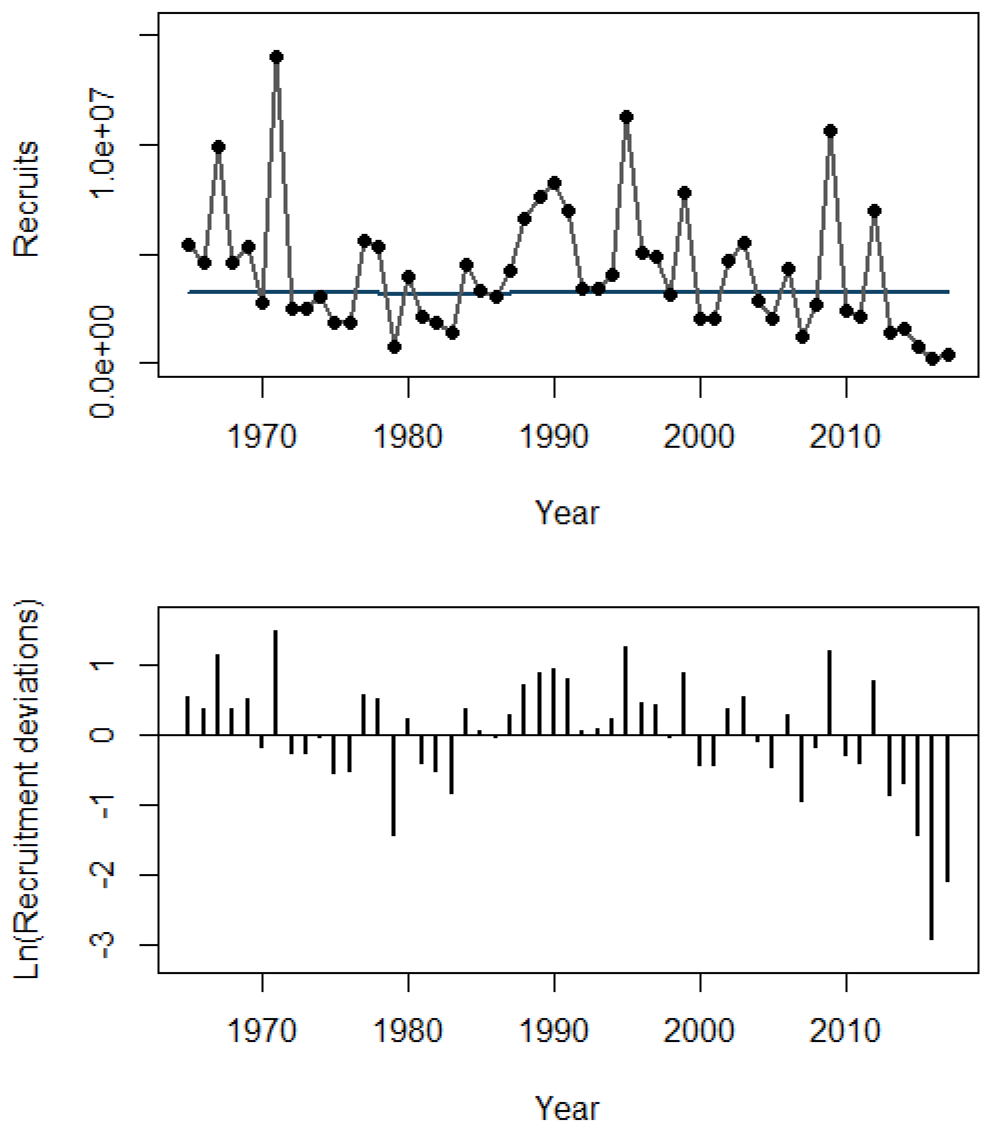


Figure B4- 22 Atlantic herring stock-recruit plot with year of recruitment as points from the base ASAP model

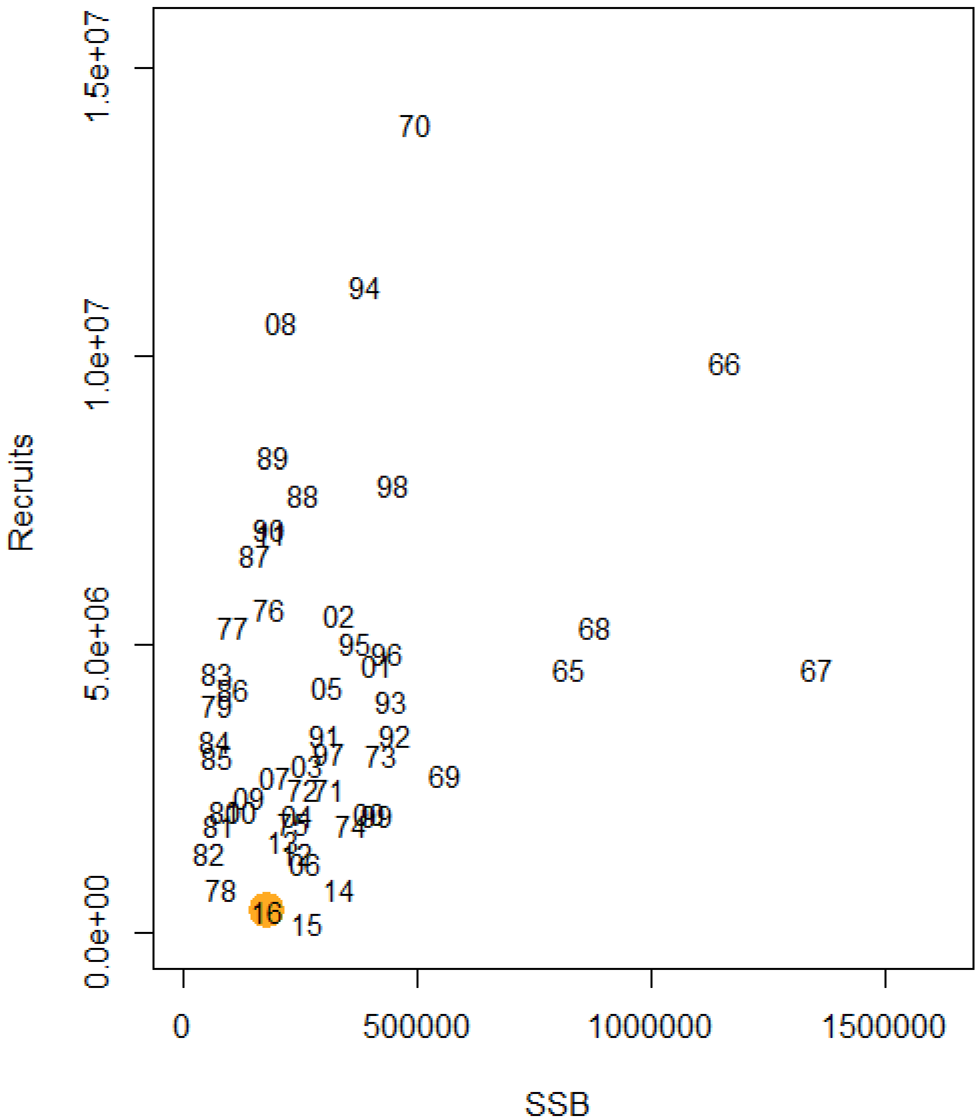


Figure B4- 23 Atlantic herring SSB, fully selected F (F.full) and average F over ages 7-8 (F.report) from the base ASAP model

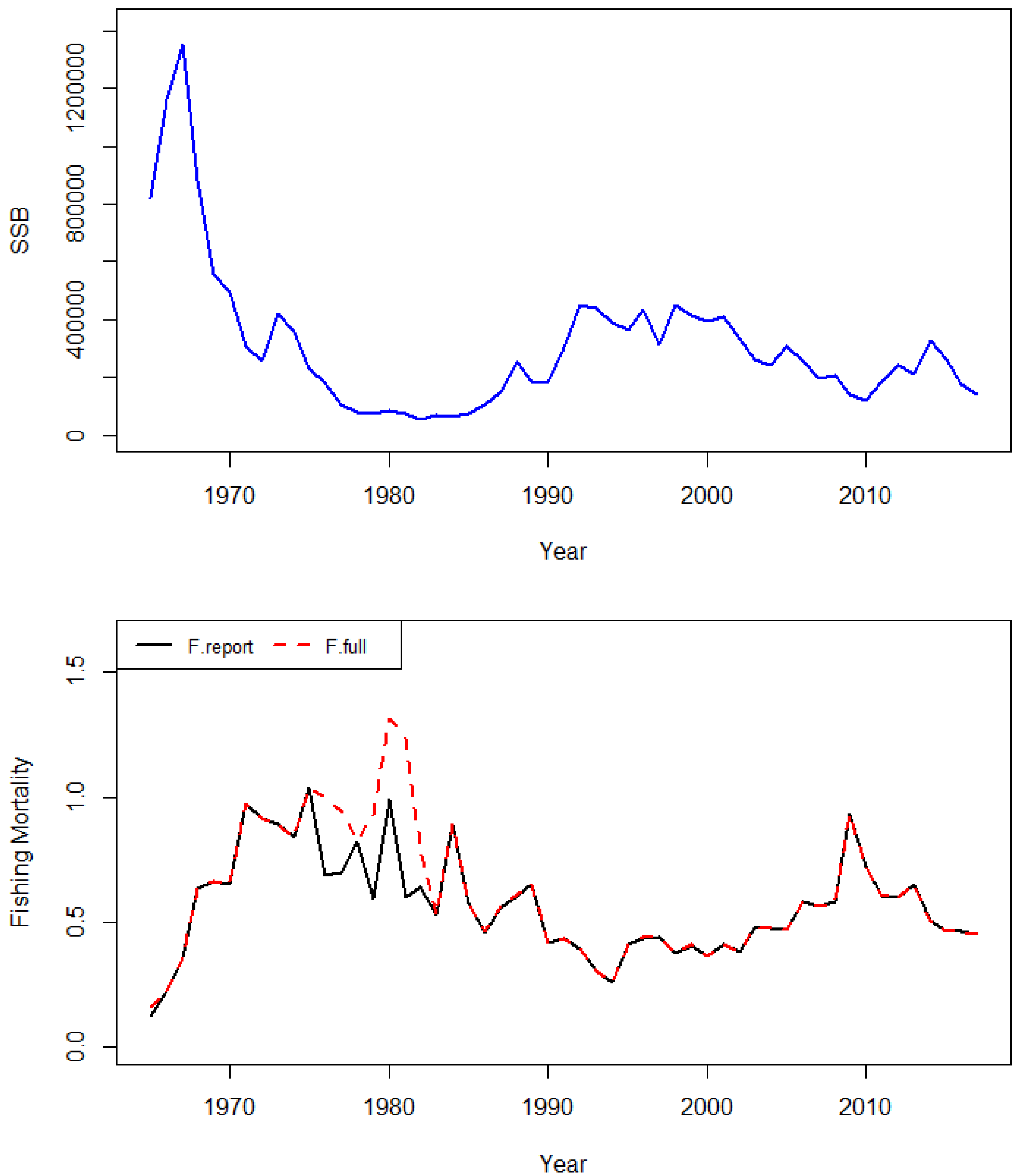
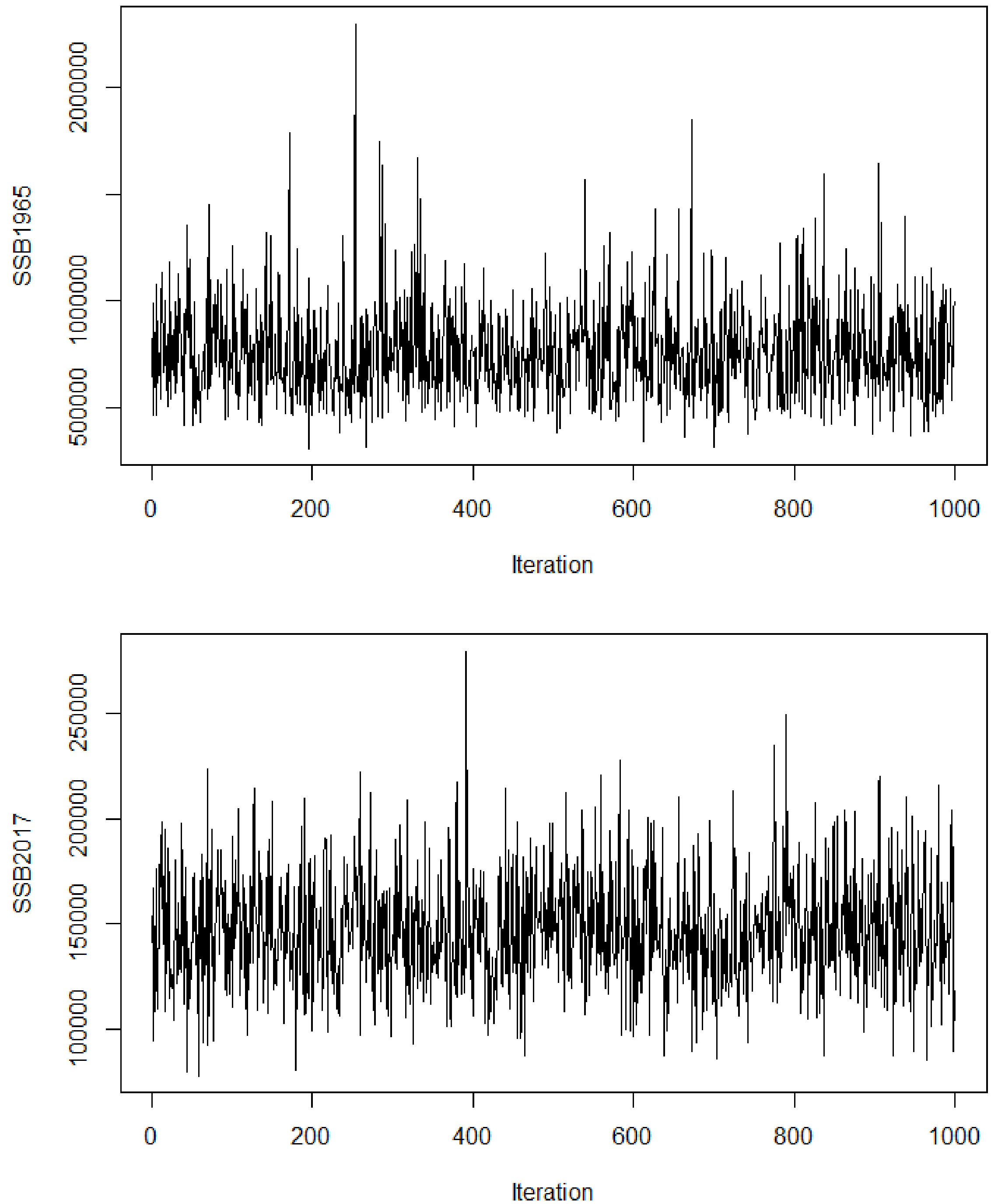
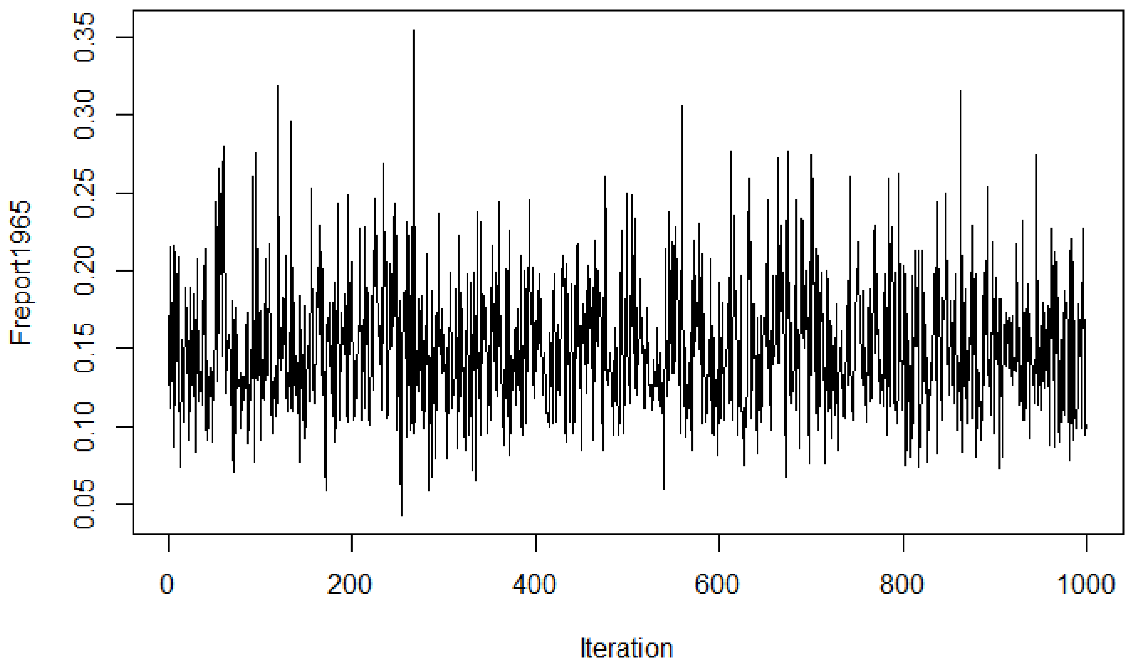


Figure B4- 24 Trace plots for SSB and F in 1965 and 2017 from MCMC of the base ASAP model




Freport1965
0.05
0.10
0.15
0.20
0.25
0.30
0.35
0
200
400
600
800
1000
Iteration

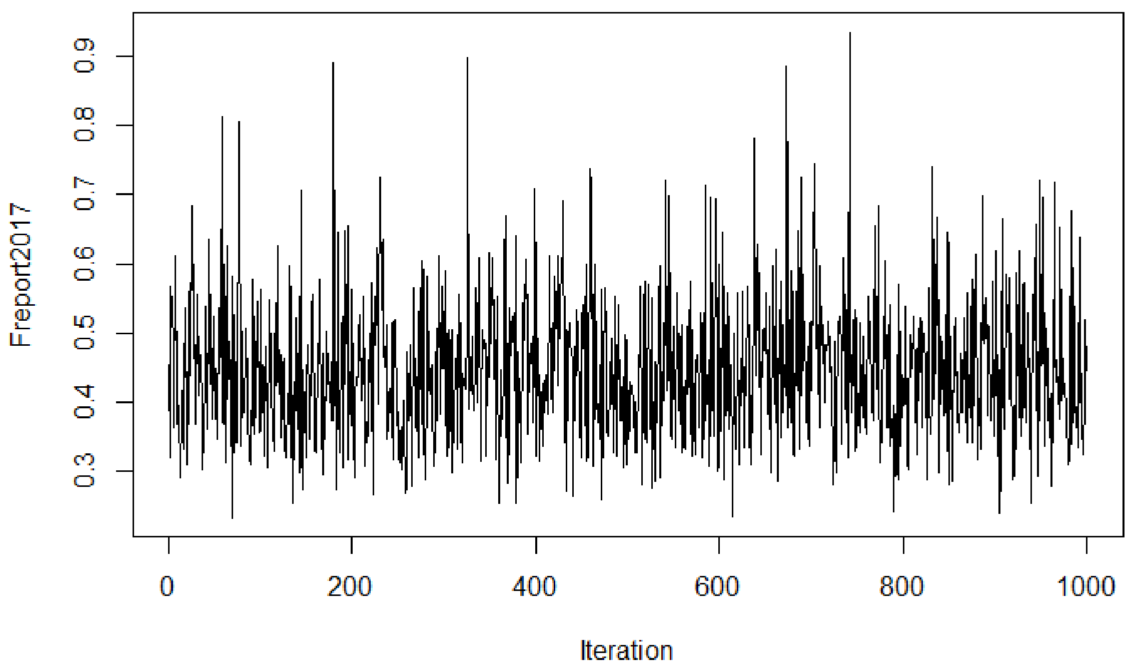
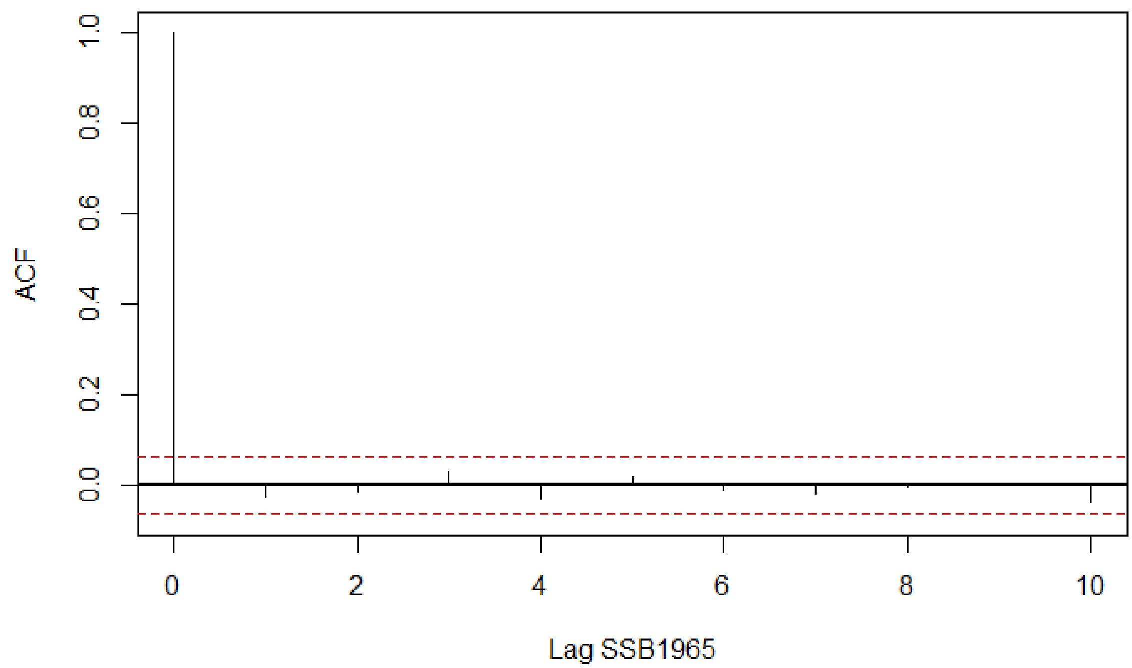
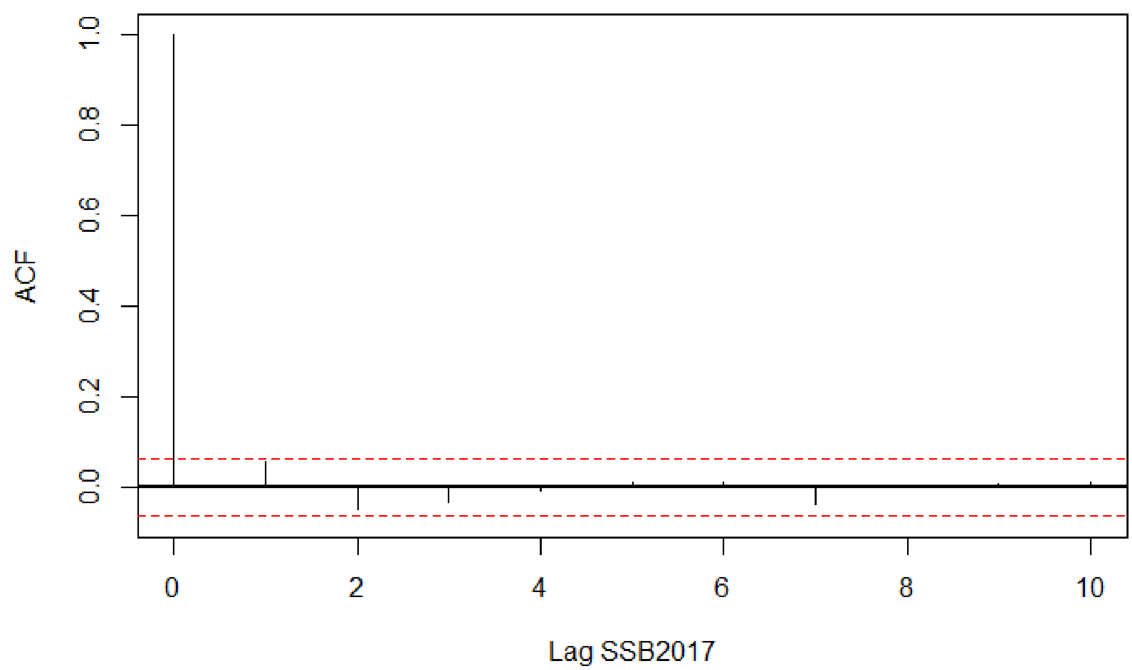
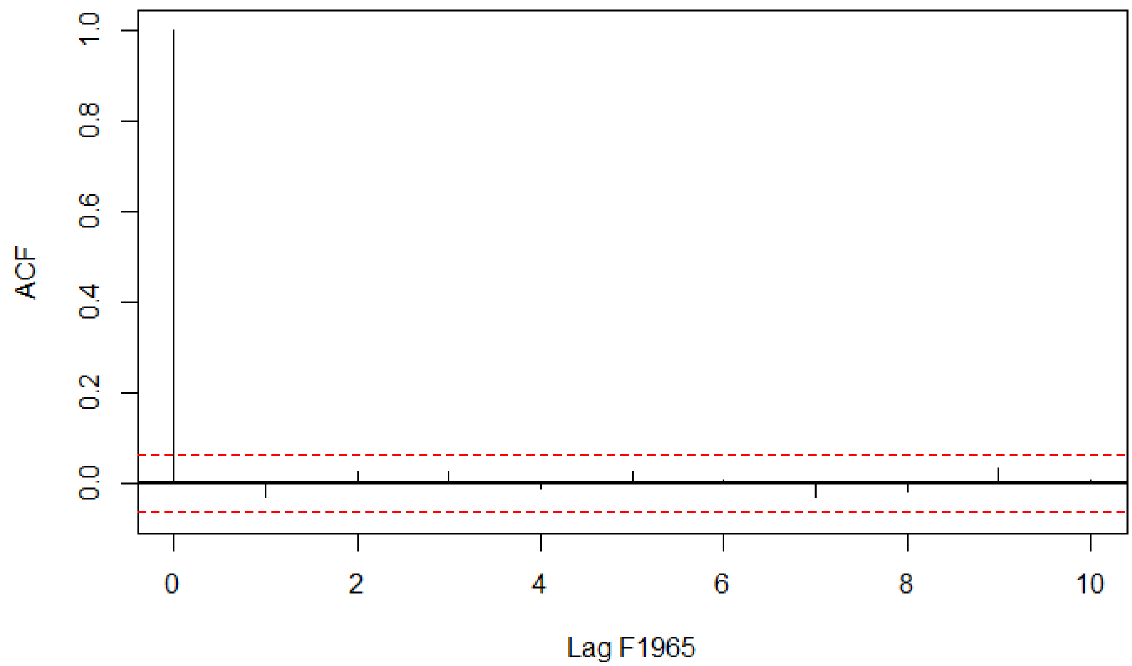

Freport2017
0.3
0.4
0.5
0.6
0.7
0.8
0.9
0
200
400
600
800
1000
Iteration

Figure B4- 25 Correlations of lags for SSB and F in 1965 and 2017 from the MCMC of the base ASAP model






ACF
1.0
0.8
0.6
0.4
0.2
0.0
0
2
4
6
8
10
Lag F1965

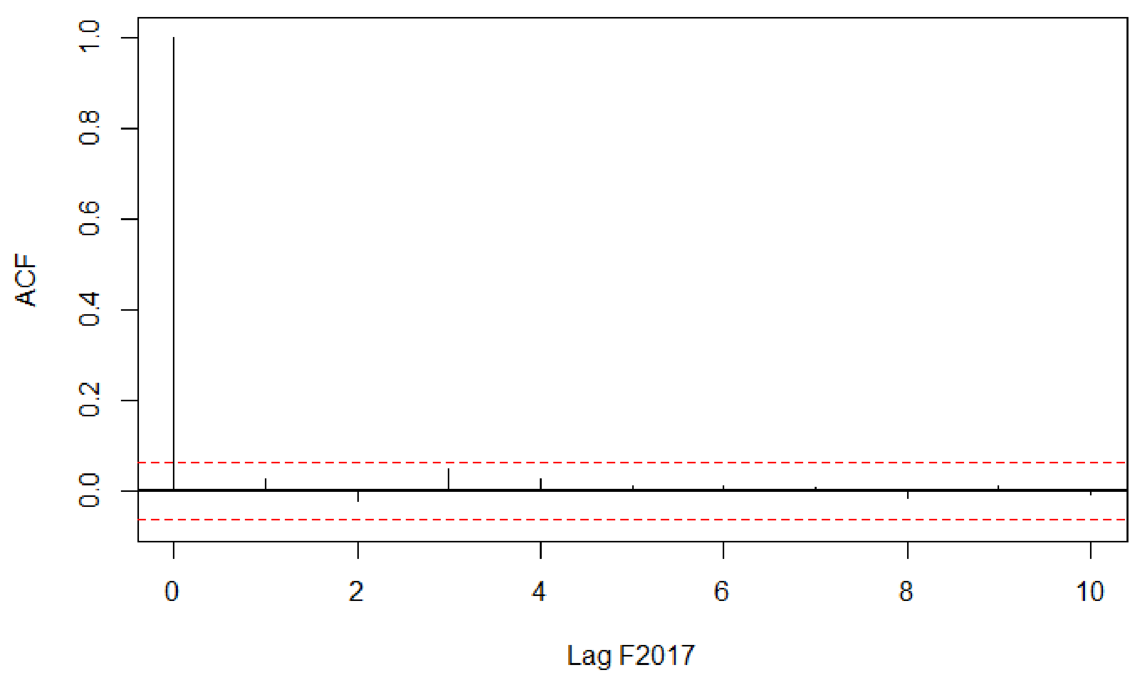
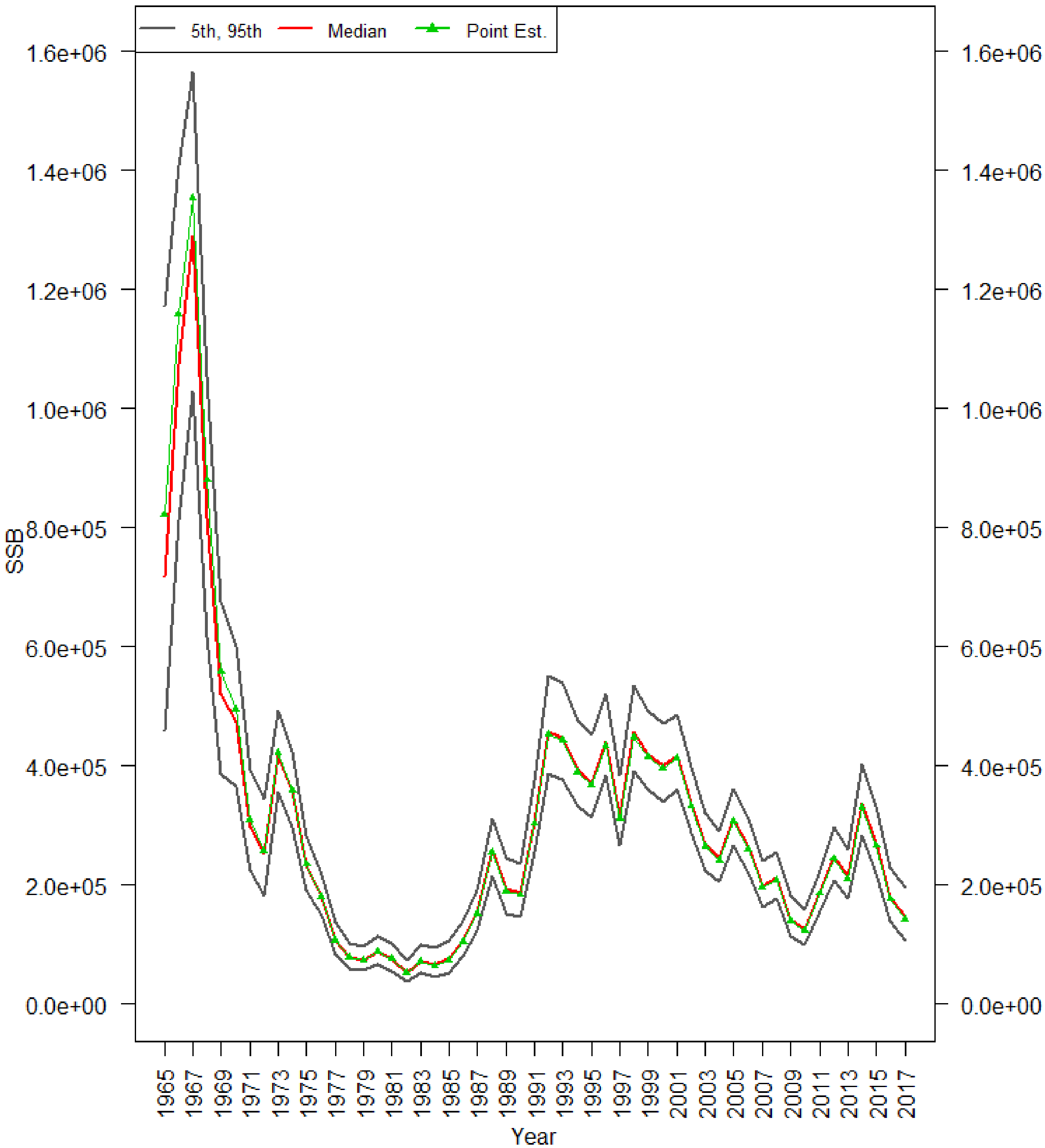
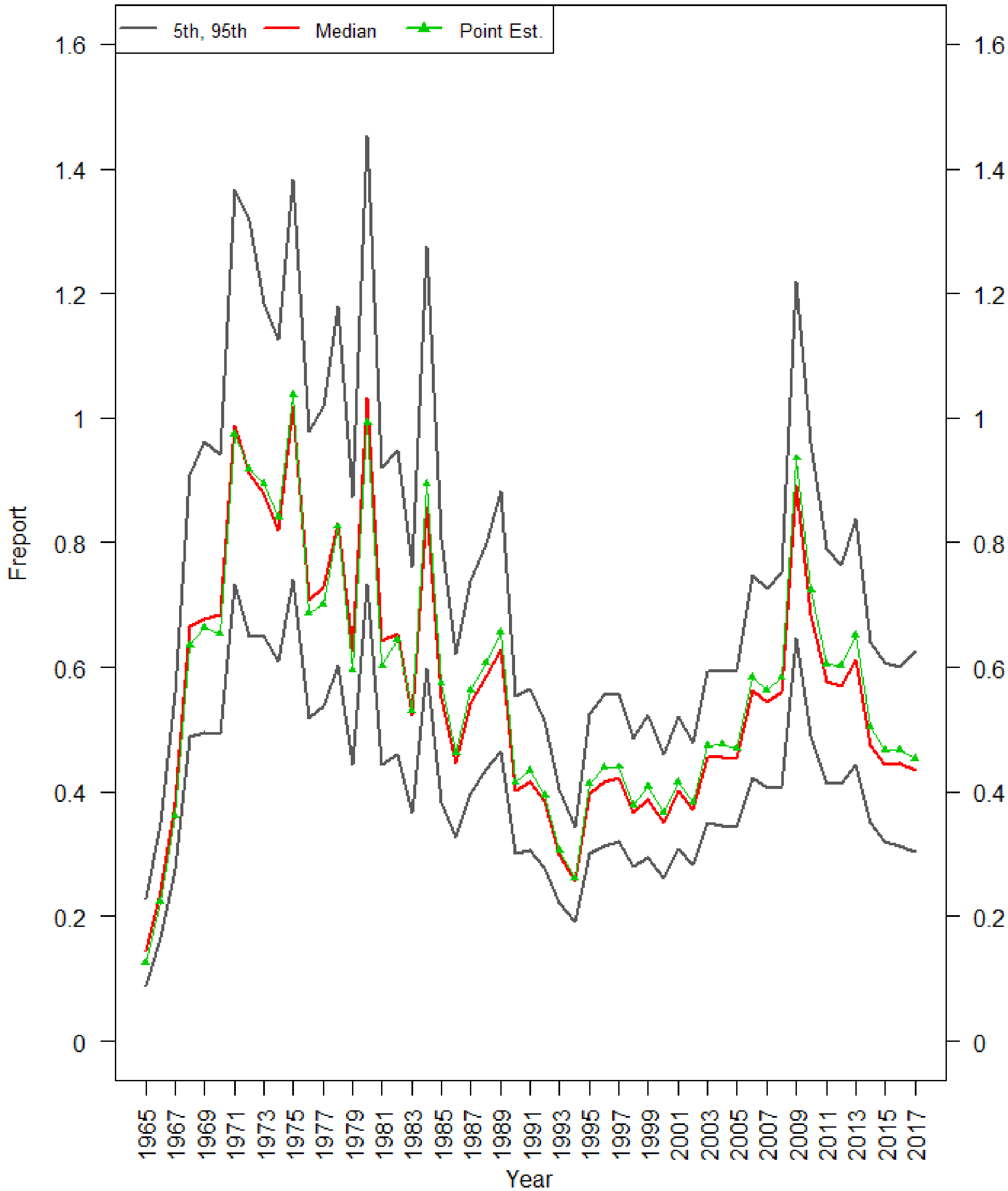

ACF
1.0
0.8
0.6
0.4
0.2
0.0
0
2
4
6
8
10
Lag F2017

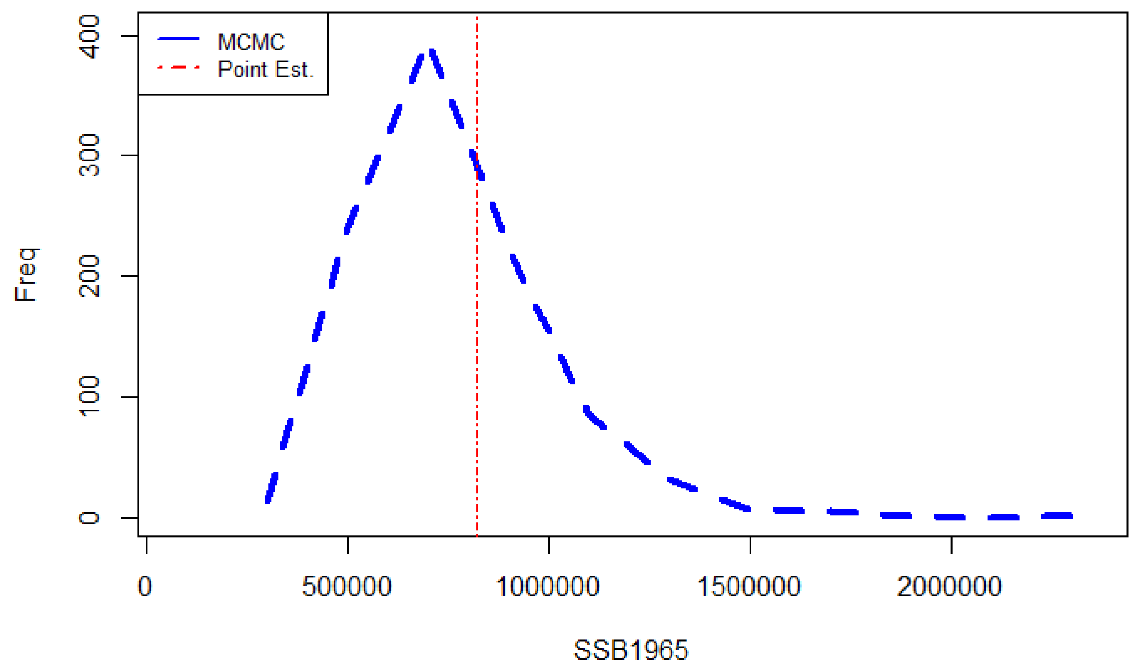
Figure B4- 26 Point estimates, median, and 90% probability intervals of Atlantic herring SSB and F from the MCMC of the base ASAP model

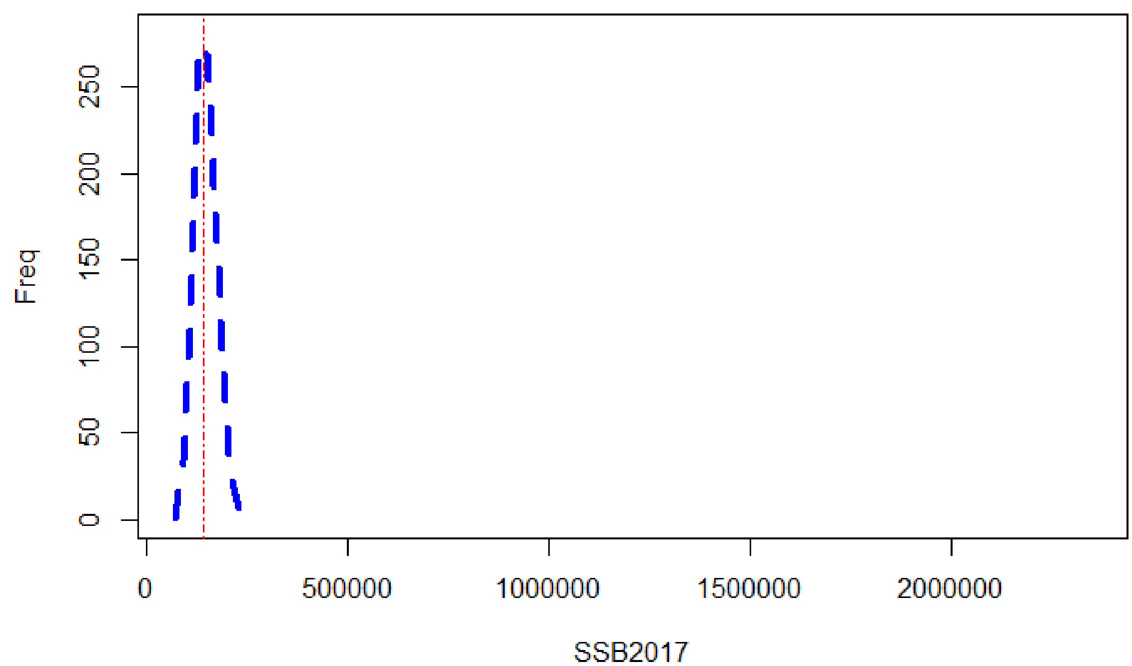


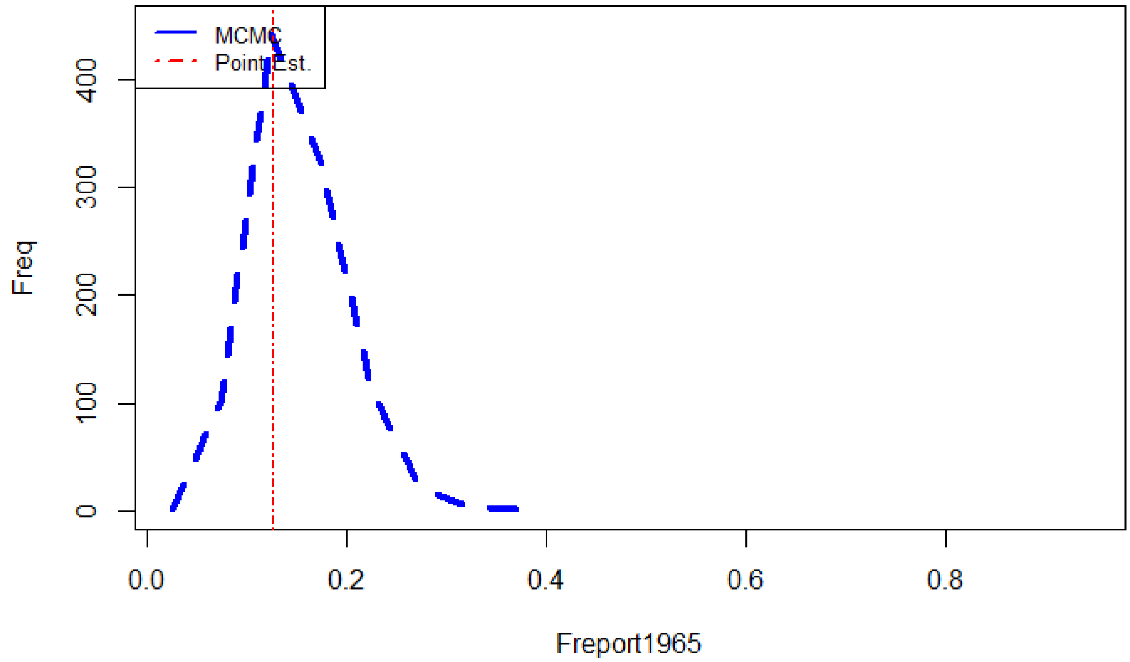


5th, 95th
Median
Point Est.
Freport
Year
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1965
1967
1969
1971
1973
1975
1977
1979
1981
1983
1985
1987
1989
1991
1993
1995
1997
1999
2001
2003
2005
2007
2009
2011
2013
2015
2017

Figure B4- 27 Posterior density of Atlantic herring SSB and F in 1965 and 2017 from the MCMC of the base ASAP model






MCMC
Point Est.
Freq
0
100
200
300
400
0.0
0.2
0.4
0.6
0.8
Freport1965

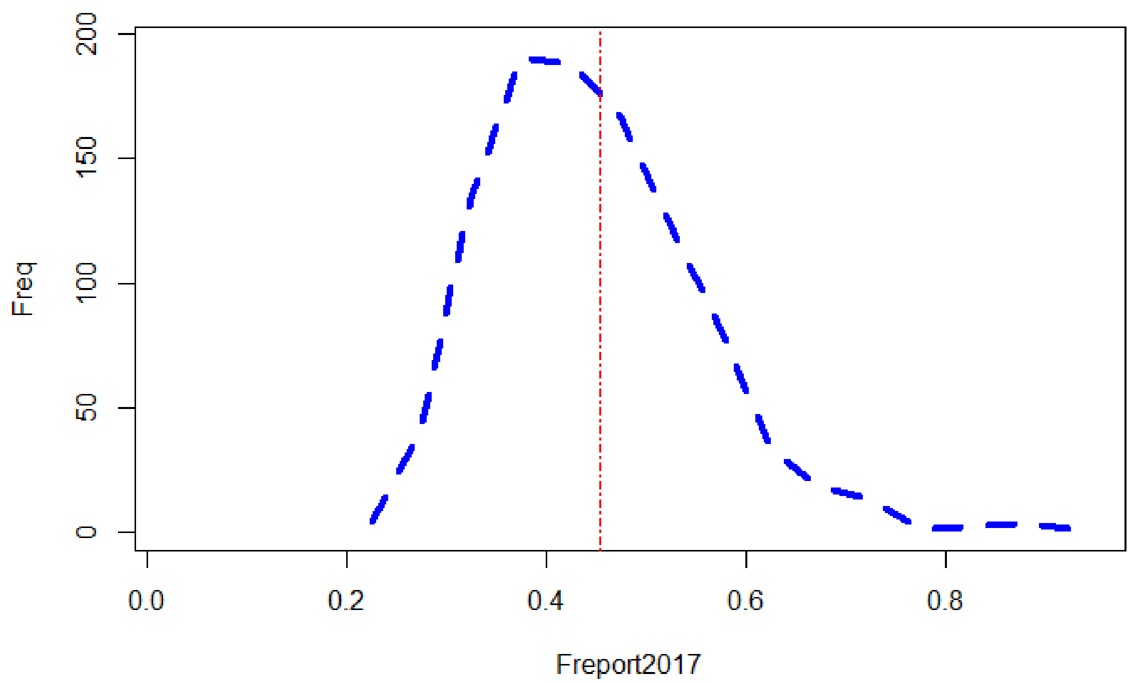

Freq
0
50
100
150
200
0.0
0.2
0.4
0.6
0.8
Freport2017

Figure B4- 28 Internal retrospective pattern for F, SSB, and recruitment for the base ASAP model

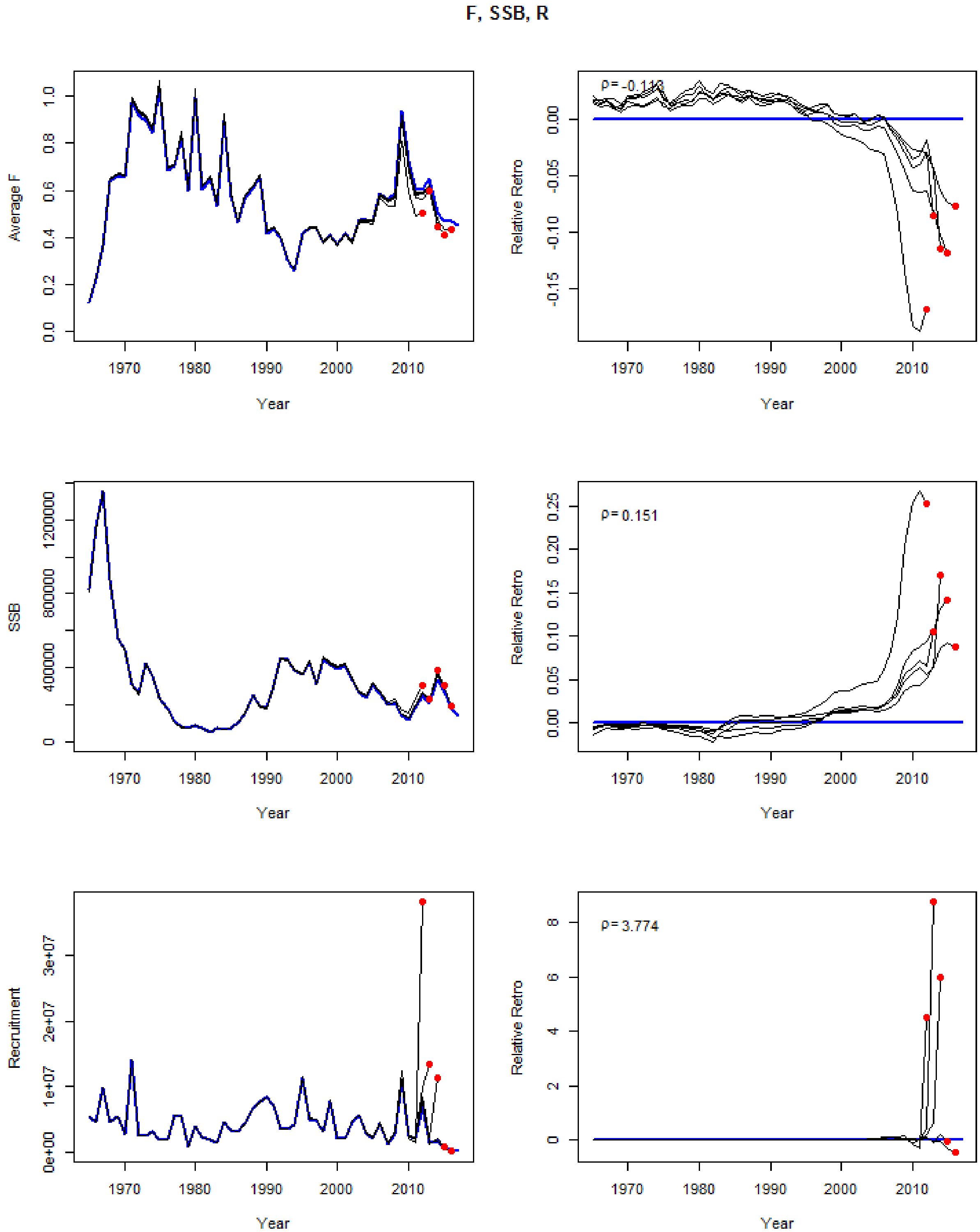
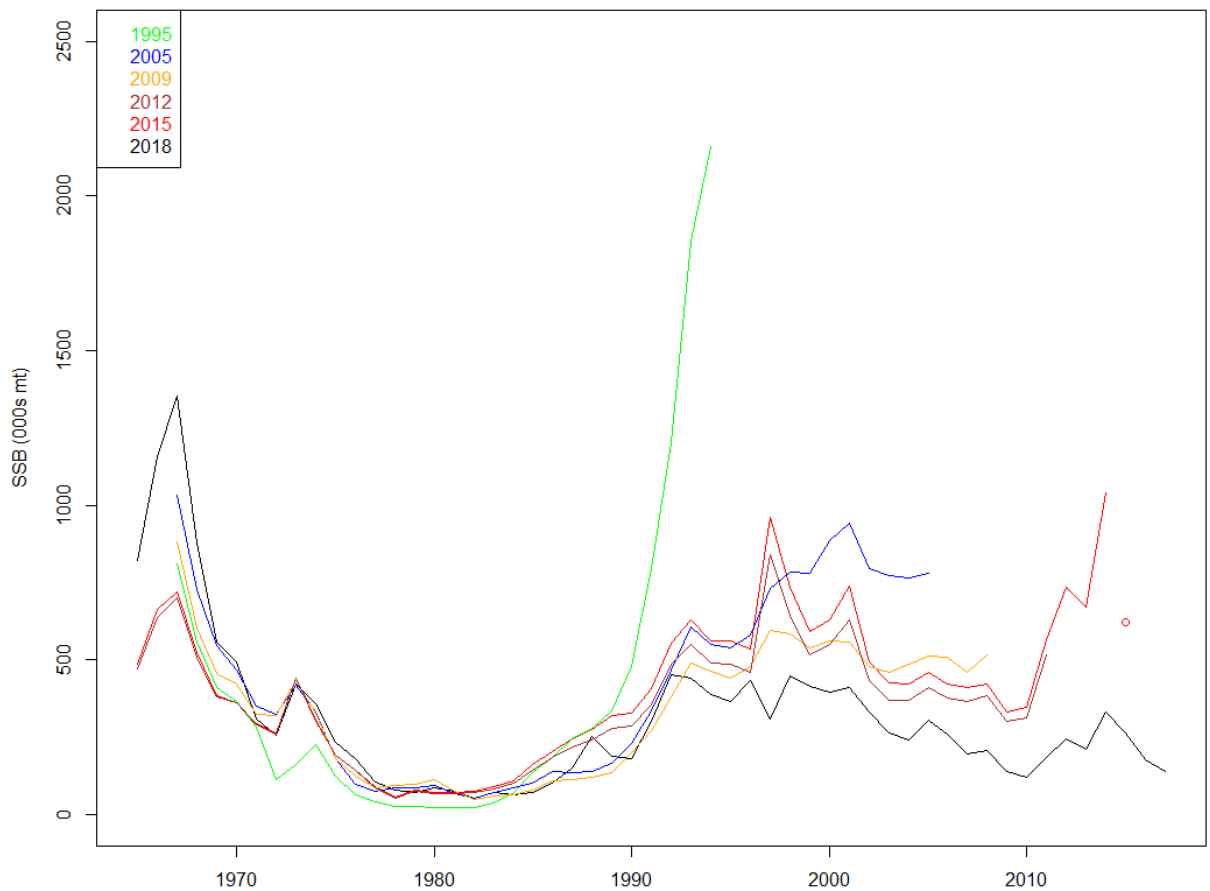
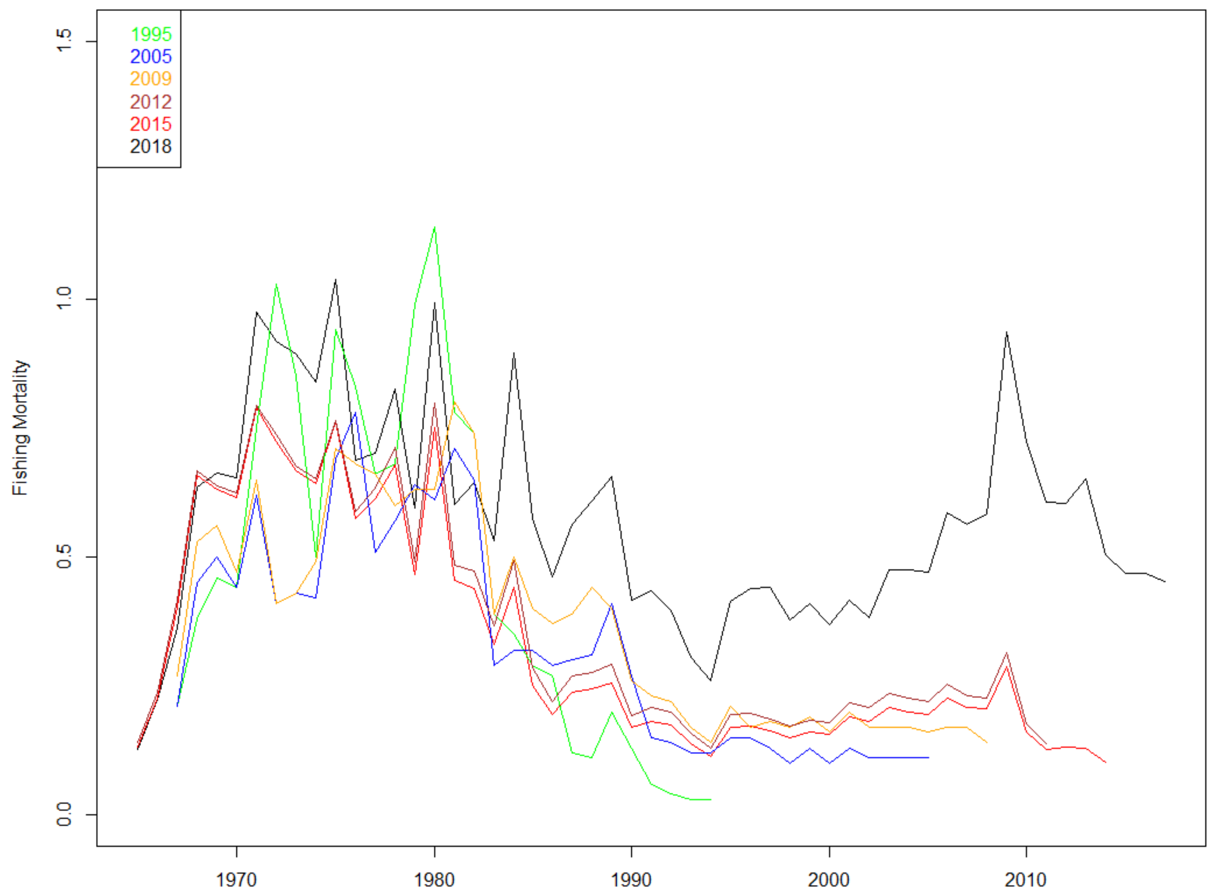


Figure B4- 29 Atlantic herring historic retrospective pattern for SSB, F (not directly comparable), and F rescaled by each time series mean to make the trends more readily comparable




1995
2005
2009
2012
2015
2018
Fishing Mortality
0.0
0.5
1.0
1.5
1970
1980
1990
2000
2010

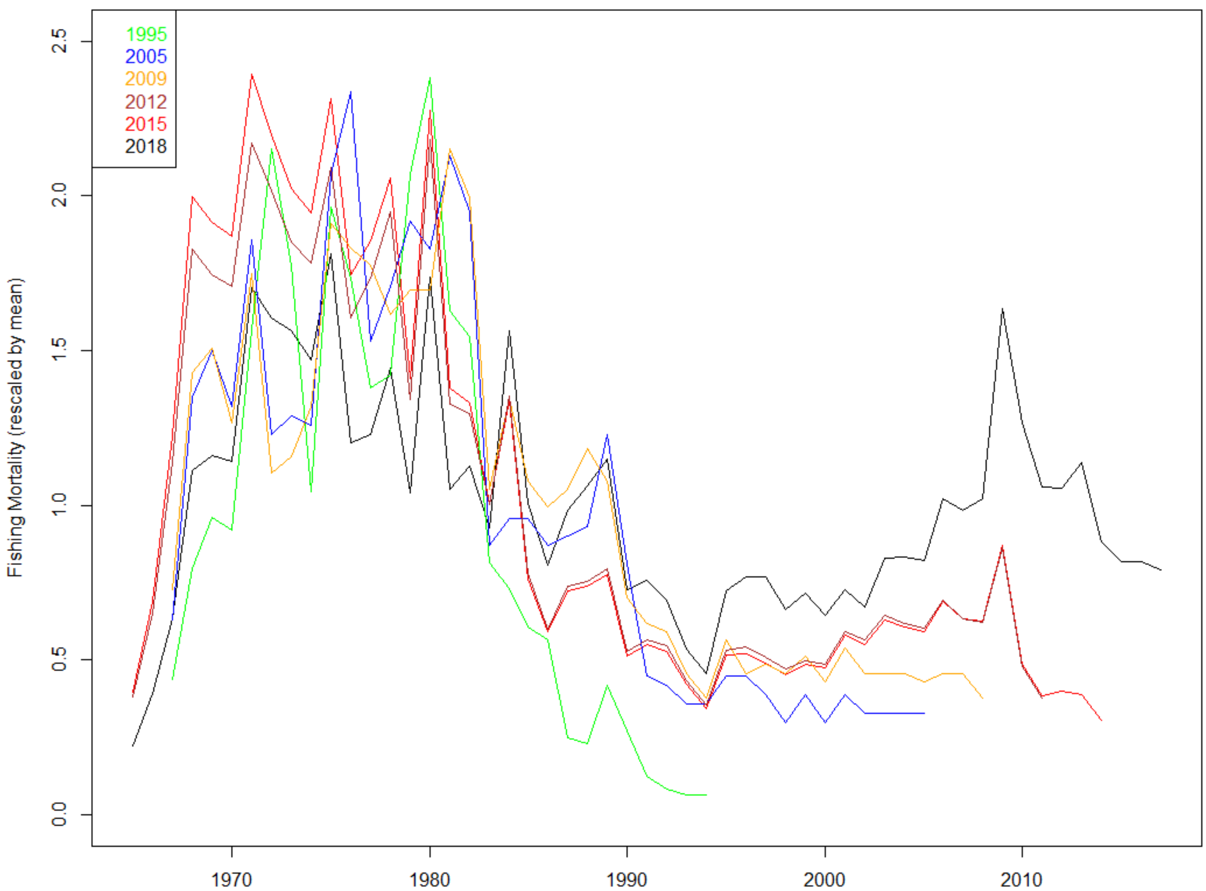

1995
2005
2009
2012
2015
2018
Fishing Mortality (rescaled by mean)
0.0
0.5
1.0
1.5
2.0
2.5
1970
1980
1990
2000
2010

Figure B4- 30 Atlantic herring SSB and recruitment time series for the base ASAP model (Base) and the base model amended to have age- and time-varying M (VaryM)

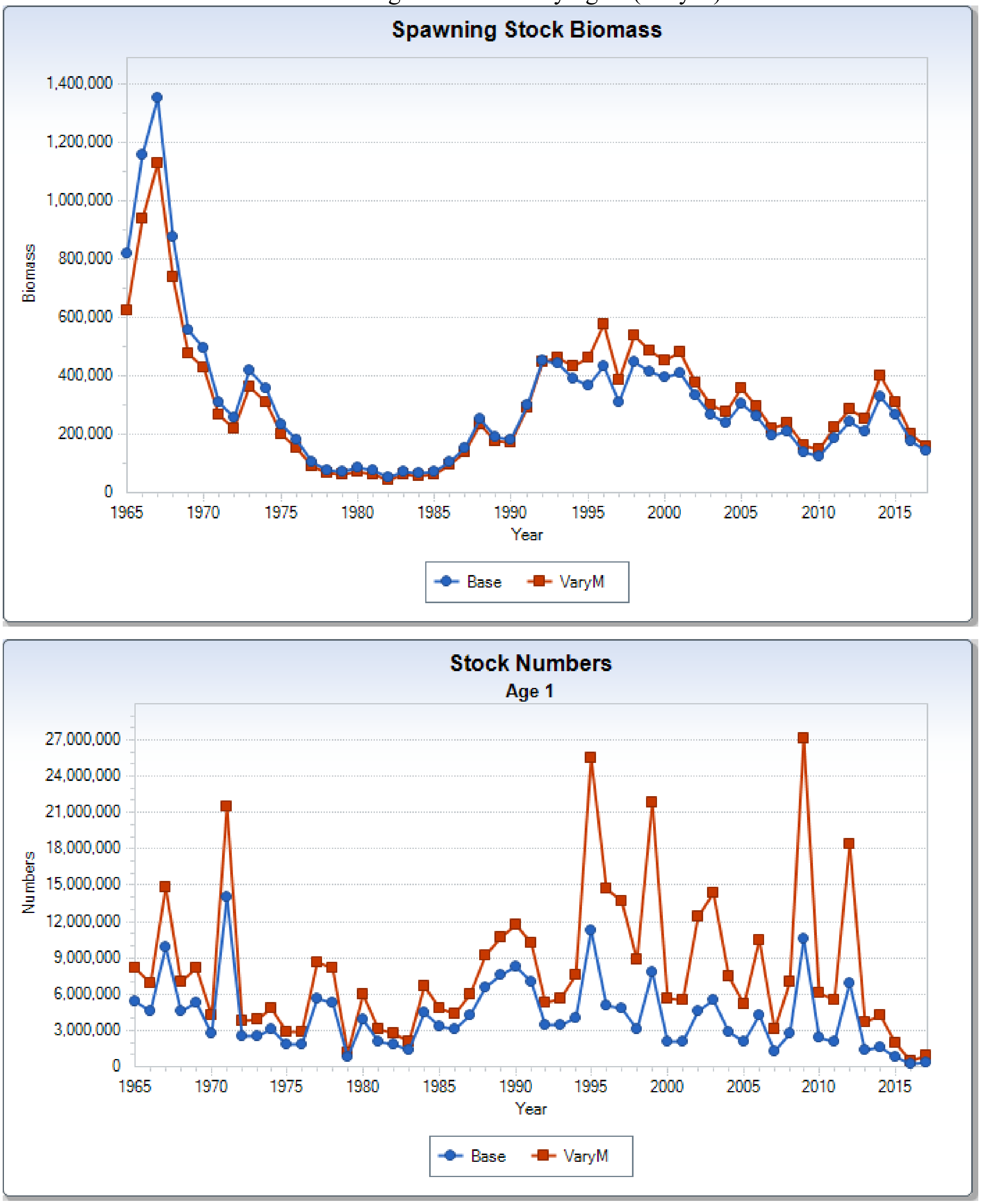


Figure B4- 31 Retrospective patterns for the base model except with age- and time-varying M

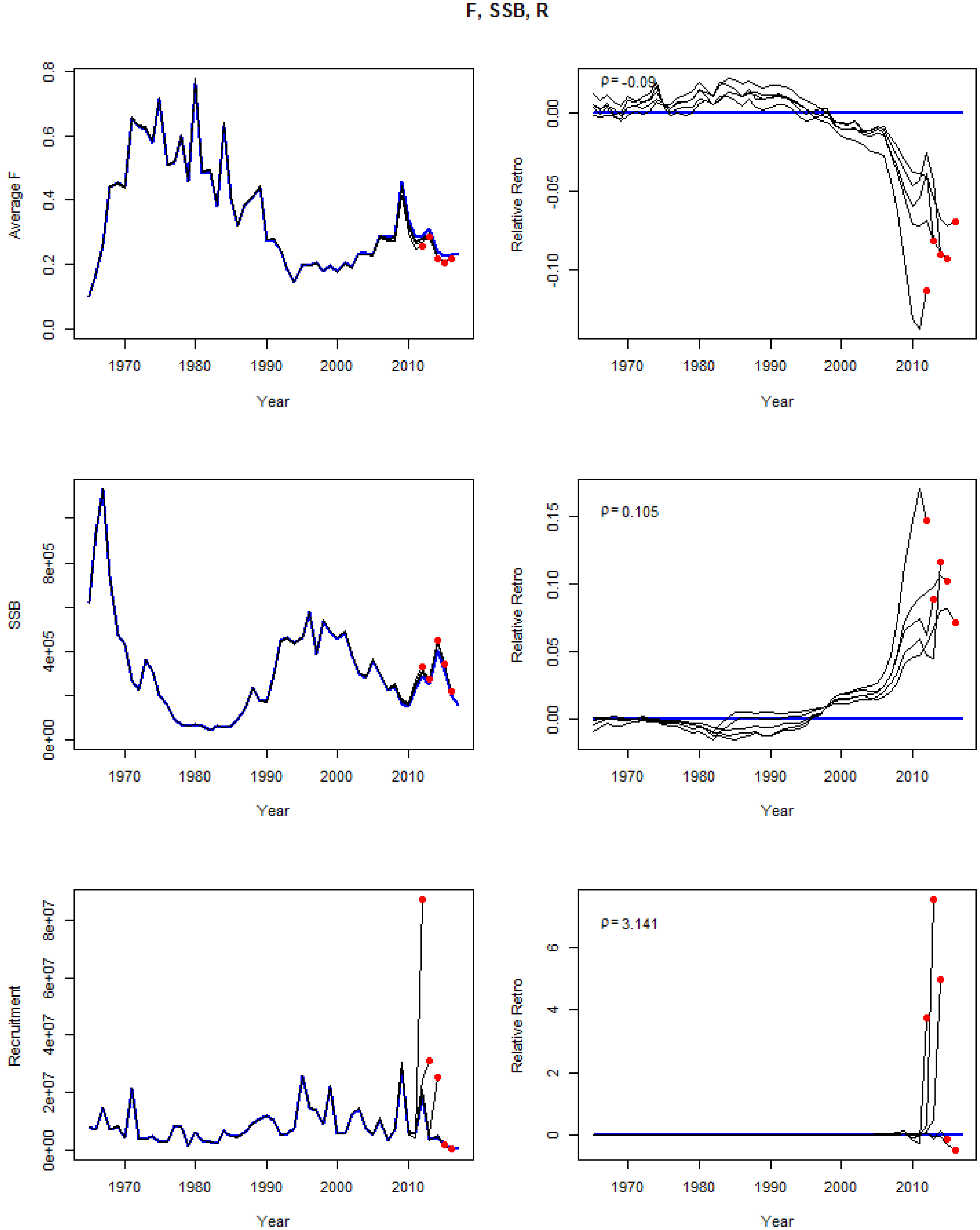
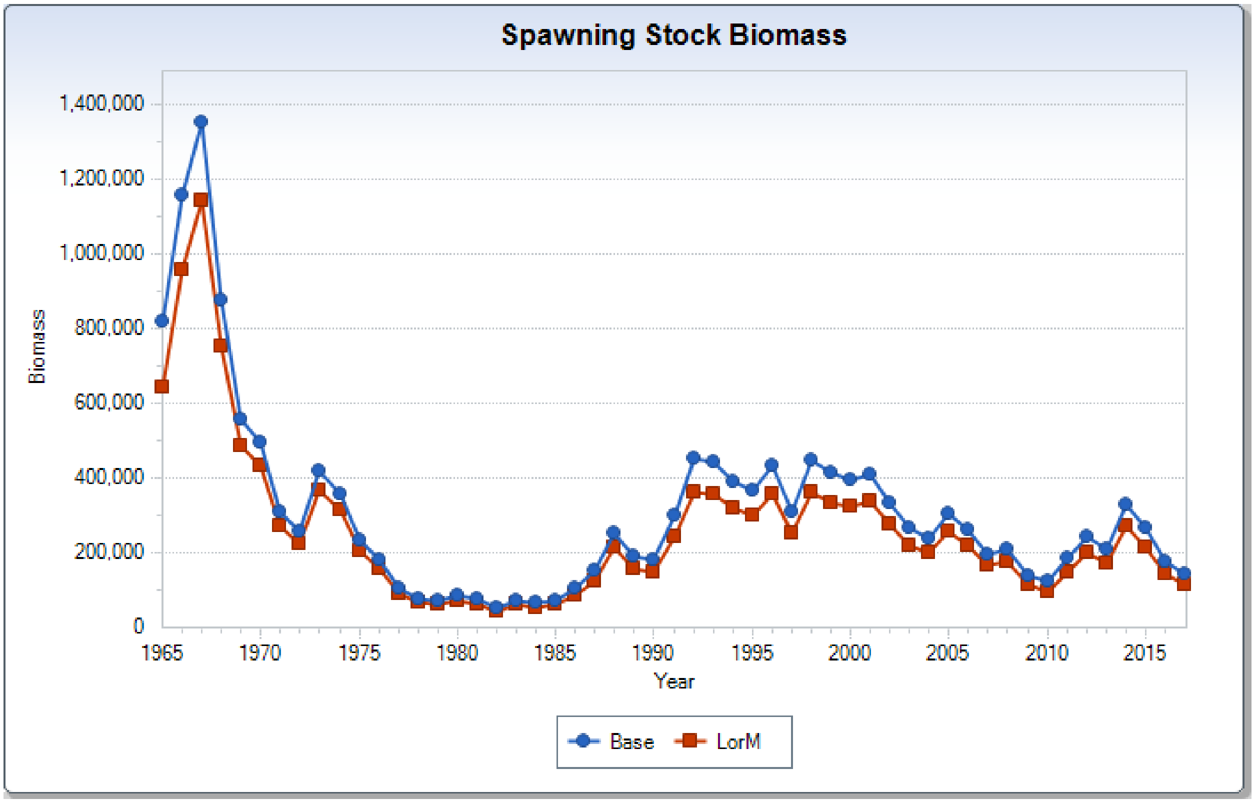


Figure B4- 32 Atlantic herring SSB and recruitment time series for the base ASAP model (Base) and the base model amended to have age-varying M (LorM)



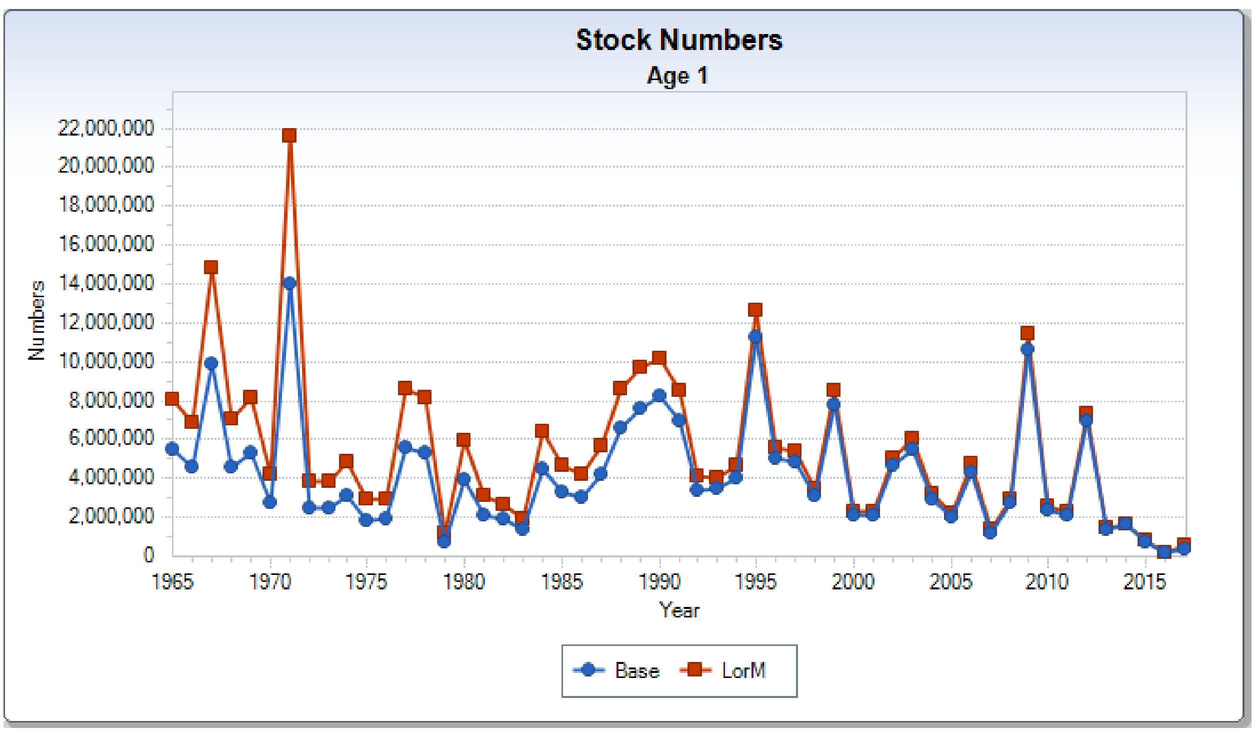


Figure B4- 33 Retrospective pattern for the base model amended to have age-varying M

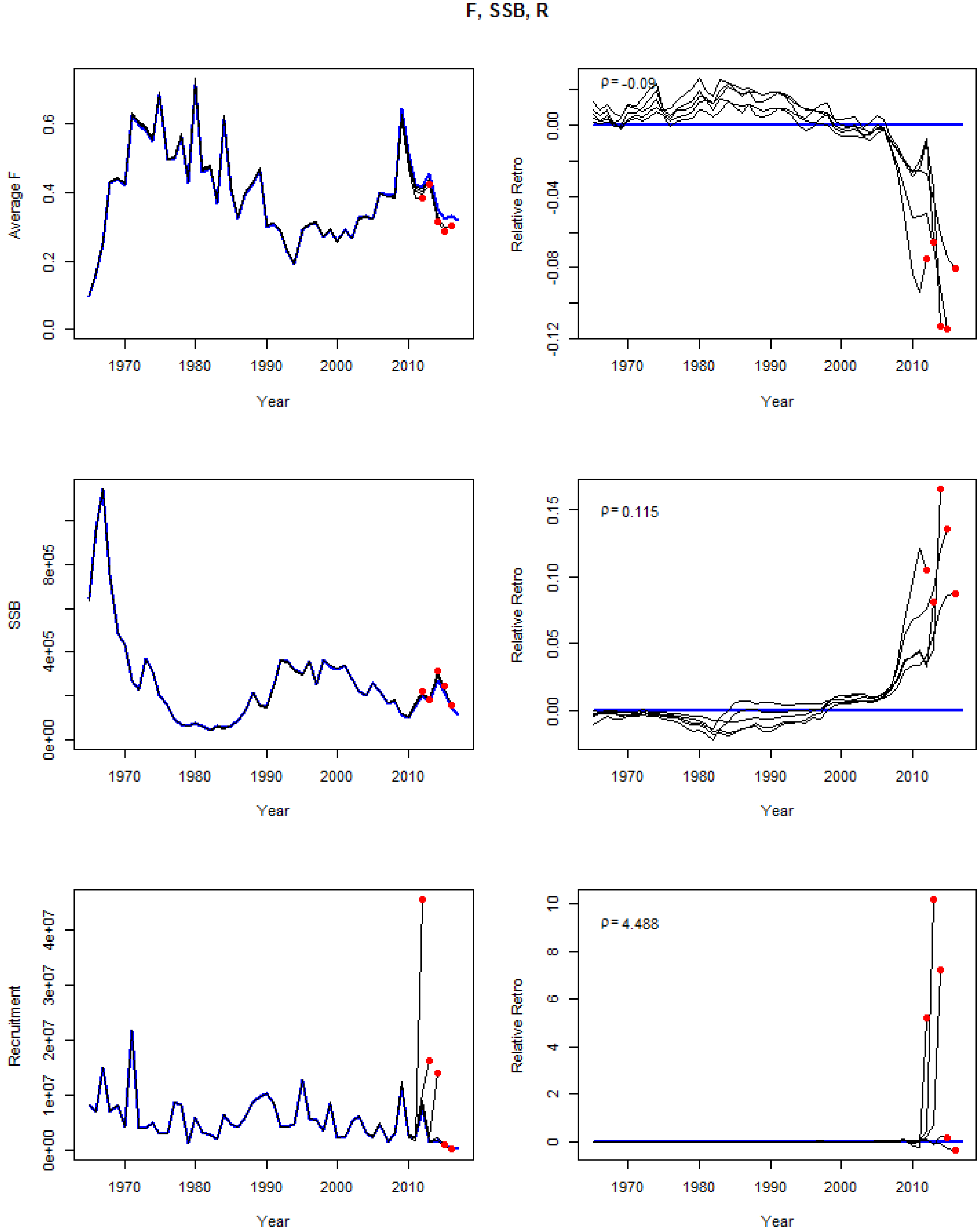
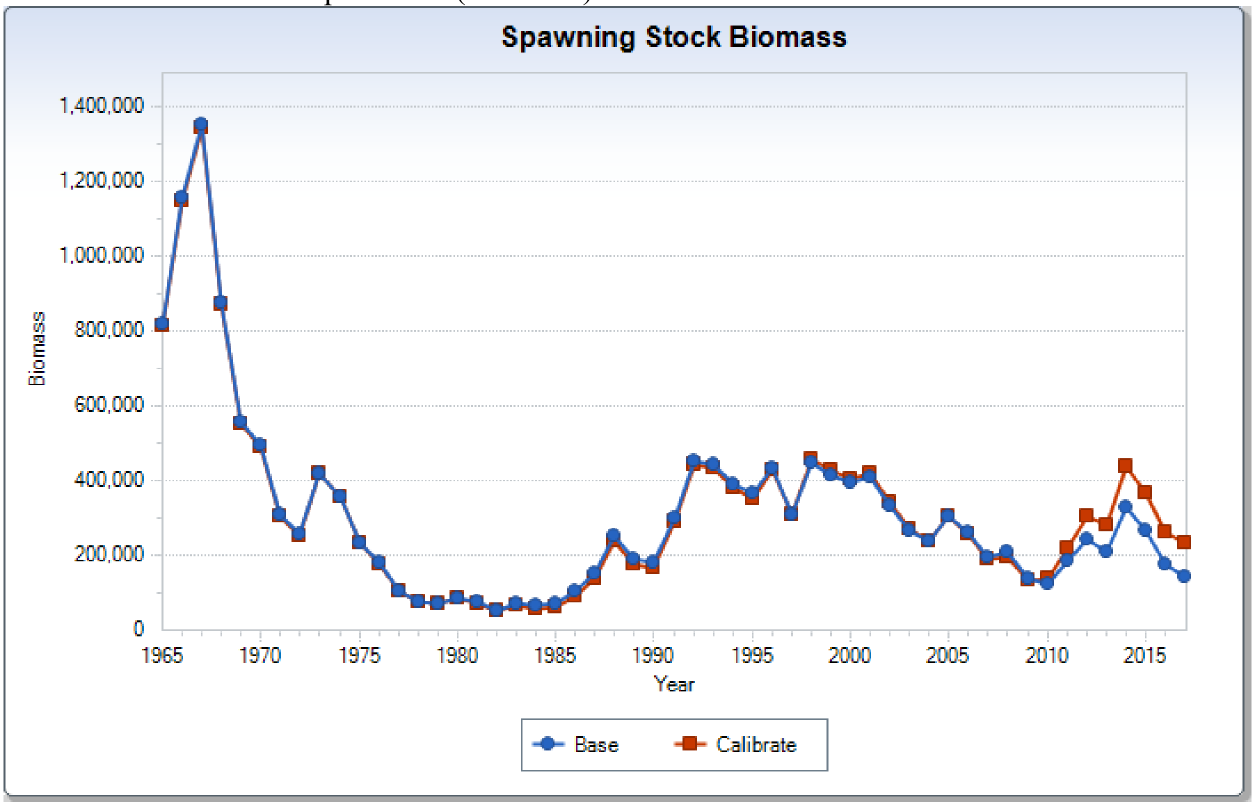


Figure B4- 34 Atlantic herring SSB and recruitment time series for the base ASAP model (Base) and the base model amended to with the NMFS spring and fall Bigelow years (2009-2017) calibrated to Albatross equivalents (Calibrate)



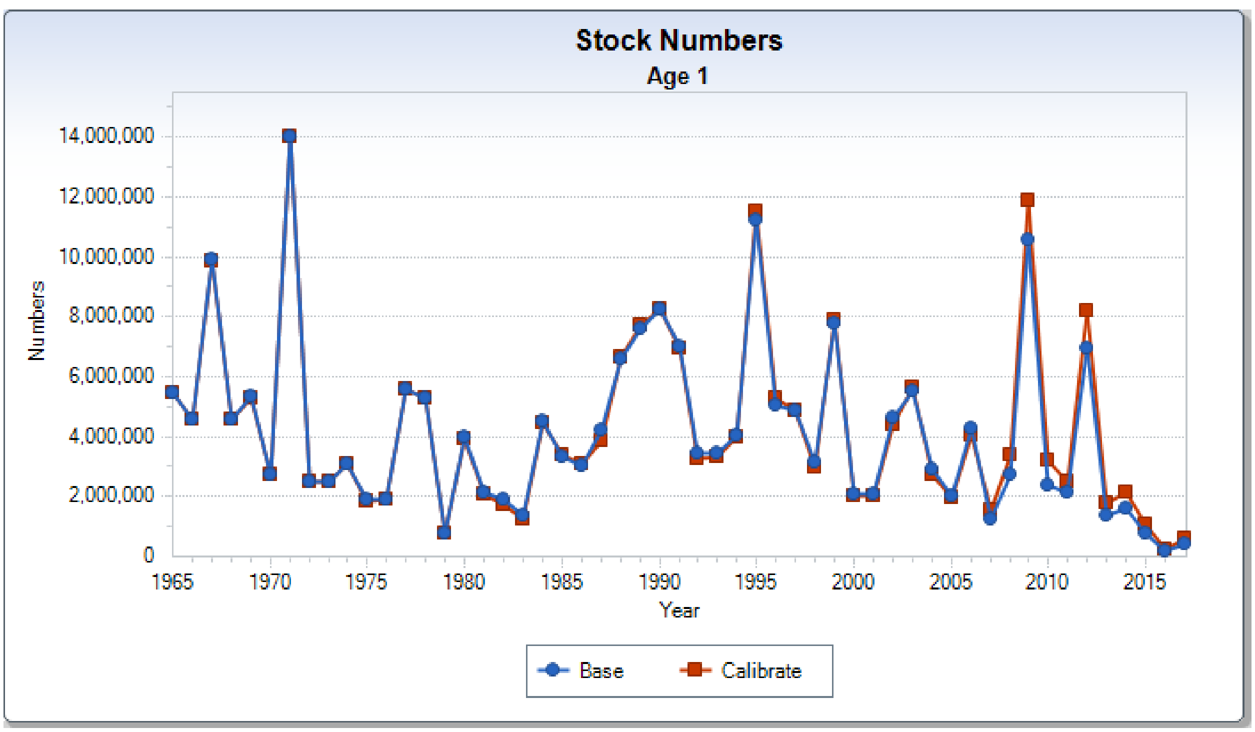


Figure B4- 35 Retrospective pattern for the base model amended to with Bigelow catches (2009-2017) calibrated to Albatross equivalents

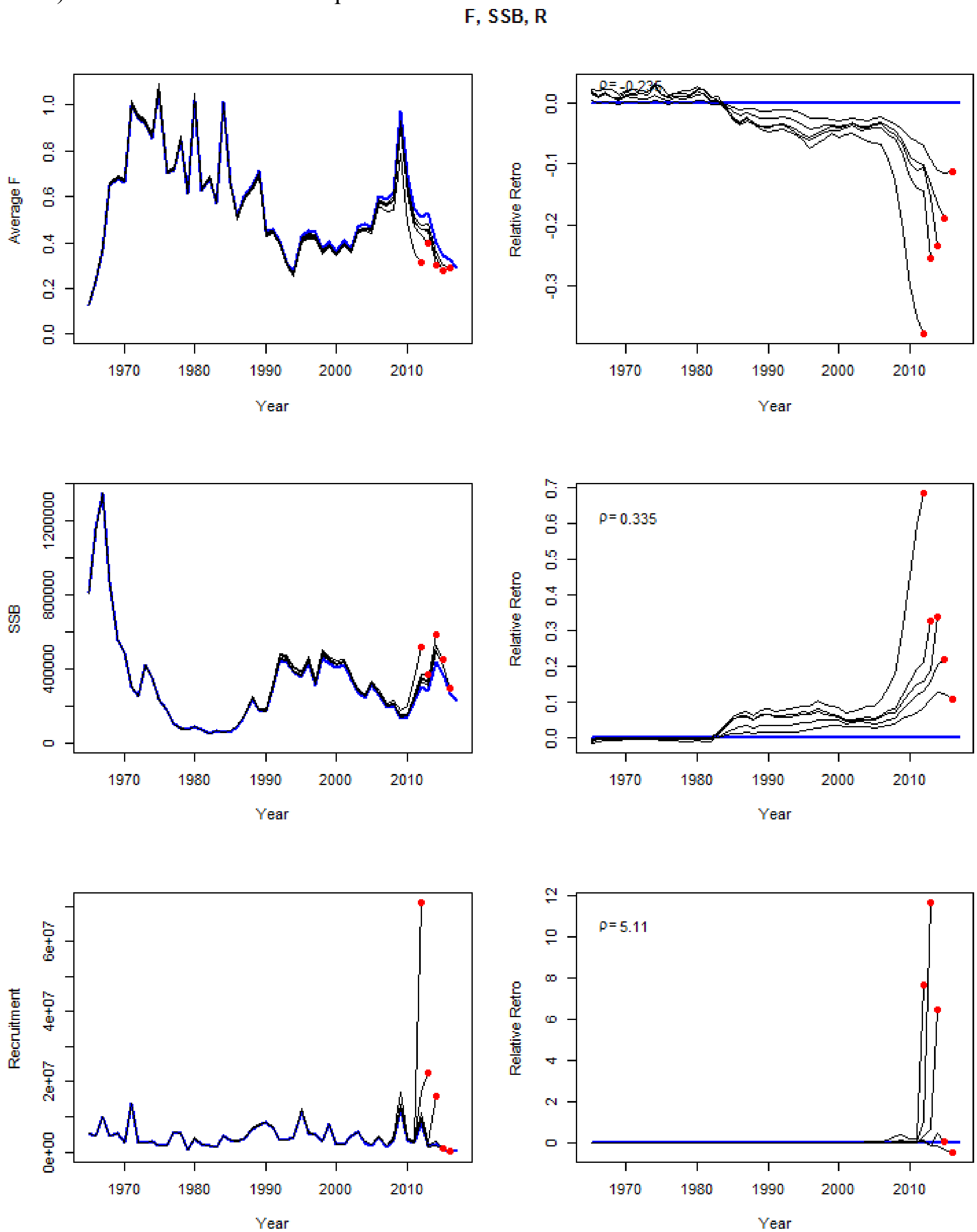


Figure B4- 36 Ratio of Bigelow to Albatross catchability as estimated by ASAP and using paired tow experiments (Conversion Coeff.). Bottom panel is the black bar value divided by the blue bar value in the top panel

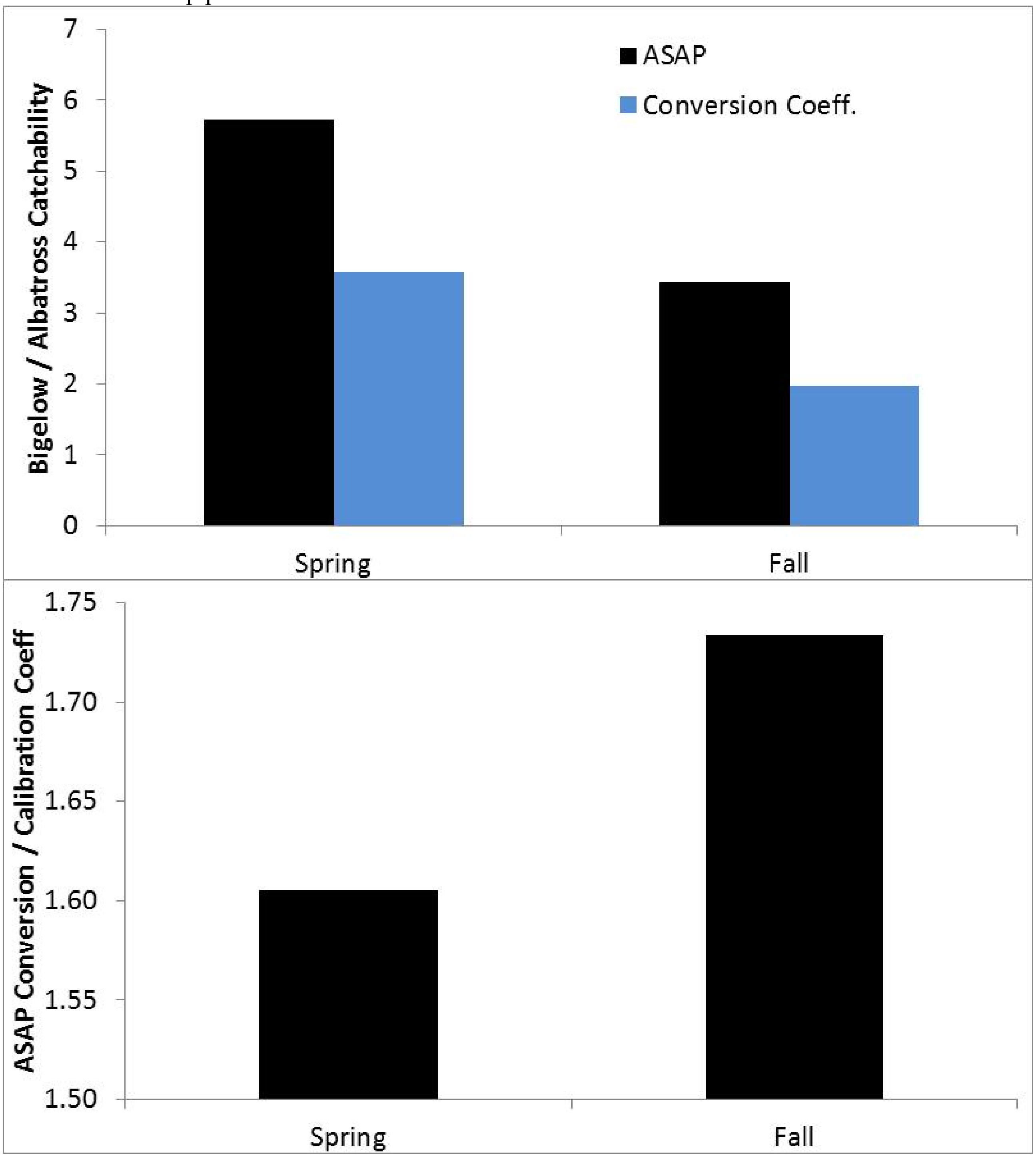
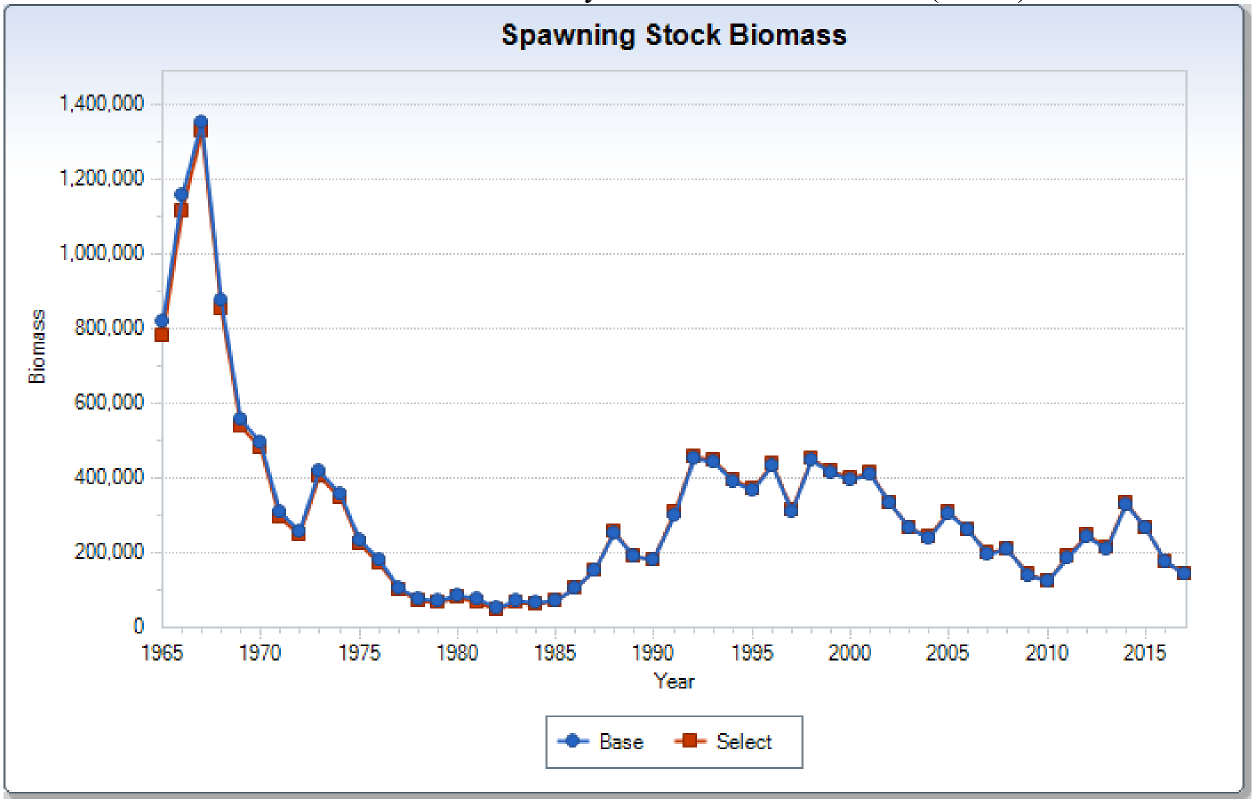


Figure B4- 37 Atlantic herring SSB and recruitment time series for the base ASAP model (Base) and the base model amended with a selectivity block in the mobile fleet (Select)



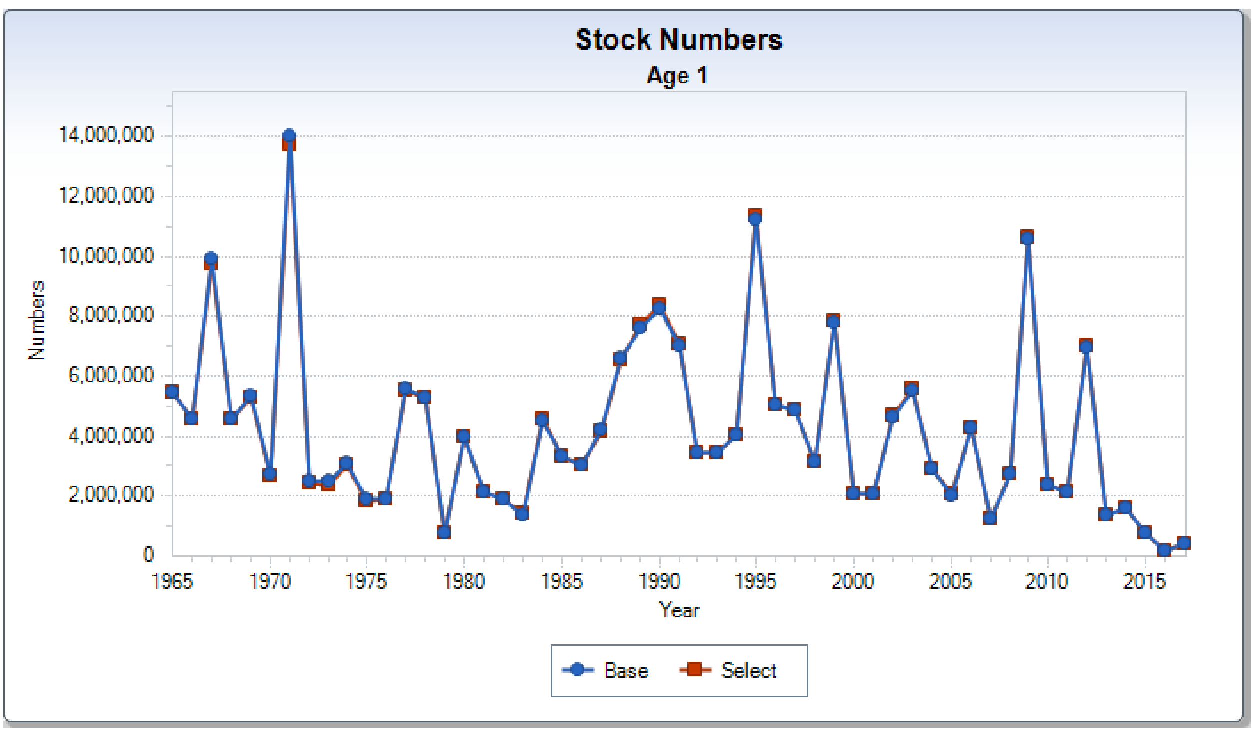


Figure B4- 38 Retrospective pattern for the base model amended with a selectivity block in the mobile fleet

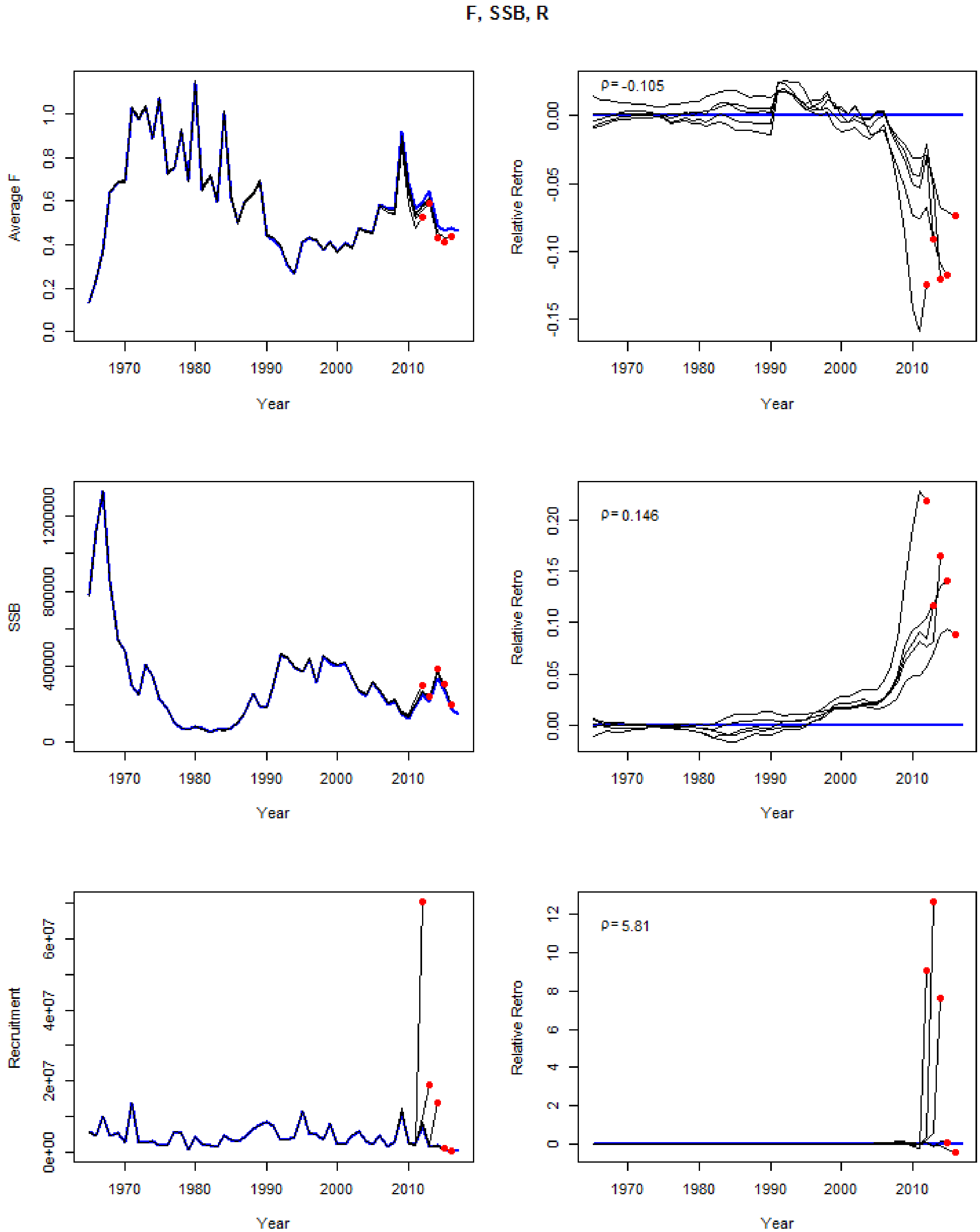
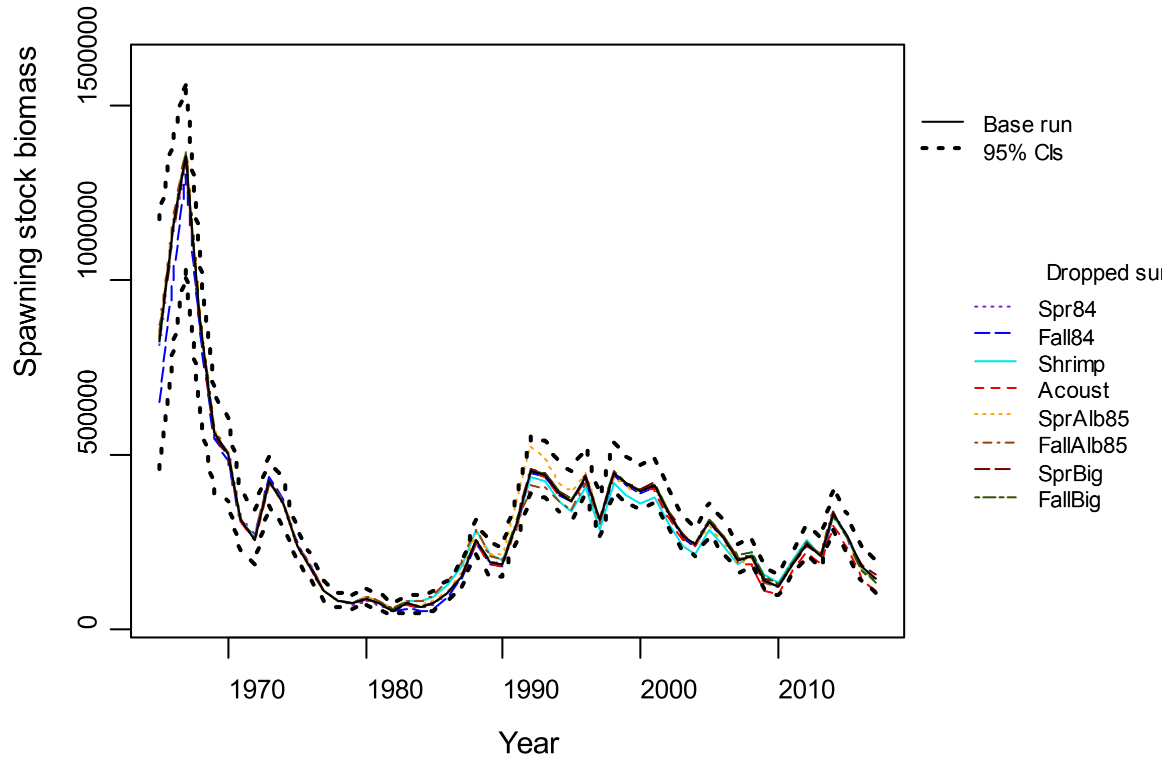


Figure B4- 39 Atlantic herring SSB time series produced by excluding one survey at a time from the base model (top panel) and highlighting the difference between the base and excluding the acoustic index (bottom panel)



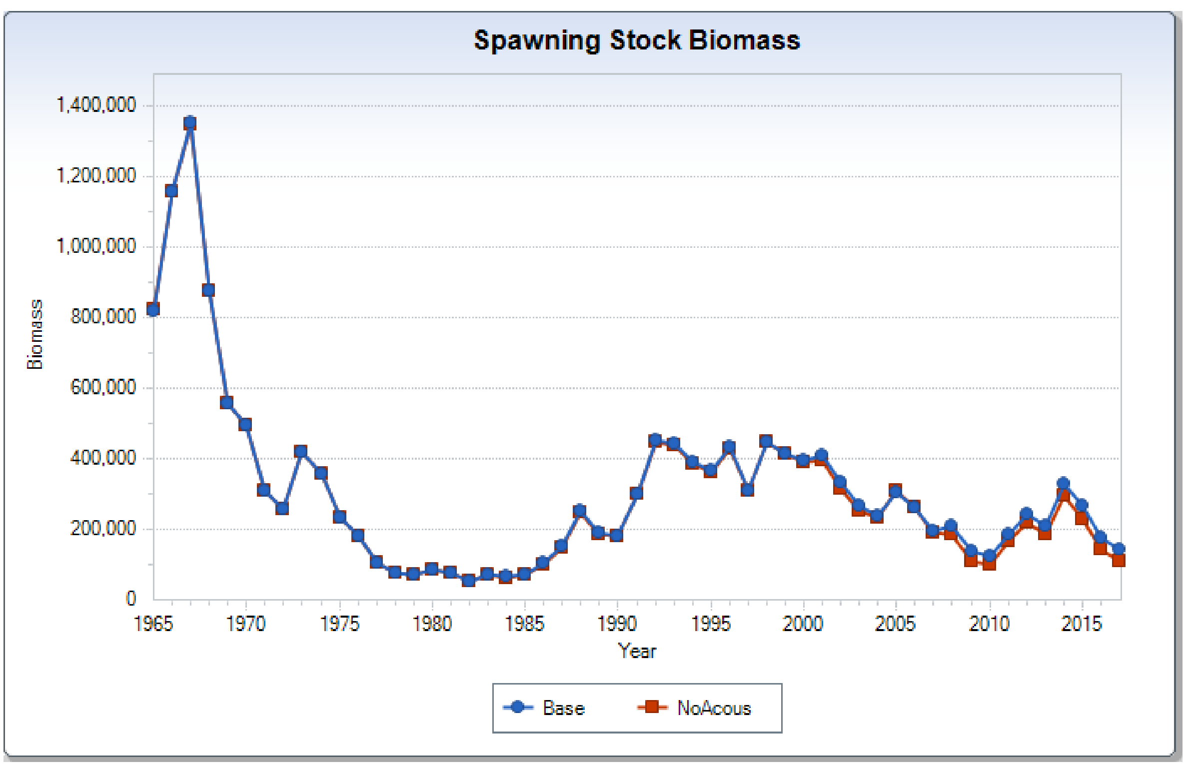


Figure B4- 40 Retrospective pattern for the base model except with the acoustic index excluded from the fit

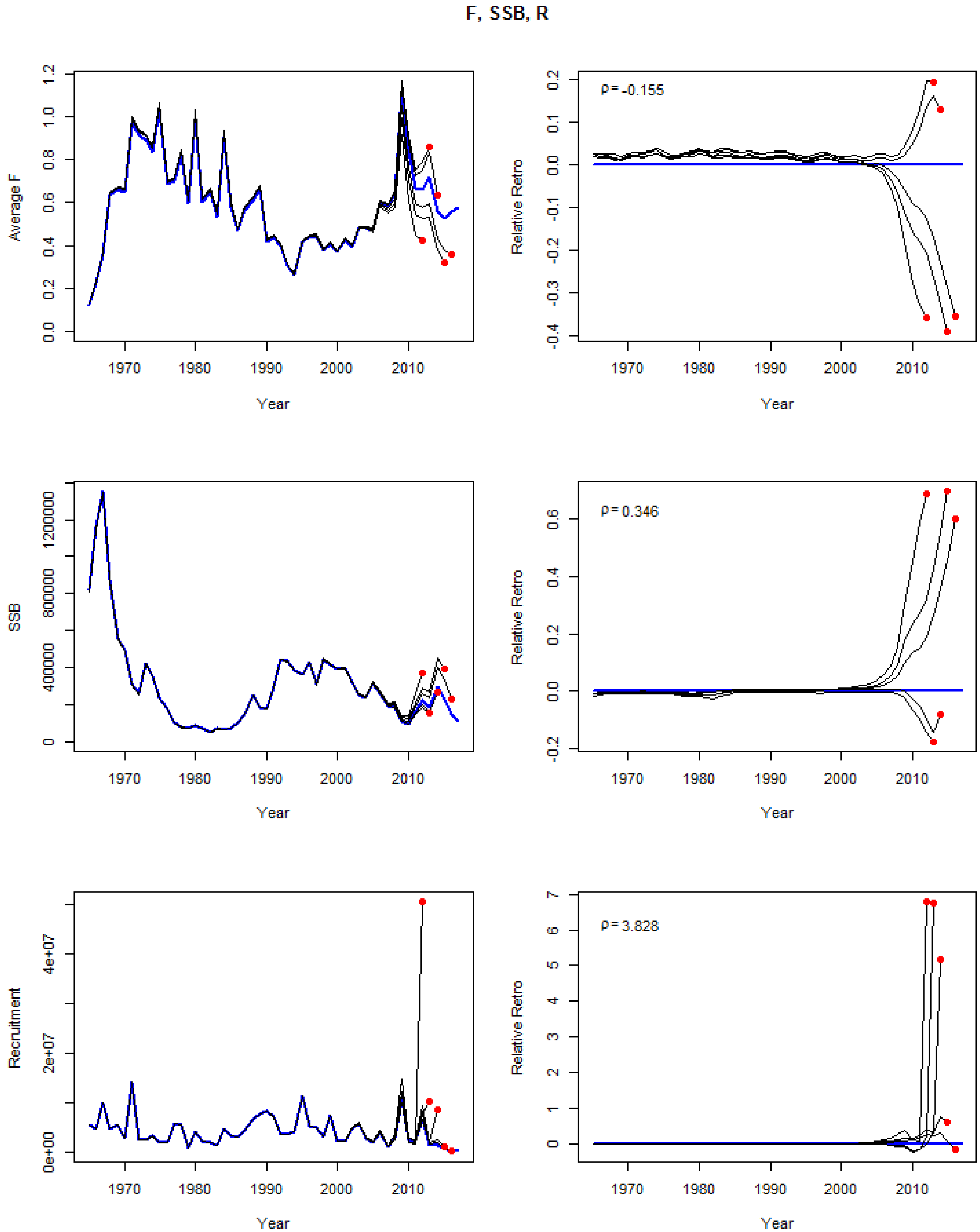


Figure B4- 41 Stock status for the Atlantic herring base model except with the exclusion of the acoustic index from the fit

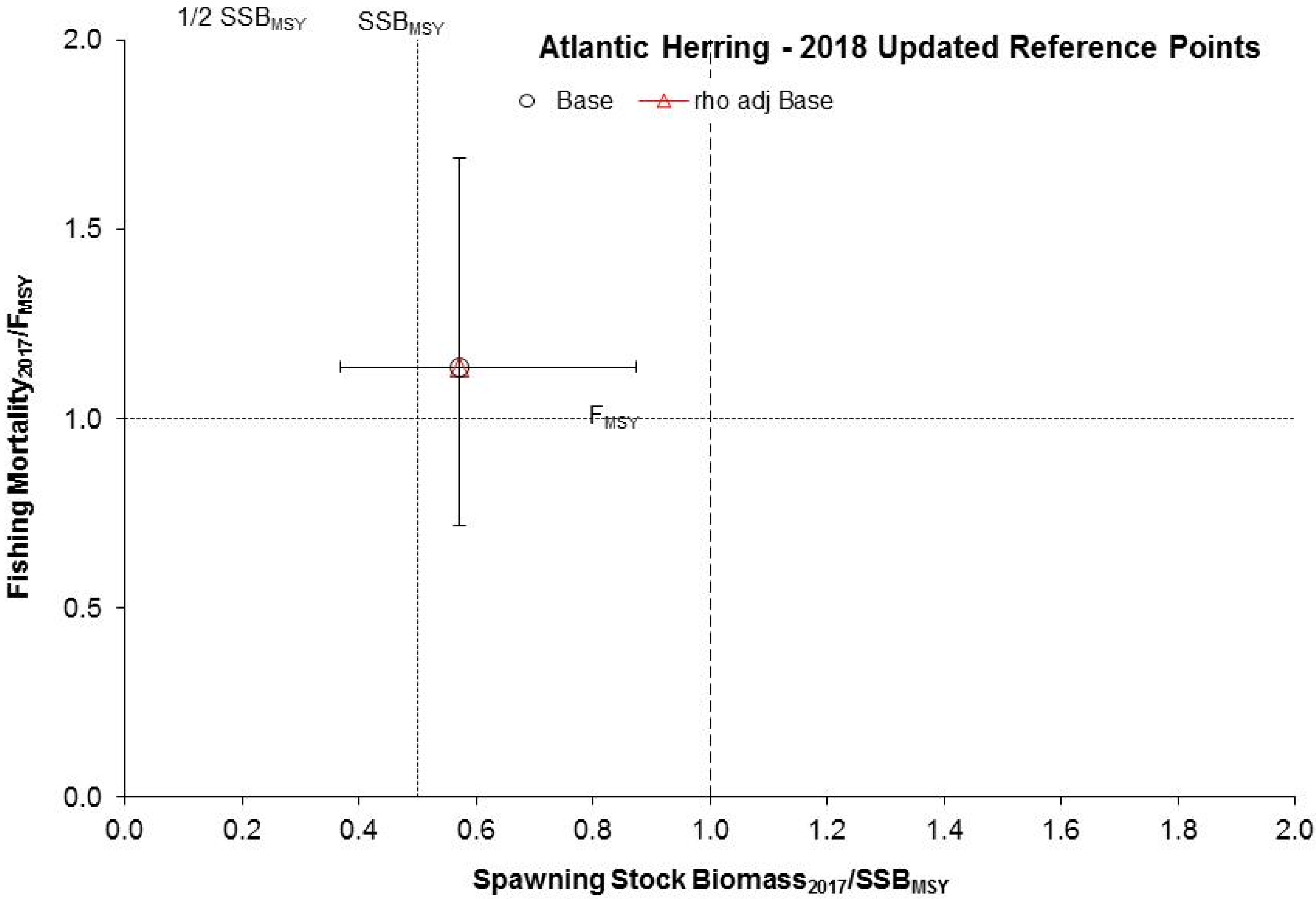


Figure B4- 42 Atlantic herring SSB time series for the base model and the base model with the addition of the index of abundance derived from food habits data

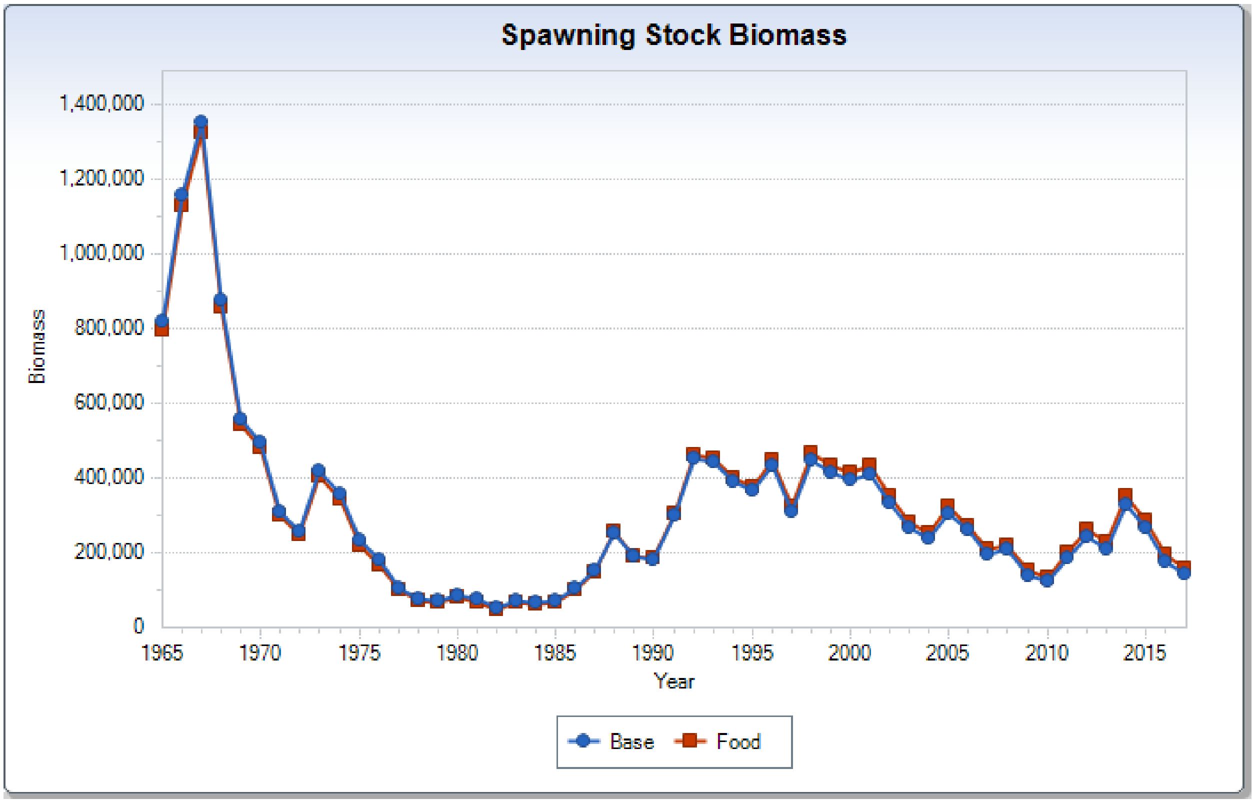
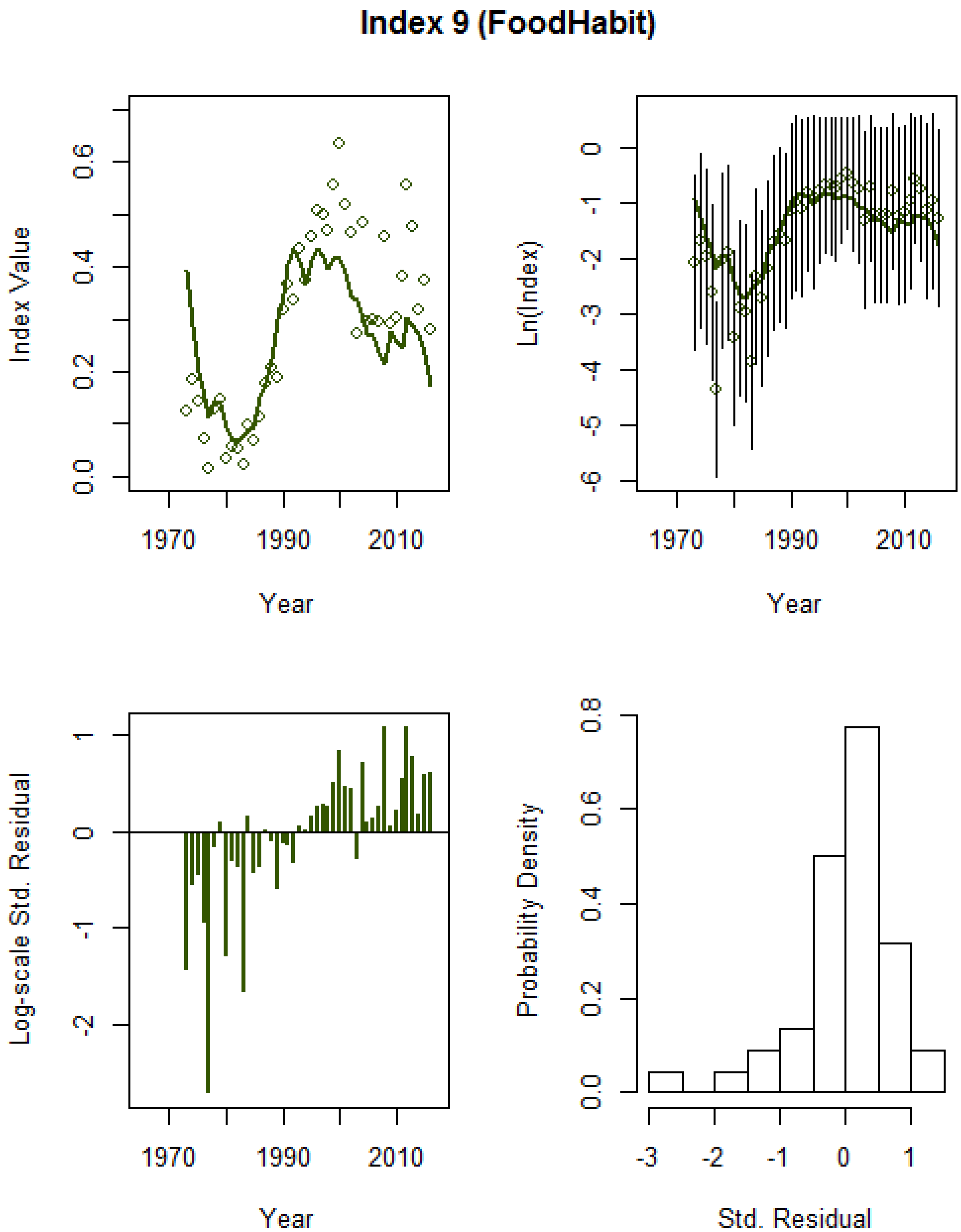


Figure B4- 43 Fit to the food habits index when added to base ASAP model



TOR B5: *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 BRPSs. Comment on the scientific adequacy of existing BRPs and the "new" (i.e. updated, redefined , or alternative) BRPs.*

The existing MSY reference points were based on the fit of a Beverton-Holt stock-recruitment relationship, estimated internally to the ASAP model, and inputs (e.g., weights-at-age, natural mortality) from the terminal year of the assessment (i.e., 2014). Point estimates of the MSY BRPs equaled: MSY = 77,247 mt, F_{MSY} = 0.24, and SSB_{MSY} = 311,145 mt.

No stock-recruit relationship was able to be estimated in the base ASAP model, therefore $F_{40\%}$ was used as a proxy for F_{MSY} and long-term projections were used to derive other MSY BRP proxies. The average of the last five years (2013-2017) of weights at age and maturity at age were used to calculate $F_{40\%}$ and in long-term projections. Selectivity at age equaled the catch weighted average of the selectivities at age from the mobile and fixed fleets over the last five years, which produced selectivity generally similar to the mobile fleet given that this fleet accounts for most of the catch in those years. Recruitment in each year of the projections was drawn from the empirical cumulative distribution of the estimated recruitments from 1965-2015. The estimates of recruitment from 2016-2017 were excluded because they were imprecisely estimated with CVs equal to 95% and 251%, respectively (as a point of comparison the CV for 2015=38%; Figure B4- 19). In drawing recruitments from the empirical distribution, a uniform random value is drawn between 0-1 each year, and the recruitment associated with that probability from the cumulative distribution is applied. Thus, any recruitment between the minimum and maximum in the estimated time series has an equal probability of selection each year. F_{MSY} proxy = 0.51, SSB_{MSY} proxy = 189,000 mt (½ SSB_{MSY} = 94,500 mt), and MSY proxy = 112,000 mt.

Metric	Point Estimate	80% probability interval
F_{MSY}	0.51	--
SSB_{MSY}	189,000 mt	128,000 – 278,000 mt
MSY	112,000 mt	78,000 – 157,000 mt

The existing MSY reference points were based on estimates of a Beverton-Holt stock-recruit curve fit internally to the ASAP model (NEFSC 2012; Deroba 2015). The ability to estimate the stock-recruit curve seems to have deteriorated in this assessment and was not supported. The deterioration in the models ability to estimate a stock-recruit curve is likely related to changes in model structure, such as in M and various likelihood penalties (see TOR B4). Although, the 2012 assessment (NEFSC 2012) reported similar estimation issues as in this assessment (e.g., flat likelihood profile over steepness; steepness and unfished SSB highly correlated), and so the ability of previous models to estimate a stock-recruit curve was also tenuous. The newly proposed reference points no longer rely on a poorly estimated stock-recruit relationship.

TOR B6: *Make a recommendation about what stock status appears to be based on the existing model (from previous peer reviewed accepted assessment) and based on a new model or model formulation developed for this peer review.*

a. Update the existing model with new data and evaluate stock status (over fished and overfishing) with respect to the existing BRP estimates.

Given the Working Group's conclusion that MSY reference points based on the estimation of a stock-recruit curve were unjustified, and were likely unjustified in previous assessments, the existing BRPs are not meaningful. Similarly, evaluating stock status of the existing model with updated data to the existing MSY BRPs is not informative.

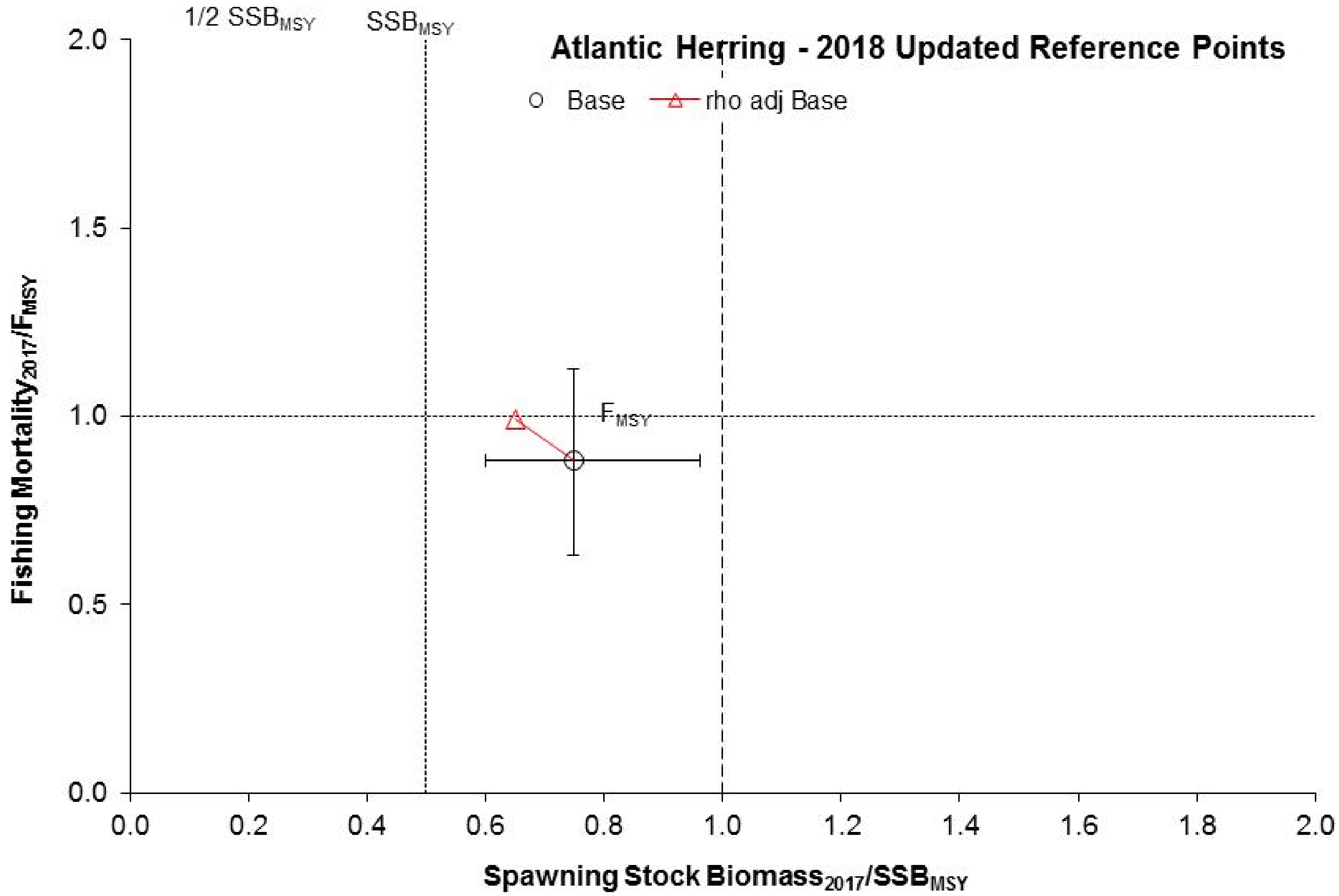
b. Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR B5).

The base ASAP model estimated $F_{7\text{-}8}$ (see TOR B5) in 2017 to be 0.45 and SSB in 2017 was 141,000 mt. Since the retrospective adjusted values do not fall outside of the confidence intervals of the base model estimates, no retrospective adjustment was warranted. A comparison of the base model values to the new MSY proxy reference points suggest that overfishing is not occurring and that the stock is not overfished (Figure B6- 1). The error bars for $F_{7\text{-}8}$, however, included overfishing (Figure B6- 1).

c. Include descriptions of stock status based on simple indicators/metrics.

The estimated numbers at age in 2017 indicate that the population is characterized by more age 6 fish than age 1 and age 2 combined. This result suggests a reliance on the ageing 2011 cohort (age 6 in 2017). If the estimated record low recruitments in recent years hold true, then the SSB is likely to remain relatively low and put the stock at relatively high risk of becoming overfished. Without improved recruitment, the probability of overfishing under recent catch levels is also likely relatively high.

Figure B6- 1 Stock status, “Kobe plot”, for the base ASAP model ± 80% CI.



TOR B7: Develop approaches and apply them to conduct stock projections.

a. Provide numerical annual projections (through 2021) and the statistical distribution (i.e. probability density function) of the catch at F_{MSY} or an F_{MSY} proxy (i.e. overfishing level, OFL) (see Appendix to the SAW TORs). 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).

Short-term projections of future stock status were conducted based on the results of the base ASAP model. The projections did not account for any retrospective pattern because the Mohn's Rho adjusted values for stock status were within the 80% probability intervals of the 2017 point estimates of $F_{7\text{-}8}$ and SSB (Figure B6- 1; TOR B6). Numbers at age for ages 2-8+ in 2018 (the first year of the projection) were drawn from 1000 vectors of numbers at age produced from MCMC simulations of the base ASAP model (see TOR B4 for description of MCMC). Age 1 recruitment in 2018 was drawn from 1000 values, with each value representing the geometric mean of the estimated recruitments for 2013-2017 from each of the 1000 MCMC simulations of the base ASAP model. Age 1 recruitment in 2019-2021 was drawn from the empirical cumulative distribution of the estimated recruitments from 1965-2015 from the base ASAP model (2016 and 2017 were excluded due to imprecision; TOR B5). All other inputs were the same as described in TOR B5.

Projections were repeated with catch in 2018 equal to: 1) the 2018 allowable biological catch (111,000 mt), or 2) half the 2018 allowable biological catch (55,000 mt). Regardless of the catch value in 2018, fishing mortality in 2019-2021 equaled the Fmsy proxy (0.51; TOR B5), and so the row of "Catch (mt)" in the tables below represents the catch at the Fmsy proxy.

	2018	2019	2020	2021
Catch (mt)	111,000	13,700	31,000	55,700
Catch 80% CI	--	4,000-36,600	16,000-62,700	32,100-95,500
F_{7-8}	1.7	0.51	0.51	0.51
F_{7-8} 80% CI	0.83-4	--	--	--
SSB (mt)	32,900	19,700	31,700	85,800
SSB 80% CI	4,700-78,600	5,200-58,700	16,500-71,300	47,500-159,000
P(overfishing)	0.95	--	--	--
P(overfished)	0.96	0.94	0.93	0.58

	2018	2019	2020	2021
Catch (mt)	55,000	28,900	38,000	59,400
Catch 80% CI	--	17,200-53,100	22,700-70,800	35,300-99,600
F_{7-8}	0.58	0.51	0.51	0.51
F_{7-8} 80% CI	0.4-0.86	--	--	--
SSB (mt)	75,300	43,500	42,600	91,000
SSB 80% CI	46,900-112,100	25,800-86,100	26,400-87,900	52,400-166,100
P(overfishing)	0.69	--	--	--
P(overfished)	0.76	0.92	0.91	0.53

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. Identify reasonable projection parameters (recruitment, weight -at-age, retrospective adjustments, etc.) to use when setting specifications.

The Working Group agreed that the 2018 ABC of 111,000mt is unlikely to be fully utilized and that a value of 55,000mt is more realistic. The exact value for 2018 catch that should ultimately be used is best left to the Atlantic herring Plan Development Team of the New England Fishery Management Council. Other uncertainties were addressed in TOR B4. The projections assume the future recruitment will approach the mean for the time series. If recruitment continues to be below average, the projected catch increases may be overly optimistic.

c. Describe the stock's vulnerability (see "Appendix to the SARC TORs") to becoming overfished, and how this could affect the choice of ABC (or DEF, possibly even GH&I).

The unknown contributions of the Scotian Shelf (4WX), Gulf of Maine, and Georges Bank stocks can affect the stocks vulnerability to becoming overfished. For example, if the Scotian Shelf stock is contributing a significant amount of fish and that contribution decreases, the vulnerability to overfishing would increase. The vulnerability of the stock has been demonstrated by the historical collapse of the Georges Bank component in the 1980s, which also demonstrated that the multiple spawning groups can be differentially impacted by fishing. Varying contributions from the Scotian Shelf (4WX) stock may also contribute to a retrospective pattern (see below).

In the short-term, the relatively poor recruitments in 2013-2017 will increase the vulnerability of the stock to becoming overfished. The 2016 and 2017 cohorts were imprecisely estimated and so estimates of these cohorts may change significantly in either direction in future assessments, and decisions should likely consider this uncertainty. Growth (i.e., weight at age) also continues to be relatively low when compared to the 1990s, and this seems to be a longer-term feature of the stock that also reduces production. The stock, however, seems to be capable of producing relatively large and small year classes regardless of growth, and so recruitment is likely the more significant driver of short-term vulnerability.

While this assessment had a retrospective pattern that did not warrant adjustments (i.e., via Mohn's Rho), the history of the Atlantic herring stock assessment suggests that resolutions to retrospective patterns are ephemeral (NEFSC 2012; Deroba 2015). Given concerns that estimating catchability separately for the Bigelow years in 2009-2017 may also be aliasing other causes of the retrospective pattern (TOR B2; TOR B4), a safe assumption is that future herring assessments will have worsening retrospective patterns. Retrospective patterns are indicative of model misspecification, and this would increase the vulnerability of the stock to becoming overfished.

TOR B8: *If possible, make a recommendation about whether there is a need to modify the current stock definition for future assessments.*

Previous assessments (NEFSC 2012) concluded that there is likely sub-stock structure unaccounted for in the assessment, but that there is no ability to distinguish mixed survey and fishery catches to stock of origin. This lack of information on stock of origin precludes accounting for the sub-stock structure. In this assessment, a Stock Synthesis model was attempted (Appendix B4) that accounted for stock structure on a coarse level (i.e., Inside Gulf of Maine and Outside Gulf of Maine). In order to attempt this, however, assumptions were required that were likely incorrect, and model diagnostics were poor. The consequences of not accounting for stock structure are unclear, and therefore the need to modify the stock definition is also unclear. More certain, however, is that changing the stock definition and accounting for stock structure in the assessment is currently not possible. Continued research on the topic is warranted (see TOR B9).

TOR B9: *For any research recommendations listed in SARC and other recent peer reviewed assessment and review panel reports, review, evaluate and report on the status of those research recommendations. Identify new research recommendations.*

Higher priority research recommendations are in bold.
2018 Atlantic Herring Research recommendations:

- **Further research on the use of acoustic technology for inclusion in stock assessment, including information using industry based platforms. Specifically:**
    - Investigate methods for converting herring acoustic indices to biomass.
    - Investigate refinements in target strength conversion to abundance estimates in acoustic data
    - Evaluate statistical design implications in acoustic data from surveys and ships of opportunity.
    - Additional research to better understand species identification using acoustic signals
- Investigate use of length data, stock structure and movement within assessment models (e.g. SS3)
- **Evaluate data collected in study fleet program for informing assessment data. Development research ideas that can be addressed within the context of the study fleet**.
    - Explore fisheries selectivity in greater depth. Perhaps with study fleet and with historical perspective with industry.
    - Research on depth preferences of herring in the water column through time to inform selectivity and catchability.
- Continue work related to understanding sources of variation in stomach contents, especially as this relates to the (GAMM) models used to develop an index of herring abundance.

General assessment recommendations:

- **Evaluate the ability of state-space models to reliably estimate observation and process error variances under a range of scenarios, as well as their ability to estimate quantities of management interest**.

- Develop a list of standards for evaluating data for possible use in stock assessment. Also develop standards for evaluating model diagnostics and inclusion criteria of indices.
- Develop protocols for multi model inference to provide management advice from stock assessments based on NEFSC experience as well as other input (e.g. model averaging approaches).
- Develop simulations to evaluate diagnostics that are useful under different scenarios (e.g. use of likelihoods, retrospective patterns for diagnostics, etc.).

2012 SARC Research Recommendations

a. *More extensive stock composition sampling including all stocks (i.e. Scotian Shelf).*

No additional work completed. Research in other areas suggests that parasite composition may be informative.

b. *Develop (simple) methods to partition stocks in mixed stock fisheries.*

No simple methods completed. Work ongoing using SS3 model to address mixed stock issue.

c. *More extensive monitoring of spawning components.*

Work completed at NEFSC examining extended spawning season in a subset of the mixed stock. Egg survey data analyzed for use as SSB index.

d. *Analyze diet composition of archived mammal stomachs. Improve size selectivity of mammal prey. Also sea birds.*

No work completed for assessment however additional information added to recent herring MSE.

e. *Consider alternative sampling methods such as HabCam.*

Acoustic index evaluated and used in the 2018 assessment model.

f. *Research depth preferences of herring.*

Evaluation attempted using Study Fleet information but data incomplete for such analysis. Acoustics has offered some insights (e.g., Jech and Stroman 2012).

g. *Simulation study to evaluate ways in which various time series can be evaluated and folded into model.*

On-going work under SEAGRANT funding to Essington and Deroba related to the sampling and subsequent utility of diet data. Similarly, Trijoulet et al. (In review Ecology) have done some simulation/estimation studies to inform how diet data should be treated in the fitting of multi-species stock assessment models.

h. *Evaluate use of Length-based models (Stock Synthesis and Chen model).*

SS3 initiated but needs additional work before consideration for use in assessment. Chen model no longer supported.

i. *Develop indices at age from shrimp survey samples.*

Average age-length key developed for application to survey samples. Will make request for a collection of age samples in shrimp survey.

j. *Evaluate prey field to determine what other prey species are available to the predators that could explain some of the annual trends in consumption.*

Some work done regarding sand lance but otherwise not completed.

k. *Develop statistical comparison of consumption estimates and biomass from model M.*

No additional work completed

l. *Consider information on consumption from other sources (i.e. striped bass in other areas) and predators inshore of the survey.*

No additional work completed. No information available.

m. *Investigate why small herring are not found in the stomachs of predators in the NEFSC food habits database.*

No additional quantitative work completed, however discussions suggest a potential spatial mismatch between our survey coverage and small herring.

n. *Develop an industry-based LPUE or some other abundance index (Industry Based Survey).*

No additional work completed, however ongoing discussion regarding use of acoustic information collected by industry.

o. *Develop objective criteria for inclusion of novel data streams (consumption, acoustic, larval, etc) and how can this be applied.*

Criteria for inclusion already in place, although not completely documented. (see new recommendations).

2012 CIE Research Recommendations

1. *Alternative catch scenarios could be developed to account for uncertainty in the stock boundary, particularly including catches from the Scotian Shelf. This would also allow examination of whether catch underestimation (e.g. inclusion of Scotian shelf catch) can contribute to the reduction in the retrospective pattern and contribute to or explain the need for increased M.*

No additional work completed

2. *Look at the effect of adding a penalty to encourage the NMFS survey trawl door-change q ratios to be similar in spring and fall.*

No indication based on calibration experiment that this is necessary.

3. *Using simulation/estimation methods, evaluate consequences of alternative harvest policies in light of uncertainties in model formulation, presence of retrospective patterns, and incomplete information on magnitude and variability in M (see term of reference 9).*

Considered to some extent in recent MSE work.

References

Azarovitz, T.R. 1981. A brief historical review of the Woods Hole Laboratory trawl survey time series. In: Doubleday WG; Rivard D., eds. Bottom trawl surveys. Can Spec. Publ. Fish. Aquat. Sci. 58; p 62-67.

Azarovitz, T.R., S.H. Clark, L. Despres, and C.J. Byrne. 1997. The Northeast Fisheries Science Center bottom trawl survey program. ICES C. M. 1997/Y33. 23 p.

Bajkov, A.D. 1935. How to estimate the daily food consumption of fish under natural conditions. Trans. Amer. Fish. Soc. 65:288-289.

Belleggia, M., Mabragaña, E., Figueroa, D.E., Scenna, L.B., Barbini, S.A., Díaz de Astarloa, J.M. 2008. Food habits of the broad nose skate, *Bathyraja brachyurops* (Chondrichthyes, Rajidae), in the south-west Atlantic. Sci. Mar. 72:701-710.

Braccini, J.M. 2008. Feeding ecology of two high-order predators from south-eastern Australia: the coastal broadnose and the deepwater sharpnose sevengill sharks. Mar. Ecol. Prog. Ser. 371:273-284.

Brodziak, J., J. Ianelli, K. Lorenzen, and R.D. Methot Jr. (editors). 2011. Estimating natural mortality in stock assessment applications. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/SPO-119, 38 pp.

Collette, B.B. and G. Klein-MacPhee. 2002. Bigelow and Schroeder's Fishes of the Gulf of Maine, Third Edition. Smithsonian Institution Press, Washington D.C.

Deroba, J.J. 2015. Atlantic herring operational assessment report 2015. US COC, NEFSC Ref Doc 15-16; 30p.

Deroba, J. J. 2018. Sources of variation in stomach contents of predators of Atlantic herring in the Northwest Atlantic during 1973–2014. ICES Journal of Marine Science, doi:10.1093/icesjms/fsy013.

Durbin, E.G., Durbin, A.G., Langton, R.W., Bowman, R.E. 1983. Stomach contents of silver hake, *Merluccius bilinearis*, and Atlantic cod, *Gadus morhua*, and estimation of their daily rations. Fish. Bull. 81:437-454.

Eggers, D.M. 1977. Factors in interpreting data obtain by diel sampling of fish stomachs. J. Fish. Res. Board Can. 34:290-294.

Elliot, J.M., Persson, L. 1978. The estimation of daily rates of food consumption for fish. J.

Anim. Ecol. 47:977-991.

Foote, K.G., Knudsen, H.P., Vestnes, G., MacLennan, D.N., Simmonds, E.J. 1987. Calibration of acoustic instruments for fish density estimation: A practical guide. ICES Cooperative Research Report 44. 69 pp.

Francis, R.I.C.C. 2011. Data weighting in statistical fisheries stock assessment models. CJFAS 68: 1124-1138.

Gerritsen, H.D., McGrath, D., and Lordan, C. 2006. A simple method for comparing age-length keys reveals significant regional differences within a single stock of haddock. ICES Journal of Marine Science 63: 1096-1100.

Hare, J.A., Morrison, W.E., Nelson, M.W., Stachura, M.M., Teeters, E.J., Griffis, R.B., Alexander, M.A., Scott, J.D., Alade, L., Bell, R.J., Chute, A.S., Curti, K.L., Curtis, T.H., Kircheis, D., Kocik, J.F., Lucey, S.M., McCandless, C.T., Milke, L.M., Richardson, D.E., Robillard, E., Walsh, H.J., McManus, M.C., Marancik, K.E., and Griswold, C.A. 2016. A Vulnerability Assessment of Fish and Invertebrates to Climate Change on the Northeast U.S. Continental Shelf. PLOS ONE **11**(2): e0146756. doi:10.1371/journal.pone.0146756.

Jech, J.M., 2014. Post-Processing of Scientific Echo-Sounder Data from the NOAA Ships Albatross IV and HB Bigelow: 1998 – 2012. NOAA NEFSC Technical Memorandum (*in press*).

Jech, J.M., W. Michaels, W. Overholtz, W. Gabriel, T. Azarovitz, D. Ma, K. Dwyer, and R. Yetter. 2000. Fisheries acoustic surveys in the Gulf of Maine and on Georges Bank at the Northeast Fisheries Science Center, in Proceedings of the Sixth International Conference on Remote Sensing for Marine and Coastal Environments, 1-3 May, Charleston, South Carolina, USA. Veridian ERIM International, Ann Arbor, Michigan, USA. Vol. 1, pp. 168-175.

Jech, J.M. and W.L. Michaels. 2006. A multifrequency method to classify and evaluate fisheries acoustics data. Can. J. Fish. Aquat. Sci. 63: 2225-2235.

Jech, J.M., and F. Stroman. 2012. Aggregative patterns of pre-spawning Atlantic herring on Georges Bank from 1999-2010. Aquat. Living Resourc., 25: 1-14.

Jech, J.M., and P.J. Sullivan. 2014. Distribution of Atlantic herring (*Clupea harengus*) in the Gulf of Maine from 1998 to 2012. Fish. Res. 156: 26-33.

Kitchell, J.F., Stewart, D.J., Weininger, D. 1977. Applications of a bioenergetics model to yellow perch (*Perca flavescens*) and walleye (*Stitzostedion vitreum vitreum*). J. Fish. Fes. Board Can. 34:1922-1935.

Koen Alonso, M., Crespo, E.A., García, N.A., Pedraza, S.N. 2002. Fishery and ontogenetic driven changes in the diet of the spiny dogfish, *Squalus acanthias*, in Patagonian waters, Argentina. Environ. Biol. Fish. 63:193-202.

Latour, R.J., Gartland, J., Bonzek, C.F., Johnson, R.A. 2008. The trophic dynamics of summer flounder (Paralichthys dentatus) in Chesapeake Bay. Fish. Bull. 106:47-57.

Legault, C. M., and V. R. Restrepo. 1999. A flexible forward age-structured assessment program. ICCAT Coll. Vol. Sci. Pap. 49(2): 246-253.

Link, J.S., Almeida, F.P. 2000. An overview and history of the food web dynamics program of the Northeast Fisheries Science Center, Woods Hole, MA. NOAA Tech. Memo. NMFS-NE-159, 60 p.

Link, J.S., Garrison, L.P. 2002. Changes in piscivory associated with fishing induced changes to the finfish community on Georges Bank. Fish. Res. 55:71-86.

Link, J.S., Garrison, L.P., Almeida, F.P. 2002. Interactions between elasmobranchs and groundfish species (Gadidae and Pleuronectidae) on the Northeast U.S. Shelf. I: Evaluating predation. N. Am. J. Fish. Mange. 22:550-562.

Methot, R.D. Jr., and Wetzel, C.R. 2013. Stock synthesis: a biological and statistical framework for fish stock assessment and fishery management. Fisheries Research 142: 86-99.

Miller, T.J., C. Das, P.J. Politis, A.S. Miller, S.M. Lucey, C.M. Legault, R.W. Brown, and P.J. Rago. 2010. Estimation of Albatross IV to Henry B. Bigelow calibration factors. Northeast Fisheries Science Center Reference Document, 10-05. 233 p.

Northeast Fisheries Center (NEFC). 1988. An evaluation of the bottom trawl survey program of the Northeast Fisheries Center. NOAA Tech. Memo. NMFS-F/NEC-52, 83 p.

Northeast Fisheries Science Center (NEFSC). 2007. 44th Northeast Regional Stock Assessment Workshop (44th SAW): 44th SAW assessment report. NEFSC Ref. Doc. 07-10; 661 p.

NEFSC. 2008. Assessment of 19 Northeast Groundfish Stock through 2007: Report of the 3rd Groundfish Assessment Review Meeting (GARM III), Northeast Fisheries Science Center, Woods Hole, MA, August 4-8, 2008. US DOC, NEFSC Ref Doc 08-15; 884p.

NEFSC. 2012. 54th Northeast Regional Stock Assessment Workshop (54th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 12-18; 600 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://www.nefsc.noaa.gov/nefsc/publications/

Northeast Fisheries Science Center (NEFSC). 2015. Operational Assessment of 20 Northeast Groundfish Stocks, Updated Through 2014. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 15-24; 251 p.

NEFSC. 2018. 64th Northeast Regional Stock Assessment Workshop (64th SAW) Assessment Report. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 18-04; 529 p. Available from: http://www.nefsc.noaa.gov/publications/

Nielsen, A., Berg, C.W. 2014. Estimation of time-varying selectivity in stock assessments using state-space models. Fisheries Research 158: 96-101.

Overholtz, W.J., Link, J.S., Suslowicz, L.E. 1999. Consumption and harvest of pelagic fishes in the Gulf of Maine-Georges Bank ecosystem: Implications for fishery management. Proceedings of the 16th Lowell Wakefield Fisheries Symposium – Ecosystem Considerations in Fisheries Management. AK-SG-99-01:163-186.

Overholtz, W.J., Link, J.S., Suslowicz, L.E. 2000. The impact and implications of fish production on pelagic fish and squid on the eastern USA shelf. ICES J. Mar. Sci. 57:1147-1159.

Overholtz, W.J., Link, J.S. 2007. Consumption impacts by marine mammals, fish, and seabirds on the Gulf of Maine-Georges Bank Atlantic Herring (*Clupea harengus*) complex during 1977-2002. ICES J. Mar. Sci. 64:83-96.

Pennington, M. 1985. Estimating the average food consumption by fish in the field from stomach contents data. Dana 5:81-86.

Reid, R.N., Almeida, F.P., Zetlen, C.A. 1999. Essential fish habit source document: fishery-independent surveys, data sources, and methods. NOAA Tech. Memo. NMFS-NE-122, 39 p.

Richards RA. 2016. 2016 Monkfish Operational Assessment. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 16-09; 109 p.

Richards, R.A., and Jacobson, L.D. 2016. A simple predation pressure index for modeling changes in natural mortality: application to Gulf of Maine northern shrimp stock assessment. Fisheries Research 179: 224-236.

Rivard, D. 1982. APL programs for stock assessment (revised). Can. Tech. Rep. Fish. Aquat.

Sci. 1091: 146 pp.

Smith, B.E., Link, J.S. 2010. The trophic dynamics of 50 finfish and 2 squid species on the northeast US continental shelf. NOAA Tech Memo. NMFS-NE-216, 640 p.

Taylor, M.H., Bascuñán, C. 2000. CTD data collection on Northeast Fisheries Science Center Cruises: Standard Operating Procedures. NEFSC Ref. Doc. 00-11; 28 p.

Taylor, M.H., Bascuñán, C., Manning, J.P. 2005. Description of the 2004 oceanographic conditions on the northeast continental shelf. NEFSC Ref. Doc. 05-03; 90 p.

Tsou, T.S., Collie, J.S. 2001a. Estimating predation mortality in the Georges Bank fish community. Can. J. Fish. Aquat. Sci. 58:908-922.

Tsou, T.S., Collie, J.S. 2001b. Predation-mediated recruitment in the Georges Bank fish community. ICES J. Mar. Sci. 58:994-1001.

Ursin, E., Pennington, M., Cohen, E.B., Grosslein, M.D. 1985. Stomach evacuation rates of Atlantic cod (*Gadus morhua*) estimated from stomach contents and growth rates. Dana 5:63-80.

Wigley, S.E., J. Blaylock, P.J. Rago, J. Tang, H.L. Haas, and G. Shield. 2011. Standardized bycatch reporting methodology 3-year review report – 2011 Part 1. Northeast Fisheries Science Center Reference Document 11-09.

B. Atlantic Herring – List of appendices

Appendix B1.

Herring ageing: the history and recent exchanges

Jonathan J. Deroba, Eric Robillard, Gary Shepherd, Matt Cieri

Introduction

Estimates of abundance (biomass), fishing mortality (*F*), recruitment, and management quantities (e.g., recommended yield) can be biased when using age-based stock assessments with imprecise or biased age information (Bradford 1991; Eklund et al., 2000; Reeves 2003; Bertignac and Pontual 2007; Yule et al., 2008). Imprecise ageing can cause estimates of abundance or *F* to be biased in scale, but not necessarily trend, while recruitment estimates may be biased in scale and falsely autocorrelated (Bradford 1991; Reeves 2003). Biased age information can cause estimates of abundance, *F*, and recruitment to be biased in scale and trend, and result in inappropriate catch and management advice (Bradford 1991; Eklund et al., 2000; Reeves 2003; Bertignac and Pontual 2007; Yule et al., 2008).

Atlantic herring *Clupea harengus* in the northwest Atlantic Ocean have been assessed using age-based stock assessment models (Anthony 1977; NEFSC 1993; NEFSC 1998; Overholtz et al., 2004). The age-based assessments rely on ages from multiple agencies. Commercial catch samples are aged by the Canadian Department of Fisheries and Oceans (DFO) and the Maine Department of Marine Resources (DMR). Survey catch samples are aged by the US National Marine Fisheries Service (NMFS). Periodic evaluations of the accuracy and precision of herring ageing, however, have revealed disagreements among ageing labs and potential biases (Dery and Chenoweth 1979; Overholtz et al., 2004; Libby et al., 2006). Results of an otolith exchange among agencies conducted during the 2003 stock assessment suggested that age readers from DMR and NMFS were generally in agreement, except for about a 10% difference for fish older than about age-4, with the NMFS reader concluding that fish were younger ("underageing") than the DMR reader (Overholtz et al., 2004). Significant differences of greater than 50% at some older ages were found between the DFO lab and both US facilities, with the DFO concluding that fish were younger than both US readers. An ageing workshop and second otolith exchange were conducted in 2006 (Libby et al., 2006). Generally, agreement among the ageing labs in the second otolith exchange was worse than during the 2003 assessment. Age readers from DMR and NMFS agreed 54% of the time and DMR tended to conclude that fish were younger than NMFS, which is the reverse of the discrepancies found in 2003. Age readers from DFO and NMFS agreed only 39% of the time and DFO generally concluded that fish were younger than NMFS. Similarly, DFO and DMR agreed 58% of the time, with DFO concluding that fish were younger than DMR.

While otolith exchanges provide information on ageing precision and differences among labs, they do not inform accuracy. Using bomb radiocarbon dating to evaluate herring ageing accuracy, Melvin and Campana (2010) concluded that herring aged six and older were often underaged, while ages of younger fish were relatively well determined. The inaccuracy of ageing for older fish is consistent with the results of the otolith exchanges that also found greater disparity at older ages. Since 2003, DFO and DMR have re-aged much of their historical catalogue using techniques agreed to during ageing workshops (Libby et al., 2006), but concerns about herring age accuracy and precision have lingered (NEFSC 2012).

During the 2012 Atlantic herring stock assessment, systematic differences were found between age-length keys (ALKs) from commercial samples aged by the DMR and survey samples aged by NMFS (see below; NEFSC 2012). One possible explanation for these differences is ageing errors. This manuscript describes work that has been done since 2012 to evaluate the potential for ageing errors in the Atlantic herring stock assessment.

Methods

Examinations of ageing data

Prior to the 2012 stock assessment, ALKs for herring from commercial and survey samples were combined to eliminate lengths for which no age data were available (i.e., “fill holes”), increase sample sizes and precision, and allow for survey age compositions to extend prior to 1987, the year when ages were first sampled for herring during NMFS surveys. Combining ALKs among gears, spatial areas, and time, however, can induce bias in the subsequent age compositions and stock assessments (Westrheim and Ricker 1978; Quist et al., 2012; Gerritsen et al., 2006). During the 2012 stock assessment, the practice of combining ALKs from commercial and survey sources was evaluated by plotting the proportion of fish at length assigned to each age by the commercial mobile gear fishery ALK in the first semester of each year (i.e., January-June) with the NMFS spring survey ALK for each year from 1987-2010 (NEFSC 2012). Using only commercial gear samples from the first semester of the year was intended to control for growth within the year that might affect the ALKs. These plots were then visually compared, with general consistency suggesting that ALKs could likely be combined, and inconsistency suggesting that ageing error or some other issue may be problematic and ALKs should not be combined. Consistency between the DMR and

NMFS ageing labs was evaluated by plotting the mean age in 5cm length bins in each year estimated from samples collected from the commercial mobile gear fishery in the first semester of each year with mean age estimated using NMFS spring survey samples. The same plots were created using samples collected from the commercial mobile gear fishery in the second semester of each year and the NMFS fall survey. Similarly, samples from all years were combined and mean age in 1 cm length bins was estimated from samples collected from the commercial mobile gear fishery in the first semester and plotted with mean age estimated using NMFS spring survey samples, and a similar plot was created using samples collected from the commercial mobile gear fishery in the second semester of all years and the NMFS fall survey. These plots were visually compared, with consistency suggesting no evidence of ageing error, but systematic differences suggesting the opposite conclusion.

Recent otolith exchanges

To make sure that all labs providing ages followed the same protocols, otoliths exchanges between labs occurred in 2014, 2016 and 2017. The following measures were used to characterize the results of tests of ageing consistency between the labs:

Coefficient of Variation (CV)

The mean coefficient of variation (CV, Campana *et al*. 1995, Chang 1982) is a relatively robust approach to quantifying agreement in fish ages. It yields results which are easier to compare between species and structures. Also, the contribution each fish makes to the CV is relative to the average age assigned to that fish; i.e., a 2-year error in ageing a young fish would increase the measure more than would a 2-year error in an older fish, as the percentage change in age is greater for younger ages.

The CV is based on the differences between the mean age and each given age for each fish, and then these values are averaged over the entire sample set. When two ages are assigned to each fish, the CV is calculated as follows:

$$CV = 100\% \times \frac{1}{N} \sum_{j=1}^{N} \frac{\sqrt{\sum_{i=1}^{2} (X_{ij} - X_j)^2}}{X_j}$$

where X_{ij} is the ith age for the jth fish, X_j is the mean age of the jth fish, and N is the sample size.

Campana (2001) indicates that many ageing laboratories around the world view CVs under 5% to be acceptable among species of moderate longevity and ageing complexity. His description applies to all the herring exchanges that have occurred since 2012.

Percent Agreement

The Fishery Biology Program has used this measure since the group's inception, and considers levels of over 80% to be adequate. It is calculated based on the percentage of ages agreed upon relative to the total number aged:

$$\text{Percent Agreement} = 100 \times \frac{\text{Number of agreements}}{\text{N}}$$

For this measure, an error in ageing a young fish changes the measure by the same amount as would a similar error for an old fish. Therefore, this statistic is harder to compare between samples sets with different age distributions.

Bowker's Test of Symmetry

For both types of precision test, a Bowker's test (Hoenig *et al.* 1995, Bowker 1948) was used to test for departures from symmetry within the age-frequency table. Such asymmetries indicate the presence of a bias, although the test has low sensitivity when few disagreements exist. Where ages differ from one another, the Bowker's test compares values on the age-frequency table which represent symmetric errors, such as the paired ages (3,4) and (4,3). If all such values are dissimilar, the test will return a significant P value.

This test statistic is calculated as a chi-square variable, as follows:

$$\chi^2 = \sum_{i=1}^{m-1} \sum_{j=i+1}^{m} \frac{(n_{ij} - n_{ji})^2}{n_{ij} + n_{ji}}$$

where m is the maximum age in the data set, and n_{ij} is the number of fish in the *i*th row and *j*th column (Hoenig *et al.* 1995, Bowker 1948). The value of the degrees of freedom is equal to the number of non-zero n_{ij}-n_{ji} comparisons in this calculation, to a maximum of m(m-1)/2.

Results

Examinations of ageing data

All plots of the proportion of fish at length assigned to each age for each year can be found in NEFSC (2012), and so only example plots were provided here. The proportion of fish at length assigned to each age was generally similar between the commercial mobile gear fishery ALK in the first semester of the year and the NMFS spring survey ALK from 1987-1992 (Figure 1-top). The proportion of fish at length assigned to each age was also similar for ages 1-2 during 1993-2010 (Figure 1-bottom). For ages 3 and older, however, the NMFS spring survey ALK generally assigned a larger proportion of fish to each age at smaller lengths and a smaller proportion of fish to each age at larger lengths than the commercial mobile gear fishery ALK during 1993-2010 (Figure 1-bottom).

Mean ages in 5cm length bins were generally similar for the DMR mobile commercial samples and NMFS spring and fall survey samples for the 15-19cm and 20-24cm bins (Figure 2). The exception was between the DMR mobile commercial samples from the second semester and the NMFS fall survey for the 15-19cm length bin during 1987-1991, when the NMFS fall survey mean ages were approximately one year less than the DMR samples (Figure 2).

Mean ages in 5cm length bins differed for the DMR commercial samples and the NMFS spring and fall survey samples for the 25-29cm and 30-35cm bins, and the differences trended among years (Figure 3). Although the severity of differences and trends varied among length bins and seasonal surveys, some patterns were similar. Mean age from the NMFS surveys were less than the mean ages from commercial samples from 1987 until the mid-1990s, similar from the mid-1990s to the early 2000s, and greater than the commercial samples for the remainder of the time series (Figure 3).

Mean ages in 1cm length bins for all years combined (1987-2013) were similar from the smallest bins until about 28cm, after which the survey mean ages were about 1-3 years less than the DMR commercial samples (Figure 4). Mean ages from the DMR commercial samples increased relatively smoothly with length, as might be expected from a von Bertalanffy growth curve, whereas the mean ages from the surveys suggested an irregular increase in age with length beginning at about age-6 (Figure 4).

Recent otolith exchanges

Following Campana's 2001 recommendations, ageing labs around the world consider to have acceptable ages if there is 80% or higher agreement and a CV of 5 % and under. All herring exchanges between the labs fit in this category, with had high percent agreement with low variation (figure 5). Of the seven exchanges, only one had a 73.3% agreement but the CV still met the standard with 3.72%. The average agreement and CV between the 7 exchanges was 83.78% and 2.1% respectively. Bowker's test showed there was a bias between DFO and NEFSC in 2014 and between Maine and NEFSC in 2016. There seemed to be no pattern to the bias has it was two different labs, and were followed by other exchanges in which the bias did not show up.

Conclusions

Results suggest some systematic differences between commercial and survey ages. Consistency among DMR and NMFS ageing labs was also worse for larger and older fish (Figures 1-4), which was consistent with results from previous ageing workshops on herring (Overholtz et al., 2004; Libby et al., 2006) and the work of Melvin and Campana (2010). Results also suggest that the severity of the problem, be it ageing error or some other source, may vary through time. The comparison of the ALKs from commercial and survey sources (Figure 1) and mean age in various length bins (Figures 2-3) show temporal trends. The plots of mean age in various length bins suggest greater ageing discrepancies from about 1987 to the mid-90s, better agreement from the mid-90s to mid-00s, and increased discrepancies in recent years.

This research cannot definitively conclude that ageing error or differences in ageing methods is a problem, but other explanations for the patterns in the data seem unlikely. One alternative is that cohorts of herring school independently of each other, such that the mean age at length of fish from one school would differ from the mean age at length of catch from another school. For this to be a valid explanation, however, the pattern of mean age in 1cm length bins (Figure 4) would require that the NMFS randomized survey systematically misses schools from older cohorts, while a commercial fishery targets schools of older fish. This explanation is unlikely, especially considering that schools of older fish would likely be smaller than schools of younger fish, and therefore inefficient for the fishery to target. This age-based segregation has also never been observed in Atlantic herring in this area. The NMFS survey catches could be detecting signals from cohorts outside of the Georges Bank/Gulf of Maine complex. This explanation, however, is also unlikely because the confounding in the signal does not seem to start until older ages and the fishery operates over much of the same area as the NMFS surveys.

Results of recent otolith exchanges suggest consistent aging methodologies and generally trustworthy ages, regardless of source. Ultimately, combining ALKs from different sources should be abandoned for Atlantic herring, especially in previous years where no explanation is available for discrepancies in the data.

References

Anthony, V.C. 1977. June 1977 Assessments of herring from the Gulf of Maine and Georges Bank areas. Northeast Fisheries Science Center Reference Document 77-16, p. 17. http://nefsc.noaa.gov/publications/series/

Bertignac, M., and de Pontual, H. 2007. Consequences of bias in age estimation on assessment of the northern stock of European hake and on management advice. ICES Journal of Marine Science 64: 981-988.

Bradford, M.J. 1991. Effects of ageing errors on recruitment time series estimated from sequential population analysis. Canadian Journal of Fisheries and Aquatic Sciences 48: 555-558.

Deroba, J.J. 2014. Evaluating the consequences of adjusting fish stock assessment estimates of biomass for retrospective patterns using Mohn's rho. North American Journal of Fisheries Mangement 34: 380-390.

Campana, S.E. 2001. Accuracy, precision, and quality control in age determination, including a review of the use and abuse of age validation methods. J. Fish Bio. 59:197-242.

Dery, L., and Chenoweth, J. 1979. Recent problems in ageing sea herring from the Gulf of Maine. Northeast Fisheries Science Center Reference Document 79-38, p. 7. http://nefsc.noaa.gov/publications/series/

Eklund, J., Parmanne, R., and Aneer, G. 2000. Between-reader variation in herring otolith ages and effects on estimated population parameters. Fisheries Research 46: 147-154.

Francis, R.I.C.C. 2011. Data weighting in statistical fisheries stock assessment models. Canadian Journal of Fisheries and Aquatic Sciences 68: 1124-1138.

Gerritsen, H.D., McGrath, D., and Lordan, C. 2006. A simple method for comparing age-length keys reveals significant regional differences within a single stock of haddock. ICES Journal of Marine Science 63: 1096-1100.

Libby, D.A., Burnett, J.M., and Melvin, G.D. 2006. Proceedings of the Atlantic herring otolith age estimation workshop, 10-11 January 2006, West Boothbay Harbor, Maine. Transboundary Resource Assessment Committee Working Paper 2006/04. p. 18. http://www2.mar.dfo-mpo.gc.ca/science/trac/rd.html

Melvin, G.D., and Campana, S.E. 2010. High resolution bomb dating for testing the accuracy of age interpretations for a short-lived pelagic fish, the Atlantic herring. Environmental Biology of Fishes 89: 297-311.

Mohn, R. 1999. The retrospective problem in sequential population analysis: an investigation using cod fishery and simulated data. ICES Journal of Marine Science 56: 473-488.

NEFSC. 1993. Report of the 16th northeast regional stock assessment workshop (16th SAW). Northeast Fisheries Science Center Reference Document 93-18, p. 120. http://nefsc.noaa.gov/publications/series/

NEFSC. 1998. 27th northeast regional stock assessment workshop (27th SAW). Northeast Fisheries Science Center Reference Document 98-14, p. 80. http://nefsc.noaa.gov/publications/series/

NEFSC. 2012. 54th Northeast Regional Stock Assessment Workshop (54th SAW) Assessment Report. Northeast Fisheries Science Center Reference Document 12-18, p. 600. http://nefsc.noaa.gov/publications/series/

Overholtz, W.J., Jacobson, L.D., Melvin, G.D., Cieri, M., Power, M., Libby, D., Clark, K. 2004. Stock assessment of the Gulf of Maine-Georges Bank Atlantic herring complex, 2003. Northeast Fisheries Science Center Reference Document 04-06, p. 290. http://nefsc.noaa.gov/publications/series/

Quist, M.C., Pegg, M.A., and DeVries, D.R. 2012. Age and growth. Pages 677-731 in A.V. Zale, D.L. Parrish, and T.M. Sutton, editors. Fisheries Techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Reeves, S.A. 2003. A simulation study of the implications of age-reading errors for stock assessment and management advice. ICES Journal of Marine Science 60: 314-328.

Stewart, I.J., and Martell, S.J.D. 2014. A historical review of selectivity approaches and retrospective patterns in the Pacific halibut assessment. Fisheries Research 158: 40-49.

Westrheim, S.J., and Ricker, W.E. 1978. Bias in using an age-length key to estimate age-frequency distributions. Journal of the Fisheries Research Board of Canada 35: 184-189.

Yule, D.L., Stockwell, J.D., Black, J.A., Cullis, K.I., Cholwek, G.A., and Myers, J.T. 2008. Transactions of the American Fisheries Society 137: 481-495.

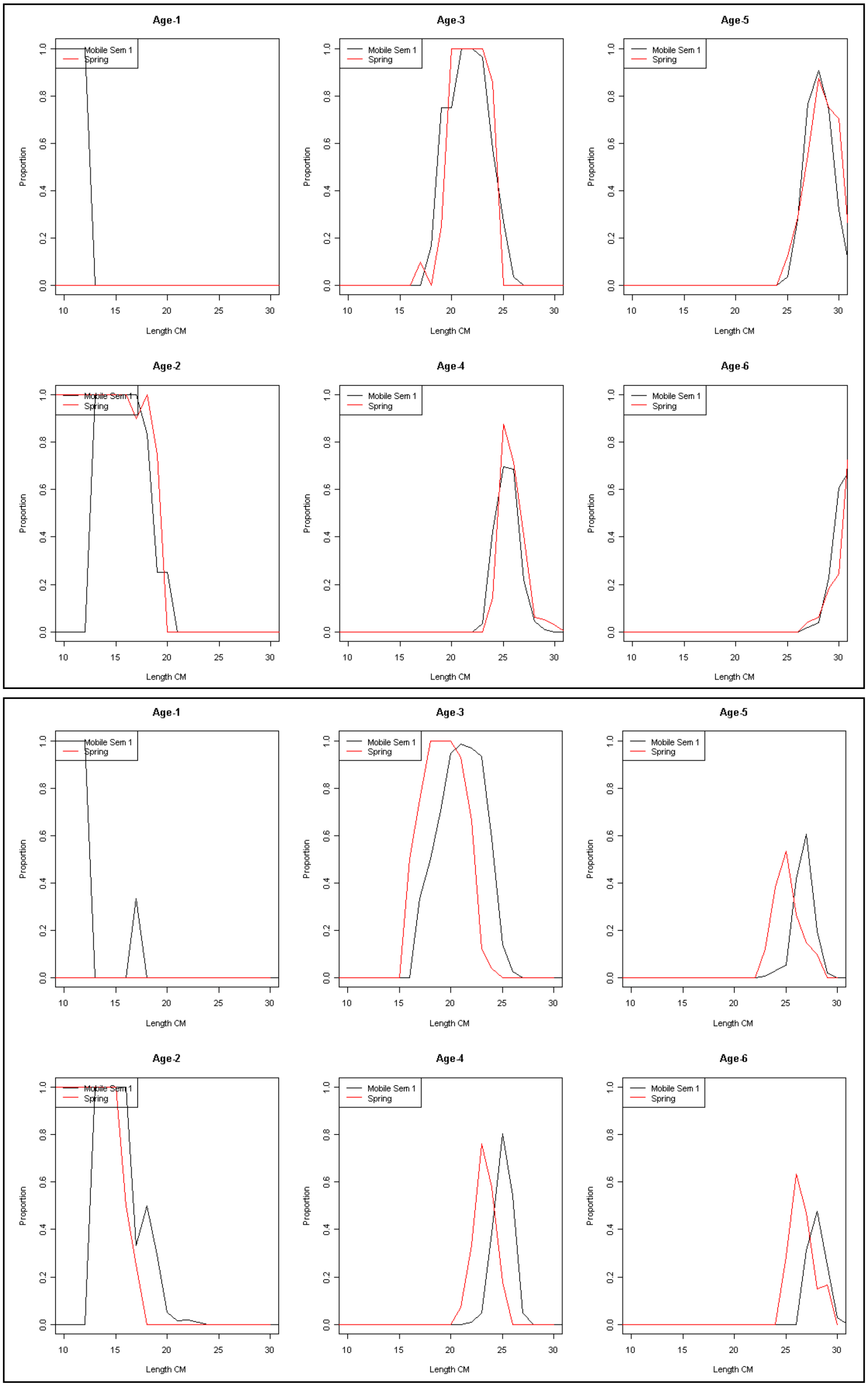
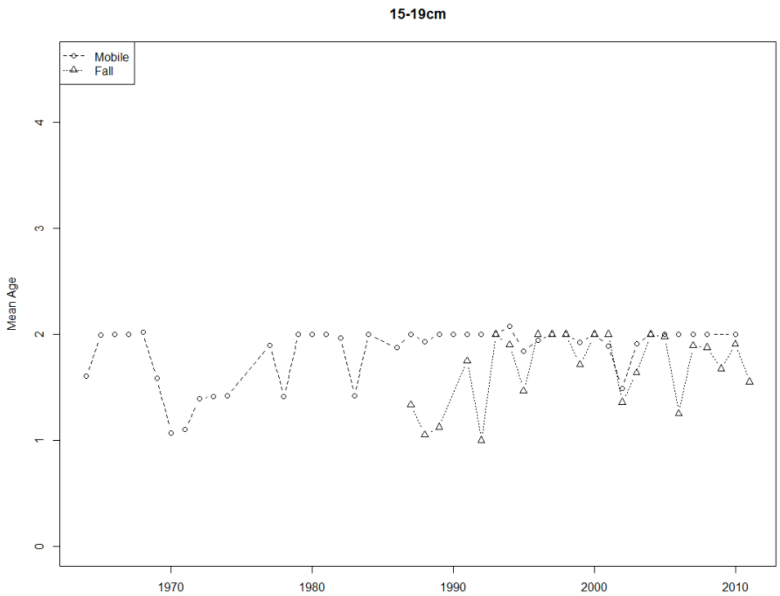


Figure 1. The proportion of fish at length assigned to each age using the commercial mobile gear fishery ALK from semester one of each year (black line) and the NMFS spring survey ALK (red line) in 1988 (top) and 1997 (bottom).



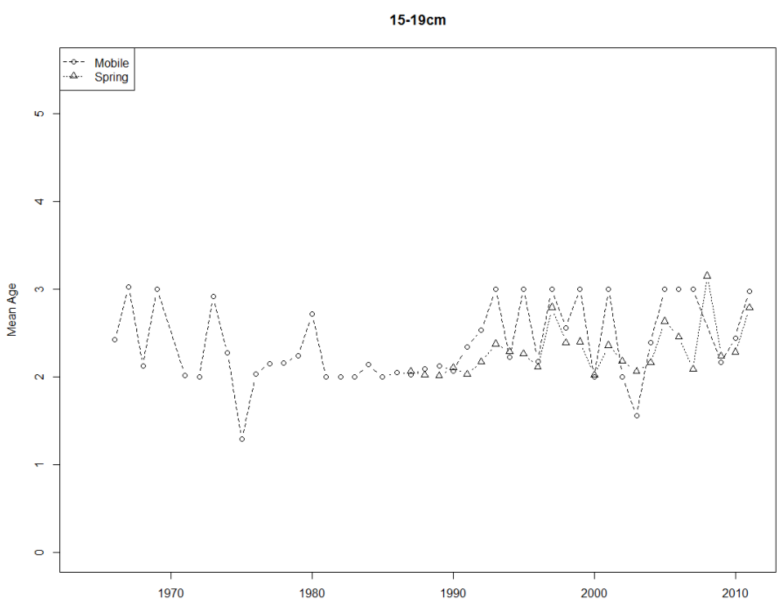
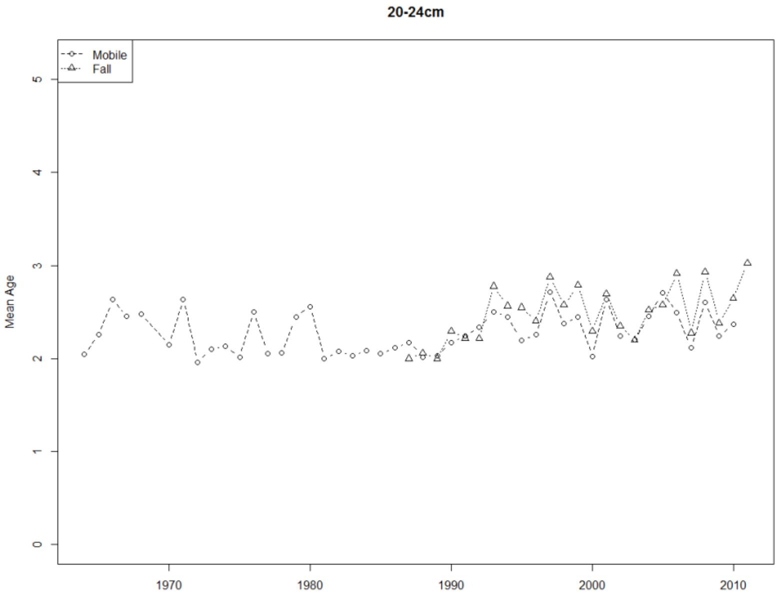


Figure 2. Mean age of herring in 5cm length bins for mobile commercial samples from semester one of each year and NMFS spring survey samples or mobile commercial samples from semester two of each year and NMFS fall survey samples during 1987-2011.



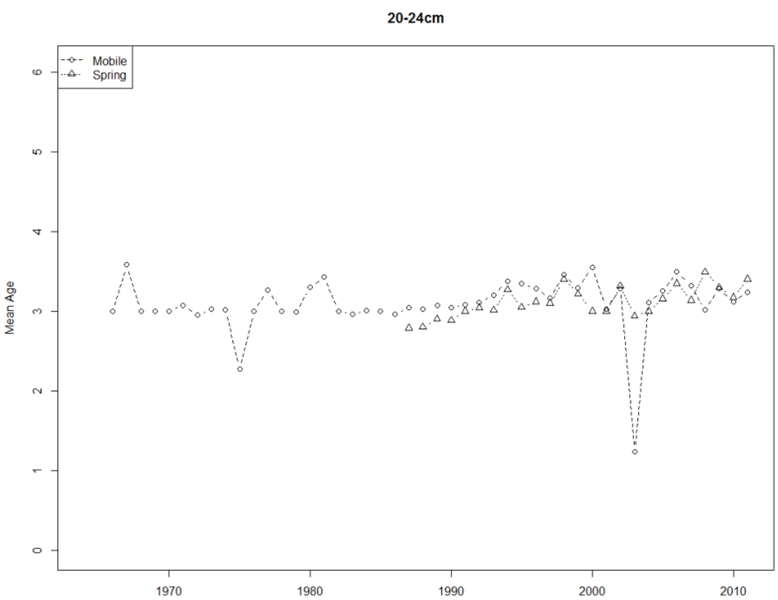
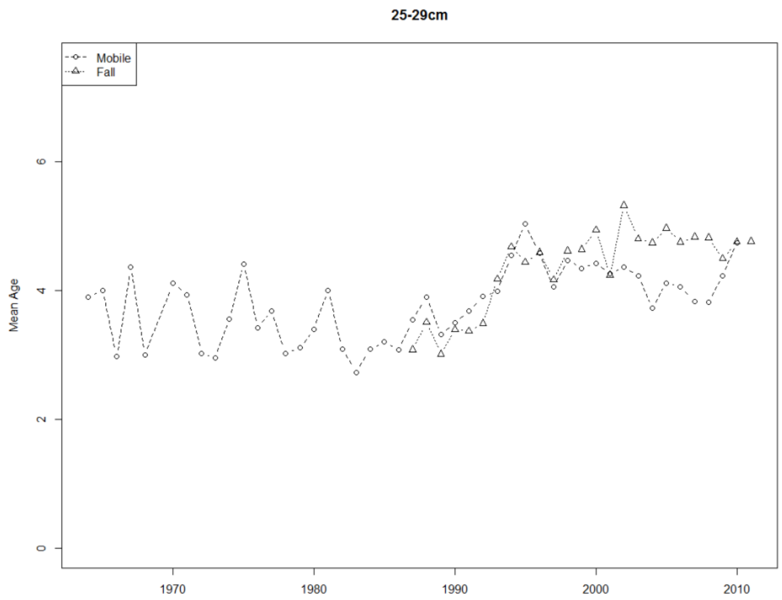


Figure 2 (continued). Mean age of herring in 5cm length bins for mobile commercial samples from semester one of each year and NMFS spring survey samples or mobile commercial samples from semester two of each year and NMFS fall survey samples during 1987-2011.



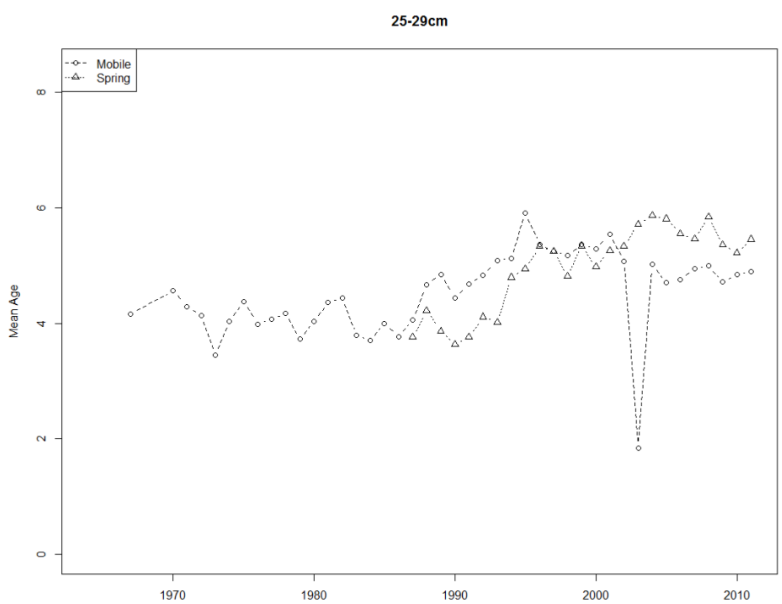
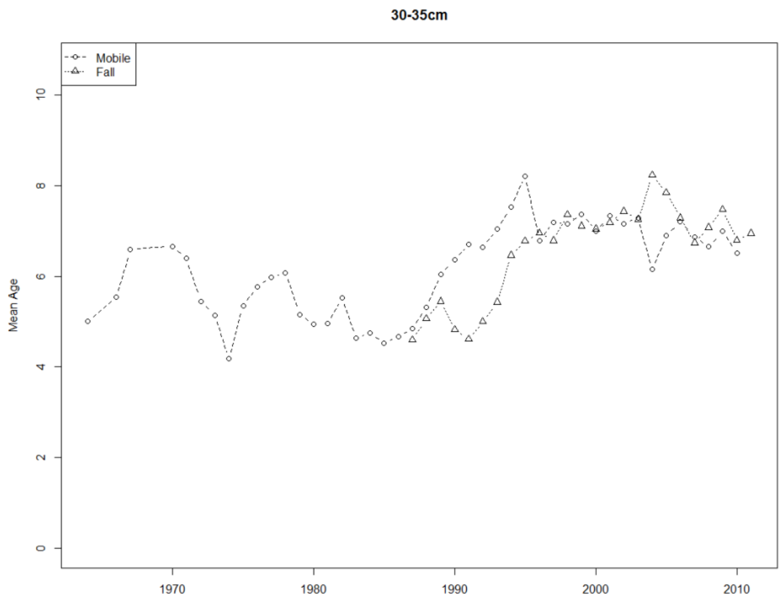


Figure 3. Mean age of herring in 5cm length bins for mobile commercial samples from semester one of each year and NMFS spring survey samples or mobile commercial samples from semester two of each year and NMFS fall survey samples during 1987-2011.



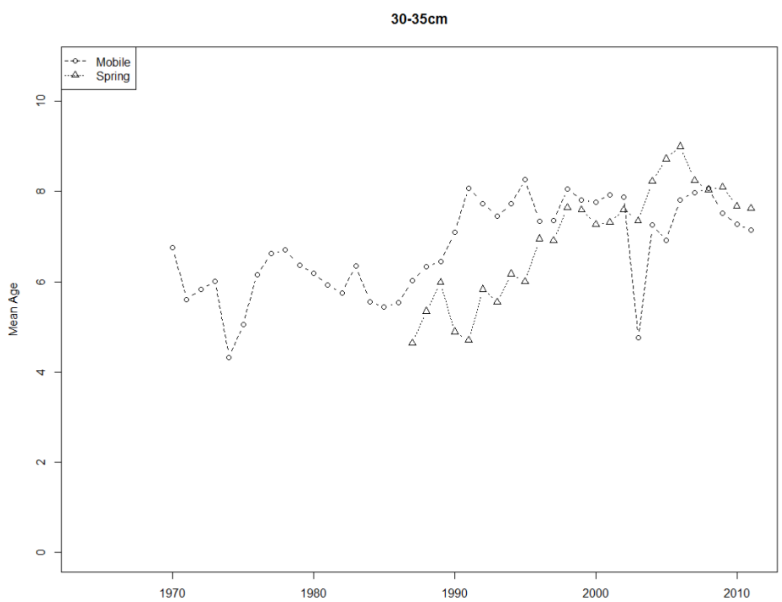


Figure 3 (continued). Mean age of herring in 5cm length bins for mobile commercial samples from semester one of each year and NMFS spring survey samples or mobile commercial samples from semester two of each year and NMFS fall survey samples during 1987-2011.

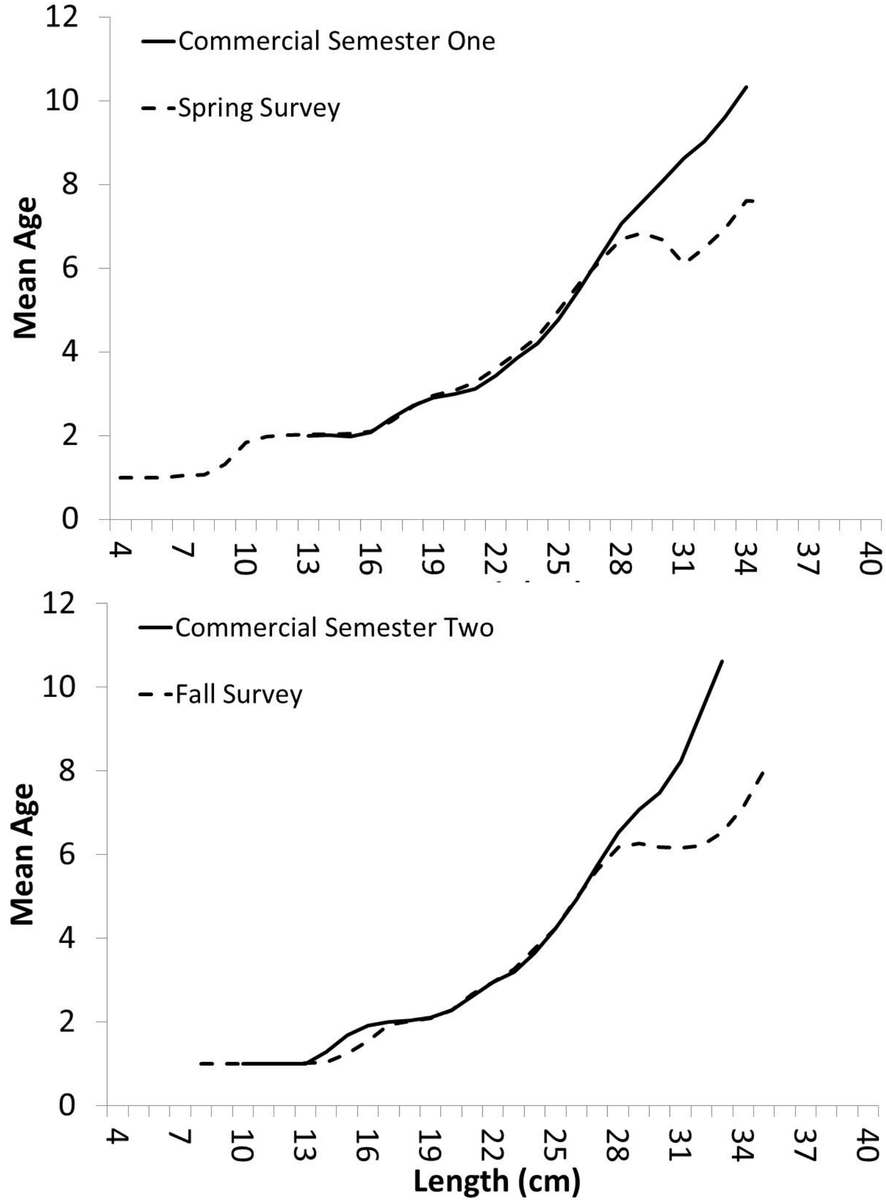


Figure 4. Mean age of herring in 1cm length bins for mobile commercial samples from semester one and NMFS spring survey samples or mobile commercial samples from semester two and NMFS fall survey samples for all years combined from 1987-2013.

Figure 5. Herring otoliths exchanges between labs

Date	Who	%	CV	Bias
9/29/2014	Maine vs DFO	91.3	1.08	no
9/29/2014	DFO vs NEFSC	80.2	2.8	yes
9/29/2014	Maine vs NEFSC	89	1.57	no
6/1/2016	Maine vs NEFSC	85.6	2.32	yes
7/1/2016	Maine vs NEFSC	73.3	3.72	no
8/1/2017	NEFSC vs Ref	83.3	3.46	no

Appendix B2

A State-Space Stock Assessment Model (SAM) for Gulf of Maine – Georges Bank Atlantic Herring

Jonathan J. Deroba
NOAA NMFS NEFSC
Woods Hole, MA 02543

Introduction

Fish stock assessments rely on observations (e.g., survey indices, catch, age composition) to inform fishing, survival, and reproduction processes (e.g., fishing mortality, selectivity). The observations and the processes are both subject to error. Observations are collected through sampling procedures that are subject to measurement error, while some processes like selectivity and survival are not directly observed and so are subject to process errors not reflected in the observed data.

Stock assessment approaches vary in the degree to which observation and process errors are acknowledged. Virtual population analyses do not allow any observation or process errors because data are assumed perfectly known. Statistical catch-at-age (SCAA) models permit observation errors and limited process error in recruitment, but the extent of the errors are user specified and the models estimate relatively many parameters (e.g., a fishing mortality rate and recruitment for each year). State-space models can separate observation and process errors using relatively few parameters (Nielsen and Berg 2014). This efficiency is achieved by estimating the variances of the assumed distributions for the observation and process errors, and the fishing mortality and abundance states are predictions from the assumed distributions, as opposed to free parameters as in SCAA models.

The objective of this working document was to apply a SAM model to Gulf of Maine – Georges Bank Atlantic herring. I provide an overview of the model here, but details can be found in Nielsen and Berg (2014) and Berg and Nielsen (2016). Notation generally follows that of Nielsen and Berg (2014).

Methods

Observations

Catch and index observations are assumed to have lognormal errors, with separate variance parameters applied to different user selected age groups:

$$\log(C_{a,y}) = \log\left(\frac{F_{a,y}}{Z_{a,y}}(1 - e^{-Z_{a,y}})N_{a,y}\right) + e_{a,y}^{(o)} ;$$

$$e_{a,y}^{(o)} \sim N(0, \hat{\sigma}_{o,a}^{2}) ;$$

$$\log(I_{a,y}) = \log(\hat{q}N_{a,y}) + e_{a,y}^{(s)} \;;$$

$$e_{a,y}^{(s)} \sim N(0, \hat{\sigma}_{s,a}^2) \;.$$

Age groups were defined to share variance parameters based on AIC and residual patterns.

Processes

SAM allows for process errors in recruitment, survival between sequential ages, and age specific fishing mortality rates. The recruitment and survival processes are assumed to follow lognormal distributions:

$$\log(R_{a=1,y}) = \log\left(f(SSB_{y-1} \; or \; R_{a=1,y-1})\right) + \gamma_{a=1,y} \;;$$

$$\gamma_{a=1,y} \sim N(0, \hat{\sigma}_R^2) \;;$$

$$\log(N_{a,y}) = \log(N_{a-1,y-1}) - F_{a-1,y-1} - M_{a-1,y-1} + \gamma_{a>1,y} \;;$$

$$\gamma_{a>1,y} \sim N(0, \hat{\sigma}_{a>1}^2) \;.$$

Recruitment in all model runs was assumed to follow a random walk. As with the observation variances, age groups were defined to share survival process variance parameters based on AIC and residual patterns.

Fishing mortality rates can be age-specific or groups of ages can be coupled to share fishing mortality rates, and these rates follow a random walk between years. The random walk fishing mortality rates can be correlated among the age couplings, for example, with a correlation of 0.0 producing independent random walks among age couplings and a correlation of 1.0 producing parallel time trajectories in fishing mortality rates among age couplings (i.e., time invariant selectivity). This results in age- and year-specific random walk increments following a multivariate normal distribution:

$$\log(F_{a,y}) = \log(F_{a,y-1}) + \delta_y \;;$$

$$\delta_y \sim \boldsymbol{N}(0, \widehat{\boldsymbol{E}}) \;.$$

The degree of correlation in the random walks can be fixed at 0.0 (i.e., independent) or estimated, and both were attempted. Age groups were defined to share fishing mortality states and process variances based on AIC and residual patterns.

Input Data

The input data were similar to that used in the ASAP base model, but SAM can only fit to age-based indices or indices of SSB. That is, SAM cannot fit to annual, aggregate index observations with user specified selectivity. Consequently, the SAM model only fit to NMFS spring, fall, and summer (shrimp) bottom trawl surveys for the years 1987-2017. In summary, input data were:

- Catches-at-age for ages 1-8+, with age 8 as a plus group, for the years 1965-2018.
- The NMFS spring and fall bottom trawl surveys for ages 2-8+ from years that used the vessel Albatross, 1987-2008.
- The NMFS spring and fall bottom trawl surveys for ages 2-8+ from years that used the vessel Bigelow, 2009-2017.
- The NMFS summer (shrimp) bottom trawl survey for ages 2-8+ from 1987-2017.
- Natural mortality equaled 0.35 for all ages and years.
- Age- and year-specific maturity was the same as the base ASAP model, as were weights at age.

Results

More than 20 models were run in the development of the SAM model. Presenting the AIC values and diagnostic plots that led to the final model structure would be voluminous. Consequently, only the final model structure is described. Supporting figures are at the end of this document.

Observations

Two separate observation variances were estimated for fishery catches, one that applied to ages 1-6 and another applied to ages 7-8+.

The spring NMFS survey for the Albatross years had separate catchabilities for age 2, 3, 4-6, and 7-8+, and different observation variances for age 2, 3-6, and 7-8+. The fall NMFS survey for the Albatross years had separate catchabilities for age 2, 3, 4, and 5-8+, and different observation variances for age 2-6 and 7-8+. The spring NMFS survey for the Bigelow years had separate catchabilities for age 2, 3, 4, and 5-8+, and different observation variances for age 2-3, 4-8+. The fall NMFS survey for the Bigelow years had separate catchabilities for age 2, 3, 4-6, and 7-8+, and a single observation variance that applied to all ages. The summer NMFS survey had separate catchabilities for age 2, 3, 4, 5, 6, and 7-8+, and different observation variances for age 2, 3-7, and 8+.

Processes

Unique fishing mortality rates were specified for age-1, age-2, age-3, age-4, age-5, age-6, and ages 7-8+. The fishing mortality rates were assumed to follow independent random walks. A model that estimated the degree of correlation among the fishing mortality rates improved the model fit based on log-likelihood, but did not resolve any residual patterns and so this parameter was not estimated.

Separate process variances were estimated for the fishing mortality rates at age 1, 2-4, 5-6, and 7-8+. Process variance in recruitment was estimated separately from a survival process variance shared among ages 2-8+.

Summary of Final SAM Model Structure

- Two fishery catch observation variances (2 parameters).
- Eleven observation variances among all the surveys (11 parameters).
- 22 catchability parameters among all the surveys (22 parameters).
- Four fishing mortality rate process variances (4 parameters).
- Process variance for recruitment and a survival process variance for ages 2-8+ (2 parameters).
- 41 total parameters.

Overview of Final SAM Model Estimates and Results ("Run 13")

The time-varying fishing mortality rates suggested a generally flat-topped selectivity, with ages 7-8+ having the highest fishing mortality rates in most years and age 1 having the lowest selectivity in all years. The fishing mortality rates and subsequent selectivity at ages 2-6, however, were relatively variable. Age-2 had a relatively high selectivity in the 1970s due to higher catches from fixed gear sources during those years, but has since declined as mobile gears have become more dominant. Selectivity at other ages changes through time in a near parallel and cyclic manner.

Fishing mortality rates at age 1 had the largest of the process variances, followed by recruitment. Observation variances differed among ages and data sources.

The model did not exhibit a retrospective pattern. Fitting the model without each of the surveys resulted in time series that were withing the 95% confidence intervals of the base SAM model. Fits to the catch and survey observations varied by data source, with relatively few patterns visible for some inputs (e.g., spring Albatross years), but year effects evident for some surveys (e.g., summer survey). Process residuals did not have any obvious patterns.

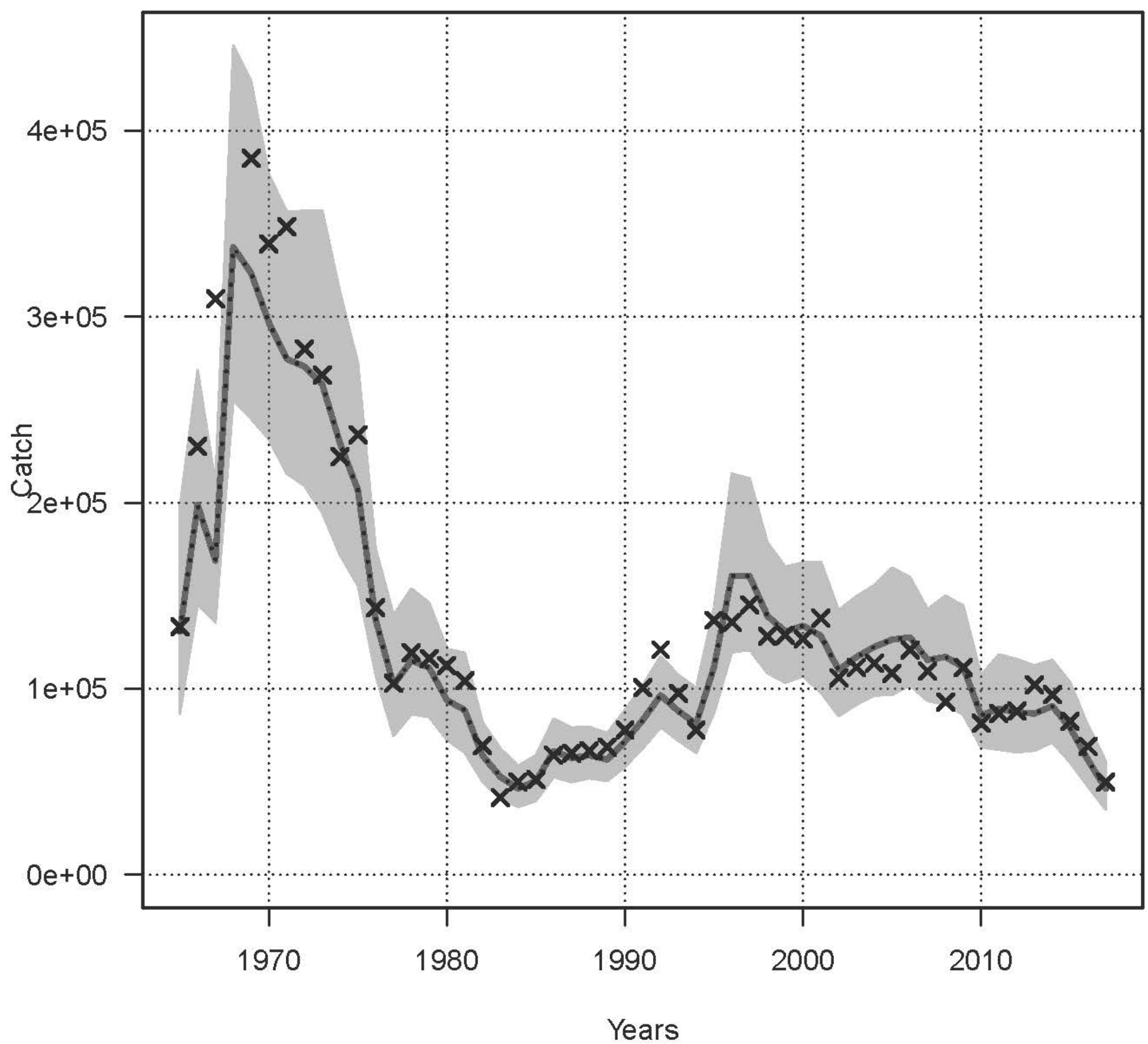
Time series estimates of recruitment, fishing mortality rate, and biomass (abundance) were generally similar to the final ASAP run.

Maximum sustainable yield proxy reference points were calculated using similar methods as for the base ASAP model. More specifically, the 5-year average of life history traits from 2013-2017 (e.g., maturity, weights-at-age) were used to calculate $F_{40\%}$ as an F_{MSY} proxy. Given that selectivity varies through time in the SAM model, the selectivities at age from 2013-2017 were also averaged for purposes of reference point calculation. Consequently, the $F_{40\%}$ value is not identical to that produced by the base ASAP model, nor is the corresponding B_{MSY} proxy. The B_{MSY} proxy was determined for the SAM model by conducting a 50 year projection at $F_{40\%}$, which was of sufficient length for the projection to reach equilibrium. Projected recruitments each year were resampled from the full time series of recruitment estimates from the SAM model. Various aspects about how process variance is carried forward in projections for the SAM model were not clear to the Working Group, and best practices for reference point calculation from a state-space model have not been developed. Consequently, the reference points and stock status from the SAM model should be used only for informational purposes and not considered for use in management. The $F_{40\%}$ MSY proxy equaled 0.39 and the corresponding B_{MSY} proxy equaled 197,000mt. Based on the SAM model, the stock is overfished and overfishing is occurring. Measures of uncertainty about stock status, however, were not readily available, but the uncertainty would likely be larger than that from the ASAP model due to the inclusion of process errors in SAM.

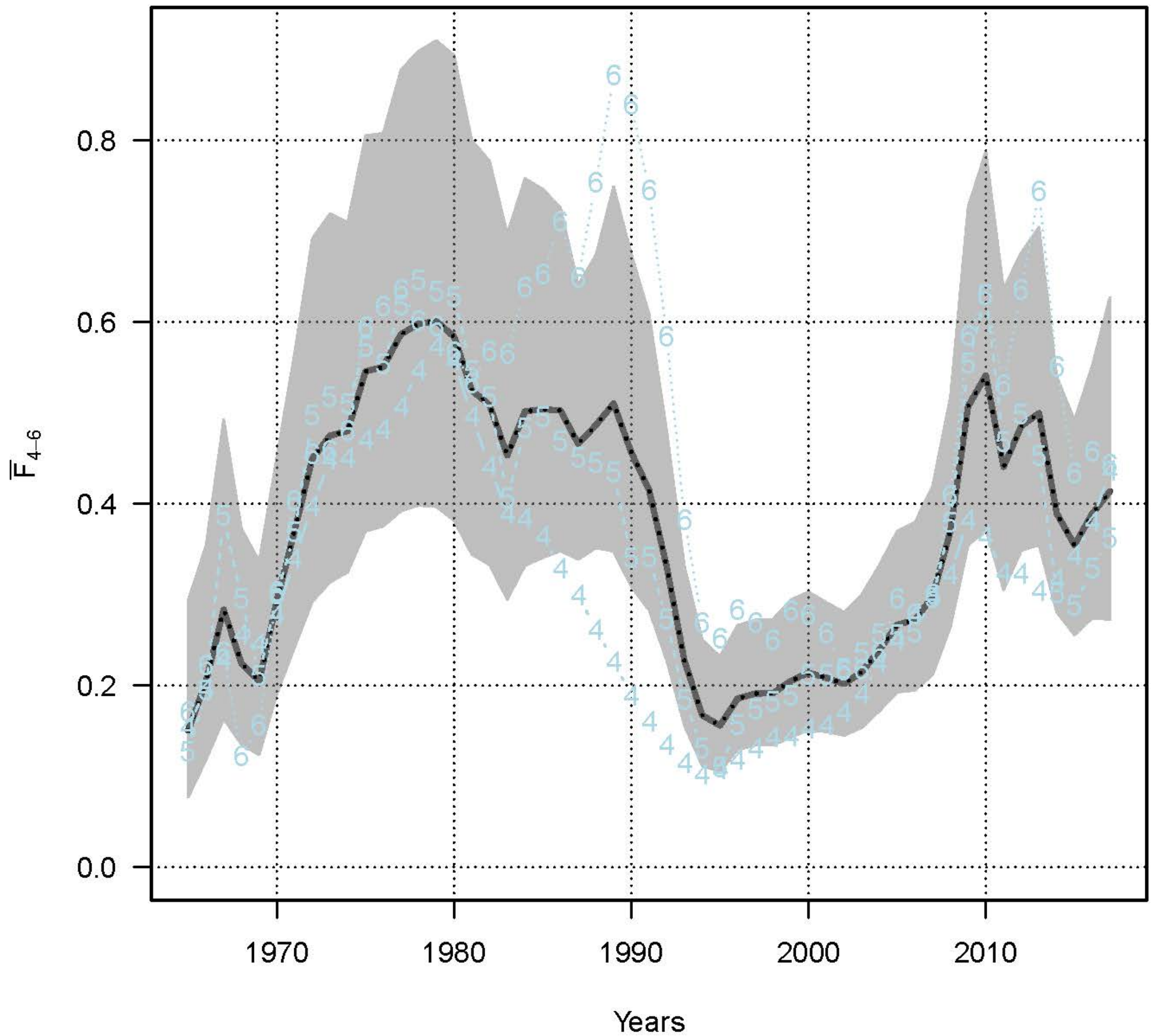
References

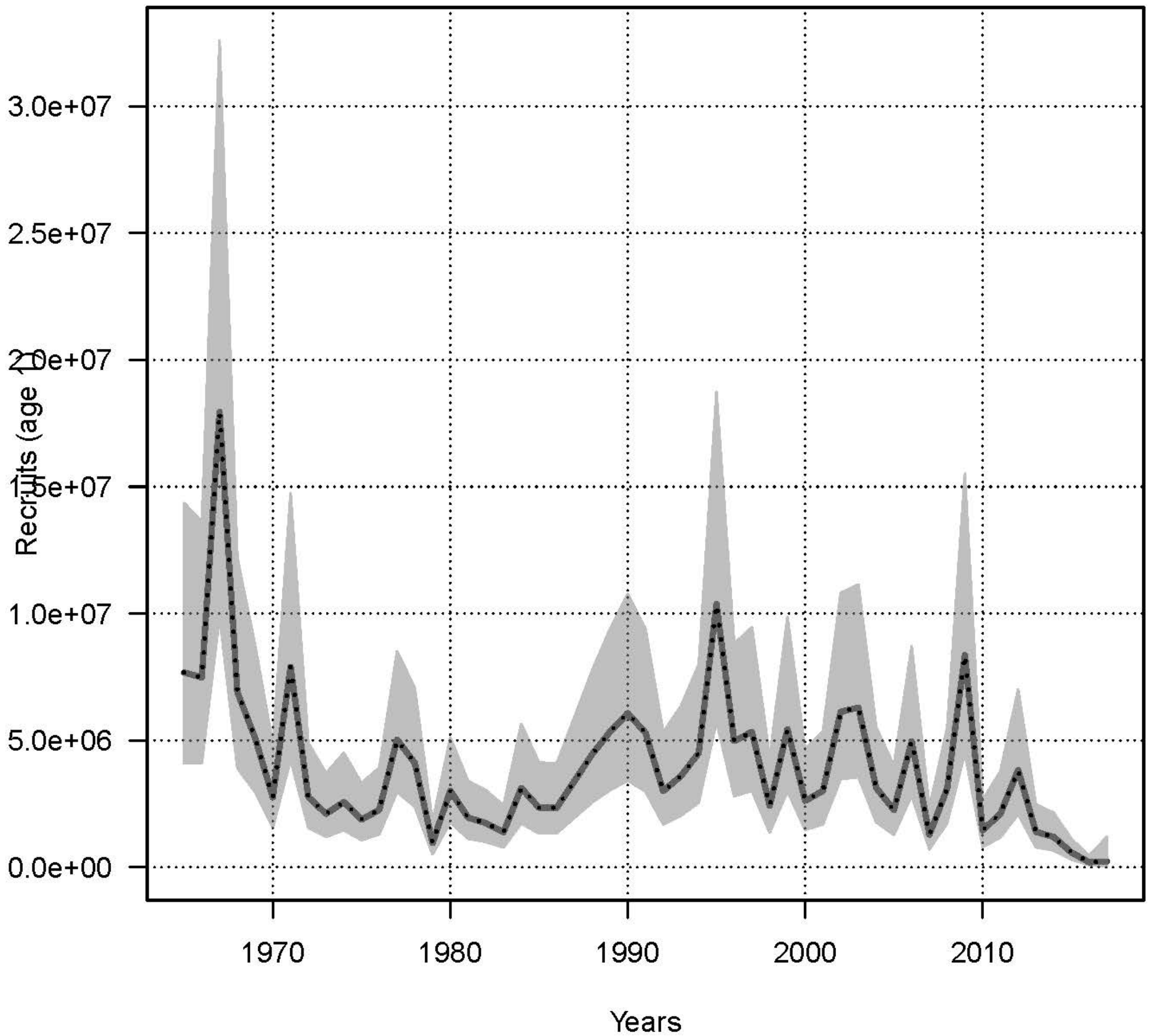
Berg, C.W., and Nielsen, A. 2016. Accounting for correlated observations in an age-based state-space stock assessment model. ICES Journal of Marine Science 73: 1788-1797.

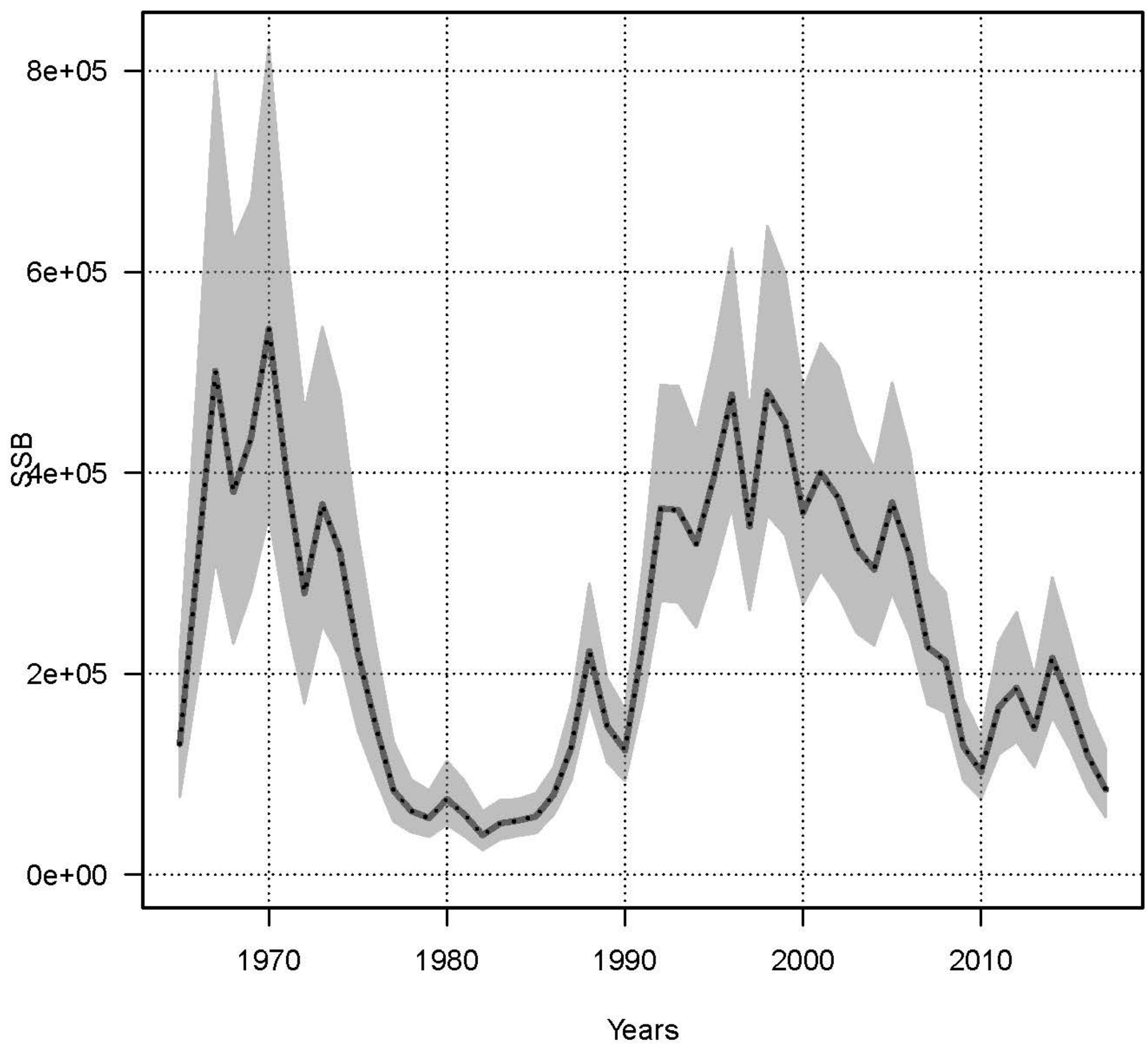
Nielsen, A., and Berg, C.W. 2014. Estimation of time-varying selectivity in stock assessments using state-space models. Fisheries Research 158: 96-101.

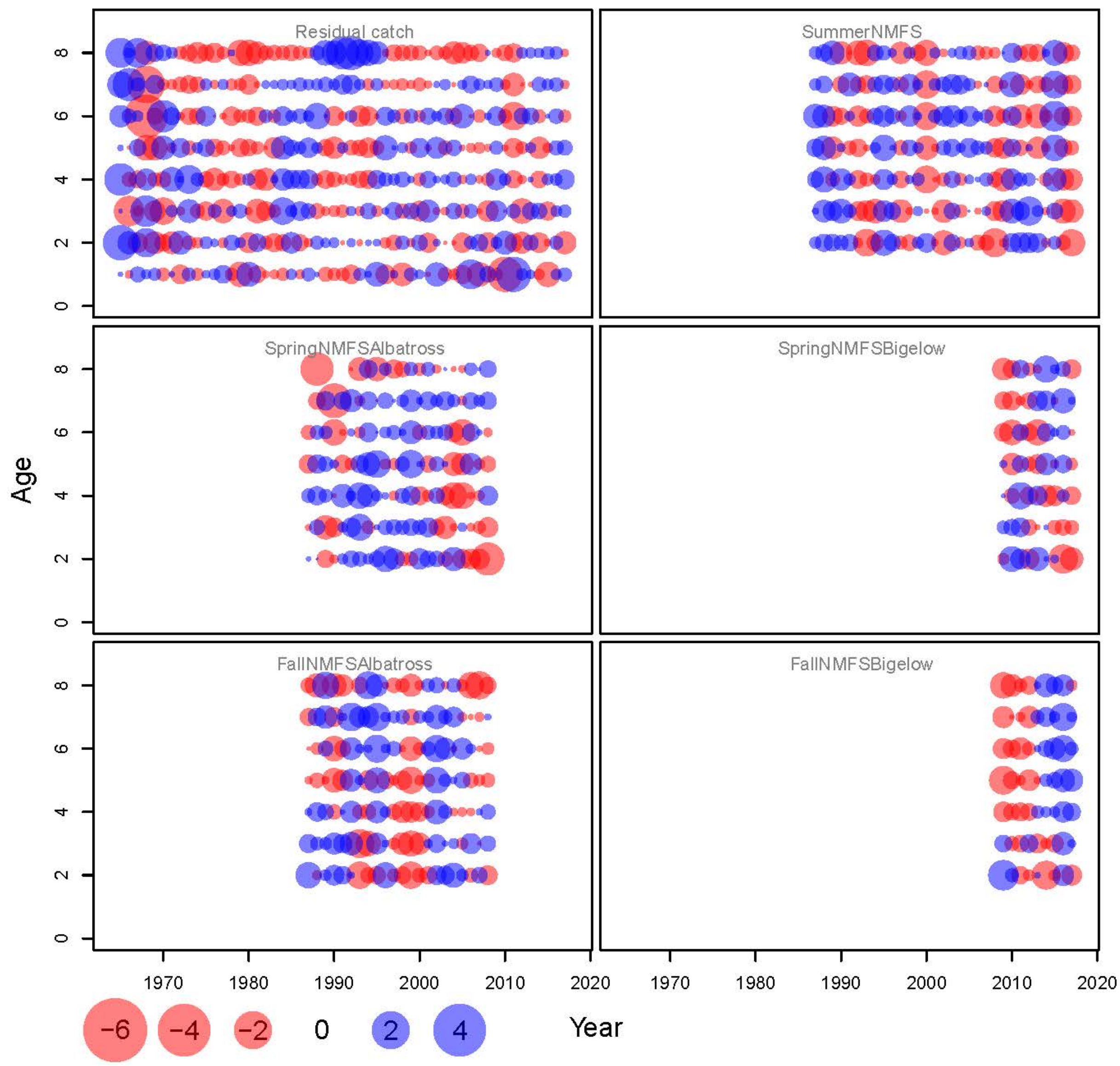


4e+05
3e+05
2e+05
1e+05
0e+00
Catch
1970
1980
1990
2000
2010
Years

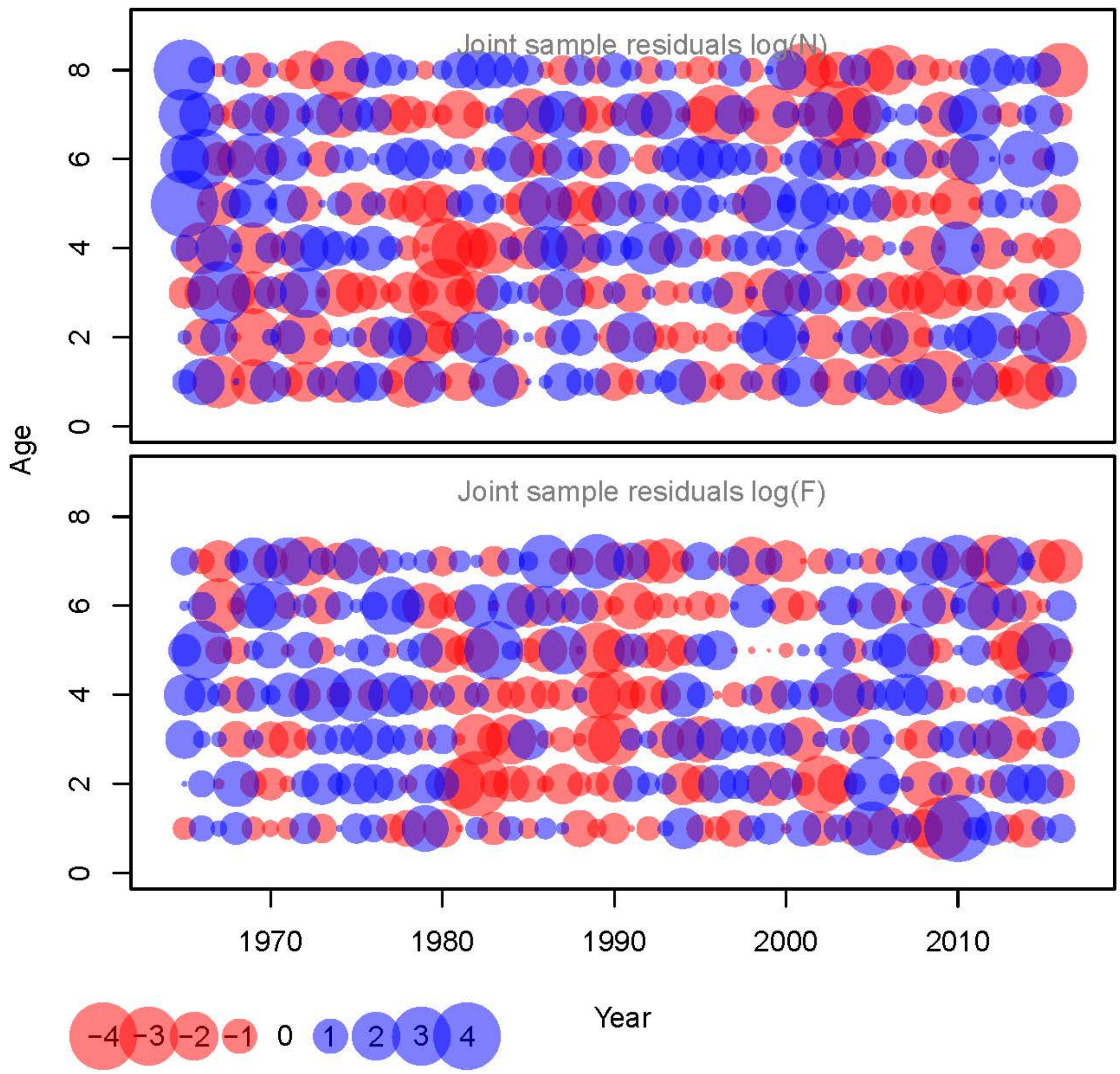

0.8
0.6
0.4
0.2
0.0
$\overline{F}_{4-6}$
1970
1980
1990
2000
2010
Years


3.0e+07
2.5e+07
2.0e+07
1.5e+07
1.0e+07
5.0e+06
0.0e+00
Recruits (age 1)
1970
1980
1990
2000
2010
Years

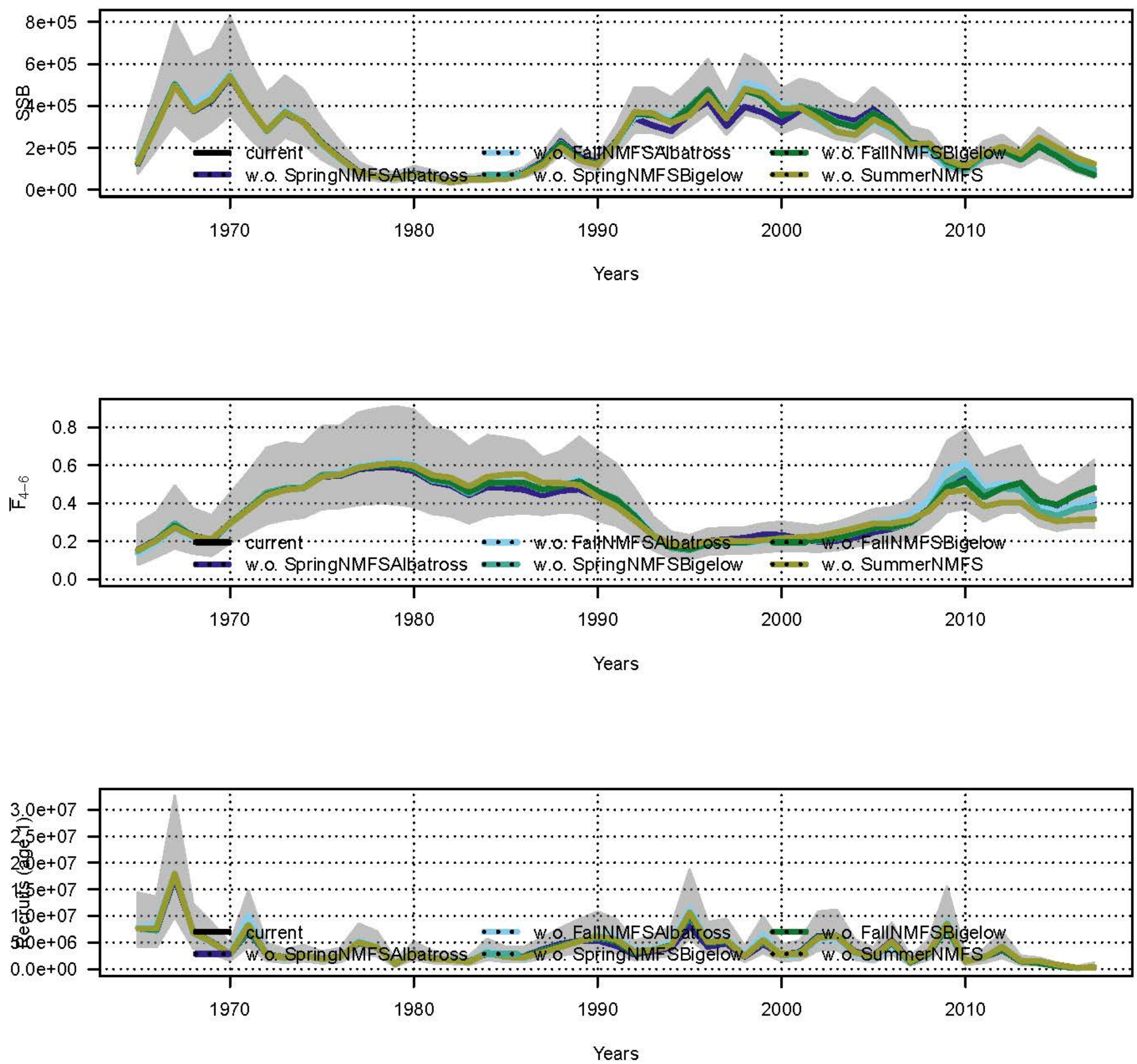

8e+05
6e+05
4e+05
2e+05
0e+00
SSB
1970
1980
1990
2000
2010
Years

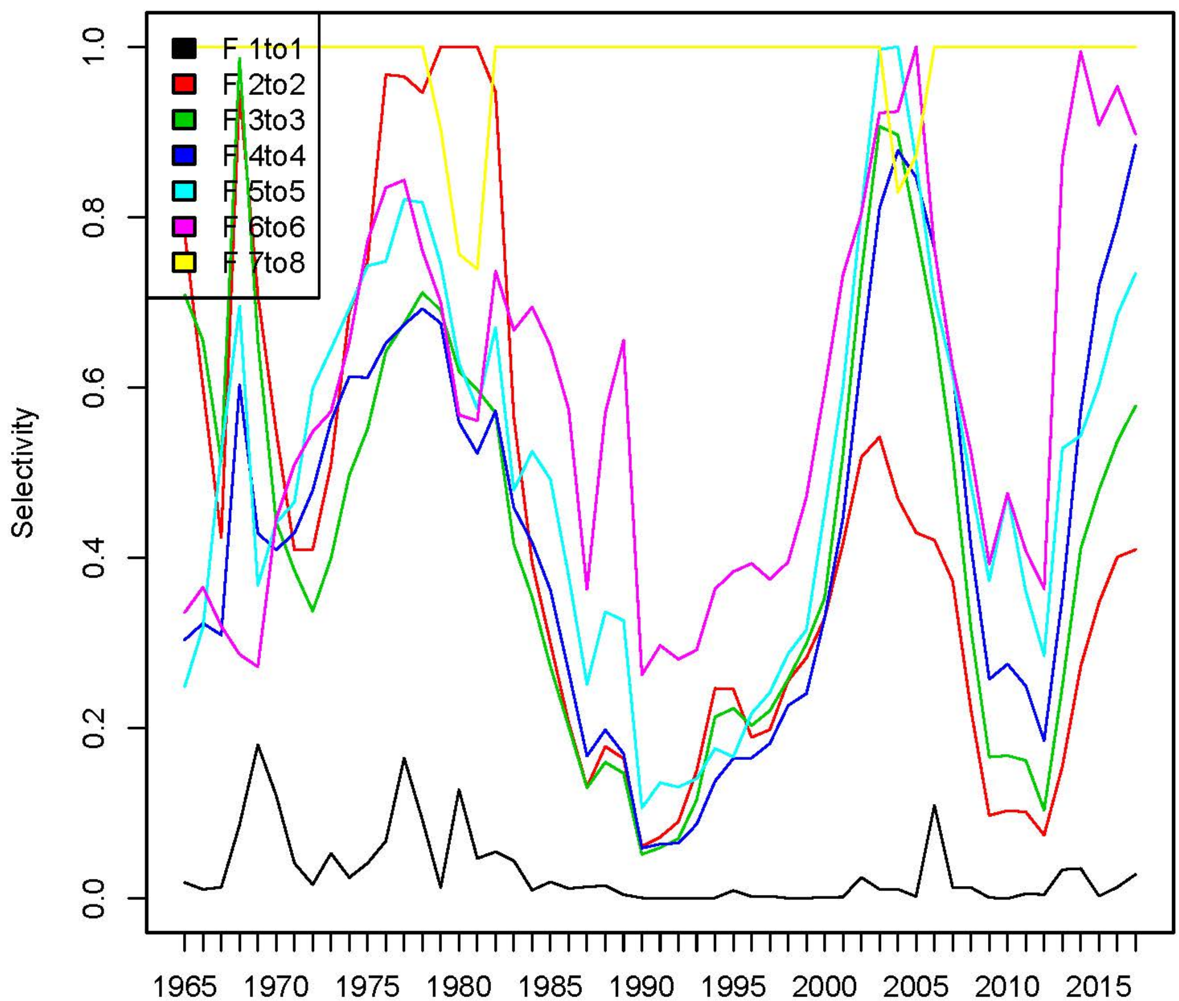


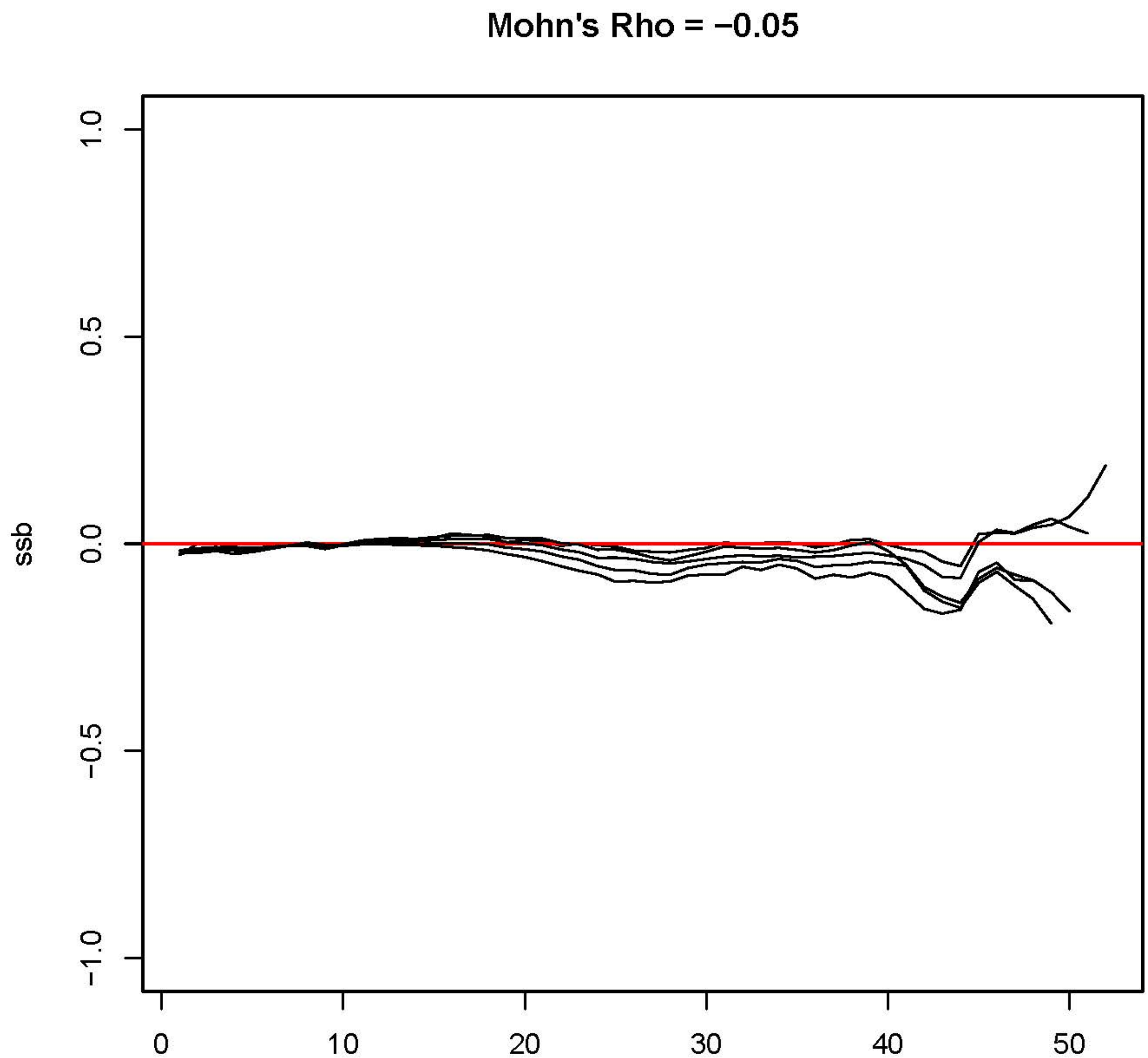
Residual catch
SummerNMFS
SpringNMFSAlbatross
SpringNMFSBigelow
FallNMFSAlbatross
FallNMFSBigelow
Age
0
2
4
6
8
1970
1980
1990
2000
2010
2020
Year
−6
−4
−2
0
2
4



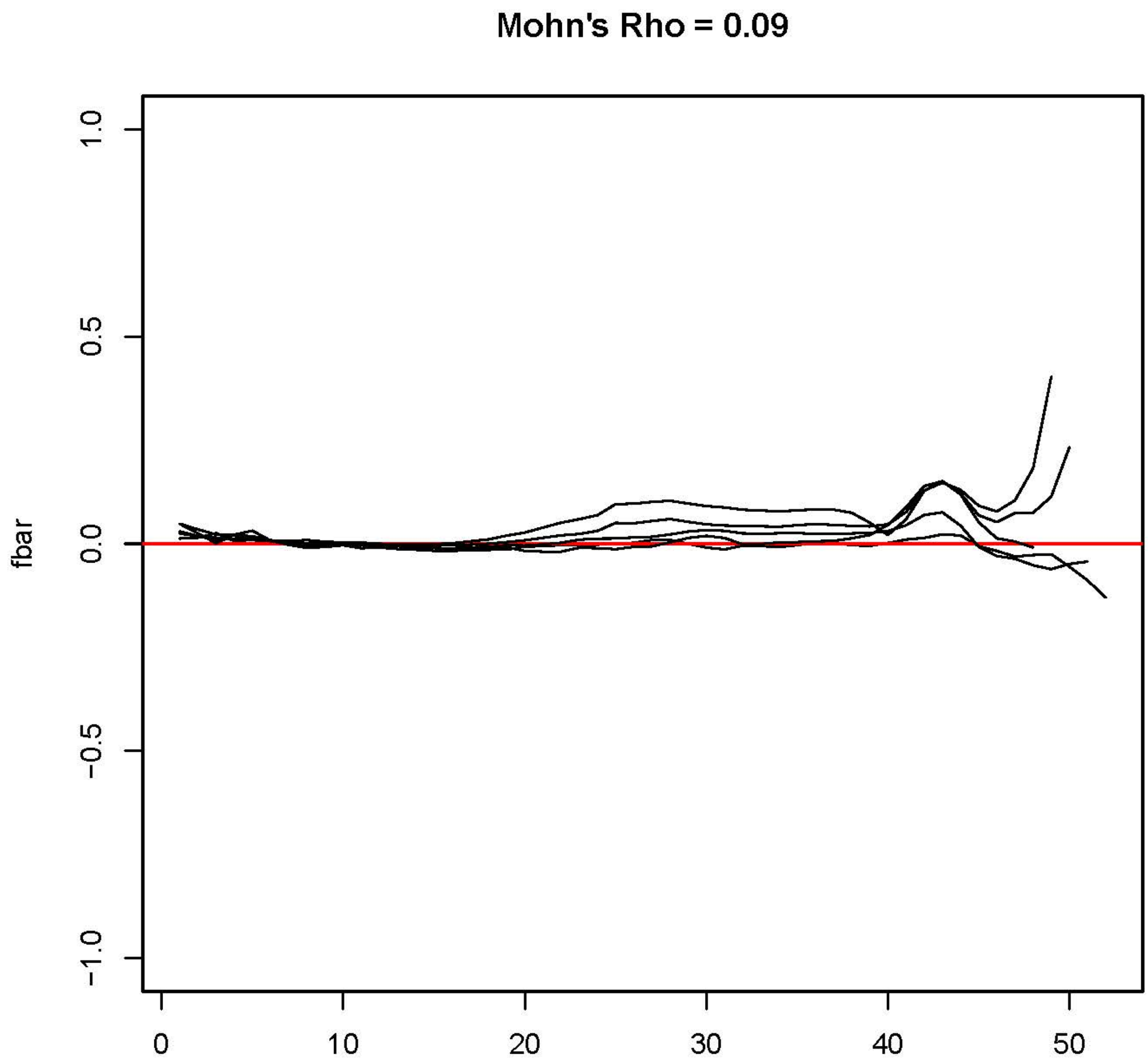
Joint sample residuals log(N)
Joint sample residuals log(F)
Age
Year
0
2
4
6
8
1970
1980
1990
2000
2010
-4
-3
-2
-1
0
1
2
3
4


SSB
Years
current
w.o. SpringNMFSAlbatross
w.o. FallNMFSAlbatross
w.o. SpringNMFSBigelow
w.o. FallNMFSBigelow
w.o. SummerNMFS
$\overline{F}_{4-6}$
Recruits (age 1)

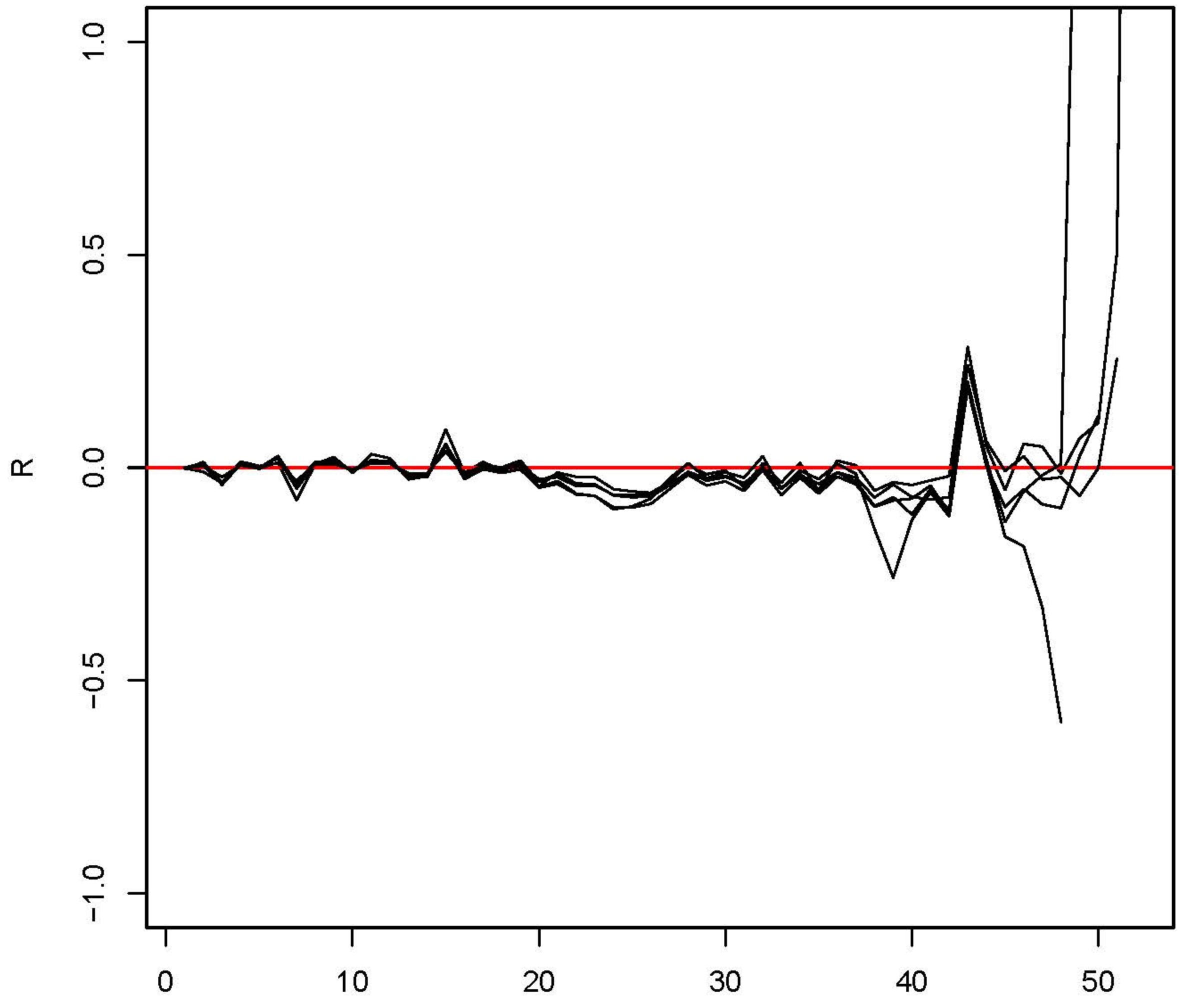

F 1to1
F 2to2
F 3to3
F 4to4
F 5to5
F 6to6
F 7to8
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015

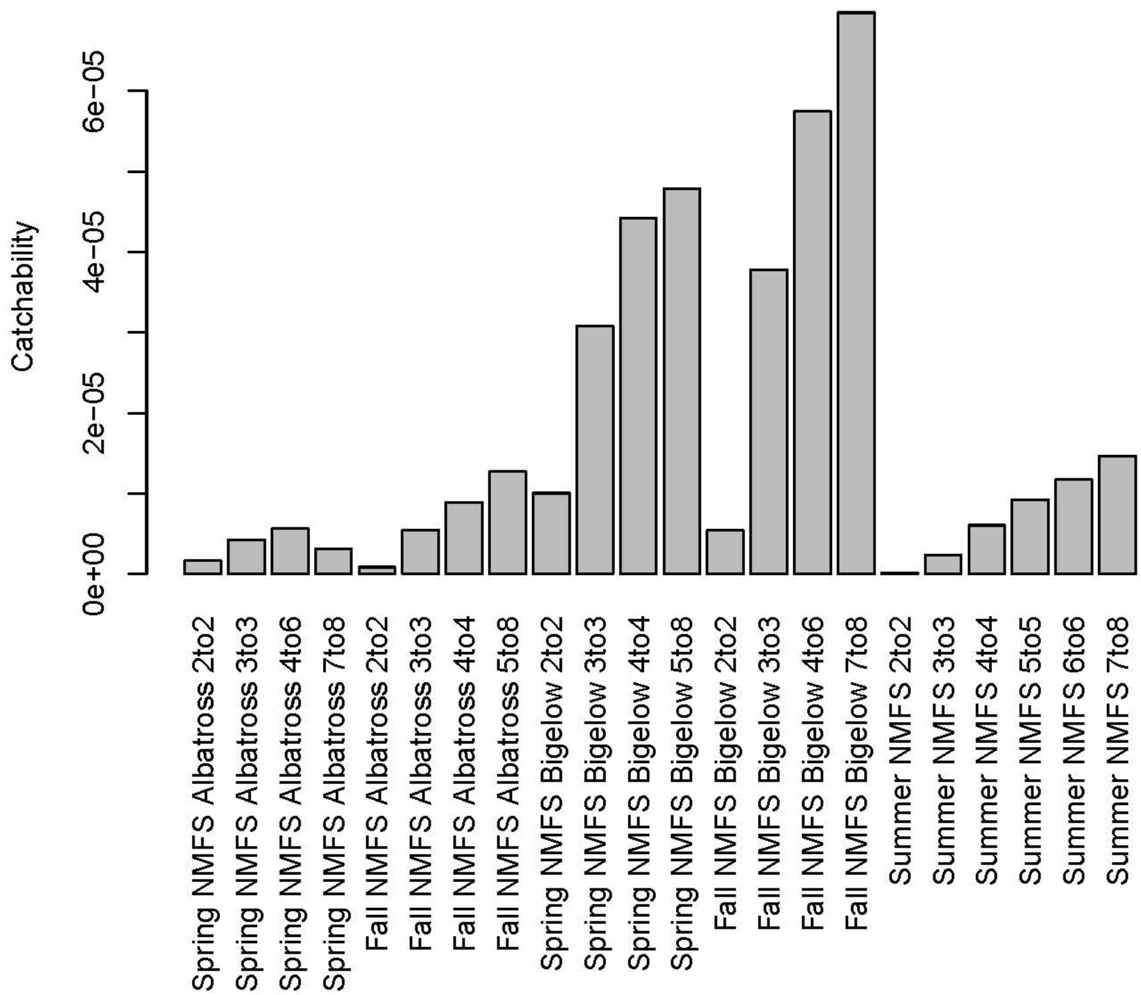


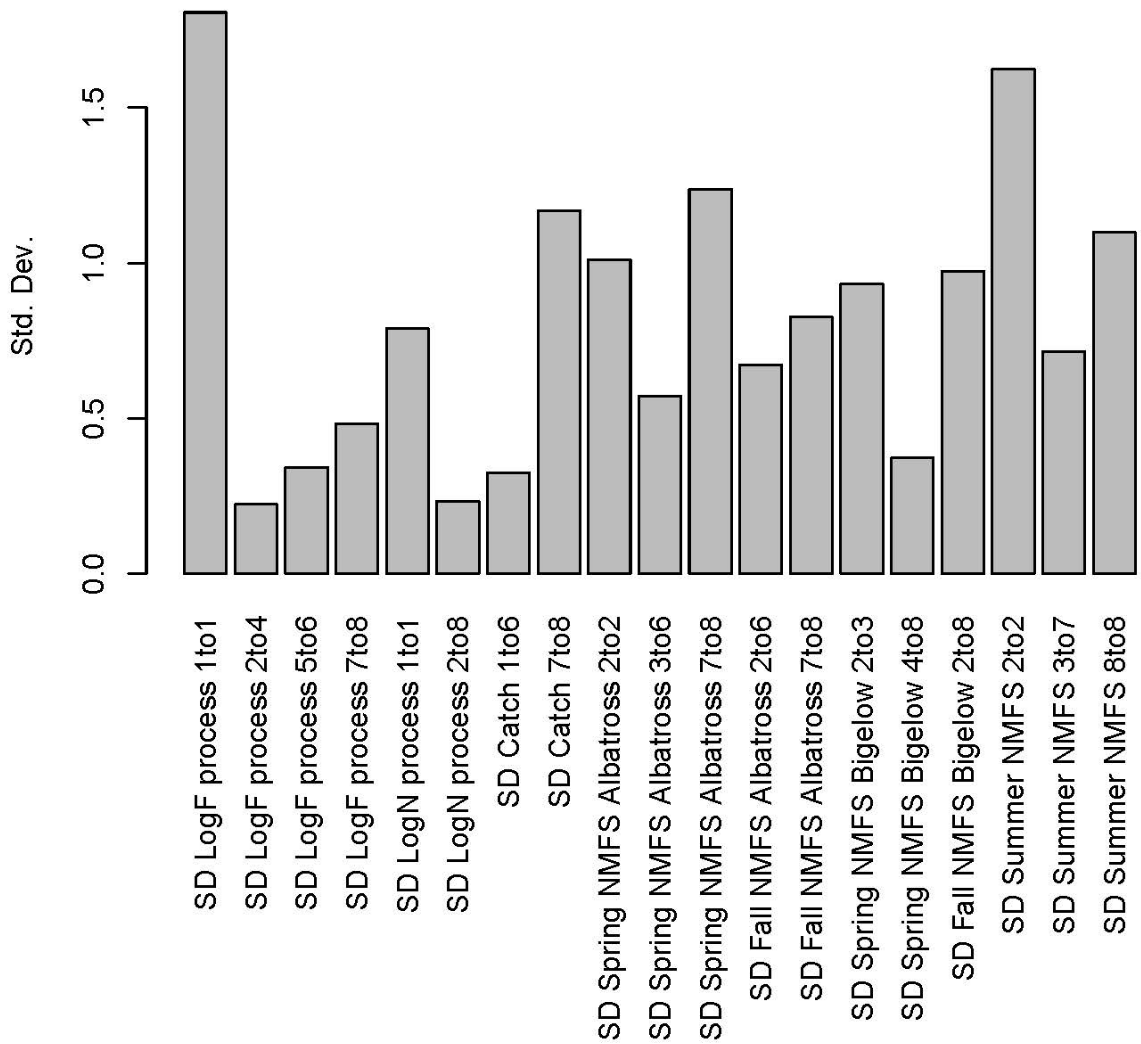
Mohn's Rho = −0.05
ssb
1.0
0.5
0.0
−0.5
−1.0
0
10
20
30
40
50

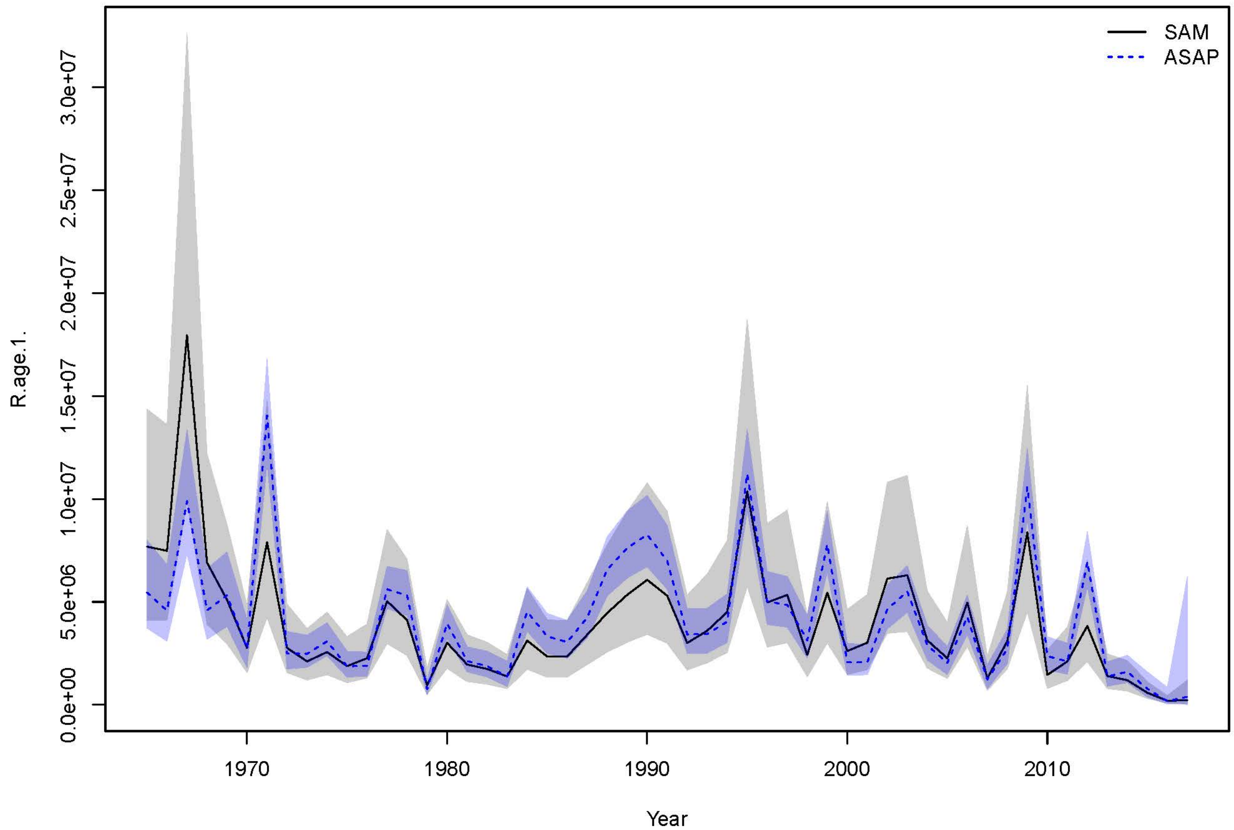

Mohn's Rho = 0.09
fbar
1.0
0.5
0.0
−0.5
−1.0
0
10
20
30
40
50

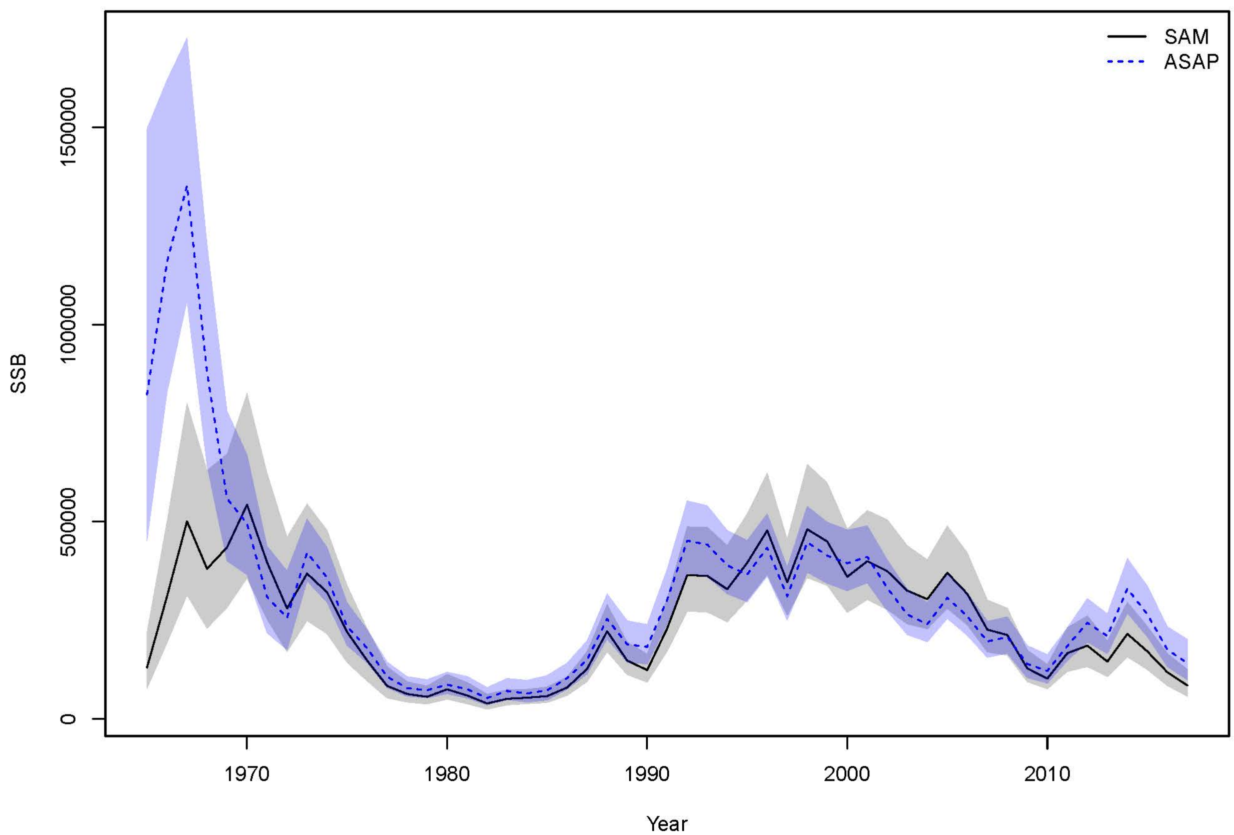
Mohn's Rho = 1.1



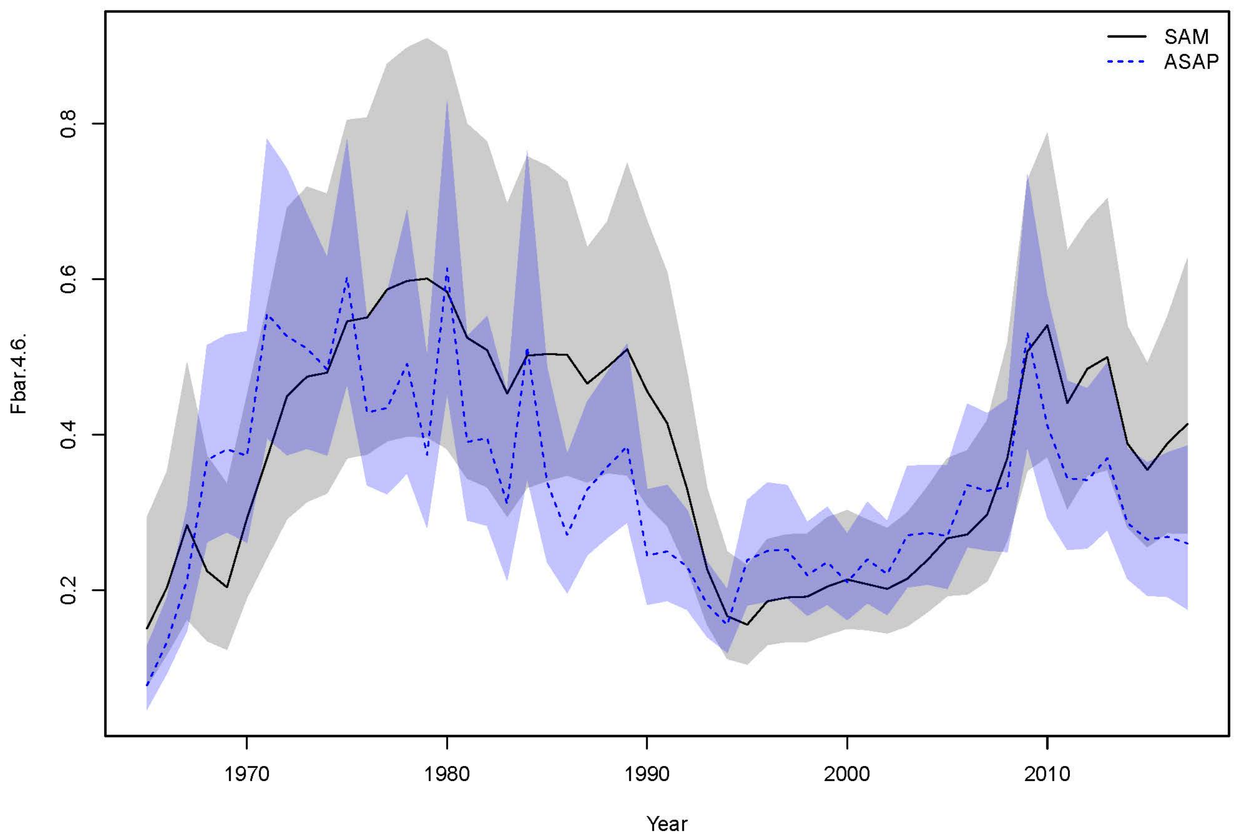

Catchability
0e+00
2e-05
4e-05
6e-05
Spring NMFS Albatross 2to2
Spring NMFS Albatross 3to3
Spring NMFS Albatross 4to6
Spring NMFS Albatross 7to8
Fall NMFS Albatross 2to2
Fall NMFS Albatross 3to3
Fall NMFS Albatross 4to4
Fall NMFS Albatross 5to8
Spring NMFS Bigelow 2to2
Spring NMFS Bigelow 3to3
Spring NMFS Bigelow 4to4
Spring NMFS Bigelow 5to8
Fall NMFS Bigelow 2to2
Fall NMFS Bigelow 3to3
Fall NMFS Bigelow 4to6
Fall NMFS Bigelow 7to8
Summer NMFS 2to2
Summer NMFS 3to3
Summer NMFS 4to4
Summer NMFS 5to5
Summer NMFS 6to6
Summer NMFS 7to8


Std. Dev.
0.0
0.5
1.0
1.5
SD LogF process 1to1
SD LogF process 2to4
SD LogF process 5to6
SD LogF process 7to8
SD LogN process 1to1
SD LogN process 2to8
SD Catch 1to6
SD Catch 7to8
SD Spring NMFS Albatross 2to2
SD Spring NMFS Albatross 3to6
SD Spring NMFS Albatross 7to8
SD Fall NMFS Albatross 2to6
SD Fall NMFS Albatross 7to8
SD Spring NMFS Bigelow 2to3
SD Spring NMFS Bigelow 4to8
SD Fall NMFS Bigelow 2to8
SD Summer NMFS 2to2
SD Summer NMFS 3to7
SD Summer NMFS 8to8

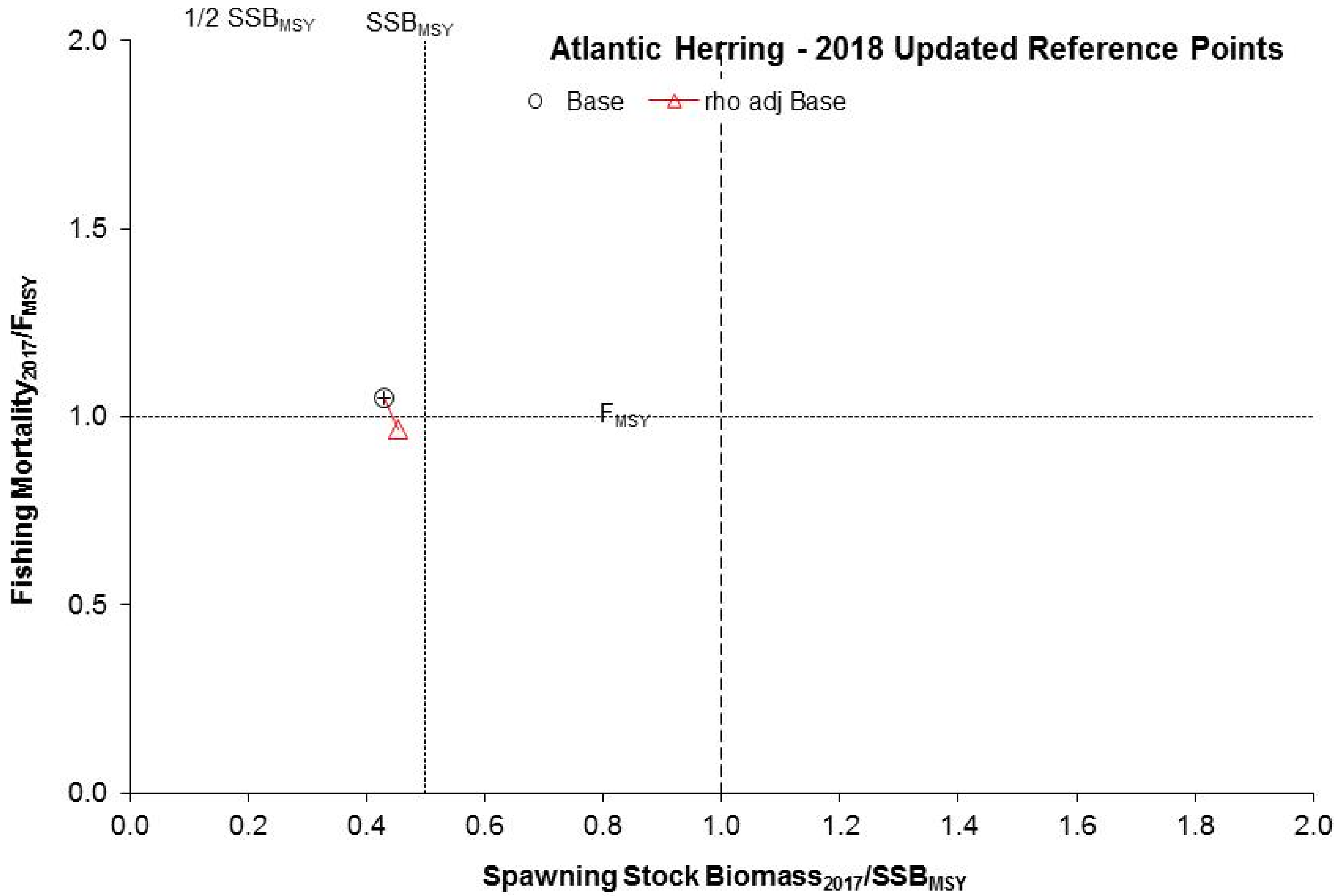

SAM
ASAP
R.age.1.
3.0e+07
2.5e+07
2.0e+07
1.5e+07
1.0e+07
5.0e+06
0.0e+00
1970
1980
1990
2000
2010
Year



SAM
ASAP
SSB
0
500000
1000000
1500000
1970
1980
1990
2000
2010
Year



SAM
ASAP
0.2
0.4
0.6
0.8
Fbar.4.6.
1970
1980
1990
2000
2010
Year


Atlantic Herring - 2018 Updated Reference Points
Base
rho adj Base
1/2 SSBMSY
SSBMSY
FMSY
Fishing Mortality2017/FMSY
Spawning Stock Biomass2017/SSBMSY
2.0
1.5
1.0
0.5
0.0
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
2.0

Appendix B3
Multi-model inference for stock assessments: options and advantages

Sarah Gaichas, Jonathan Deroba, and Kiersten Curti

Introduction

The tradition in stock assessment has been to select one preferred model to estimate quantities of interest for management. However, structural differences between models can reflect legitimate alternative hypotheses regarding uncertain dynamics, which lead to different estimates (Hill et al. 2007). Capturing model structural uncertainty is desirable, especially for management systems that use uncertainty in assessment outputs to specify total allowable catches to manage the risk of overfishing (PFMC, MAFMC).

Model ensembles and multi-model inference are used for analysis and operational forecasting in many fields (weather: Tracton and Kalnay (1993), Zhou and Du (2010), and see https://www.ncdc.noaa.gov/data-access/model-data/model-datasets/north-american-multi-model-ensemble, hurricane: Krishnamurti et al. (2016); Leonardo and Colle (2017); and see https://www.nhc.noaa.gov/modelsummary.shtml?) and also for long term climate prediction (Tebaldi and Knutti 2007, Semenov and Stratonovitch 2010). General methods range from unweighted model averaging through several types of weighted averaging through use of bayesian posterior model probabilities (Burnham and Anderson 2002, King et al. 2009). Methodolgy continues to develop in the weather forecasting and climate fields (e.g., Du et al. 2018, Narapusetty et al. 2018). Multiple models are used in these situations to address structural uncertainty, with the added benefit that ensemble forecast quantities of interest have been found to be more accurate than estimates from individual model ensembles and ensemble members (Hagedorn et al. 2005, Zhou and Du (2010)). Given appropriate resources and review frameworks, fisheries management could benefit from multiple model and ensemble methods similarly.

For multimodel inference (MMI) to be practical in a fisheries stock assessment context, it needs to be transparent for stakeholders with straightforward, standardized methods that do not add substantially to the workload of assessment scientists. Another high priority within an assessment process is to address all of the terms of reference (ToRs), so a multimodel/ensemble approach must be able to do this. Finally, demonstrating that the multimodel/ensemble produces added benefits for management over single model selection is important. This will likely require simulation analysis, as fisheries scientists and managers almost never get to see the true state of the system the way weather forecasters do.

MMI has been used for recruitment modeling in fisheries (Brodziak and Legault 2005), as well as long term forecasting (Ianelli et al. 2016). MMI can be used to include multiple plausible parameters or functional forms within a modeling framework to address multiple hypotheses (Millar et al. 2015). Models with different structures and assumptions about uncertainty can also be compared to determine which management measures may be most robust (Gårdmark et al. 2013). Finally, status and trend can be evaluated with MMI (Anderson et al. 2017).

Here, we apply a simple MMI approach to two herring stock assessment models. The objective is to "field test" a simple method during a working group meeting so that we can see what would

would be necessary to evaluate an MMI approach during a benchmark assessment process. We then make suggestions to build a framework for the use of MMI in the future for providing advice to fishery managers.

Methods

For the purposes of this example, there was no opportunity to develop a set of candidate models based on alternative structural hypotheses (Burnham and Anderson 2002), although this happens iteratively through the working group process within a single model framework as alternative model configurations and dataset combinations are discussed, tested, and accepted or rejected as improvements on a baseline model. We consider a "model" to be a structurally different population dynamics package, rather than an implementation within the same package using different datasets and or parameter values. It might be reasonable to consider multiple equally plausible implementations within the same package to be an ensemble, but we did not explore that here. Instead, we included both models that had been implemented for Atlantic herring by the lead assessment scientist: ASAP (Miller and Legault (2015)) and SAM (Nielsen and Berg (2014)); see the main body of the text and appendix B2 for descriptions. The models are similar in that they are age-structured single species population dynamics models. They differ in their approach to parameter estimation and treatment of uncertainty. Both models were developed iteratively by the lead assessment scientist to produce the desired level of fit diagnostics, although the ASAP model received further iterative development during the working group meeting while the SAM model did not.

Stock assessment models produce many estimated parameters and outputs. We determined quantities of interest for MMI using the ToRs for the assessment. Therefore, we focused on model derived quantities: spawning stock biomass (SSB), recruitment, fishing mortality (F), and reference points. Uncertainty in these quantities is of interest in some management frameworks as well. We did not attempt to apply MMI to projections and catch advice, for this example, but discuss possible approaches below.

In this example, we take a simple average of the derived estimates from the "best" model from each modeling framework. We calculate the confidence intervals using the minimum CI from the two models as the lower bound and the maximum CI as the upper bound at each point in the time series. For these examples, approximate 80% confidence bounds are shown. This approach does not consider the uncertainty and information potential in all possible models or even all tested models, but is consistent with the objective to keep the ensemble process simple and achievable within an assessment timeframe.

Results

Taking the simple average of output SSB, recruitment, and F from ASAP and SAM demonstrates where the models are similar and where they diverge (Figs 1-3). Population indices such as SSB and recruitment are fairly similar, with the model average not greatly different from individual models. Estimates of fishing mortality are shown, but we note that the selectivities estimated by the models are very different, such that comparing Fs is conceptually more difficult. Nevertheless, the two models estimate similar Fs for some portions of the time series, and divergent Fs in others.

Perhaps more important than a measure of central tendency between models is the estimate of uncertainty when considering both. The "envelope" around the model average estimate contains both models by design, but still shows where our certainty is greater or lower across the estimation period (Figs 4-6).

Estimates of stock status were derived from the model outputs and as expected, the model average estimate falls directly between them (Fig 7). In this case, each model estimated different status, and the model average is right on the borderline with wide intervals.

Discussion

The working group discussed two options for using the models presented:

1. Average the two available models, with stock status to come from average of relative reference points, rather than the average of the absolute reference points from each model (as presented in the Results section).

2. Use ASAP as the primary model with SAM as a check on basic consistency (as in the main body of the text).

The working group elected option 2 for the herring assessment, but recommended further research into developing a protocol for using multimodel inference to provide managemnet advice from stock assessments. We outline a start on this below. The main reasons for selecting one model (ASAP) were that the SAM model had received no iterative review during the process (which the ASAP model had), and that there is less experience with all aspects of the SAM model in this region, including use of projections. A secondary concern was that the same data was not included in both models.

To address these concerns in the future in order to apply MMI, we suggest the following:

1. A full review of all models at the same level of scrutiny is required. Further, while the simplest approach may be to average the results from the "best" models as we have shown here, a working group would need similar diagnostic info on all the models to determine which models are best and to determine how to weight the models if desired (see below).

2. Models should be developed to make the best use of their features and available data, rather than for consistency with an initially preferred model (e.g. the 2017 Atlantic mackerel assessment alternative models were built to approximate the primary ASAP model as a consistency check and therefore would be less useful in MMI).

3. Given a wide range of models, descisions on which models to include in the ensemble would be required, and should be based on a consistent set of pre-determined performance criteria (e.g., fit diagnostics, retrospective patterns.) These performance criteria should be provided in advance to different modeling teams as pre-requisites for introducing models to the assessment process.

4. Model weighting can be useful but complicated. A huge literature exists on this; but a smaller subset of options would be necessary for working groups to proceed with decisions. Considerations could include weighting by model skill as determined from simulation analysis, model fit as determined by AIC or BIC, etc, other model diagnostics such as retrospective pattern. Equally weighted models could be a simple option if working groups are convinced that in using the "best" version of each model structure, the assumptions are different but equally plausible. There are also algorithms for evaluation of combined model and parameter posterior probability that could be implemented (King et al. 2009).

5. A method for doing projections from multimodel ensembles must be established. Again, there are multiple options including projecting with all models and averaging projections, or projecting with a single method using averaged parameters. Working groups would need a short list or a single option for addressing projections determined by assessment ToRs.

6. Decisions regarding the operational use of MMI are also important: if one model fails after a benchmark used MMI, what is the plan? These contingencies would need to be part of transparent advance plan.

Some general guidelines for MMI could be developed through simulation analysis, although this may be a resource intensive process.

A final working group reccomendation was to fully document the SAM model to have as part of this benchmark. The previous appendix and this one may lay the groundwork for using MMI in a future Atlantic herring assessment.

References

Anderson, S.C., Cooper, A.B., Jensen, O.P., Minto, C., Thorson, J.T., Walsh, J.C., Afflerbach, J., Dickey-Collas, M., Kleisner, K.M., Longo, C., Osio, G.C., Ovando, D., Mosqueira, I., Rosenberg, A.A., and Selig, E.R. 2017. Improving estimates of population status and trend with superensemble models. Fish and Fisheries **18**(4): 732–741. doi:10.1111/faf.12200.

Brodziak, J., and Legault, C.M. 2005. Model averaging to estimate rebuilding targets for overfished stocks. Canadian Journal of Fisheries and Aquatic Sciences **62**(3): 544–562. doi:10.1139/f04-199.

Burnham, K.P., and Anderson, D.R. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. *In* 2nd editions. Springer-Verlag, New York. Available from //www.springer.com/us/book/9780387953649 [accessed 11 May 2018].

Du, J., Berner, J., Buizza, R., Charron, M., Houtekamer, P., Hou, D., Jankov, I., Mu, M., Wang, X., Wei, M., and Yuan, H. 2018. Ensemble Methods for Meteorological Predictions. *In* Handbook of Hydrometeorological Ensemble Forecasting. Springer, Berlin, Heidelberg. pp. 1–52. doi:10.1007/978-3-642-40457-3_13-1.

Gårdmark, A., Lindegren, M., Neuenfeldt, S., Blenckner, T., Heikinheimo, O., Müller-Karulis, B., Niiranen, S., Tomczak, M.T., Aro, E., Wikström, A., and Möllmann, C. 2013. Biological ensemble modeling to evaluate potential futures of living marine resources. Ecological Applications **23**(4): 742–754. doi:10.1890/12-0267.1.

Hagedorn, R., Doblas-Reyes, F.J., and Palmer, T.N. 2005. The rationale behind the success of multi-model ensembles in seasonal forecasting – I. Basic concept. Tellus A **57**(3): 219–233. doi:10.1111/j.1600-0870.2005.00103.x.

Hill, S.L., Watters, G.M., Punt, A.E., McAllister, M.K., Quéré, C.L., and Turner, J. 2007. Model uncertainty in the ecosystem approach to fisheries. Fish and Fisheries **8**(4): 315–336. doi:10.1111/j.1467-2979.2007.00257.x.

Ianelli, J., Holsman, K.K., Punt, A.E., and Aydin, K. 2016. Multi-model inference for incorporating trophic and climate uncertainty into stock assessments. Deep Sea Research Part II: Topical Studies in Oceanography **134**: 379–389. doi:10.1016/j.dsr2.2015.04.002.

King, R., Morgan, B.J., Gimenez, O., and Brooks, S.P. 2009. Bayesian Analysis for Population Ecology. Chapman & Hall/CRC. Available from https://www.crcpress.com/Bayesian-Analysis-for-Population-Ecology/King-Morgan-Gimenez-Brooks/p/book/9781439811870 [accessed 11 May 2018].

Krishnamurti, T.N., Kumar, V., Simon, A., Bhardwaj, A., Ghosh, T., and Ross, R. 2016. A review of multimodel superensemble forecasting for weather, seasonal climate, and hurricanes: MULTIMODEL SUPERENSEMBLE FORECASTS. Reviews of Geophysics **54**(2): 336–377. doi:10.1002/2015RG000513.

Leonardo, N.M., and Colle, B.A. 2017. Verification of Multimodel Ensemble Forecasts of North Atlantic Tropical Cyclones. Weather and Forecasting **32**(6): 2083–2101. doi:10.1175/WAF-D-17-0058.1.

Millar, C.P., Jardim, E., Scott, F., Osio, G.C., Mosqueira, I., and Alzorriz, N. 2015. Model averaging to streamline the stock assessment process. ICES Journal of Marine Science **72**(1): 93–98. doi:10.1093/icesjms/fsu043.

Miller, T.J., and Legault, C.M. 2015. Technical Details for ASAP version 4, US Dept Commer, Northeast Fish Sci Cent Ref Doc. 15- 17. Available from https://www.nefsc.noaa.gov/nefsc/publications/crd/crd1517/crd1517.pdf [accessed 14 June 2018].

Narapusetty, B., Collins, D.C., Murtugudde, R., Gottschalck, J., and Peters-Lidard, C. 2018. Bias correction to improve the skill of summer precipitation forecasts over the contiguous United States by the North American multi-model ensemble system. Atmospheric Science Letters **0**(0). doi:10.1002/asl.818.

Nielsen, A., and Berg, C.W. 2014. Estimation of time-varying selectivity in stock assessments using state-space models. Fisheries Research **158**: 96–101. doi:10.1016/j.fishres.2014.01.014.

Semenov, M.A., and Stratonovitch, P. 2010. Use of multi-model ensembles from global climate models for assessment of climate change impacts. Climate Research **41**(1): 1–14. Available from http://www.jstor.org/stable/24870469 [accessed 13 June 2018].

Tebaldi, C., and Knutti, R. 2007. The use of the multi-model ensemble in probabilistic climate projections. Philosophical Transactions of the Royal Society of London A: Mathematical, Physical and Engineering Sciences **365**(1857): 2053–2075. doi:10.1098/rsta.2007.2076.

Tracton, M.S., and Kalnay, E. 1993. Operational Ensemble Prediction at the National Meteorological Center: Practical Aspects. Weather and Forecasting **8**(3): 379–398. doi:10.1175/1520-0434(1993)008<0379:OEPATN>2.0.CO;2.

Zhou, B., and Du, J. 2010. Fog Prediction from a Multimodel Mesoscale Ensemble Prediction System. Weather and Forecasting **25**(1): 303–322. doi:10.1175/2009WAF2222289.1.

Figures
Herring Examples

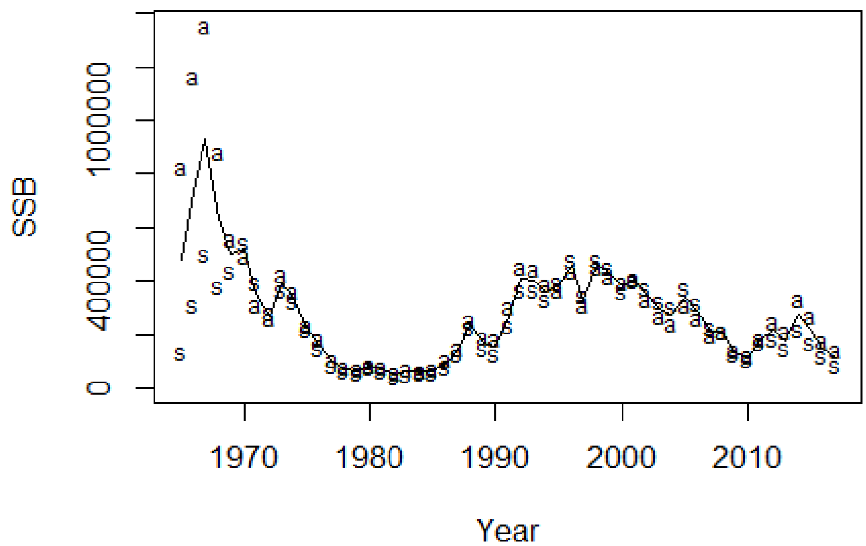


Figure 1: Herring SSB estimates from two models with average

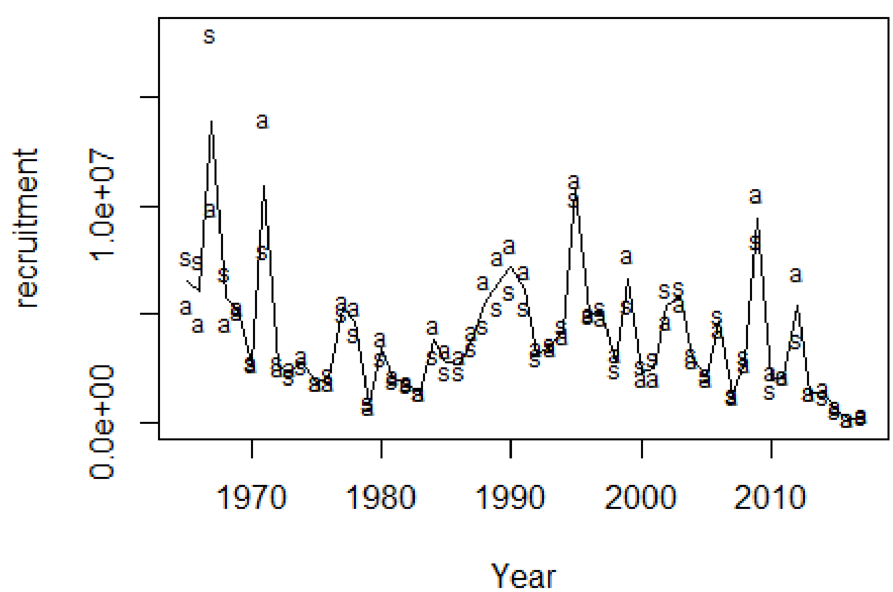


Figure 2: Herring recruitment estimates from two models with average

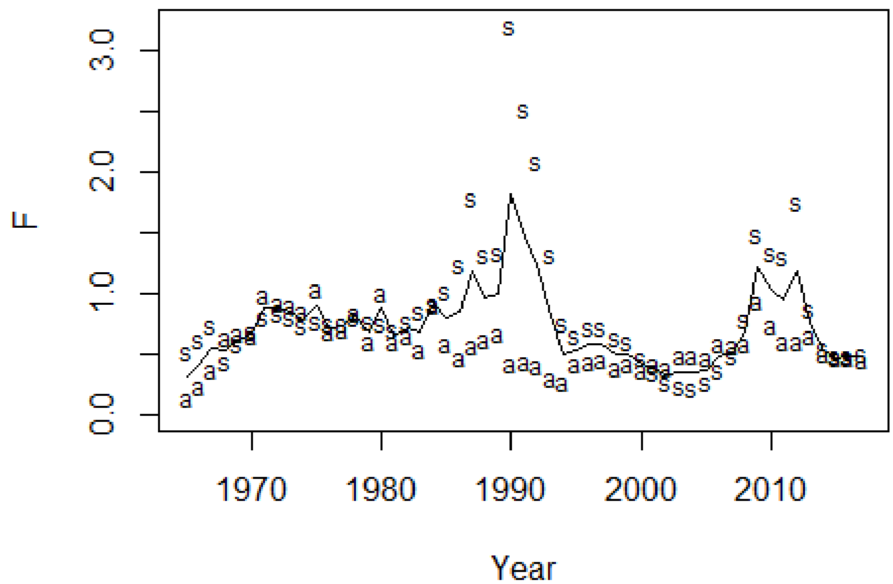


Figure 3: Herring fishing mortality estimates from two models with average

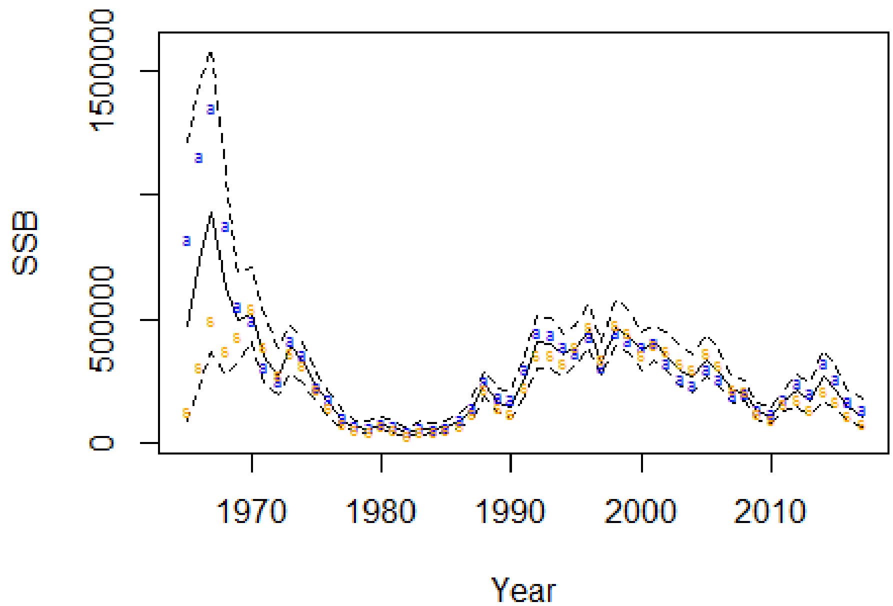


Figure 4: Average Herring SSB estimates with CIs

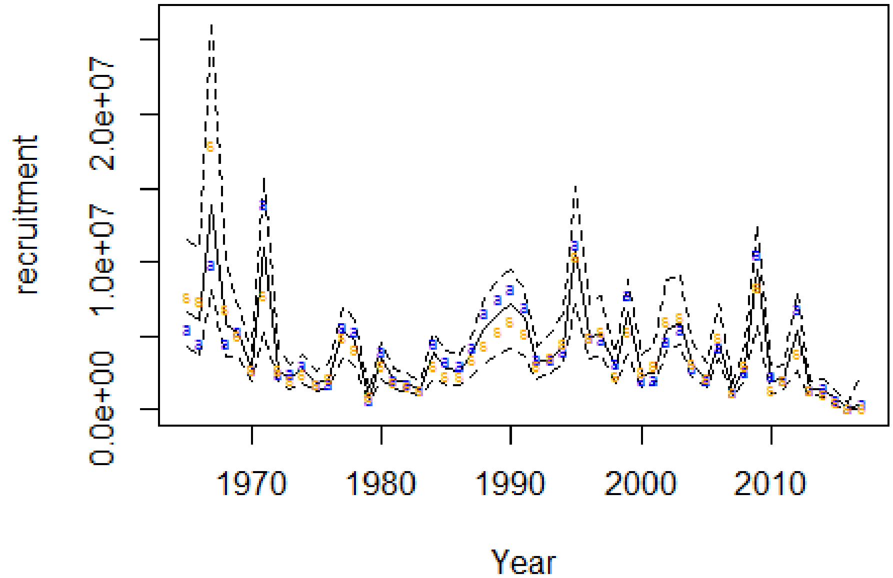


Figure 5: Average Herring recruitment estimates with CIs

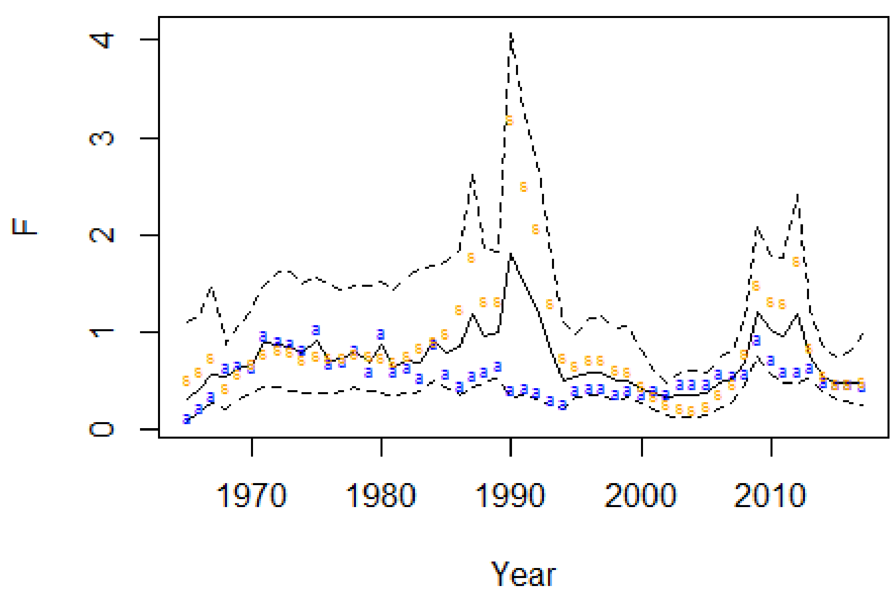


Figure 6: Average Herring F estimates with CIs

Stock status

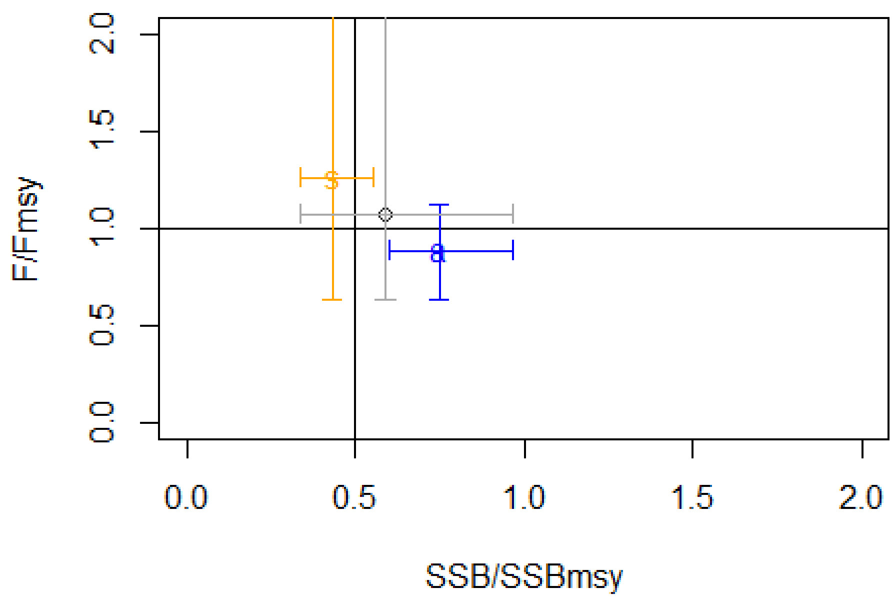


Figure 7: Herring stock status from ASAP (a), SAM (s) and average

Appendix B4

Development of an Atlantic herring Stock Synthesis model

Gavin Fay[1], Amanda Hart[1], Ashley Weston[1], Jonathan Deroba[2]
[1]Department of Fisheries Oceanography, School for Marine Science and Technology, University of Massachusetts Dartmouth, 836 South Rodney French Boulevard, New Bedford, MA 02744
[2]NOAA Fisheries Northeast Fisheries Science Center, 166 Water Street, Woods Hole MA 02543
June 2018

---

1. Background

Largely in response to research recommendations from previous assessments, a Stock Synthesis (SS) model was developed (Methot and Wetzel 2013). The research recommendations that the SS model was primarily intended to address were the ability to fit to length composition data and consideration of stock structure. So, a two area SS model was developed that fit to a broad range of data types, including length and conditional age-at-length composition data.

2. Data

2.1. Fishery landings

- Eight fishing fleets were modeled. Landings were assumed known, and are not fit to within the model. Landings were available for all fleets:
- Gulf of Maine fixed gear season 1
- Gulf of Maine fixed gear season 2
- Gulf of Maine mobile gear season 1
- Gulf of Maine mobile gear season 2
- Georges Bank (“other”) fixed gear season 1
- Georges Bank (“other”) fixed gear season 2
- Georges Bank (“other”) mobile gear season 1
- Georges Bank (“other”) mobile gear season 2

2.2. Fishery length compositions

- Length data were modeled in 1cm length bins, from 3cm - 39cm. The number of tows by fleet/season/year were used as the initial input sample size for length compositions.
- Length composition data were available for all fleets except for Georges Bank fixed gear in the fall (and very few observations for Georges Bank fixed gear in spring). In recent years (>1995) most fishery composition data is from the mobile gear fleets.

2.3. Fishery age-at-length compositions

- The distribution of ages at length were used rather than age composition information.
- Raw numbers of aged fish sampled per length bin were used as the initial sample size for age-at-length compositions for each fleet/season/year combination.

- Age-at-length composition data were available for all fleets except for Georges Bank fixed gear in the fall (and very few observations for Georges Bank fixed gear in spring). In recent years (>1995) most fishery composition data is from the mobile gear fleets.

2.4. Survey abundance indices

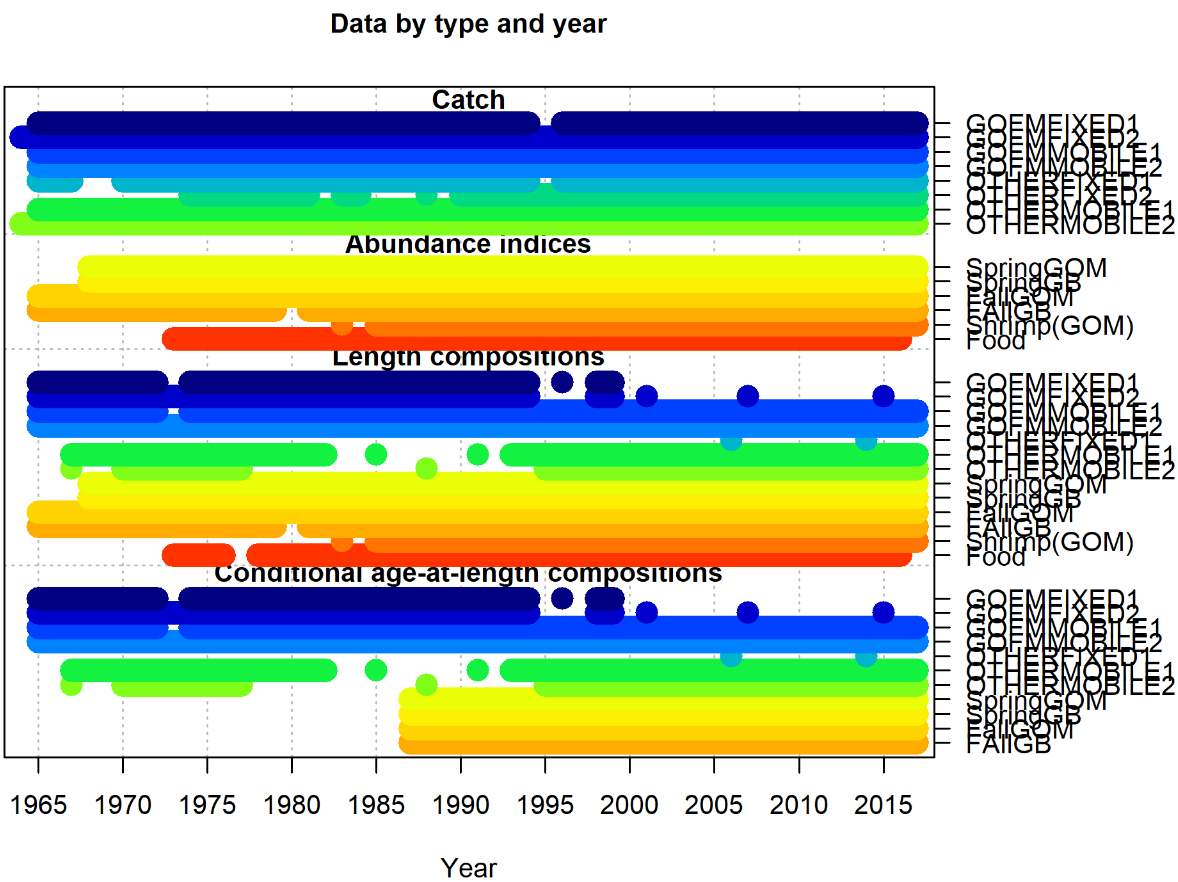
- Time series of survey indices (numbers per tow) were split to the 2 areas (NMFS bottom trawl surveys) and input as separate surveys. Raw CVs from stratified abundance estimates were used as initial inputs for the variance of survey observations.
- Spring Gulf of Maine survey
- Spring Georges Bank survey
- Fall Gulf of Maine survey
- Fall Georges Bank survey
- Gulf of Maine shrimp survey
- Food habits index - an index of predator consumption was used as an abundance index. This was assigned to the Gulf of Maine.

2.5. Survey length compositions

- Length composition data were available for all surveys.
- The number of tows by fleet/season/year were used as the initial input sample size for length compositions.

2.6. Survey age-at-length compositions

- Age-at-length composition data were available for the four NMFS bottom trawl surveys (season/area)
- Raw numbers of aged fish sampled per length bin were used as the initial sample size for age-at-length compositions for each fleet/season/year combination



Time series of availability of data by fishing fleet and survey.

3. Stock assessment model

3.1. Population dynamics model structure

A single-stock population was modeled, with spatial and seasonal structure, and seasonal movement between areas. The model represented two areas (Georges Bank and Gulf of Maine) and two seasons (Spring/Fall) for the years 1965-2017. It represented a single sex and ages 1-17, with age 17 modeled as a plus group.

3.2. Biology

3.2.1. Growth

- Von Bertalanffy growth model
- Two growth patterns were modeled; one for each area, allowing for area differences in growth parameters (fish are assigned to growth patterns at recruitment and therefore do not inherit growth curve parameters from the area in which they find themselves over lifespan).
- Time-varying growth by area where L_∞ was allowed to vary through time with blocks before and after 1993.

3.2.2. Mortality

- Constant at 0.3

3.2.3. Maturity

- Time invariant maturity-at-age; same for both growth patterns (insert picture)

3.2.4. Movement

- Estimated between areas after season 2, but assumed fish returned to their area of origin at the end of the first season.

3.3. Recruitment

3.3.1. Stock-recruit relationship

- Beverton-Holt stock-recruitment relationship
- Steepness fixed at 0.57
- Unfished recruitment estimated
- σ_R fixed at 0.8
- Spawning biomass determining recruitment modeled as that present halfway through season 2 (start of month 10)

3.3.2. Recruitment deviations

- Estimated for years 1965-2015

3.3.3. Recruitment allocation

- Time-varying allocation of the proportions of the total annual recruitment by area estimated.
- Recruits enter population at the start of season 1 as age 1s.

3.4. Fleets

- 8 fishery fleets (see above)

3.5. Surveys

- 6 survey fleets (see above)

3.6. Catchability

3.6.1. Surveys

- Time-varying catchability estimated in three blocks; before 1985, 1985-2009, post 2009 for the Spring/Fall Georges Bank and Gulf of Maine surveys
- Estimated for shrimp and food surveys; constant through time

3.7. Selectivity

Selectivity modeled and estimated as a function of length for all fleets and surveys

3.7.1. Fishing fleets

- Dome shaped; Gulf of Maine/Georges Bank fixed gear fall and spring. The same selectivity pattern was used for fleets in both areas (but varied by season), due to lack of composition data for Georges Bank fixed gear.
- Logistic shaped; Gulf of Maine/Georges Bank mobile gear fall and spring. In each area, the same selectivity pattern was used in both seasons.

3.7.2. Surveys

- Time-varying selectivity in three blocks for NMFS bottom trawl surveys; before 1985, 1985-2009, post 2009.
- Dome-shaped selectivity estimated for spring NMFS bottom trawl surveys, logistic for fall and for the Gulf of Maine shrimp and food habits data surveys

3.8. Initial conditions

3.8.1. Equilibrium catches

- To adjust the population from unfished in first modeled year, initial equilibrium levels of fishing mortality were estimated. Data on the equilibrium catch for each season and area were input and fit to - these were assigned to the fleets taking the majority of the catch in the early years of the time series (Gulf of Maine fixed gear and Georges Bank mobile gear). Values used for the equilibrium catches were the average total landings from 1964-1969 by season and area. These data were given a CV of 10%.

3.8.2. Initial fishing mortality rates

- Estimated for Gulf of Maine fixed gear and Georges Bank mobile gear in both seasons 1 and 2.

4. Parameter estimation

Estimation of model parameters was performed using AD Model Builder in Stock Synthesis to obtain maximum posterior density estimates.

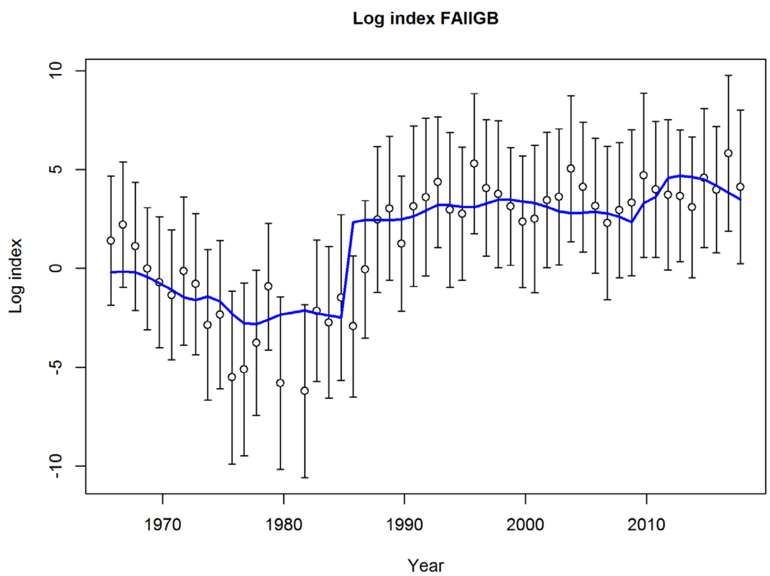
Parm_Name	Value	Phase	Min	Max	Init	Status	Parm_StDev	Gradient
L_at_Amin_Fem_GP_1	6.45846	3	1	25	9.55062	OK	0.5083	-0.00112
L_at_Amax_Fem_GP_1	30.0398	3	18	47	31.1163	OK	0.1621	-0.00484
VonBert_K_Fem_GP_1	0.665104	4	0.01	2.3	0.53984	OK	0.0258	-0.00705
CV_young_Fem_GP_1	3.62722	5	0	5	3.16291	OK	0.2072	0.00006
CV_old_Fem_GP_1	1.42012	5	0	4	0.910216	OK	0.0622	-0.00005
L_at_Amin_Fem_GP_2	11.491	3	1	25	13.6344	OK	0.2927	-0.00094
L_at_Amax_Fem_GP_2	32.4246	3	18	47	32.7838	OK	0.2681	-0.00547
VonBert_K_Fem_GP_2	0.443269	4	0.01	2.3	0.411623	OK	0.0157	-0.00574
CV_young_Fem_GP_2	1.86485	5	0	5	1.97692	OK	0.1045	-0.00012
CV_old_Fem_GP_2	1.12463	5	0	4	1.20628	OK	0.0774	-0.00025
RecrDist_GP_2	-1.48545	6	-4	4	0	OK	0.1166	-0.00433
MoveParm_A_seas_2_GP_1from_1to_2	-14.9988	3	-15	15	-0.19266	LO	6.5470	0.00000
MoveParm_B_seas_2_GP_1from_1to_2	-14.9852	4	-15	15	-0.56806	LO	51.9196	-0.00001
MoveParm_A_seas_2_GP_2from_2to_1	-4.16614	3	-15	15	0	OK	1.9396	-0.00008
MoveParm_B_seas_2_GP_2from_2to_1	2.25818	4	-15	15	0	OK	1.4026	-0.00004
L_at_Amax_Fem_GP_1_BLK2repl_1993	26.5088	3	18	47	28.1651	OK	0.1179	-0.00059
L_at_Amax_Fem_GP_2_BLK2repl_1993	27.3685	3	18	47	29.248	OK	0.1833	-0.00050
SR_LN(R0)	15.8399	1	10	25	18	OK	0.0409	-0.07635
InitF_seas_1_flt_2GOFMFIXED2	0.352132	1	0.00001	3	0.05	OK	0.0559	0.00061
InitF_seas_1_flt_8OTHERMOBILE2	0.088196	1	0.00001	3	0.05	OK	0.0138	0.00341
InitF_seas_2_flt_2GOFMFIXED2	1.50699	1	0.00001	3	0.05	OK	0.2307	0.00098
InitF_seas_2_flt_8OTHERMOBILE2	0.183986	1	0.00001	3	0.05	OK	0.0304	0.00577
LnQ_base_SpringGOM(9)	-14.7658	1	-17	5	0	OK	0.1230	-0.00003
LnQ_base_SpringGB(10)	-12.2169	1	-17	5	0	OK	0.1366	-0.00013
LnQ_base_FallGOM(11)	-13.1754	1	-17	5	0	OK	0.1451	0.00168
LnQ_base_FAllGB(12)	-15.3249	1	-17	5	0	OK	0.4536	-0.00003
LnQ_base_Shrimp(GOM)(13)	-11.8804	1	-17	5	0	OK	0.1717	-0.00095
LnQ_base_Food(14)	-16.6949	1	-17	5	0	OK	0.2092	-0.00012
LnQ_base_SpringGOM(9)_BLK1repl_1985	-12.4206	1	-17	5	0	OK	0.1538	-0.00021
LnQ_base_SpringGOM(9)_BLK1repl_2009	-11.2772	1	-17	5	0	OK	0.2503	-0.00032
LnQ_base_SpringGB(10)_BLK1repl_1985	-10.5043	1	-17	5	0	OK	0.1398	-0.00033
LnQ_base_SpringGB(10)_BLK1repl_2009	-9.35495	1	-17	5	0	OK	0.2094	-0.00011
LnQ_base_FallGOM(11)_BLK1repl_1985	-11.7518	1	-17	5	0	OK	0.2120	-0.00057
LnQ_base_FallGOM(11)_BLK1repl_2009	-11.0027	1	-17	5	0	OK	0.1844	-0.00010
LnQ_base_FAllGB(12)_BLK1repl_1985	-10.6149	1	-17	5	0	OK	0.4074	0.00002
LnQ_base_FAllGB(12)_BLK1repl_2009	-9.4842	1	-17	5	0	OK	0.6689	-0.00004
SizeSel_P1_GOFMFIXED1(1)	14.6022	2	5	35	15	OK	0.4599	-0.00123
SizeSel_P2_GOFMFIXED1(1)	-9.79968	2	-10	4	-2	OK	5.7922	-0.00001
SizeSel_P3_GOFMFIXED1(1)	3.23061	2	0.01	9	5.1	OK	0.1435	0.00030
SizeSel_P4_GOFMFIXED1(1)	4.05786	2	0.01	9	5.9	OK	0.1331	-0.00109
SizeSel_P1_GOFMFIXED2(2)	15.2439	2	5	35	15	OK	0.3307	-0.00288
SizeSel_P2_GOFMFIXED2(2)	-9.83771	2	-10	4	-2	OK	4.7665	0.00000
SizeSel_P3_GOFMFIXED2(2)	2.64609	2	0.01	9	5.1	OK	0.1305	0.00161
SizeSel_P4_GOFMFIXED2(2)	3.76354	2	0.01	9	5.9	OK	0.0900	-0.00081
SizeSel_P1_GOFMMOBILE1(3)	18.1876	2	-10	35	19.8575	OK	0.3307	-0.00103
SizeSel_P2_GOFMMOBILE1(3)	5.38491	2	0	15	5.49468	OK	0.3552	0.00006
SizeSel_P1_OTHERMOBILE1(7)	21.232	2	-10	35	19.9999	OK	0.3134	-0.00498
SizeSel_P2_OTHERMOBILE1(7)	4.23062	2	0	15	3.16861	OK	0.2850	0.00005
SizeSel_P1_SpringGOM(9)	11.963	2	5	35	15	OK	0.7664	-0.00001

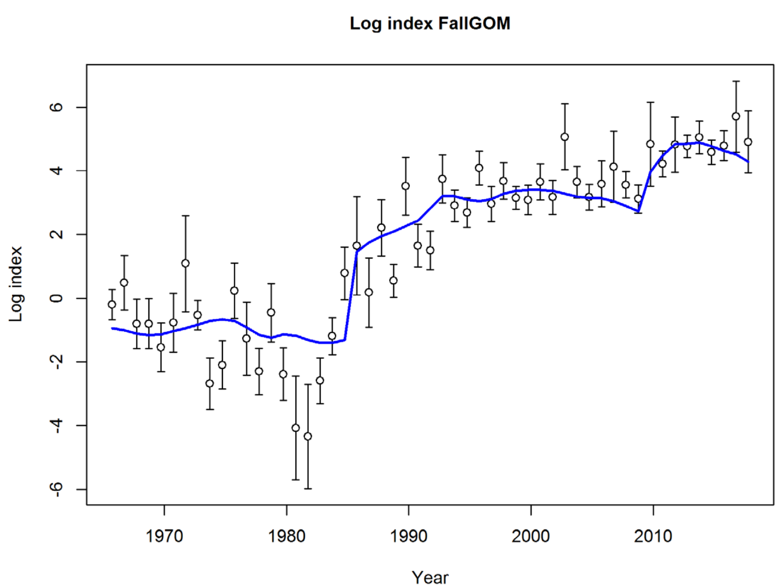
Parm_Name	Value	Phase	Min	Max	Init	Status	Parm_StDev	Gradient
SizeSel_P2_SpringGOM(9)	0.9542	2	-10	4	-2	OK	0.2525	-0.00033
SizeSel_P3_SpringGOM(9)	1.89469	2	0.01	9	5.1	OK	0.5105	0.00003
SizeSel_P4_SpringGOM(9)	0.051086	2	0.01	9	5.9	LO	1.2795	-0.00001
SizeSel_P5_SpringGOM(9)	-6.7098	3	-10	10	-9	OK	3.7902	0.00000
SizeSel_P6_SpringGOM(9)	-0.64426	3	-10	10	-9	OK	3.8854	0.00001
SizeSel_P1_SpringGB(10)	26.5528	2	5	35	15	OK	0.5621	-0.00004
SizeSel_P2_SpringGB(10)	-8.89381	2	-10	4	-2	OK	24.6235	0.00000
SizeSel_P3_SpringGB(10)	4.18975	2	0.01	9	5.1	OK	0.1743	-0.00002
SizeSel_P4_SpringGB(10)	1.75725	2	0.01	9	5.9	OK	0.4736	0.00001
SizeSel_P5_SpringGB(10)	-4.10322	3	-10	10	-9	OK	0.5287	0.00002
SizeSel_P6_SpringGB(10)	-3.0465	3	-10	10	-9	OK	1.0921	-0.00004
SizeSel_P1_FallGOM(11)	34.9955	2	-10	35	19.5	HI	0.1466	0.00003
SizeSel_P2_FallGOM(11)	14.9889	2	0	15	2.39E-06	HI	0.3374	0.00000
SizeSel_P1_FAllGB(12)	24.4204	2	-10	35	17.9357	OK	1.1756	0.00003
SizeSel_P2_FAllGB(12)	6.72547	2	0	15	0.002769	OK	1.2848	0.00002
SizeSel_P1_Shrimp(GOM)(13)	24.8019	2	-10	35	19.9996	OK	0.6153	0.00026
SizeSel_P2_Shrimp(GOM)(13)	6.6486	2	0	15	0.008	OK	0.5349	0.00000
SizeSel_P1_Food(14)	15.4069	4	-10	35	19.9996	OK	2.0724	-0.00002
SizeSel_P2_Food(14)	14.8336	4	0	15	0.008	OK	2.3886	0.00000
SizeSel_P1_SpringGOM(9)_BLK1repl_1985	13.8729	2	5	35	15	OK	0.4895	0.00000
SizeSel_P1_SpringGOM(9)_BLK1repl_2009	12.0426	2	5	35	15	OK	0.2837	0.00065
SizeSel_P2_SpringGOM(9)_BLK1repl_1985	-9.21916	2	-10	4	-2	OK	18.7243	0.00000
SizeSel_P2_SpringGOM(9)_BLK1repl_2009	-3.65537	2	-10	4	-2	OK	2.3139	0.00001
SizeSel_P3_SpringGOM(9)_BLK1repl_1985	1.99621	2	0.01	9	5.1	OK	0.3222	-0.00005
SizeSel_P3_SpringGOM(9)_BLK1repl_2009	0.015498	2	0.01	9	5.1	LO	0.1759	0.00004
SizeSel_P4_SpringGOM(9)_BLK1repl_1985	1.37981	2	0.01	9	5.9	OK	0.5881	-0.00002
SizeSel_P4_SpringGOM(9)_BLK1repl_2009	1.23774	2	0.01	9	5.9	OK	1.2923	0.00001
SizeSel_P5_SpringGOM(9)_BLK1repl_1985	-6.07275	3	-10	10	-9	OK	1.5021	0.00000
SizeSel_P5_SpringGOM(9)_BLK1repl_2009	-5.74968	3	-10	10	-9	OK	1.8121	-0.00001
SizeSel_P6_SpringGOM(9)_BLK1repl_1985	-1.85251	3	-10	10	-9	OK	0.1807	-0.00008
SizeSel_P6_SpringGOM(9)_BLK1repl_2009	-1.11193	3	-10	10	-9	OK	0.3300	-0.00008
SizeSel_P1_SpringGB(10)_BLK1repl_1985	26.6063	2	5	35	15	OK	0.7005	0.00005
SizeSel_P1_SpringGB(10)_BLK1repl_2009	17.6221	2	5	35	15	OK	1.5252	0.00002
SizeSel_P2_SpringGB(10)_BLK1repl_1985	-8.80609	2	-10	4	-2	OK	26.0950	0.00000
SizeSel_P2_SpringGB(10)_BLK1repl_2009	0.314444	2	-10	4	-2	OK	1.4966	-0.00001
SizeSel_P3_SpringGB(10)_BLK1repl_1985	4.81908	2	0.01	9	5.1	OK	0.2716	-0.00002
SizeSel_P3_SpringGB(10)_BLK1repl_2009	3.56531	2	0.01	9	5.1	OK	0.3972	-0.00001
SizeSel_P4_SpringGB(10)_BLK1repl_1985	1.37361	2	0.01	9	5.9	OK	0.6565	0.00000
SizeSel_P4_SpringGB(10)_BLK1repl_2009	0.601037	2	0.01	9	5.9	OK	12.4371	0.00000
SizeSel_P5_SpringGB(10)_BLK1repl_1985	-4.12978	3	-10	10	-9	OK	0.9804	-0.00005
SizeSel_P5_SpringGB(10)_BLK1repl_2009	-9.69039	3	-10	10	-9	OK	7.8749	0.00000
SizeSel_P6_SpringGB(10)_BLK1repl_1985	-4.77976	3	-10	10	-9	OK	3.1894	-0.00001
SizeSel_P6_SpringGB(10)_BLK1repl_2009	-4.7997	3	-10	10	-9	OK	76.0245	0.00000
SizeSel_P1_FallGOM(11)_BLK1repl_1985	24.4991	2	-10	35	19.5	OK	0.8911	0.00017
SizeSel_P1_FallGOM(11)_BLK1repl_2009	18.1251	2	-10	35	19.5	OK	1.0240	0.00008
SizeSel_P2_FallGOM(11)_BLK1repl_1985	6.29023	2	0	15	2.39E-06	OK	0.8771	-0.00002
SizeSel_P2_FallGOM(11)_BLK1repl_2009	3.99482	2	0	15	2.39E-06	OK	1.6323	0.00000
SizeSel_P1_FAllGB(12)_BLK1repl_1985	23.0246	2	-10	35	17.9357	OK	1.0826	-0.00006
SizeSel_P1_FAllGB(12)_BLK1repl_2009	20.9684	2	-10	35	17.9357	OK	0.8627	-0.00002
SizeSel_P2_FAllGB(12)_BLK1repl_1985	8.47617	2	0	15	0.002769	OK	1.2487	0.00000
SizeSel_P2_FAllGB(12)_BLK1repl_2009	3.43223	2	0	15	0.002769	OK	0.9394	0.00000

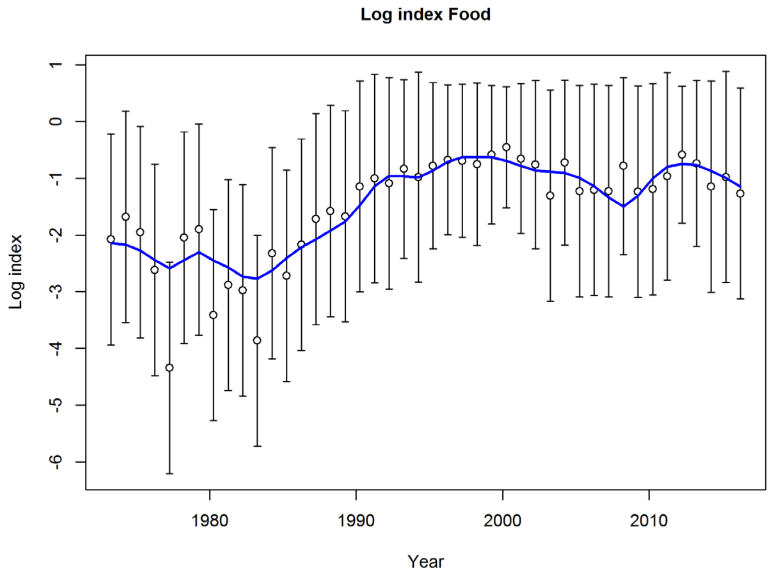
5. Results

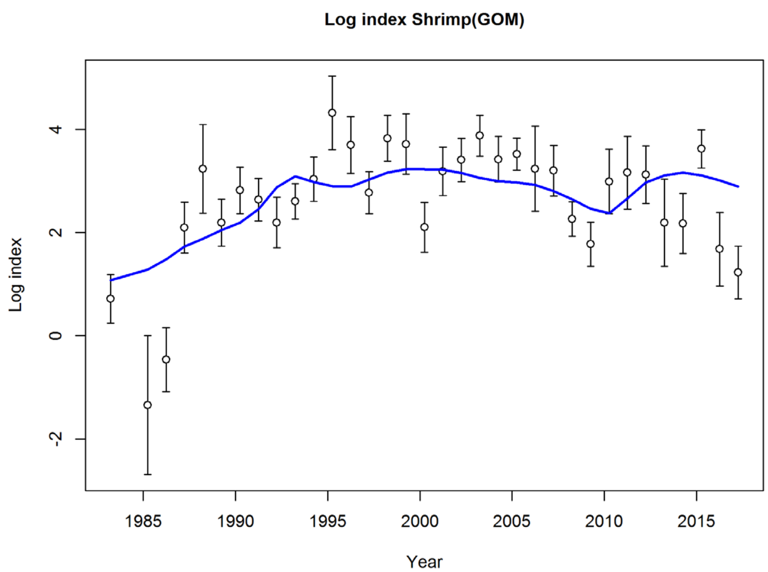
Given that the Working Group did not consider this SS model viable for consideration as the base assessment, only some example results were provided. Furthermore, providing fits for all the data sources would be voluminous and likely unnecessary to demonstrate the main outcomes of this application of SS.

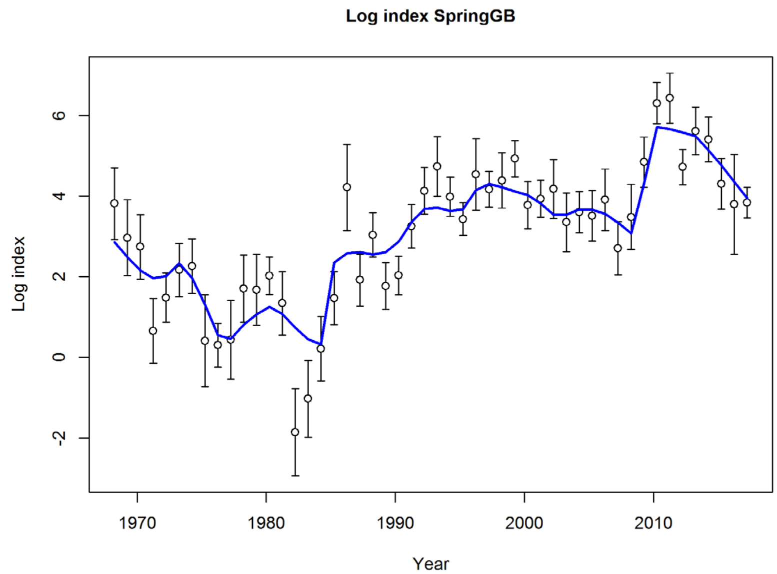
The SS model fit the survey trends relatively well. Estimates of catchability in the spring and fall NMFS bottom trawl surveys increased through time as a result of door (1984/1985) and vessel changes (2008/2009).

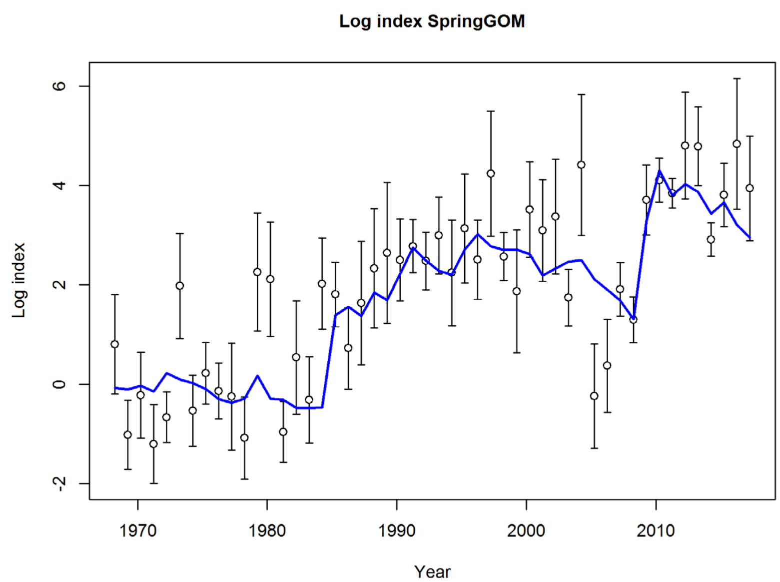


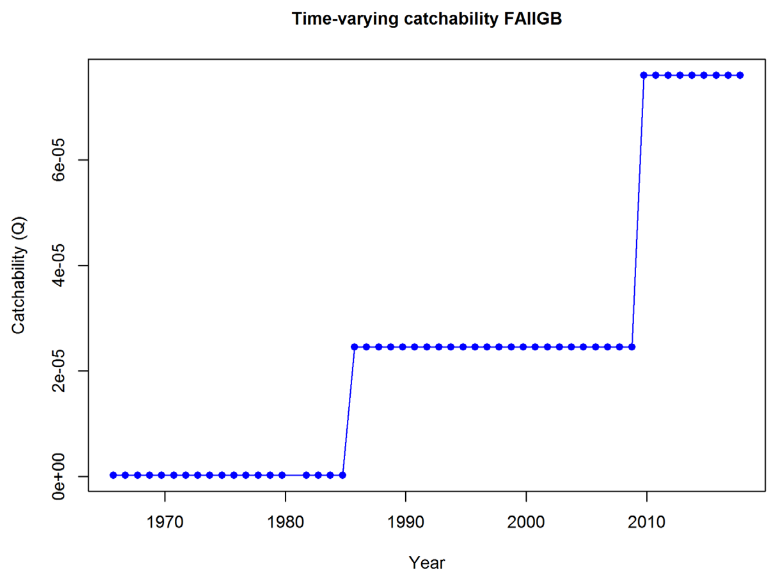



Log index Food
Log index
Year
1980
1990
2000
2010
1
0
-1
-2
-3
-4
-5
-6

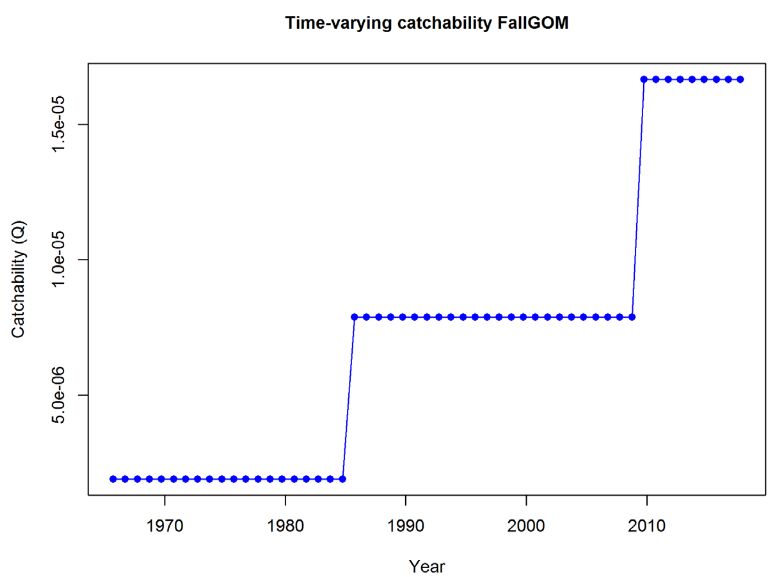

Log index Shrimp(GOM)
Log index
Year
1985
1990
1995
2000
2005
2010
2015
4
2
0
-2


Log index SpringGB
Log index
Year
6
4
2
0
-2
1970
1980
1990
2000
2010


Log index SpringGOM
Log index
Year
6
4
2
0
-2
1970
1980
1990
2000
2010

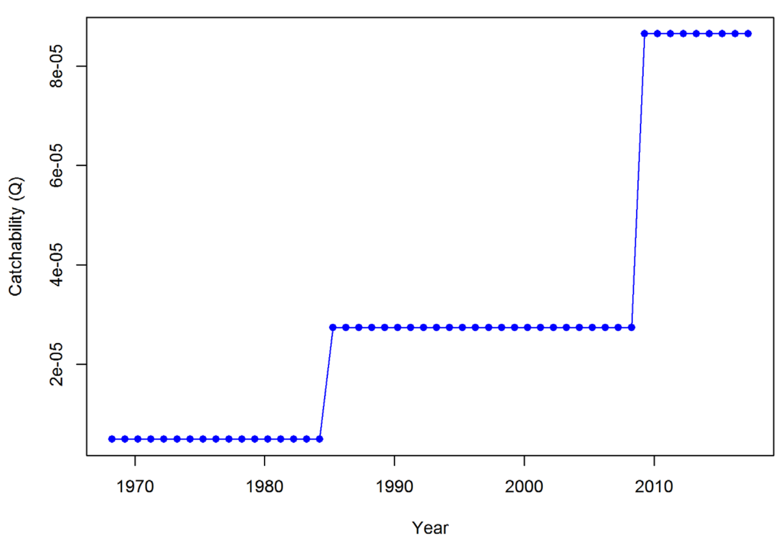


Time-varying catchability FAllGB
Catchability (Q)
0e+00
2e-05
4e-05
6e-05
1970
1980
1990
2000
2010
Year

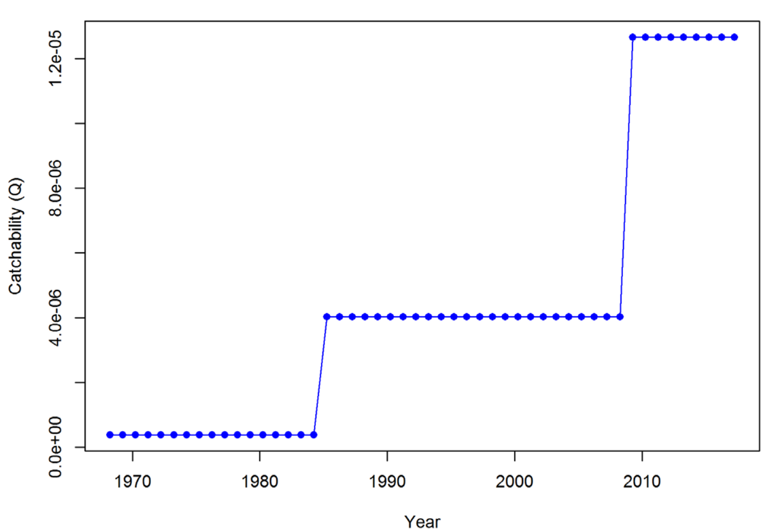


Time-varying catchability FallGOM
Catchability (Q)
5.0e-06
1.0e-05
1.5e-05
1970
1980
1990
2000
2010
Year

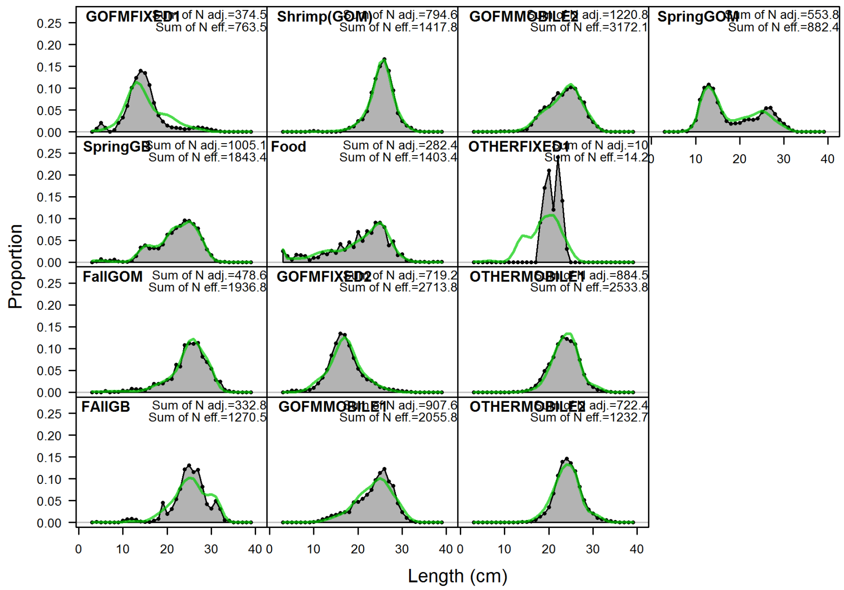
Time-varying catchability SpringGB

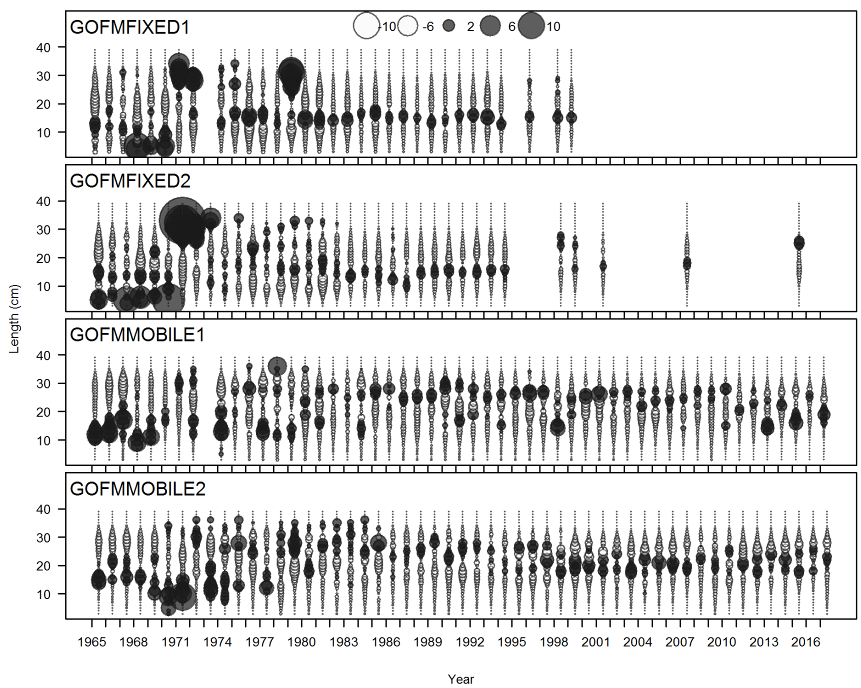


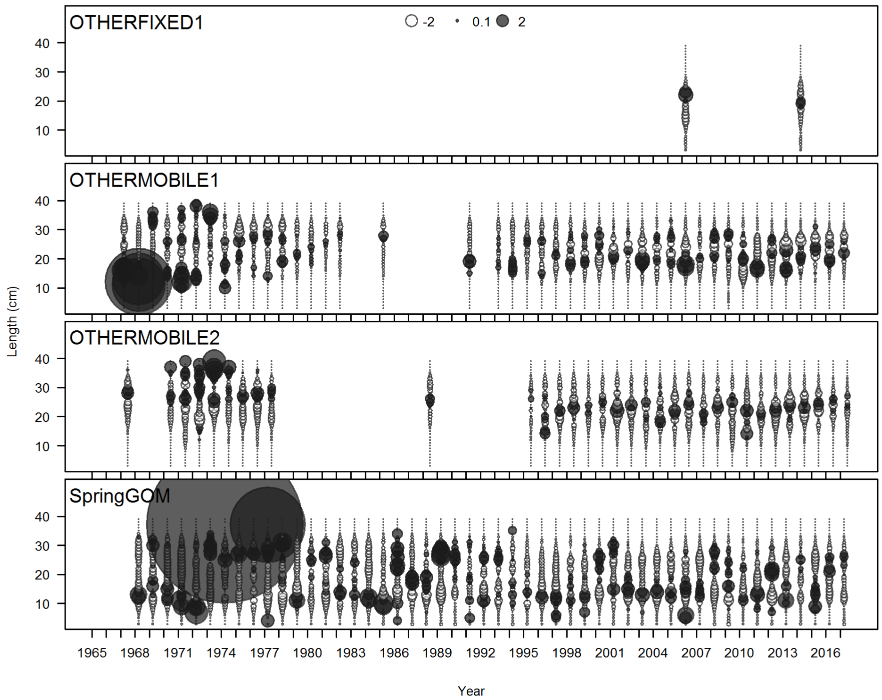
Time-varying catchability SpringGOM

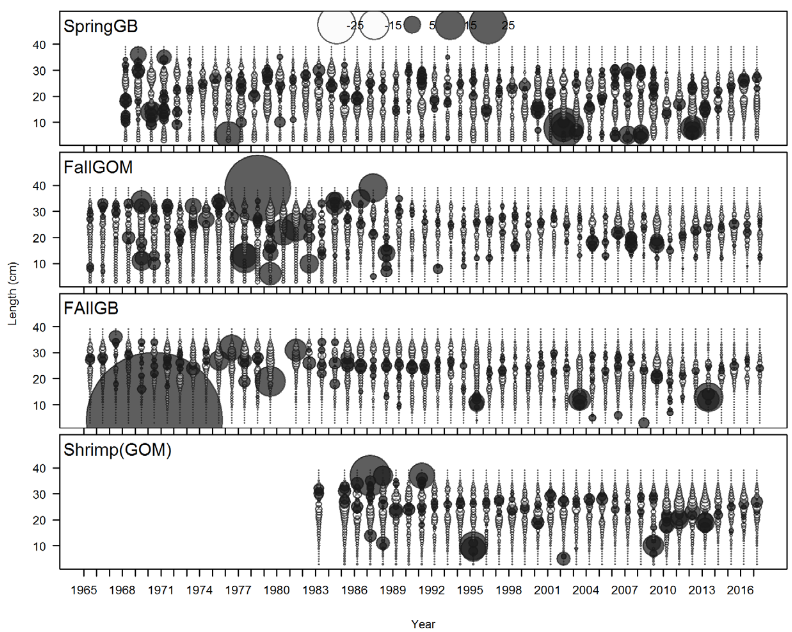


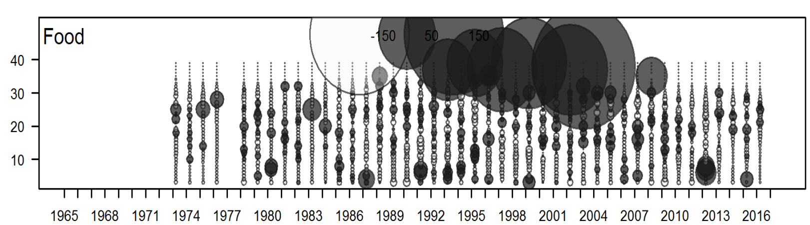
Fits to length composition data aggregated among years were relatively good. These relatively good fits to the aggregate data did not always translate to relatively good fits when examined by fleet, year, and length. Many of the disaggregated fits exhibited residual patterns, most likely suggesting mis-specification of growth and/or selectivity, or conflicts among data sources.





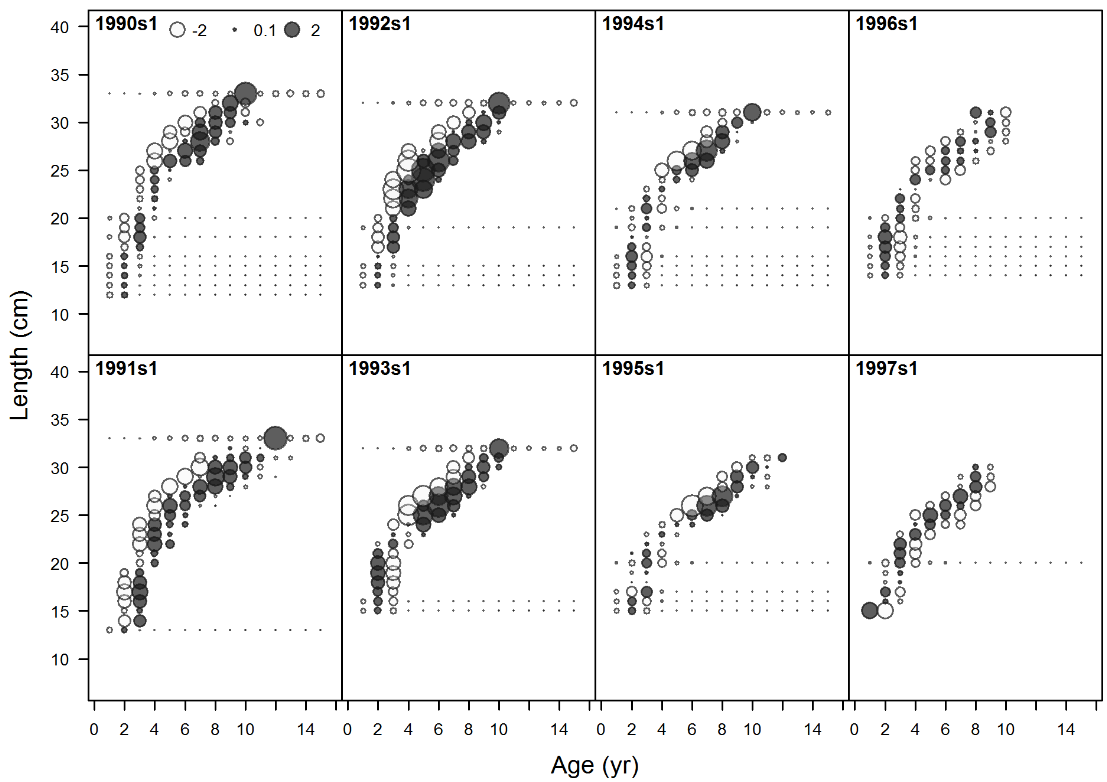

OTHERFIXED1
-2
0.1
2
OTHERMOBILE1
OTHERMOBILE2
SpringGOM
Length (cm)
Year
1965 1968 1971 1974 1977 1980 1983 1986 1989 1992 1995 1998 2001 2004 2007 2010 2013 2016

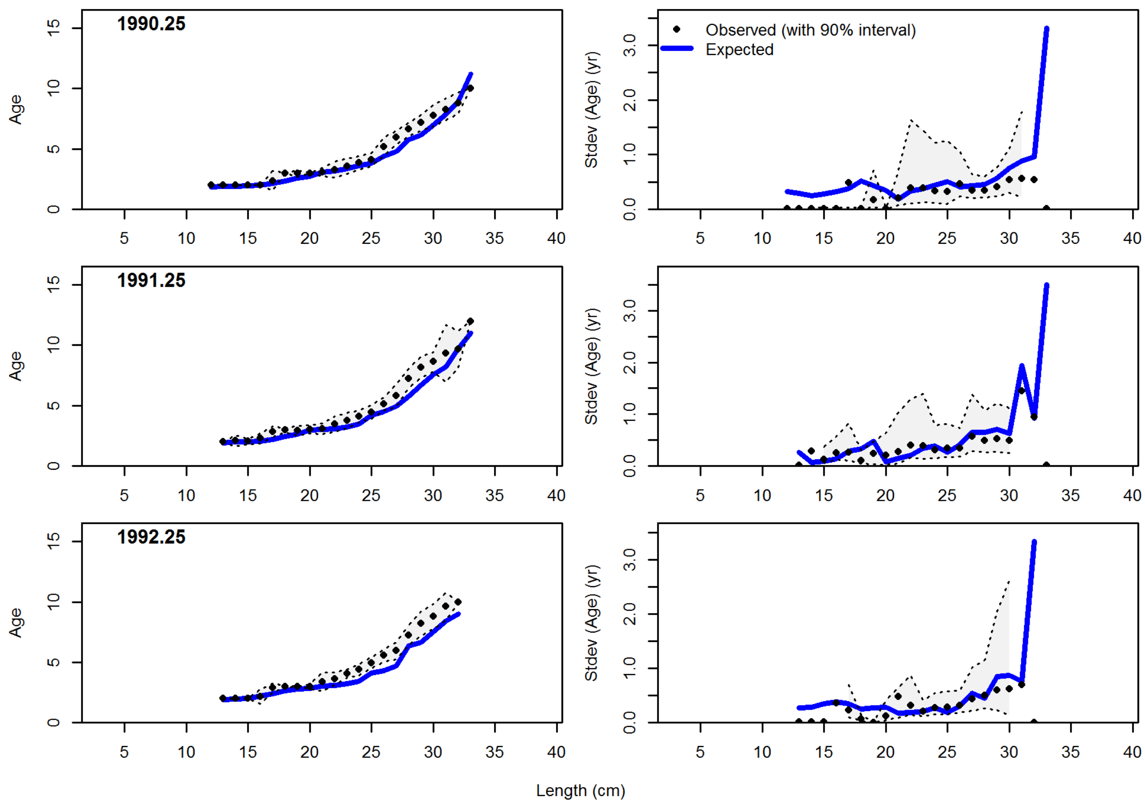

SpringGB
-25
-15
5
15
25
FallGOM
FAllGB
Shrimp(GOM)
Length (cm)
Year
1965 1968 1971 1974 1977 1980 1983 1986 1989 1992 1995 1998 2001 2004 2007 2010 2013 2016


Food
-150
50
150
1965 1968 1971 1974 1977 1980 1983 1986 1989 1992 1995 1998 2001 2004 2007 2010 2013 2016

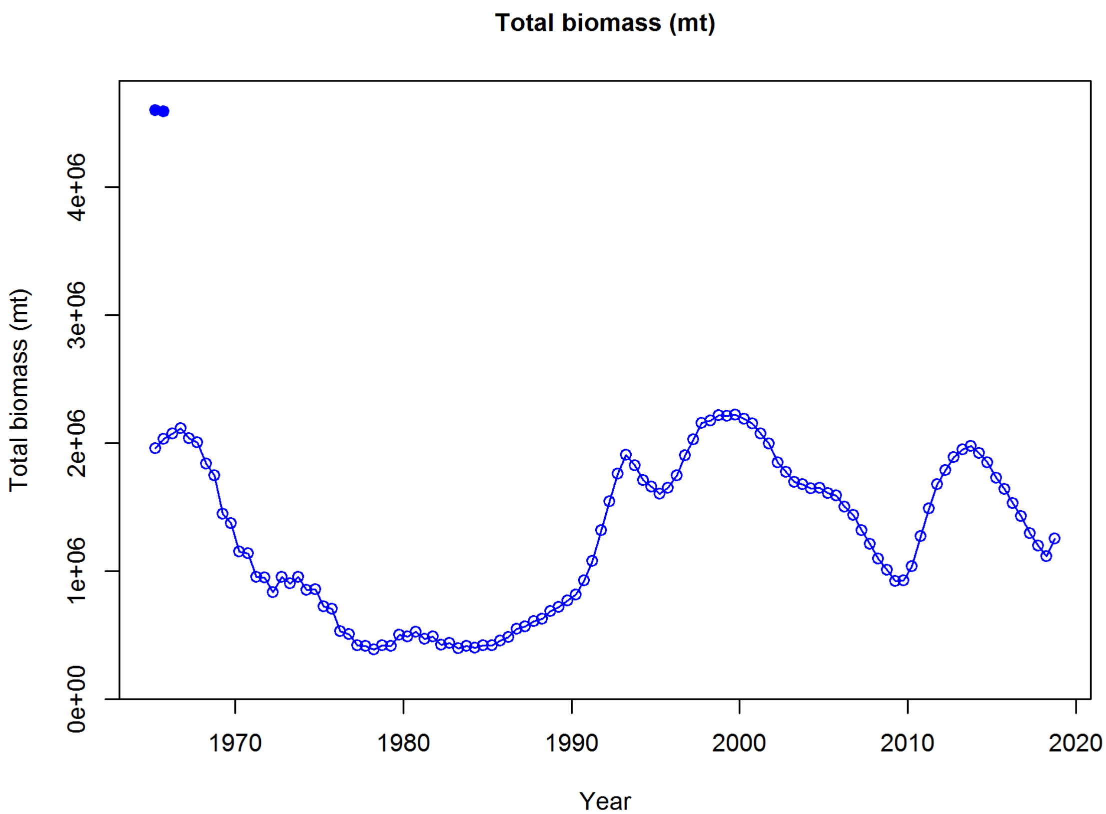
Fits to conditional age at length data varied by season and fleet. Patterns in the fits were evident in most cases, however. Again, these patterns most likely suggest mis-specification of growth and/or selectivity, or conflicts among data.

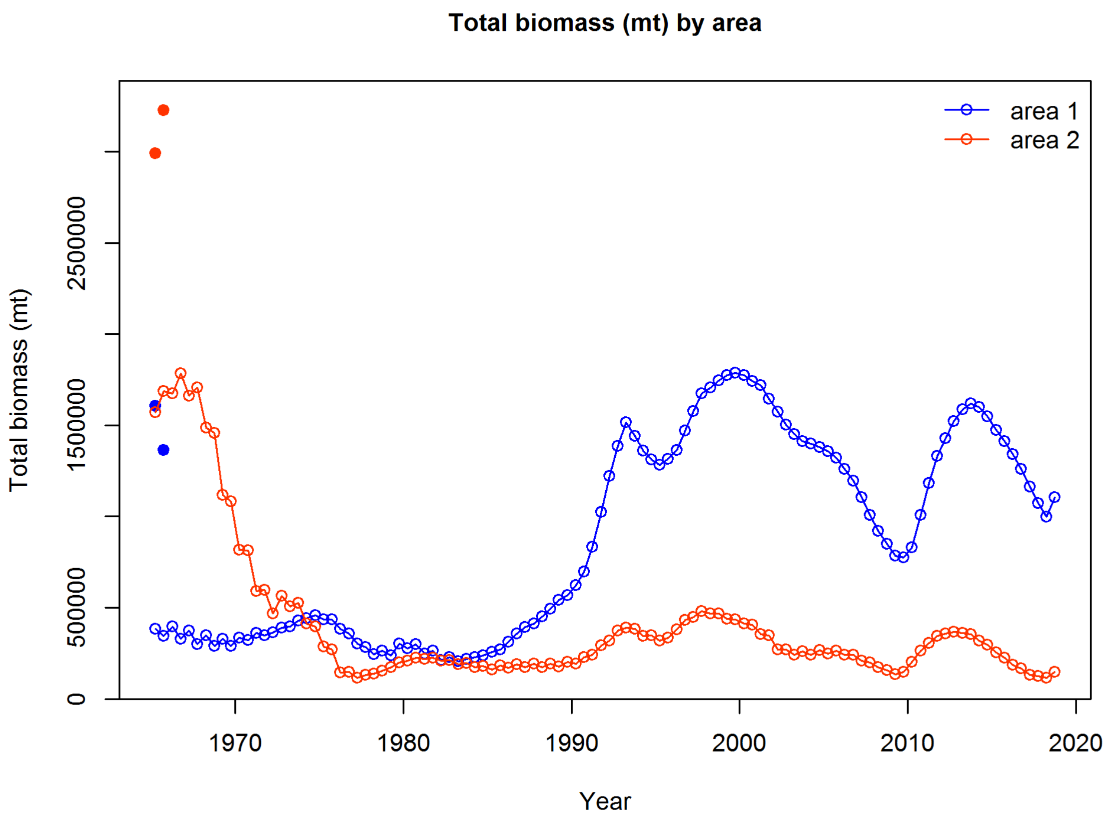
For example, below is the bubble plot for the mobile fleet in the Gulf of Maine in semester one of the years 1990-1997 (top) and the fit to mean age at length for the same fleet during the years 1990-1992 (bottom).



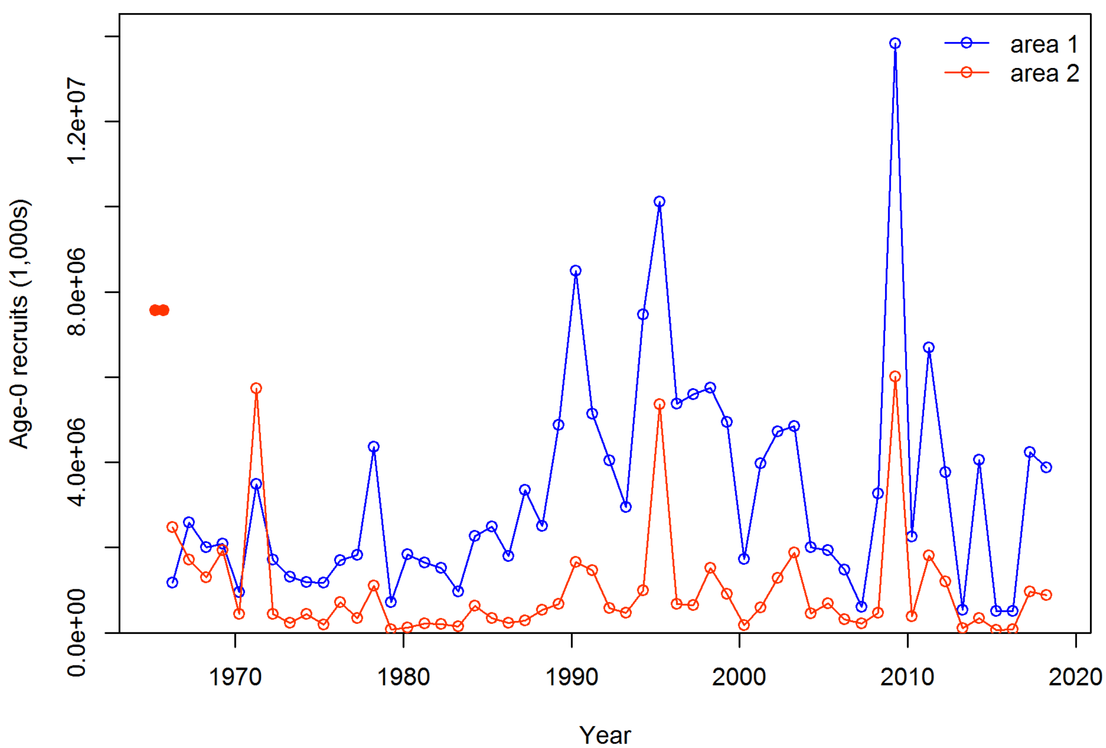


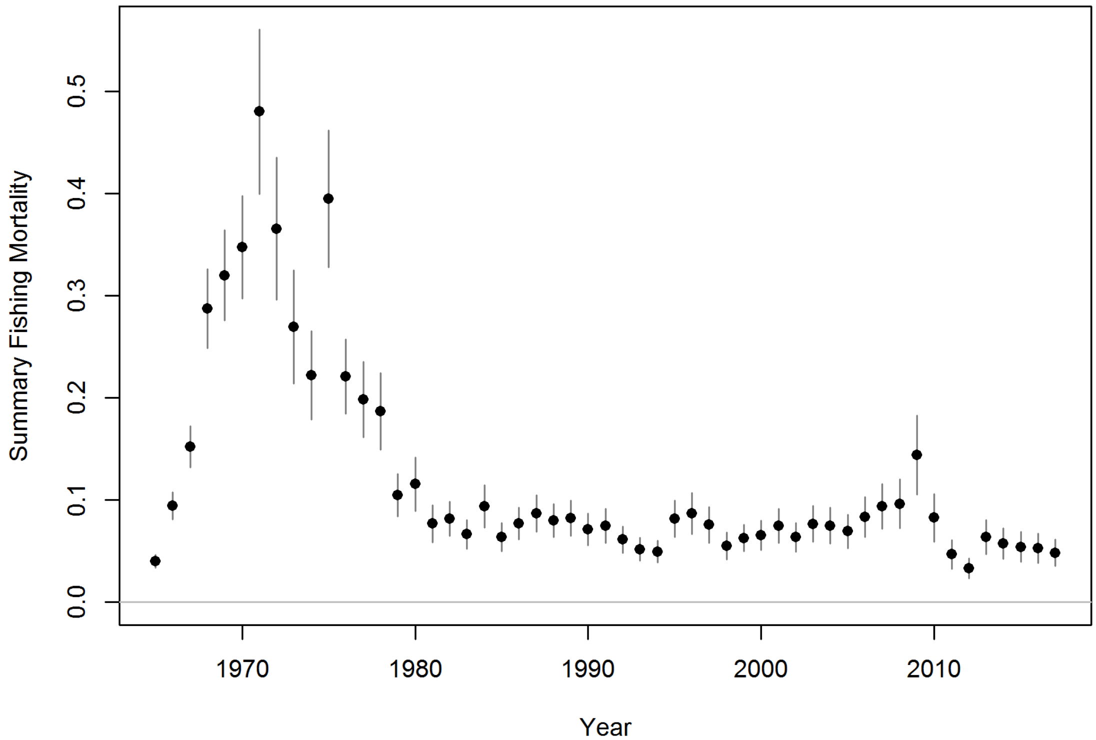
Total biomass declined from the beginning of the time series (1965) through the mid-1980s, reached a second peak in the early 2000s, and a third in ~2012. Total biomass has declined since about 2012. The SS model estimates that most of the biomass occurs in the Gulf of Maine (area1), and this is likely a direct outcome of the model estimating a larger portion of recruitment occurring in the Gulf of Maine. The model could not reliably estimate movement rates between areas, however, which would change the relative biomass by area and potentially the recruitment apportionment. So this result is not trustworthy. Average fishing mortality (averaged over ages 5-15) was highest in the early 1970s and has been relatively low in more recent years.


Total biomass (mt)
Total biomass (mt)
0e+00
1e+06
2e+06
3e+06
4e+06
1970
1980
1990
2000
2010
2020
Year


Total biomass (mt) by area
area 1
area 2
Total biomass (mt)
0
500000
1500000
2500000
1970
1980
1990
2000
2010
2020
Year

Age-0 recruits (1,000s) by area





6. Discussion

- Results of this model should not be used to inform management.
- The model could not reliably estimate movement rates between areas, with parameters often hitting bounds (see above) or the model not converging. The inability to estimate movement likely stems from a lack of information in the data. The model currently assumes that all fish from each area return to their area of birth to reproduce in the second season of each year. Consequently, fishery and survey catches in each area in the second season were assumed to be comprised of all fish from a given area. These assumptions are likely not valid. Without additional information on stock composition or movement rates (e.g., tagging, morphometrics), conducting a multi-area assessment with movement may not be feasible. That said, spatial model runs without movement were conducted with generally similar estimated outcomes.
- The model estimated an increasing amount of biomass in the Gulf of Maine than in other areas. This is consistent with kernel density plots based on the NMFS bottom trawl surveys (TOR B3), although the surveys are not independent of the model results as they were used to fit the SS model.

- The recruitment time series estimated by SS was generally consistent with the ASAP base model (TOR B4), especially the relatively poor recruitment in recent years. The SS model did not estimate the recruitments in 2016 or 2017, however. Rather, they were set equal to the underlying mean recruitment, and so are not comparable to the ASAP results.
- The general trend in herring biomass estimated by the SS model is generally consistent with the ASAP base model, exhibiting a decline, an increase into the early 2000s, and declines over about the last 5 years.
- The SS model's ability to fit to length composition data may increase the utility of surveys without age composition data, because SS could estimate selectivity as opposed to fixing an age-based selectivity pattern. The use of length composition data, however, requires estimation (or pre-specification) of growth parameters, and these will likely have to be time-varying, which will increase model complexity, the number parameters, run-time, and may prove challenging, as it was here. Examples of SS assessments modeling time-varying growth are however extant.
- Continued analysis of the diagnostics from the SS model (e.g., fits to conditional age at length and length composition) may reveal sources of conflict in the data that may not be evident using other modeling platforms that are strictly age based.

Appendix B5

Working Paper: Predation Pressure Index to Inform Natural Mortality
Jonathan J. Deroba

Objective

Develop an index of predation pressure (Richards and Jacobson2016) to inform time-varying natural mortality (M) in Atlantic herring.

Methods

Food habits and data to estimate indices of herring predator biomass were collected on NMFS Northeast Fisheries Science Center spring and fall bottom trawl surveys. Details about the methods for sampling food habits data, including the stomach contents data used here, can be found in Link and Almeida (2000) and Smith and Link (2010). Details about bottom trawl survey design and sampling can be found in Grosslein (1969), Azarovitz (1981), and Miller et al. (2010).

A predation pressure index (PPI) was estimated for predators that had at least 10 stomachs containing herring and positive occurrences of herring in at least 0.1% of stomachs, combined among all years and seasons. These criteria were met by 15 predators (Table 1), and the list is similar to that used to estimate annual consumption in recent herring assessments (NEFSC 2012).

A percent frequency of occurrence of herring in predator stomachs was estimated as the percentage of stomachs containing herring (P), combined among years and seasons:

$$P_p = \frac{\sum_y S_{y,p}}{\sum_y T_{y,p}} \times 100;$$

where p is predator, y is year, S is the number of stomachs containing herring (i.e., positive occurrences), and T is the total number of stomachs examined.

Annual indices of predator biomass (B; stratified mean kg per tow) were estimated for the spring and fall bottom trawl surveys for each of the predators (except striped bass and sea raven, see below). Indices for each predator were estimated as in their respective stock assessments (index values were downloaded from the PopDy Branch "ADIOS" system on June 8, 2017). For predators that have indices estimated separately by region or sex (Table 1), values were summed to obtain season specific (spring and fall) annual indices. Sea raven were excluded from the analysis because this species has no stock assessment and indices were unavailable, but they likely account for a relatively small amount of herring predation (NEFSC 2012), and so results and conclusions are likely robust to this omission. Time series of indices of biomass and percent frequency of occurrence began in 1968 in the spring and 1963 in the fall.

The striped bass stock assessment does not use the spring and fall bottom trawl survey data for indices of biomass because the gear does not provide a suitable index. In order to accommodate striped bass in the calculation of PPI, estimates of total striped bass biomass from the stock assessment were rescaled to equal the average of all the other predator biomass indices among seasons and years, and this annual quantity was used for both seasons in the calculation of PPI:

$$B_{y,bass} = \frac{E_y}{\bar{B}};$$

where $B_{y,bass}$ is the value treated as the year specific index of biomass for striped bass in both seasons, $E$is the estimate of total striped bass biomass from the stock assessment, and $\bar{B}$ is the mean index of predator biomass among all other predators, years, and seasons. This rescaling of the striped bass total biomass estimates was done so that the scale of the index values used for striped bass in the calculation of PPI were similar to other predators. The PPI was calculated with and without striped bass included, and results were presented separately. The striped bass stock assessment begins in 1982, and so PPI calculations that included striped bass also began in 1982.

Season and year specific PPI was calculated as the weighted average of the predator indices of biomass (B):

$$PPI_{y,s} = \sum_p B_{y,s,p} \times P_p;$$

where s denotes season (spring or fall).

Time-varying M was calculated by adjusting a base M by annual deviations in PPI from the mean PPI:

$$M_{y,s} = \frac{M_b \times PPI_{y,s}}{\overline{PPI_s}};$$

where M_b was a baseline level of natural mortality, which equaled 0.35 for demonstration purposes, but was derived from Hoenig (1983) and has been used in previous herring stock assessments (NEFSC 2012).

Results

Temporal trends in PPI were similar between seasons, with and without striped bass. Without striped bass, PPI declined from the beginning of each time series, varied without trend below the time series means after ~1990, and increased since ~2010 to near the time series means in the most recent year (Figure 1). With striped bass, PPI has generally varied without trend near the time series means (Figure 1). Results for M were similar to PPI. Without striped bass, M declined from the beginning of each time series, varied without trend below the below the baseline rate after ~1990, and increased since ~2010 to near the baseline level (Figure 2). With striped bass, M generally varied without trend near the baseline level (Figure 2).

Discussion

The 2012 herring stock assessment increased M from 0.35 (averaged among ages) to 0.50 beginning in 1996 (NEFSC 2012). This increase in M eliminated a retrospective pattern and produced generally consistent amounts of consumption between that implied by the input M and that estimated from stomach contents data. These two justifications for increased M no longer held in the 2015 updated herring stock assessment, with a worsened retrospective pattern and consumption of herring implied by the input M being higher than that estimated from stomach contents data (Deroba 2015). The trends in PPI, and subsequent deviations from M_b, were also inconsistent with the increased M rates used in the 2012 and 2015 herring stock assessments.

The PPI and consumption estimates both use some of the same stomach contents data, but suggest different conclusions about variation in M. This inconsistency could be related to caveats in the calculation of PPI, consumption, the herring stock assessment, or a combination. The estimates of consumption, for example, have been criticized as likely to be biased in scale and trend due to reliance on estimates of predatory biomass from stock assessments and other sources with different underlying structural assumptions and uncertainties (Brooks and Deroba 2015). The strength of evidence for an increase in M provided by the estimates of consumption also depends on aspects of the herring assessment itself. Assumptions and input data to the herring assessment determine the scale and trend of the resulting assessment estimates, and estimates of consumption from the stomach contents data are compared to consumption implied by the input M and this requires use of herring assessment estimates. The assumptions and input data for the herring assessment, however, are also subject to uncertainties.

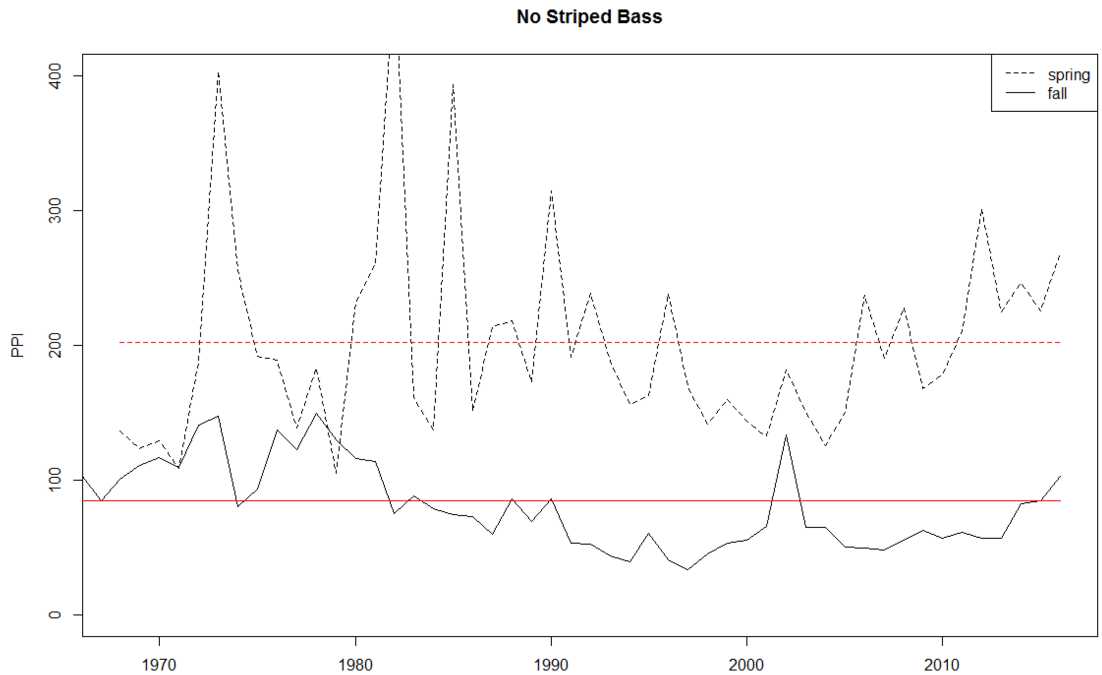
The PPI and consumption calculations also ignore spatial and seasonal variation (other than spring and fall) in the overlap and efficiency of predators and Atlantic herring. The probability of a predator stomach containing herring and the amount of herring in a stomach vary seasonally and spatially (Deroba 2018). Ignoring such variation may cause bias and a false sense of precision in the PPI and consumption estimates.

References

Azarovitz, T.R. 1981. A brief historical review of the Woods Hole Laboratory trawl survey time series. Can. Spec. Publ. Fish. Aquat. Sci. 58: 62-67.

Brooks, E.N., Deroba, J.J. 2015. When “data” are not data: the pitfalls of post hoc analyses that use stock assessment model output. Canadian Journal of Fisheries and Aquatic Sciences 72: 634-641.
Deroba, J.J. 2015. Atlantic herring operational assessment report 2015. US Dept. Commer. NEFSC Ref. Doc. 15-16; 30pp.
Deroba, J. J. 2018. Sources of variation in stomach contents of predators of Atlantic herring in the Northwest Atlantic during 1973–2014. ICES Journal of Marine Science, doi:10.1093/icesjms/fsy013.
Grosslein, M.G. 1969. Groundfish survey program of BCF at Woods Hole. Commer. Fish. Rev. 31: 22-35.
Hoenig, J.M. 1983. Empirical use of longevity data to estimate mortality rates. Fishery Bulletin 82(1): 898-903.
Link, J.S., Almeida, F.P. 2000. An overview and history of the food web dynamics program of the Northeast Fisheries Science Center, Woods Hole, MA. NOAA Tech. Mem. NMFS-NE-159.
Miller, T., Das, C., Politis, P., Long, A., Lucey, S., Legault, C., Brown, R., Rago, P. 2010. Estimation of Henry B. Bigelow calibration factors. NEFSC Reference Document 10-05, 230pp.
NEFSC. 2012. 54th Northeast Regional Stock Assessment Workshop (54th SAW) Assessment Report. US Department of Commerce, Northeast Fisheries Science Center Reference Document 12-18; 600 p.
Richards, R.A., Jacobson, L.D. 2016. A simple predation pressure index for modeling changes in natural mortality: Application to Gulf of Maine northern shrimp stock assessment. Fisheries Research 179: 224-236.
Smith, B.E., Link, J.S. 2010. The trophic dynamics of 50 finfish and 2 squid species on the Northeast US Continental Shelf. NOAA Tech. Mem., NMFS-NE-216.

Figure 1.—Predation pressure index (PPI) for Atlantic herring calculated in the spring and fall, without (top panel) and with (bottom panel) striped bass. Red horizontal lines are time series means in the spring and fall.



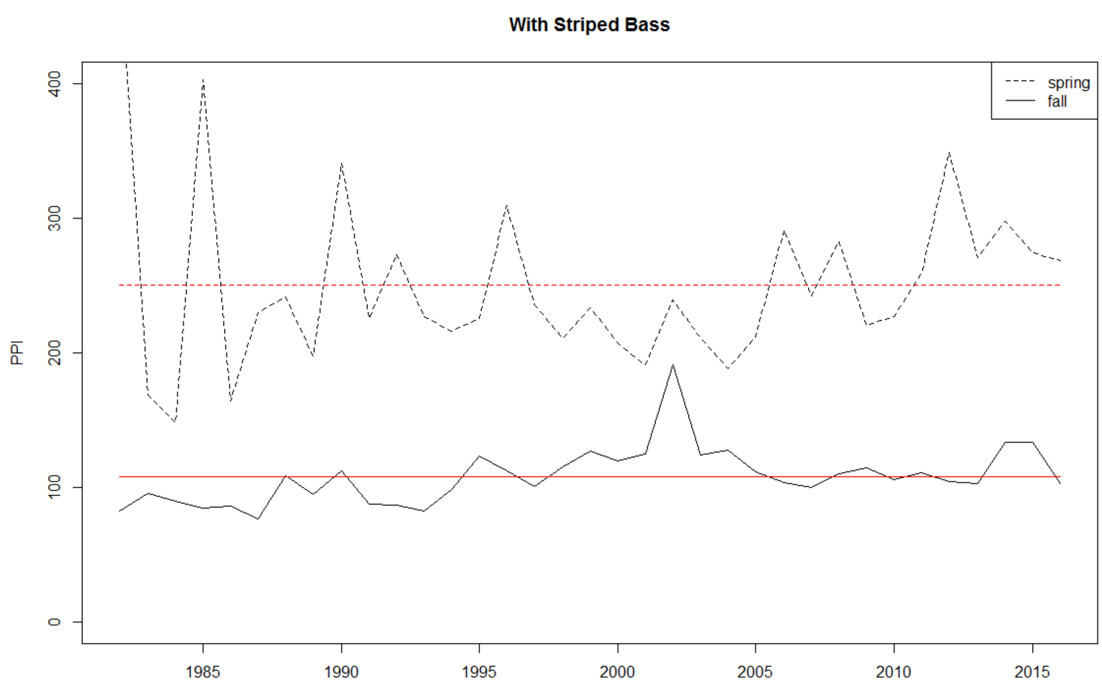


Figure 2.—Natural Mortality (M) for Atlantic herring calculated in the spring and fall, without (top panel) and with (bottom panel) striped bass. Red horizontal lines indicate a baseline M level of 0.35.

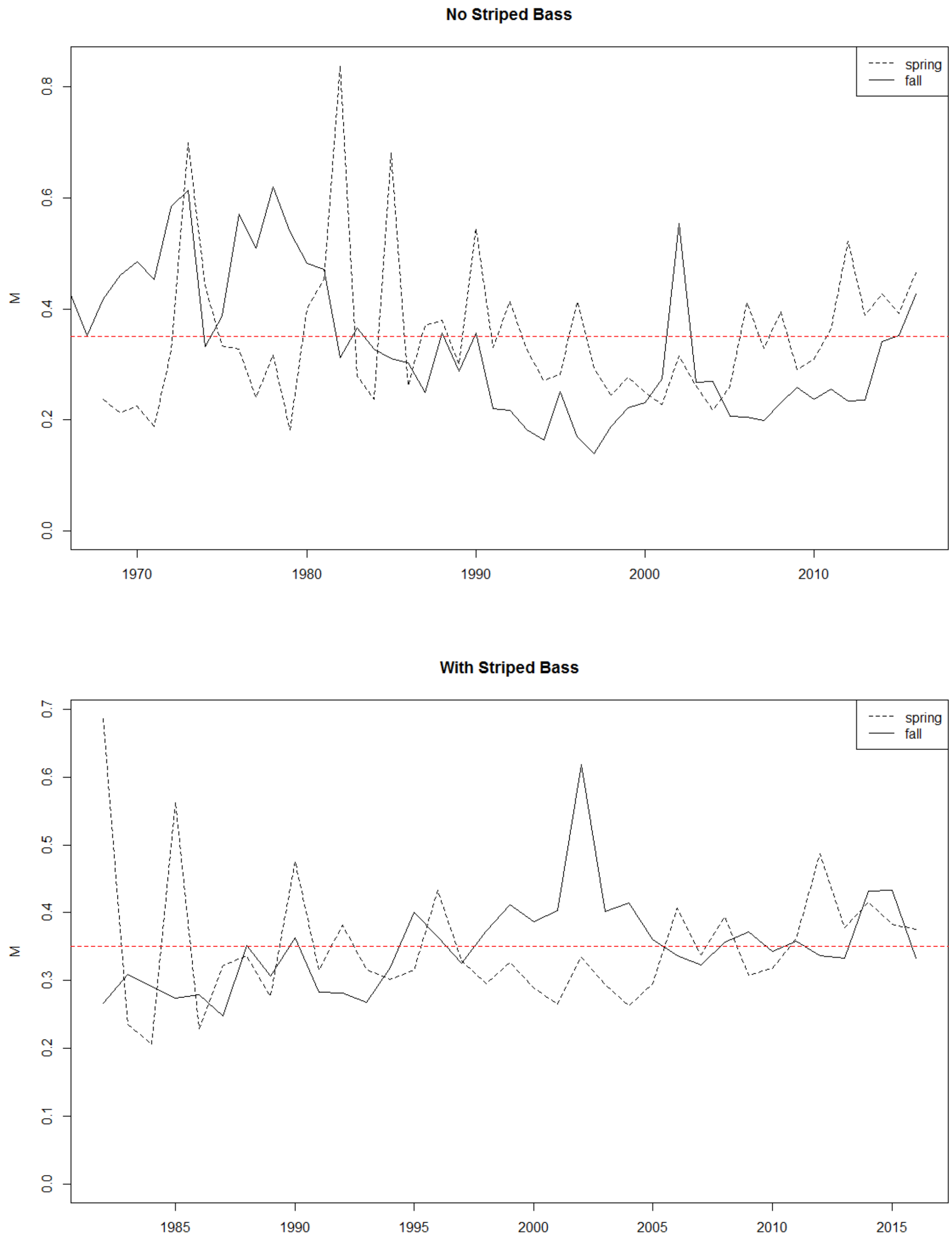


Table 1.—Herring predators that had at least 10 stomachs containing herring and positive occurrences of herring in at least 0.1% of stomachs, combined among all years and seasons, and used to estimate a predation pressure index.

PREDATOR	**Region (R) or Sex (S) distinctions**
ATLANTIC_COD	R – Georges Bank, Gulf of Maine
ATLANTIC_HALIBUT	NA
BLUEFISH	NA
GOOSEFISH	R – North, South
HADDOCK	R – Georges Bank, Gulf of Maine
POLLOCK	NA
RED_HAKE	R – North, South
SEA_RAVEN	NA
SILVER_HAKE	R – North, South
SPINY_DOGFISH	S – Male, Female, Unidentified
STRIPED_BASS	NA
SUMMER_FLOUNDER	NA
THORNY_SKATE	NA
WHITE_HAKE	NA
WINTER_SKATE	NA

Appendix B6

The NEFSC Study Fleet Program's Fisheries Logbook Data and Recording Software and its use the Atlantic Herring Fishery

Submitted by

Christopher L Sarro – NEFSC Cooperative Research Program

The Northeast Fisheries Science Centers (NEFSC) Cooperative Research Branch's Study Fleet program began development in late 2000. The pilot program had two main goals: 1) assemble a 'study fleet' of commercial New England groundfish vessels capable of providing high resolution (temporal and spatial) self-reported data on catch, effort and environmental conditions while conducting normal fishing operations; and 2) developing and implementing electronic reporting hardware and software for the collection, recording and transferring of more accurate and timely fishery-based data (Palmer et al 2007).

The program was developed to provide stock assessment scientists with fisheries dependent data that could provide more precise estimates of fishing effort and spatially-specific catch and discard rates. The collaborative nature of the program could also provide a means of communication between industry and science for better understanding of factors driving fishing effort and catch, as well as serve as platform for future collaborative projects (Palmer et al 2007).

Phase I began in late 2002, with a fleet of 15 paid participants, to develop an electronic logbook (ELB) and test supporting hardware. Phase II began in September 2004, with 30 participating vessels, to continue developing the ELB and explore satellite communication (Palmer et al 2007). Study Fleet is currently in Phase III, with a fully functioning ELB for data collection and transfer, auditing and utilization of data and enhanced biological sampling. Currently, there are over 40 contracted vessels in Study Fleet with homeports ranging from North Carolina to New Hampshire. Participating vessels range from New Hampshire to North Carolina with concentrations in Cape May, New Jersey and Point Judith, Rhode Island. The majority of vessels fish bottom trawl gear, though there are also gillnet, longline and scallop vessels participating.

The ELB developed for use in the Study Fleet program was the Fisheries Logbook and Data Recording Software (FLDRS). This is free software developed by the NEFSC, which is capable of reporting on the haul-by-haul and subtrip levels. FLDRS is currently on its fourth version with version five in development. On all vessels, FLDRS connects to a GPS unit or satellite compass and polls the unit every 20 seconds for accurate location information. FLDRS can be integrated with the depth sounder for depth information and/or a vessel monitoring system for rapid data transmission via satellite. The newest version of FLDRS is also capable of sending trip and GPS data via email if the software can access a Wi-Fi connection.

Gear-mounted temperature/depth probes are also deployed on vessels. The temperature/depth probes collect temperature and depth data every 90 seconds. Earlier models would poll continually and needed to be downloaded every 30-90 days. Current models are depth triggered and the data is uploaded after each tow to an onboard computer via a Bluetooth connection. Future improvements to FLDRS will allow for email transmission of temperature/depth files as well.

FLDRS can collect data on both the trip and haul levels. On the trip level, FLDRS collects program code, vessel and operator information, sail and landing date and port, number of efforts, aggregated fishing time, catch, apportionment and dealer information. On the haul level, FLDRS also collects fishing gear, tow specific location, duration, depth, statistical area and catch information.

FLDRS is also capable of collecting 'Dynamic Data'. These are additional data elements that are specific to certain gear types or program code. The Herring Program Code was developed with Massachusetts Division of Marine Fisheries (MADMF) to assist with their River Herring Bycatch Avoidance Program. Under the Herring Program Code, captains record which herring management sub-area they intend on fishing, the percent river herring catch and estimated river herring weight if a fisheries observer is present.

During installation, FLDRS is customized to each vessel and its fishing activities. All the various gears that a vessel uses, with the necessary gear characteristics (gear code, sweep length, mesh size and mesh type) are saved in FLDRS. Each gear is also associated with a customized species list of the most common kept and discarded species caught with that particular gear. All dealers that a vessel sells to are added to a dealer list. Finally, vessels' defaults are set up; these are the operator, gear, port, crew size and trip type that populate in the software automatically.

In July 2011, the Greater Atlantic Regional Office (formerly the Northeast Regional Office) approved the use of electronic Vessel Trip Reporting (eVTR) for a segment of the groundfish fleet and was expanded to other fisheries in 2013. FLDRS is now one of six approved eVTR platforms. During 2014-2015, the Cooperative Research Branch collaborated with the Pacific States Marine Fisheries Commission (PSFMC) to expand electronic reporting in the Northeast fisheries. This effort made funding available for up to 120 participating vessels to receive computers, installation, hardware and training in the use of FLDRS. A subset of these vessels also received temperature/depth probes. To date, 234 vessels have used FLDRS for haul-by-haul and subtrip reporting for research and eVTR purposes (Figure 1).

Use of FLDRS in the Atlantic herring Fishery:

Atlantic herring catches start in the Cooperative Research database in 2006. That year only a single vessel landed more than 2,200 lbs of Atlantic herring on an individual trip. The number of participant vessels participating in the Atlantic herring fishery ranged from one in 2006 to seven in 2013 with the vast majority of fishing effort coming from the small-mesh bottom trawl fishery off Rhode Island. Cooperative Research staff, through coordination with MADMF, made a concerted effort to install FLDRS on the mid-water and paired mid-water vessels through the collaboration with the PSMFC. This increase in vessels has greatly increased the amount of data collected from Atlantic herring fishery including fishing effort and the geographic footprint of the fishery. (Figures 2-4).

Midwater gear (both single and paired) is the most commonly occurring gear type in the time series. However, some small-mesh bottom trawl vessels out of Point Judith, RI will report using gear code 097OTM. The summer purse seine fishery in management sub-area 1A is not strongly represented with only one boat reporting using FLRDS in 2016 and two in 2017 (Figure 5). Vessels reporting haul-by-haul using FLDRS represented only 0.23 % of the total Atlantic herring landings in 2006. However, in 2016, vessels using FLDRS accounted for 41 % of the total Atlantic herring landings (Figure 6).

The participation of the commercial Atlantic herring fishery in haul-by-haul reporting has allowed for the collection of detailed information on effort and catch. Cooperative Research is attempting to integrate more of the onboard equipment such as net mensuration equipment for more accurate estimation of fishing time and catch per unit effort. This information combined with future improvements to FLDRS should be able address specific research and management questions.

Cooperative Research staff has fostered a close relationship with the commercial Atlantic herring industry, includes sailing on commercial vessels to examine trends in river herring bycatch and conducting a dedicated study to evaluate a predictive river herring distribution model in the small-mesh bottom trawl fishery. The direct lines of communication between Cooperative Research staff and members of the commercial Atlantic herring industry also provide insight into factors affecting fishing effort beyond availability. Variables such as fuel prices, market forces, seasonal closures, catch caps and availability of other species can influence catch beyond availability of the target species. Providing this information to stock assessment scientists and fisheries managers could prove valuable moving forward.

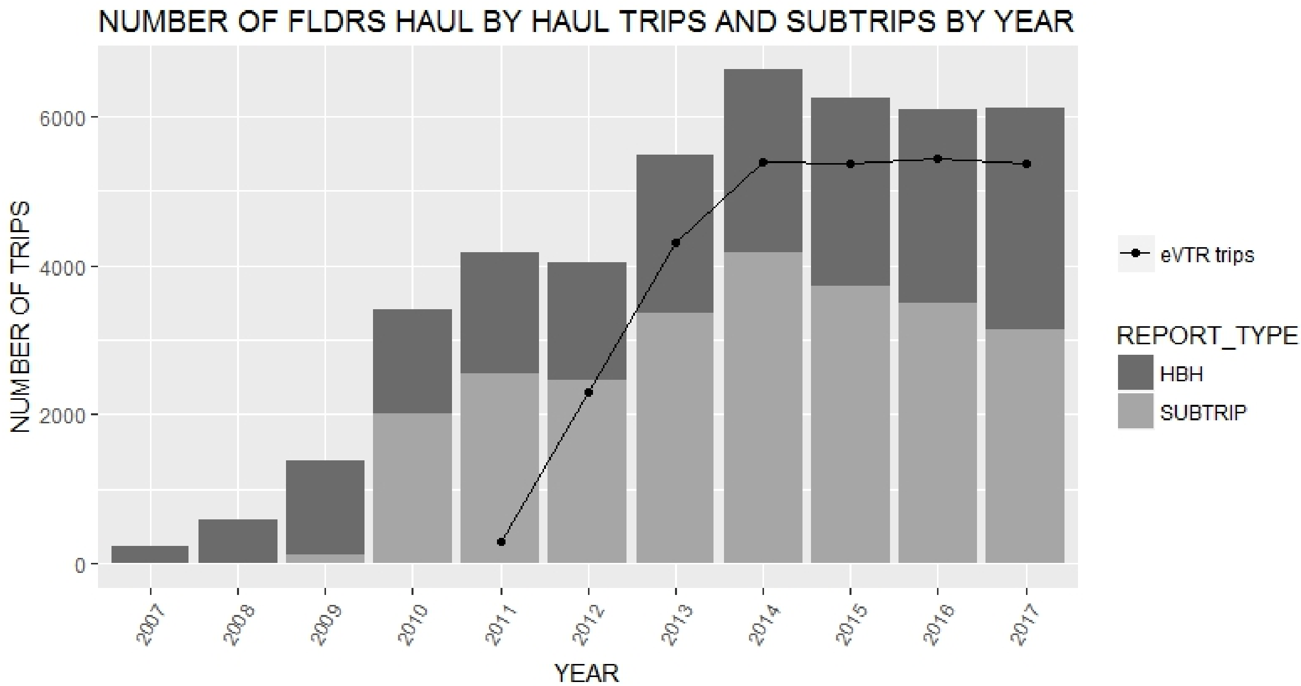


Figure 1: The number of trips (haul-by-haul and subtrip) and eVTRs per year from vessels using FLDRS.

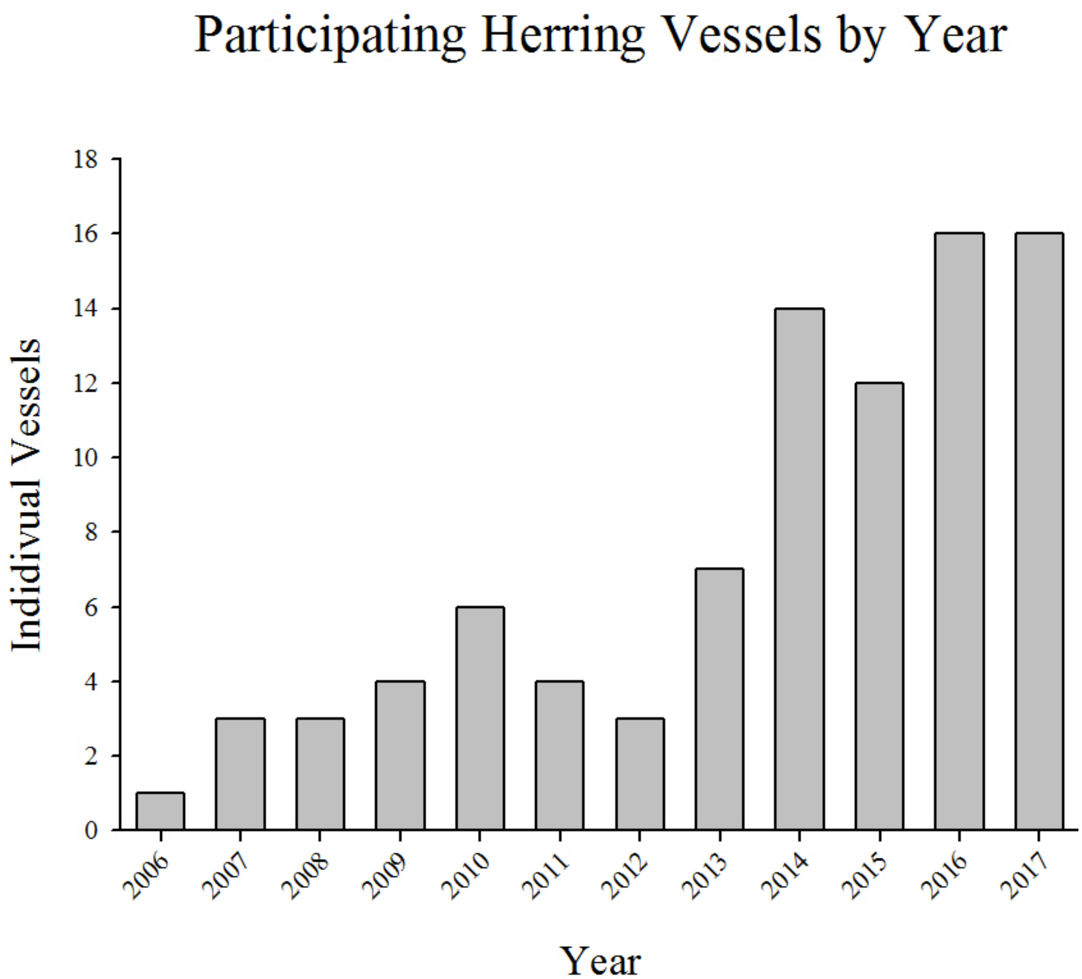


Figure 2: The number of individual vessel permit numbers that reported at least one haul-by-haul trip that landed > 2,200 lbs of Atlantic herring.

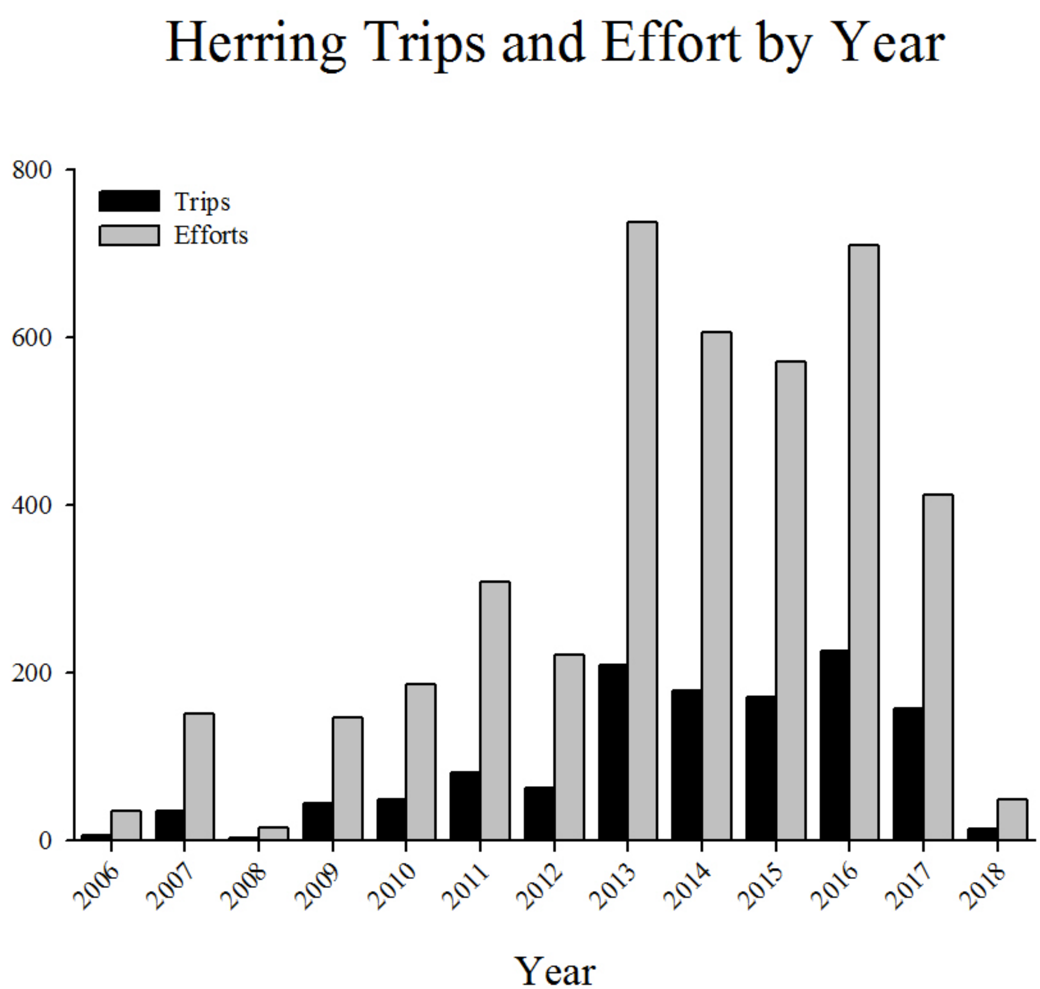


Figure 3: The number of trips and efforts that landed > 2,200 lbs of Atlantic herring. Trips and efforts from paired midwater trawlers were counted together as a single trip or effort.

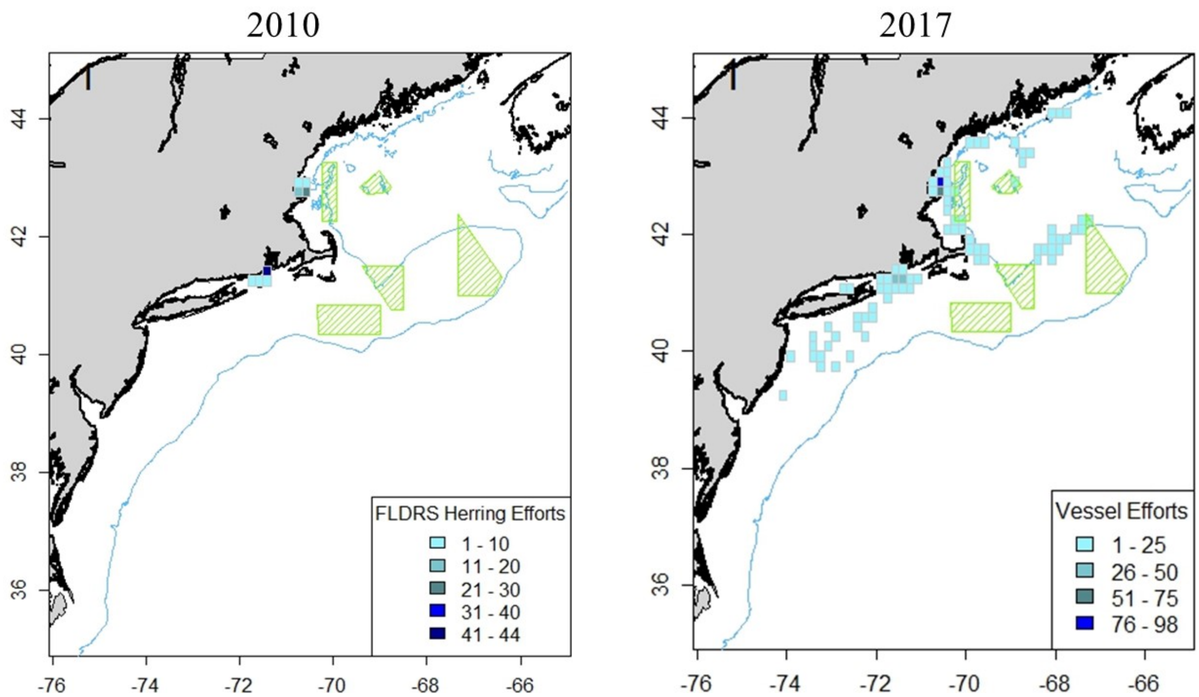


Figure 4: The number of efforts per 10 minute squares on trips landing more than 2,200 lbs of Atlantic herring from Cooperative Research Participants using FLDRS in 2010 and 2017.

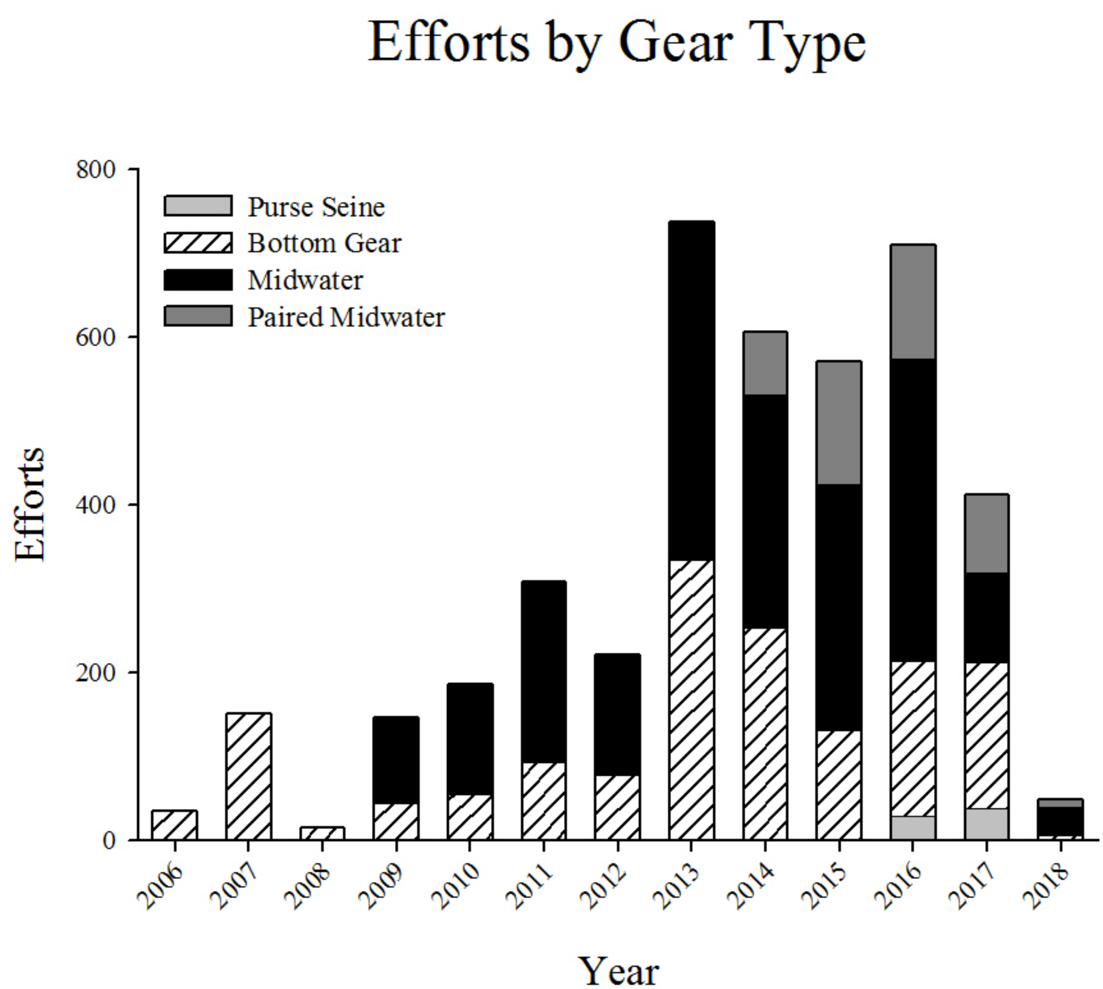


Figure 5: The number of efforts made using the various gear types by year. Bottom gear includes gear codes 090OTO, 092OTF and 092OTR.

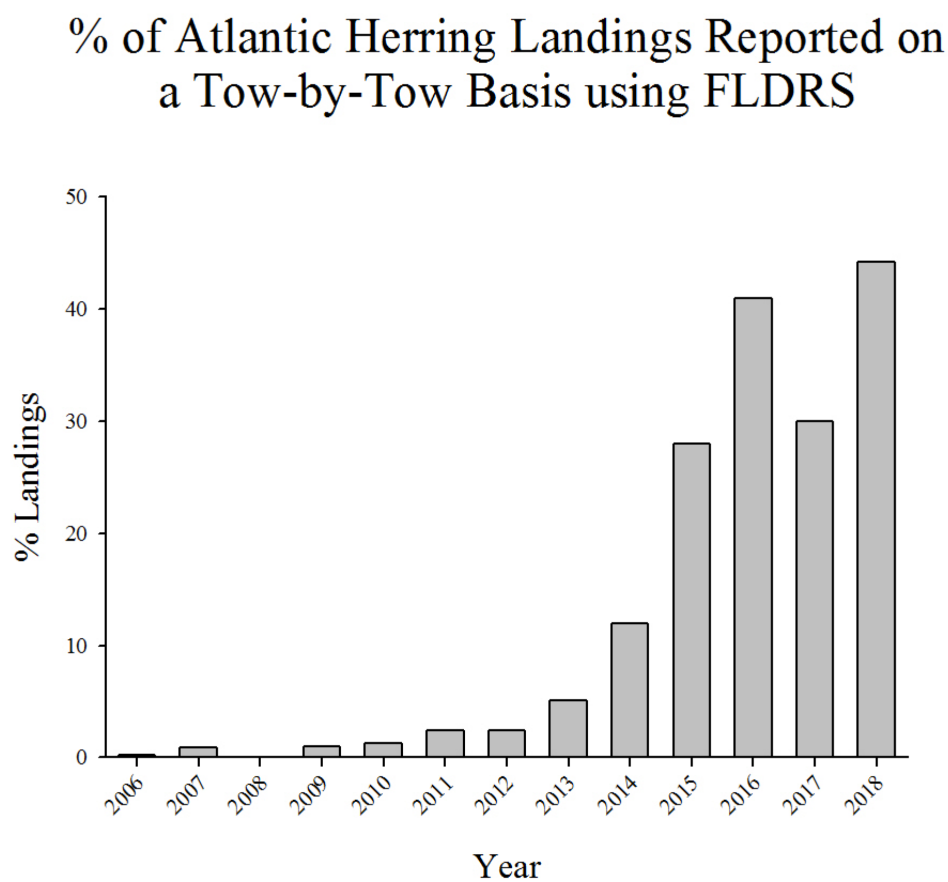


Figure 6: The percentage of Atlantic herring landings from vessels using FLDRS at the haul-by-haul level. The FVTR Apportion table was used to estimate catch and the CFDBS CFDERS tables were used to estimate landings.

References:

Palmer, Michael C., Wigley, Susan E., Hoey, John J., and Palmer, Joan E (2007) An Evaluation of the Northeast Region's Study Fleet pilot program and Electronic Logbook System: Phases I and II. NOAA Technical Memorandum NMFS-NE-204, 78 pp.

Appendix B7

Working Paper:

Maturity and spawning seasonality of Atlantic herring (*Clupea harengus*) in US waters: QA/QC of data collected from fishery dependent and independent sources, with an evaluation of the consequences of skipped or spring spawning for stock-recruit relationships and stock assessment

Mark J. Wuenschel and Jonathan J. Deroba

Introduction and Objective

Altantic herring (*Clupea harengus*) is of commercial importance throughout its range across the eastern and western north Atlantic. Over this broad geographic range, reproductive plasticity is evident; stock specific size and age at maturity, spawning seasonality, egg sizes, and spawning areas (Iles 1964, van Damme et al. 2009). In addition to the diversity of reproductive strategies, skip spawning (i.e. not all mature fish spawn in every year) has become increasingly apparent in several fish stocks (Rideout and Tomkiewicz 2011, Skjaeraasen 2009, Skjaeraasen 2012), and has been reported for herring in the eastern Atlantic (Engelhard and Heino, 2005, Kennedy et al., 2011, Bucholtz et al. 2013). Although recruitment in the western Atlantic is highly variable (Anthony and Fogarty 1985), a recent evaluation of spawning strategies in the region that considers the possibility of skipped spawning is lacking.

Commercial fishery catch samples from the third quarter of each year (July-September) have been used to define annual maturity ogives for input into Atlantic herring stock assessments (NEFSC 2012; Deroba 2015). During the 2015 Operational Assessment of Atlantic herring (Deroba 2015), systematic differences in the maturity-at-age of herring were found between commercial samples and NMFS fall bottom trawl survey samples, with the commercial samples generally having smaller proportions of herring mature-at-age than the survey, especially for age-3 (Figure 1). The commercial sample based age-3 maturity-at-age also had larger interannual variation than the survey samples. Exploratory analyses by length bins (across ages and for age-3) indicated a similar tendency towards lower proportions mature at length (especially for smaller sizes) in the commercial samples (Figure 2). These differences and the variation in age-3 maturity were a noteworthy uncertainty during the 2015 assessment because: 1) spawning stock biomass at maximum sustainable yield (SSB_{MSY}) varies with age-3 maturity at a constant F_{MSY}, 2) a relatively large year class was age-3 in 2014, the terminal year of the assessment, which contributed to a 2014 SSB that exceeded the SSB_{MSY} reference point by more than two-fold, and 3) the assessment estimates a stock-recruit curve that assumes maturity is known without error in the estimation of SSB each year. While none of these uncertainties or concerns were considered grounds to reject the assessment, incorrectly specifying maturity-at-age could lead to bias in MSY reference points, bias in annual estimates of stock and recruitment, and ultimately to incorrect conclusions about stock status and inappropriate management advice.

One possible source of the differences between the commercial and survey samples is that commercial samples are taken by a port sampler with the State of Maine while the survey samples are taken by NMFS. The State of Maine and NMFS use different maturity classification schemes (Table 1).

The accuracy of macroscopic based maturity determinations for Gulf of Maine and Georges Bank Atlantic herring from both the commercial port samples and fishery-independent samples has not been formally investigated. Inaccuracy of maturity staging of herring using macroscopic criteria has been reported elsewhere (McPherson et al. 2011). Oocyte development and maturity classification of herring based on microscopic characteristics has been documented in other areas (van Damme et al. 2009, McPherson et al. 2011, Kennedy et al., 2011, Bucholtz et al. 2013), but has yet to be applied in the NW Atlantic. In the Gulf of Maine and Georges Bank, Atlantic herring are considered to be fall spawners, with spring spawning reported but not quantified or considered in assessments. An understanding of oocyte development is necessary to determine skipped spawning, especially for 'resting' type (Rideout and Tomkiewicz 2011), which is further complicated by the possibility of spring or fall spawning in the region.

Gonad histology is considered the most accurate method for determining maturity. In addition to basic maturity information, histological methods can establish spawning seasonality. Preliminary work has suggested that not all herring spawn in the fall, and so may be either skip spawners or a spring spawning contingent. The occurrence of skip or spring spawning would suggest a violation of the current assumption of all fall spawners in the assessment, and could lead to biased estimates of SSB and reference points. Histological analysis of herring ovaries and the size frequency distributions of developing oocytes indicate whether an individual has spawned recently or is preparing to spawn in the near or more distant future (5-6 months). This information, along with prior studies of oocyte development for the species is used to identify whether apparent 'non-participatory' fish collected in fall are indeed skip spawners or if they spawn in a different season (spring).

In this study we apply histological (microscopic) methods to document oocyte development through the year for both spring and fall spawning herring. Using oocyte stages and other histological characters, we develop criteria to assign maturity stages, spawning seasonality, skipped spawning, and assess the accuracy of macroscopic maturity determinations from both commercial port samples and the fishery-independent surveys in Gulf of Maine and Georges Bank herring. Since the stock assessment of Atlantic herring assumes all mature fish spawn in fall, we then evaluate the implications of observed reproductive diversity on stock-recruit models and the stock assessment.

Methods

Oocyte development and histology-based maturity classification

We obtained gonad samples of Atlantic herring from multiple sources operating at different times of the year. Samples obtained from NEFSC spring and autumn bottom trawl surveys (SBTS and ABTS, respectively) were processed at sea; maturity classified macroscopically following Burnett et al. 1989 (Table 1), fish weight and gonad weight (+/- 0.1 g). Samples obtained from the NEFSC Cooperative Research Program (Study Fleet) were held on ice and transported to the laboratory where they were processed; fish weight and gonad weight +/- 0.001g, otoliths removed for ageing. Gonad samples were also obtained from the Maine DNR sampling of the commercial catch. The Maine DNR samples were usually frozen (but were fresh in some cases), and processed in the laboratory; maturity classified macroscopically using a different scheme than NEFSC samples (Table 1), fish weight and gonad weight +/- 1 g. In all cases, after weighing the gonad, a small portion was preserved in 10% buffered formalin for histology. Preserved tissue samples were processed following standard protocols; dehydrated in ETOH, embedded in paraffin, thin sectioned and stained with Mallory's trichrome. Histology was viewed with a digital microscope (Nikon Coolscope II) and oocyte were staged following (Brown-Peterson et al., 2011). Additional microscopic characters were recorded; the thickness of the gonad wall, presence and stage of post ovulatory follicles, presence and stage of atresia (Figure 3). Diameters of oocytes (~ 60-80 per fish) sectioned through the nucleus were measured using image analysis (ImageJ) from non-overlapping images from histological sections of representative individuals from each month available.

The histological characters and oocyte diameters (Figure 4) from all months sampled were used to develop classification algorithm to assign maturity stages and spawning seasonality (Table 2). The histology-based classifications were compared to macroscopic assessments at-sea for ABTS 2014, ABTS 2015, SBTS 2016, and

2015 Q3 commercial samples. The sampling protocol for histological samples collected on NEFSC surveys for verification was as follows; at each station, after determining maturation stage of individuals sampled for age and growth, one fish of each macroscopic maturity stage was selected for histology sampling until a total of 100 was reached. This protocol ensured histological samples covered all stages encountered, and came from a wide region. Similarly, 100 random histological samples were requested from the third quarter of 2015 sampled by the Maine DMR staff processing the commercial samples with the following objectives; to cover as broad a range in dates, areas, and macroscopic stages as possible, with a preference for fresh (not frozen) samples which produce higher quality histology.

QA/QC of macroscopic maturity estimation.

The accuracy of macroscopic maturity staging for Atlantic herring was assessed for NEFSC surveys (Fall 2014, 2015; spring 2016) and ME DNR (third quarter 2015). The NEFSC Northeast Cooperative Research Program (Study Fleet) collections were used to inform histological characters (oocyte stages, POF persistence) and oocyte size distributions in months not sampled from the other sources and were used solely in the development of classification algorithms (no comparison of macroscopic vs. histologic determinations were performed).

Estimation of spring and/or skipped spawning

Based on the histological characteristics and month of collection, we were able to assign spawning seasonality for mature and maturing fish. For immature fish that have not yet initiated secondary development of oocytes, it was not possible to identify spawning seasonality. The identification of skip spawners is limited to discrete portions of the year, and in the case of Atlantic herring this is further complicated by potential for spring or fall spawners. We established criteria that would indicate skip spawners based on the month of collection, oocyte stages, POFs and atresia (Table 2).

Stock-recruit modeling

Estimates of biomass and recruitment (age-1 abundance) from the 2015 stock assessment (Deroba 2015) were used to evaluate the effect of spring spawning or skipped spawning on estimates of Beverton-Holt stock-recruit parameters. Beverton-Holt models were fit using three different definitions of spawning stock biomass (*SSB*) that corresponded to spring spawning, skipped spawning, or 100% fall spawning. For spring spawning, some fraction of the stock was assumed to have spawned in May, while the remainder of surviving fish spawned in October. The annual SSB related to recruitment in the following year was the sum of the spring and fall spawners:

$$SSB_{y,spr} = SSB_{y,Jan1} * p_s * \exp\left({}^{5}/_{12} * Z_y\right) + SSB_{y,Jan1} * p_f * \exp\left({}^{10}/_{12} * Z_y\right);$$

where $SSB_{y,spr}$ was spawning stock biomass in year *y* and *spr* denoted that the calculation included spring spawners, $SSB_{y,Jan1}$ was spawning stock biomass on January 1, p_s was the fraction of the stock that spawned in spring, $p_f = 1 - p_s$ was the fraction of the stock that spawned in fall, and Z was total instantaneous mortality:

$$Z_y = M + F_y;$$

where M was instantaneous natural mortality, and F was fully-selected instantaneous fishing mortality estimated for each year in the 2015 stock assessment (Deroba 2015). Skipped spawning (SSB_{skip}) was approximated by not including the spring spawners in the calculation of the annual SSB:

$$SSB_{y,skip} = SSB_{y,Jan1} * p_f * \exp\left({}^{10}/_{12} * Z_y\right).$$

In this context, p_s equates to the proportion of the stock that skips spawning. Fall spawning (SSB_{fall}), as assumed in recent stock assessments, was calculated as in skipped spawning except with $p_f = 1$:

$$SSB_{y,fall} = SSB_{y,Jan1} * \exp\left({}^{10}/_{12} * Z_y\right).$$

These methods for calculating SSB assumed the same maturity ogive applied in both seasons and ignored within year growth. A Beverton-Holt stock-recruit model was fit using each of the definitions for SSB:

$$R_{y+1} = \frac{\alpha * SSB_{y,x}}{\beta + SSB_{y,x}};$$

where x denoted one of the three definitions of SSB (*spr, skip, or fall*), R was estimated recruitment from 1966-2013 from the 2015 stock assessment (Deroba 2015), and α and β were parameters. This method assumed that the expected recruitment at a given level of SSB was the same in both seasons (i.e., spring spawners were the "same" as fall spawners).

Models were fit using a range of M and p_s (p_f) values (Table 3). The mean stock-recruit curve from each fit was plotted, and plots were qualitatively examined for differences.

Sensitivity of the stock assessment

The stock assessment was evaluated for sensitivity to spring spawning and skipped spawning. With the exception of modifying the SSB calculation, all inputs and settings were identical to the 2015 stock assessment (Deroba 2015). The calculation of SSB in the stock assessment was modified as in the case of spring spawning $\left(SSB_{y,spr}\right)$ and skipped spawning $\left(SSB_{y,skip}\right)$ in the stock-recruit modeling methods, but the distinction in this case was that the estimation of the stock-recruit relationship was done internal to the assessment model and estimated with all the other associated parameters. Models were fit using a range of p_s $\left(p_f\right)$ values (Table 3). Time series plots of estimates of SSB, recruitment, and fully-selected fishing mortality were qualitatively examined for differences between the assessment modified for spring or skipped spawning and the 2015 assessment. Values of estimated steepness and unexploited SSB were also compared.

Results and Discussion

Using microscopic verification we found the macroscopic method to be reasonably accurate for Atlantic herring (direct agreement 60-87%), however errors in determination of sex (2-7%) and maturity (0-13%) were evident in all surveys (Tables 4-8). Errors in maturity were highest in the spring survey period. Subtle disagreements (not affecting maturity) between the histologic and macroscopic methods were also evident in all surveys. During spring, many fish classified as resting at sea were undergoing early development which was only visible via histology.

Misclassification of sex occurred in all surveys (summarized in Table 8). Most of these misclassifications occurred for immature fish, where it is more difficult to differentiate sex macroscopically. Additionally, for most individuals during the spring survey period, the gonads are very small, making it more difficult to distinguish males from females macroscopically. This likely led to the higher rates of incorrect maturity in the spring (Table 6), and supports the continued estimation of maturity from samples obtained closest to the main spawning season in fall. The results from the spring also indicate histology was able to identify early developing fish before this was evident macroscopically (ED fish classified as resting at sea). This is not surprising since the characters that define early developing are not readily visible with the naked eye. Several late developing and one spent fish were collected in the spring, a clear indication of spring-early summer spawning. Interestingly, the spent fish was classified as ripe and running at sea. This individual contained advanced and mature oocytes that histology indicated were 'residual' (i.e. left over). During winter and spring, the difference in ovary condition between spring and fall spawners is obvious (Figures 5 and 6). Therefore, the estimation of spring spawning from developing and spawning active herring in the spring is considered reliable. Nonetheless, spring spawning was relatively rare (2.5-13%, Table 8). A summary herring maturity from the SBTS time series across regions (Figure 7) indicates a latitudinal trend in the proportion of spring spawning herring (pre-spawning and spawning) encountered. Proportions of spring spawning increased with latitude; 0-10% in the Middle Atlantic Bight, 5-20% on Georges Bank, 10-40% in the Gulf of Maine, and 10-80% on the Scotian Shelf. Although rates of spring spawning were higher on the Scotian Shelf, fewer fish were sampled in that region.

Because of the sampling scheme used, wherein samples were requested across stages, and not in proportion to the stages encountered, the error rates reported here do not depict actual error rates. To arrive at overall errors in macroscopic classifications in the surveys, one would need to apply stage specific error rates to all fish examined in the surveys. In fall most herring are developing, which is sometimes confused with resting (regenerating), but was never confused with immature. Therefore, proportionally more developing fish in fall would dilute the effect of errors in the rarer stages (i.e. resting), reducing the overall error rate. In a similar vein, although error rates in spring were higher, the late developing and spawning active females were accurately identified, confirming spring spawning in the region. The commercial Q3 collection was the most precise (Table 8), possibly due to being performed by a single experienced technician. In contrast, the NEFSC survey data is collected by multiple individuals per survey, with varying backgrounds and experience levels with respect to herring maturity. Annual training workshops on fish maturity are held at the NEFSC to address this potential source of error.

The spatial distribution of samples from the NEFSC surveys differed from the Q3 commercial samples analyzed by the Maine DNR which were predominately from inshore stat areas (512, 513, 514; Figure 8). Most immature fish remain inshore in fall (i.e. do not undergo spawning migrations offshore), so samples inshore will have more immature fish (overall and at a given age). This is illustrated in maps showing the spatial distribution in age 2 and 3 fish in 2014 and 2015 (Figs 9 and 10). The proportion mature at age 2 and 3 varies in the time series (Deroba 2015), but it appears immature individuals are found closer to shore in fall. In spring Atlantic herring are more widely distributed, including immature individuals (Figure 11). The spatial difference in maturity likely contributes to observed differences in estimated proportions mature at age from survey and commercial data sources in fall. When analysis of survey data is constrained to inshore regions, the differences in maturity decrease (Figure 12).

Stock-recruit modeling

The fits of stock-recruit models with spring spawning and fall spawning were generally similar, but skipped spawning produced higher recruitment at a given level of SSB (Figure 13). Natural mortality had a negligible effect on the fits, especially relative to the fraction of the stock that skipped spawning (p_s in the context of skipped spawning). At low levels of skipped spawning, differences among all fits were generally similar, but the skipped spawning stock-recruit relationships became more distinct at higher levels of skipped spawning (Figure 13).

Sensitivity of the stock assessment

Results for spring spawning were insensitive to the value of p_s, and so only results for $p_s = 0.3$ were reported here. Time series plots were generally similar between the assessment model with spring spawning and the 2015 assessment (Figure 14). Estimates of steepness and unfished SSB were also similar (Table 9). In the case of skipped spawning, differences in the time series plots with the 2015 assessment were only evident for SSB, and SSB was less than the 2015 assessment (Figure 14). The degree of difference increased with the value of p_s, but only results with $p_s = 0.3$ were reported for simplicity (Figure 14). Estimated steepness with skipped spawning was similar to the 2015 assessment, but unfished SSB was less (Table 9). Skipped spawning seems to scale SSB and related reference points, with little other consequences.

This analysis could be improved by accounting for in-season growth and different maturity ogives between seasons. The methods assumed, however, that the maturity and weight-at-age matrices from the fall spawning season also applied in the spring. Spring spawners are less likely to be mature-at-age and have smaller weights-at-age than fall spawners. Consequently, this analysis falsely inflated the contribution that spring spawners would have, and so accounting for these seasonal differences would likely only result in reducing any differences already observed among model fits.

The differences in the stock-recruit modeling between assuming spring spawning or skipped spawning serve to demonstrate the consequence of falsely concluding one mechanism or the other. Results suggest, however, that making such a false conclusion would be of little consequence until the fraction of spring or skipped spawners was relatively high, at which point other indications of spring or skipped spawning would likely become evident in survey or commercial catch samples.

At the levels of possible spring or skipped spawning reported here, the effect on the stock assessment can likely be ignored. Levels of measurement error, process error, and other structural uncertainty also likely far exceed the uncertainty induced by spring or skipped spawning suggested by this analysis, which also supports the conclusion that the effect of spring or skipped spawning can be ignored.

Conclusions

Histological analysis of herring gonads from multiple sources, and inclusion of reproductive diversity in the stock assessment indicated the following:

- error rates of the macroscopic method to determine maturity were low
- there were not any systematic biases between NEFSC and ME DNR maturity methods
- spring spawning was confirmed at low levels
- skipped spawning was not observed
- the spatial distribution of immature herring differs from that of mature herring in fall
    - differences in maturity estimated from NEFSC survey and Q3 commercial are likely due to spatial heterogeneity of the population with respect to maturity
- skipped spawning scales SSB and related reference points, with little other consequences
    - incorporating observed rates of spring and/or skip spawning had little effect on the stock assessment

Acknowledgments

We thank the following individuals and groups for assisting with the collection and processing of samples: Emilee Tholke, David McElroy, Study fleet staff, Eric Robillard, ME DNR (Mathew Cieri, Lisa Pinkham), agers (Blanche Jackson, Katey Rogers), and NEFSC Ecosystems Survey Branch.

References

Brown-Peterson, N.J., Wyanski, D.M., Saborido-Rey, F., Macewicz, B.J., and Lowerre-Barbieri, S.K. (2011). A Standardized Terminology for Describing Reproductive Development in Fishes. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science **3**: 52-70.

Bucholtz, R. H., J. Tomkiewicz, J. R. Nyengaard and J. B. Andersen (2013). Oogenesis, fecundity and condition of Baltic herring (*Clupea harengus* L.): A stereological study. Fisheries Research **145**: 100-113.

Burnett, J., L. O'Brien, R. K. Mayo, J. A. Darde, and M. Bohan. (1989). Finfish Maturity Sampling and Classification Schemes Used During Northeast Fisheries Center Bottom Trawl Surveys, 1963-1989. NOAA Technical Memorandum: NMFS-F/NEC-76.

Deroba, J.J. (2015). Atlantic herring operational assessment report 2015. US Dept. Commer, NEFSC Ref Doc. 15-16; 30 pages.

Deroba, J.J. 2015. Atlantic herring operational assessment report 2015. US Department of Commerce, Northeast Fisheries Science Center Reference Document 15-16; 30p.

NEFSC, Northeast Fisheries Science Center. (2012). 54th Northeast Regional Stock Assessment Workshop (54th SAW) Assessment Report. US Department of Commerce, Northeast Fisheries Science Center Reference Document 12-18; 600p.

Engelhard, G. H. and M. Heino (2005). Scale analysis suggests frequent skipping of the second reproductive season in Atlantic herring. Biology Letters **1**(2): 172-175.

Kennedy, J., J. E. Skjaeraasen, R. D. M. Nash, A. Slotte, A. J. Geffen and O. S. Kjesbu (2011). Evaluation of the frequency of skipped spawning in Norwegian spring-spawning herring. Journal of Sea Research **65**(3): 327-332.

McPherson, L. R., K. Ganias and C. T. Marshall (2011). Inaccuracies in routinely collected Atlantic herring (*Clupea harengus*) maturity data and correction using a gonadosomatic index model. Journal of the Marine Biological Association of the United Kingdom **91**(7): 1477-1487.

Rideout, R. M., G. A. Rose, and M. P. M. Burton. (2005). Skipped spawning in female iteroparous fishes. Fish and Fisheries **6**: 50-72.

Rideout, R. M. and J. Tomkiewicz (2011). Skipped Spawning in Fishes: More Common than You Might Think. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science **3**: 176-189.

Skjaeraasen, J. E., J. Kennedy, A. Thorsen, M. Fonn, B. N. Strand, I. Mayer and O. S. Kjesbu (2009). Mechanisms regulating oocyte recruitment and skipped spawning in Northeast Arctic cod (*Gadus morhua*). Canadian Journal of Fisheries and Aquatic Sciences **66**(9): 1582-1596.

Skjaeraasen, J. E., R. D. M. Nash, K. Korsbrekke, M. Fonn, T. Nilsen, J. Kennedy, K. H. Nedreaas, A. Thorsen, P. R. Witthames, A. J. Geffen, H. Hoie and O. S. Kjesbu (2012). Frequent skipped spawning in the world's largest cod population. Proceedings of the National Academy of Sciences of the United States of America **109**(23): 8995-8999.

van Damme, C. J. G., M. Dickey-Collas, A. D. Rijnsdorp and O. S. Kjesbu (2009). Fecundity, atresia, and spawning strategies of Atlantic herring (*Clupea harengus*). Canadian Journal of Fisheries and Aquatic Sciences **66**: 2130-2141.

Table 1. Macroscopic and histological maturity classification schemes used in NEFSC surveys (Table 11 in Burnett et al. 1989) and ME DNR sampling of commercial catch (Table 3B in Burnett et al. 1989). Corresponding histological classes are listed (following Brown-Peterson et al. 2011, with potential skip spawning following Rideout et al. 2005); in some cases multiple stages in one scheme are represented by a single stage in another scheme.

Macroscopic Classes NEFSC	**Macroscopic Classes ME DNR**	**Histological Classes**
Immature	I (Immature) II (Immature will spawn next season)	Immature Immature First Maturing
Developing	III (Ripening, Early stage) IV (Ripening mid stage)	Early Developing Late Developing Spawning Capable Skip Spawning (Reabsorbing)
Ripe Running Ripe	V (Ripe) VI (Spawning)	Spawning Active
Spent	VII (Spent)	Regressing
Resting	VIII (Resting)	Regenerating Skip Spawning (Resting)

Table 2. Microscopic characteristics for each histological maturity stage.

Histological Classes	Characteristics
Immature	Ovaries small with thin ovarian wall and little space between oocytes; only oogonia and PG oocytes present
Immature First Maturing	PG, CA, oocytes present with thin ovarian wall.
Early Developing (repeat)	Ovaries with PG, CA; thick ovarian wall and/or late stage POFs indicating prior spawning.
Late Developing	Enlarging ovaries with Vtg1, Vtg2 oocytes.
Spawning Capable	Large ovaries with Vtg3 oocytes present. Atresia of vitellogenic oocytes may be present. Early stages of OM can be present.
Skip Spawning (Reabsorbing)	Mass atresia of vitellogentic oocytes.
Spawning Active	Oocytes undergoing late GVM, GVBD, hydration, or ovulation.
Regressing	Flaccid ovaries with thick ovarian wall; atresia and recent POFs present. Most advanced oocyte stage is primary growth, with some residual secondary or tertiary growth oocytes possible.
Regenerating	Small ovaries with thick ovarian wall. Late stage atresia or POFs may be present. Only oogonia and PG oocytes present.
Skip Spawning (Resting)	Small ovaries with thick ovarian wall. Only oogonia or PG oocytes present. No indication of participation in proximal spawning season; no secondary or tertiary growth oocytes, recent POFs, or atresia.

Table 3. Range of natural mortality (M) and proportion of spring (p_s) and fall (p_f) spawners evaluated.

M	p_s	p_f
0.20	0.10	0.90
0.35	0.20	0.80
0.50	0.30	0.70

Table 4. QA/QC results for the 2014 fall bottom trawl survey (2015ABTS). In fall IFM fish are not expected to spawn until the following calendar year, so they should not be included in SSB (i.e. not mature). Green cells indicate direct agreement, red cells indicate incorrect maturity.

	Immature	Developing	Ripe	Ripe & Running	Spent	Resting
Immature	8	0	0	0	0	3
Immature First Developing	0	0	0	0	0	0
Early Developing	0	1	0	0	0	2
Late Developing	0	7	6	0	0	0
Spawning Capable	0	1	2	0	0	0
Spawning Active	0	0	0	0	0	0
Regressing	0	1	2	0	0	3
Regenerating	0	4	1	0	0	51
Skip Spawner	0	0	0	0	0	0

Table 5. QA/QC results for the 2015 fall bottom trawl survey (2015ABTS). In fall IFM fish are not expected to spawn until the following calendar year, so they should not be included in SSB (i.e. not mature). Green cells indicate direct agreement, red cells indicate incorrect maturity.

	Immature	Developing	Ripe	Ripe & Running	Spent	Resting
Immature	7	0	0	0	0	0
Immature First Developing	1	0	0	0	0	1
Early Developing	0	0	0	0	0	2
Late Developing	0	14	2	0	0	0
Spawning Capable	0	1	2	0	0	0
Spawning Active	0	0	0	0	0	0
Regressing	0	2	1	0	0	3
Regenerating	3	1	0	1	1	46
Skip Spawner	0	0	0	0	0	0

Table 6. QA/QC results for the 2015 Q3 commercial samples (ME2015 Q3). In fall IFM fish are not expected to spawn until the following calendar year, so they should not be included in SSB (i.e. not mature). Green cells indicate direct agreement, red cells indicate incorrect maturity.

	I	II	III	IV	V	VI	VII	VIII
Immature	2	1	0	0	0	0	0	0
Immature First Developing	0	2	0	0	0	0	0	0
Early Developing	0	2	10	0	0	0	0	0
Late Developing	0	0	34	10	0	1	0	0
Spawning Capable	0	0	1	2	10	7	0	0
Spawning Active	0	0	0	0	0	5	0	0
Regressing	0	0	0	0	0	0	4	0
Regenerating	0	0	0	0	0	0	0	7
Skip Spawner	0	0	0	0	0	0	0	0

Table 7. QA/QC results for the 2016 Spring bottom trawl survey (2016SBTS). In the spring, IFM fish would be expected to spawn in the calendar year (that fall) so they should be included in SSB (i.e. not immature). Green cells indicate direct agreement, red cells indicate incorrect maturity.

	Immature	Developing	Ripe	Ripe & Running	Spent	Resting
Immature	1	0	0	0	0	1
Immature First Developing	5	0	0	0	0	0
Early Developing	4	4	0	0	0	24
Late Developing	0	7	2	0	0	0
Spawning Capable	0	0	0	0	0	0
Spawning Active	0	0	0	0	0	0
Regressing	0	0	0	1	0	0
Regenerating	3	3	0	0	0	40
Skip Spawner	0	0	0	0	0	0

Table 8. Summary of sex and spawning group determinations for the four data sources. For Males the percentages listed in parentheses are the percentage males incorrectly classified as females macroscopically. For females, the percentages listed in parentheses are the percentage of mature females in that spawn group. The direct agreement is the sum of the diagonal green cells for survey (Tables 3-6), and the incorrect maturity is the sum of the red cells in each table. Percentages included for QA/QC are calculated for females only.

Sex	Spawn group	2014 (ABTS)	2015 (ABTS)	ME2015 (Q3)	2016 (SBTS)
Males (incorrect sex)		7 (7.0%)	4 (4.3%)	2 (2.0%)	5 (5.0%)
Females		93 (88.6%)	89 (95.7%)	98 (98.0%)	95 (95.0%)
Mature	Spring	3 (3.7%)	2 (2.5%)	12 (13.0%)	10 (11.4%)
	Fall	78 (96.3%)	77 (97.5%)	81 (87.0%)	78 (88.6%)
	Skip	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Immature	Unknown	11	9	5	7
QA/QC	Direct agreement	68 (73.1%)	69 (77.5%)	85 (86.7%)	57 (60%)
	Incorrect Maturity	3 (3.2%)	3 (3.4%)	0 (0%)	13 (13.6%)

Table 9.—Estimates of unfished spawning stock biomass and steepness from assessment with all fall spawning and 30% spring spawning.

	Fall Spawn Only (2015 Assessment)	With Spring Spawning	Skip Spawning
Unfished SSB	845176	885784	591623
Steepness	0.44	0.43	0.44

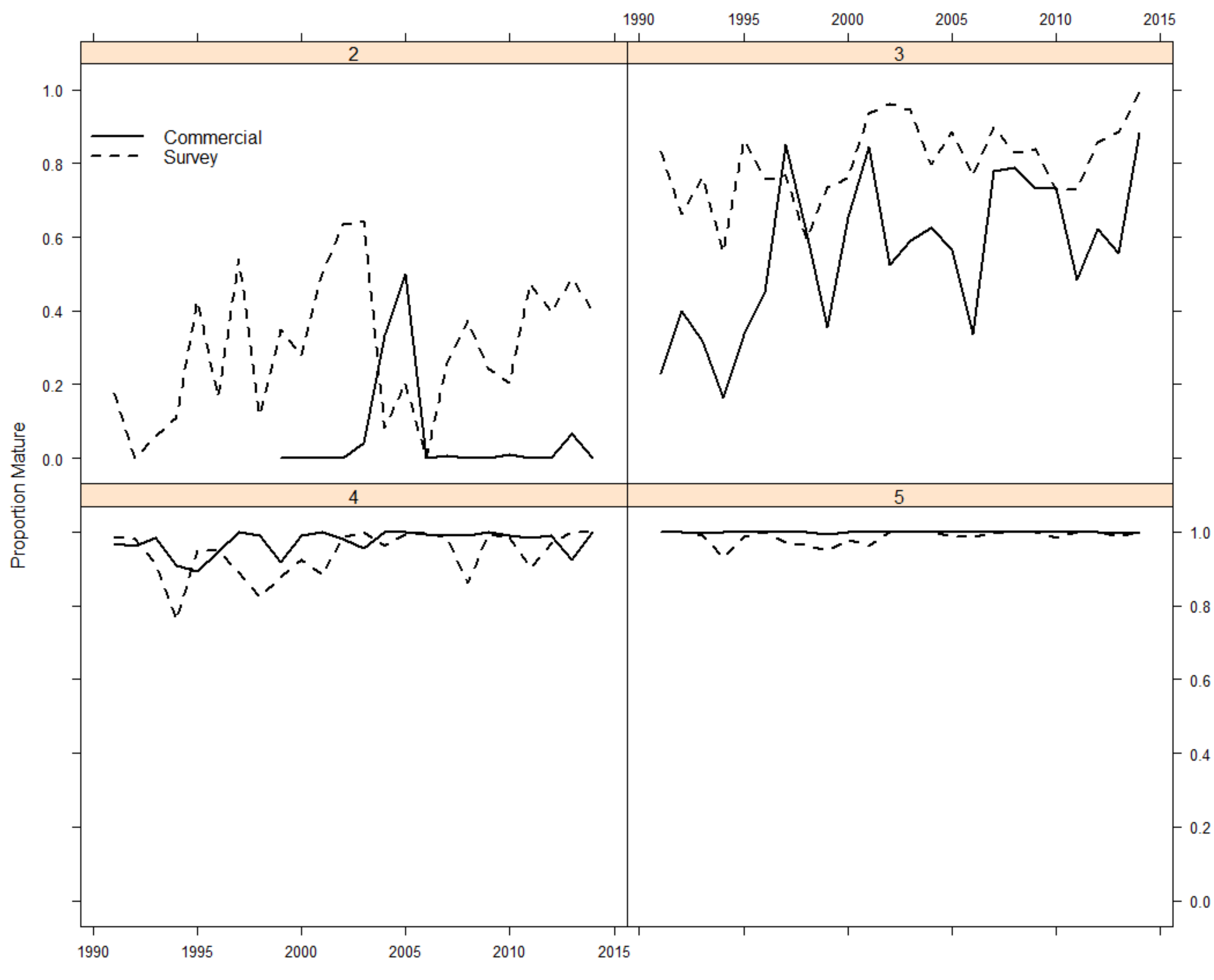


Figure 1. Proportion mature at age (age specified in the “strip” of each panel) from quarter three commercial fishery herring samples and the NMFS fall survey.

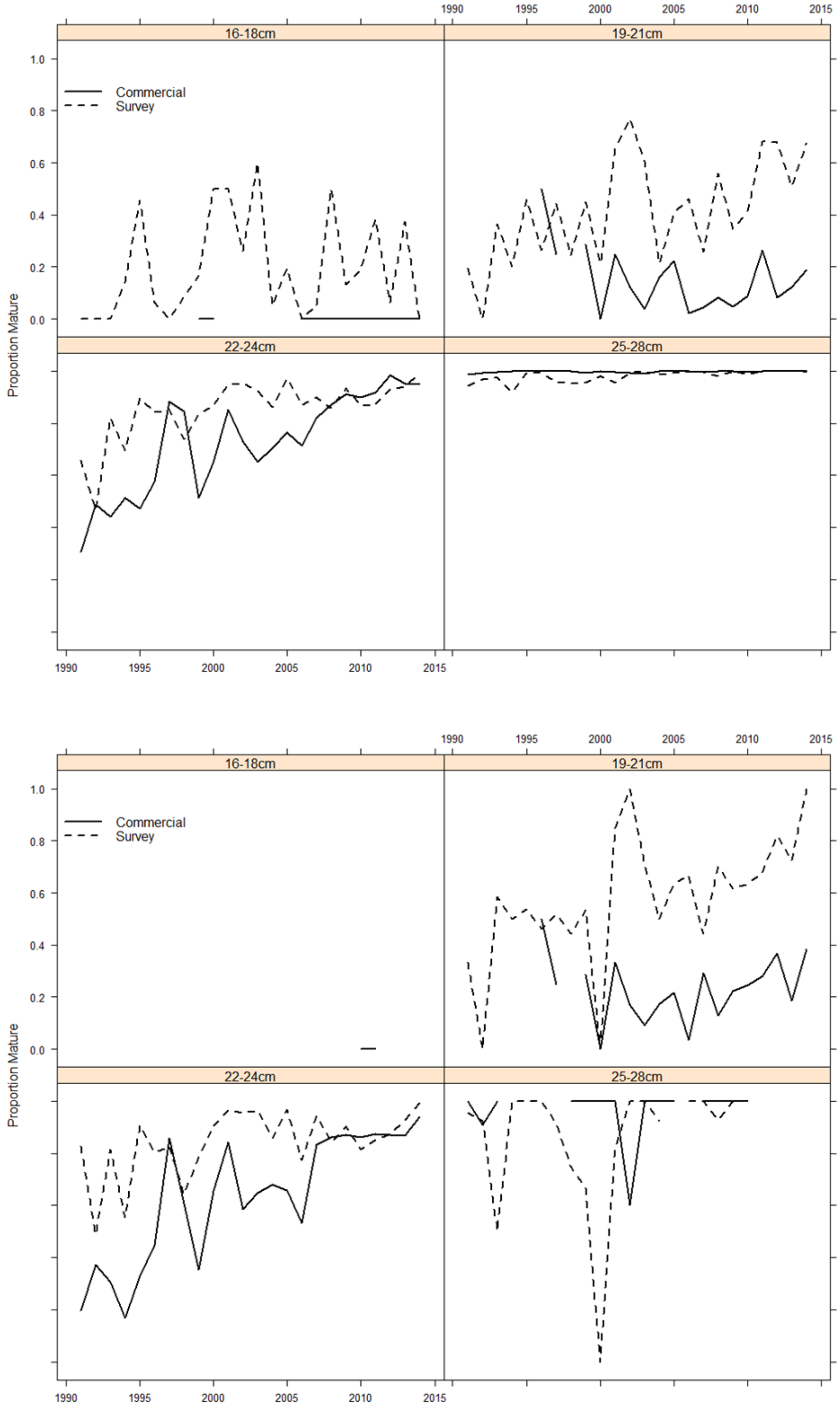


Figure 2. Proportion mature at length (length specified in the “strip” of each panel) for all ages (top) and age-3 (bottom) from quarter three commercial fishery herring samples and the NMFS fall survey.

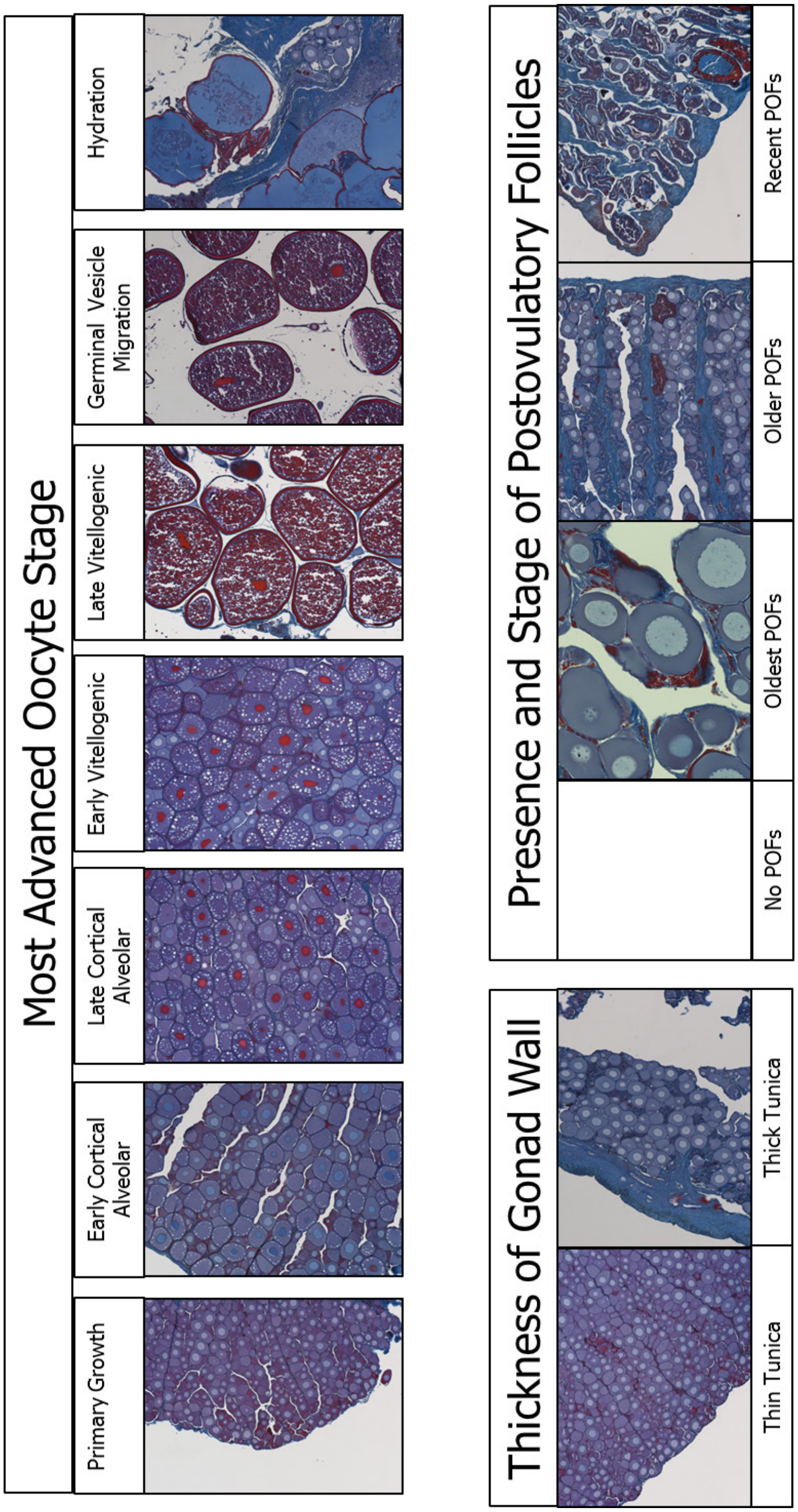


Figure 3. Histological criteria used to assess maturity of Atlantic herring. All images are at the same magnification, except for the 'Oldest POF' which is at a higher magnification.

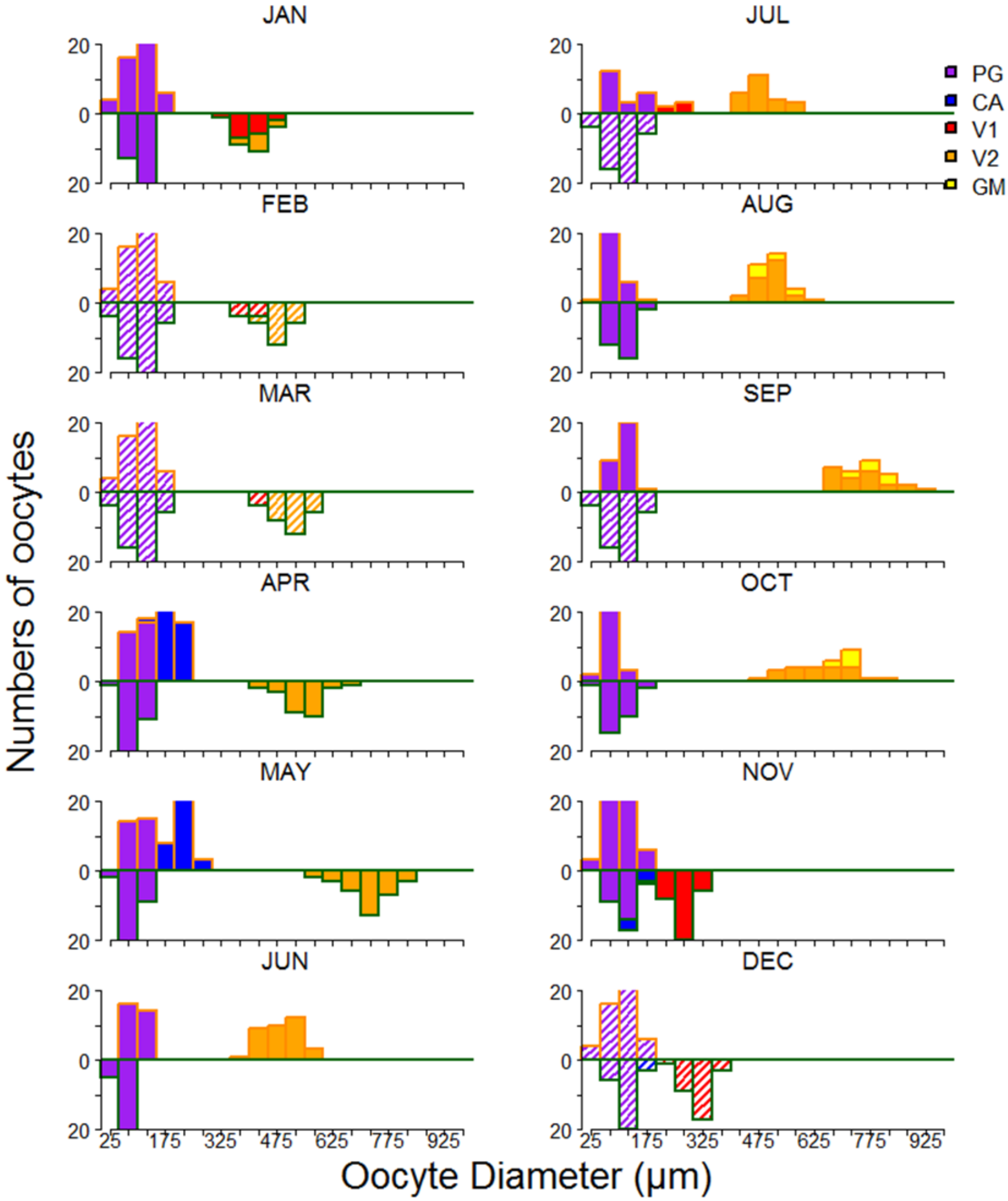


Figure 4. Monthly oocyte size distributions for representative Atlantic Herring (solid bars) or estimated from adjacent months and observed oocyte growth rates (shaded bars). Within each month, the distribution on top represents a fall spawner, and that on the bottom a spring spawner. PG, Primary Growth; CA, Cortical Alveolar; V1, Early Vitellogenic; V2, Late Vitellogenic; GM, Germinal Vesicle Migration.

Figure 5. Photographs of herring sampled January 22, 2015 from commercial catch (NEFSC Study Fleet). Top, resting female (Fall spawner); bottom, developing female (Spring spawner).

Figure 6. Herring photographed April 25, 2017 during the NEFSC spring bottom trawl survey. Top, resting female (Fall spawner); Bottom, developing female (Spring spawner).

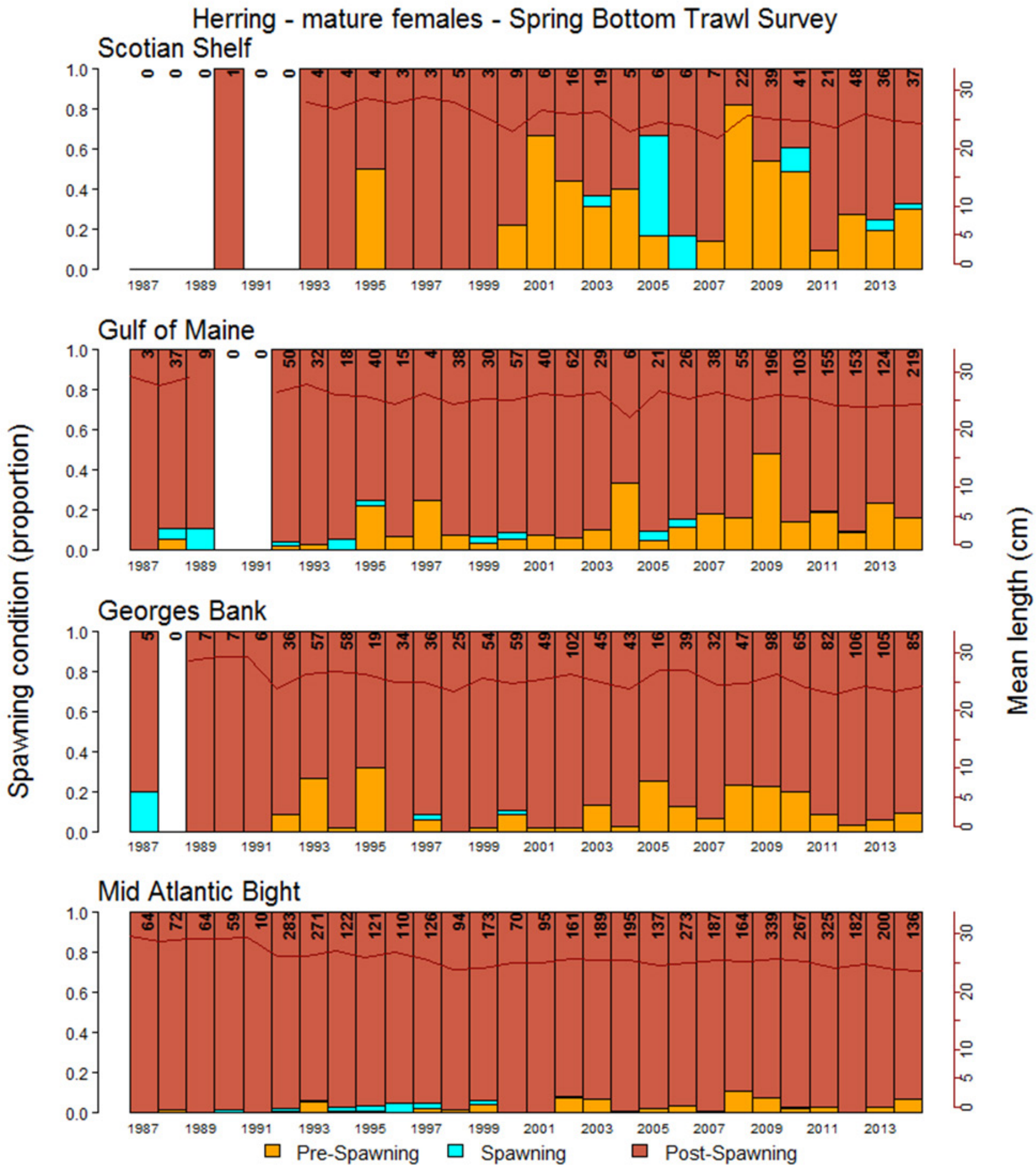
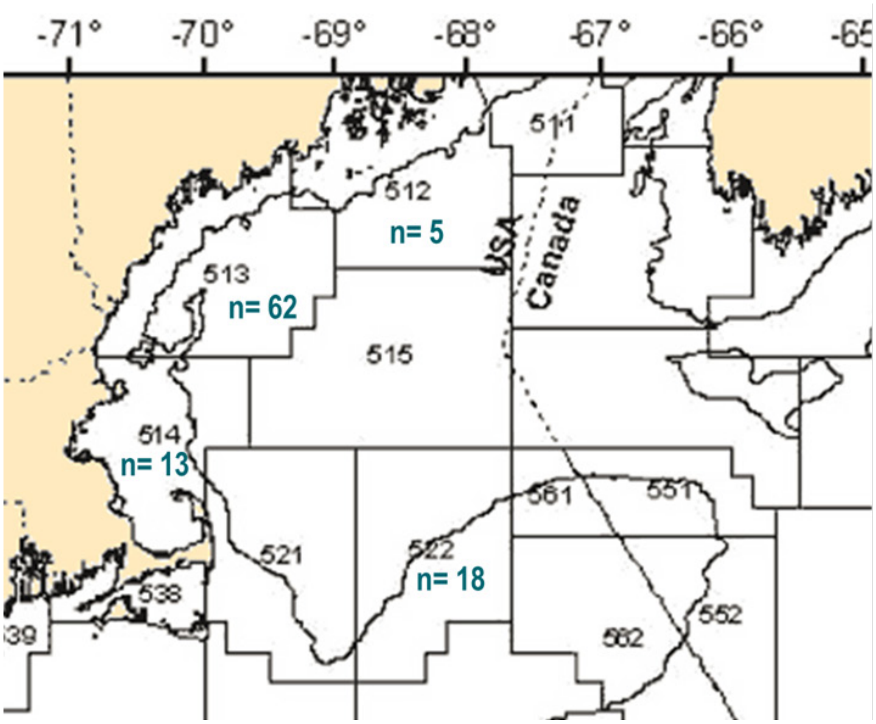


Figure 7. Time series of mature female macroscopic maturity collected on the NEFSC spring bottom trawl survey. For simplification, maturity classes are aggregated to illustrate spawning seasonality (Pre-Spawning = Developing, Spawning = Ripe and Ripe and Running, Post-Spawning = Spent and Resting). Pre-Spawning and Spawning groups represent spring spawning herring.



Stat Area	Sample size
512	5
513	62
514	13
522	18

Figure 8. Distribution and summary of female herring from Q3 commercial that were analyzed histologically.

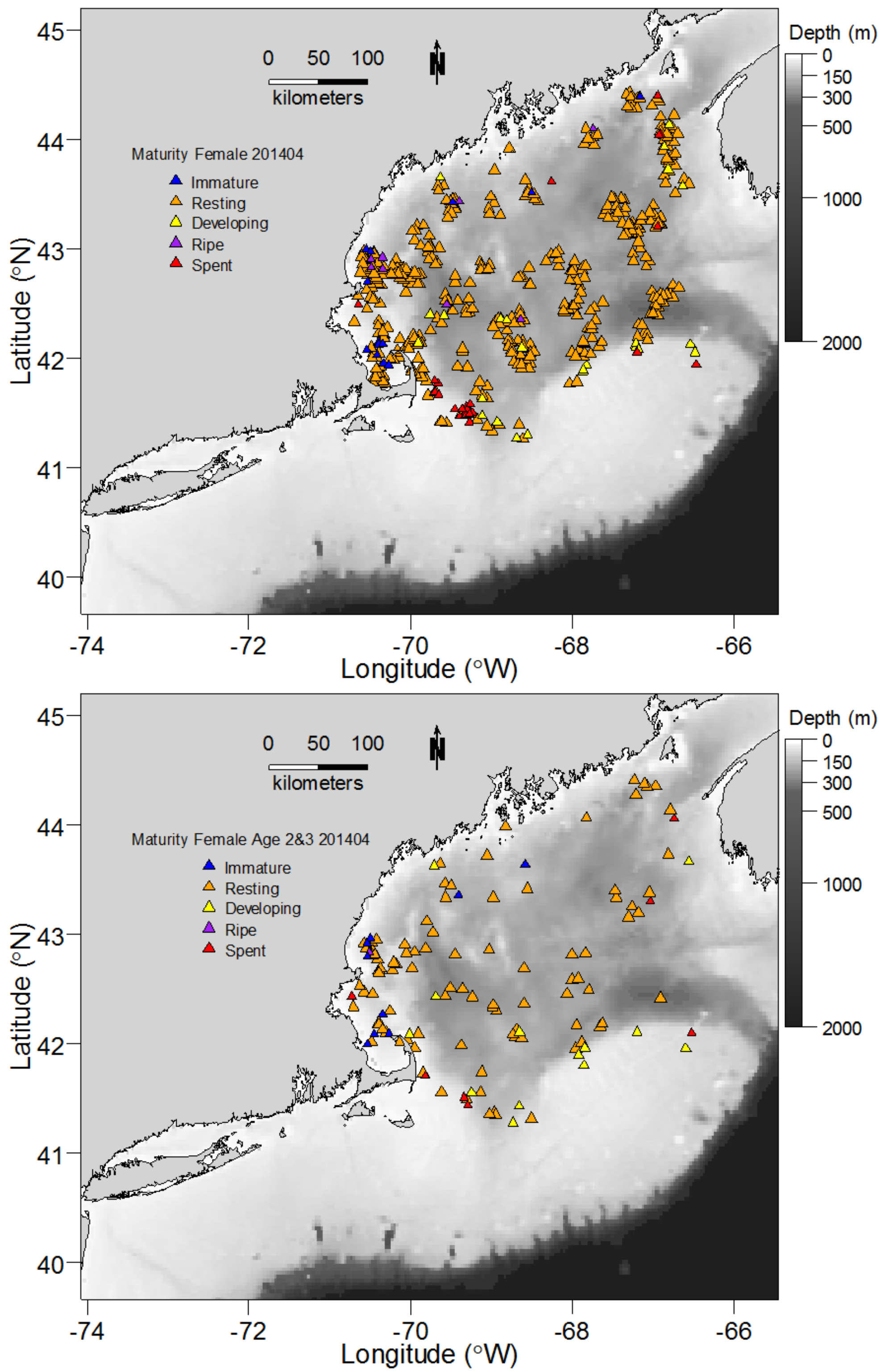


Figure 9. Distribution of all (top) and age 2 and 3 (bottom) females sampled for age, growth and maturity on the 2014 NEFSC fall bottom trawl survey. Points are jittered to reduce over-plotting.

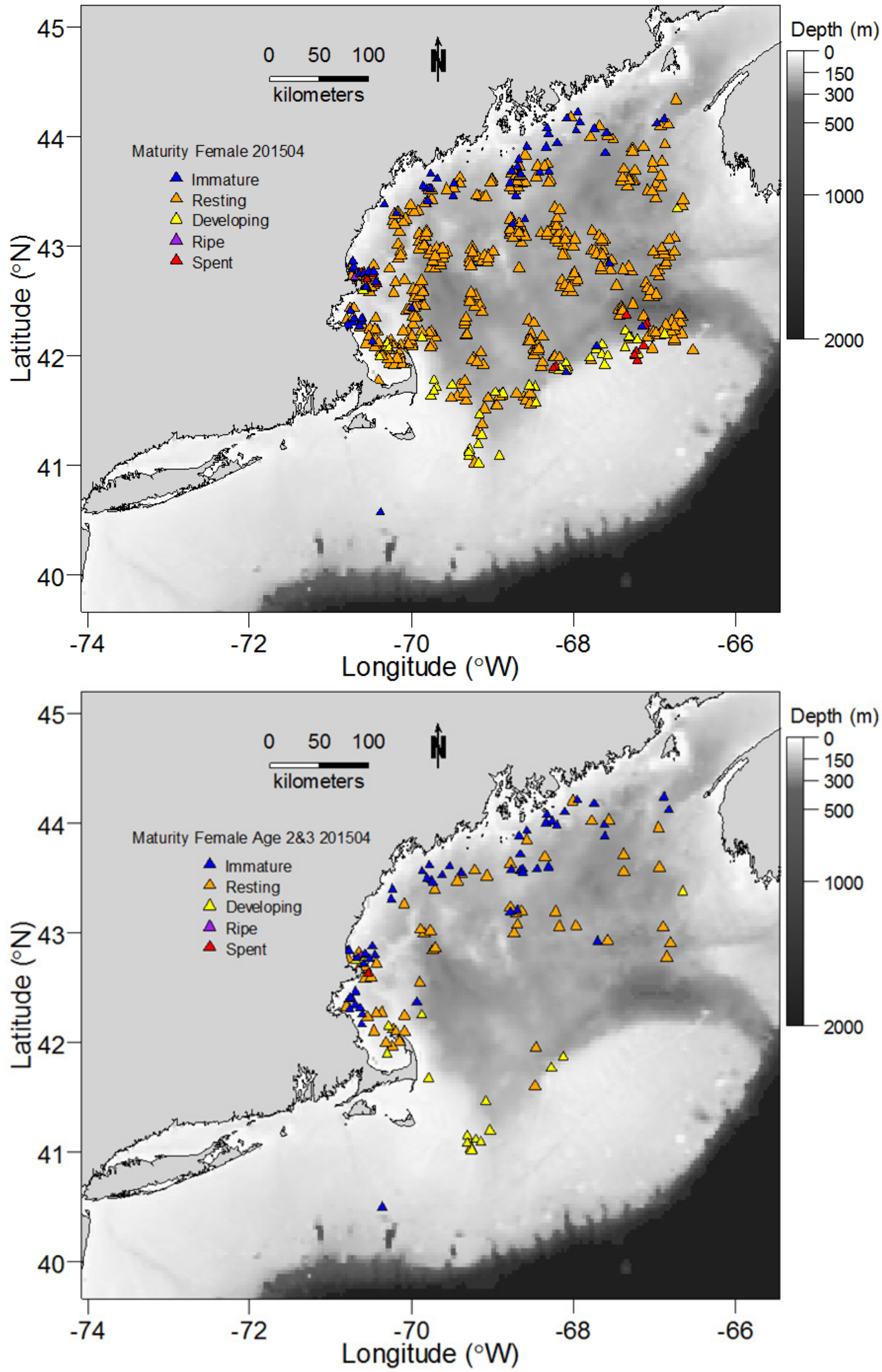


Figure 10. Distribution of all (top) and age 2 and 3 (bottom) females sampled for age, growth and maturity on the 2015 NEFSC fall bottom trawl survey. Points are jittered to reduce over-plotting.

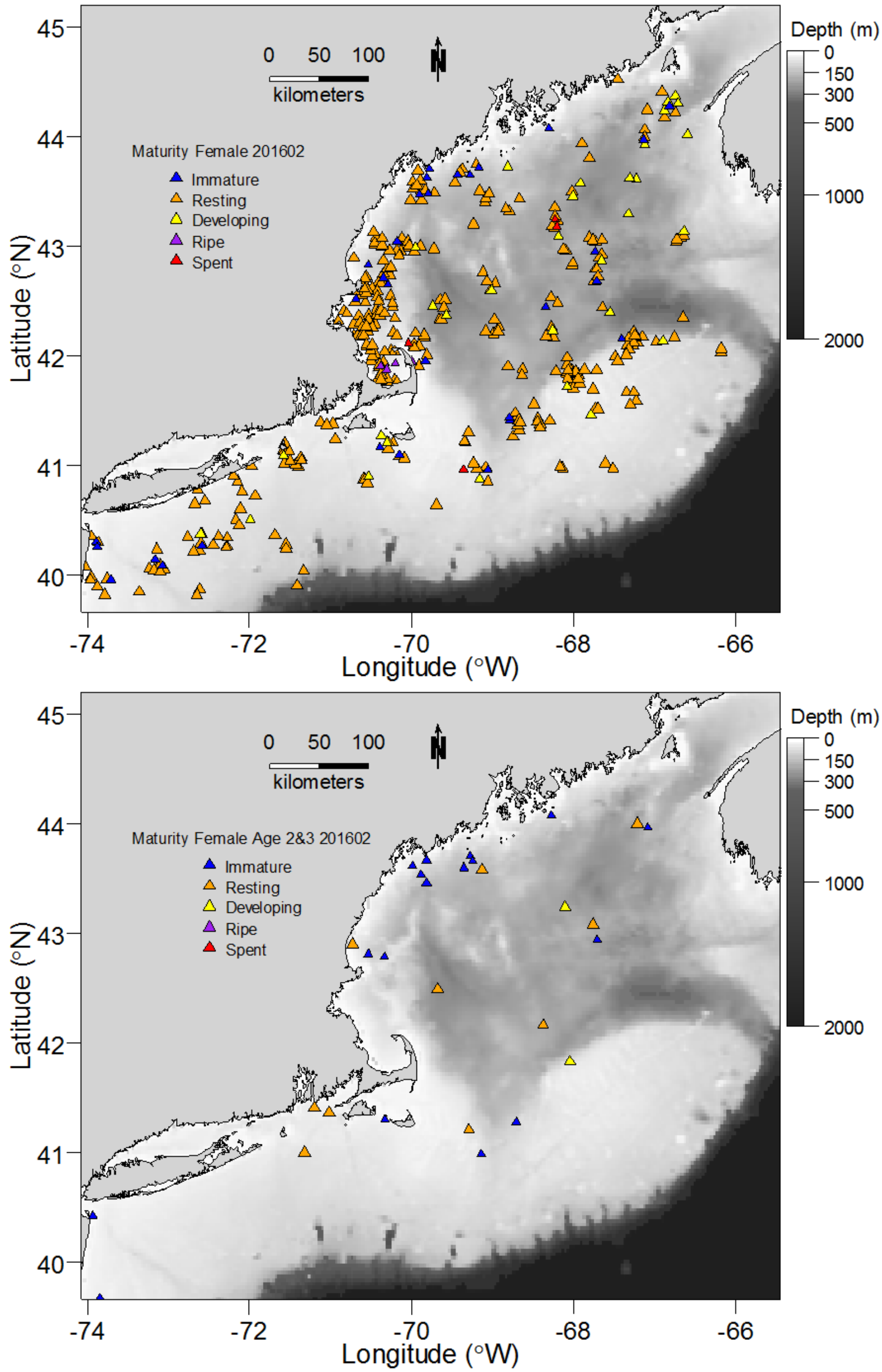


Figure 11. Distribution of all (top) and age 2 and 3 (bottom) females sampled for age, growth and maturity on the 2016 NEFSC spring bottom trawl survey. Points are jittered to reduce over-plotting.

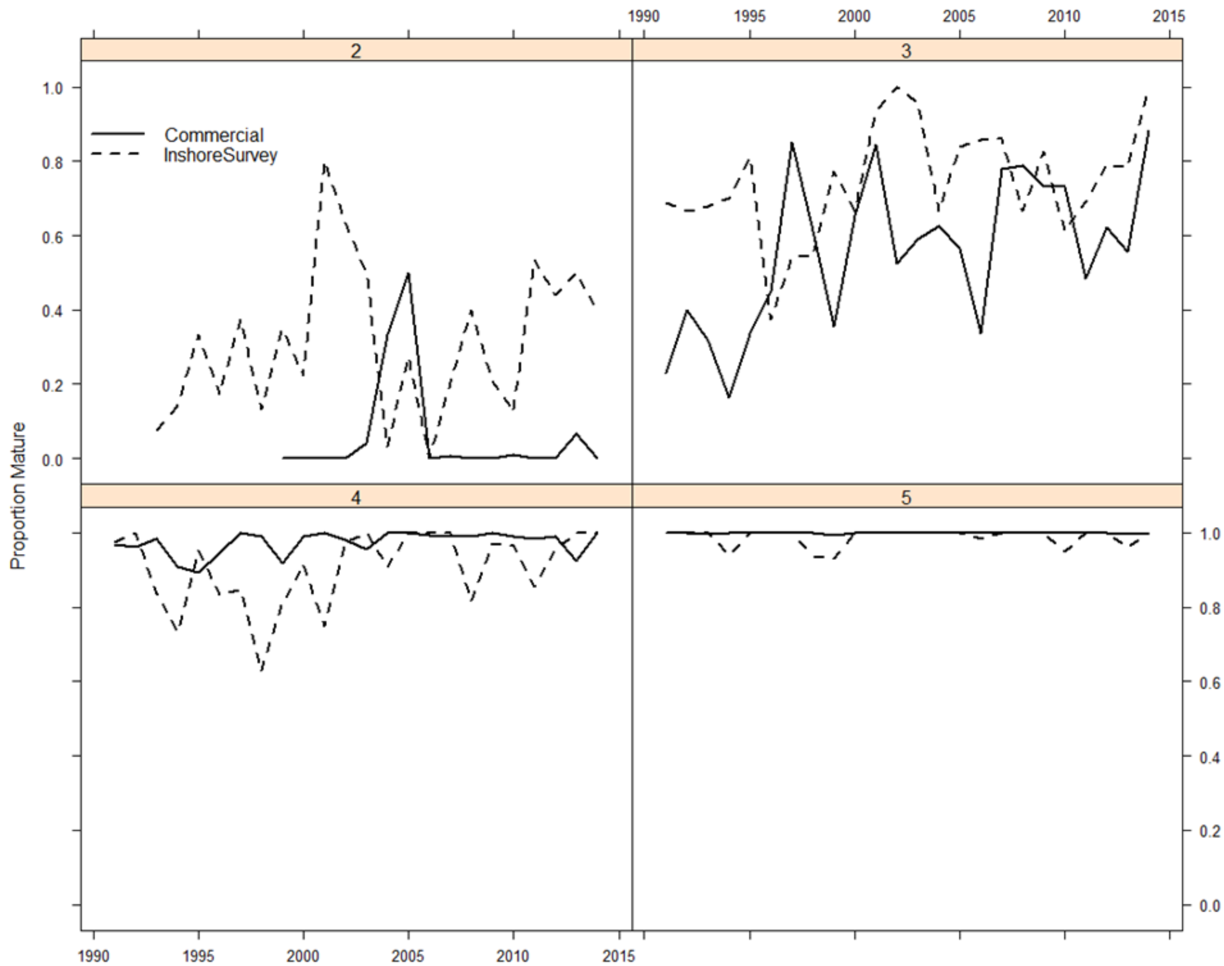


Figure 12. Proportion mature at age (age specified in the "strip" of each panel) from quarter three commercial fishery herring samples and the inshore strata (strata 26-27, 37-40) of the NMFS fall survey.

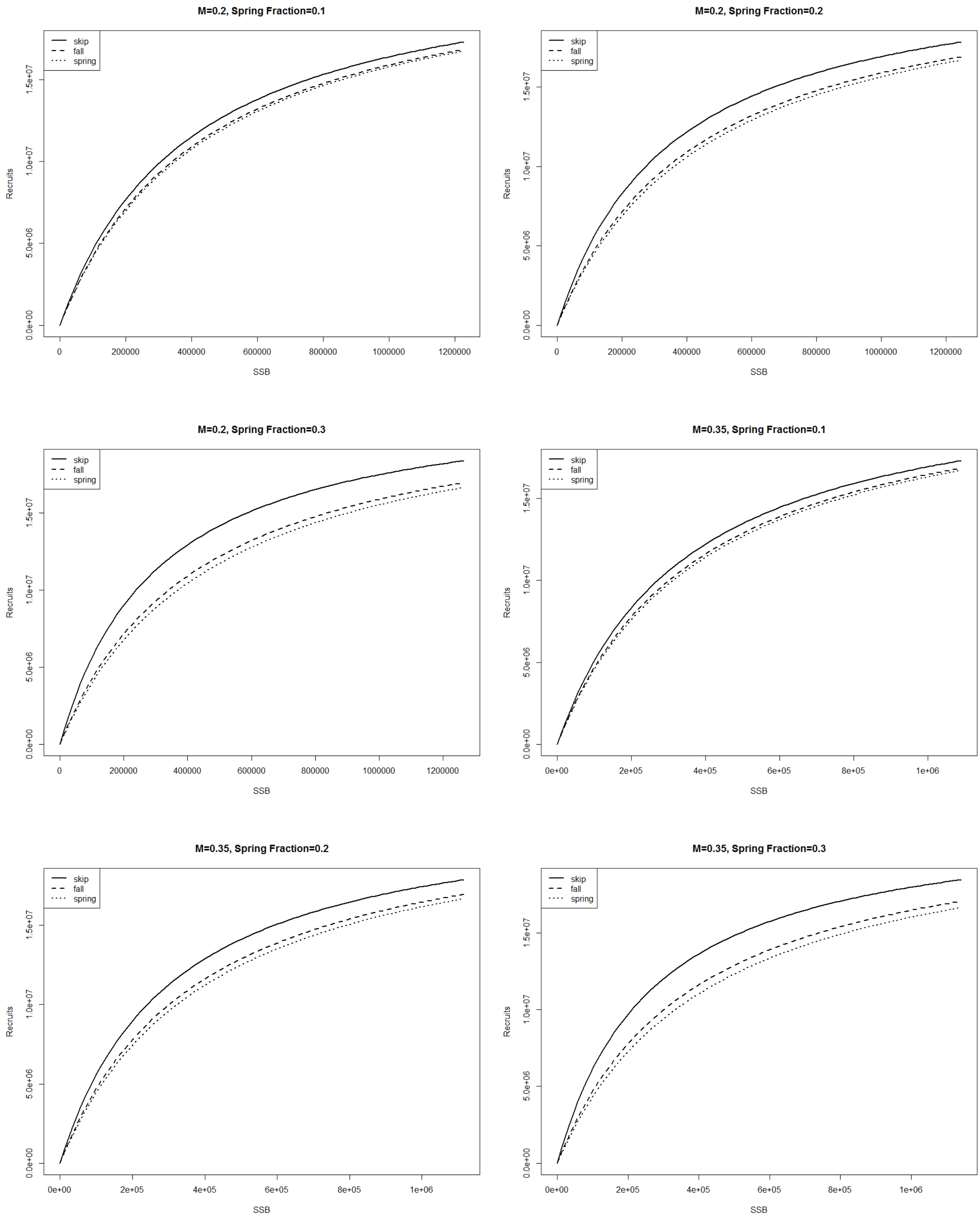


Figure 13.—Beverton-Holt stock-recruit model fits.

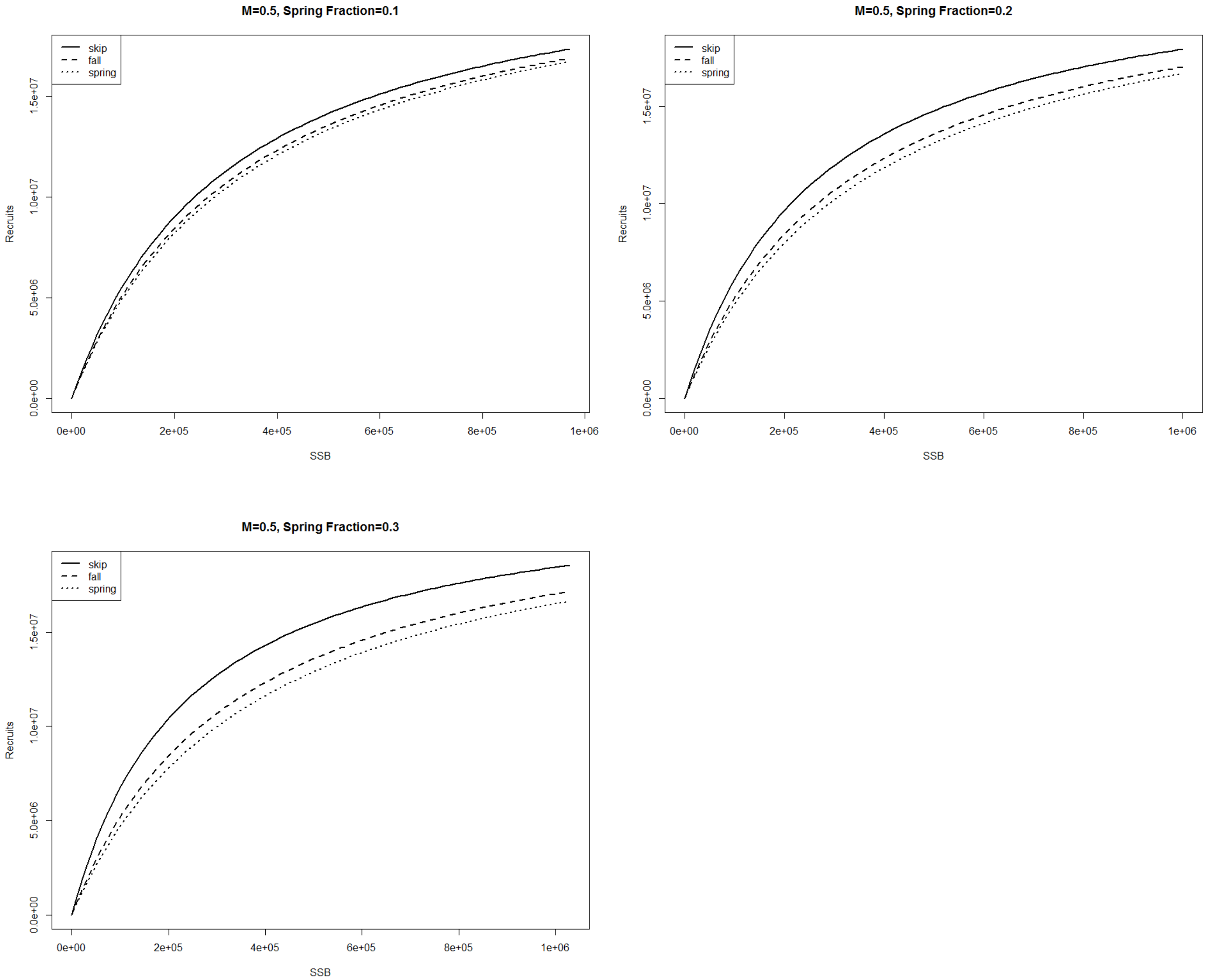
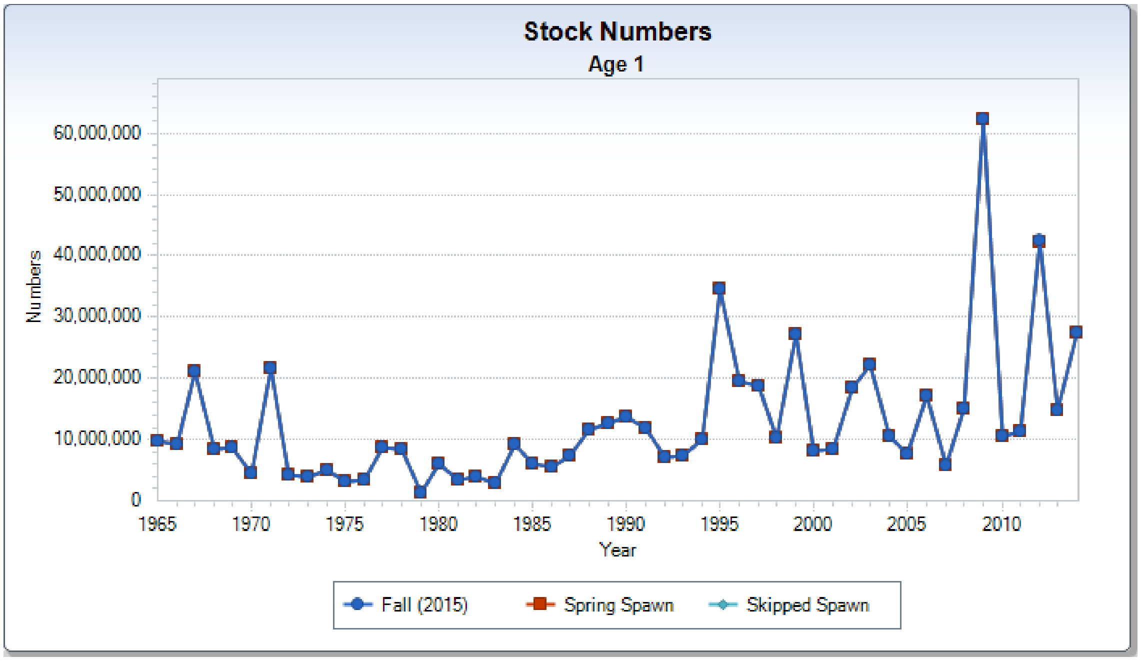


Figure 13. (continued) Beverton-Holt stock-recruitment model fits.



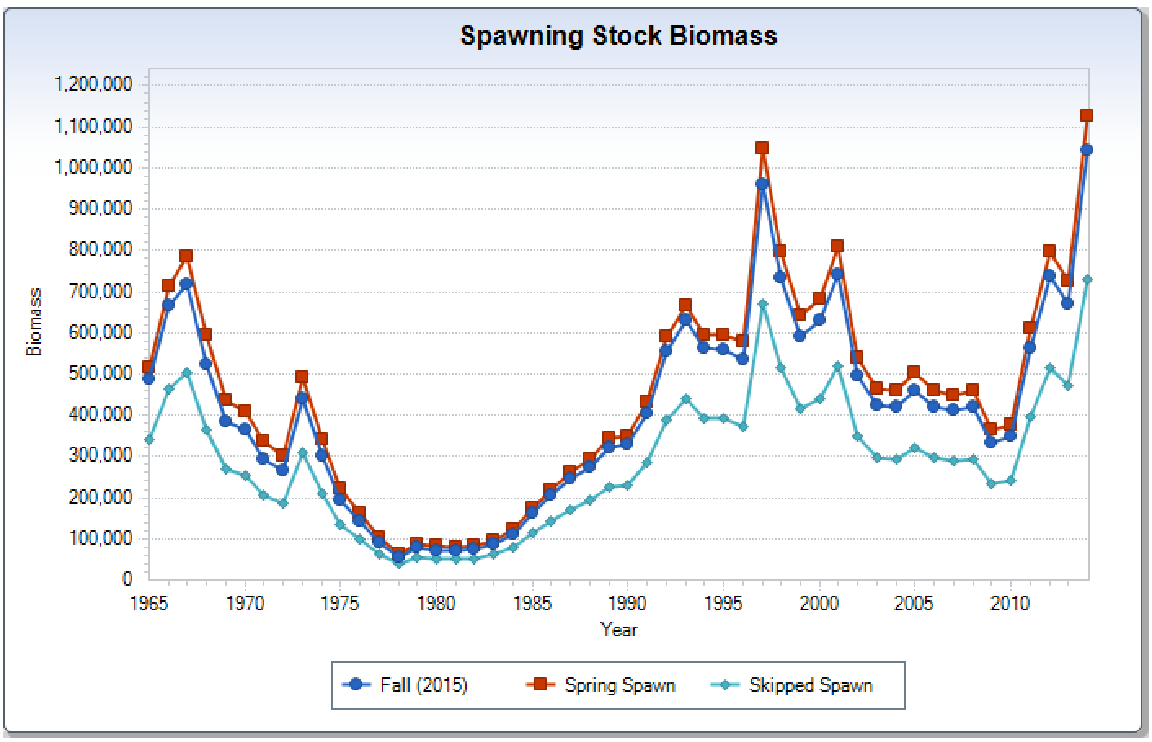


Figure 14. Time series estimates from stock assessment models assuming all fall spawning and 30% spring or skipped spawning.

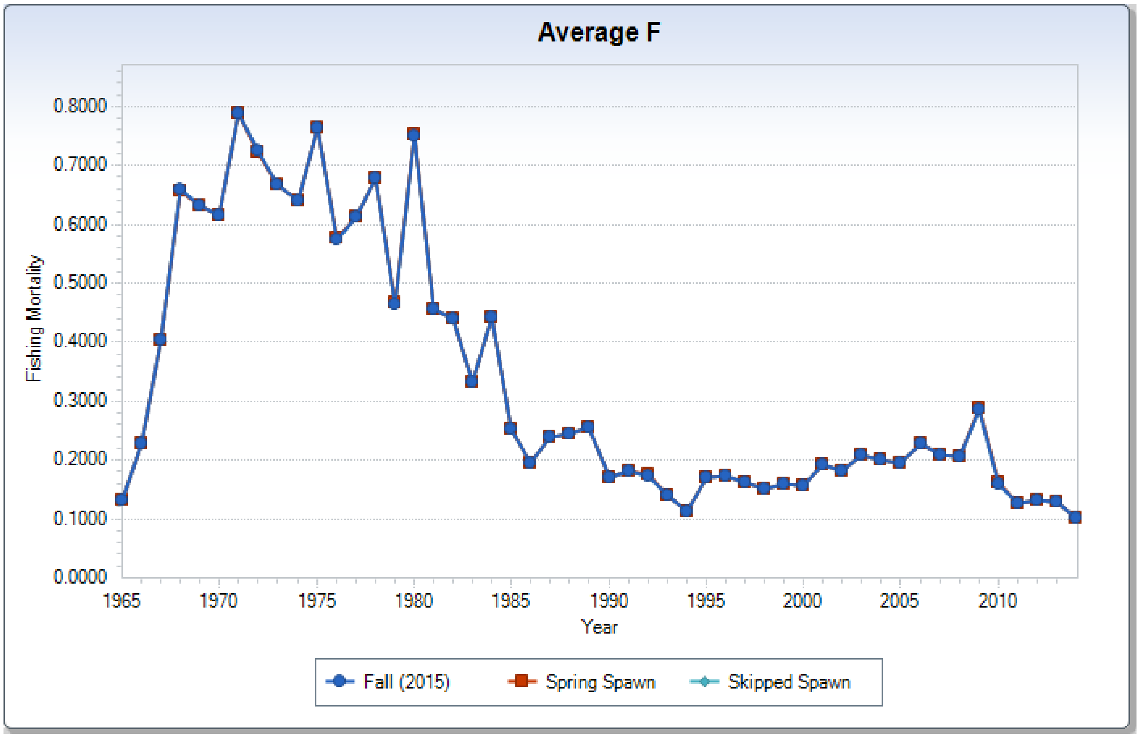


Figure 14 (continued). Time series estimates from stock assessment models assuming all fall spawning and 30% spring or skipped spawning.

Appendix B8

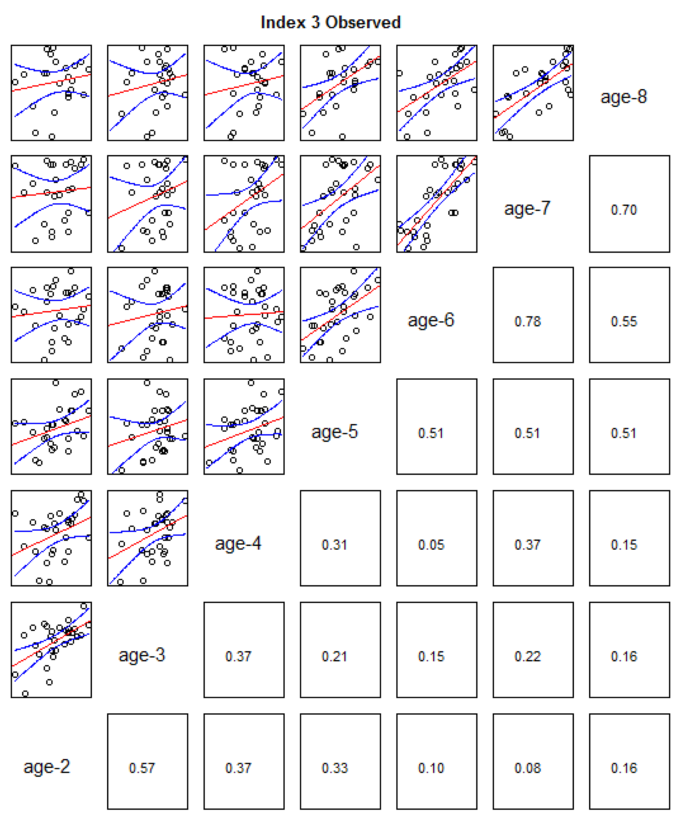
Work requested and completed during the review

TOR B2

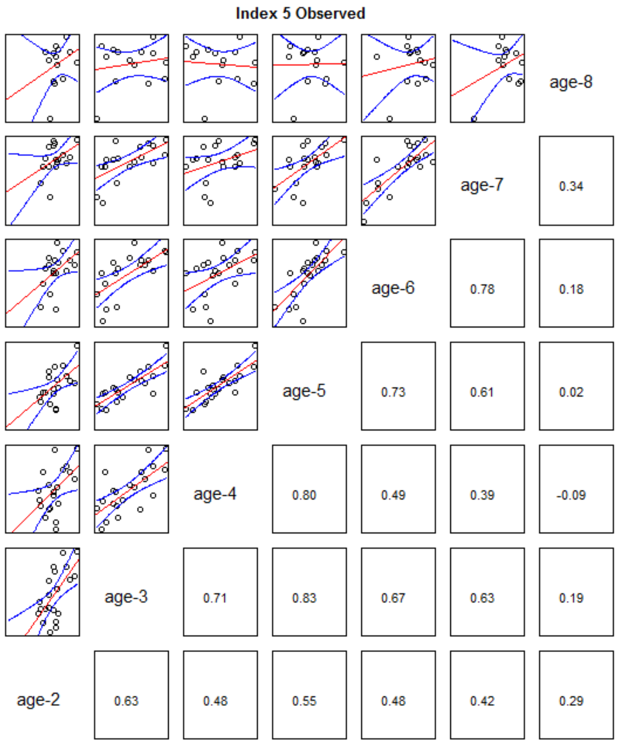
Correlation plots were requested for each age-based survey to evaluate self-consistency in tracking cohorts among years.

A plot of all the survey time series on a single plot was requested so that among survey consistency could be evaluated. Such a plot has to be interpreted with consideration that each survey may have different selectivities, and so may have lags in correlation.

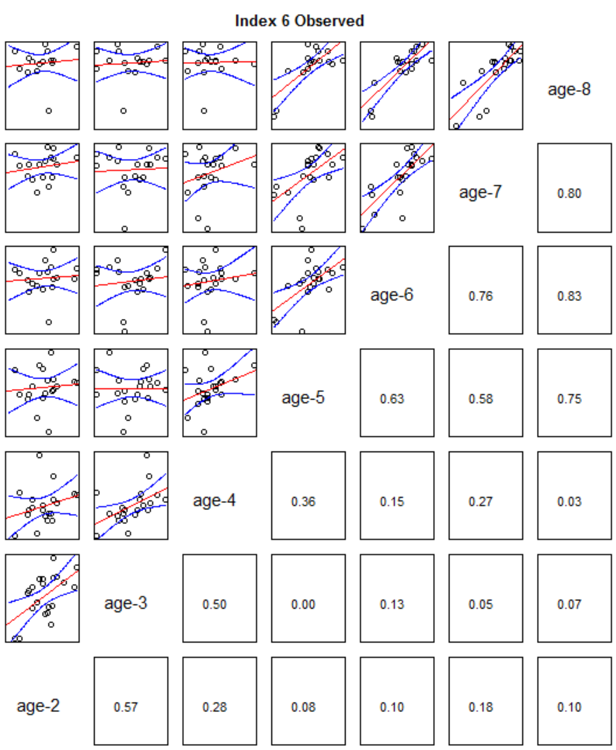
Shrimp/Summer survey



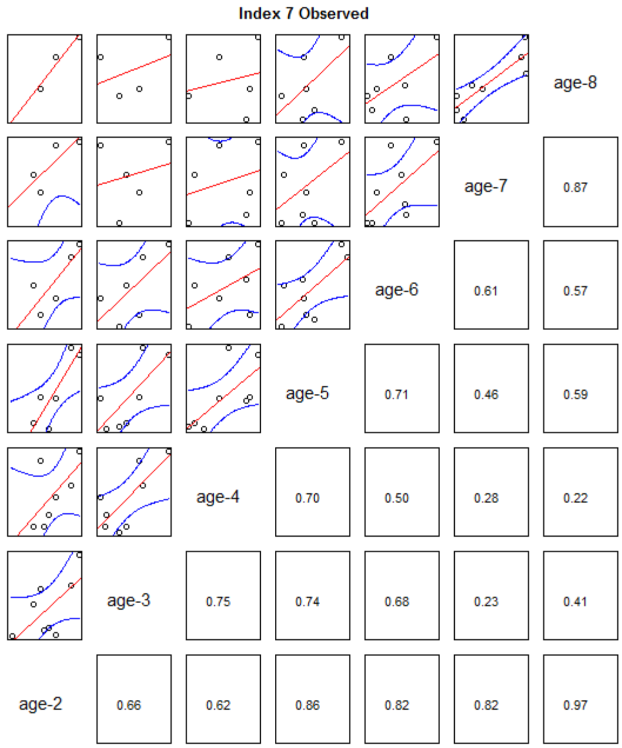
Spring Survey 1987-2008



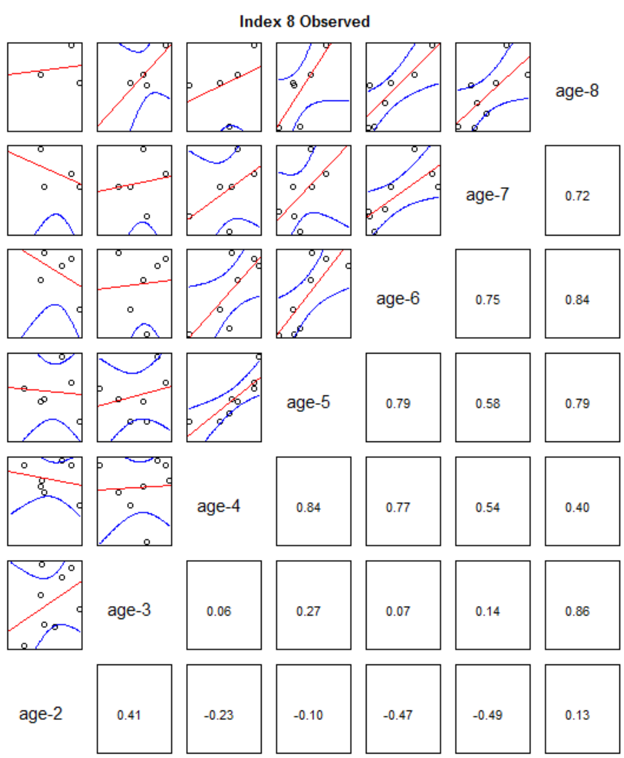
Fall Survey 1987-2008



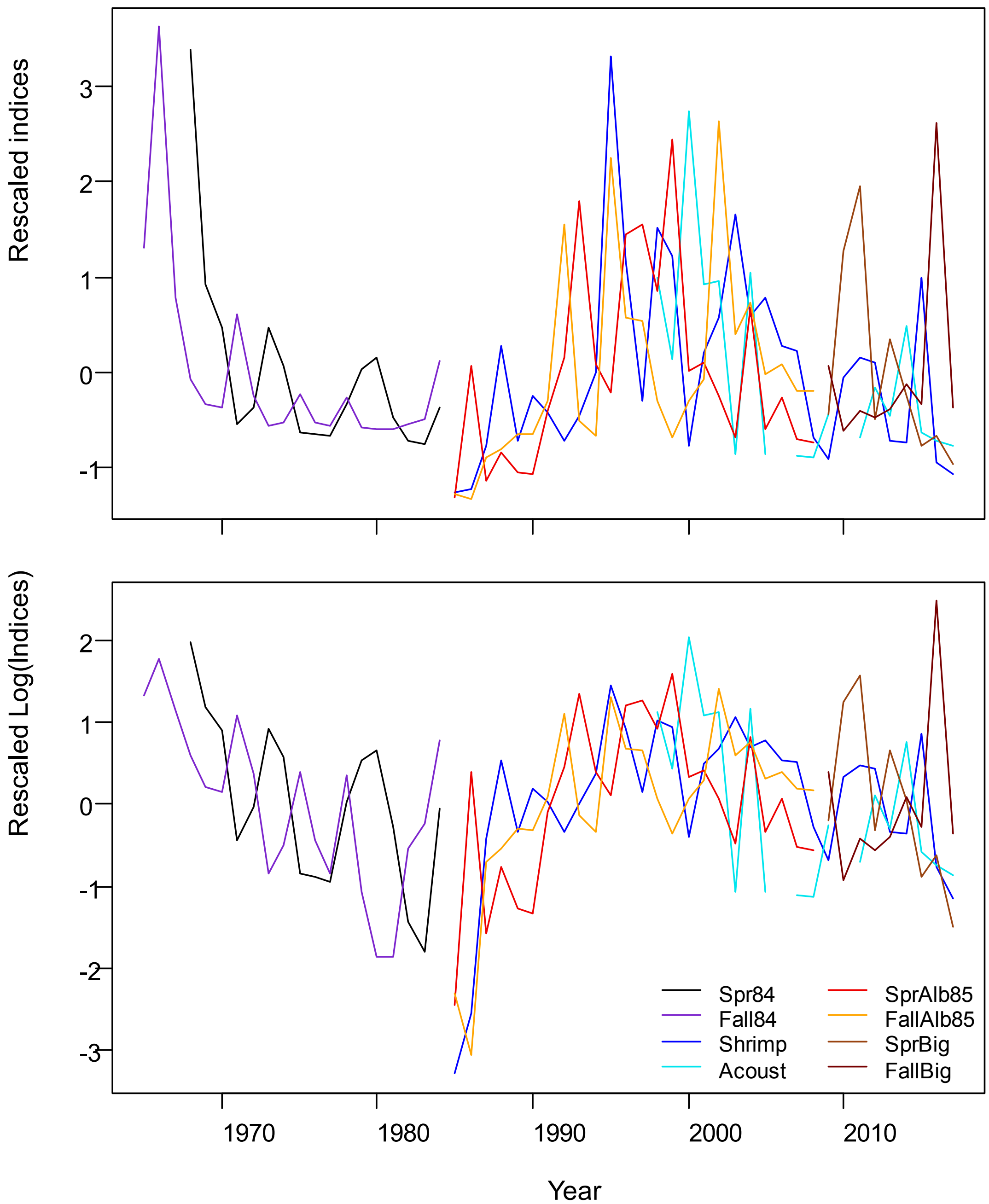
Spring Survey 2009-2017



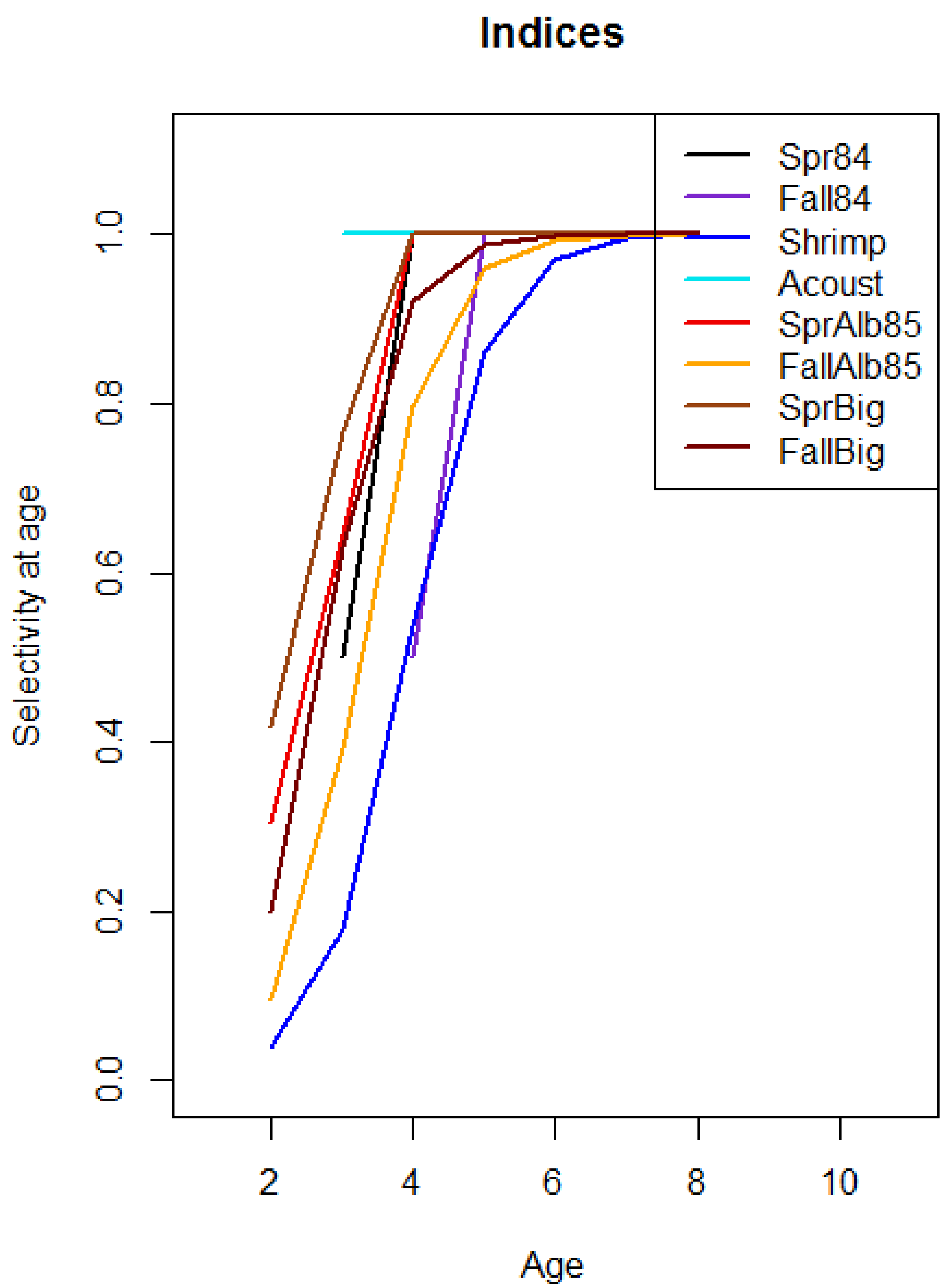
Fall Survey 2009-2017



Rescaled survey time series on a single plot



Survey selectivities



TOR B3

The 2012 herring assessment (NEFSC 2012) considered consumption of herring by highly migratory species, marine mammals, and sea birds. These consumption values were not updated for this assessment.

There is ongoing work to improve estimates of diet composition for marine mammals, in particular for pinnipeds, by the NEFSC Protected Species Branch. Population estimates for marine mammals are still largely missing to scale the diet composition to a consumption estimate, but some trend analysis is also in progress at PSB. This information will be included in updated consumption estimates as it becomes available.

Consumption of herring by predators in the Northeast US has been considered previously in the literature (Overholtz *et al.*, 2000; Overholtz and Link, 2007; Smith *et al.*, 2015), and this information was updated for the 2012 benchmark assessment. Consumption of herring by groundfish accounts for the highest proportion of total consumption, followed by marine mammals, with highly migratory species and seabird consumption contributing <9% and <3% to total consumption, respectively (Overholtz and Link, 2007). In 2018, groundfish diet and abundance information was updated and examined by the working group. However, there were no updates to diet information for marine mammals since (Smith *et al.*, 2015), and there remains no time series of abundance for marine mammals. Similarly, there are no updates to diet information for highly migratory species or seabirds, and no update to abundance information for seabirds. A ballpark update of consumption by highly migratory species is possible given than bluefin tuna accounted for the majority of highly migratory species herring consumption, and there is an updated assessment for the western stock of bluefin tuna (http://old.iccat.int/Documents/SCRS/DetRep/BFT_ASS_ENG.pdf). Using the same assumptions regarding herring diet proportion and daily ration as in Overholtz and Link, (2007), the bluefin tuna consumption estimate from 2002 can be updated to 2015 by scaling up using the ratio of 2015/2002 tuna SSB, approximately 2500/1500 t, or 1.67. This would increase the consumption estimate to approximately 41,667 t from 25,000 t for bluefin tuna. This compares with the estimated groundfish consumption ranging from 80,000-250,000 tons.

To inform the various life stages over which the consumption estimates might apply, length compositions of herring collected from predator stomachs were examined.

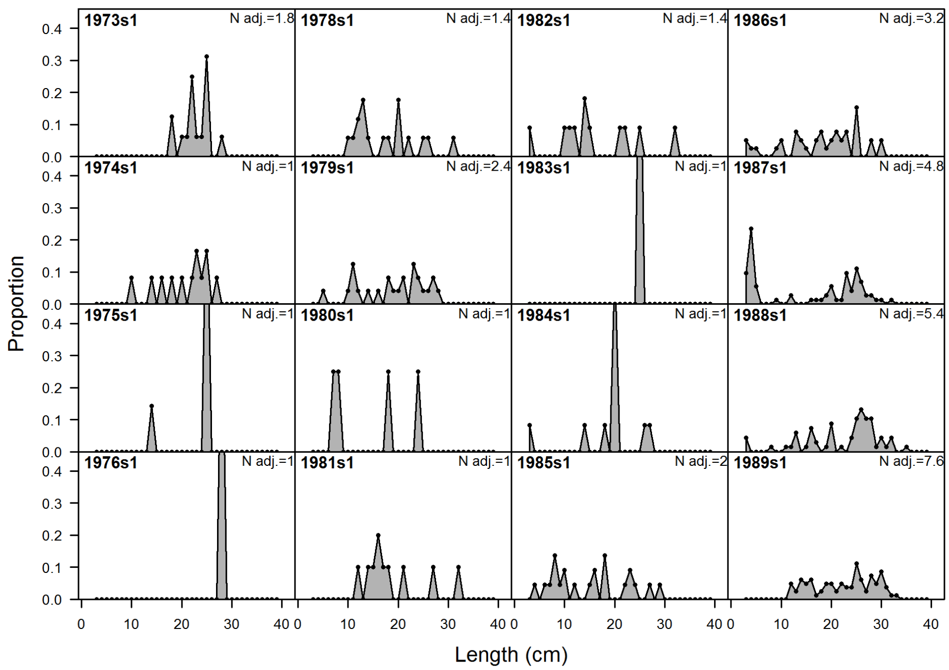
To supplement the work on the possibility of the herring stock shifting spatially in response to environmental changes, centers of gravity were examined for the spring, fall, and summer surveys.

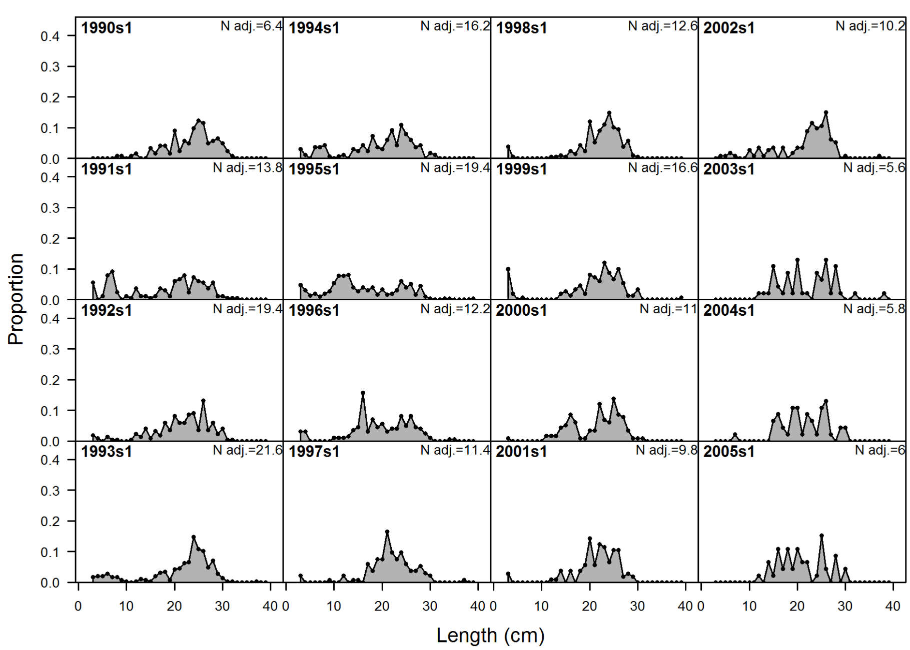
Overholtz, W. J., Link, J. S., and Suslowicz, L. E. 2000. Consumption of important pelagic fish and squid by predatory fish in the northeastern USA shelf ecosystem with some fishery comparisons. ICES Journal of Marine Science: Journal du Conseil, 57: 1147–1159.

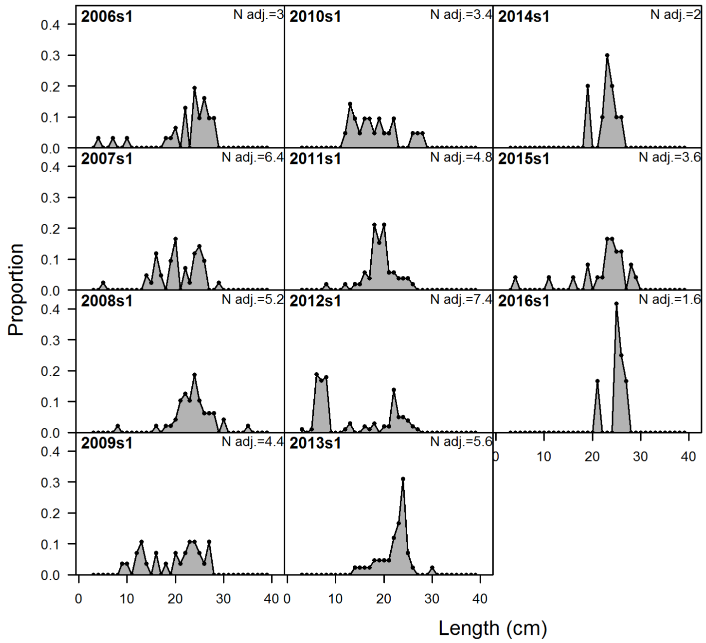
Overholtz, W. J., and Link, J. S. 2007. Consumption impacts by marine mammals, fish, and seabirds on the Gulf of Maine–Georges Bank Atlantic herring (Clupea harengus) complex during the years 1977–2002. ICES Journal of Marine Science: Journal du Conseil, 64: 83–96.

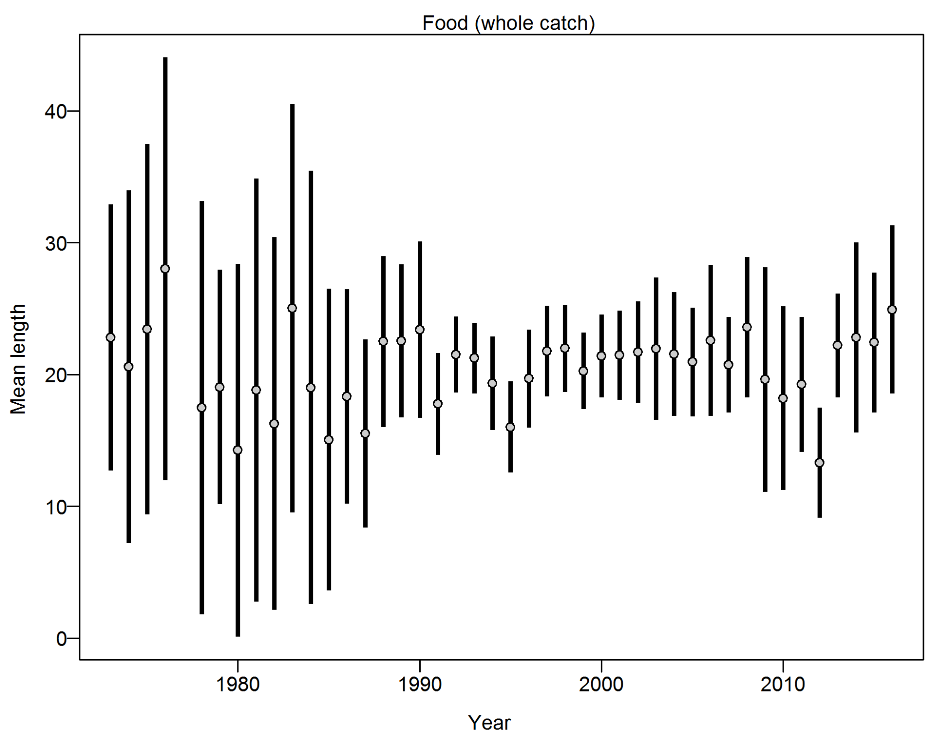
Smith, L. A., Link, J. S., Cadrin, S. X., and Palka, D. L. 2015. Consumption by marine mammals on the Northeast U.S. continental shelf. Ecological Applications, 25: 373–389.

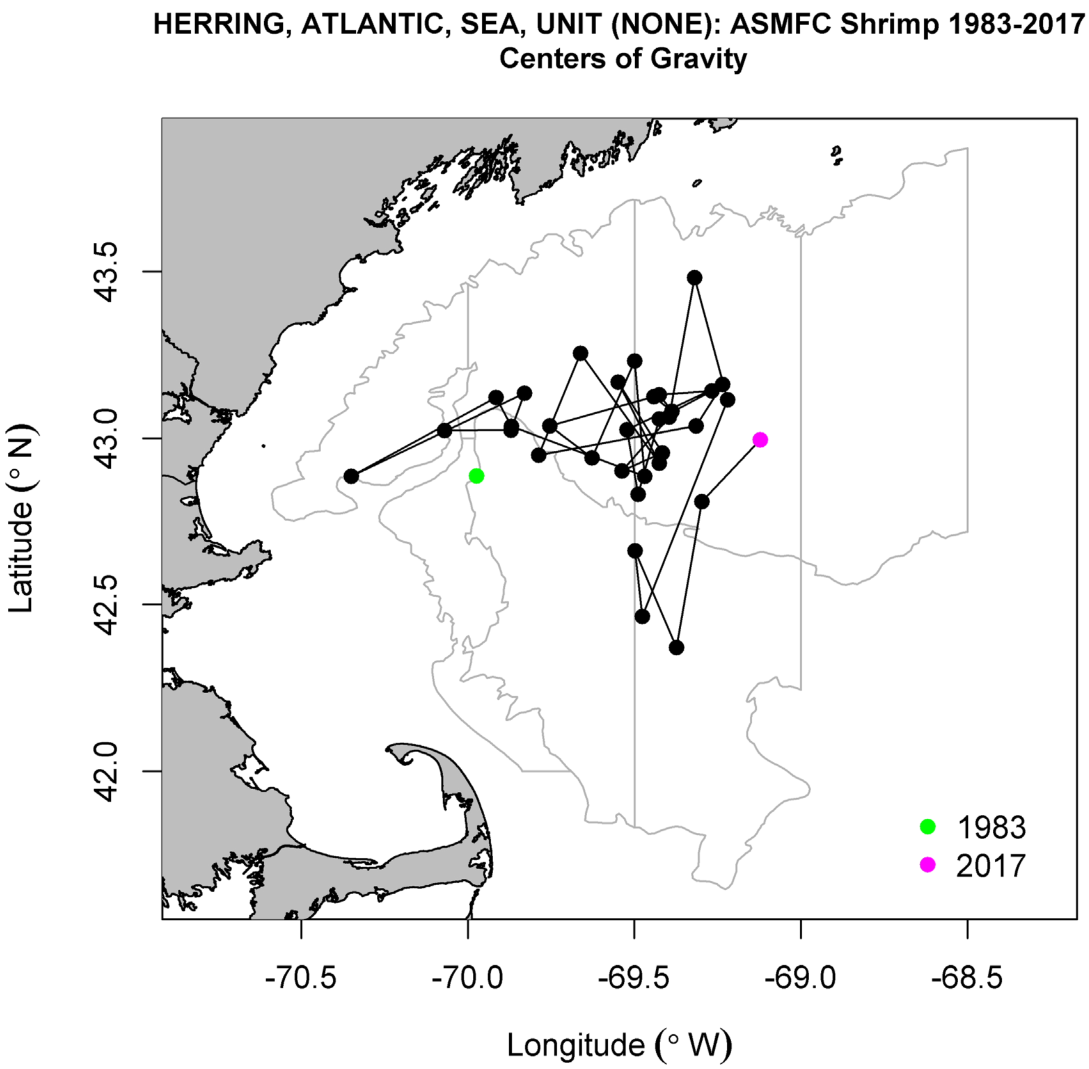
Length composition of herring identified in predator stomachs.





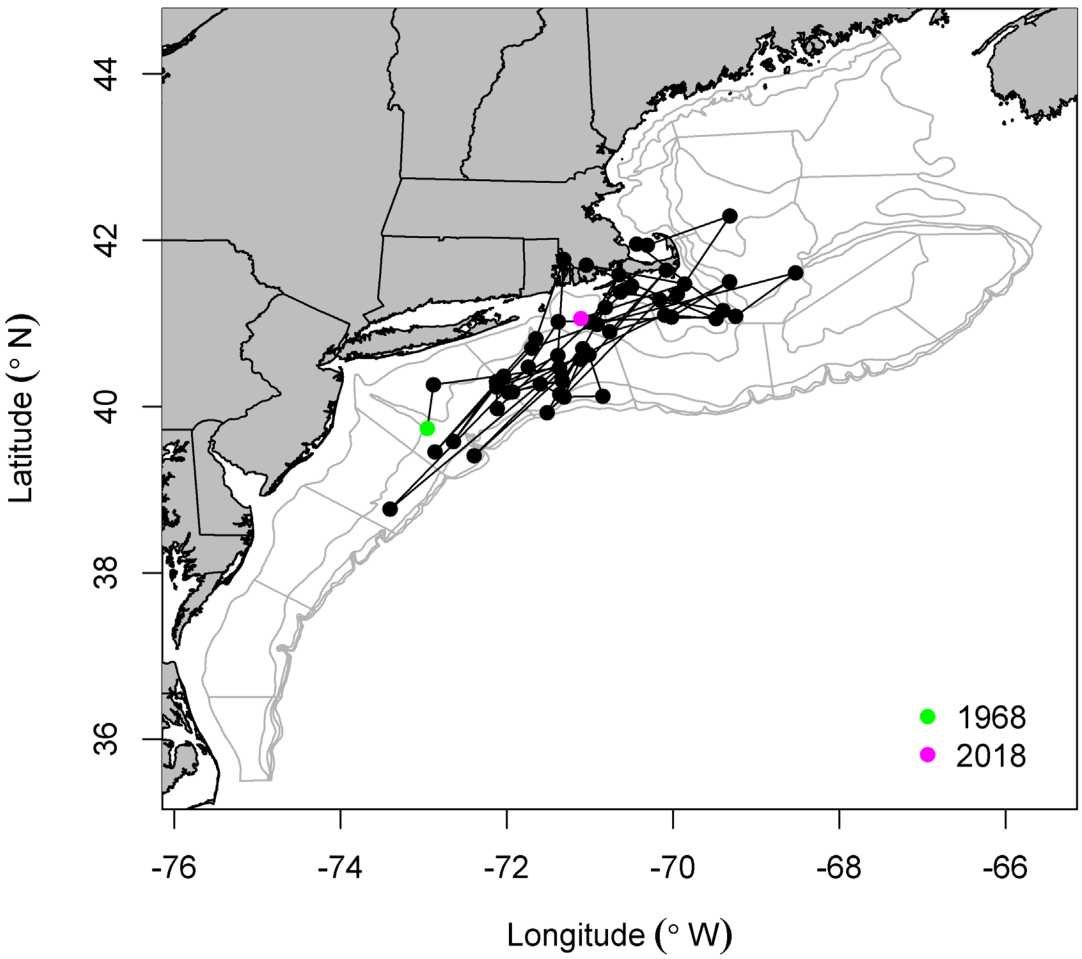

2006s1
N adj.=3
2010s1
N adj.=3.4
2014s1
N adj.=2
2007s1
N adj.=6.4
2011s1
N adj.=4.8
2015s1
N adj.=3.6
2008s1
N adj.=5.2
2012s1
N adj.=7.4
2016s1
N adj.=1.6
2009s1
N adj.=4.4
2013s1
N adj.=5.6
Proportion
Length (cm)


Food (whole catch)
Mean length
Year

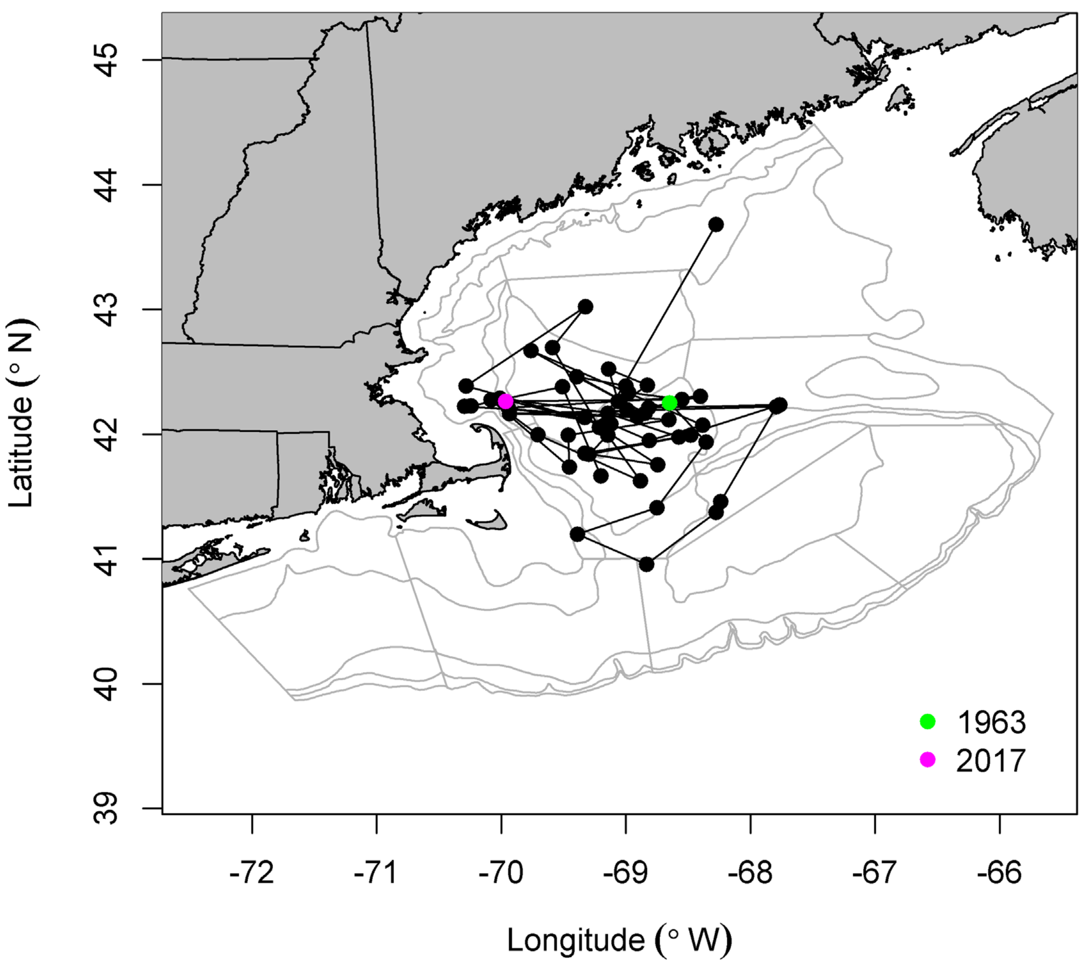


HERRING, ATLANTIC, SEA, UNIT (NONE): ASMFC Shrimp 1983-2017
Centers of Gravity
Latitude (° N)
43.5
43.0
42.5
42.0
-70.5
-70.0
-69.5
-69.0
-68.5
Longitude (° W)
1983
2017

HERRING, ATLANTIC, SEA, UNIT (NONE): NEFSC Spring BTS 1968-2018
Centers of Gravity



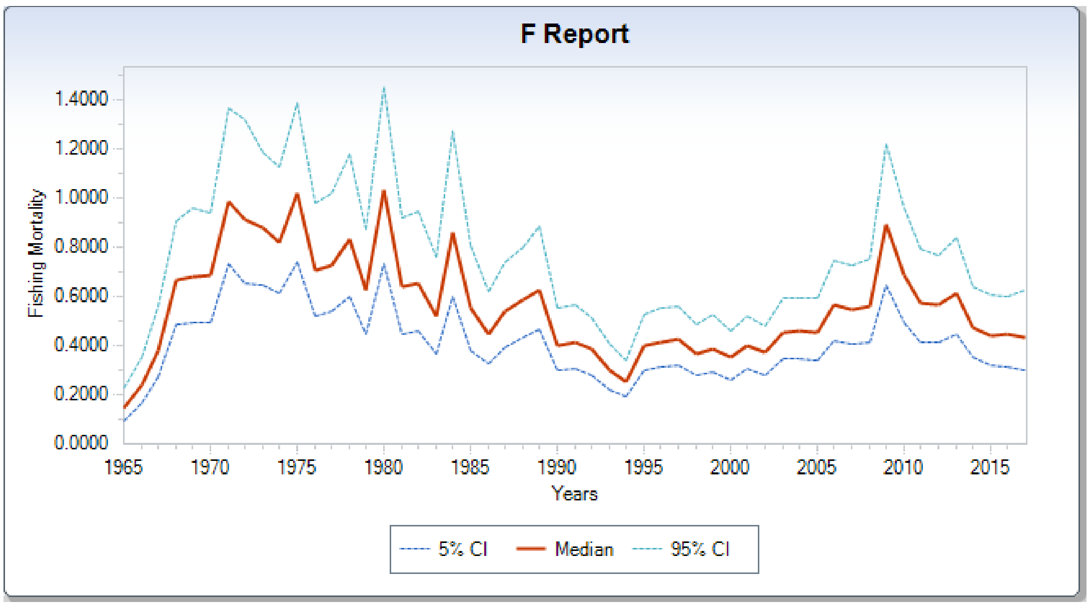
HERRING, ATLANTIC, SEA, UNIT (NONE): NEFSC Fall BTS 1963-2017
Centers of Gravity

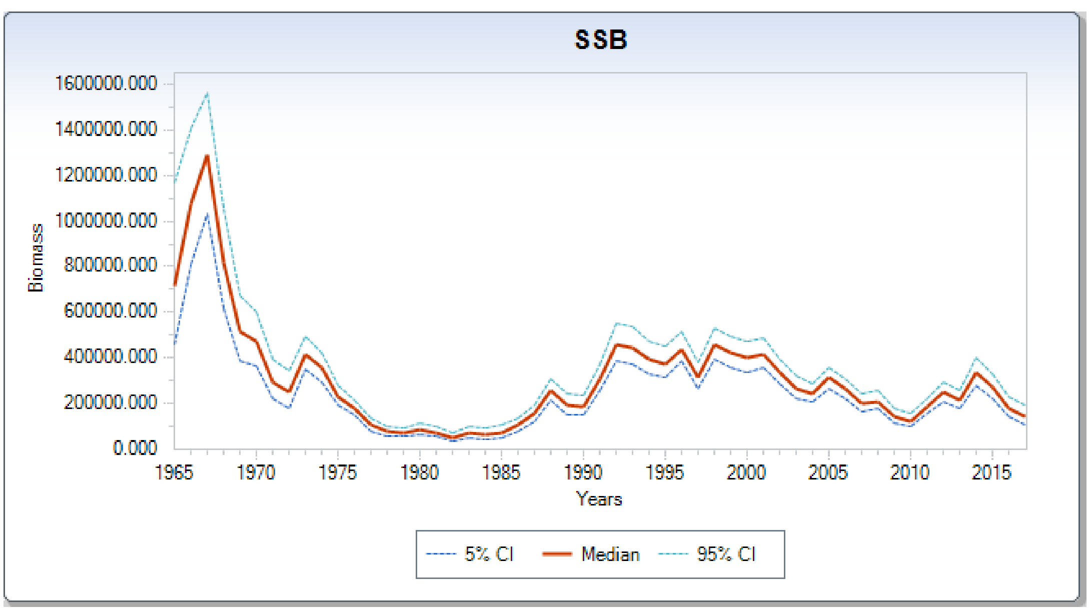


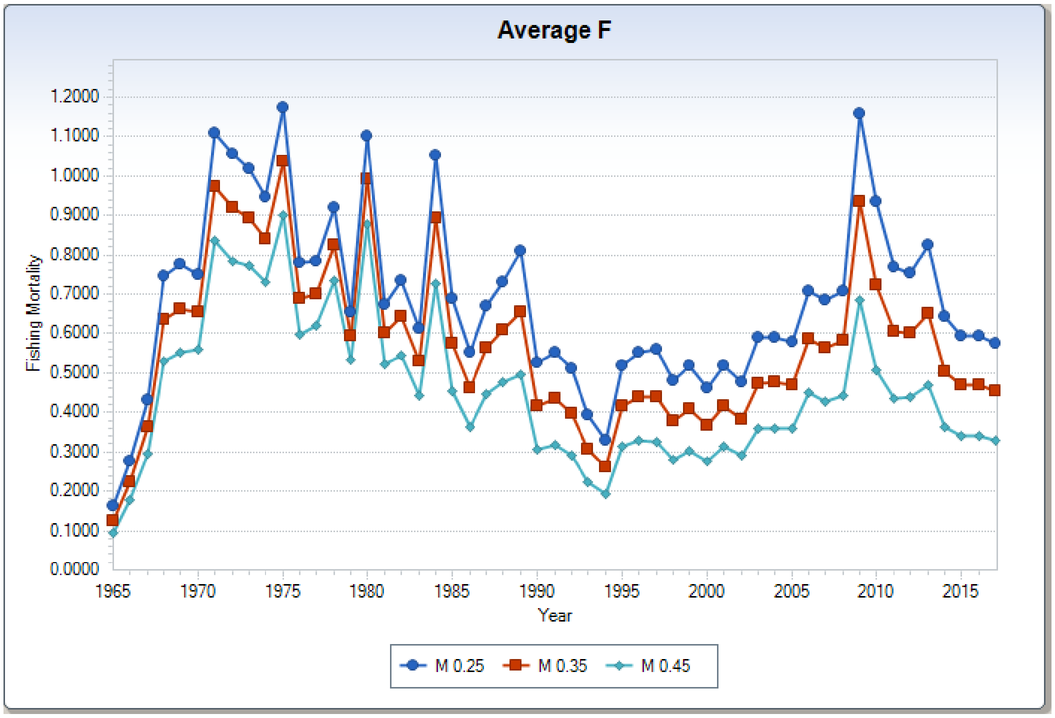
TOR B4

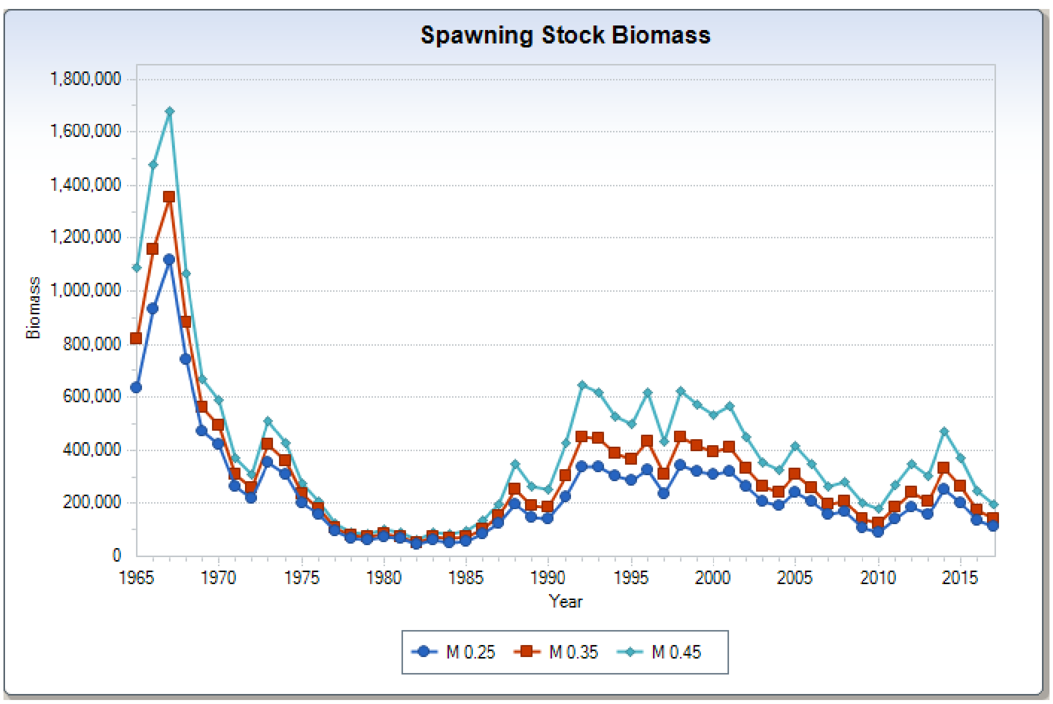
Plots of SSB and fishing mortality (averaged over ages 7 and 8 as in the main text) with confidence intervals were requested.

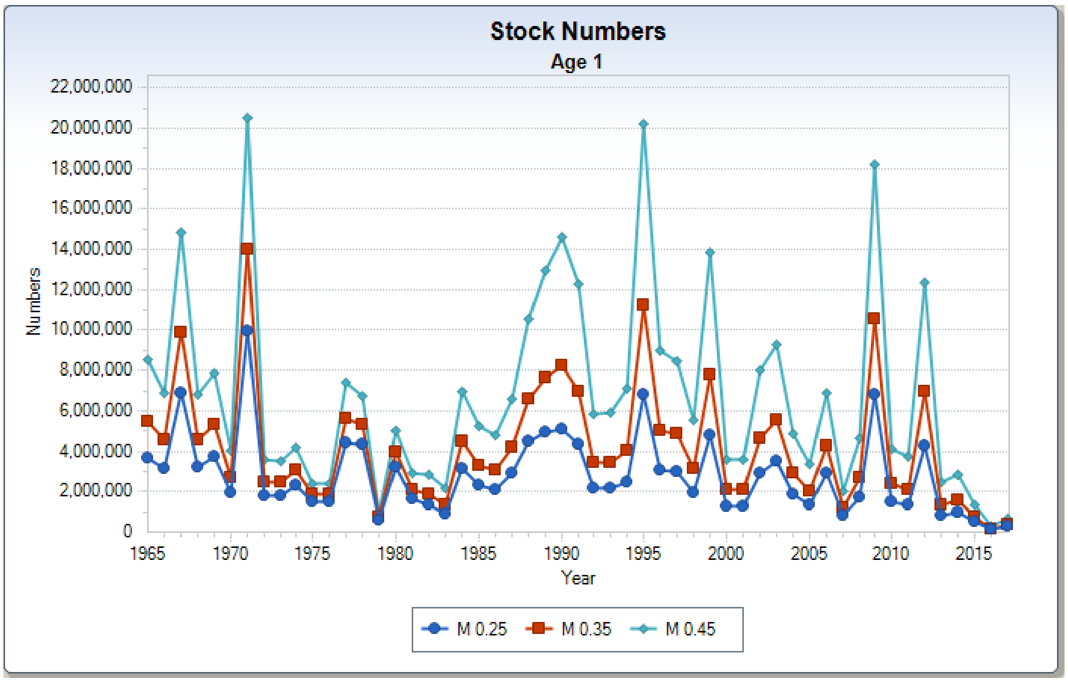
Time series estimates of SSB, F, and recruitment were requested for a range of values for input natural mortality. The base model used M=0.35.






Average F
Fishing Mortality
1.2000
1.1000
1.0000
0.9000
0.8000
0.7000
0.6000
0.5000
0.4000
0.3000
0.2000
0.1000
0.0000
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
Year
M 0.25
M 0.35
M 0.45

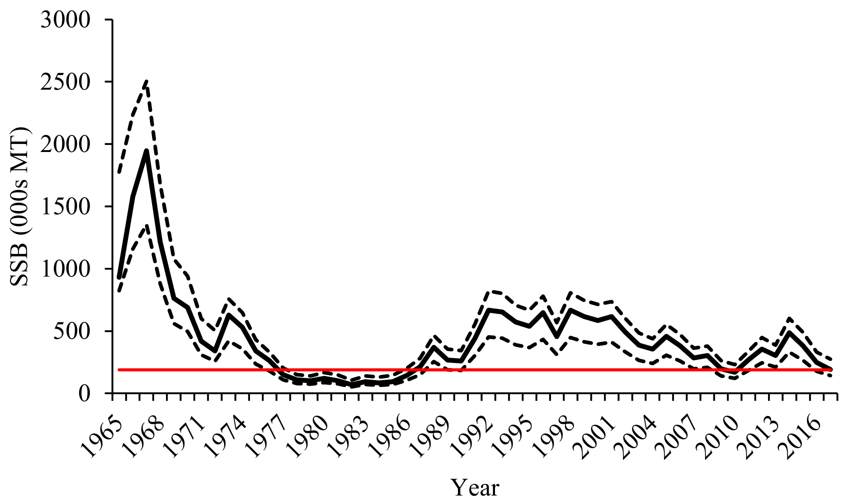

Spawning Stock Biomass
Biomass
1,800,000
1,600,000
1,400,000
1,200,000
1,000,000
800,000
600,000
400,000
200,000
0
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
Year
M 0.25
M 0.35
M 0.45

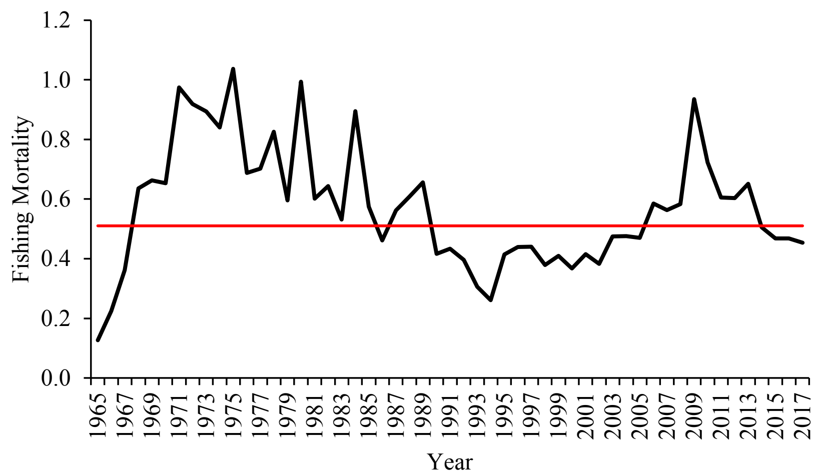

Stock Numbers
Age 1
22,000,000
20,000,000
18,000,000
16,000,000
14,000,000
12,000,000
10,000,000
8,000,000
6,000,000
4,000,000
2,000,000
0
Numbers
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
Year
M 0.25
M 0.35
M 0.45

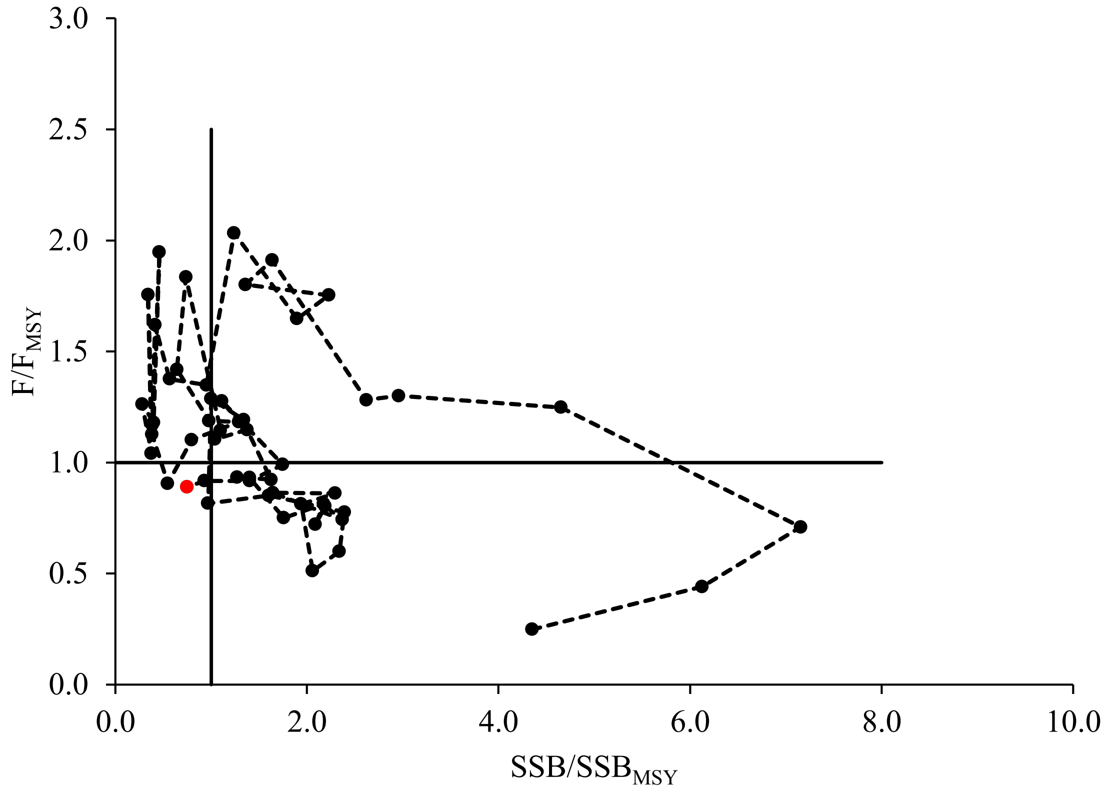
TOR 6

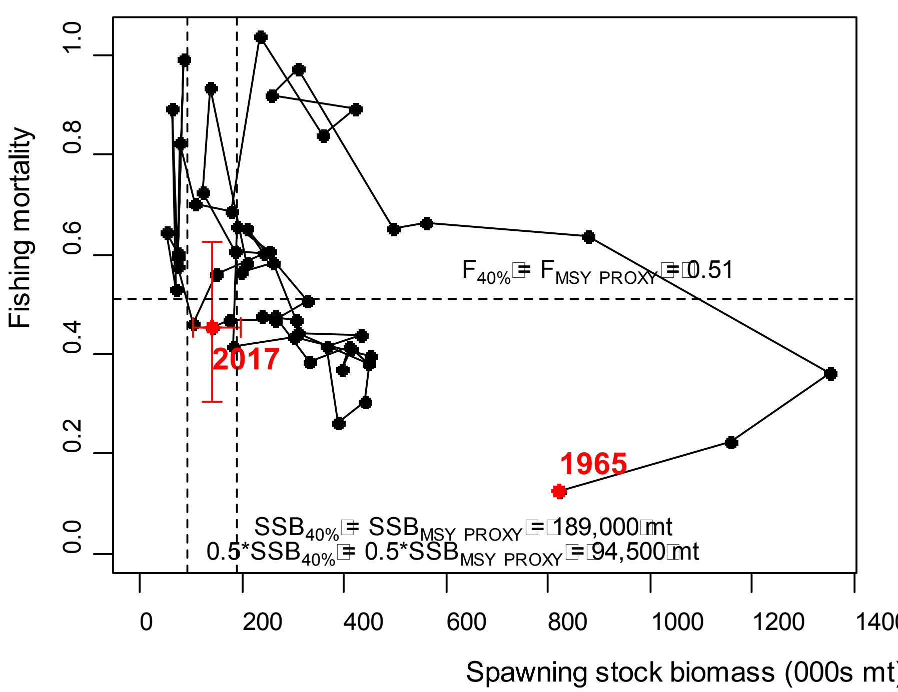
Time series plots of SSB and F were requested with horizontal lines indicating the location of MSY proxy reference points. Similarly, phase plots (Kobe plots) were requested showing the time series of stock status.

The probability of overfished and overfishing in 2017 were requested. The probability of overfished was 2% and the probability of overfishing was 24%.






3.0
2.5
2.0
1.5
1.0
0.5
0.0
F/F_{MSY}
0.0
2.0
4.0
6.0
8.0
10.0
SSB/SSB_{MSY}


Fishing mortality
0.0
0.2
0.4
0.6
0.8
1.0
$F_{40\%}$ = $F_{MSY\ PROXY}$ = 0.51
2017
1965
$SSB_{40\%}$ = $SSB_{MSY\ PROXY}$ = 189,000 mt
0.5*$SSB_{40\%}$ = 0.5*$SSB_{MSY\ PROXY}$ = 94,500 mt
0
200
400
600
800
1000
1200
Spawning stock biomass (000s mt)

TOR 7

A sensitivity analysis for short-term projections was conducted with the age-1 recruitment estimates from 1965-2015 equal to half their estimated value. The projected recruitments in 2019-2021 were drawn from the empirical cumulative distribution of the halved estimates. No other inputs were altered. This sensitivity was intended to demonstrate the consequences of recruitment remaining relatively low in the short-term.

	2018	**2019**	**2020**	**2021**
Catch (mt)	111,000	13,368	20,423	31,476
Catch 80% CI	--	3,717-36,194	10,504-41,883	18,432-55,294
$F_{7\text{-}8}$	1.7	0.51	0.51	0.51
$F_{7\text{-}8}$ 80% CI	0.83-4	--	--	--
SSB (mt)	32,900	18,800	24,350	49,700
SSB 80% CI	4,700-78,600	4,300-57,900	11,800-63,000	28,200-94,300
P(overfishing)	0.95	--	--	--
P(overfished)	0.96	0.94	0.94	0.90

	2018	**2019**	**2020**	**2021**
Catch (mt)	55,000	28,500	27,500	35,000
Catch 80% CI	--	16,900-52,600	17,000-50,400	21,500-59,800
$F_{7\text{-}8}$	0.58	0.51	0.51	0.51
$F_{7\text{-}8}$ 80% CI	0.4-0.86	--	--	--
SSB (mt)	75,300	42,500	35,100	55,000
SSB 80% CI	46,900-112,100	25,000-85,400	21,500-80,000	46,300-121,000
P(overfishing)	0.69	--	--	--
P(overfished)	0.76	0.92	0.92	0.87

Procedures for Issuing Manuscripts in the *Northeast Fisheries Science Center Reference Document* (CRD) Series

Clearance

All manuscripts submitted for issuance as CRDs must have cleared the NEFSC's manuscript/abstract/webpage review process. If any author is not a federal employee, he/she will be required to sign an "NEFSC Release-of-Copyright Form." If your manuscript includes material from another work which has been copyrighted, then you will need to work with the NEFSC's Editorial Office to arrange for permission to use that material by securing release signatures on the "NEFSC Use-of-Copyrighted-Work Permission Form."

For more information, NEFSC authors should see the NEFSC's online publication policy manual, "Manuscript/abstract/webpage preparation, review, and dissemination: NEFSC author's guide to policy, process, and procedure," located in the Publications/Manuscript Review section of the NEFSC intranet page.

Organization

Manuscripts must have an abstract and table of contents, and (if applicable) lists of figures and tables. As much as possible, use traditional scientific manuscript organization for sections: "Introduction," "Study Area" and/or "Experimental Apparatus," "Methods," "Results," "Discussion," "Conclusions," "Acknowledgments," and "Literature/References Cited."

Style

The CRD series is obligated to conform with the style contained in the current edition of the United States Government Printing Office Style Manual. That style manual is silent on many aspects of scientific manuscripts. The CRD series relies more on the CSE Style Manual. Manuscripts should be prepared to conform with these style manuals.

The CRD series uses the American Fisheries Society's guides to names of fishes, mollusks, and decapod crustaceans, the Society for Marine Mammalogy's guide to names of marine mammals, the Biosciences Information Service's guide to serial title abbreviations, and the ISO's (International Standardization Organization) guide to statistical terms.

For in-text citation, use the name-date system. A special effort should be made to ensure that all necessary bibliographic information is included in the list of cited works. Personal communications must include date, full name, and full mailing address of the contact.

Preparation

Once your document has cleared the review process, the Editorial Office will contact you with publication needs – for example, revised text (if necessary) and separate digital figures and tables if they are embedded in the document. Materials may be submitted to the Editorial Office as files on zip disks or CDs, email attachments, or intranet downloads. Text files should be in Microsoft Word, tables may be in Word or Excel, and graphics files may be in a variety of formats (JPG, GIF, Excel, PowerPoint, etc.).

Production and Distribution

The Editorial Office will perform a copy-edit of the document and may request further revisions. The Editorial Office will develop the inside and outside front covers, the inside and outside back covers, and the title and bibliographic control pages of the document.

Once both the PDF (print) and Web versions of the CRD are ready, the Editorial Office will contact you to review both versions and submit corrections or changes before the document is posted online.

A number of organizations and individuals in the Northeast Region will be notified by e-mail of the availability of the document online.

Research Communications Branch
Northeast Fisheries Science Center
National Marine Fisheries Service, NOAA
166 Water St.
Woods Hole, MA 02543-1026

MEDIA MAIL

Publications and Reports of the Northeast Fisheries Science Center

The mission of NOAA's National Marine Fisheries Service (NMFS) is "stewardship of living marine resources for the benefit of the nation through their science-based conservation and management and promotion of the health of their environment." As the research arm of the NMFS's Northeast Region, the Northeast Fisheries Science Center (NEFSC) supports the NMFS mission by "conducting ecosystem-based research and assessments of living marine resources, with a focus on the Northeast Shelf, to promote the recovery and long-term sustainability of these resources and to generate social and economic opportunities and benefits from their use." Results of NEFSC research are largely reported in primary scientific media (*e.g.*, anonymously-peer-reviewed scientific journals). However, to assist itself in providing data, information, and advice to its constituents, the NEFSC occasionally releases its results in its own media. Currently, there are three such media:

NOAA Technical Memorandum NMFS-NE -- This series is issued irregularly. The series typically includes: data reports of long-term field or lab studies of important species or habitats; synthesis reports for important species or habitats; annual reports of overall assessment or monitoring programs; manuals describing program-wide surveying or experimental techniques; literature surveys of important species or habitat topics; proceedings and collected papers of scientific meetings; and indexed and/or annotated bibliographies. All issues receive internal scientific review and most issues receive technical and copy editing.

Northeast Fisheries Science Center Reference Document -- This series is issued irregularly. The series typically includes: data reports on field and lab studies; progress reports on experiments, monitoring, and assessments; background papers for, collected abstracts of, and/or summary reports of scientific meetings; and simple bibliographies. Issues receive internal scientific review and most issues receive copy editing.

Resource Survey Report (formerly *Fishermen's Report*) -- This information report is a regularly-issued, quick-turnaround report on the distribution and relative abundance of selected living marine resources as derived from each of the NEFSC's periodic research vessel surveys of the Northeast's continental shelf. This report undergoes internal review, but receives no technical or copy editing.

TO OBTAIN A COPY of a *NOAA Technical Memorandum NMFS-NE* or a *Northeast Fisheries Science Center Reference Document*, either contact the NEFSC Editorial Office (166 Water St., Woods Hole, MA 02543-1026; 508-495-2350) or consult the NEFSC webpage on "Reports and Publications" (http://www.nefsc.noaa.gov/nefsc/publications/). To access *Resource Survey Report*, consult the Ecosystem Surveys Branch webpage (http://www.nefsc.noaa.gov/femad/ecosurvey/mainpage/).