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(This will be a chapter in the 56th SAW Assessment Report when the full report is published. There may be additional editing to this draft.)

STOCK ASSESSMENT FOR ATLANTIC SURFCLAMS IN THE US EEZ FOR 2013

A report of the SAW Invertebrate Working Group, reviewed during SARC56, Feb. 2013

Terms of reference for Atlantic surfclam

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

2. Present the survey data being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, age-length data, relevant cooperative research, etc.). Investigate the utility of commercial LPUE as a measure of relative abundance. Characterize the uncertainty and any bias in these sources of data.

3. Evaluate the current stock definition in terms of spatial patterns in biological characteristics, population dynamics, fishery patterns, the new cooperative survey, utility of biological reference points, etc. If appropriate, recommend one or more alternative stock definitions, based on technical grounds. Integrate these results into TOR-4.

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

5. State the existing ***stock status*** *definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for* B_{MSY}, $B_{THRESHOLD}$, F_{MSY} *and MSY) and provide estimates of their uncertainty. This should be carried out using the existing stock definition and, if possible, for the recommended "alternative" stock definitions from TOR-3. If analytic model-based estimates are unavailable, consider recommending alternative measurable proxies for BRPs. Comment on the appropriateness of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs.*

6. Evaluate stock status with respect to the existing assessment model and with respect to any new assessment model. Determine stock status based on the existing stock definition and, if appropriate and if time permits, for "alternative" stock definitions from TOR-3.

- *a.When working with the existing model, update it with new data and evaluate stock status (overfished and overfishing) with respect to the existing BRP estimates.*
- *b. Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR-5).*

7. Develop approaches and apply them to conduct stock projections and to compute the statistical distribution (e.g., probability density function) of the OFL (overfishing level) and candidate ABCs (Acceptable Biological Catch; see Appendix to the SAW TORs).

- *a. Provide numerical annual projections (3-5 years). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. Use a sensitivity analysis approach in which a range of assumptions about the most important uncertainties in the assessment are considered (e.g., terminal year abundance, variability in recruitment).*

b. Comment on which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.
c. Describe this stock's vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

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

Executive Summary

TOR 1. Commercial fishery

About 20,000 mt of surfclam meats (18,600 mt from federal waters) were landed during 2011. Total landings were down slightly from the last assessment (22,519 mt in 2008). Landings during 2011 were mostly from the New Jersey (NJ 64%), Southern New England (SNE 13%) and the Georges Bank (GBK 13%) regions. The Long Island (LI) and Delmarva (DMV) regions supplied about 10% of total landings. About 74% of the total effort in 2011 occurred in NJ, with an additional 15% occurring in SNE. Landings per unit effort (LPUE) were near record low levels, approximately 40 – 60 bushels (bu) per hour except in GBK where they were approximately 290 bu h^{-1}. Commercial surfclam data are considered accurate and precise relative to many fisheries because there is no discarding and few active permits. Landings are reported both in log books and by dealers. TOR 1 is discussed starting on page 8.

TOR 2. Survey

NEFSC survey data were collected in 2011 aboard the *RV Delaware II*. Recruitment of small surfclams (50 – 119 mm) for the whole EEZ stock has increased since 2005 based on survey data. Survey catch of larger surfclams recruited to the fishery (120+ mm) has been stable since 2005. Despite positive trends, both recruitment and number per tow were below average for the time series. NEFSC, Industry and academic collaborators conducted depletion and selectivity experiments from the *FV Pursuit* in 2011. New estimates of survey dredge efficiency, and selectivity were produced, as well as refinements to shell height to meat weight relationships and growth curve estimates. Age and size composition data from survey catches were used in the primary assessment model for the first time. TOR 2 is discussed starting on page 12.

TOR 3. Stock definition

The current definition is a single EEZ surfclam stock which extends from Georges Bank (GBK) in the north to Southern Virginia – SVA. An alternative definition would divide the surfclam stock into northern (GBK) and southern (Southern Virginia - SVA to SNE) components. The Invertebrate Subcommittee discussed the technical merits of both approaches but no consensus was reached and conclusions were left to reviewers. The SARC56 Panel concluded the material presented did not contain sufficient information to allow it to reach a decision on stock definition. The SARC Panel noted that this does not prevent the stock assessment from being conducted by subareas, nor does it preclude area-based management. Arguments for and against both options are presented concisely in tabular form with a brief introduction starting on page 23.

TOR 4. Model results

The primary assessment model was a statistical catch at size model, Stock Synthesis (SS3), instead of the biomass dynamic delay difference model (KLAMZ), used previously. Using SS3 allowed the working group to make use of age and size composition data for the first time. Additional changes to the assessment model included: new estimates of capture efficiency, size selectivity, growth curves, shell length to meat weight formulas, and a new approach to modeling the stock, where the GBK and southern areas were modeled separately. Results indicate that biomass was higher and fishing mortality rates that were lower than in previous assessments. In general, population trends appear well estimated while population scale (overall level of biomass in mt) was uncertain. Discussion of TOR 4 begins on page 25.

TOR 5. Stock status definitions

The current overfished threshold for surfclams is ½ B_{MSY} proxy = ¼ B_{1999} and the biomass target is ½ B_{1999}. The overfishing threshold is F=M=0.15. The fishing mortality reference point was considered adequate under either the current or alternative stock definition and no changes were recommended in this assessment.

Biomass reference points depend on which stock definition is adopted. The biomass reference point was considered adequate for the current stock definition and for the southern part of the resource. However, it was not possible to estimate B_{MSY} or a proxy for GBK in the time available because surfclams on GBK have had little exploitation, biomass has changed substantially there in the absence of fishing, environmental conditions are changing and the response of surfclams to fishing could not be predicted. A B_{MSY} proxy for GBK may be an important topic for future research but the question does not affect status determinations in this assessment given that the GBK area is essentially unexploited and cannot, by definition, be overfished. TOR 5 is discussed starting on page 31.

TOR 6. Stock status

The surfclam population is not overfished and overfishing is not occurring under either the current or alternative stock definitions. TOR 6 is discussed starting on page 33.

TOR 7. Projections

Projections indicate that the population is unlikely to be overfished and that overfishing is unlikely to occur by 2021 under either, the current or alternative stock definitions and a wide range of assumed catches. TOR 7 is discussed starting on page 34.

TOR 8. Research recommendations

Research recommendations are discussed starting on page 35.

Introduction

Distribution and biology

Atlantic surfclams are large fast growing bivalves distributed along the coast of North America from the southern Gulf of St. Lawrence to Cape Hatteras (Figure A1), with major concentrations on Georges Bank, the south shore of Long Island, New Jersey and the Delmarva Peninsula. Surfclams are found from the intertidal zone to a depth of 128m but the highest concentrations are found at depths of less than 40m. Off of the Delmarva Peninsula where the water is warmest, they are distributed in slightly deeper, cooler water. Surfclams, which burrow energetically, inhabit medium-grained sand, although they can also be found in fine or silted sand.

Surfclams are the largest bivalves in the western North Atlantic, reaching a maximum size of about 22 cm (Ropes 1980). Individuals larger than 16 cm shell length (SL - the distance across the longest part of the shell) are relatively common in Northeast Fisheries Science Center (NEFSC) surveys. Growth to commercial size (12 cm) takes about 6-7 years. Weinberg (1998), and Weinberg and Helser (1996), show that growth rates vary among regions, over time, and in response to surfclam density levels. Slower growth in surfclams in DMV and NJ during recent years coincides with mortality in near shore areas probably due to warm water (Weinberg et al 2005)

Surfclams taken in the NEFSC clam surveys are aged regularly. The surfclam shells are sectioned through the chondrophore (the attachment surface for the “hinge” ligament) and the annuli (rings) are counted. Surfclams age 30+ are relatively common and the maximum observed age exceeds 37. Most surfclams have recruited to the fishery (reached a shell length of 12 cm) by the time they are six or seven years old.

Surfclams can reach sexual maturity at three months of age (Cargnelli et al.1999). Sexes are separate, but are not distinguished in either commercial or NEFSC survey data. Spawning is thought to occur from late spring through early fall, generally depending on latitude, with more southern clams spawning earlier. Eggs and sperm are shed directly into the water column. Settlement to the bottom occurs after 19 to 35 days, depending on the temperature. Relationships between age/size, functional maturity and effective fecundity have not been precisely quantified.

There are two subspecies of Atlantic surfclam: The offshore subspecies *Spisula solidissima solidissima*, to which this assessment refers, and the smaller coastal subspecies (*Spisula solidissima similis*) that occupies relatively southern inshore habitats (Weinberg et al 2010). The geographic distributions of the two subspecies overlap to a limited extent in the south and in some inshore waters to the north. However, *S. s. similis* is reproductively isolated from *S. s. solidissima* and not important to the federal commercial fishery. It is likely that all *Spisula solidissima similis* along the northeast coast belong to the same biological population.

See Cargnelli et al. (1999) for a more detailed review of life history and distributional information.

Management

Surfclams are common in both state waters (3 miles or less from shore) and federal waters (the Exclusive Economic Zone - EEZ, between 3 and 200 miles from shore). This stock assessment applies only to the segment of the surfclam population in federal waters because the EEZ is the management unit specified in the Atlantic Surfclam Fishery Management Plan (FMP). Surfclams in New Jersey and New York state waters support valuable fisheries that are managed by state authorities. The state of the inshore portion of the resource is discussed in Appendix A1.

Atlantic surfclams in the US Exclusive Economic Zone (EEZ) are considered a single stock for management purposes, though state and federal stocks are not biologically distinguishable. There are, however, substantial regional differences in biological properties and population dynamics.

Because the surfclam fishery is highly localized and the resource is sedentary, stock conditions are often described for regions, rather than the whole stock area. Names and abbreviations for the stock assessment regions are listed from south to north below (and see Figure A1)
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Abbreviation	Assessment region name
SVA	Southern Virginia and South Carolina
DMV	Delmarva
NJ	New Jersey
LI	Long Island
SNE	Southern New England
GBK	Georges Bank

The southern area consists of the regions from SVA to SNE, excluding only GBK (Figure A2). SVA is at the southern end of the species range and of relatively little importance to the stock as whole.

Georges Bank was closed to surfclam harvesting between 1989 and 2009 due to the presence of paralytic shellfish poisoning (PSP) toxins in surfclam meats. With the recent development of fast, accurate tests for these toxins, fishermen have been able to test catches at sea and determine if they are safe for consumption. Since 2009, limited fishing on GBK has been allowed under an exempted fishing permit for the purposes of testing the PSP safety protocols developed by industry. GBK is open for fishing as of January 1, 2013, contingent on continuous testing and the absence of PSP.

The fisheries for Atlantic surfclams and ocean quahogs (*Arctica islandica*) in the EEZ are unique in being the first US fisheries managed under an individual transferable quota (ITQ) system. ITQ management was established during 1990 by the Mid-Atlantic Fishery Management Council under Amendment 8 to the Fishery Management Plan for the Atlantic Surfclam and Ocean Quahog Fisheries (FMP). Management measures include an annual quota for EEZ waters and mandatory logbooks that describe each fishing trip to a spatial resolution of at least one ten-minute square (TMS, 10’ lat. by 10’ longitude).

Murawski and Serchuk (1989) and Serchuk and Murawski (1997) provide detailed information about the history and operation of the fishery.

Previous assessments

Stock assessments are generally done after NMFS clam surveys, which are conducted every 2-3 years. Surfclams were previously assessed in 1992, 1994, 1997, 1999, 2003, 2005, and 2008 (NEFSC 1993, 1995, 1998, 2000, 2003, 2007, 2010). The most recent stock assessment for surfclams, NEFSC (2010) concluded that the stock was above the biomass threshold (the stock was not overfished) and that fishing mortality was below the overfishing threshold (overfishing was not occurring). However, biomass was projected to decline gradually through 2014, because recent recruitment had been low and was likely to remain low over the next five years. The uncertainty of these predictions was high due to uncertainty regarding future conditions. A “historical retrospective” analysis in this assessment includes biomass and fishing mortality estimates from previous assessments.

During the NEFSC clam surveys aboard the *R/V Delaware II*, clams were sampled with a 3.2 ton hydraulic dredge, similar to that used by industry but about half the size. A submersible pump, mounted above the dredge, shot water into the sea bottom just ahead of the 1.5m-wide dredge mouth. Commercial dredges have blades 8-12 feet (2.4-3.7m) wide and higher pressure water jets. These jets of water turn the sea bottom into a fluid, which allows the clams to be captured more easily.

Uncertainty in assessment results and the necessity for additional research on abundance were highlighted by NEFSC (1995) because survey catch rates were anomalously high during the 1994 survey in some regions. The anomalously high catch rates were apparently due to a change in voltage supplied to the pump on the survey dredge towed by the *R/V Delaware II*, which increased capture efficiency. Subsequently, a major effort has been made to monitor and improve understanding of the performance of the dredge used in NMFS clam surveys.

Sensors, first deployed in 1997, are used in clam surveys to monitor the performance of the dredge during each tow. Data collected include ship speed and position, dredge angle, voltage and amperage of electrical current that powers the pump on the dredge, manifold pressure (hydraulic pressure just upstream of the nozzles), water depth and water temperature. The sensor data allow for more accurate estimates of distance towed as well as identification of problematic tows. The dredge has been operated in a consistent fashion using the same survey protocols and gear since 1997. In particular, the criteria used to reject bad tows for trend analysis have not changed. Sensor data are used most extensively in analysis of depletion study data to estimate capture efficiency, and in estimation of efficiency corrected swept area biomass.

Cooperative depletion experiments are an important part of surfclam stock assessments. Depletion studies are conducted in collaboration with academia and the clam industry. An industry vessel fishes repetitively to "deplete" a site where the *R/V Delaware II* has already made a small number of non-overlapping tows. As described below, a spatially explicit statistical model (the “Patch” model, Rago et al., 2006) is used to analyze the depletion study data and estimate surfclam density and capture efficiency for the survey and commercial vessels. This assessment includes analysis of data from four new depletion experiments.

This assessment (also described in NEFSC 2013) estimates fishing mortality and stock biomass with efficiency-corrected swept-area biomass calculators, the KLAMZ model, and Stock Synthesis, the main assessment model.

Commercial Catch (TOR-1)

Commercial landings are reported as meat weights in this assessment for ease in comparison to survey data and in calculations, but were originally recorded in units of industry cages. One cage equals 32 industry bushels, and one industry bushel is assumed to produce 17 lbs or 7.711 kg of usable meats. Landings per unit of fishing effort (LPUE) data are reported in this assessment as landings in bushels per hour fished, based on clam logbook reports. The spatial resolution of the clam logbook reports is usually one ten-minute square.

Unit	Equivalent
1 cage	32 bushels
1 bushel	1.88 ft^3
1 bushel	17 lbs meats
1 bushel	7.71 kg meats

As in previous assessments (NEFSC 2010), for all stock assessment analyses "catch" is defined as the sum of landings, plus 12% of landings, plus discards. The 12% figure accounts for potential incidental mortality of clams in the path of the dredge. It is an upper bound; actual incidental mortality is likely to be lower. Incidental mortality to the total surfclam resource is likely low because the total area fished (e.g. 155 km^2 during 2004) is small relative to the spatial area of the resource (Wallace and Hoff, 2005). The ITQ fishery operates with little or no regulation-induced inefficiency (e.g. area closures, trip limits, size limits, etc.) so that fishing effort and incidental mortality are limited.

Recreational catch is near zero, although small numbers of surfclams are taken recreationally in shallow inshore waters for use as bait. Surfclams are not targeted recreationally for human consumption.

Discard data

Discards were zero during 2008-2011 (since the last assessment). Some discards occurred during 1979-1993 (Table A1). No new information about discards was available for this assessment.

Age and size at recruitment to the fishery

Age at recruitment to the surfclam fishery depends on growth rates which vary geographically. Recruitment appears to occur earlier in northern regions. In previous assessments (and in the KLAMZ model discussed in this assessment), commercial selectivity was assumed be knife-edged at 120 mm. Growth curves used in stock assessment modeling (described later) indicate that surfclams reach 120 mm SL and recruit to the fishery at the estimated age of about 6 y south of Georges Bank where most fishing occurs (Figure A2). The age at recruitment depends on the area being modeled (north vs. south), the time period in question, as growth may change over time. Size at recruitment depends on the fishery selectivity estimated in the model. This issue is discussed in detail in the section describing stock assessment modeling (TOR 4).

Landings, fishing effort and prices

Landings and fishing effort data for 1982-2011 were from mandatory logbooks (similar but more detailed than Vessel Trip Reports used in the groundfish fishery) with information on the location, duration and landings of each trip. Data for earlier years were from NEFSC (2003) and MAFMC (2006).

Landings data from surfclam logbooks are considered accurate in comparison to other fisheries because of the ITQ system. However, effort data are not reliable for 1985-1990 due to regulations that restricted the duration of fishing to 6 hours. Effort data are reliable for years before 1985 and after 1990.

Surfclam landings were mostly from the US EEZ during 1965 to 2011 (Table A2 and Figure A3). EEZ landings peaked during 1973-1974 at about 33 thousand mt, and fell dramatically during the late 1970s and early 1980s before stabilizing beginning in about 1985. The ITQ system was implemented in 1990. EEZ landings were relatively stable and varied between 18 and 25 thousand mt during 1985 to 2011. Landings have not reached the quota of 26,218 mt since it was set in 2004 because of limited markets. The quotas themselves are set at levels much lower than might be permitted under the FMP.

The bulk of EEZ landings were from the DMV region during 1979-1980. After 1980, the bulk of landings were from the NJ region (Table A3 and Figure A4). During recent years, EEZ landings from the NJ region have been about 64% of the total, DMV about 8%, and LI and SNE combined about 16%. Landings from LI were modest but appreciable starting in 2001. Landings from SNE were modest but appreciable starting in 2004. Recent LI and SNE landings reflect the tendency of the fishery to move north towards lightly fished areas where catch rates were higher. Landings from GBK were 13% of the total in 2011. Only three vessels were allowed to fish there, and were under the restrictions of an Experimental Fishing Permit. The high proportion of landings on GBK reflects the high catch rates there (see below).

Fishing effort has increased substantially since 1999, particularly in the DMV and NJ regions (Table A4 and Figure A5). The bulk of the fishing effort is in areas where the majority of landings come from. Fishing effort, however, has been increasing in the DMV and NJ regions as the LPUE has declined (see below).

Nominal ex-vessel prices for the inshore and EEZ fisheries have been stable, fluctuating around $9 to $11 per bushel since the mid-1990s (Table A5 and Figure A6). Ex-vessel prices (1991 dollars) decreased steadily in real terms from about $9 per bushel during the mid-1990s to less than $6.50 per bushel during 2008, before stabilizing at approximately $6.80 between 2009 and 2011. Nominal revenues for surfclam during 2011 were about $29 million, making the ITQ surfclam fishery one of the most valuable single species fisheries in the US. In 2011, the ITQ component accounted for 93% of total landings and revenues (Figure A3).

Landings per unit effort (LPUE)

Nominal landings per unit effort (LPUE) based on logbook data was computed as total landings divided by total fishing effort for all vessels and all trips (Table A6. and Figure A7.). Standardized LPUE was not estimated for this assessment because the data are not used analytically and because NEFSC (2007) showed that nominal and standardized trends were almost identical when standardized trends were estimated in separate general linear models for each region with vessel and year effects.

Nominal LPUE has been declining steadily across all regions (except GBK) since 2000. LPUE levels in, NJ, LI and SNE have been at or near record lows, falling to an estimated 41 to 44 bushels per hour in 2011. The only region aside from GBK showing a recent increase in LPUE is DMV which increased from 49 to 60 bushels per hour between 2010 and 2011. LPUE in GBK reached 352 bushels per hour in 2010 and 285 bushels per hour in 2011.

LPUE is not an ideal measure of fishable biomass trends for sessile and patchy stocks like surfclams because fishermen target high density beds and change their operations to maintain relatively high catch rates as stock biomass declines (Hillborn and Walters 1992). However, trends in LPUE and NEFSC clam survey biomass data are highly correlated for DMV and NJ where fishing has been heaviest and fishing grounds are widespread (NEFSC 2010).

Spatial patterns in fishery data

Annual landings, fishing effort and LPUE were calculated by ten-minute square (TMS) from 1979-2011 (Appendix A2) and mean landings, fishing effort and LPUE were calculated by TMS for five time periods: 1980-1990, 1991-1995, 1996-2000, 2001-2005 and 2006-2011 (Figures A8 – A10). Only TMS where more than ten bu of surfclams (estimated by weight) were caught over the time period were included in the maps. TMS with reported landings less than 10 bu were probably in error, or from just a few exploratory tows. Inclusion of TMS, with less than 10 bu distorted the graphical presentations because the area fished appeared unrealistically large.

Figures A8 – A10 show the spatial patterns of the surfclam fishery over the past 32 years. In all the years, the greatest concentration of fishing effort and landings occurred in the same thirty or so TMS in the NJ region, with intermittent fishing activity in other regions. For example, during the first ten-year time period, from 1981 to1990, the highest landings and fishing effort were still concentrated off NJ, but there were some landings and fishing effort mostly offshore in DMV and SVA, and some fishing activity in SNE off of Martha's Vineyard (about 41^{o}N 70^{o}W). During 1996-2000, there were little landings or effort in SVA or SNE, reduced activity in DMV, and increased activity in NJ with expansion to offshore regions. During 2001-2005, fishing effort in DMV increased and fishing effort expanded eastward along the south shore of Long Island. During 2006-2011, some landings came from a small offshore area in DMV, and fishing north of NJ has been mostly limited to the waters adjacent to Long Island and the experimental fishing on GBK.

TMS with the highest LPUE levels over time have been mostly in the NJ and DMV regions with irregular contributions from GBK and the Nantucket Shoals region of SNE. The exception is DMV during 2006-2011, where LPUE is noticeably lower.

Important TMS

TMS “important” to the fishery were identified by choosing the 10 TMS from with the highest mean landings during each of the following time periods 1980-1990, 1991-1995, 1996-2000, 2001-2005 and 2006-2011. For example, a TMS important during 1991-1995 could be selected regardless of its importance during earlier or later time periods. The list contains a total of 28 important TMS, because of overlap between the time periods and because the same TMS tend to remain important. The large majority of important TMS were in the NJ region (18), with 6 in the DMV region, 2 in SNE 1 in GBK. LI and SVA did not qualify in any of the time periods we examined. These plots are complicated by the “rule of three” which states that fine scale fishing location data cannot be shown for areas fished by three or fewer vessels due to confidentiality concerns. Therefore, some otherwise important TMS cannot be depicted here because they were fished by a small number of vessels. Trends in landings, effort and LPUE were plotted (Figures A11 – A13) for each TMS to show changes in conditions over time within individual TMS.

Landings and especially effort have increased recently in one TMS in the DMV region that has historically been lightly fished, but trends show most of the important TMS in the DMV region have seen declining effort and landings over time. Several have not had any reported landings in recent years. Landings and effort have increased in two important TMS in NJ and two in SNE, and appear to be increasing recently (although they are still at low levels) in one of the two NJ TMS that have continuously supported the highest landings in the region for the last 30 years.

With the exception of GBK, there are very few important ten-minute squares in which the LPUE has trended upwards in recent years, if they are still being fished. Most are currently at or below about 100 bushels per hour.

Fishery length composition

Since 1982, port samplers have routinely collected shell length measurements from ~30 random landed surfclams from selected fishing trips each year (Table A7.). During 1982-1986, length data were collected from over 5,000 clams in each of the DMV and NJ regions, where most surfclams are landed. Since 1986 an average of about 1000 lengths from DMV and 1500 from NJ have been collected each year. Surfclams were measured from SNE landings every year from 1982 to 1990, although in small numbers with a maximum of 810 in 1988. Samplers began collecting from SNE once again in 2010 and collected over 2000 lengths in 2011. Port samplers began taking measurements from landings from the LI region in 2003 and have been collecting them consistently ever since, but only about 400 lengths are measured per year on average.

Port sample length frequency data from the four regions show modest variation in size of landed surfclams over time (Figures A14 – A18). Surfclams from the SNE region are larger than surfclams from more southern areas. Care should be taken in interpreting these due to small sample sizes in some cases (especially LI and SNE), but in general the data indicate that most landed surfclams have been larger than 120mm SL, with the distribution of sizes being wider some years than others on both ends of the distribution. Commercial size distributions are discussed in detail in the SS3 model section (see below).

NEFSC and Cooperative clam surveys (TOR-2)

Survey data used in this assessment were from NEFSC clam surveys conducted during 1982-2011 by the *R/V Delaware II* during summer (June-July), using a standard NEFSC survey hydraulic dredge with a submersible pump. The survey dredge had a 152 cm (60 in) blade and 5.08 cm (2 in) mesh liner to retain small individuals of the two target species (surfclams and ocean quahogs). The survey dredge differed from commercial dredges because it was smaller (5 ft instead of 8-12.5 ft blade), had the small mesh liner, and because the pump was mounted on the dredge instead of the deck of the vessel. The survey dredge was useful for surfclams as small as 50 mm SL (size selectivity described below). Changes in ship construction, winch design, winch speed and pump voltage that may have affected survey dredge efficiency were summarized in Table A7 of NEFSC (2004). Each of these factors has been constant since the 2002 survey.

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

NEFSC clam surveys are organized around NEFSC shellfish strata and stock assessment regions (Figure A1). Most surfclam landings originate from areas covered by the survey. The survey did not cover Georges Bank (GBK) during 2005 and provided marginal coverage in 1982, 1983, and 1984. Individual strata in other areas were sometimes missed. Strata and regions not sampled during a particular survey were “filled” for assessment purposes by borrowing data from the same stratum in the previous and/or next survey, if these data were available (Table A8.). Survey data were never borrowed from surveys behind the previous, or beyond the next survey. Despite research recommendations, a model based approach to filling survey holes has not yet been adopted. A model-based imputation was investigated for this assessment, but the imputation tended to over-emphasize unsampled years and areas. Alternative approaches to imputing missing strata remain a possibility but were not further pursued in this assessment.

Surveys follow a stratified random sampling design, allocating a pre-determined number of tows to each stratum. A standard tow is nominally 0.125 nm (232 m) in length (i.e. 5 minutes long at a speed of 1.5 knots) although sensor data used on surveys since 1997 show that tow distance increases with depth, varies between surveys and is typically longer than 0.125 nm (Weinberg et al., 2002). For trend analysis, changes in tow distance with depth were ignored and survey catches were adjusted to a standard tow distance of 1.5 nm based on ship's speed and tow start/ stop times recorded on the bridge.

Stations used to measure trends in surfclam abundance were either random or "nearly" random. The few nearly random tows were added in some previous surveys in a quasi-random fashion to ensure that important areas were sampled. This generally occurred when stake holders or the assessment lead wished to increase sampling intensity in a stratum of particular interest. Stations added this way were different from other random stations in that they deviated from the pre-determined sampling design described above. They were otherwise random with respect to location within a stratum and thus are called "quasi random". Other non-random stations are occupied for a variety of purposes (e.g. depletion experiments) but not used to estimate trends in abundance.

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

Following almost all survey tows, all Atlantic surfclams in the survey dredge were counted and shell length was measured to the nearest mm. A few very large catches were subsampled. Mean meat weight (kg) per tow was computed with shell length-meat weight (SLMW) equations (updated in this assessment) based on fresh meat weight samples obtained during the 1997-2011 surveys (see below).

Locations and catches of all stations in the 2011 survey have been mapped (Figure A19.) and maps for previous surveys can be found in Appendix A3.

Survey tow distance and gear performance based on sensor data

There are some applications where it is desirable to know the tow distance with more certainty than is provided using the nominal tow distance. Beginning with the 1997 survey, sensors were used to monitor depth (ambient pressure), differential pressure (the difference in pressure between the interior of the pump manifold and the ambient environment at fishing depth), voltage, frequency (hertz) and amperage of power supplied to the dredge, x-tilt (port- starboard angle, or roll), y-tilt (fore-aft angle, or pitch) and ambient temperature during survey fishing operations. At the same time, sensors on board the ship monitor electrical frequency, GPS position, vessel bearing and vessel speed. Most of the sensor data are averaged and recorded at 1 second intervals. These metrics of tow performance can be used to accurately gauge the true distance fished by the dredge.

Analysis of sensor data from the 2011 NEFSC survey

The survey sensor package (SSP) was deployed on the NEFSC clam survey dredge during the 2011 survey. The SSP provided differential pressure measurements on 187 out of 430 total tows. On other tows (generally between tows 161 and 371) the SSP did not function properly. Back up sensors (Vemco Minilog depth/temperature recorders) failed to produce useful information due a gradual calibration drift that overlapped the period during which no SSP data was recorded. Because the shift in baseline pressure was systematic and began at an unknown point, no data from the Minilog recorders was used. Electric current supplied to the pump on the survey dredge was successfully logged for every tow (Figure A20).

A predictive relationship exists between the electric current supplied to the dredge and the differential pressure in the dredge pump manifold (Figure A21). This relationship was explored in the previous assessment (NEFSC 2009). The previous assessment provided a tolerance point for minimum differential pressure of 35 PSI based on analysis of dredge operation (NEFSC 2009). The current approach maintains that minimum tolerance but does not use the previous upper bound for differential pressure (40 PSI), because pump pressure was generally higher in 2011 (Figure A22).

The parameters estimated in 2009 do not provide a good fit to the data from the 2011 survey. It is likely that the operating specifications have changed somewhat due to alterations in procedure and equipment. For example, the dredge pump was rebuilt and the electrical supply line was replaced after the 2009 survey. These pieces of equipment will have slightly different properties from those used in 2009, and thus produce a subtly different relationship between current and differential pressure.

We compared four different models for predicting differential pressure from current supplied to the pump. We used only current measured while the dredge was fishing (fishing seconds - see below). Current was the smoothed mean (7 second moving average) of three different amperage meters on the research vessel. Our models were fit to the smoothed (7 second moving average) differential pressure recorded by the SSP for the 187 tows where it functioned (Figure A21). The models tested were: a simple power function (M1), the model fit to the data from 2009 (M2), a cubic spline (M3) and a Loess spline (M4, Figure A23). Model selection was based on the models ability to correctly distinguish the tows with SSP data in which differential pressure that was above or below tolerance (35 PSI). Predicted differential pressure was plotted against observed values. Where predicted and observed values were together above or below the tolerance line, the model was considered to have segregated correctly. When the predicted and observed values did not agree on whether or not the differential pressure was above 35 PSI, the model failed to segregate correctly. The cubic spline model produced the highest percentage of correctly segregated points (Figure A24).

The cubic spline fit was then used to predict the differential pressure for all tows, including those for which we measured differential pressure. If the model predicted differential pressure was below 35 PSI for more than 25% of the fishing seconds that tow was considered a "bad" and not used in this assessment for calculating swept area abundance or biomass from surveys since 1997 (Table A9). These tows were, however, used in conventional trend analysis, unless there was an obvious problem noted by the survey crew, because historical surveys did not have sensors.

<u>Determination of time fishing</u>

The determination of time fishing, the "fishing seconds" for each tow was based on a measurement of the pitch of the dredge during each second of the tow. Pitch was recorded by two different instruments: the SSP, which functioned intermittently, and a Star Oddi inclinometer which functioned consistently. Data from each instrument was smoothed using a 7 second moving average and then parsed for time below the "critical angle".

The choice of critical angle has implications for the calculation of tow distance for each tow. When the dredge is above the critical angle it is assumed to be pitched too steeply for the blade to penetrate the sediment. If the dredge is pitched below the critical angle, it assumed to be near enough to horizontal that the blade should penetrate and thus be actively fishing.

An ideal critical angle is as close to zero as possible. When the dredge is bouncing over rough terrain it is unlikely to be fishing effectively and those seconds should be excluded. There is however, a certain amount of pitch that is within fishing tolerance and a certain amount of noise in the data. If the critical angle is too small, many seconds when the dredge was actually fishing would be excluded, which would tend to bias estimates of tow distance down. It is therefore important to find a critical angle for tow distance that is neither too small, nor too large.

The critical angle in the last assessment was 5.16 degrees, a value chosen because it represents a blade penetration of 1 inch (in.) on level ground. Our examination of the sensor data from 2011 provided no compelling reason to use a different critical angle (Figure A25). That is, shifting the critical angle upwards produced only slightly longer tows on average and this shift was not sufficient to trigger a reconsideration of the mechanically derived, blade penetration based estimate, used previously. Therefore the critical angle used in the current assessment was also 5.16 degrees.

NEFSC clam survey trends and size composition

NEFSC clam survey data (Table A10.) were tabulated for small (50-119 mm SL, Figure A26.) and large (120+ mm SL, Figure A27) surfclams by year, region and for the entire stock. Only trends in mean numbers per tow were plotted because trends in mean kg per tow were similar. Approximate asymmetric 95% confidence intervals were based on the CV for stratified means and assume that the means were log normally distributed.

Survey trends for small surfclams (Figure A26.) show low recruitment levels during recent years in the Delmarva (DMV) and New Jersey (NJ) regions, approximately average recent recruitment levels in Southern Virginia (SVA), and Southern New England (SNE), high recruitment levels in Long Island (LI) and low recruitment in GBK. Recruitment appears to be increasing in SVA, LI, and possibly DMV. Survey trends for fishable (120+mm) surfclams (Figure A27.) show low abundance in the SVA, DMV and NJ region during recent years. In comparison, the other regions are either increasing (GBK and possibly LI) or variable (SNE). Based on survey data for the entire stock, recruitment was increasing, but fishable abundance was slightly below average during 2011 (Figures A28 – A29).

Shell length composition data (Figure A30.) are compatible with patterns in trend data. In particular, abundance and recruitment appear low in the southern DMV and NJ regions while abundance is higher and recruitment is at near average levels in the northern LI, SNE and GBK regions.

NEFSC survey age composition

Surfclam ages are considered to be reliable and the aging process has been studied in detail (See Appendix A4 NEFSC 2009; Jacobson et al 2006; and http://www.nefsc.noaa.gov/fbp/QA-QC/data/surfclam/).

In this assessment, "recognizable" recruitment events are year classes that are strong enough to be detected by visual examination. "Strong" recruitment events are year classes that are obviously large relative to other years.

Survey age-length keys and stratified mean length composition data were used to estimate the age composition of surfclams in NEFSC clam survey catches and the stock as a whole by year and region. Age composition was estimated for the years between 1982 and 2011when surveys occurred. Ages ranged from 1-37 (Figures A31 – A36). Specific year classes and trends in age composition are discussed in the context of the assessment model (see TOR 4).

Dredge efficiency

Estimation of dredge efficiency is based primarily on the results of depletion experiments conducted with industry and academic collaborators aboard commercial vessels (NEFSC 2009). In 2011 additional depletion experiments were carried out aboard the *FV Pursuit* (see below). Procedures for estimating dredge efficiencies were modified considerably for this assessment based on Hennen et al (2011) and the incorporation of previously unrecognized uncertainty.

Dredge position during depletion experiments was approximated by vessel position, which was measured via GPS every one second. The true start and stop times for a tow were determined using a Star Oddi inclinometer mounted on the dredge which recorded the angle of the dredge every 1 second. The inclinometer data were smoothed with a 7 second moving average. The dredge was assumed to be fishing when the smoothed dredge angle was less than a_{crit} degrees and the dredge was assumed not fishing when the smoothed inclinometer subsequently increased to an angle greater than a_{crit} degrees. The value a_{crit} was determined by testing critical angles between 2 and 12 degrees and comparing the total tow distance and average tow distance across all depletion experiments (Figure A37). There was an asymptote at angles greater than 8 degrees. That is, total tow distance and average tow distance did not change appreciably with any critical angle between 8 and 12 degrees. We selected 10 degrees as a critical angle. The time stamps for the true start and stop times were used to determine the vessel position during the tow. These data were smoothed with a loess spline (span =0.75, degree = 2) to both longitude and latitude. The choice of smoothing algorithm did not make appreciable differences in the total tow distance across depletion experiments or in the average distance per tow within an experiment (Figure A38). The smoothed vessel positions were used in the patch model to determine tow paths.

The previous assessment (NEFSC 2009) used an estimator for survey dredge capture efficiency that was based on the ratio of observed density in the "set up tows" with the density estimate derived from depletion experiments conducted at the same site. Set up tows were conducted aboard the *RV Delaware II* using the survey dredge described above. They were 5 parallel tows evenly spaced over 1 km at the sites selected for depletion experiments. The set up tows were oriented perpendicularly to the expected direction of depletion tows. The estimator was:

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

where *e* is estimated survey efficiency, *d* is the observed density in setup tows and *D* is the estimated depletion experiment density. The implicit assumption of this analysis is that *d* and *D* are estimating the same true density. The estimated survey efficiency used for several calculations in this assessment was the median of all the usable depletion experiments (NEFSC 2009).

Survey dredge efficiency has been difficult to estimate with reasonable precision. It is likely that dredge efficiency is affected by local conditions such as substrate properties, currents and wind. It may be highly variable from site to site. We found that although the quantity *d* was reasonably stable from site to site it carried a high variance (Figure A39.) relative to the quantity *D*. This variance was ignored in previous assessments. Uncertainty in d was carried into the estimate of *e* in this assessment.

We considered a suite of independent variables that might provide additional information about *e*. In 2008, a series of repeat tows were conducted using survey gear in the same location towed previously by the NMFS survey (NEFSC 2009). These "repeat stations" thus provide information about the ability of the survey gear to capture clams when compared to commercial gear. The commercial gear has relatively well understood selectivity. The density observed in the commercial gear was scaled to approximate true density, using its estimated selectivity curve $D_L = \frac{D_{L(obs)}}{Slx_L}$. Thus the observed catch in the survey dredge divided by the rescaled catch in the commercial dredge provided a second measure of survey dredge efficiency.

The selectivity stations (described below) were also a potential source of information on survey dredge efficiency. At selectivity stations, the observed survey density was compared to the rescaled (see above) commercial catch at the same site.

The data from these three sources were truncated. All values larger than 1.0 were discarded due to implausibility (catch in the survey dredge must be less than or equal to the total number of available clams). All sites where 0 clams were caught were not used based on the assumption that if clams were available, the gear would catch at least one of them during a 5 minute tow.

The resulting estimates of survey dredge efficiency from all of these sources of information together provide the set of prior knowledge on survey dredge efficiency (Figure A40.). Each individual estimate has an associated CV. For the depletion sites the CV was estimated directly from the numerical estimation procedure used to fit the Patch model. For the repeat and selectivity sites the CV was based on the pure error variance derived from the set of combined estimates. These values were bootstrapped 100000 times using a weighted bootstrap procedure in which the weights were proportional to the inverse CV associated with each estimate. A bounded (0,1) log normal prior distribution was fit to the bootstrapped data set (Figure A41.). The mean and CV of the log normal distribution were 0.234 and 1.32, respectively. The log normal distribution described by these parameters was the prior distribution for survey *q* used in the assessment models. The mean is similar to the estimate of survey dredge efficiency used in the last assessment (0.256), though the CV is considerably larger when compared to the previous value (0.13).

New Depletion Experiments

The 2011 depletion experiments were analyzed using standard Patch methodology with one exception. We employed a new method for calculating the hit matrix (Hennen et al, 2011). Three of the four SC depletion experiments worked well. Estimated densities ranged from 0.184 – 0.416 clams per m^2 (Table A11). Estimated efficiencies ranged from 0.556 – 0.738. These values are similar to values from previous assessments.

Maps of the tow sequences from the depletion plots show thorough coverage of study sites with high degrees of overlap between tows, which follows procedures recommended by (Hennen et al, 2011) (Figure A42). Recommended patch model diagnostics include examining the catch vs. expected catch, the catch per unit of effective area and the likelihood residuals (Figure A43-A46). We generated likelihood profiles for each of the three estimated parameters for each experiment (Figure A47-A49). The confidence intervals shown in Table 1 are based on the likelihood profiles.

The one depletion study that did not produce reasonable estimates (SC11-04) suffered from a very low catch in the 13th tow of the depletion sequence. Altering this value toward the expected catch changes the Patch model results to estimated values that closely agree with results from the other three SC depletion experiments. We examined all the available logs for tow 13 and found no errors. Inclinometer and pressure sensors did not indicate any mechanical problems during this tow and the tow was of normal length. In short there was no *a priori* reason to exclude this tow from the depletion sequence.

Size selectivity

Survey dredge selectivity was previously calculated using Millar's (1992) SELECT model and precision was estimated using Miller's beta-binomial model (NEFSC 2009). Selectivity was estimated for this assessment using a generalized linear mixed model (Pinheiro and Bates 2000). The data were collected by the *R/V Delaware II* and *F/V Pursuit* during cooperative selectivity experiments in 2008 and 2011. Data from the experiments were used to estimate size-selectivity for the NEFSC clam survey dredge which is used on the *R/V Delaware II*. The data were also used to estimate size selectivity for the commercial dredge used by the *F/V Pursuit* when repeating NEFSC 2008 and 2011 clam survey stations. The commercial dredge was configured for survey operations, rather than commercial fishing operations. Thus, the size selectivity estimates for the commercial dredge used by the *F/V Pursuit* during cooperative survey work are not applicable to commercial catch data. They may be useful, however, in anticipating the size selectivity of commercial dredges configured for use in cooperative surveys.

As described below, the size selectivity experiments analyzed for this assessment had a paired-tow design, because the tows were conducted in the same general area. R/V and F/V stations more than 300 m apart based on GPS position data were not used.

The data available for each selectivity study site included shell length data from: one R/V tow; one F/V repeat tow with the modified commercial dredge; and one F/V selectivity tow with a commercial dredge lined with wire mesh.

The *F/V Pursuit* has two dredges, each 12.5 feet (3.8 m) wide, which are towed separately. The knives on both dredges were set at 5.25 inches (13.3 cm) for surfclam cooperative survey operations. The starboard dredge used for F/V selectivity tows was lined with 1-inch (2.54 cm) hexagonal wire mesh to maximize retention of small surfclams.

After F/V repeat tows, the catch was dumped into the port or starboard hoppers and then moved mechanically onto a larger, centralized belt to a shaker table and then onto a sorting belt where sampling occurred following F/V repeat tows. The large belt before the shaker table was about 4 feet (1.2 m) wide and 10 feet (3 m) long. Alongside the belt was a large metal stand where the catch could be sampled before it reached the shaker table where mechanical sorting occurred. The average spacing between the rolling bars on the shaker table was 0.73 (+/- 0.10) inches which was narrower than during normal commercial operations.

Surfclams were measured to the nearest mm. F/V repeat tows used the port (unlined) commercial dredge. R/V and F/V repeat tows were 5-minutes in duration. F/V repeat tow catches were allowed to run over the shaker table and onto the sorting belt in the normal fashion before sampling, to measure the effects of both the dredge and shaker table on shell length data. The entire catch was measured following R/V tows following standard survey protocols. The number of bushels was counted for F/V tows and a subsample of three full bushels was measured.

For F/V selectivity tows, the lined dredge was towed for 45 seconds along a track adjacent to the F/V repeat tow. The catch was sorted before going over the shaker table to avoid loss of small surfclams due to mechanical sorting on deck. All clams in three full bushel samples were measured to the nearest mm. Inclinometer data used elsewhere to measure area swept were not available for F/V selectivity tows with the lined dredge. Positions were measured at the start and stop of each selectivity tow by GPS.

Shell length data from selectivity experiments were tabulated using 1 mm shell length size groups. Survey size selectivity was estimated using data from R/V (survey and repeat) tows and FV selectivity data from 40 total sites (10 mm bin summaries in Table A12 – A13).

Previous selectivity estimates

In the last assessment, the Invertebrate Subcommittee decided that the dome shaped curve was the best estimate of size selectivity for the NEFSC survey dredge (NEFSC 2009). Beta-binomial confidence intervals suggested that the domed shaped pattern was real although most of the evidence was based on only two SL groups (160 and 170 mm SL).

The dome shaped size selectivity curve seems biologically plausible. Large surfclams (150+ mm SL) have long siphons and live deeper in the sediments. They may be difficult to dislodge using the light survey dredge with relatively low pressure at the nozzles (about 40 psi compared to about 80 - 120 psi on a commercial dredge).

The selectivity experiments conducted in 2011 were designed to address questions about the appropriateness of a domed shape selectivity curve.

Current selectivity estimates

All R/V and F/V data were combined so that there was a single set of R/V, F/V repeat and F/V selectivity data (Table A12.; Figure A50.).

Selectivity was modeled as a generalized additive model (GAM) where the shell length bin was a factor, predicting the binomial proportion of the survey catch over the total catch (R/V + F/V).

$$p_L = e^{a+s(L)+s(sta)+offset(s.a.ratio)}$$

Where p_L is the binomial proportion (logit link) estimated for shell length L with intercept α and vector of model terms evaluated over L. The *s()* terms indicate a spline over the indicated variables, in this case shell length (L) and a random effect due to station and year. The final term is an offset (MacCullagh and Nelder, 1989) based on the ratio of swept areas between the respective tows at each station. For example, at station 7 the lined dredge swept 242.4 m^2 while the research dredge was towed 318.2 m^2 (Figure A51). Area swept by each gear is a potential source of bias because clams can be unevenly distributed on the sea floor. The nominal time fished for the lined dredge is 45 s compared to 5 min. for a nominal survey tow. The commercial dredge however, is much larger and is towed at a faster speed, which tends to minimize the differences between the gears in area swept.

Using the GAM methodology allowed greater flexibility in the model, when compared to assuming any particular shape. The basis dimension (*k*) in a spline determines the amount of "wiggle" allowed in the spline. Wood (2009)[1] suggests an objective method for choosing a basis dimension in splines. This method allows the data to determine the shape required to adequately fit them rather than the modeler.

The last assessment assumed a double logistic shape when modeling selectivity (though the fit from the double logistic was contrasted with a logistic fit, which allowed for a comparison of at least two shape families in the model selection process). The double logistic shape is described by a monotonic increase to a peak value, and a subsequent horizontal surface, followed by a monotonic decrease. The current approach estimates a spline along the range shell lengths and thus the peak may occur at any point and multimodal shapes are allowed.

The inclusion of random effects based on station is important because there is a great deal of variation in selectivity between stations. Variation across stations is essentially a nuisance parameter in our assessment because we are interested in the general selectivity over all possible stations, rather than

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[1] See R package mgcv documentation: http://127.0.0.1:19246/library/mgcv/html/choose.k.html

the differences between them. Because we believe that clams taken from a particular place and time would tend to experience similar selectivity when compared to clams taken from a different place and time, it is appropriate to model selectivity using random effects.

Approximate confidence intervals were estimated using

$$CI_L = elogit(\rho_L \pm 1.96 * \sigma_L)$$

Where $CI_L$is the approximate confidence interval for length L, $\rho_L$is the corresponding selectivity estimate, σ_L is its standard error and *elogit* is the inverse of the logit function.

It is clear from the model results (Figure A52) that the domed selectivity curve estimated in the last assessment is appropriate. It is also clear that the domed shape is present in most of stations we sampled (Figure A53.). That is, the dome shape is not driven by data from a single site.

The $\rho_L$estimates were rescaled in some applications so that the highest value was fully selected, that is, equal to 1.0 (Figure A54.). This was necessary because selectivity may be used in product with gear capture efficiency which is defined as the probability of capture (between zero and one) for an organism fully selected by the sampling gear.

Rescaled selectivity was applied to the survey data using the inverse estimated ρ_L as a multiplier for the aggregate animals of each size on each tow. That is, if n_L animals in size class L were caught on a survey tow, we multiplied n_L by $1/\rho_L$, thus n_L/ρ_L rather than n was used to compute the stratified means for the survey index used in the KLAMZ assessment models. The SS3 models estimated selectivity internally and this adjustment to the survey data was not made.

Fishery selectivity

Fishery selectivity experiments were conducted on the F/V Pursuit. A modified fishery dredge (described above) was towed for five minutes as part of the selectivity sequence. The catch by size from this tow was compared to the lined dredge catch at each site. The selectivity estimates for each size class were found using models similar to the ones described above. Data from 2008 was combined with data from 2011. The same model (eq. 1) with offsets based on swept area ratios (Figure A55.) was preferred by AIC. Rescaled fishery selectivity estimates were useful for comparison to internally estimated commercial selectivity from SS3 (Figure A54.).

Shell length, meat weight relationships

The shell length-meat weight (SLMT) relationships are important because they are used to convert numbers of surfclams in survey catches to meat weight equivalents. The survey meat weight equivalents are inputs in the stock assessment models used to estimate stock biomass, which is reported in units of meat weight.

Meat weights for surfclam include all of the soft tissues within the shell. All meat weights greater than 0.5 kg were assumed to be data entry error, and were removed from the analysis.

Generalized linear mixed models (GLMM; Venables & Dichmont 2004) were used to predict clam meat weight, using equations of the form:

$$MW = e^{a+b_0 ln(L)+b_1 ln(c_1)+b_2 ln(c_2)+\cdots+b_n ln(c_n)}$$

where MW was meat weight, L was shell length, $c_1,\ldots,c_n$ were covariate predictors (e.g., region; in the basic model these are absent), and a and the b_i were parameters to be estimated. Examination of the variance of the weights as a function of shell length indicated that weight increased approximately linearly with shell height, implying that the Poisson family was appropriate for the distributions of meat weights (McCullagh & Nelder 1989). The GLMMs in all analyses therefore used the Poisson family with a log link. Because shell length/weight relationships for clams at the same station are likely to be more similar than those at other stations, we considered the sampling station as a grouping factor ("random effect") in the analysis.

We fit models with fixed effects for year and region (Table A14.). Neither of these factors proved to be important using AIC (Table A14). The best model by AIC and BIC was a model with fixed effects for shell length and depth and random effects for shell length slope and the intercept, using both the year and the station as the grouping variables.

$$E(MW) = exp\big(\alpha(1 + r_{sta}) + \beta(lnL + r_{sta}) + \gamma lnD + \delta_{Reg} + \epsilon_{Yr}\big)$$

where $E(MW)$ is the expected meat weight (in g) and r_{sta} is the grouping variable for the random effects (station). The important predictors of meat weight are: ln(length), ln(depth), region and year.

Random effects improved the model fit (i.e., decreased the AIC, Table A14.) in all analyses, demonstrating that individuals at the same sampling site are more similar to each other than to the general population. When multiple samples are collected at each site and random effects are not accounted for, the results typically overstate the precision of parameter estimates. This occurs because the analysis assumes that within-site observations are independent when, in fact, they often are highly correlated.

The GLMM approach also allows specification of the appropriate variance structure of the response variable, while a log-transformed regression implicitly assumes that variance increases with the square of the mean; an assumption that appears incorrect for clam weights.

The curves from (NEFSC 2009) and the current assessment are not substantially different at common commercial meat weights though the current model predicts somewhat heavier meats at small shell lengths and lighter meats at large shell lengths (Figure A56.). The largest observed clam used in the model fitting was 190 mm. The curve for the current assessment was generated using a depth of 33 m, which is the average depth of the survey stations over all years used in the analysis.

Regional differences in meat weight are meaningful, though some of the differences between regions can be explained by the different depths found there (Figure A57.). The largest meats at length, given constant depth were found in Georges Bank, but the largest meats given the depths actually observed in each region were found in Southern New England.

Age and growth

Surfclams in age and growth samples were measured at sea and the shells were retained for aging in the laboratory. Shells for aging were collected based on a length stratified sampling plan. A recent study confirmed that rings on shells collected during the summer clam survey are annuli that can be used to estimate age (NEFSC 2009).

Age and length samples are available for most regions but not from every survey (Table A15). DMV and NJ were the most consistently sampled regions (Table A15). GBK was the least consistently sampled.

Plots of age vs. shell length by year and region (Figures A58 – A62) indicate that growth patterns have been relatively constant in most regions over time with DMV and NJ being notable exceptions. As described in the last assessment (NEFSC 2009), maximum size was lower after 1994 in DMV and NJ.

Von Bertalanffy parameters for growth in shell length were estimated for each region and each survey year for which sufficient data existed (Table A16). The Von Bertalanffy growth curve used in the calculations was:

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

Where L_a is size (meat weight in g or SL in mm) at age a, and L_∞, K and t_0 are Von Bertalanffy parameters (the curves for growth in SL and weight have different parameter values). DMV and NJ have experienced significant declines in L_∞ through time. This result follows from weighted regression of the year specific parameter estimates against time, where the weights were the inverse standard errors of the parameters in question (Figures A63 - 64). NJ has experienced a significant decline in the growth constant K as well, demonstrating that clams in NJ are taking longer to reach a smaller size than they once did (Figure A65). Weighted regressions of parameter estimates in other regions did not indicate any significant trends over time.

Commercial LPUE

Commercial LPUE was not considered an adequate measure of relative abundance for this assessment because of the sessile nature of the species and the corresponding behavior exhibited by fishers. In general clam fishers use a fine spatial scale area until catch rates drop below economically profitable levels. They then move to another location and repeat the process. Thus catch rates tend to remain relatively stable over time even when population abundances fluctuate (See Appendix A2)

Stock Definitions (TOR-3)

Surfclams and ocean quahogs in the US EEZ (federal waters) have been managed as a single stock by the Mid-Atlantic Fishery Management Council for the last 35 years. The inshore portions of the resource off the coast of each state (<3 nm from shore) have been managed independently by state authorities. Two options for defining stocks in the EEZ surfclam resource were evaluated on technical grounds (biology, applicability of MSY reference points, fishing patterns and survey coverage) while excluding policy related considerations. The first (status-quo) option defines a single stock that extends over the entire range of the EEZ resource from Cape Hatteras in the south to the northern edge of Georges Bank. The second option defines two stocks by separating Georges Bank (GBK) from the area to the south along a traditional boundary based on NEFSC shellfish survey (depth) strata lines (Figure A66). The southern area (SNE - SVA) extends from Southern New England (just southwest GBK) in the north to Cape Hatteras in the Southern Virginia/North Carolina region in the south.

This discussion and TOR were triggered by difficulties noted in recent assessments (SARC 49 NEFSC 2010, page 43) and recommendations by SARC reviewers (SARC 49 summary report; NEFSC 2010, pages 9-11). The Invertebrate Working Group did not achieve consensus on this issue and so the decision about which approach is better is left to reviewers. Arguments for and against defining two stocks are presented in Table A17 – A18.

The working group did agree on a shared working definition of a stock for use in its deliberations. The definition, extracted from the NOAA Fisheries Glossary (Blackhart, et al. 2006; *http://www.st.nmfs.gov/st4/documents/F_Glossary.pdf*), reads:

> *A part of a fish population usually with a particular migration pattern, specific spawning grounds, and subject to a distinct fishery. A fish stock may be treated as a total or a spawning stock. Total stock refers to both juveniles and adults, either in numbers or by weight, while spawning stock refers to the numbers or weight of individuals that are old enough to reproduce.*[6]
>
> *Comment: In theory, a unit stock is composed of all the individual fish in an area that are part of the same reproductive process. It is self-contained, with no emigration or immigration of individuals from or to the stock. On practical grounds, however, a fraction of the unit stock is considered a "stock" for management purposes (or a management unit), as long as the results of the assessments and management remain close enough to what they would be on the unit stock.*[5]
>
> [5] *United Nations Food and Agricultural Organization. Fisheries Glossary.* http://www.fao.org/fi /glossary/default.asp
>
> [6] *Northeast Fisheries Science Center. Definition of Fisheries Technical Terms. http://www.nefsc.noaa.gov/techniques/tech_terms.html*

Some recent developments in the fishery are relevant. The GBK region was closed to fishing due to risk of PSP contamination in 1990 and is nearly virgin. The fishing industry developed protocols during 2008-2011 for determining if PSP is present prior to fishing and subsequent laboratory testing once clams from GBK are landed. The protocols were tested during experimental fishing on

GBK during 2011 and 2012 and have been approved. GBK will open for fishing by all permitted vessels during 2013. Industry sources expect landings from the GBK region will amount to about1 million bu per year (about 1/3 of recent landings) over the next few years.

Fishing on GBK involves long (multiday) trips by a small number of vessels (currently 3) which are substantially larger than the rest of the fleet, capable of fishing with two large dredges simultaneously and generally able to work under rough conditions. In contrast, smaller boats make day trips with a single and often smaller dredge in southern regions. The surfclam resource is believed to be lightly exploited.

Abundance has trended down in the south and up on GBK due to environmental effects but is near its target biomass as a whole. Under either the current or alternative stock definitions, surfclams are not likely to be overfished, nor is overfishing likely to be occurring.

Assessment model results (TOR 4)

Stock Synthesis (SS3[2]) replaced KLAMZ (Appendix A4) as the primary model in this assessment (Methot, in press). SS3 was preferable because it made better use of survey age data in estimating recruitment and in making forecasts. In addition, the SS3 model was more flexible and capable of handling multiple assessment areas as might be needed in future. SS3 models for surfclam were explored in the previous assessment, but the KLAMZ model was used to provide management advice (Appendix 2 in NEFSC 2010). KLAMZ models were updated for this assessment, and discussion and results, including the bridge to the current assessment, are available in Appendix A5.

Separate SS3 models were developed for surfclams in the southern and GBK areas. No final SS3 model is available for the combined southern plus GBK region assumed in KLAMZ models and previous assessments. Preliminary models that combined the two areas with no internal spatial subdivision were developed but abandoned after a great deal of work. Divergent population dynamics (i.e. different biomass and mortality trends, changes in proportion of total biomass in the two areas over time, very limited fishing on GBK, and differences in occurrence of strong year classes) made it too difficult to estimate “average” population dynamics for the areas combined. Also, data were lost when the areas were combined because surveys were not available for the entire combined assessment region in some years. In this assessment, biomass, fishing mortality, recruitment and other estimates for the combined regions were estimated by combining estimates for the southern and GBK areas.

Fishery and survey selectivity were functions of size rather than age in SS3 models (Table A20). Conditional ages at length data, rather than traditional age composition data, were used in fitting models. The conditional age vector with elements $n_{t,a,L}$ for example, gives the proportion or number of observed ages (a) from samples of length L in year t of the NEFSC clam survey. The major advantage of the conditional approach is that more information about growth (including variance in size at age) and yearclass strength is preserved. Size compostion data are not used twice (once as size composition data and once in calculation of traditional catch at age). Finally, the sampling distribution of condtional age data is probably easier and more accurately characterized as a multinomial conditional on the number of ages $n_{t,L}$ actually sampled. The traditional type of age data was included in the model for

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[2] Stock Synthesis Model version SS-V3.24f compiled for 64-bit linux.

qualitative for use in evaluating goodness of fit and recruitment patterns. Traditional age composition data had no effect on model estimates.

The SS3 models for surfclams were more complex than KLAMZ, but relatively simple compared with many other SS3 models. We estimated fewer parameters relative to other models for many other species because NEFSC clam surveys are carried out every three years, the fishery is relatively uncomplicated, and because no other survey data were available (Table A20-A21). Simple approaches with relatively few parameters increased model stability, and aligned with the philosophy of KLAMZ models used in previous surfclam assessments. The same types of data were available for both areas, although more precise and numerous data were available for the southern area (Figures A68 – A69). The additional data for the south made it possible to estimate additional catchability and selectivity parameters, as well as biomass and mortality over a longer time period. It was necessary to borrow these parameter estimates from the south in modeling surfclams on GBK because data were so limited and catches were nearly zero.

Dome shaped survey selectivity curves with parameters fixed at field study estimates were used in SS3 models for surfclams in the south and on GBK. Field estimates were used because they were relatively precise, based on a great deal of data, and were obtained from designed experiments carried out in association with the stratified random survey using actual survey sampling gear (Figure A54). When survey selectivity parameters were estimated by SS3 in preliminary runs, different selectivity curves with broader domes were obtained. Estimating selectivity improved goodness of fit, but retrospective and other analyses indicated that model stability was substantially reduced. Moreover, field study survey selectivity estimates were relatively precise and were considered likely to be directly applicable to survey catches.

The number of trips sampled by port agents was used as initial effective sample sizes for fishery length data in each year. The number of survey tows that caught surfclams was used as initial effective sample size for survey size composition data in each year. The number of fish aged in each size group and year was used as the initial effective sample size for survey conditional catch at age data. Initial log scale standard deviations for survey abundance trend data were derived from the CV for mean numbers per tow in each year assuming that errors were lognormal. These initial specifications for length and age data were “tuned” (adjusted up or down) based on preliminary model fits by multiplying the values for each type of data by a constant that was the same for all observations of the same data type. The initial standard deviations for survey trend data were tuned based on preliminary model fits by adding a constant to the standard deviation for each observation in the time series.

In three anomalous cases for length data in the southern area (fishery length data for 1982 and 1989 and survey length data for 1984), effective samples sizes were fixed at a low value (effective N=10) to avoid distorting fit to the rest of the data in the model (see below). The survey length data for 1984 was anomalous because of a single very large catch of surfclams (the largest catch in the survey time series) that consisted almost entirely of 7-8.9 cm SL surfclams.

Prior for survey dredge capture efficiency

A prior distribution based on field study estimates of survey dredge capture efficiency was used to help estimate the catchability parameter for minimum swept area abundance from clam survey data. Survey dredge efficiency is key in estimating surfclam abundance in SS3, particularly because fishing mortality rates appear to be quite low (Figure A41). The model ignored the trend in swept-area abundance (likelihood weight=10^{-5}) but goodness of fit to the prior was included in the objective function. Catchability (q) and capture efficiency (e) are closely related:

$$I = qN$$
$$q = \frac{aeu}{A}$$

where I is mean number per tow in the survey, N is stock abundance (fully selected by the survey dredge for this derivation), A is stock area, a is the area swept by the dredge and u accommodates the change from survey units (mean number per standardized tow) to population abundance.

The time series of minimum survey swept-area abundance estimates (N') were developed assuming e=1 for use with the prior. These estimates were for surveys conducted beginning in 1997, when sensors were used to monitor dredge performance and to calculate area swept accurately. Minimum swept area abundance was calculated:

$$N' = \frac{AI}{au}$$

where survey mean number per tow (I) was calculated after adjusting the catches in each survey tow to a standard tow distance (a) based on sensor measurement of tow distance and after discarding a few tows with poor dredge performance due to problems identified using sensors (see TOR 2). Stock area (A) was the area covered by the survey (assumed to be the stock area) reduced by an estimate of the fraction of the stock area which is untowable by the survey dredge (untowable ground was assumed to be unsuitable habitat). In theory, catchability for the swept area abundance data is the same as capture efficiency because $q=N'/N=e$. Thus, the catchability coefficient from SS3 was an estimate of dredge capture efficiency that could be compared to the prior for capture efficiency based on field studies.

The prior for log efficiency in SS3 was normally distributed because the prior distribution for efficiency was lognormal. The original lognormal distribution had a mean of 0.234 and a CV of 1.304. The standard deviation of the normal prior for log efficiency was $\sigma = \sqrt{log(1 + CV^2)} = 0.997$ and the mean was $log(0.234) - 0.5\sigma^2 = -1.95$.

Comparing SS3 and KLAMZ

Care is required in comparing estimates from KLAMZ and SS3. Biomass results from SS3 were for ages 6+ (south) and 7+ (GBK where growth is slower) on January 1 (unless noted otherwise) to approximate the biomass of surfclams 12+ cm SL estimated in KLAMZ. Annual exploitation rates from SS3 were catch weights divided by biomass of ages 6+ (south) and 7+ (GBK) on January 1 and should be roughly comparable in both models.

Fishery selectivity assumptions and fishing mortality estimates differ in SS3 and KLAMZ and make comparisons more difficult. Fishing mortality rates were not comparable because estimates from SS3 related catch numbers to area abundance for fully recruited size groups (about 15-17 cm SL in the southern region and 14+ cm in GBK). Estimates from KLAMZ related catch weight to population biomass, assuming that all surfclams 12+ cm SL were fully recruited to the fishery.

Recruitment estimates from the two models were not comparable because recruitment was estimated as a smooth random walk in KLAMZ and as independent estimates around a constant mean in SS3. Age composition data used in SS3 were informative and made it possible to model recruitment in a more complicated and realistic manner. Moreover, recruitment was the biomass of clams 12-12.9 cm SL (approximately age 6 y) in KLAMZ and numbers of age 0 recruits on January 1 in SS3.

Issues

The primary issues encountered in using SS3 in preliminary runs for surfclams in the southern area were: 1) choice of growth parameters to be estimated, 2) fit to fishery size composition data for sizes 14+ cm SL, 3) lack of fit to survey data (overall trends as well as size composition data for 1982, 1983 and 1986), and 4) lack of fit to commercial size data for the largest surfclams. The most important issue in using SS3 for GBK surfclams was sparse data that limited estimation of key parameters and contributed additional uncertainty.

Decisions about growth parameters were important because growth assumptions were key elements in fitting the age structured SS3 model to commercial and survey size data and because growth has changed over time in the southern area. SS3 uses von Bertalanffy growth curves with five parameters. L_{min} was the predicted size at a_{min}, L_{max} was the predicted size at a_{max}, K was the von Bertalanffy growth rate parameter, where a_{min}=5 y and a_{max}=30 y are user specified ages. SD_{min} was the standard error in size for surfclams at age a_{min}, and SD_{max} was the standard error in size at age a_{max}. In addition, growth is assumed to linear between 0 and L_{min} for ages 0 to a_{min}. For GBK, growth parameters were assumed constant over time and fixed at estimates made externally from survey data.

L_{min}, L_{max} and K for the 1975-2006 cohorts in the southern area were estimated in three separate preliminary model runs as random walks. Cohorts born before 1975 or after 2006 were assumed to have the same growth curve as the 1975 or 2006 cohorts. Annual steps in the random walk were assumed to have log scale standard deviations of 0.05 so that parameters might change by about 5% per year on average. Results suggested relatively fast growth to large size (high K and L_{max}) for the 1978-1983 cohorts (Figure A70). The variability in L_{max} was unrealistically large (about 12-23 cm SL compared to about 16 cm SL from external estimates). The working group concluded that the apparent variability in L_{max} was probably due to anomalous survey size data for 1982-1984 and 1986 which remain unexplained (see below). In the absence of an explanation for the survey size data, growth parameters were assumed to be constant over time in the south. The group assumed that the obvious changes in growth after 1994 in the southern areas were relatively unimportant for the stock as a whole because abundance and biomass there was a relatively small fraction of the total after 1994.

Next, fifteen preliminary model runs were carried out estimating individual growth parameters or sets of growth parameters with all parameters assumed constant over time (Table A22 and Figure A71). External parameter estimates from growth curves were used as starting values for estimated parameters or for parameters not estimated. The two best models, based on total negative log likelihood (NLL) estimated relatively high L_{min}, low K values, and implausible growth curves. In contrast, the model with the third lowest NLL, which estimated L_{min} and L_{max} only, seemed to provide relatively good fit and a plausible growth curve. Therefore L_{min} and L_{max} were estimated in final SS3 models for the southern area with other growth parameters fixed at initial values.

SS3 did not fit survey trend data as well as initially expected based on KLAMZ model results (Figure 2 in Appendix A5). A sensitivity analysis was carried out with a preliminary model that used a large likelihood weight (λ=100) for survey fit. This caused the fit to the survey trend data to improve. Fit to all length and age data, however, degraded substantially (Table A23). Estimated trends were similar except during the late 1980s and early 1990s (Figure A72) The working group concluded that the survey trend data were relatively noisy and that SS3 did not fit the trend closely because there was no evidence in the length and age data that the variability in the survey trend was real.

Three sensitivity runs with a preliminary model were used to address lack of fit to the very peaked survey length composition data for 1982-1983 and 1986 in the southern area. Run 1 placed a high weight (λ=100) on all of the survey size data in the model. Run 2 increased the weight on just the 1982-1983 and 1986 survey size data by multiplying the assumed effective samples sizes by 10. Run 3 dropped the survey size data for 1982-1983 and 1986 entirely. The run with a high weight on all survey sizes indicated faster growth in area biomass to a higher level during the early 1980s. However, the working group noted that the lack of fit seemed relatively unimportant because: 1) biomass estimates for 1988-2011 were similar in all runs (Figure A73), 2) there were no problems fitting survey age data for 1982-1983 or 1986, and 3) the survey size data for 1984 (down weighted due to one large tow) were not as peaked as in the problematic years. Based on these considerations, the Working Group decided to include lack of fit to early survey size composition data as a research recommendation but to ignore it otherwise in SS3 models.

The lack of fit to commercial size composition data at large sizes (14-18 cm SL) suggests that natural mortality (*M*) increased for large surfclams or that commercial selectivity was dome shaped such that large clams were less likely to be caught. Natural mortality has been fixed at 0.15 in surfclam assessments since 2000 (NEFSC 2000, see appendix 7 in NEFSC 2009 for a discussion of M estimates for surfclam). Sensitivity analyses were run with a preliminary model that estimated natural morality rates for clams age 7+ y, 8+ y, etc. while maintaining M=0.15 y^{-1} for younger ages. The estimated natural mortality rates were always about 0.15 y^{-1}. These results indicate that the model was able to fit the survey age data (which show surfclams 30+ y in age routinely) reasonably well under the assumption that M=0.15 y^{-1} for all ages and size groups. In contrast, the lack of fit to commercial size composition data at large sizes was nearly eliminated when a dome-shaped fishery selectivity curve was estimated in the model.

The improvement in model fit with dome-shaped fishery selectivity in the south was puzzling. External estimates of commercial fishery selectivity based on field experiments indicate that the commercial clam dredges used to harvest surfclams (Figure A54) and ocean quahogs (Thorarinsdottir et al. 2010) have logistic, rather than domed fishery selectivity patterns. Industry contributors to the Working Group reported that clam dredges are designed to collect large surfclams with high efficiency because large clams provide a higher meat yield.

Based on these considerations, the Working Group concluded that the lack of large individuals in commercial samples from the southern area was probably due to removal of large surfclams by relatively heavy fishing on the productive grounds where the fishery is concentrated. In other words, the apparently domed relationship between length composition and fishery length samples from the southern area was probably due to logistic gear selectivity combined with removal of large clams (relative to the area as a whole) on fishing grounds.

Based on the considerations above, a dome shaped fishery selectivity pattern was estimated in the basecase model for the southern area. However, Georges Bank is essentially virgin. Therefore, the Working Group assumed that the fishery selectivity pattern for Georges Bank had the same shape (same parameters) as estimated for the southern area on the left hand side for small surfclams. The right hand side for large surfclams was assumed to be asymptotic resulting in a typical logistic selectivity pattern. No selectivity parameters were estimated for GBK because commercial size data for GBK were too few and too noisy.

Fit and estimates from basecase models

Goodness of fit for final basecase models (Tables A24) was generally good, with the exception of the early survey size composition data described above. The estimated catchability (survey dredge capture efficiency) estimate for swept area abundance in the south (*e*=0.33) was larger than the mode and mean of the experimentally derived prior (see TOR 2), but seems plausible. Fit to conditional age at length was good based on observed and predicted mean age and variance in ages at size, although there were patterns in bubble plots for age at length residuals (see Appendix A6). The models fit traditional survey age composition data very well even though they were not used in fitting the model, which relied on conditional age at length information. Strong year classes estimated by the models were clearly visible in the traditional age composition data, indicating that the conditional and traditional age data convey the same information. Full diagnostics of the model fit are available in Appendix A6.

In the southern area, biomass and fishing mortality were estimated with reasonable precision, while recruitment trends were relatively uncertain in recent years (Figures A74 – A76, Table A25). Biomass and recruitment were less precisely estimated in the northern area (Figures A77 – A79, Table A26).

Likelihood profile analysis

Likelihood profile analyses was an important uncertainty analysis that was carried out for surfclams in the southern area by fixing the catchability coefficient for the NMFS clam survey at successive values that bracketed the best estimate and estimating all of the other parameters in the model. To ease interpretation, results were presented in terms of the catchability coefficient for swept-area abundance in each run (i.e. for survey dredge efficiency). The profile was not carried out using dredge efficiency *per se* as the fixed variable for southern area runs because dredge efficiency interacts with its prior distribution. Instead, we report the dredge efficiency estimate that was obtained for each fixed value of clam survey catchability. Points where the negative log likelihood in profile analysis was the minimum value + 1.92 likelihood units were used to approximate 95% confidence bounds (Figure A80).

Likelihood profile results for the south indicate that goodness of fit for the survey trend was best near the basecase model run (Table A27). Fishery and survey length data support higher dredge efficiency estimates (lower biomass) while survey age data support lower dredge efficiency estimates (higher biomass). Biomass estimates were sensitive to dredge efficiency but trends and the status ratio (B2011/B1999) were not (Figure A80). The 95% confidence interval for dredge efficiency based on the profile analysis was about 0.24 to 0.43, the confidence interval for biomass was about 625,000 to 1,025,000 mt, and the confidence interval for B2011/B1999 was about 0.43 to 0.49 (Figure A80).

Preliminary runs showed that the likelihood surface for the GBK region was nearly the same over a relatively wide range of fixed dredge efficiency values. In other words, none of the data provided information about the overall abundance of GBK surfclams. Therefore, no likelihood profile analysis was performed for GBK and the working group concluded that biomass estimates for GBK were no more (and possibly much less) certain that the estimated dredge efficiency from the south.

Internal retrospective

The internal retrospective pattern for the southern area was minimal, Mohn's rho was only ρ = 0.02 for a nine year "peal" (after dropping nine 2002-2010) (Figure A81). The retrospective pattern in the GBK area was more substantial (Mohn's ρ = 0.30), but the confidence bounds of each successive peel overlapped considerably, indicating the retrospective probably did not constitute a substantial bias (Figure A82). Given limitations in the data for GBK (including no 2005 survey) it is not clear that better results could be expected.

Whole stock results

Whole stock biomass estimates for clams 12+ cm SL were the sum of the biomass estimates from each area $B_W = B_S + B_N$. Because the estimation error associated with the two areas was independent, the variance of the sum of the biomasses was $\sigma_W^2 = \sqrt{\sigma_N^2 + \sigma_S^2}$. Whole stock fishing mortality was $F_W = \frac{(C_S + C_N)}{(\bar{N}_S + \bar{N}_N)}$ where C_S and C_N were the catch in numbers from each area and $\bar{N}_S$ and $\bar{N}_N$ were average fully selected abundances $\bar{N} = \sum_L s_L \frac{N_L(1-e^{-Z_L})}{Z_L}$, where the total mortality rate (Z) was based only on fully selected lengths and s_L was commercial fishery size selectivity. Whole stock results are discussed in TOR 6 and are listed in Table A26B.

Historical retrospective

When the summary biomass estimates from both the northern and southern areas area were summed, the results were higher than biomass estimates from previous assessments (Table A28, Figure A83). Direct comparability is nuanced because the current assessment makes use of new data sources (e.g. age and size structure), and because the comparison of age 6+ (south) and 7+ (north) to animals greater than 12 cm is only approximately direct.

Older versions of the surfclam assessment used swept area biomass estimates as the primary means of determining stock status. These analyses were updated in appendix (A8).

Performance of historical projections

The previous assessment projected a combined GBK + south biomass of 868 thousand mt in 2011. This estimate was based on the “industry estimate” catch (20 – 23 thousand mt including incidental mortality). Actual catch was within this range. The current assessment estimated 1,100 thousand mt. The current estimate is outside the approximate 95% asymptotic confidence bounds (717 – 1,051 thousand mt) implied by the CV of the previous estimate (0.10). It is, however, difficult to compare forecast and current estimates because of the changes in estimates described above.

Updated and redefined biological reference points and scientific adequacy of existing and redefined BRPs (TOR 5)

According to the FMP for Atlantic surfclams, overfishing occurs whenever the annual fishing mortality rate on the entire (GBK + south) surfclam resource (stock) is larger than the over fishing limit (OFL). The OFL for Atlantic surfclam is based on the F_{MSY} proxy. The stock is overfished if total biomass falls below $B_{Threshold}$, which is estimated as ½ B_{MSY} proxy. When stock biomass is less than the biomass threshold, the fishing mortality rate threshold is reduced from F_{MSY} to zero in a linear fashion.

The current proxy for $F_{MSY} = M = 0.15$ y^{-1} was not revised in this assessment. However, its interpretation is revised because of the change in stock assessment models. In the KLAMZ model used previously, F=0.15 y^{-1} was effectively a biomass weighted mortality measure that corresponded (under certain conditions) to the standard abundance weighted mortality rates estimated in SS3. Moreover, fishery selectivity was assumed knife-edged at 120+ mm in KLAMZ but was estimated in SS3 to be dome-shaped with selectivity near one at sizes 160+ mm on GBK and 160-170+ mm SL in the south. At the OFL, all surfclams 120+ mm SL would experience F=0.15 based on the KLAMZ model but only surfclams 160+ or 160-170+ mm SL would experience F=0.15 based on the SS3 model. In effect, the OFL under SS3 is lower from a biological perspective than under KLAMZ. The potential split into two stocks (GBK and south) does not affect the current proxy because it can be applied under any set of stock definitions.

The current proxy for B_{MSY} in the current stock unit (GBK + south) is one-half of the estimated fishable biomass during 1999. The current proxy for $B_{Threshold}$ (which is used to identify overfished stocks) is $B_{MSY}/2$ or $B_{1999}/4$. Biomass in 1999 and related biological reference points under the current stock definition were re-estimated in this assessment (see below).

Current Stock Definition (GBK + southern areas)

Reference Point	Last assessment	Revised
F_{MSY}	M=0.15 y^{-1}	Same
B_{1999}	1086 thousand mt meats	1944 thousand mt meats
$B_{MSY} = ½B_{1999}$ **(target)**	543 thousand mt meats	972 thousand mt meats
$B_{Threshold}$ = ½ B_{MSY}	272 thousand mt meats	486 thousand mt meats
MSY	NA	98 thousand mt meats

The possible revision of the stock definition for surfclams which would separate GBK and the southern region complicates biological reference points to some extent. The Invertebrate Subcommittee noted that B_{1999} was almost identical (probably fortuitously) to estimated virgin biomass in the basecase SS3 model for the southern area and in sensitivity analysis and preliminary runs. The Subcommittee therefore agreed that $B_{1999}/2$ was still a suitable proxy for B_{MSY} in the southern region. The Subcommittee concluded that B_{1999} was preferable to a formal virgin biomass estimate from an assessment model as the basis for biomass reference points because the stability of estimated trends substantially reduces uncertainty in the ratio $B_{Current}/B_{Threshold}$ when $B_{Threshold} = B_{1999}/4$ and because of uncertainty about ongoing environmental trends. The group concluded that ratio of $B_{Current}$ over an estimate of B_{MSY} was thought unlikely to be robust particularly due to uncertainties about B_{MSY} in the face of environmental change.

The Invertebrate Subcommittee found no technical basis for establishing a B_{MSY} proxy for GBK. GBK is virgin, biomass has varied considerably there in the absence of fishing due presumably to environmental effects (Figure A77), and data for the GBK region is limited. The Subcommittee agreed that this uncertainty does not present any practical problems for determining legal status in this assessment because GBK is virgin and could not, by any definition, be overfished. Therefore, B_{MSY} for GBK is not defined but is considered an important research topic for the next assessment.

Southern Area

Reference Point	Last assessment	Revised
F_{MSY}	M=0.15 y^{-1}	Same
B_{1999}	1,086 thousand mt meats	1488 thousand mt meats
$B_{MSY} = ½B_{1999}$ **(target)**	543 thousand mt meats	744 thousand mt meats
$B_{Threshold}$ = ½ B_{MSY}	272 thousand mt meats	372 thousand mt meats
MSY	NA	74 thousand mt meats

Northern Area

Reference Point	Last assessment	Revised
F_{MSY}	M=0.15 y^{-1}	Same
B_{1999}	NA	NA
B_{MSY} =½B_{1999} **(target)**	NA	Undefined
$B_{Threshold}$ = ½ B_{MSY}	NA	Undefined
MSY	NA	29 thousand mt meats

Revised biomass reference points are higher than previous values primarily because of new information regarding the efficiency of the dredge used in NEFSC clam surveys and SS3 models that included age and length data. Conclusions about stock status are robust and would not change unless either the natural mortality estimate or biomass threshold was changed substantially.

Scientific adequacy of reference points

The current proxy for F_{MSY} (M = 0.15) is a common approach used in many fisheries. However, the productivity of the surfclam stock appears low for a species with M=0.15 and surplus production in surfclams may be negative for periods up to one or two decades. The performance of the simulated surfclam stock in projection analyses under the F_{MSY} proxy policy indicates that M=0.15 may not be an ideal proxy for F_{MSY} in the surfclam fishery. In addition, there is uncertainty about natural mortality in surfclams, which likely varies temporally and spatially. Reductions in biomass of surfclam in inshore southern regions are probably due, in part, to changes in environmental conditions and increasing natural mortality. On the other hand, the occurrence of old clams (> 35 y) in survey catches implies that the natural mortality rate may be lower than assumed. Sensitivity analysis indicated that the surfclam population in the south was adequately modeled using M=0.15. While there are indications that the current F_{MSY} proxy could be improved, there are no compelling reasons to change it at this time.

Stock status evaluation with respect to BRPs (TOR-6)

Current stock definition

The Atlantic surfclam stock in the US EEZ (current stock definition, GBK+south) has a low probability of being overfished ($B_{2011} > B_{Threshold}$) because the 95% confidence intervals for the biomass and reference point estimates do not overlap). The estimated stock biomass during 2011 for surfclams 120+ mm SL was 1060 thousand mt meats (CV=0.15) with a 95% confidence interval of approximately 791 to 1420 thousand mt meats. The biomass threshold is 1/4 of the biomass estimate for 1999; $B_{Threshold}$= 486 thousand mt meats (CV= 0.14) with a 95% confidence interval of 374 to 633 thousand mt meats (Figure A84, Table A29).

Surfclam biomass in 2011 was probably above its target biomass level ($B_{2011} < B_{Target}$) because the 95% confidence intervals for the target and current biomass levels do not overlap. The biomass target is ½ of the estimated biomass during 1999; B_{Target} = 972 thousand mt (CV 0.135) with a 95% confidence interval of 747 to 1235 thousand mt (Figure A84).

The Atlantic surfclam stock in the US EEZ is not experiencing overfishing ($F_{2011} < F_{MSY}$). Fishing mortality for the entire resource (F_W) was based on a numerically weighted average of the annual fishing mortality in each area, accounting for different selectivities. The estimated fishing mortality during 2011 was $F = 0.027$ y^{-1}, with 95% confidence intervals of (0.016 – 0.045), which is below the management threshold OFL of $F = M = 0.15$ y^{-1}. The confidence interval suggests that there is virtually no probability that F exceeded the OFL during 2011 (Figure A85, Table A30).

Alternative stock definition

The alternative stock definition would separate GBK and area to the south as separate stocks. There are no reference points currently defined for the GBK area (see TOR 5). The stock was not fished between 1989 and 2009 and is essentially virgin. Therefore the stock is not overfished and overfishing is not occurring.

The estimated stock biomass in the southern area during 2011 for surfclams age 6+ (~120+ mm SL) was 703 thousand mt meats (CV=0.2) with a 95% confidence interval of approximately 481 to 1028 thousand mt meats (Figure A74). The biomass threshold is 1/4 of the biomass estimate for 1999; $B_{Threshold}$= 392 thousand mt meats (CV= 0.17) with a 95% confidence interval of 268 to 516 thousand mt meats (Figure A86, Table A31). The confidence intervals associated with B2011 and the threshold reference point in the southern area overlap. Therefore there is a possibility that the southern area is overfished. Overfished probability was calculated using the approach detailed in Shertzer et al. (2008). The distributions for B_{2011} and $B_{THRESHOLD}$ were assumed to be log normal, with means equal to their point estimates and variances equal to their delta method variances (B_{2011}~LogN(6.55,0.194); $B_{THRESHOLD}$~LogN(5.92,0.167)). 10,000,000 possible threshold values were drawn from correlated distributions with means and variances as described above, where the correlation between them was equal to the correlation between $B_{THRESHOLD}$ and B_{2011} estimated in the model (0.90). Each pair of draws was compared. Overfished status occurred when the threshold draw was greater than the biomass draw. Probabilities were equal to the number of overfished occurrences divided by the number of comparisons made. The probability of being overfished was <1% (Figure A87).

The southern area is not experiencing overfishing ($F_{2011} < F_{MSY}$). The estimated fishing mortality during 2011 was $F = 0.040$ y^{-1}, with 95% confidence intervals of (0.025 – 0.056), which is below the management threshold OFL of $F = M = 0.15$ y^{-1}. The confidence interval suggests that there is virtually no probability that F exceeded the OFL during 2011 (Figure A88, Table A32).

Projections (TOR 7)

Basecase SS3 models were used to project biomass of surfclams approximately 120+ mm SL (age 6+ y in the south and 7+ y on GBK), landings (mt and bu), fully recruited fishing mortality, and annual exploitation rates (catch weight/biomass) in the southern area, GBK area, and the combined areas during 2012-2021 (Table A33 – A35 and Figures A89 – A95). Three harvest policies were assumed: 1) F=0.15 y^{-1} (at the OFL), 2) status-quo catch (23,357 mt y^{-1}, equivalent to landings of 20,854 mt or 2.7 million bu y^{-1}) and 3) the maximum allowed catch under the current FMP or "quota level" catch (29,359 mt y^{-1}, equivalent to 26,213 mt or 3.4 million bu y^{-1}) in the combined areas (Table A34).

There is a positive probability that the stock will be overfished within the next five years. The maximum probability of overfished status coincides with the minimum biomass estimate over the five year time horizon. Using the Shertzer et al. (2008) method, the probability of the whole stock being overfished ranged from 0.005 to 0.035, depending on the projection scenario being considered (Figure A96). Under the alternate stock definition the probability of the southern area being overfished in the next 5 years ranged from 0.015 – 0.044 (Figure A97).

The most likely fishing scenario is probably status quo, because the fishery is market limited and has been fishing under quota since 2004 (Table A2). The quota scenario is therefore a reasonable upper bound on likely fishing pressure over the next five years. Using the quota scenario and the maximum probability of being overfished in *any one year* in next five ($P*$ = 0.005, or 0.015, for the whole stock and southern area respectively) the cumulative probability of being overfished *at any time* during the next five years is $1 - \prod_y \left(1 - P_y^*\right)$ = 0.015 and 0.056 (Table A36), for the whole stock and southern area respectively, where P_y^* is the $P*$ value for each year (see Shertzer et al, 2008).

Catches were landings + 12% to account for assumed incidental mortality. Catches and landings during 2012 were assumed the same as during 2011. For lack of better information, catches on GBK during 2013-2021 were assumed to be the same in the status-quo catch and quota level catch scenarios. This assumption is likely reasonable for the first few years because of processor infrastructure and fleet range limitations. Thus, any differences in total catch between scenarios or over time would probably be due to differences in southern catches. Catches from GBK may, however, increase at some point if additional vessels capable of fishing on GBK, and additional processing infrastructure, are built in the north.

Projected total landings, biomass and exploitation levels for the combined area were obtained by adding estimates for the southern and GBK areas. Fishing mortality was not computed exactly for the combined area because fishery selectivity differs between the southern and GBK areas and numbers at size was not a projection output. Approximate fishing mortality was based on numerically weighted average fishing mortality from each area.

Projected fishing mortality levels are lower than the fishing mortality threshold F=0.15 y^{-1} for the entire resource under the current stock definition under all scenarios except F=M=OFL (Figure A91; Table A36). Under the alternative stock definition, neither the southern area nor the GBK area are likely to experience overfishing under the status quo or quota scenarios (Figures A93 and A95; Table A36).

Probability distributions of the catch at the OFL were generated by repeated draws from the sampling distribution of biomass in each year. B_i, the biomass in year *i* was assumed to have a log normal distribution $B_i \sim Lognormal(\beta_i, \sigma_i)$, where β_i is point estimate of biomass in year *i* and σ_i is the delta method standard deviation estimated in the model for biomass in year *i*. The overfishing limit F=M=0.15 was applied to each of 1,000,000 draws from the distribution for B_i, resulting in a probability distribution of catch (Figures A98 – A200; Table A37).

Additional sensitivity analyses and decision tables based on projections are available in appendix A9.

Research recommendations (TOR 8)

The following are previous research recommendations (not in priority order):

i) Continue surfclam recruitment research. *This assessment incorporates length and age data. Age structure provides some new information that was not previously leveraged in forecasting. This change should allow for more precise estimation of the magnitude of incoming year classes and* *thus improve our ability to predict important recruitment events. Including age and size structure have also broadened the scope of hindcast recruitment analysis by allowing the inclusion of younger ages into the assessment model. Recruits in the old assessment were animals approximately five years old. We now use age zero animals.*

ii) Port samples should be taken from the SNE and GBK (if fishing resumes there) regions. *Collected since 2010.*

iii) Determine how much of Georges Bank is good surfclam habitat, and if depletion and selectivity experiments done in the mid-Atlantic are applicable to the Georges Bank region. *We have begun exploratory work with existing HabCam*[3] *images, attempting remote identification of bivalves using siphon anatomy. We hope that automated identification of live surfclam is possible and will lead to a better understanding of habitat use by surfclam. If this turns out to be too difficult it is possible that visual inspection of HabCam images will lead to habitat identification through other means, such as identifiable shell piles or shell hash. This project is still in exploratory stages, though we have applied twice for funding.*

iv) Fecundity and maturity at length information is required to improve reference point calculations and predict management effects. *No progress. This issue is technically difficult to resolve in situ and is unlikely to be addressed in the near term. Direct studies of fecundity would require specialized laboratory facilities. It is possible that academic partners may pursue this research topic.*

v) Data on the number of clams per bushel landed at different ports over time would be useful. *No progress.*

vi) Commercial length data for surfclams should be more accessible. *Commercial length data is summarized in this document and is available by request through NEFSC.*

vii) Determine whether the carrying capacity of surfclams has changed over time. *No progress. Surfclam are experiencing a range contraction as habitat degrades in the southern extreme of the historical species extent due to climate change. Carrying capacity has certainly changed over time, and clearly continues to change, though this topic has not been directly addressed analytically.*

viii) Estimate densities of spawning surfclams necessary to produce good recruitment. Is reproduction likely to be impaired if relatively dense beds of surfclams are reduced? *No progress.*

New research recommendations (not in priority order)

i) Biomass reference points need to be reconsidered.
ii) Has surfclam biomass shifted offshore into deeper water over time?
iii) Look into a better way to implement regime change into the SS3 model. Look into patterns which may match other species and climate indices.
iv) Determine the best spatial and temporal distribution to use for surfclam assessment models

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[3] See http://habcam.whoi.edu

v) Look at habitat on GBK
vi) Given the increasing importance of GBK re-evaluate the optimal sampling design for the survey.
vii) Look into area specific recruitment streams for SS3 and how to accommodate the 2012 and 2013 surveys.

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Table A1. Surfclam discard estimates from 1982 through 1994. A minimum size regulation was in effect from 1982 through 1990. Within two years of dropping the minimum size regulation (1993) the discard rate had dropped to zero and has remained zero since then.

Year	Discard (mt meats)					Landings (mt meats)	Discards / Landings	Catch	Size limit (mm)
	NNJ	SNJ	NJ	DMV	Total				
1982	3,684	215	3,899	2,295	6,194	16,688	37%	22,882	140
1983	2,122	385	2,507	2,127	4,634	18,592	25%	23,226	140
1984	2,266	458	2,724	2,015	4,739	22,888	21%	27,627	133
1985	1,938	248	2,186	1,725	3,911	22,480	17%	26,391	127
1986	2,328	233	2,561	239	2,800	24,520	11%	27,320	127
1987	1,414	61	1,475	415	1,890	21,744	9%	23,634	127
1988	1,317	13	1,330	106	1,436	23,377	6%	24,813	127
1989	1,048	6	1,054	258	1,312	21,887	6%	23,199	127
1990	1,089	57	1,146	123	1,269	24,018	5%	25,287	127
1991	495	36	531	5	536	20,615	3%	21,151	--
1992	918	102	1,020	4	1,024	21,685	5%	22,709	--
1993	0	0	0	0	0	21,859	0%	21,859	--
1994	0	0	0	0	0	21,942	0%	21,942	--

Table A2. (Following page) Atlantic surfclam landings and EEZ surfclam quotas. All figures are meat weights in mt. Total landings for 1965-1981 are from NEFSC (2003) and while figures for other years were from a dealer database (CFDBS). EEZ landings for 1965-1982 are from NEFSC (2003) while figures from later years are from a logbook database (SFOQVR). Landings for state waters are total landings - EEZ landings.

Year	Total (dealer data)	EEZ (logbooks)	State waters (dealer-logbooks)	Proportion from EEZ	EEZ Quota
1965	19,998	14,968	5,030	0.75	
1966	20,463	14,696	5,767	0.72	
1967	18,168	11,204	6,964	0.62	
1968	18,394	9,072	9,322	0.49	
1969	22,487	7,212	15,275	0.32	
1970	30,535	6,396	24,139	0.21	
1971	23,829	22,704	1,125	0.95	
1972	28,744	25,071	3,673	0.87	
1973	37,362	32,921	4,441	0.88	
1974	43,595	33,761	9,834	0.77	
1975	39,442	20,080	19,362	0.51	
1976	22,277	19,304	2,973	0.87	
1977	23,149	19,490	3,659	0.84	
1978	17,798	14,240	3,558	0.8	13,880
1979	15,836	13,186	2,650	0.83	13,880
1980	17,117	15,748	1,369	0.92	13,882
1981	20,910	16,947	3,963	0.81	13,882
1982	21,727	16,688	5,039	0.77	18,506
1983	23,631	18,592	5,038	0.79	18,892
1984	30,530	22,889	7,641	0.75	18,892
1985	28,316	22,480	5,835	0.79	21,205
1986	35,073	24,521	10,552	0.7	24,290
1987	27,231	21,744	5,486	0.8	24,290
1988	28,506	23,378	5,128	0.82	24,290
1989	30,081	21,888	8,194	0.73	25,184
1990	32,628	24,018	8,610	0.74	24,282
1991	30,794	20,615	10,179	0.67	21,976
1992	33,164	21,686	11,478	0.65	21,976
1993	32,878	21,859	11,019	0.66	21,976
1994	32,379	21,943	10,436	0.68	21,976
1995	30,061	19,627	10,434	0.65	19,779
1996	28,834	19,827	9,008	0.69	19,779
1997	26,311	18,612	7,700	0.71	19,779

1998	24,506	18,234	6,272	0.74	19,779
1999	26,677	19,577	7,100	0.73	19,779
2000	31,093	19,778	11,315	0.64	19,779
2001	31,237	22,017	9,220	0.7	21,976
2002	32,645	24,006	8,639	0.74	24,174
2003	31,526	25,017	6,509	0.79	25,061
2004	28,322	24,197	4,125	0.85	26,218
2005	26,882	21,163	5,719	0.79	26,218
2006	27,176	23,573	3,604	0.87	26,218
2007	27,094	24,915	2,179	0.92	26,218
2008	27,750	22,519	5,231	0.81	26,218
2009	22,972	20,149	2,823	0.88	26,218
2010	19,978	18,102	1,876	0.91	26,218
2011	19,908	18,587	1,320	0.93	26,218
Min	15,836	6,396	1,125	0.21	13,880
Max	43,595	33,761	24,139	0.95	26,218
Mean	27,022	19,983	7,039	0.75	21,850

Table A3. EEZ surfclam landings (mt meats) by stock assessment area and year prorated based on NEFSC (2003) for 1979 and logbook data for 1980-2011. Landings from unknown areas in each year were prorated to known areas based on logbook proportions of landings in known areas.

Year	SVA	DMV	NJ	LI	SNE	GBK	Other	Total EEZ
1979	0	11,836	1,350	0	0	0	0	13,186
1980	64	12,788	2,878	17	0	0	0	15,748
1981	568	7,472	8,820	88	0	0	0	16,947
1982	1,705	6,679	8,086	94	125	0	0	16,688
1983	2,225	7,173	8,095	264	836	0	0	18,592
1984	1,797	5,979	11,905	7	382	2,766	54	22,889
1985	741	7,856	11,246	0	452	2,185	0	22,480
1986	529	2,853	17,730	17	1,223	1,991	177	24,521
1987	378	1,303	18,017	0	1,140	907	0	21,744
1988	558	1,149	19,420	0	1,512	739	0	23,378
1989	439	3,123	16,532	0	1,361	433	0	21,888
1990	1,502	3,546	17,887	0	998	7	79	24,018
1991	0	1,634	18,913	15	33	0	21	20,615
1992	0	1,221	20,399	61	5	0	0	21,686
1993	0	3,414	18,365	62	3	0	14	21,859
1994	0	3,454	18,418	71	0	0	0	21,943
1995	0	2,752	16,497	0	378	0	0	19,627
1996	0	2,239	17,479	26	82	0	0	19,827
1997	0	1,540	16,999	73	0	0	0	18,612
1998	0	484	17,511	117	121	0	0	18,234
1999	0	648	18,755	157	16	0	0	19,577
2000	0	2,042	17,513	121	103	0	0	19,778
2001	0	3,282	17,719	935	81	0	0	22,017
2002	64	4,489	18,271	1,130	52	0	0	24,006
2003	0	1,432	21,693	1,625	267	0	0	25,017
2004	0	1,482	19,197	906	2,612	0	0	24,197
2005	0	1,668	16,850	759	1,885	0	0	21,163
2006	0	2,773	19,660	245	895	0	0	23,573
2007	0	3,073	20,268	1,117	458	0	0	24,915
2008	0	3,261	17,517	1,317	423	0	0	22,519
2009	0	1,978	14,881	1,827	1,451	11	0	20,149
2010	0	1,583	11,144	1,184	2,888	1,302	0	18,102
2011	0	1,427	11,908	437	2,420	2,397	0	18,587
Min	0	484	1,350	0	0	0	0	13,186
Max	2,225	12,788	21,693	1,827	2,888	2,766	177	25,017
Mean	320	3,565	15,513	384	673	386	10	20,851

Table A4. EEZ fishing effort (hours fished by all vessels) for surfclam, by stock assessment area and year based on logbook data. The fraction of logbook effort from unknown areas in each year was prorated to known areas based on effort in known areas. Effort data prior to 1981 are less reliable due to restrictions on hours fished per day.

Year	SVA	DMV	NJ	LI	SNE	GBK	Other	Total EEZ
1982	2,790	18,050	24,636	225	137	0	0	45,838
1983	4,191	18,805	23,584	536	1,130	0	0	48,245
1984	2,603	8,972	20,819	27	1,264	1,732	42	35,459
1985	397	4,686	10,518	0	1,702	2,608	0	19,911
1986	236	1,629	10,764	38	2,516	1,610	675	17,469
1987	262	722	11,910	0	3,780	1,006	0	17,680
1988	322	593	13,175	0	5,274	587	0	19,950
1989	228	1,615	11,794	0	4,741	389	0	18,768
1990	1,150	2,065	12,437	0	3,032	0	898	19,582
1991	0	1,254	17,243	21	107	0	293	18,917
1992	0	797	21,379	67	0	0	0	22,243
1993	0	2,423	18,232	57	15	0	5	20,731
1994	0	1,930	21,495	70	0	0	0	23,495
1995	0	1,560	18,625	0	1,059	0	0	21,244
1996	0	1,577	20,994	40	287	0	0	22,899
1997	0	1,098	20,383	77	0	0	0	21,558
1998	0	289	19,608	134	518	0	0	20,550
1999	0	734	18,146	151	149	0	0	19,180
2000	0	1,859	16,787	115	368	0	0	19,128
2001	0	2,536	18,461	962	148	0	0	22,108
2002	112	5,505	19,826	1,241	62	0	0	26,747
2003	0	2,367	25,034	1,828	176	0	0	29,405
2004	0	3,161	26,409	1,244	1,093	0	0	31,907
2005	0	2,654	24,379	1,207	1,364	0	0	29,604
2006	0	5,883	27,102	343	1,022	0	0	34,350
2007	0	7,065	34,664	1,587	960	0	0	44,276
2008	0	8,154	33,916	2,308	541	0	0	44,920
2009	0	5,669	33,648	4,195	2,528	12	0	46,053
2010	0	4,201	32,103	3,314	5,614	479	0	45,712
2011	0	3,067	35,043	1,361	7,339	1,084	0	47,894
Min	0	289	10,518	0	0	0	0	17,469
Max	4,191	18,805	35,043	4,195	7,339	2,608	898	48,245
Mean	410	4,031	21,437	705	1,564	317	64	28,527

Table A5. Real and nominal prices for surfclams based on dealer data. Average price was computed as total revenues divided by total landed meat weight during each year, rather than as annual averages of prices for individual trips, to reduce bias due to small deliveries at relatively high prices. The consumer price index (CPI) used to convert nominal dollars to 2010 equivalent dollars is for unprocessed and packaged fish, which includes shellfish and finfish.

Year	CPI	Prices ($ / bu)		Revenue (million $)	
		Nominal	Real ($2010)	Nominal	Real ($2010)
1982	0.50	8.94	17.89	25.186	50.406
1983	0.52	7.57	14.58	23.207	44.678
1984	0.54	8.37	15.54	33.156	61.521
1985	0.56	9.34	16.82	34.303	61.780
1986	0.57	9.20	16.21	41.841	73.725
1987	0.58	7.83	13.40	27.644	47.336
1988	0.60	7.80	12.91	28.826	47.721
1989	0.63	7.78	12.40	30.330	48.384
1990	0.65	7.66	11.76	32.393	49.755
1991	0.67	7.51	11.13	29.975	44.464
1992	0.69	7.40	10.72	31.832	46.125
1993	0.71	7.83	11.10	33.369	47.307
1994	0.72	9.82	13.64	41.241	57.261
1995	0.74	10.58	14.39	41.246	56.098
1996	0.75	10.24	13.66	38.275	51.085
1997	0.76	10.31	13.53	35.189	46.151
1998	0.77	9.19	11.92	29.200	37.869
1999	0.78	8.79	11.24	30.421	38.881
2000	0.80	9.43	11.80	38.025	47.568
2001	0.82	9.76	11.95	39.555	48.390
2002	0.83	9.45	11.37	39.988	48.141
2003	0.85	9.64	11.37	39.427	46.487
2004	0.87	9.59	10.99	35.209	40.377
2005	0.90	9.50	10.55	33.123	36.764
2006	0.93	10.19	10.95	35.908	38.608
2007	0.96	10.49	10.96	36.844	38.497
2008	0.98	10.96	11.20	39.441	40.316
2009	0.99	11.43	11.56	34.050	34.442
2010	1.00	11.67	11.67	30.240	30.240
2011	1.02	11.52	11.28	29.732	29.110

Table A6. Nominal landings per unit effort (LPUE, bushels h^{-1}) for surfclam fishing (all vessels) in the US EEZ from logbooks. LPUE is defined as total landings in bushels divided by total hours fished. Landings and fishing effort from unknown areas were prorated to area before LPUE was calculated.

Year	SVA	DMV	NJ	LI	SNE	GBK	Other	All areas
1982	79	48	43	54	118			47
1983	69	49	45	64	96			50
1984	89	86	74	35	39	207	165	84
1985	242	217	139		34	109		146
1986	291	227	214	59	63	160	34	182
1987	187	234	196		39	117		159
1988	224	251	191		37	163		152
1989	249	251	182		37	144		151
1990	169	223	187		43		11	159
1991		169	142	95	40		9	141
1992		199	124	119				126
1993		183	131	143	28		390	137
1994		232	111	132				121
1995		229	115		46			120
1996		184	108	85	37			112
1997		182	108	122				112
1998		217	116	114	30			115
1999		115	134	135	14			132
2000		142	135	137	36			134
2001		168	124	126	71			129
2002	74	106	120	118	108			116
2003		78	112	115	197			110
2004		61	94	94	310			98
2005		82	90	82	179			93
2006		61	94	93	114			89
2007		56	76	91	62			73
2008		52	67	74	101			65
2009		45	57	56	74	120		57
2010		49	45	46	67	352		51
2011		60	44	42	43	287		50
Min	74	45	44	42	14	120	9	50
Max	74	232	142	143	310	352	390	141
Mean	74	127	102	101	86	253	199	104

Table A7. Numbers of commercial trips sampled and numbers of surfclams measured in port samples from landings during 1982-2011, by region. Numbers of trips during 1982-1999 were estimated assuming 30 individuals sampled per trip, as specified in port sample instructions.

	DMV		NJ		LI		SNE		GBK	
Year	Trips	Lengths	Trips	Lengths	Trips	Lengths	Trips	Lengths	Trips	Lengths
1982	259	7756	249	7477	1	30				
1983	197	5923	375	11253	Unk.	Unk.	1	30		
1984	102	3066	425	12751	3	90				
1985	61	1832	256	7674	5	150				
1986	42	1260	171	5130	11	330				
1987	24	730	30	900	19	569				
1988	14	420	30	900	27	810				
1989	29	866	31	919	15	449				
1990	30	892	30	901	7	209				
1991	36	1080	76	2272						
1992	39	1170	57	1710						
1993	46	1392	31	928	Unk.	Unk.				
1994	4	119	30	900						
1995	24	720	17	510						
1996	38	1154	37	1117						
1997	54	1622	32	957						
1998	52	1560	23	690						
1999	57	1720	29	856						
2000	20	600	111	3315	1	30				
2001	33	970	42	1260						
2002	7	210	37	1111						
2003	2	60	80	2455	5	150				
2004	36	1080	2	60						
2005	19	581	61	1834	11	330				
2006	50	1541	49	1482	23	690				
2007	68	2215	72	2409	16	508				
2008	57	1712	65	1950	21	632				
2009	31	932	59	1771	43	1296				
2010	25	751	43	1293	36	1086	3	90	15	450
2011	28	780	126	3706	52	1460	70	2097	7	240
Min	2	60	17	510	1	30	1	30	7	240
Max	259	7,756	425	12,751	23	690	27	810	15	450
Mean	53	1,584	92	2,768	11	343	10	296	11	345

Table A8. Number of successful random tows in NEFSC clam surveys used for survey trends and efficiency corrected swept area biomass. “Holes” (unsampled survey strata in some years) were filled by borrowing from adjacent surveys were possible (borrowed totals are negative numbers in gray-shaded boxes). Holes that could not be filled have zeros in black boxes. Survey strata are grouped by region. Survey strata not used for surfclams are not shown.

	Years												
Stratum	**1982**	**1983**	**1984**	**1986**	**1989**	**1992**	**1994**	**1997**	**1999**	**2002**	**2005**	**2008**	**2011**
SVA													
1	**-10**	10	14	7	10	10	10	10	**-10**	0	0	0	0
2	0	0	0	**-1**	1	2	1	1	**-1**	0	0	0	0
5	4	9	13	8	8	8	7	8	**-16**	8	8	**-17**	9
6	1	1	1	1	1	1	1	1	**-3**	2	1	**-1**	0
80	**-6**	6	9	3	7	7	8	7	**-7**	0	0	0	0
81	**-4**	4	7	3	5	5	5	5	**-10**	5	**-10**	5	0
DMV													
9	30	26	35	29	37	37	38	37	37	38	37	31	15
10	2	2	3	3	3	3	3	3	3	3	3	2	4
13	19	18	25	20	20	20	21	20	19	20	18	15	7
14	2	2	3	3	3	3	5	3	3	3	3	**-26**	23
82	1	1	1	1	1	1	1	1	2	2	**-3**	1	0
83	2	2	2	2	2	2	2	2	2	2	2	2	0
84	4	3	3	4	4	4	4	4	3	4	4	4	4
85	5	5	4	5	5	5	5	5	5	5	5	5	5
86	2	2	3	3	3	2	3	3	3	3	3	3	5
NJ													
17	11	11	18	12	12	12	12	12	12	12	12	12	5
18	3	3	**-6**	3	3	3	3	3	3	3	3	3	5
21	18	18	22	19	20	20	20	20	33	27	20	28	15
22	3	3	**-6**	3	3	3	5	3	3	3	3	3	5
25	9	9	13	8	9	9	9	9	8	9	9	13	8
26	2	2	**-5**	3	3	3	3	3	3	3	3	3	3
87	8	7	10	9	9	9	9	9	9	16	8	9	6
88	15	15	24	17	20	20	20	21	21	20	17	19	6
89	15	15	21	15	18	17	18	19	18	18	15	18	4
90	2	2	3	2	2	2	2	2	2	2	2	1	4

Table A8. Cont...

							Years						
Stratum	**1982**	**1983**	**1984**	**1986**	**1989**	**1992**	**1994**	**1997**	**1999**	**2002**	**2005**	**2008**	**2011**
							LI						
29	11	10	**-20**	10	10	10	10	10	10	10	10	16	10
30	7	8	**-14**	6	6	6	6	6	5	6	7	12	4
33	4	4	**-8**	4	4	4	5	4	4	4	4	10	4
34	2	2	**-4**	2	2	2	5	2	1	2	2	8	6
91	3	2	4	4	3	3	3	3	3	3	3	5	11
92	2	2	3	2	2	2	2	2	2	2	2	5	11
93	1	2	2	1	1	1	1	1	1	2	1	4	6
							SNE						
37	7	4	**-7**	3	**-6**	3	5	4	4	3	**-3**	3	2
38	3	2	**-5**	3	3	3	5	3	3	3	2	3	7
41	6	5	7	5	6	6	6	6	5	6	6	6	4
45	3	7	9	4	4	4	4	4	4	2	4	4	7
46	2	5	5	3	2	3	5	3	3	2	3	3	6
47	4	3	4	2	2	4	4	4	3	1	7	4	8
94	1	2	**-2**	0	**-1**	1	2	2	**-4**	2	**-2**	2	5
95	4	14	11	4	4	4	4	4	4	4	**-8**	4	5
96	**-12**	12	**-13**	1	1	3	2	4	**-4**	0	**-1**	1	0
							GBK						
54	0	**-3**	3	3	**-6**	3	3	3	**-3**	0	**-2**	2	2
55	3	**-3**	**-3**	3	1	3	3	3	2	2	**-4**	2	3
57	0	0	**-2**	2	1	2	5	2	2	2	**-4**	2	11
59	1	4	**-5**	1	2	6	5	5	4	5	**-9**	4	16
61	8	1	**-6**	5	**-12**	7	5	6	6	6	**-11**	5	5
65	0	0	**-3**	3	**-5**	2	4	3	**-4**	1	**-1**	1	3
67	0	**-5**	5	5	7	7	7	7	**-7**	0	**-2**	2	1
68	1	**-8**	7	3	6	6	5	5	**-5**	0	**-6**	6	0
69	2	5	**-11**	6	6	6	7	6	8	**-8**	**-4**	4	1
70	1	2	**-6**	4	**-8**	4	4	4	3	2	**-6**	4	19
71	0	**-2**	2	3	1	2	3	3	1	2	**-3**	1	3
72	2	**-10**	8	1	8	8	8	8	6	**-6**	**-4**	4	5
73	1	1	**-4**	3	6	6	6	6	5	6	**-9**	3	5
74	3	**-4**	1	3	**-7**	4	4	4	3	3	**-6**	3	11

Table A9. NEFSC clam survey stations for which the model predicted differential pressure below the threshold (35 PSI) for more than 25% of fishing seconds. These stations were not used in the current assessment.

Station	Strata	Depth	Lat	Lon	Region
143	13	42	38.27442	74.5733	DMV
145	14	54	38.30777	74.23925	DMV
70	87	27	39.06597	74.40457	NJ
254	26	48	39.88967	73.32147	NJ
46	26	65	40.14597	73.65233	NJ
31	29	33	40.43415	73.34963	LI
292	38	55	40.91837	71.60237	SNE
294	37	39	41.27432	71.40202	SNE
481	94	28	41.3911	71.23802	SNE
482	94	28	41.44353	71.38292	SNE
343	57	70	40.81365	68.01625	GBK
342	57	65	40.84938	68.01197	GBK
341	57	64	40.85402	68.0533	GBK
375	59	62	40.90093	67.91472	GBK
376	70	53	40.97942	67.84257	GBK
377	70	57	40.98083	67.77793	GBK
394	59	73	41.022	67.17712	GBK
390	59	59	41.10465	67.51712	GBK
391	59	58	41.14662	67.4156	GBK
409	73	46	41.43885	67.35357	GBK
419	74	53	41.79002	67.36272	GBK
430	72	54	41.9348	67.45007	GBK
180	23	55	38.89438	73.53642	OTH

Table A10. (On the following pages.) NEFSC clam survey data for surfclam abundance (mean N/tow) and biomass (mean kg/tow). Data are for three size groups: prerecruits (50-119mm), fishable clams (120+mm) and all clams greater than 50mm. Survey holes (strata with no sampling) are filled by borrowing, but no imputed data were used for this table.

		Prerecruits (50-119 mm SL)				**Large fishable (120+ mm SL)**				**All surfclams 50mm and above**						
	Year	N / Tow	CV	KG / Tow	CV	N / Tow	CV	KG / Tow	CV	N / Tow	CV	KG / Tow	CV	N Tows	Pos. Tows	N Strata
SVA	1982	3.53	0.88	0.19	0.90	3.73	0.92	0.404995	0.86	7.26	0.90	0.595757	0.872	25	6	5
	1983	6.60	0.62	0.35	0.64	5.71	0.62	0.649399	0.59	12.31	0.58	0.994758	0.565	30	12	5
	1984	7.85	0.37	0.43	0.40	21.82	0.31	2.536182	0.294	29.66	0.30	2.961469	0.287	44	17	5
	1986	1.50	0.35	0.08	0.42	22.20	0.75	2.413548	0.735	23.69	0.72	2.495099	0.72	23	13	6
	1989	3.11	0.75	0.11	0.70	9.78	0.83	1.199442	0.819	12.89	0.81	1.310352	0.808	32	13	6
	1992	18.15	0.86	1.22	0.91	12.10	0.77	1.279377	0.783	30.25	0.65	2.497773	0.648	33	18	6
	1994	43.38	0.46	1.03	0.31	6.38	0.44	0.656494	0.355	49.76	0.40	1.689041	0.276	33	19	6
	1997	10.31	0.44	0.42	0.46	0.49	0.46	0.047867	0.44	10.80	0.43	0.4673	0.448	32	14	6
	1999	9.32	0.41	0.33	0.36	1.22	0.46	0.134403	0.473	10.54	0.38	0.460503	0.331	47	21	6
	2002	13.69	0.61	0.49	0.62	5.66	0.55	0.641627	0.55	19.35	0.58	1.132064	0.565	15	7	3
	2005	3.65	0.66	0.07	0.57	0.00	0.00	0	0	3.65	0.66	0.068276	0.573	14	4	3
	2008	10.23	0.30	0.24	0.29	0.00	0.00	0	0	10.30	0.29	0.24407	0.286	18	11	2
	2011	15.40	0.29	0.38	0.28	0.14	1.00	0.010603	1	15.54	0.29	0.395325	0.27	9	8	1
DMV	1982	157.13	0.46	9.58	0.46	21.36	0.23	3.524782	0.32	178.49	0.42	13.10507	0.407	68	47	9
	1983	30.68	0.54	1.98	0.62	31.21	0.46	3.855335	0.364	61.88	0.49	5.831617	0.439	61	41	9
	1984	184.10	0.74	6.94	0.62	34.91	0.28	4.327025	0.276	219.01	0.63	11.26841	0.395	79	58	9
	1986	58.77	0.43	3.99	0.46	74.79	0.38	8.290292	0.326	133.56	0.39	12.278	0.365	70	53	9
	1989	16.71	0.54	1.02	0.55	31.24	0.26	3.782973	0.245	47.94	0.26	4.807792	0.233	78	53	9
	1992	13.49	0.28	0.75	0.38	28.86	0.29	3.591607	0.242	42.35	0.28	4.339855	0.258	77	58	9
	1994	68.70	0.33	3.57	0.43	60.96	0.21	7.35485	0.201	129.67	0.23	10.92903	0.218	83	66	9
	1997	77.18	0.17	4.30	0.20	54.53	0.24	6.127452	0.225	131.71	0.17	10.42328	0.19	82	64	9
	1999	29.61	0.28	1.94	0.28	26.36	0.22	3.002235	0.205	55.98	0.23	4.939529	0.21	78	47	9
	2002	16.47	0.28	0.75	0.27	20.70	0.21	2.756585	0.192	37.17	0.22	3.511343	0.186	81	58	9
	2005	6.44	0.42	0.31	0.43	4.76	0.26	0.616634	0.282	11.19	0.27	0.922988	0.237	75	45	9
	2008	9.61	0.23	0.36	0.25	2.64	0.35	0.361625	0.348	12.34	0.23	0.729765	0.266	89	50	9
	2011	43.27	0.25	1.78	0.29	9.32	0.40	0.98473	0.427	51.92	0.26	2.690627	0.309	66	37	9
NJ	1982	33.10	0.30	2.18	0.32	32.78	0.22	4.690181	0.212	65.88	0.19	6.874827	0.178	85	60	10
	1983	27.78	0.51	1.88	0.55	25.38	0.22	3.434296	0.207	53.16	0.30	5.319006	0.251	85	63	10
	1984	15.93	0.23	0.80	0.23	29.97	0.20	4.038403	0.186	45.90	0.18	4.835422	0.179	126	86	10
	1986	10.33	0.21	0.55	0.21	29.68	0.18	4.44884	0.18	40.01	0.17	4.999115	0.17	91	70	10

	1989	9.88	0.29	0.52	0.30	31.53	0.15	4.439793	0.134	41.40	0.15	4.964282	0.135	99	75	10
	1992	16.46	0.33	0.94	0.43	23.22	0.16	3.357078	0.152	39.68	0.20	4.297829	0.166	98	73	10
	1994	67.39	0.20	2.93	0.19	82.77	0.17	11.57065	0.167	150.16	0.16	14.50123	0.166	103	85	10
	1997	17.91	0.16	1.07	0.17	83.72	0.13	11.78592	0.121	101.63	0.13	12.85891	0.12	112	91	10
	1999	8.02	0.25	0.42	0.31	50.58	0.21	7.266118	0.189	58.60	0.21	7.689472	0.193	120	93	10
	2002	10.68	0.16	0.49	0.15	35.03	0.17	5.6948	0.165	45.71	0.14	6.188908	0.155	115	99	10
	2005	7.81	0.20	0.41	0.22	19.09	0.18	2.874266	0.17	26.90	0.16	3.283292	0.162	92	73	10
	2008	10.07	0.14	0.44	0.14	17.05	0.16	2.537086	0.168	27.11	0.13	2.97367	0.155	109	93	10
	2011	11.70	0.21	0.52	0.21	14.12	0.18	2.063531	0.192	25.82	0.16	2.586211	0.172	61	44	10

Table A10. Cont…

		Prerecruits (50-119 mm SL)				**Large fishable (120+ mm SL)**				**All surfclams 50mm and above**						
	Year	N / Tow	CV	KG / Tow	CV	N / Tow	CV	KG / Tow	CV	N / Tow	CV	KG / Tow	CV	N Tows	Pos. Tows	N Strata
LI	1982	0.03	1.00	0.002434	1	3.99	0.61	0.743364	0.606	4.03	0.61	0.745798	0.604	29	5	7
	1983	0.17	0.61	0.004333	0.613	0.41	0.72	0.057422	0.716	0.58	0.60	0.061755	0.688	29	4	7
	1984	0.56	0.30	0.020969	0.366	1.64	0.34	0.283652	0.353	2.20	0.22	0.304621	0.319	55	14	7
	1986	0.58	0.39	0.020603	0.403	1.72	0.61	0.305768	0.61	2.30	0.45	0.32637	0.567	29	8	7
	1989	2.24	0.87	0.088874	0.871	3.48	0.72	0.504931	0.726	5.72	0.78	0.593806	0.747	28	5	7
	1992	5.73	0.44	0.319383	0.476	2.54	0.33	0.295907	0.316	8.28	0.39	0.61529	0.373	28	10	7
	1994	4.23	0.17	0.211863	0.194	7.24	0.19	0.938826	0.208	11.48	0.17	1.150689	0.199	32	12	7
	1997	1.44	0.49	0.082004	0.533	4.17	0.64	0.604188	0.64	5.62	0.59	0.686193	0.622	28	6	7
	1999	1.61	0.64	0.048118	0.507	10.71	0.65	1.594682	0.607	12.32	0.65	1.6428	0.604	30	9	7
	2002	0.85	0.45	0.034689	0.439	1.94	0.67	0.331373	0.664	2.80	0.59	0.366062	0.636	29	8	7
	2005	1.42	0.34	0.062799	0.382	12.62	0.50	1.84611	0.479	14.04	0.47	1.908909	0.47	29	9	7
	2008	1.47	0.24	0.063645	0.236	3.52	0.24	0.534445	0.239	5.00	0.21	0.59809	0.23	60	22	7
	2011	4.57	0.26	0.156991	0.207	10.20	0.25	1.536774	0.253	14.76	0.21	1.693766	0.241	52	33	7
SNE	1982	2.58	0.29	0.131607	0.354	12.40	0.41	2.293756	0.418	14.99	0.33	2.425363	0.392	42	19	9
	1983	0.84	0.40	0.048743	0.435	7.88	0.39	1.712466	0.387	8.72	0.38	1.761209	0.385	54	24	9
	1984	0.81	0.36	0.042455	0.44	10.84	0.34	2.285845	0.336	11.65	0.34	2.3283	0.337	63	26	9
	1986	1.12	0.14	0.032305	0.252	4.12	0.68	0.872532	0.701	5.24	0.54	0.904837	0.678	25	11	8
	1989	1.18	0.43	0.051921	0.429	4.57	0.33	0.93215	0.332	5.75	0.31	0.984071	0.326	29	12	9
	1992	1.15	0.56	0.036055	0.482	2.49	0.58	0.558217	0.584	3.64	0.44	0.594272	0.55	31	9	9
	1994	1.26	0.52	0.077467	0.612	1.69	0.53	0.366591	0.549	2.96	0.45	0.444058	0.502	38	11	9
	1997	2.95	0.31	0.150038	0.362	12.28	0.30	2.555287	0.308	15.23	0.25	2.705325	0.298	34	15	9
	1999	2.60	0.42	0.102415	0.454	4.30	0.66	1.009042	0.663	6.90	0.45	1.111458	0.604	34	16	9

	2002	1.01	0.69	0.066557	0.719	3.85	0.27	0.825208	0.221	4.86	0.31	0.891765	0.229	24	9	8
	2005	1.33	0.08	0.052673	0.083	1.62	0.24	0.402845	0.241	2.95	0.14	0.455517	0.215	35	14	9
	2008	1.46	0.10	0.062659	0.126	5.01	0.63	1.03101	0.582	5.37	0.47	0.866775	0.545	32	11	9
	2011	1.35	0.09	0.051196	0.088	1.97	0.29	0.437128	0.278	3.07	0.18	0.434453	0.249	45	13	9
GBK	1986	20.00	0.79	0.783168	0.776	4.97	0.52	0.822095	0.549	24.97	0.68	1.605262	0.527	44	20	14
	1989	5.21	0.34	0.329709	0.425	24.86	0.73	3.523909	0.732	30.07	0.66	3.853617	0.704	75	37	14
	1992	15.54	0.40	0.800933	0.457	7.89	0.33	1.125339	0.342	23.43	0.33	1.926272	0.32	66	43	14
	1994	30.01	0.33	1.83765	0.347	45.84	0.39	6.734682	0.414	75.85	0.33	8.572331	0.375	70	47	14
	1997	58.55	0.31	3.402449	0.334	23.52	0.25	3.150657	0.245	82.07	0.28	6.553106	0.26	65	45	14
	1999	24.01	0.41	1.558739	0.416	29.59	0.31	3.945581	0.311	53.60	0.35	5.50432	0.337	59	34	14
	2002	22.09	0.52	1.358712	0.551	27.05	0.43	3.811007	0.417	49.15	0.46	5.169719	0.439	43	23	11
	2008	7.21	0.28	0.478127	0.335	33.02	0.25	4.605182	0.246	39.23	0.21	4.942882	0.224	45	29	14
	2011	7.62	0.21	0.513838	0.243	30.53	0.25	4.718915	0.246	43.79	0.24	6.109591	0.243	91	52	14

Table A11. Patch model results and approximate 95% confidence intervals for all surfclam depletion experiments conducted in 2011. The model for SC11-04 did not converge on a solution so no delta method confidence intervals are available.

Experiment	Tows	Density	CI	Efficiency	CI	Dispersion	CI
SC11-02	20	0.231	(0.14,0.25)	0.738	(0.53,0.90)	5.878	(2.95,10.65)
SC11-02S	18	0.184	(0.19,0.29)	0.556	(0.35,0.71)	4.904	(2.4,9.0)
SC11-03	15	0.416	(0.29,0.85)	0.571	(0.23,0.90)	4.156	(1.85,8.05)
SC11-04	17	0.163	NA	1	NA	6.438	NA

Table A12. F/V and R/V shell height composition data used to estimate NEFSC clam survey dredge selectivity for surfclams. Numbers of positive stations (e.g. R/V n positive stations) give the number of stations at which surfclams of each shell length group were captured. For example, “F/V lined dredge N positive stations” = 10 for the 20-29 mm SL group because individuals in the 20-29 mm size group were observed in F/V selectivity tows at 10 sites.

SL group	F/V lined dredge N	F/V unlined dredge N	R/V N	F/V lined dredge N positive stations	F/V unlined dredge N positive stations	R/V N positive stations
20-29	21	3	2	10	1	2
30-39	147	6	5	19	2	5
40-49	327	8	13	20	1	5
50-59	237	18	15	17	1	6
60-69	217	8	45	20	2	10
70-79	218	9	84	20	2	16
80-89	282	68	90	18	8	17
90-99	269	439	100	17	15	15
100-109	235	765	106	18	16	19
110-119	242	949	129	17	21	19
120-129	275	1256	132	18	21	20
130-139	227	1182	115	21	21	21
140-149	184	895	121	20	20	19
150-159	200	883	153	18	20	17
160-169	193	721	98	15	16	11
170-179	96	310	45	10	15	10
180-189	17	39	2	5	9	4
190-199	0	3	0	0	0	0

Table A13. Numbers of surfclams in survey dredge selectivity experiments by length bin and station (2011). For example, “3:8” means that 3 surfclams of a particular length at a particular station were measured in catches by the *R/V Delaware II* and 8 surfclams were measured in catches by the *F/V Pursuit*.

SL bin	Sta 7	Sta 23	Sta 28	Sta 34	Sta 43	Sta 49	Sta 50	Sta 51	Sta 52	Sta 53	Sta 56
6	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0
16	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:1	0:0	0:0	0:0
26	0:1	0:0	0:0	0:0	0:1	0:1	0:0	0:2	0:0	0:5	0:2
36	0:2	0:2	0:1	0:2	2:7	0:8	0:1	0:8	0:0	1:7	0:8
46	0:1	0:3	0:4	0:5	0:8	0:8	0:0	0:12	0:0	1:5	0:1
56	0:2	0:4	0:2	0:8	1:9	0:12	0:0	0:5	0:1	1:12	0:0
66	0:1	0:1	1:1	0:2	1:10	1:9	1:1	0:3	0:0	0:6	0:3
76	2:3	0:0	0:1	0:7	2:2	4:4	2:0	1:7	2:0	2:5	2:5
86	2:1	0:0	0:0	2:5	0:1	0:3	2:2	1:2	1:1	3:5	0:1
96	1:1	4:1	0:0	0:3	2:2	0:2	1:1	1:4	1:1	0:1	1:4
106	3:2	2:1	1:0	3:3	3:2	3:3	1:0	5:3	1:1	3:5	1:3
116	2:2	3:1	3:0	2:5	2:3	3:0	1:0	4:6	0:0	4:2	1:1
126	9:1	4:3	3:0	3:8	1:3	5:4	2:1	8:8	1:0	1:3	2:1
136	10:6	4:2	6:3	10:10	4:6	6:9	3:1	5:9	2:3	5:8	2:2
146	11:8	4:4	6:7	3:8	5:5	7:9	3:3	3:6	0:3	5:8	4:2
156	9:7	7:4	8:5	7:8	6:4	8:10	1:8	9:9	3:4	6:10	9:4
166	6:7	2:0	8:2	5:9	3:4	6:9	2:3	4:6	1:7	5:9	9:9
176	2:1	0:0	4:0	2:7	2:3	6:3	0:0	0:1	0:2	4:6	6:8
186	0:0	0:0	0:0	0:4	0:1	0:0	0:0	0:0	0:0	0:1	0:1

SL bin	Sta 141	Sta 156	Sta 167	Sta 234	Sta 236	Sta 239	Sta 240	Sta 247	Sta 255	Sta 279
6	0:0	0:1	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0
16	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0
26	1:6	0:1	0:2	0:0	0:1	0:0	1:1	0:2	0:1	0:1
36	1:9	2:13	0:3	1:5	0:2	0:2	0:13	0:1	0:12	0:4
46	5:10	1:15	0:3	1:9	1:12	0:1	1:11	0:0	0:6	0:3
56	6:9	3:11	0:2	0:7	1:3	0:2	1:0	0:3	0:8	0:9
66	9:12	7:12	1:3	1:7	0:3	0:9	3:5	1:8	6:8	0:4
76	8:12	6:12	2:2	1:7	0:4	2:7	6:11	2:7	9:9	2:9
86	10:11	8:10	1:2	8:10	1:1	6:11	7:11	3:9	10:11	1:9
96	10:8	8:12	3:1	4:10	0:0	7:11	4:10	3:9	9:11	0:5
106	11:9	6:12	3:2	5:10	1:1	5:10	5:9	2:6	6:9	0:2
116	12:11	6:12	4:3	4:10	3:0	7:9	3:9	5:9	12:10	0:5
126	9:10	5:12	3:1	2:9	0:1	7:11	3:7	4:8	10:8	1:4
136	3:4	3:5	2:2	2:8	4:1	5:9	2:9	8:10	5:3	5:4
146	2:2	0:3	3:2	1:8	3:1	6:8	1:4	5:6	1:2	0:4
156	0:0	1:0	0:0	0:3	1:1	0:4	2:1	4:6	0:0	0:6
166	0:0	0:0	0:0	0:2	0:3	0:0	0:0	0:2	0:0	0:4
176	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:1
186	0:0	0:1	0:0	0:0	0:0	0:0	0:0	0:0	0:0	0:0

Table A14. Estimated model parameters and (standard errors) for a selection of competing models predicting clam meat weight from shell length. Region effects are highlighted with colors corresponding to the row of the model they were estimated in.

Formula	Intercept	Length	Depth	Density	Region	AIC	BIC
MW ~ Len+(1|Sta)	-8.6041 (0.00941)	2.7249 (0.01431)				4911	4928
MW ~ Len+Dpth+(1|Sta)	-8.3705 (0.00934)	2.7227 (0.01433)	-0.0644 (0.0263)			4908	4930
MW ~ Len+(Len+1|Sta)	-8.6406 (0.0097)	2.7336 (0.02425)				4715	4742
MW ~ Len+Dpth+(Len+1|Sta)	-8.6236 (0.00966)	2.73 (0.02423)	-0.0614 (0.02721)			4712	4745
MW ~ Len+Reg+(Len+1|Sta)	-8.6383 (0.0174)	2.7276 (0.0245)			a	4695	4756
MW ~ Len+Dens+(Len+1|Sta)	-8.6347 (0.01001)	2.7363 (0.02445)	-0.00572 (0.00688)			4716	4749
MW ~ Len+(Len+1|Sta)+(Len+1|Yr)	-8.611 (0.0244)	2.7277 (0.04988)				4706	4750
MW ~ Len+Dpth+(Len+1|Sta)+(Len+1|Yr)	-8.3439 (0.02602)	2.7237 (0.04939)	-0.0714 (0.02675)			4701	4750
MW ~ Len+Reg+(Len+1|Sta)	-8.6383 (0.0174)	2.7276 (0.0245)			b	4695	4756
MW ~ Len+Dpth+Reg+(Len+1|Sta)	-7.976 (0.01687)	2.7175 (0.02426)	-0.1743 (0.03104)		c	4667	4734
MW ~ Len+Dpth+Reg+(Len+1|Sta)+(Len+1|Yr)	***-7.8622 (0.03454)***	***2.7061 (0.05402)***	***-0.1925 (0.02999)***		***d***	***4645***	***4728***
MW ~ Len+Dpth+Dens+Reg+(Len+1|Sta)+(Len+1|Yr)	-7.8391 (0.03551)	2.71 (0.05461)	-0.1951 (0.02983)	-0.0661 (0.06804)	e	4644	4732

Region	a	b	c	***d***	e
SVA	0.044 (0.07141)	0.044 (0.07141)	0.0129 (0.07043)	***-0.06 (0.06786)***	0.1714 (0.04491)
DMV	0	0	0	***0***	0
NJ	0.0162 (0.02251)	0.0162 (0.02251)	-0.00407 (0.02194)	***0.00247 (0.02111)***	-0.0824 (0.0308)
LI	-0.0219 (0.0307)	-0.0219 (0.0307)	-0.0889 (0.03172)	***-0.0816 (0.03101)***	0.2049 (0.03058)

SNE	0.1869 (0.04799)	0.1869 (0.04799)	0.1651 (0.04597)	***0.1808 (0.04497)***	-0.2668 (0.31418)
GBK	0.1141 (0.03001)	0.1141 (0.03001)	0.1792 (0.03096)	***0.2009 (0.03072)***	-0.0104 (0.0063)
OTH	-0.261 (0.32725)	-0.261 (0.32725)	-0.1631 (0.32651)	***-0.246 (0.31299)***	0.00636 (0.02111)

Table A15. Number of age samples by region and survey year.

Year	SVA	DMV	NJ	LI	SNE	GBK
1982	5	796	927	40	123	4
1983	142	422	934	6	369	0
1984	0	0	0	0	0	643
1986	64	748	1216	45	71	413
1989	60	102	566	53	42	86
1992	11	134	257	47	54	311
1994	0	299	476	0	0	0
1997	0	626	227	0	0	50
1999	0	510	496	22	50	178
2002	29	327	779	31	20	54
2005	17	322	523	21	6	0
2008	0	138	459	99	39	105
2011	26	122	144	72	17	82

Table A16. Growth curve (Von Bertalanffy) parameter estimates and standard errors for each region, by year.

Region	Year	n	L_{max}	L_{max} se	K	K se	t_0	t_0 se
DMV	1978	199	163.562	1.820	0.319	0.017	-0.010	0.096
DMV	1980	391	166.575	1.289	0.340	0.020	1.246	0.150
DMV	1981	446	173.336	1.855	0.248	0.014	0.451	0.154
DMV	1982	801	175.458	1.641	0.205	0.008	0.114	0.129
DMV	1983	564	176.522	2.512	0.214	0.013	0.113	0.190
DMV	1986	812	183.819	3.002	0.135	0.010	-1.204	0.366
DMV	1989	162	141.828	2.541	0.327	0.045	0.596	0.316
DMV	1992	145	172.122	6.760	0.161	0.025	-0.829	0.473
DMV	1994	299	149.550	1.661	0.343	0.022	1.437	0.134
DMV	1997	626	151.399	3.251	0.148	0.014	-1.472	0.395
DMV	1999	510	136.421	1.924	0.238	0.027	-0.314	0.482
DMV	2002	356	156.831	4.395	0.168	0.021	-1.223	0.434
DMV	2005	339	150.595	2.750	0.161	0.012	-0.735	0.235
DMV	2008	228	158.314	2.583	0.201	0.014	-0.607	0.197
DMV	2011	149	120.448	3.027	0.399	0.051	0.301	0.225
NJ	1978	289	163.504	2.858	0.313	0.025	0.207	0.147
NJ	1980	452	171.610	1.564	0.286	0.015	0.825	0.139
NJ	1981	641	170.430	1.330	0.316	0.013	0.703	0.094
NJ	1982	927	173.358	1.431	0.264	0.009	0.256	0.087
NJ	1983	934	176.348	1.733	0.244	0.010	0.267	0.109
NJ	1986	1216	175.558	1.866	0.177	0.008	-0.465	0.174
NJ	1989	566	162.936	2.012	0.238	0.015	0.585	0.183
NJ	1992	257	166.971	4.115	0.187	0.023	-0.422	0.432
NJ	1994	476	159.587	2.181	0.197	0.017	-0.580	0.356
NJ	1997	227	165.551	2.053	0.212	0.018	-0.046	0.291
NJ	1999	496	160.889	1.379	0.264	0.015	0.235	0.172
NJ	2002	779	163.876	1.728	0.209	0.015	-0.838	0.279
NJ	2005	523	164.111	2.418	0.150	0.013	-1.211	0.455
NJ	2008	807	158.901	2.251	0.152	0.011	-1.458	0.320
NJ	2011	145	154.582	3.475	0.216	0.031	-0.367	0.555
LI	1980	29	159.445	2.372	0.365	0.055	0.451	0.396
LI	1981	27	171.114	17.901	0.108	0.065	-5.719	4.260
LI	1982	40	156.713	1.856	0.800	0.213	2.815	0.198
LI	1986	45	165.899	3.402	0.222	0.039	0.023	0.695
LI	1989	53	163.122	3.557	0.259	0.034	0.529	0.394
LI	1992	47	155.779	3.029	0.307	0.036	0.008	0.314
LI	1999	22	167.863	4.719	0.302	0.044	0.550	0.283
LI	2002	31	174.942	8.130	0.250	0.059	0.313	0.594
LI	2005	21	160.095	7.630	0.210	0.070	-0.598	1.226
LI	2008	254	150.733	2.409	0.409	0.038	0.830	0.182

LI	2011	73	168.560	5.403	0.196	0.049	-0.784	1.258
SNE	1980	61	177.066	6.484	0.111	0.038	-7.483	3.807
SNE	1981	38	162.605	3.761	0.444	0.088	1.335	0.311
SNE	1982	123	160.352	2.398	0.222	0.025	0.642	0.378
SNE	1983	369	167.890	1.656	0.265	0.023	-0.209	0.350
SNE	1986	71	163.625	2.624	0.316	0.038	1.571	0.258
SNE	1989	42	171.995	5.179	0.422	0.079	2.009	0.350
SNE	1992	54	162.448	2.304	0.203	0.024	0.586	0.317
SNE	1999	50	174.800	6.337	0.210	0.041	-0.084	0.560
SNE	2002	20	162.292	5.311	0.452	0.118	1.539	0.525
SNE	2008	103	171.954	2.818	0.172	0.023	-1.036	0.677
SNE	2011	18	168.488	23.305	0.058	0.267	-37.007	193.965
GBK	1984	643	146.693	3.221	0.266	0.022	0.871	0.153
GBK	1986	413	148.950	3.236	0.225	0.019	0.267	0.175
GBK	1989	86	152.814	5.196	0.197	0.040	-0.250	0.765
GBK	1992	311	148.733	2.815	0.270	0.020	1.085	0.155
GBK	1997	50	138.772	7.371	0.194	0.045	-0.007	0.683
GBK	1999	178	145.613	3.129	0.355	0.033	0.581	0.160
GBK	2002	54	143.216	4.762	0.427	0.095	2.136	0.416
GBK	2008	315	147.423	2.587	0.204	0.023	-0.654	0.387
GBK	2011	83	146.346	2.053	0.486	0.189	2.249	1.109

Table A17. Points made to support splitting the Atlantic surfclams into two stocks with counterpoints. The status quo is a single stock and the alternative is two stocks with the break southwest of Georges Bank. Under this option, the Georges Bank (GBK) stock in the north would be separated from the South Virginia/ North Carolina to Southern New England (SVASNE) stock in the south. Points made to support maintaining the status quo and counterpoints are listed in Table A18.

Pro	Con	References
Spatial Patterns in Biological and Other Characteristics		
Growth curves and shell length-meat weight differ markedly between GBK and the southern region.	The differences are clinal or continuous and the split could be made elsewhere or not at all.	Table Table A14, Table A16, Figure A57, A58-62; Kim and Powell (2004); Marzec, et al. (2006); Weinberg (2005)
Post-settlement survival has decreased in the south but not on GBK.	Southern and northern portions of a large stock should respond differently to environmental change. The differences are clinal or concentrated in shallow water south of New Jersey and the split could be made elsewhere or not at all.	NEFSC 2010
Georges Bank tends to retain larvae spawned there due to a persistent gyre current. Published larval drift models for scallops show substantial movement of larvae from GBK to the south, but none from the south to GBK. A detailed unpublished surfclam larval drift presented to the Working Group indicates no movement of larvae from GBK to Southern New England and other southern areas occurs or *vice-versa* assuming no daily mortality during the assumed 35 day larval lifetime observed in culture (X. Zhang and D. Haidvogel, IMCS, Rutgers).	Larval drift models are not definitive and do not cover the whole time period of interest or all possible oceanographic conditions when substantial interchange may occur, particularly between GBK and Southern New England which is directly to the south. In certain circumstances, up to 10% of GBK larvae would reach Southern New England and these larvae would be 'unsuccessful' in the model, but near a reasonable size for metamorphosis in a biological sense.	Miller et al 1998; Werner et al 1993; Gilbert et al 2010; Tian et al 2009; Table A19
Georges Bank and MAB surfclam habitats are entirely within different and well recognized eco-regions.		Fogarty et al. (2011)

The split south of GBK crosses an area that separates the two major concentrations of the resource in the south (off New Jersey) and on GBK.	The split could be made elsewhere or not at all.	Appendix A7
Population Dynamics		
Surfclams in GBK and south resemble two independent populations based on abundance, recruitment and life history trends.	The northern and southern portions of SVASNE differ as well, why not identify three stocks?	POPULATION DYNAMICS (Figures A26, A27, A74, A75, A77 and A78)
Strong year classes occur independently and more often in the south and often over wide areas within the region.	Recruitment patterns are regional and the split could be made elsewhere or not at all.	Fig A67
Fishery Patterns		
The split south of GBK crosses an area of relatively low fishing activity and catch.		See Table A3, Figures A3,A4, and A8
Practical		
The new cooperative survey cannot sample the whole resource in one year but can be extended to include all of the SVASNE area.	Does not mean the split has to be made at GBK. Spatially explicit assessment models could be developed to handle areas incompletely sampled in annual surveys.	
Including GBK in a whole stock assessment model means that certain survey years cannot be included because GBK was not sampled in all years.	Areas can modeled separately but managed together, with results combined.	
Previous reviews of the surfclam assessment have been critical of the current stock definition.	Restoration of fishing on GBK invalidates some of these previous criticisms.	
The proposed boundary is along lines historically used to assess the stock and to collect survey data.	Historical use and best practice are not necessarily the same.	
Utility of Biological Reference Points		
”Average” biological reference points for two quasi-populations with different population dynamics do not result in MSY for either population unit, particularly when differences are as large as for GBK and the southern region.	The same argument can be made with respect to different portions of the southern area.	Hart, D. R. 2001. Can. J. Fish. Aquat. Sci. 58:2351–2358.

The surfclam stock could be removed entirely in the south or on GBK without triggering an overfishing or overfished status determination because biomass would remain > Bmsy/2 for the combined areas.	This scenario is unlikely to occur in either GBK or the southern area now that GBK is open to fishing	
Combining two quasi-populations with different population dynamics obscures the condition of both.	Assessments should contain information about both stock components and other important regions, regardless of stock definitions.	

Table A18. Points made to support maintaining the status-quo (single) stock definition for surfclams, with counterpoints. The status quo is a single stock and the alternative is two stocks with the break just southwest of Georges Bank.

Pro	Con	References
Split is a needless departure from historical precedent.	Historical precedent is not necessarily best practice particularly given biological and ecological changes.	
Scallops and ocean quahogs (other sessile bivalves) are managed as one stock	Many species (lobsters and relatively sessile fish such as goosefish and flounders) with interconnected meta-populations are managed as separate stocks. Precedent does not define best practice.	
Split made at the proposed point is not optimal - this aspect should be studied further before management action occurs	GBK is the most distinct region based on biological characteristics, oceanography, geography, larval dispersal and general ecological classifications. Additional divisions in the south can be made later if warranted.	
No genetic differences were found among samples of surfclams from Georges Bank to Virginia.	Lack of significant differences in genetic studies does not prove population homogeneity.	Weinberg, J.W. 2005. Mar. Biol. 146(4): 707-716
Recruitment in SNE may come from GBK at periods that have not been observed in models	There is insufficient age data for SNE to evaluate this hypothesis. However, the limited available data indicate that recruitment patterns differ between the major population centers (GBK in the north and New Jersey and Delmarva in the south).	TABLE A19

Table A19. Summary of unpublished results from surfclam larval drift simulation study courtesy of X. Zhang and D. Haidvogel (IMCS, Rutgers). Tables show the percentage of settlers released (columns) that settled successfully in each area (row) over 35 simulated days (the approximate larval stage duration) assuming no larval mortality. For example, of all the larvae released on Georges Bank, about 9.4% had settled on Georges Bank by the end of 35 days and none had settled elsewhere. Larvae were released from all major areas of surfclam habitat at five day intervals from May 21 to October 16, 2006-2009 (30 release dates) with results from all years and release dates summarized below. The size of each simulated larva was tracked in the model and larvae grew at a rate that depended on age, temperature and available food concentrations. Simulated larvae moved passively in horizontal directions but vertical movements were active at speeds dependent on size and water temperature. Larvae settled after they reached 260 µm, reached habitat with suitable water temperatures. They were considered dead if they had not settled in 35 days. The Regional Ocean Modeling System (ROMS) model used in simulations included forcing by rivers, tides, wind, radiation, air temperatures, humidity, etc. with a spatial resolution of 8 x 12 Km (120 x 160) grids.

		Release area (south on left, north on right)					
		Southern Virginia	DelMarva	New Jersey	Long Island	Southern New England	Georges Bank
		All years					
Settlement area (south bottom, north top)	Georges Bank	0	0	0	0	0	19.3556
	Southern New England	0	0	0	0.0167	0.3667	0
	Long Island	0	0	0.2130	37.1663	0.3333	0
	New Jersey	0	0.0683	78.7130	88.6910	0.1750	0
	DelMarva	1.9334	40.6430	80.9640	8.2167	0	0
	Southern Virginia	40.0997	85.8250	12.2463	0	0	0

Table A20. Structure of SS3 models used for surfclams in the southern and GBK areas.

Model aspect	Southern area	GBK area	Note
Natural mortality (M)	$0.15\ y^{-1}$		Constant for all ages and all years
Age bins	0-32+ y	0-30+ y	Few ages ≥ 30+ y
Population length bins	1, 2, … 19, 20 cm SL		
Time	1965-2011	1984-2011	South: starts first year with catch data and 17 y before first survey in 1982. North: starts first year with survey and catch data.
Seasons/ subareas/ morphs	None		
Commercial fleets	1		
Fishery size selectivity	Double normal (dome shaped), five parameters estimated and assumed constant over time	Double normal (logistic shaped) with left hand side from parameters estimated for south	Not estimable for GBK because of noisy and limited (2010-2011) commercial size data
Surveys	1 (2 variants)		NEFSC clam survey and minimum swept-area abundance based on clam survey data
Survey trend size selectivity	Field estimates		Double-normal selectivity curve fit externally to original GAM model estimates from field data (see parameter table)
Survey trend catchability	Estimated	Estimated	
Minimum swept area biomass size selectivity	Mirrors (same as) survey trend size selectivity		
Minimum swept area biomass catchability (capture efficiency)	Mean unbiased log scale parameter with normal prior	Fixed at estimate for southern area	Trend ignored in fitting model (weight 10-5) but catchability is calculated and compared to prior
Recruit model	Beverton-Holt with fixed steepness=0.95, estimate virgin recruitment and recruit variance		In effect, recruitments vary randomly around a constant mean estimated in the model and with a variance estimated in the model. Steepness is not important because biomass has never been low.
Recruit dev years	1965-2013	1969-2011	
Last early year with no bias adjustment	1919	1959	Adjusted based on preliminary fits
First year no full bias adjustment	1969	1974	
Last year full bias adjustment	2008	2006	
First recent year no bias adjustment	2012	2013	
Max bias adjustment	0.97	0.87	
Fishing mortality method	Hybrid method, 6 iterations (exact F)		Use Pope's approximation next time for speed if fishing mortality estimates remain low

Table A21. Parameters estimated internally and externally in SS3 models for surfclams in the southern and GBK regions. Numbers of parameters are summarized in the last rows.

Parameter	**Southern area**	**SD (if estimated)**	**GBK area**	**CV (if estimated)**	**Note**
M at ages 5 and 30 y	0.15	n/a	Same as south		
Length at age 4	10.245	0.045431	9.3017	0.10797	
Length at age 30	16.019	0.068704	14.846	0.11077	
Von Bertalanffy K	0.22379	n/a	0.253	n/a	
SD of size at ages 5 and 30 y	1.84	n/a	Same as south	n/a	
Shell length-meat weight					
Multiplier	0.000094	n/a	0.0001055	n/a	
Exponent	2.73325	n/a	2.73325	n/a	
Spawner-recruit					
Log virgin recruitment (R0)	14.893	0.13793	13.867	0.19071	
Steepness	0.95	n/a	Same as south		
Standard deviation	0.61803	0.064875	0.77469	0.086266	
Initial fishing mortality	0.016052	0.0024872	0	n/a	
Log catchability (capture efficiency) for swept area abundance	-1.1086	n/a	Same as south		This is a dummy parameter for comparison to capture efficiency prior
Size selectivity - fishery					
Peak	15.519	0.10544	15.4	n/a	GBK fishery selectivity parameters for left-hand side of double normal selectivity curve are fixed at same values as south. Parameters for right-hand side are fixed at values to ensure asymptotic pattern
Top	-9.7169	7.9249	10	n/a	
Asc-width	1.5949	0.076367	1.61	n/a	
Dsc-width	1.1254	0.1768	10	n/a	
Init	-999	n/a	-999	n/a	
Final	-999	n/a	-999	n/a	
Size selectivity - survey trend and swept-area abundance					
Peak	8.81897	n/a	Same as south		Estimated externally by fitting the double normal selectivity function to selectivity at size estimates from a mixed-effects GAM model.
Top	-0.64891	n/a			
Asc-width	2.23919	n/a			
Dsc-width	2.3557	n/a			
Init	-999	n/a			
Final	-0.817434	n/a			
N estimated parameters excluding recruit deviations	9		4		
N estimated recruit deviations	47		43		
Total N estimated parameters	56		47		

Table A22. Growth parameter estimates and goodness of fit from preliminary SS3 model runs for surfclams in the southern region. The lowest negative log likelihood values are shown in bold and the models are sorted from left (poorest fit) to right (best fit).

Statistic or growth parameter	Southern growth pars, normal prior on log q	Estimate Growth SD@Lmax	Estimate Lmax	Estimate K	Estimate Lmax and K	Estimate Growth SD@Lmin	Estimate both size@age SD	Estimate Lmin	Estimate Lmin and SD@Lmin	Estimate Lmin and Lmax	Estimate Lmin and K	Estimate all growth pars
NLL	1,248	1,245	1,241	1,235	1,234	1,216	1,205	1,167	1,166	1,156	1,128	1,122
Lmin	10.99	10.99	10.99	10.99	10.99	10.99	10.99	**11.79**	**11.76**	**11.81**	**11.91**	**11.97**
Lmax	16.19	16.19	**15.82**	16.19	**16.07**	16.19	16.19	16.19	16.19	**15.79**	16.19	**16.34**
K	0.22	0.22	0.22	**0.17**	**0.18**	0.22	0.22	0.22	0.22	0.22	**0.13**	**0.13**
SD min	1.84	1.84	1.84	1.84	1.84	**2.09**	**2.13**	1.84	**1.89**	1.84	1.84	**1.80**
SD max	1.84	**1.72**	1.84	1.84	1.84	1.84	**1.60**	1.84	1.84	1.84	1.84	**1.70**

Table A23. Goodness of fit for two preliminary SS3 models with likelihood weights on survey trend: lambda=1 and lambda=100. The lowest negative log likelihood values are shown in bold.

Label	Lambda = 1	Lambda = 100
Recruitment	**2.132**	10.016
Parm_priors	**0.051**	0.220
Survey trend	-3.768	**-7.582**
Lengths		
Fishery	**197.2**	199.4
Survey	**163.0**	176.7
Survey ages	**1,748**	1,873
Naked sum	**2,107**	2,251

SWAN Q=efficiency	0.19	0.27

B2011	1,020,610	611,096
B2011/B1999	0.49	0.36

Table A24. Data used in SS3 models for surfclams in the southern and GBK areas.

Data type	Southern area	GBK area	Note
Catches (mt meat weight)	1965-2011		Landings+discard+12% assumed incidental mortality
Historical catches (used to calculate initial biomass)	Average 1965-1969 = 12,802 mt		Landings+discard+12% assumed incidental mortality
Fishery length composition, 3-18 cm SL in 1 cm bins	N=30: 1982- 2011	N=2: 2010-2011	Southern area size data for 1982 and 1999 down-weighted (effective N=10).
Fishery age data	None		
Survey abundance data	N=13: 1982-1984, 1986, 1989, 1992, 1994, 1997, 1999, 2002, 2005, 2008, 2011	N=10: 1984, 1986, 1989, 1992, 1994,1997, 1999, 2002, 2008, 2011	Mean numbers per tow, without adjustments based on sensor data
Survey length data, 3-18 cm in cm bins	Same as survey abundance data		Southern area size data for 1984 downweighted (effective N=10) due to very large catch of surfclams almost entirely 7-8.9 cm SL
Survey age data (0-30+ y in 1 year age bins)	N=10: 1982-1983, 1986, 1989, 1992, 1999, 2002, 2005, 2008, 2011	N=9: 1984, 1986, 1989, 1992, 1997,1999, 2002, 2008, 2011	Age data were not collected from entire southern and GBK areas during some years
Minimum swept area abundance	N=6: 1997, 1999, 2002, 2005, 2008, 2011	N=5: 1997, 1999, 2002, 2008, 2011	Survey catches adjusted on a station-specific basis for tow distance using sensor data, total area adjusted for unsuitable habitat, bad tows discarded
Survey timing	0.51		Mean Julian date / 365
Likelihood weights	All 1.0 except 10^{-5} for minimum swept area abundance trend		
Initial growth parameters	External estimates		External estimates using all available age data for each region. L_{min} and L_{max} were estimated in final models (see parameter table) while other growth parameters were left at initial values.
Maturity	50% mature at age 2 1		Information about age specific fecundity limited
Age reader precision	Age data assumed unbiased with standard deviations for ageing errors increasing linearly from 0.144 y at age 0 y to 0.531 y at age 30 y		Based on between age reader comparison experiments and QA/QC experiments (ages read twice by same reader). All age data were collected by same reader.
Shell length - meat weight	External estimates		Estimates (ignoring depth effects) updated in this assessment

Table A25. Biomass (ages 6+ y or approximately 120+ mm SL, thousand mt), recruitment (10^9 age zero surfclams) and fully recruited fishing mortality (F) estimates from SS3 for the **southern area** with CVs.

Year	Biomass	CV.B	Recruitment	CV.R	F	CV.F
Virgin	1250	0.14	2937	0.14	NA	NA
1964	1160	0.14	2937	0.14	NA	NA
1965	1160	0.14	2133	0.22	0.02	0.16
1966	1157	0.14	2354	0.20	0.02	0.16
1967	1154	0.14	1767	0.21	0.02	0.16
1968	1155	0.14	2005	0.19	0.01	0.16
1969	1157	0.14	1515	0.20	0.01	0.15
1970	1162	0.14	1109	0.22	0.01	0.15
1971	1135	0.14	1109	0.21	0.03	0.15
1972	1101	0.14	1321	0.19	0.04	0.15
1973	1044	0.14	1958	0.18	0.05	0.16
1974	990	0.15	2319	0.17	0.06	0.16
1975	922	0.15	2917	0.17	0.04	0.16
1976	856	0.15	6987	0.16	0.04	0.16
1977	794	0.15	10658	0.15	0.04	0.17
1978	746	0.15	7661	0.16	0.03	0.17
1979	733	0.15	7911	0.15	0.03	0.17
1980	738	0.15	9529	0.15	0.04	0.17
1981	768	0.15	4859	0.16	0.05	0.17
1982	950	0.15	3995	0.16	0.04	0.17
1983	1277	0.15	4278	0.16	0.03	0.17
1984	1484	0.15	2822	0.18	0.03	0.17
1985	1684	0.15	2621	0.19	0.02	0.17
1986	1929	0.15	4001	0.18	0.02	0.17
1987	1974	0.15	3253	0.18	0.02	0.17
1988	1967	0.15	3094	0.19	0.02	0.17
1989	1956	0.15	3915	0.18	0.02	0.17
1990	1880	0.16	2607	0.19	0.02	0.17
1991	1789	0.16	3034	0.19	0.02	0.17
1992	1756	0.16	4698	0.18	0.02	0.17
1993	1696	0.16	3428	0.18	0.02	0.17
1994	1634	0.16	1712	0.19	0.02	0.17
1995	1608	0.16	1236	0.20	0.02	0.17
1996	1539	0.16	1672	0.19	0.02	0.17
1997	1490	0.16	1738	0.19	0.02	0.17
1998	1511	0.17	2998	0.19	0.02	0.17
1999	1488	0.17	2759	0.19	0.02	0.18
2000	1399	0.17	1465	0.20	0.02	0.18
2001	1294	0.17	552	0.24	0.03	0.18
2002	1207	0.17	849	0.22	0.03	0.18
2003	1128	0.18	851	0.23	0.04	0.18
2004	1104	0.18	1438	0.22	0.04	0.19
2005	1079	0.18	2240	0.21	0.03	0.19
2006	1013	0.18	2027	0.23	0.04	0.19
2007	912	0.19	1906	0.25	0.05	0.20
2008	827	0.19	1594	0.27	0.05	0.20
2009	750	0.19	2115	0.31	0.04	0.21
2010	706	0.20	3017	0.39	0.04	0.21
2011	703	0.20	1704	0.55	0.04	0.21

Table A26. Biomass (ages 7+ y or approximately 120+ mm SL, thousand mt), recruitment (10^9 age zero surfclams) and fully recruited fishing mortality (F) estimates from SS3 for the **northern (i.e., GBK) area** with CVs.

Year	Biomass	CV.B	Recruitment	CV.R	F	CV.F
1982	380	0.19	1053	0.19	0.00	0.00
1983	380	0.19	1053	0.19	0.00	0.00
1984	504	0.20	2056	0.24	0.01	0.20
1985	508	0.19	949	0.32	0.01	0.20
1986	522	0.19	1383	0.28	0.01	0.21
1987	523	0.19	1520	0.27	0.00	0.21
1988	532	0.18	1707	0.26	0.00	0.20
1989	521	0.19	1041	0.31	0.00	0.20
1990	518	0.19	1000	0.31	0.00	0.20
1991	541	0.19	750	0.35	0.00	0.00
1992	522	0.19	883	0.38	0.00	0.00
1993	520	0.16	3289	0.25	0.00	0.00
1994	522	0.16	3597	0.24	0.00	0.00
1995	532	0.18	1636	0.29	0.00	0.00
1996	517	0.17	1553	0.27	0.00	0.00
1997	500	0.17	1469	0.29	0.00	0.00
1998	475	0.17	1583	0.31	0.00	0.00
1999	456	0.18	849	0.39	0.00	0.00
2000	528	0.18	241	0.62	0.00	0.00
2001	610	0.18	354	0.54	0.00	0.00
2002	616	0.18	314	0.55	0.00	0.00
2003	616	0.18	234	0.51	0.00	0.00
2004	610	0.18	319	0.39	0.00	0.00
2005	608	0.18	356	0.33	0.00	0.00
2006	578	0.18	380	0.35	0.00	0.00
2007	526	0.18	300	0.43	0.00	0.00
2008	481	0.18	156	0.57	0.00	0.00
2009	437	0.18	171	0.58	0.00	0.19
2010	394	0.18	240	0.62	0.00	0.19
2011	357	0.18	385	0.69	0.01	0.19

Table A26B. Biomass (approximately 120+ mm SL, thousand mt), recruitment (10^9 age zero surfclams) and fully recruited fishing mortality (F) estimates from SS3 for the **whole stock** with CVs.

Year	Biomass	cv	Recruitment	cv	F	cv
1982	1331	0.12	5048	0.14		
1983	1657	0.12	5331	0.14		
1984	1987	0.12	4878	0.15	0.021	0.166
1985	2191	0.13	3570	0.16	0.019	0.164
1986	2451	0.13	5384	0.15	0.018	0.261
1987	2497	0.13	4773	0.15	0.016	0.261
1988	2500	0.13	4801	0.15	0.016	0.262
1989	2477	0.13	4956	0.16	0.015	0.262
1990	2398	0.13	3607	0.16	0.017	0.262
1991	2330	0.13	3783	0.17	0.015	0.262
1992	2278	0.13	5581	0.16	0.016	0.262
1993	2216	0.13	6717	0.15	0.016	0.165
1994	2156	0.13	5309	0.17	0.017	0.166
1995	2140	0.13	2872	0.19	0.015	0.167
1996	2055	0.13	3225	0.16	0.016	0.168
1997	1990	0.13	3207	0.17	0.015	0.169
1998	1986	0.13	4581	0.16	0.015	0.170
1999	1944	0.14	3608	0.17	0.017	0.171
2000	1927	0.13	1707	0.19	0.017	0.173
2001	1903	0.13	906	0.26	0.020	0.175
2002	1823	0.13	1163	0.22	0.022	0.177
2003	1744	0.13	1086	0.21	0.024	0.180
2004	1714	0.13	1758	0.19	0.024	0.184
2005	1687	0.13	2596	0.19	0.022	0.187
2006	1591	0.13	2407	0.20	0.025	0.190
2007	1439	0.14	2206	0.22	0.029	0.194
2008	1307	0.14	1749	0.26	0.028	0.198
2009	1187	0.14	2286	0.29	0.027	0.275
2010	1100	0.14	3257	0.37	0.025	0.277
2011	1060	0.14	2089	0.47	0.027	0.280

Table A27. Likelihood profile analysis for survey dredge efficiency, biomass, and biomass status (B2011/B1999) using the basecase SS3 model for surfclams in the southern area. Minimum likelihood values for each term are highlighted.

Label	Q=0.18	Q=0.26	Q=0.3	Q=0.33 (basecase)	Q=0.38	Q=0.44	Q=0.49
TOTAL	2036.0	2032.5	2031.7	**2031.5**	2032.0	2033.9	2036.1
Recruitment	3.479	3.035	**2.940**	2.948	3.124	3.791	4.728
Parm_priors	**0.057**	0.217	0.318	0.383	0.504	0.672	0.808
Parm_softbounds	0.003	0.003	0.003	0.003	0.003	0.003	**0.003**
Survey	-3.013	-3.385	-3.568	**-3.604**	-3.444	-2.738	-1.915
Lengths							
Fishery lengths	204.210	203.237	202.930	202.790	202.615	202.516	**202.515**
Survey lengths	151.100	149.685	149.213	148.976	148.614	148.219	**147.954**
Survey ages	1680.2	**1679.7**	1679.9	1680.1	1680.6	1681.4	1682.0

B2011	1,387,280	915,528	772,377	702,902	599,781	493,921	**428,446**
B2011/B1999	0.51	0.49	0.48	0.47	0.46	0.44	**0.42**

Table A28. Table comparing the biomass estimates from previous surfclam assessments. Note that in the current assessment animals greater than 120 mm are 6 and older in the southern area and 7 and older in the north, due to differing growth rates.

Year	2012	SAW 49 (NEFSC 2009)	SAW 44 (NEFSC 2007)	SAW 37 (NEFSC 2003)	SAW 30 (NEFSC 2000)	SAW 26 (NEFSC 1998)
Shell length (mm)	~120+ (age 6+ South, 7+ North)	120+	120+	120+ in NJ; 100+ elsewhere	120+ in NJ; 100+ elsewhere	All
Method	SS3	KLAMZ	KLAMZ	SWAB	KLAMZ	SWAB
Year	Biomass	Biomass	Biomass			
1981		831	1,020			
1982	1,331	862	1,036			
1983	1,657	889	1,059			
1984	1,987	916	1,083			
1985	2,191	935	1,141			
1986	2,451	954	1,225			
1987	2,497	973	1,271			
1988	2,500	988	1,290			
1989	2,477	1,003	1,289			
1990	2,398	1,021	1,285		1,200	
1991	2,330	1,029	1,283		1,200	
1992	2,278	1,045	1,290		1,200	
1993	2,216	1,059	1,476		1,200	
1994	2,156	1,070	1,613		1,200	
1995	2,140	1,082	1,709		1,200	
1996	2,055	1,088	1,780	1,146	1,200	1,113
1997	1,990	1,090	1,842		1,300	
1998	1,986	1,092	1,824	1,460	1,300	
1999	1,944	1,086	1,799			
2000	1,927	1,074	1,723			
2001	1,903	1,059	1,628	803		
2002	1,823	1,037	1,531			
2003	1,744	1,012	1,415			
2004	1,714	984	1,292			
2005	1,687	955				
2006	1,591	931				
2007	1,439	905				
2008	1,307					
2009	1,187					
2010	1,100					
2011	1,060					

Table A29. Whole stock biomass status estimates for 2011 with cv and approximate 95% confidence intervals.

	Biomass	cv	lci	uci
2011	1060	0.143	802	1401
Target	972	0.135	747	1235
Threshold	486	0.135	373	633

Table A30. Whole stock F status estimates for 2011 with cv and approximate 95% confidence intervals.

	F	cv	lci	uci
2011	0.027	0.271	0.016	0.045
Threshold	0.15			

Table A31 Southern area biomass status estimates for 2011 with cv and approximate 95% confidence intervals.

	Biomass	cv	lci	uci
2011	703	0.196	481	1028
Target	744	0.168	537	1032
Threshold	372	0.168	268	516

Table A32. Southern area F status estimates for 2011 with cv and approximate 95% confidence intervals.

	F	cv	lci	uci
2011	0.040	0.211	0.025	0.056
Threshold	0.15			

Table A33. Projected biomass and biomass status ($B/B_{threshold}$ where $B_{threshold}=B_{1999}/4$) during 2012-2021 for surflclams in the southern, GBK and combined areas.

Year	Southern area			GBK area			Southern + GBK		
	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota
Biomass (mt)									
2011	704,366	704,366	704,366	370,217	370,217	370,217	1,074,583	1,074,583	1,074,583
2012	699,480	699,480	699,480	338,866	338,866	338,866	1,038,346	1,038,346	1,038,346
2013	690,839	690,839	690,839	308,580	308,580	308,580	999,419	999,419	999,419
2014	633,310	677,921	672,888	252,941	271,536	271,536	886,251	949,457	944,424
2015	604,667	686,541	676,966	208,410	238,833	238,833	813,077	925,374	915,799
2016	617,034	731,098	717,356	175,171	212,330	212,330	792,205	943,428	929,686
2017	585,090	725,516	708,212	154,269	194,626	194,626	739,359	920,142	902,838
2018	597,117	761,170	740,671	160,621	202,314	202,314	757,738	963,484	942,985
2019	614,769	800,317	777,001	172,120	214,381	214,381	786,889	1,014,698	991,382
2020	632,270	837,938	812,136	185,038	227,946	227,946	817,308	1,065,884	1,040,082
2021	648,414	873,215	845,220	197,790	241,864	241,864	846,204	1,115,079	1,087,084
Biomass / Bthreshold (Bthreshold=B1999/4)									
1999	1,513,100			506,882			2,019,982		
Bthreshold	378,275			126,721			504,996		
2011	1.86	1.86	1.86	2.92	2.92	2.92	2.13	2.13	2.13
2012	1.85	1.85	1.85	2.67	2.67	2.67	2.06	2.06	2.06
2013	1.83	1.83	1.83	2.44	2.44	2.44	1.98	1.98	1.98
2014	1.67	1.79	1.78	2.00	2.14	2.14	1.75	1.88	1.87
2015	1.60	1.81	1.79	1.64	1.88	1.88	1.61	1.83	1.81
2016	1.63	1.93	1.90	1.38	1.68	1.68	1.57	1.87	1.84
2017	1.55	1.92	1.87	1.22	1.54	1.54	1.46	1.82	1.79
2018	1.58	2.01	1.96	1.27	1.60	1.60	1.50	1.91	1.87
2019	1.63	2.12	2.05	1.36	1.69	1.69	1.56	2.01	1.96
2020	1.67	2.22	2.15	1.46	1.80	1.80	1.62	2.11	2.06
2021	1.71	2.31	2.23	1.56	1.91	1.91	1.68	2.21	2.15

Table A34. Projected landings (mt and bu) during 2012-2021 for surflclams in the southern, GBK and combined areas.

Year	Southern area			GBK area			Southern + GBK		
	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota
	Landings (mt, catch - 12% incidental mortality)								
2011	16,089	16,089	16,089	2,127	2,127	2,127	18,216	18,216	18,216
2012	18,728	18,728	18,728	2,127	2,127	2,127	20,854	20,854	20,854
2013	60,767	13,145	18,504	28,352	7,710	7,710	89,119	20,854	26,213
2014	57,705	13,145	18,504	23,444	7,710	7,710	81,150	20,854	26,213
2015	55,609	13,145	18,504	19,570	7,710	7,710	75,178	20,854	26,213
2016	54,683	13,145	18,504	16,829	7,710	7,710	71,512	20,854	26,213
2017	54,690	13,145	18,504	15,235	7,710	7,710	69,925	20,854	26,213
2018	55,444	13,145	18,504	14,658	7,710	7,710	70,102	20,854	26,213
2019	56,660	13,145	18,504	14,827	7,710	7,710	71,488	20,854	26,213
2020	58,057	13,145	18,504	15,448	7,710	7,710	73,505	20,854	26,213
2021	59,431	13,145	18,504	16,279	7,710	7,710	75,710	20,854	26,213
Landings (bu, catch - 12% incidental mortality)									
2011	2,086,796	2,086,796	2,086,796	275,848	275,848	275,848	2,362,644	2,362,644	2,362,644
2012	2,429,011	2,429,011	2,429,011	275,848	275,848	275,848	2,704,859	2,704,859	2,704,859
2013	7,881,636	1,704,882	2,399,944	3,677,240	999,977	999,977	11,558,875	2,704,859	3,399,921
2014	7,484,494	1,704,882	2,399,944	3,040,787	999,977	999,977	10,525,280	2,704,859	3,399,921
2015	7,212,525	1,704,882	2,399,944	2,538,250	999,977	999,977	9,750,776	2,704,859	3,399,921
2016	7,092,540	1,704,882	2,399,944	2,182,694	999,977	999,977	9,275,234	2,704,859	3,399,921
2017	7,093,374	1,704,882	2,399,944	1,976,028	999,977	999,977	9,069,402	2,704,859	3,399,921
2018	7,191,136	1,704,882	2,399,944	1,901,184	999,977	999,977	9,092,320	2,704,859	3,399,921
2019	7,348,932	1,704,882	2,399,944	1,923,129	999,977	999,977	9,272,061	2,704,859	3,399,921
2020	7,530,109	1,704,882	2,399,944	2,003,590	999,977	999,977	9,533,699	2,704,859	3,399,921
2021	7,708,252	1,704,882	2,399,944	2,111,404	999,977	999,977	9,819,657	2,704,859	3,399,921

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Table A35. Projected fully recruited fishing mortality and exploitation rates (catch weight / biomass ages 6+) during 2012-2021 for surfclams in the southern, GBK and combined areas.

Year	Southern area			GBK area			Southern + GBK		
	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota	F=0.15 (M)	Status-quo catch	Quota
	Fully recruited fishing mortality								
2011	0.037	0.037	0.037	0.009	0.009	0.009	0.028	0.028	0.028
2012	0.044	0.044	0.044	0.010	0.010	0.010	0.033	0.033	0.033
2013	0.150	0.031	0.044	0.150	0.039	0.039	0.150	0.034	0.042
2014	0.150	0.031	0.044	0.150	0.044	0.044	0.150	0.035	0.043
2015	0.150	0.031	0.044	0.150	0.050	0.050	0.150	0.035	0.044
2016	0.150	0.030	0.043	0.150	0.055	0.055	0.150	0.035	0.044
2017	0.151	0.029	0.042	0.150	0.059	0.059	0.150	0.035	0.044
2018	0.151	0.028	0.040	0.151	0.061	0.061	0.150	0.035	0.043
2019	0.151	0.026	0.038	0.151	0.060	0.060	0.150	0.034	0.042
2020	0.151	0.025	0.037	0.151	0.058	0.058	0.150	0.033	0.040
2021	0.151	0.024	0.035	0.151	0.056	0.056	0.150	0.032	0.039
Exploitation rate (catch/biomass)									
2011	0.026	0.026	0.026	0.006	0.006	0.006	0.019	0.019	0.019
2012	0.030	0.030	0.030	0.007	0.007	0.007	0.022	0.022	0.022
2013	0.099	0.021	0.030	0.103	0.028	0.028	0.100	0.023	0.029
2014	0.102	0.022	0.031	0.104	0.032	0.032	0.103	0.025	0.031
2015	0.103	0.021	0.031	0.105	0.036	0.036	0.104	0.025	0.032
2016	0.099	0.020	0.029	0.108	0.041	0.041	0.101	0.025	0.032
2017	0.105	0.020	0.029	0.111	0.044	0.044	0.106	0.025	0.033
2018	0.104	0.019	0.028	0.102	0.043	0.043	0.104	0.024	0.031
2019	0.103	0.018	0.027	0.096	0.040	0.040	0.102	0.023	0.030
2020	0.103	0.018	0.026	0.094	0.038	0.038	0.101	0.022	0.028
2021	0.103	0.017	0.025	0.092	0.036	0.036	0.100	0.021	0.027

Table A36. Cumulative probability of being in overfished status in any of the years 2013 – 2017, under a variety of catch scenarios.

Catch scenario	P[overfished][1]	P[overfishing][1]
Whole stock		
Status Quo	0.019	0.000
Quota	0.022	0.000
OFL (F = M) catch	0.123	0.990
Southern Area		
Status Quo	0.053	0.000
Quota	0.061	0.000
OFL (F = M) catch	0.162	0.990
Northern Area		
Status Quo	NA	0.000
Quota	NA	0.000
OFL (F = M) catch	NA	0.990

1 Probabilities are cumulative (2013 - 2017)

Table A37. Estimated catch at the OFL for the next five years by area.

Year	Mean	Median	CV
Whole stock			
2014	92324	90886	0.179
2015	85693	84191	0.189
2016	81658	80102	0.198
2017	79908	78326	0.202
2018	80124	78516	0.203
Southern area			
2014	66202	34622	0.223
2015	63969	62304	0.233
2016	62950	61221	0.239
2017	63027	61249	0.242
2018	63908	62117	0.243
Northern area			
2014	27302	26252	0.286
2015	22879	21915	0.3
2016	19721	18860	0.306

2017	17849	17056	0.308
2018	17180	16412	0.309

Figure A1. Surfclam stock assessment regions and NEFSC shellfish survey strata. The shaded strata are where surfclams are found.

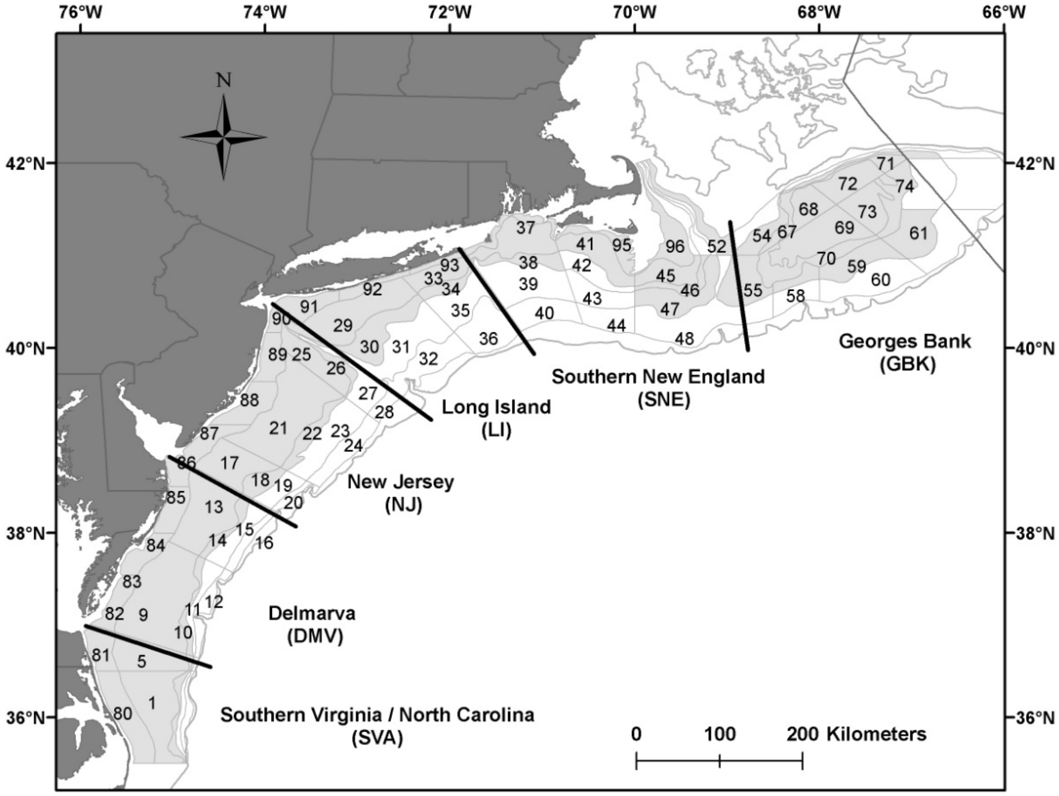


Figure A2. The surfclam regions divided into two areas.

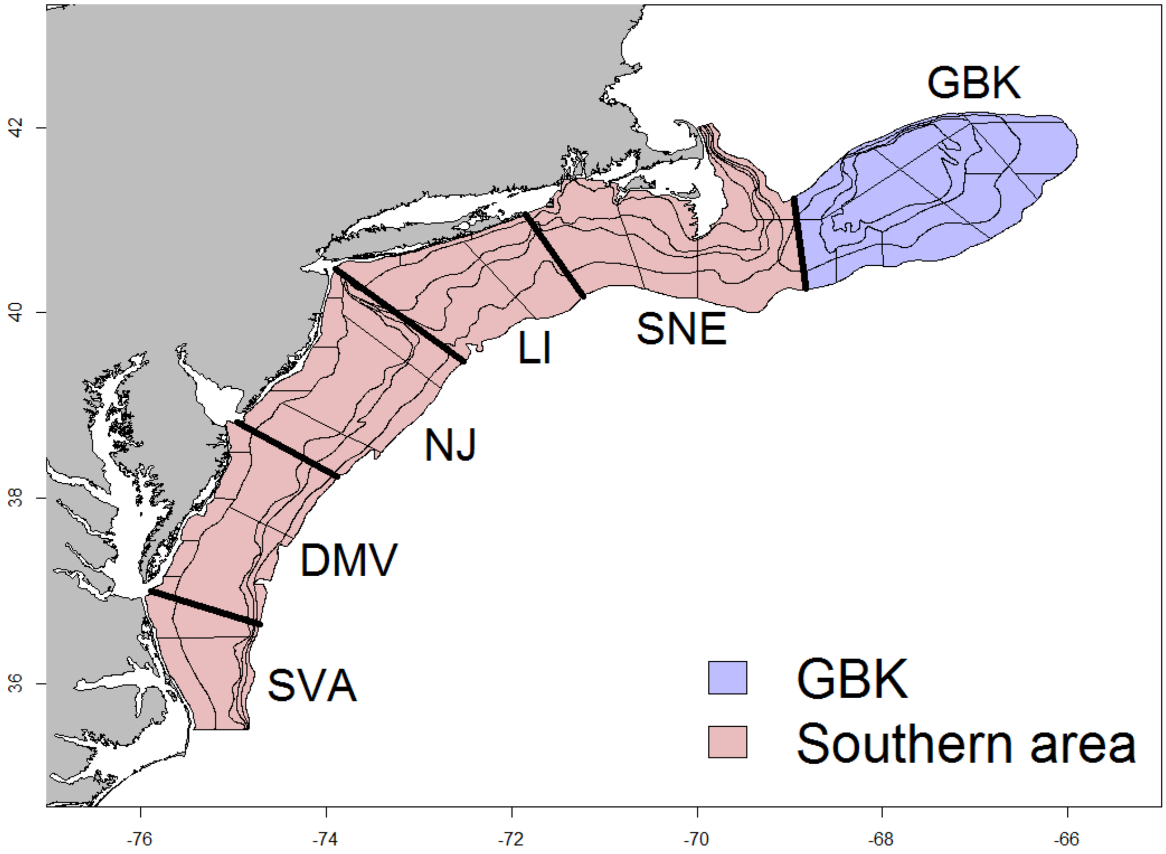


Figure A3. Surfclam landings (total and EEZ) during 1965-2011.

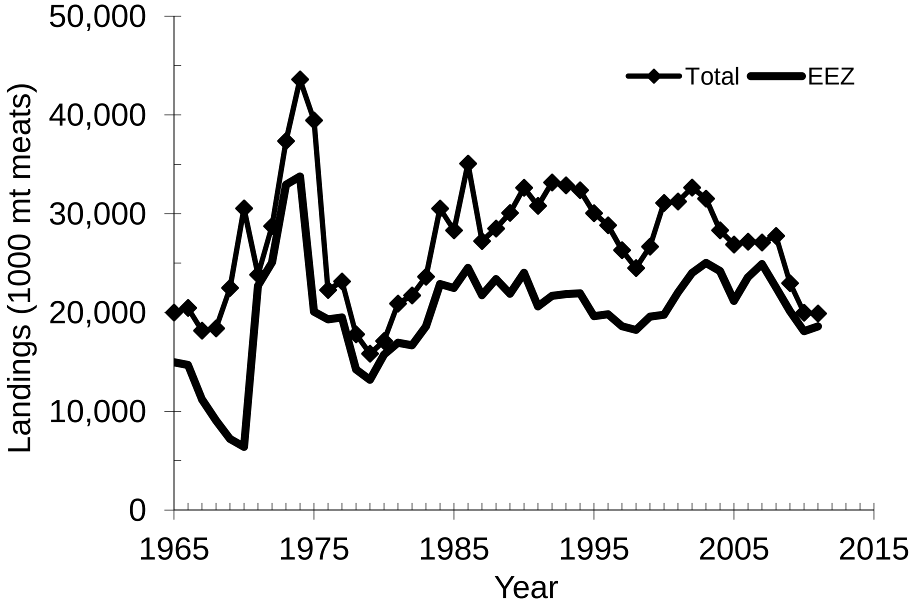


Figure A4. Surfclam landings from the US EEZ during 1979-2011, by stock assessment region.

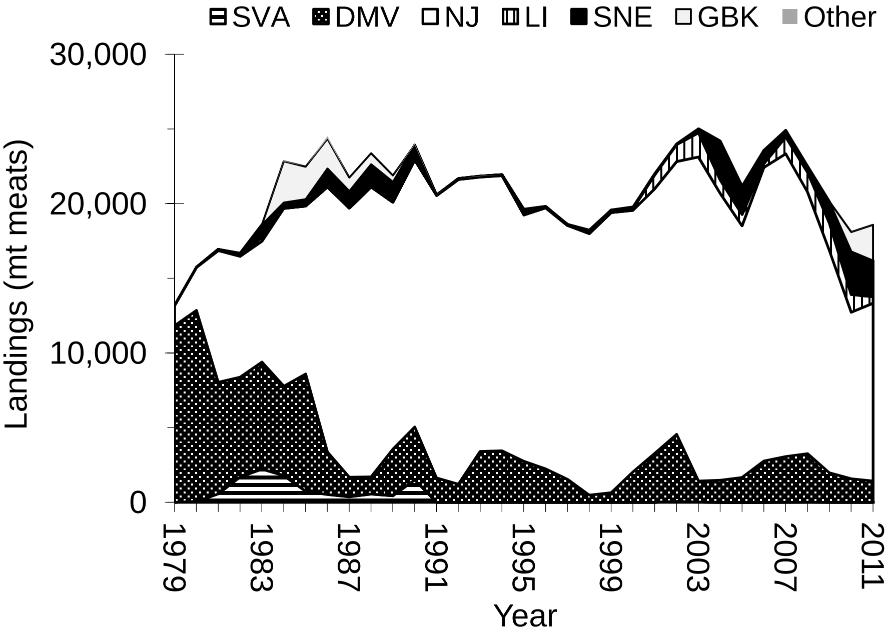


Figure A5. Surfclam hours fished from the US EEZ during 1991-2011, by stock assessment region.

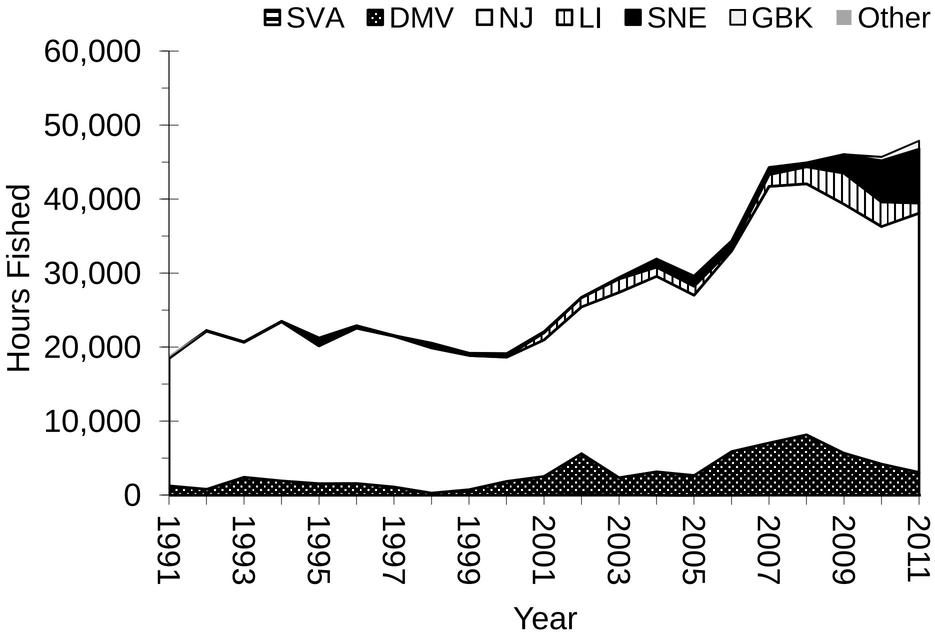


Figure A6. Nominal and 2010 dollar equivalent prices for surfclam 1981-2011.

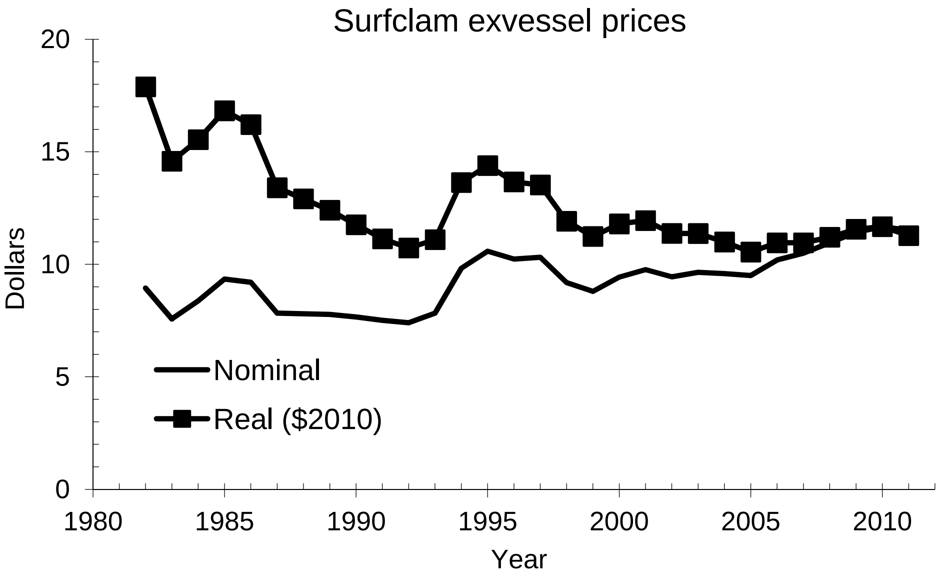
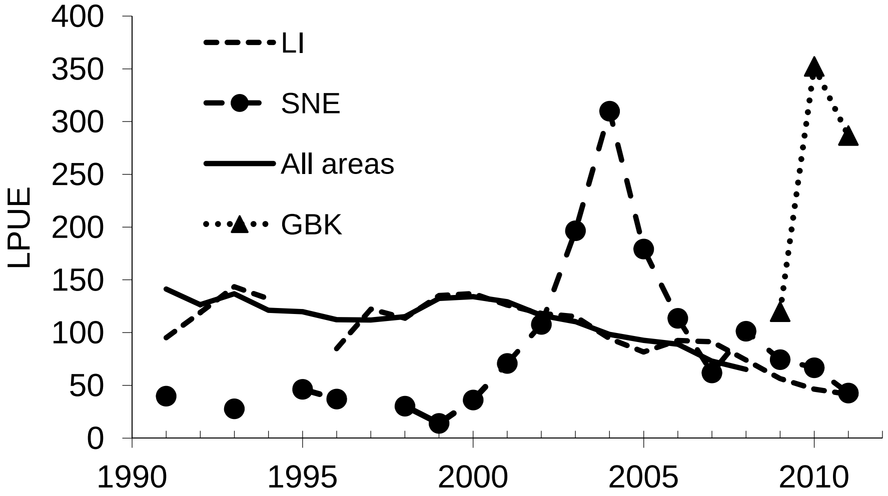


Figure A7. Nominal landings per unit effort (LPUE in bushels landed per hour fished) for surfclam, by region. LPUE is total landings in bushels divided by total fishing effort.



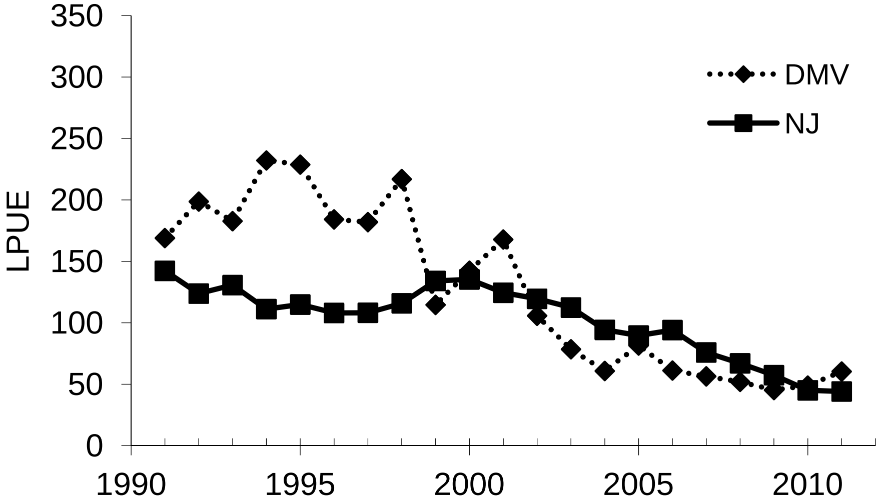


Figure A8. Average surfclams landings by ten-minute squares over time.

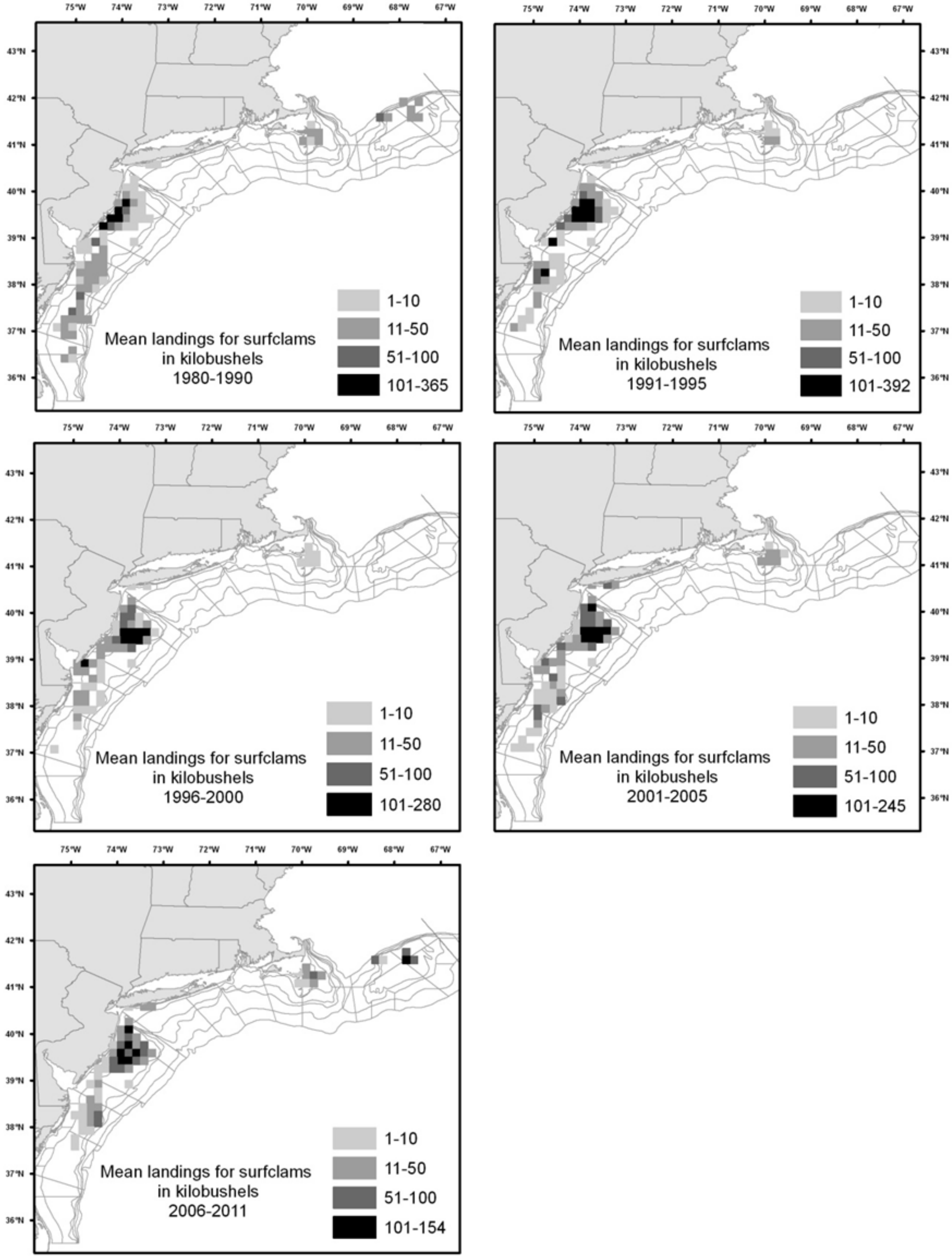


Figure A9. Average surfclam effort by ten-minute squares

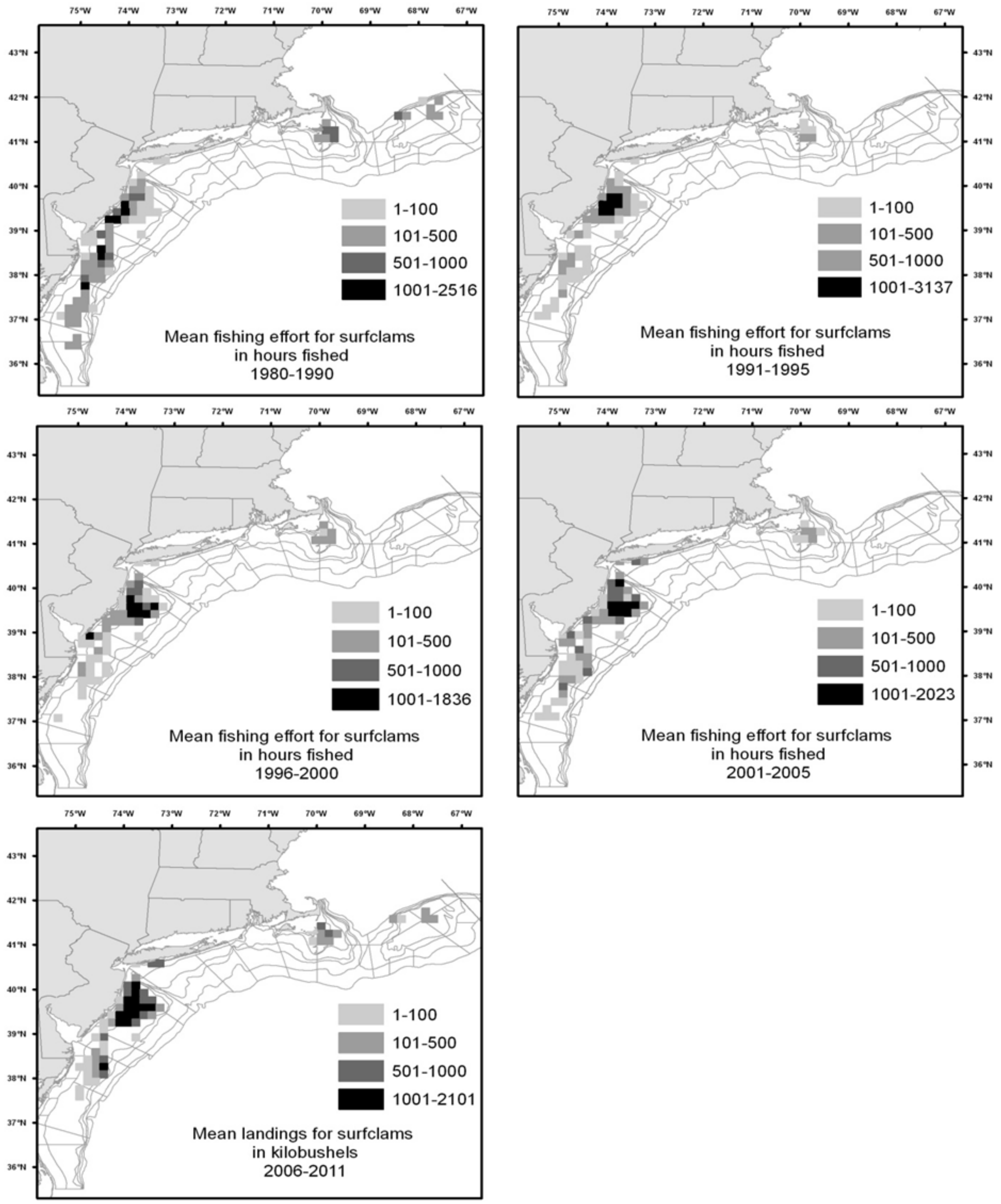


Figure A10. Average surfclam LPUE (bu. h^{-1}) by ten-minute squares over time.

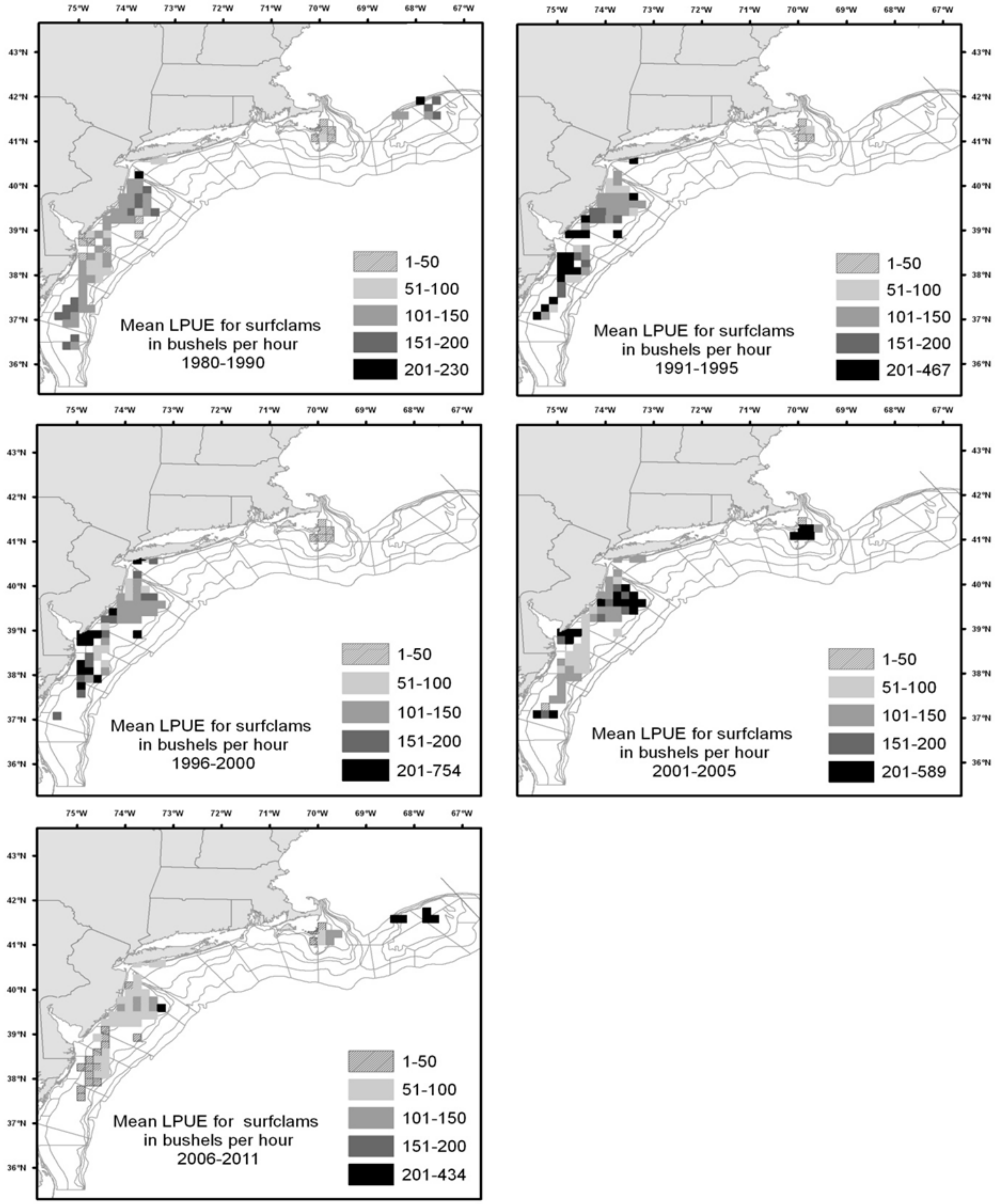


Figure A11. Annual surfclam landings in “important” ten minute squares (TNMS) during 1980-2012 based on logbook data. Important means that a square ranked in the top 10 TNMS for total landings during any five-year period (1980-1984, 1985-1989, …, 2000-2004, 2005-2009, 2010-2012). Data for 2012 are incomplete and preliminary. To protect the privacy of individual firms, data are not plotted if the number of vessels is less than 2. Instead, a “^” is shown on the x-axis to indicate where data are missing. The solid dark line is a spline intended to show trends. The spline was fit too all available data, including data not plotted.

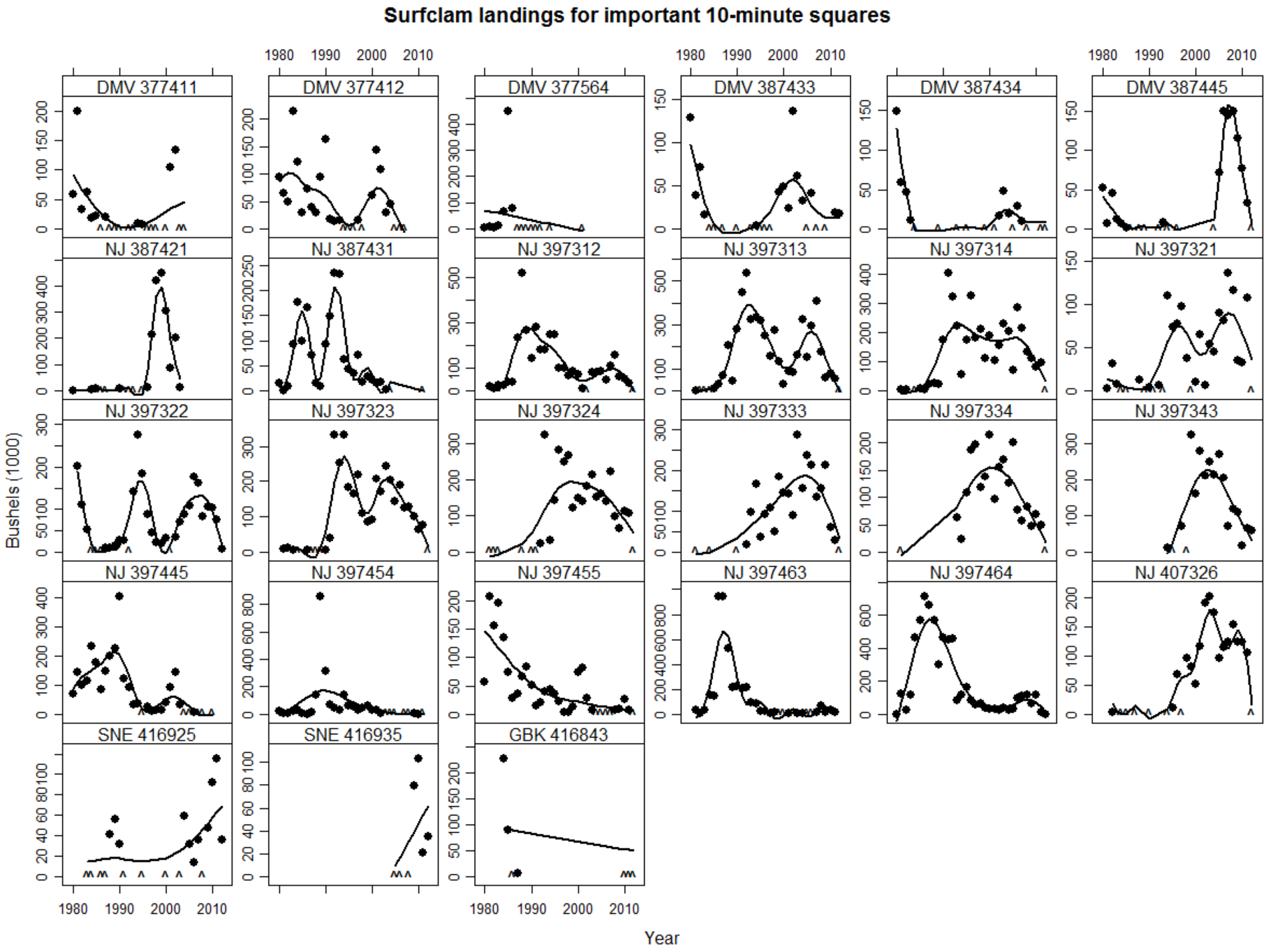


Figure A12. Annual surfclam effort (hours y^{-1}) in “important” ten minute squares (TNMS) during 1980-2012 based on logbook data. Important means that a square ranked in the top 10 TNMS for effort during any five-year period (1980-1984, 1985-1989, …, 2000-2004, 2005-2009, 2010-2012). Data for 2012 are incomplete and preliminary. To protect the privacy of individual firms, data are not plotted if the number of vessels is less than 2. Instead, a “^” is shown on the x-axis to indicate where data are missing. The solid dark line is a spline intended to show trends. The spline was fit too all available data, including data not plotted.

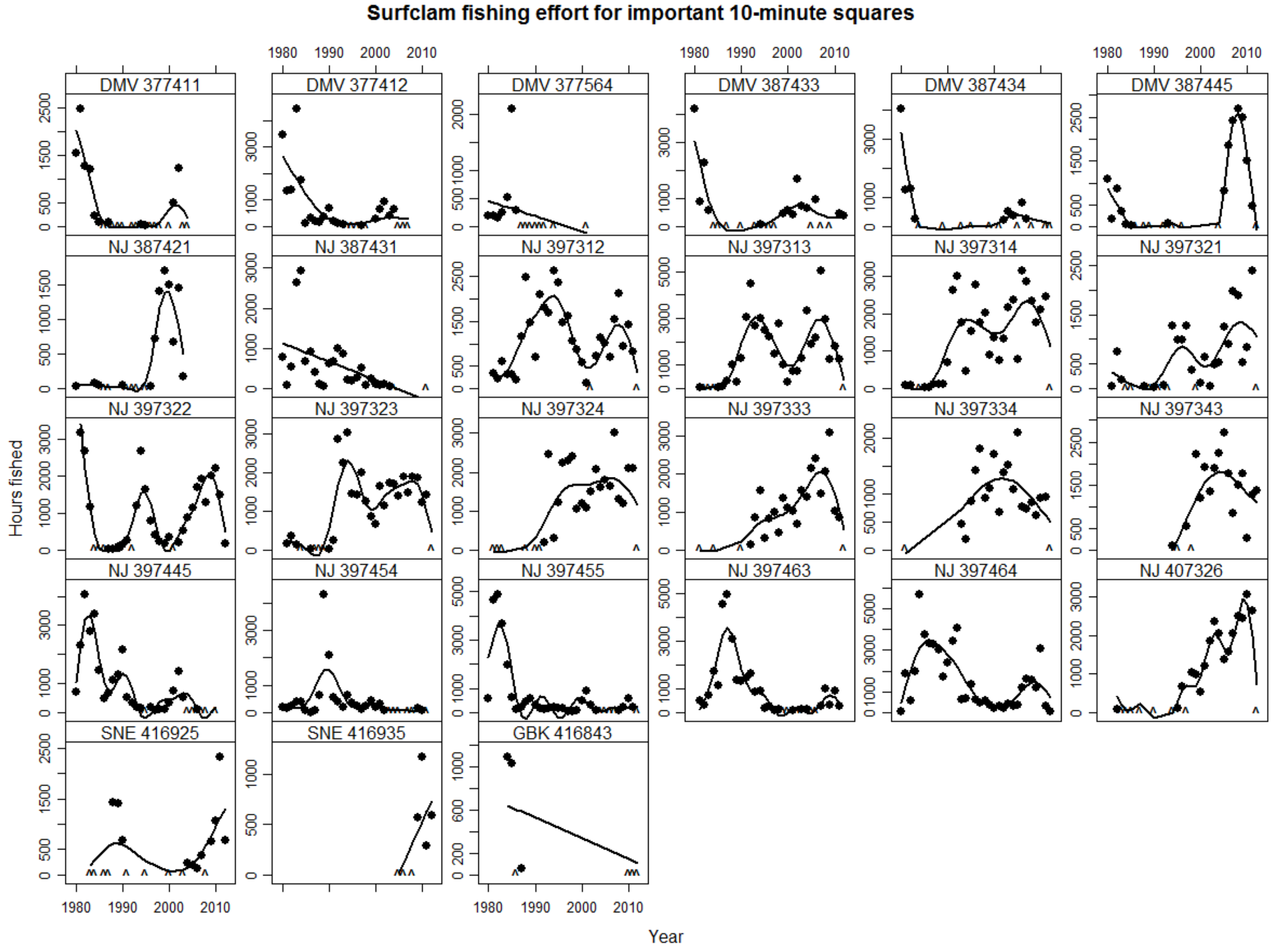


Figure A13. Annual surfclam LPUE (bu h^{-1}) in “important” ten minute squares (TNMS) during 1980-2012 based on logbook data. Important means that a square ranked in the top 10 TNMS for total LPUE during any five-year period (1980-1984, 1985-1989, …, 2000-2004, 2005-2009, 2010-2012). Data for 2012 are incomplete and preliminary. To protect the privacy of individual firms, data are not plotted if the number of vessels is less than 2. Instead, a “^” is shown on the x-axis to indicate where data are missing. The solid dark line is a spline intended to show trends. The spline was fit too all available data, including data not plotted.

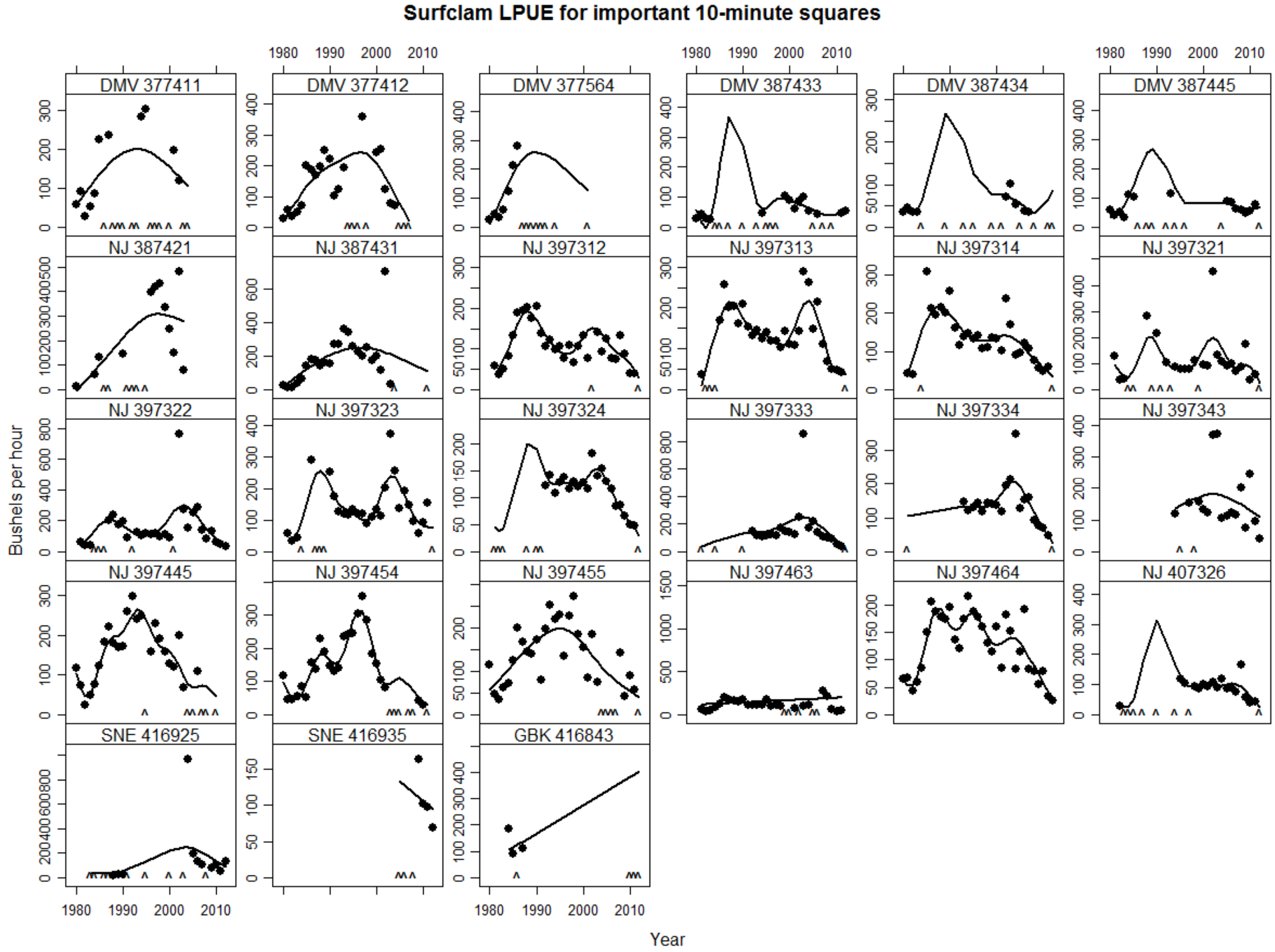


Figure A14. Length compositions of port-sampled landed surfclams from the DMV region.

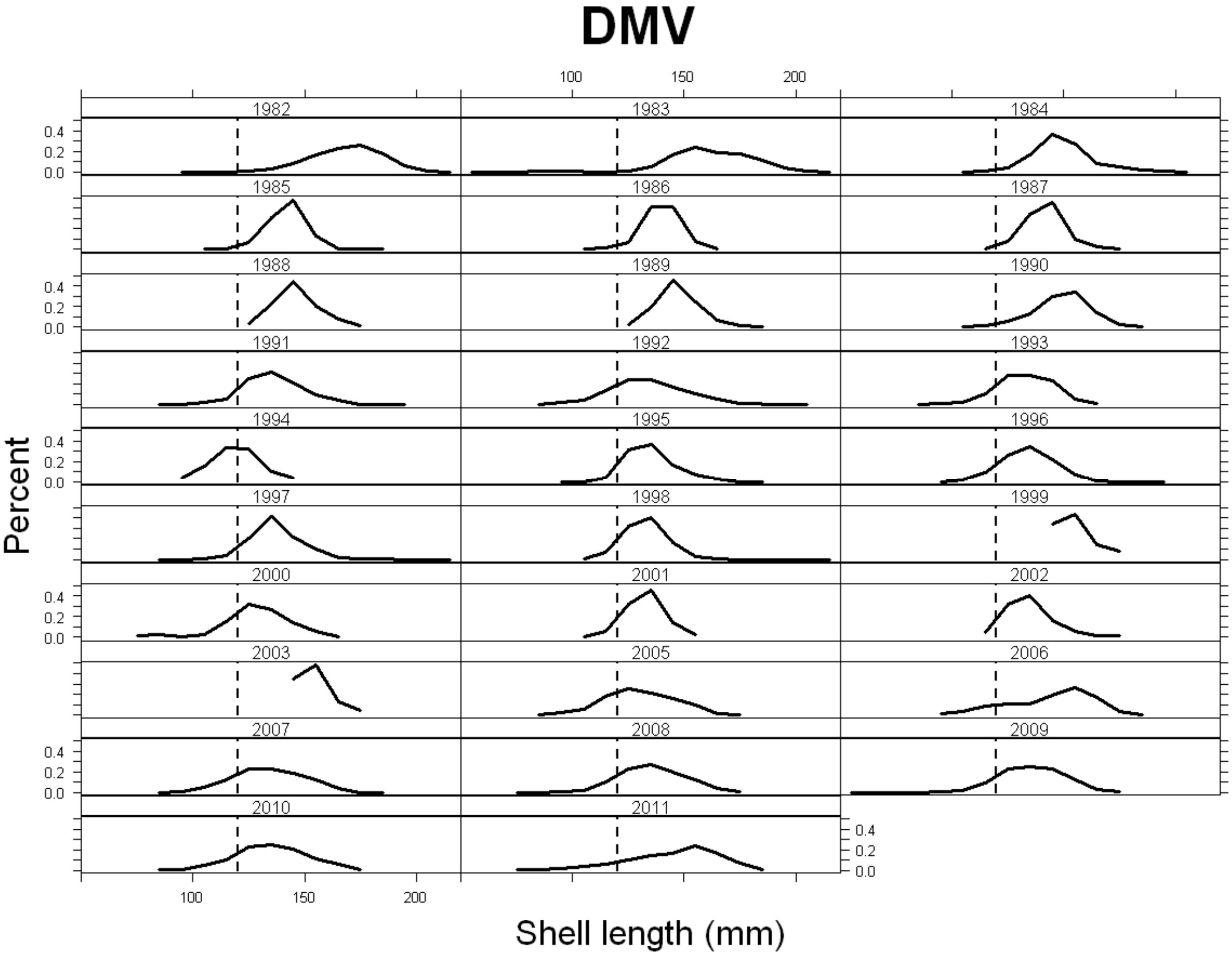


Figure A15. Length compositions of port-sampled landed surfclams from the NJ region.

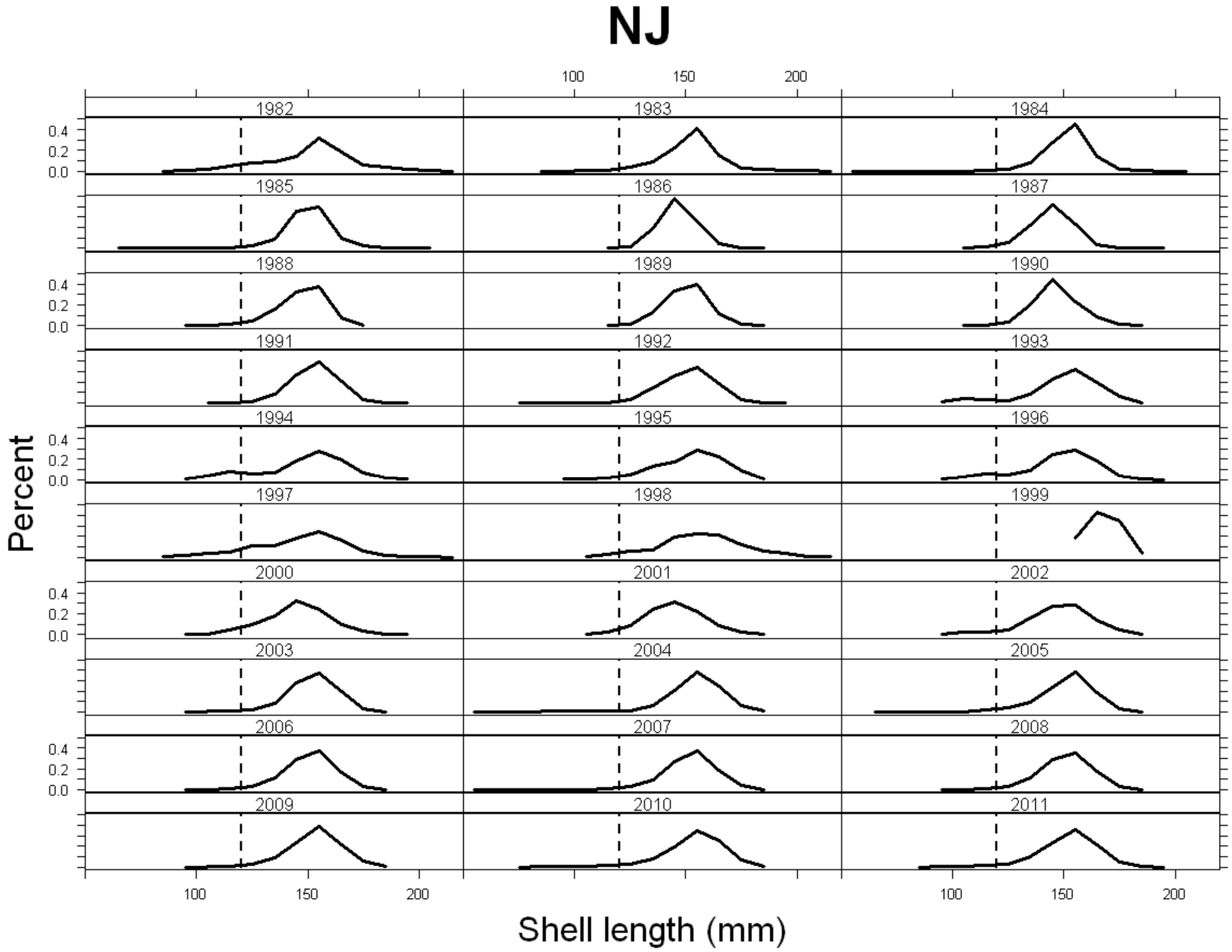


Figure A16. Length compositions of port-sampled landed surfclams from the LI region.

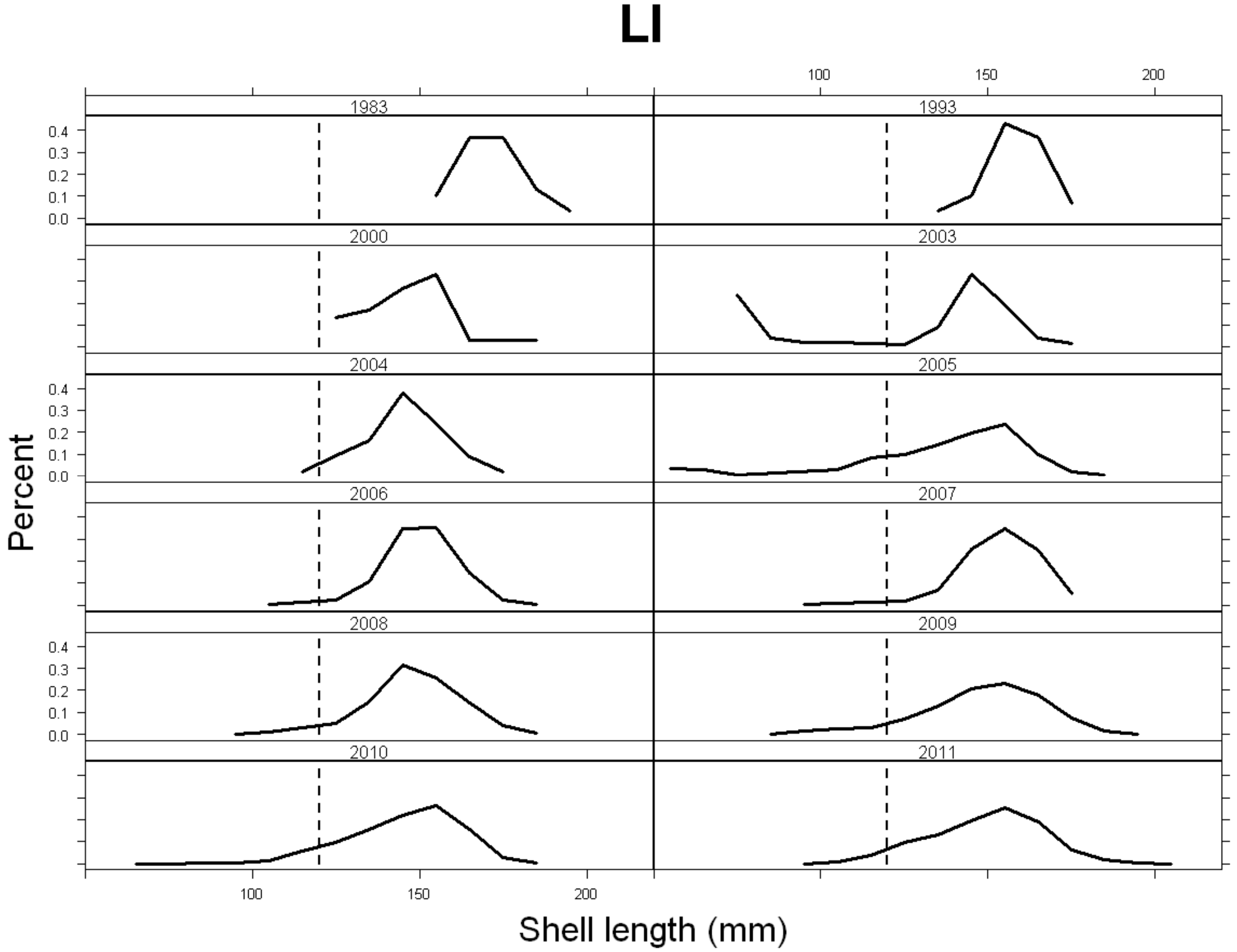


Figure A17. Length compositions of port-sampled landed surfclams from the SNE region.

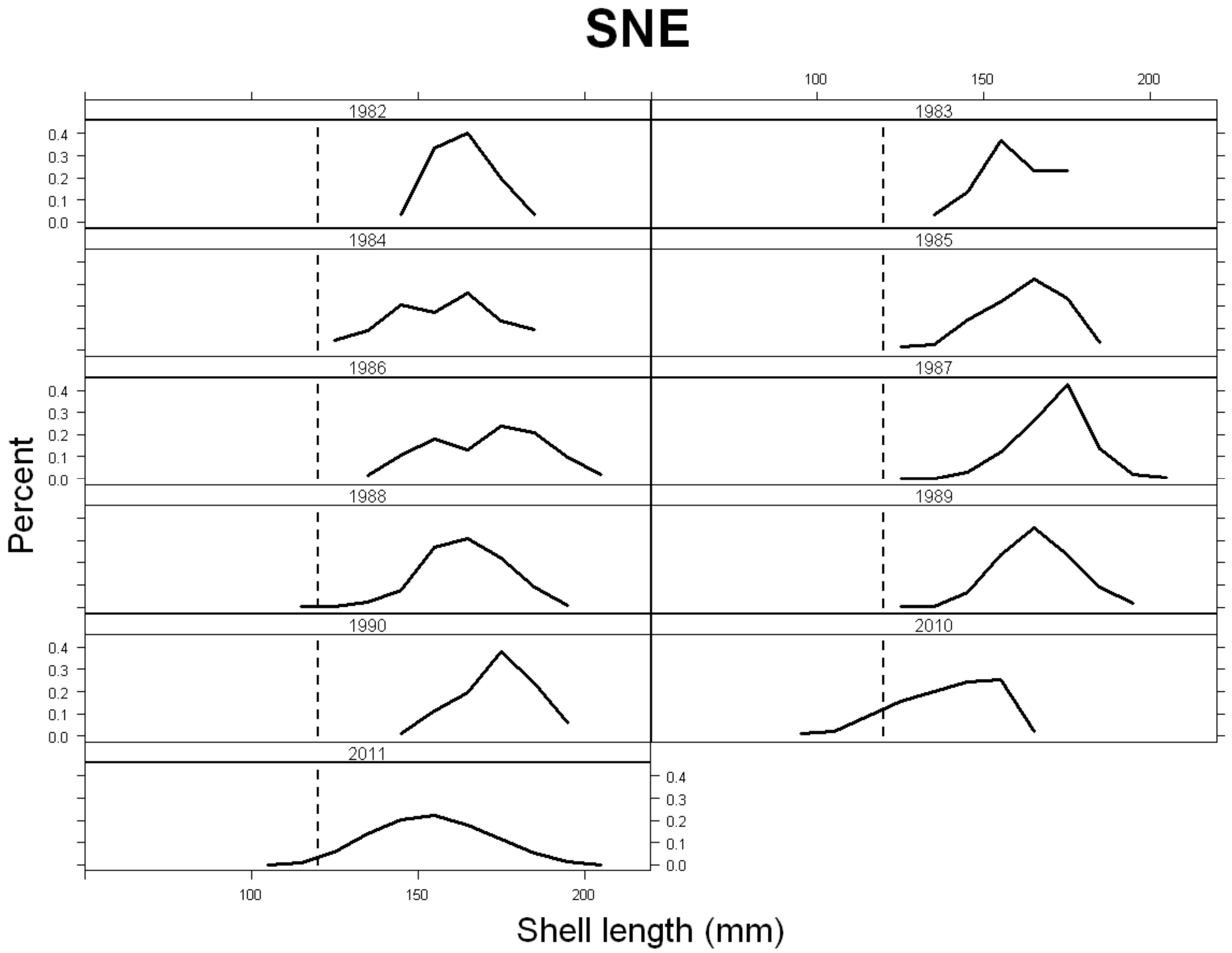


Figure A18. Length compositions of port-sampled landed surfclams from the GBK region.

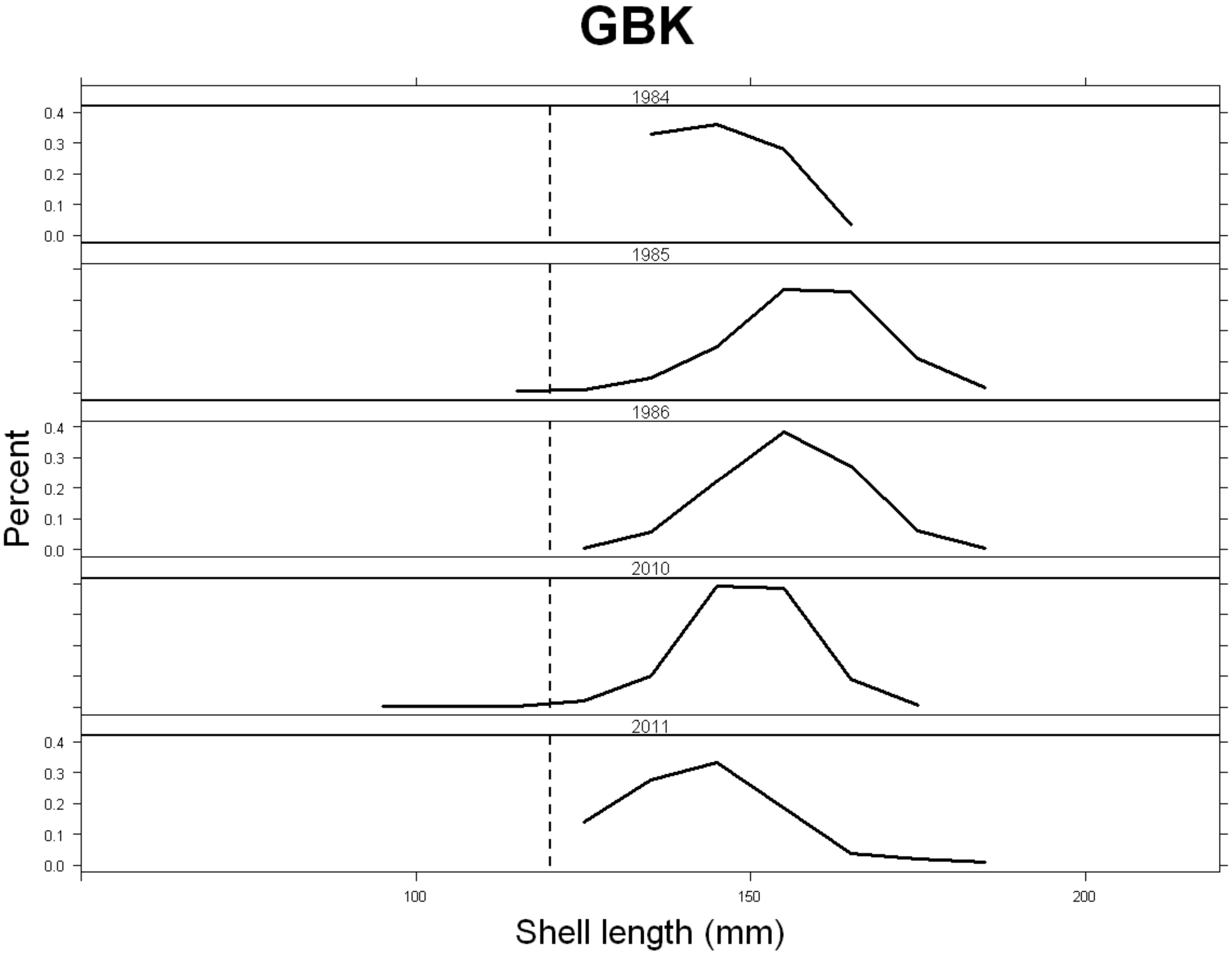


Figure A19. Station locations from the 2011 NEFSC survey.

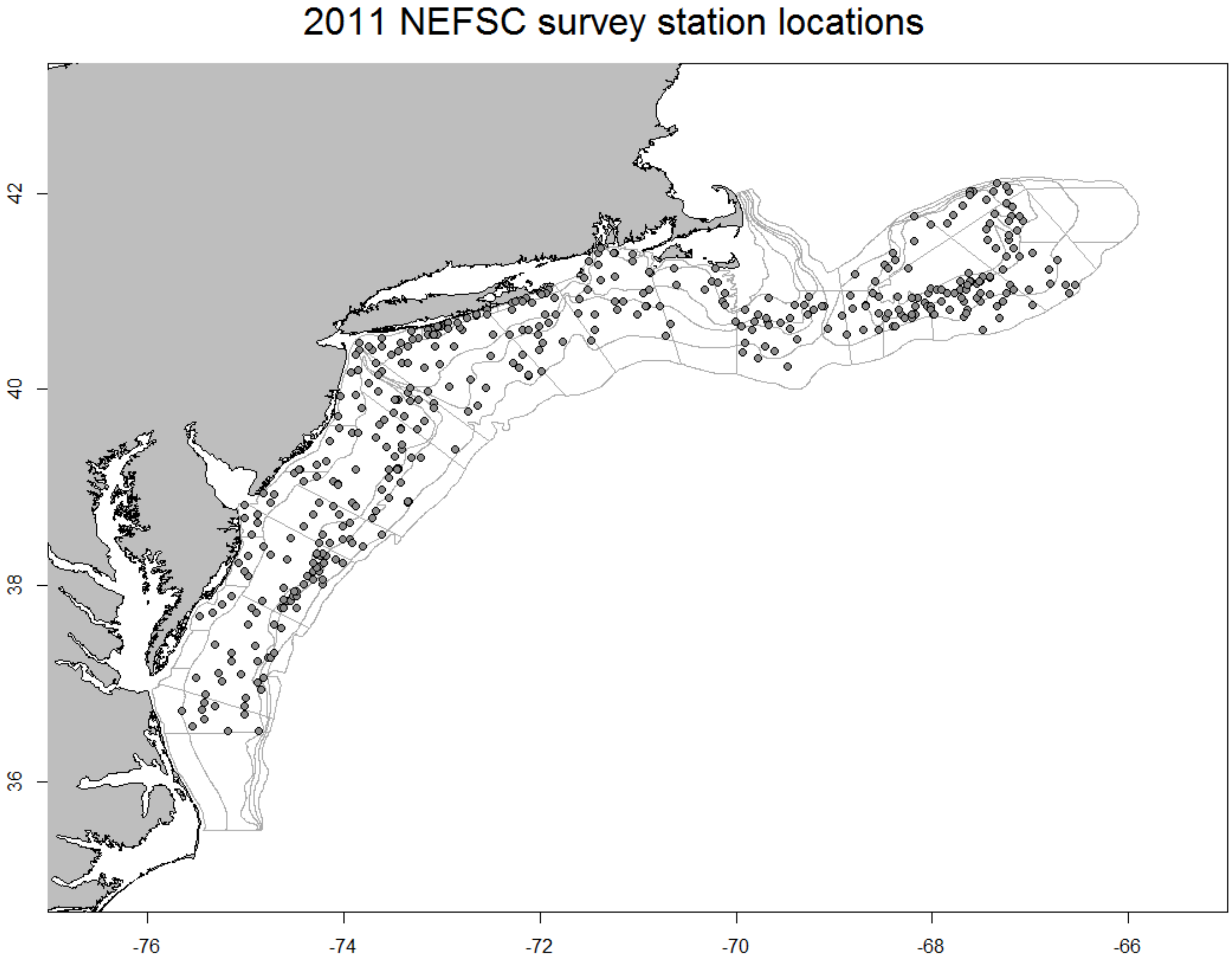


Figure A20. Amperage by tow for the 2011 NEFSC clam survey. The dashed line is for reference only.

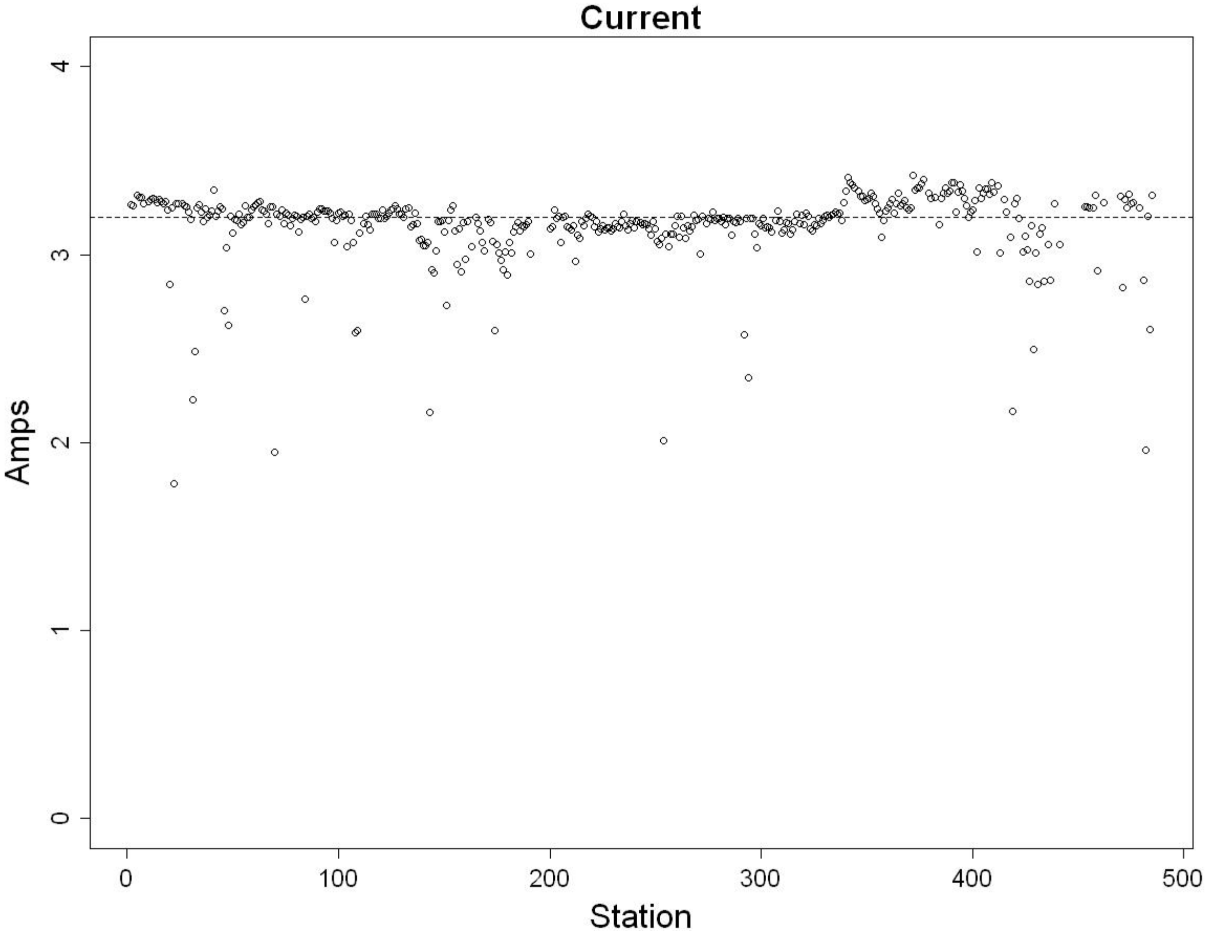


Figure A21. The relationship between amperage and differential pressure over all fishing seconds while the SSP was operational. The blue dots are observations recorded before the SSP failed at station 161 and the green dots are observations after the SSP began working again at station 371. The line plotted is the cubic spline fit to the data.

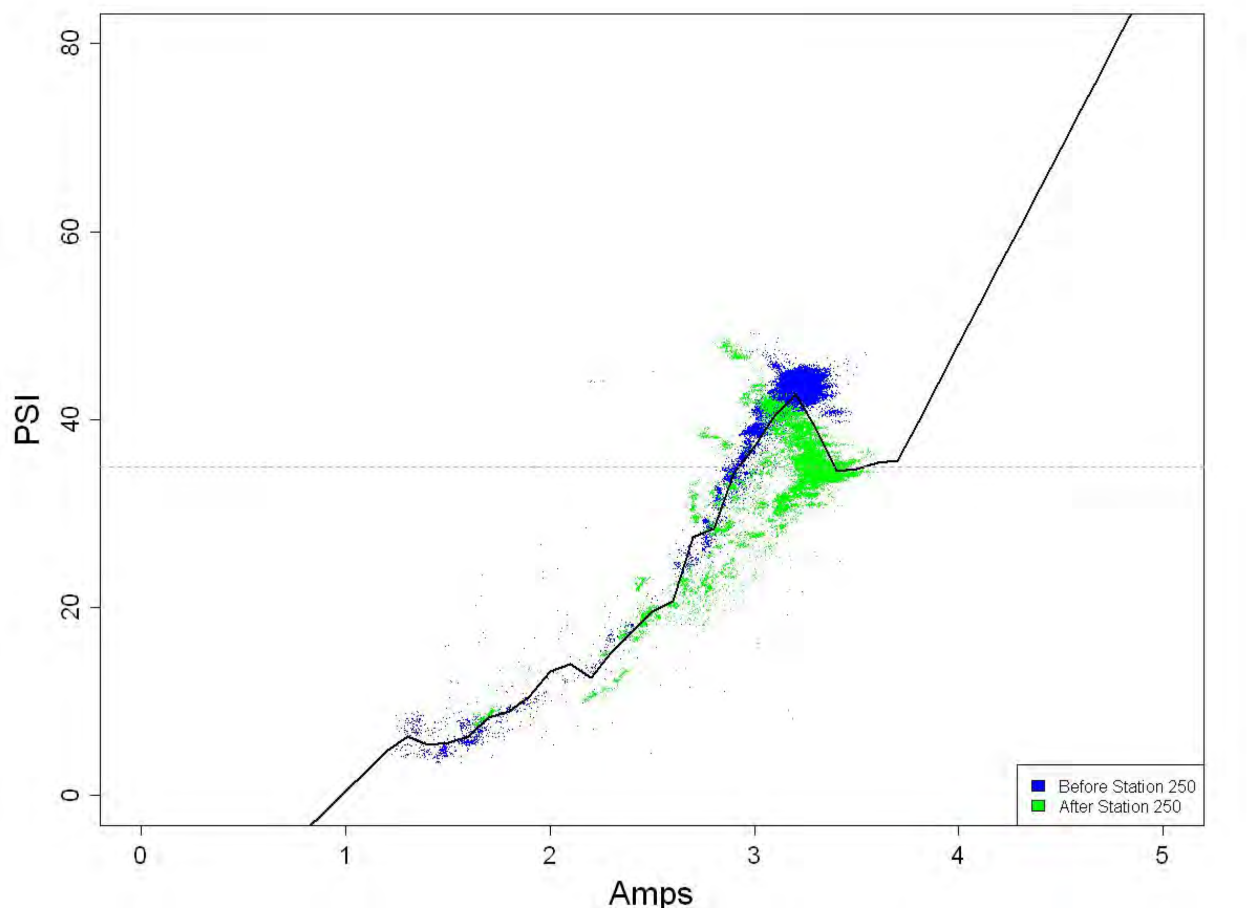


Figure A22. Differential pressure by tow during the 2011 NEFSC survey. The black circles are tows for which differential pressure was recorded by the SSP and the red circles are tows for which there is no SSP data. The dashed lines represent the upper and lower bounds for differential pressure tolerance found for the 2009 survey.

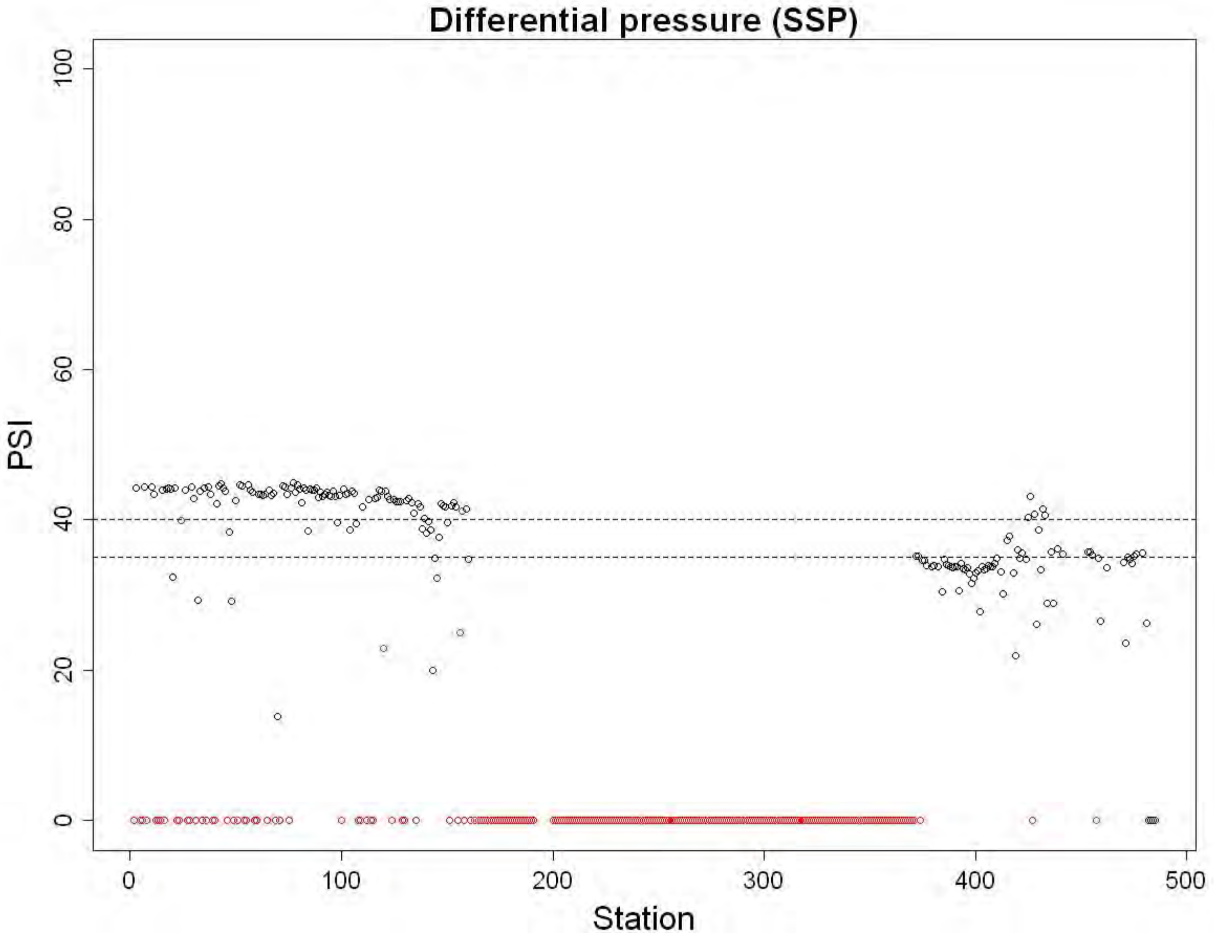


Figure A23. Model fits from four competing models to predict differential pressure from current supplied to the dredge pump on the 2011 NEFSC survey. The tolerance for adequate pump pressure (35 PSI) is shown with the dashed gray line.

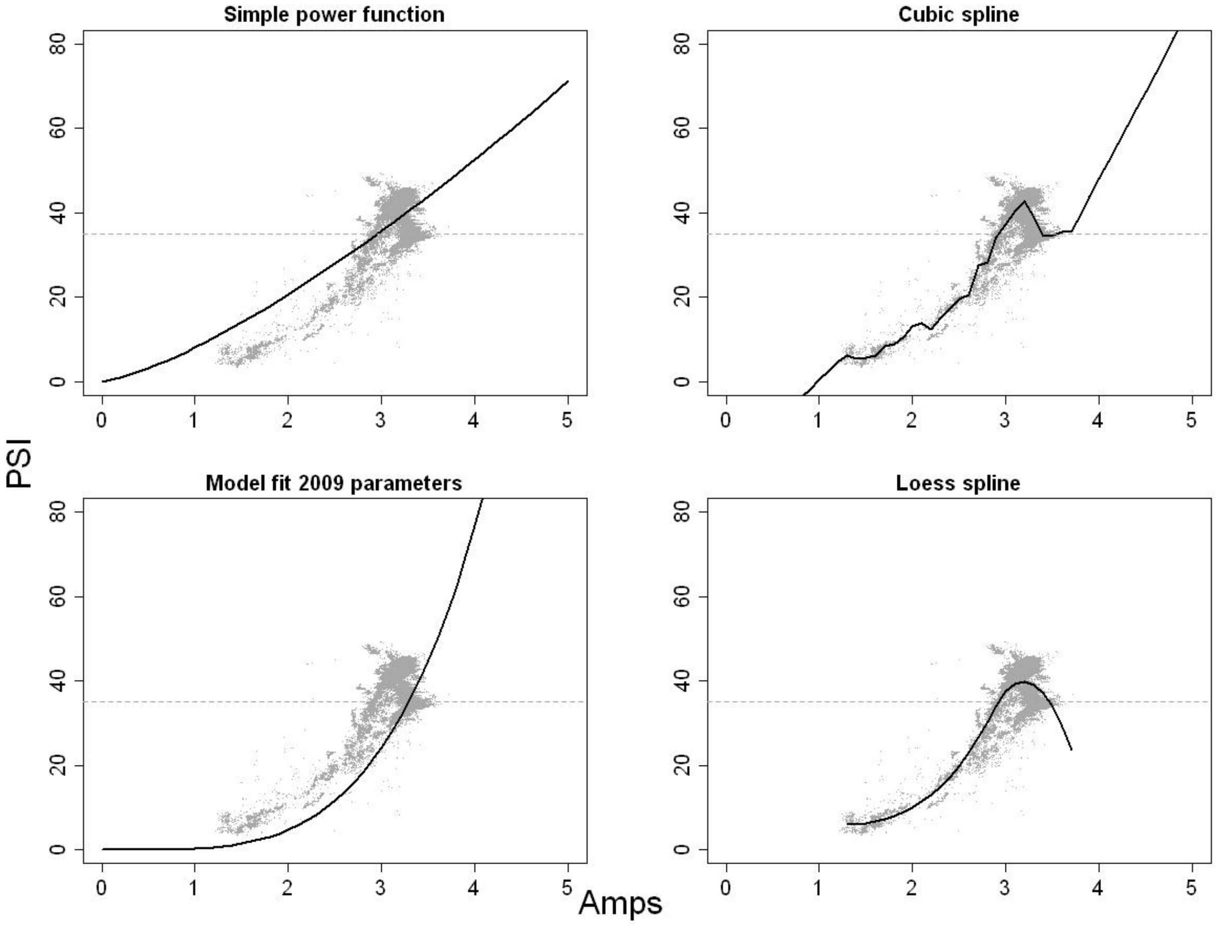


Figure A24. A comparison of four different models used to predict differential pressure from current. The shaded areas represent quadrants where the predicted and observed values disagree regarding the acceptability of a differential pressure measurement. The unshaded quadrants are areas where the predicted and observed values are in agreement. The numbers inside the plot area represent the fraction of points that fall within quadrant. Differential pressures less than 35 PSI are below tolerance for a successful fishing second. The predicted = observed line is also shown for reference.

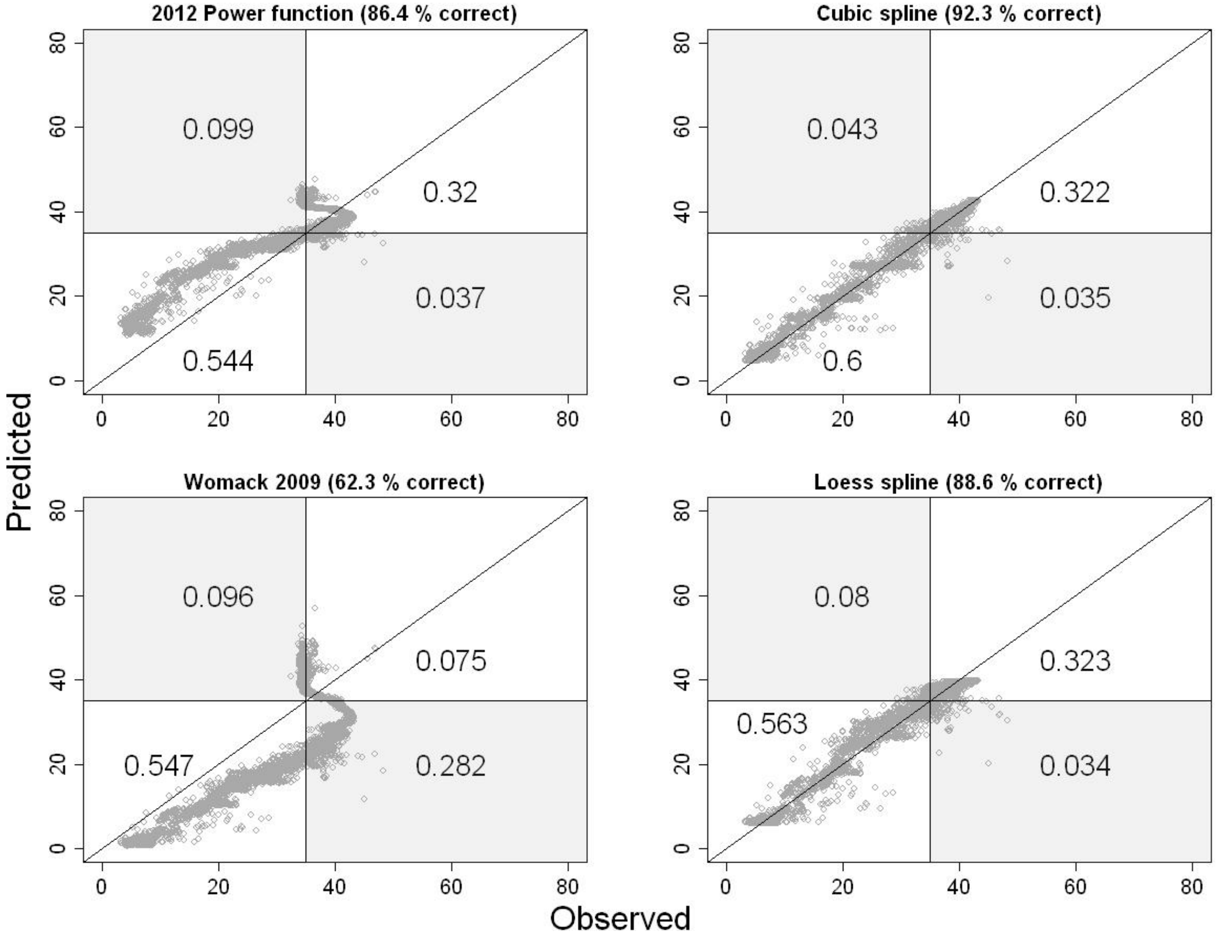


Figure A25. Average and total tow distance over all stations by critical dredge angle.

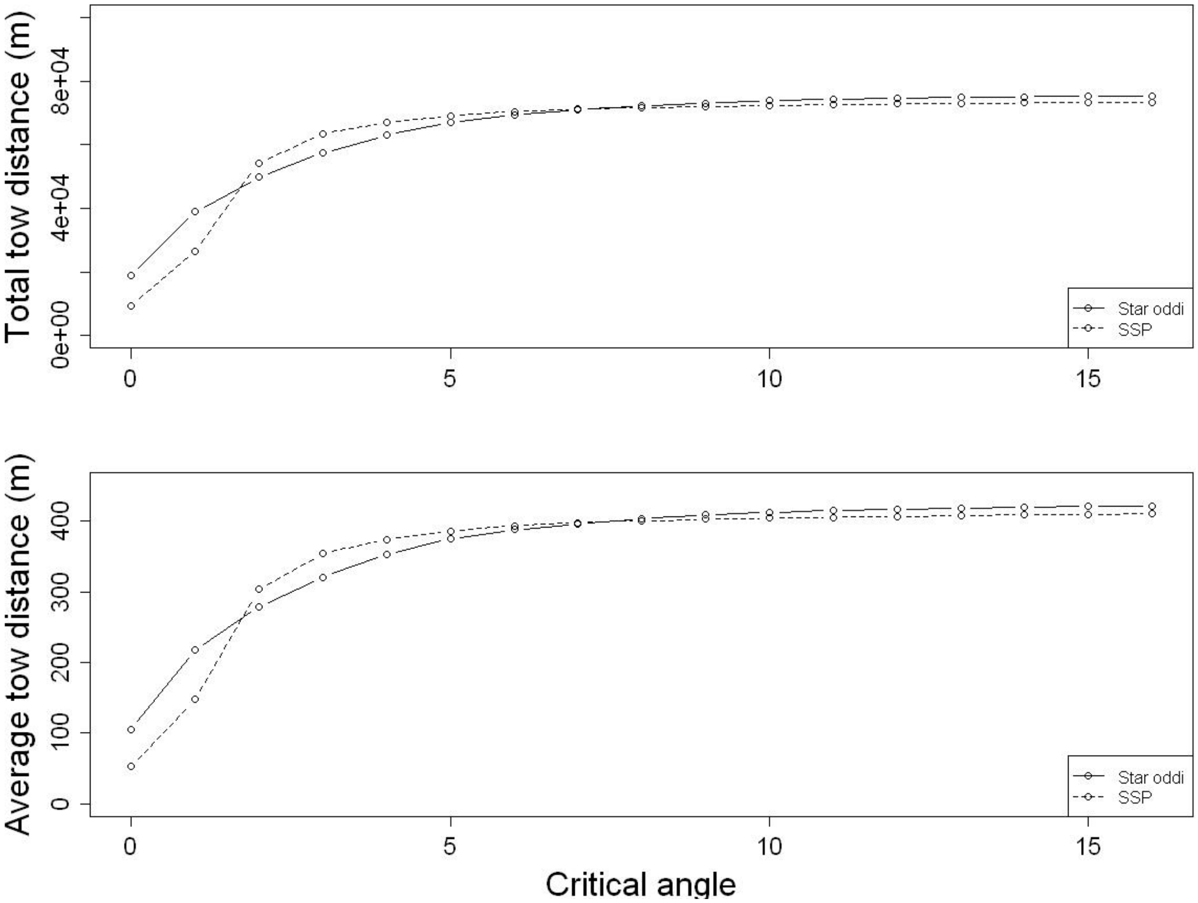


Figure A26. Surfclam 50 – 119 mm SL from NEFSC surveys adjusted for selectivity, with approximate 95% asymmetric confidence intervals, by region.

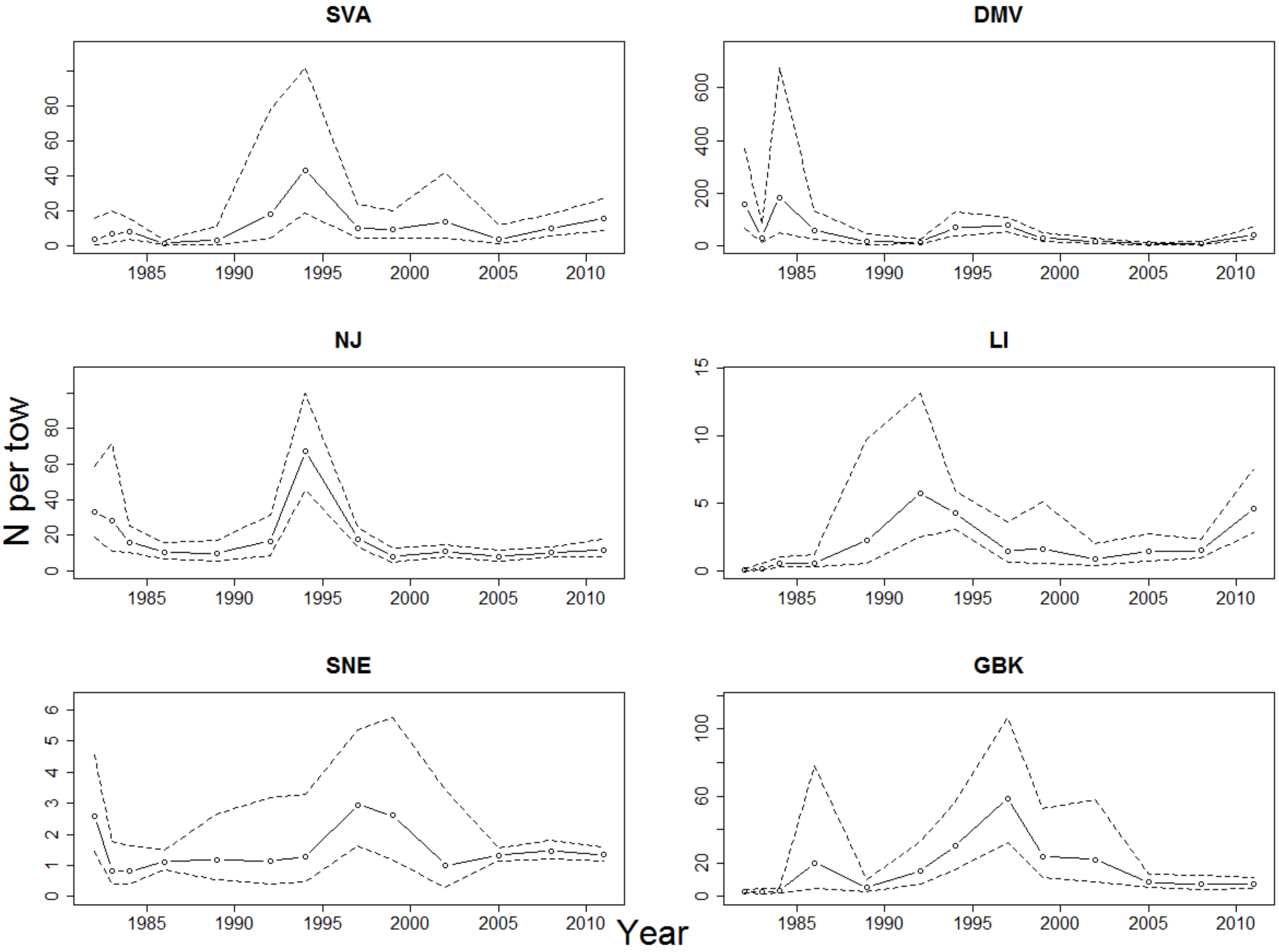


Figure A27. Surfclam larger than 120 mm SL from NEFSC surveys adjusted for selectivity, with approximate 95% asymmetric confidence intervals, by region.

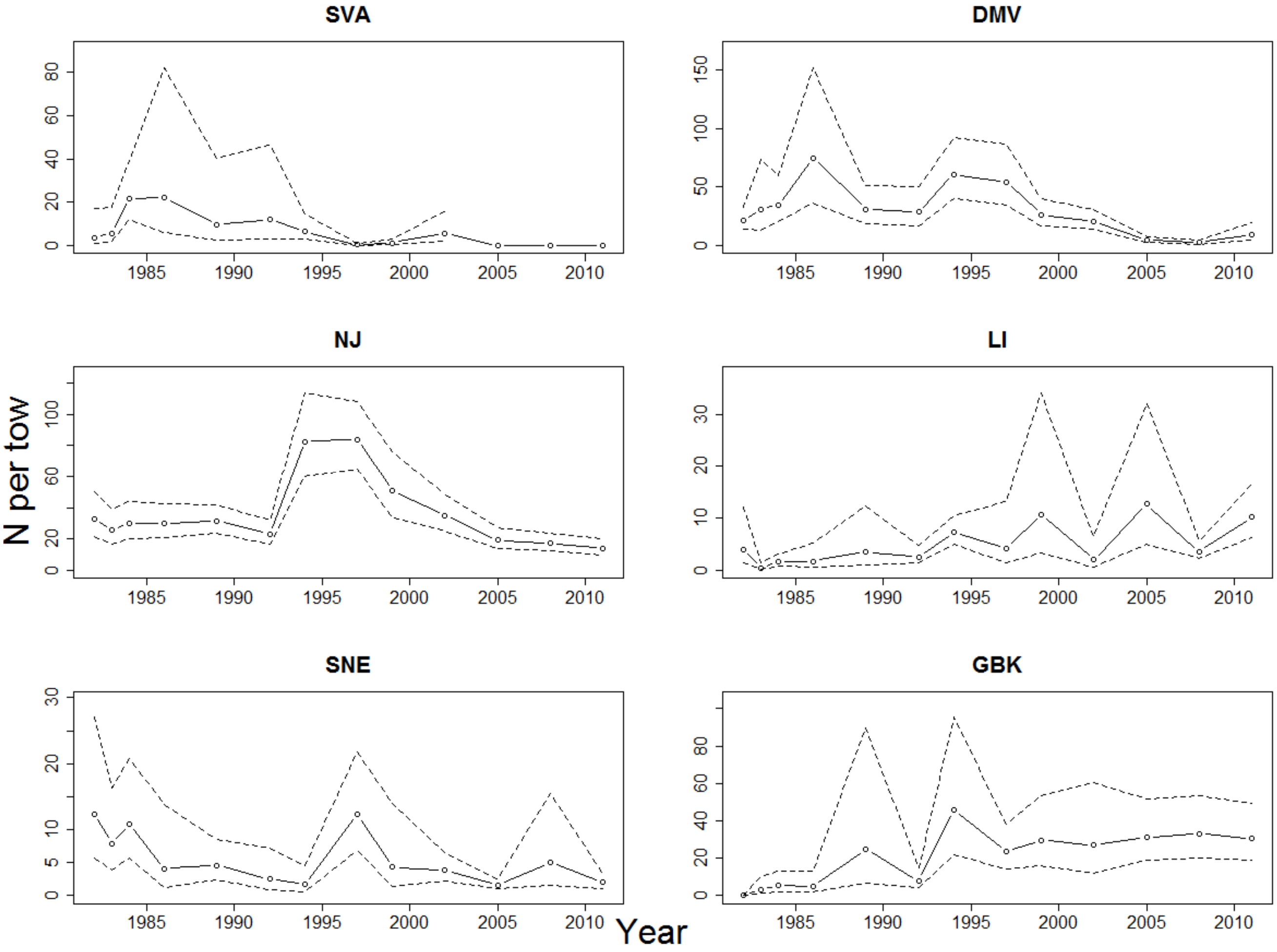


Figure A28. Surfclam 50 – 119 mm SL from NEFSC surveys adjusted for selectivity, with approximate 95% asymmetric confidence intervals for the whole stock.

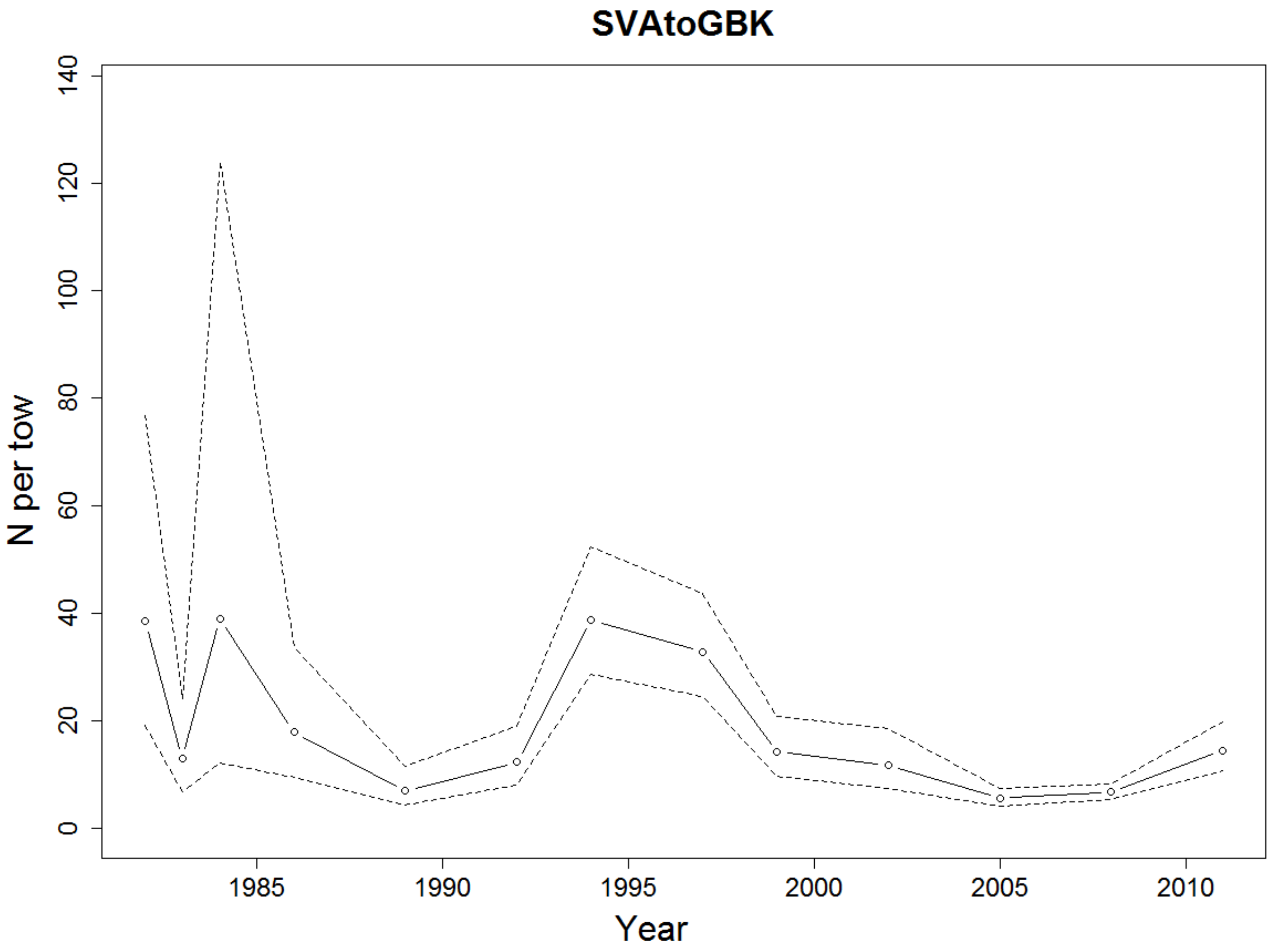


Figure A29. Surfclam larger than 120 mm SL from NEFSC surveys adjusted for selectivity, with approximate 95% asymmetric confidence intervals, for the whole stock.

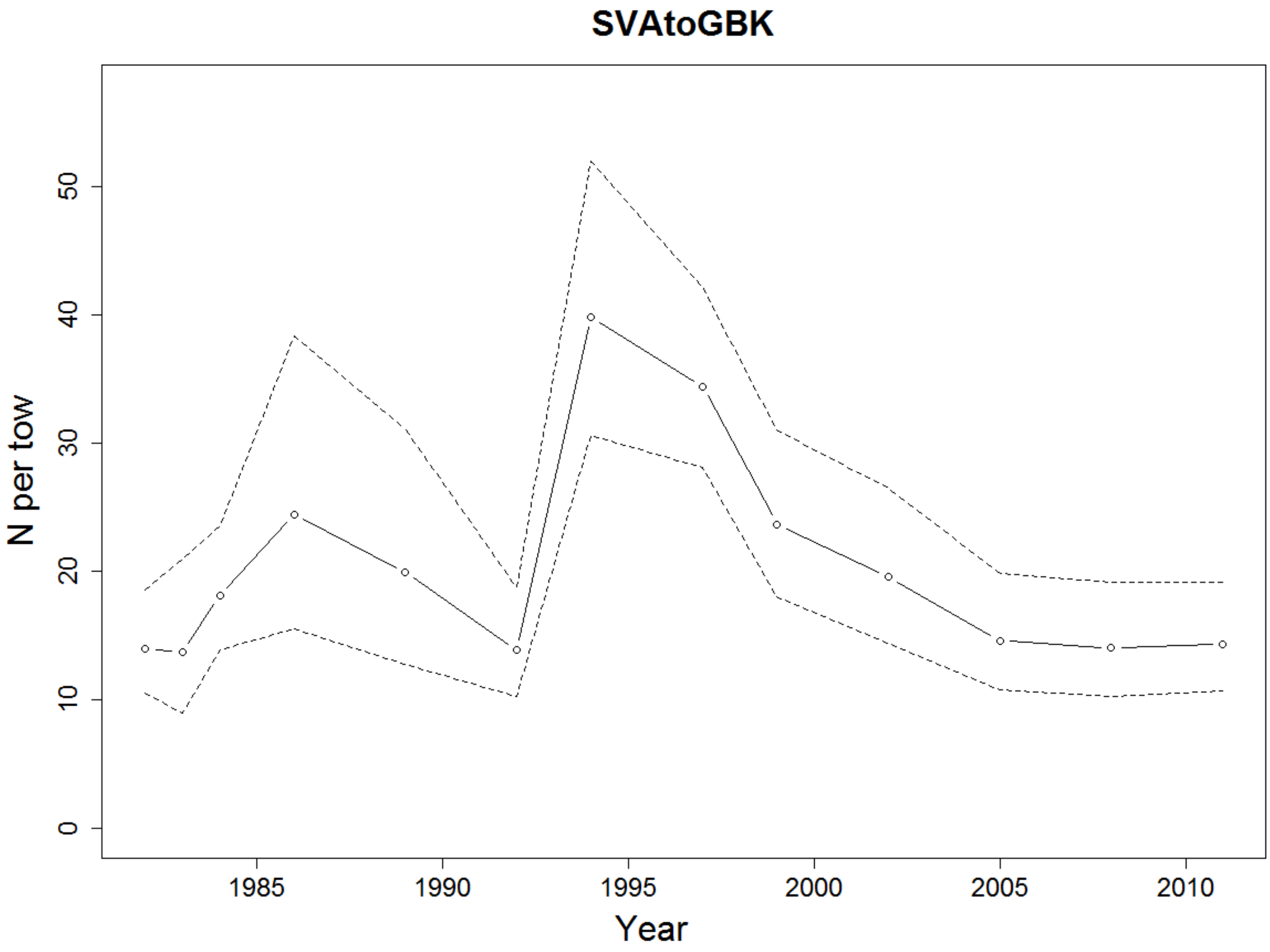
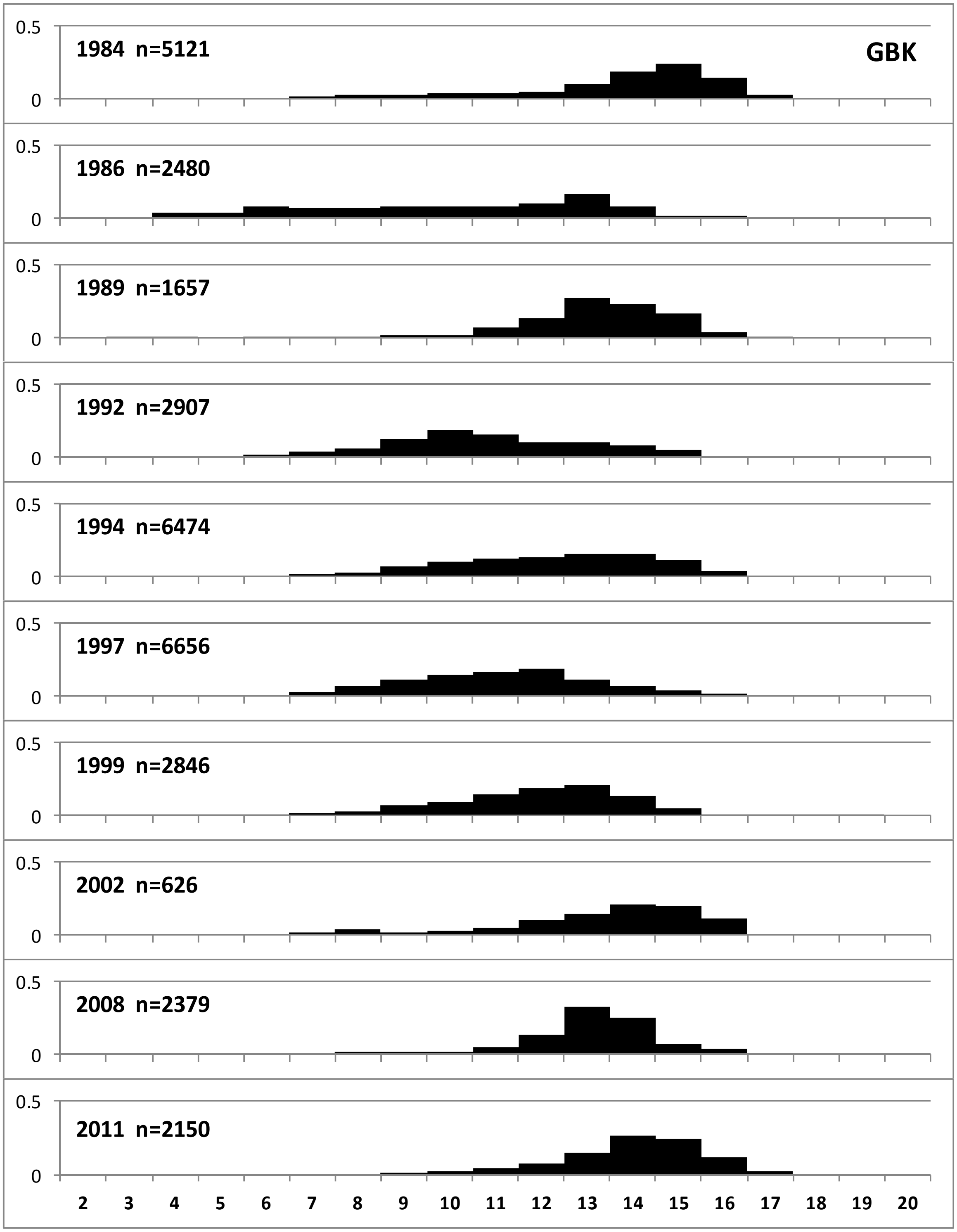
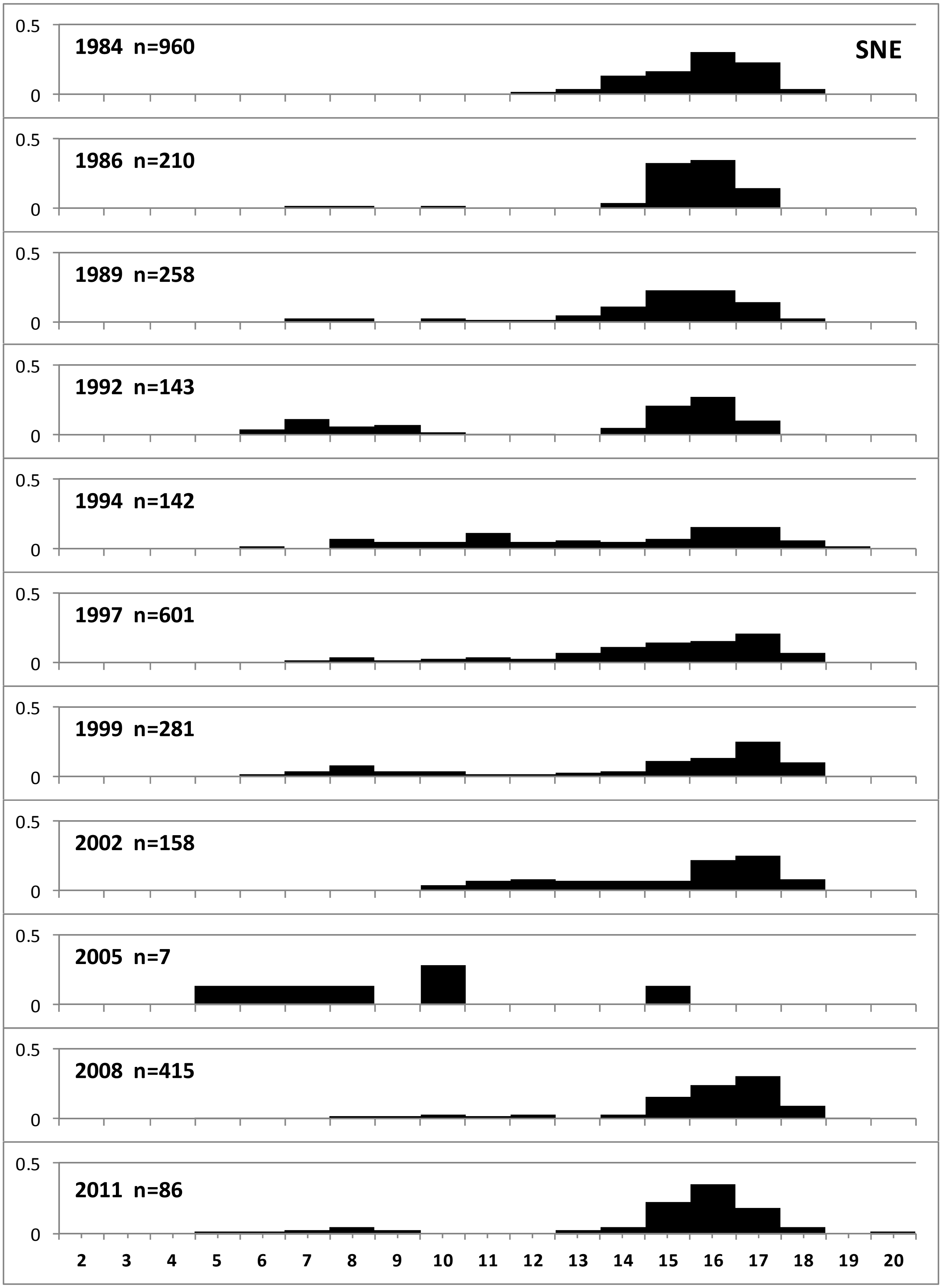
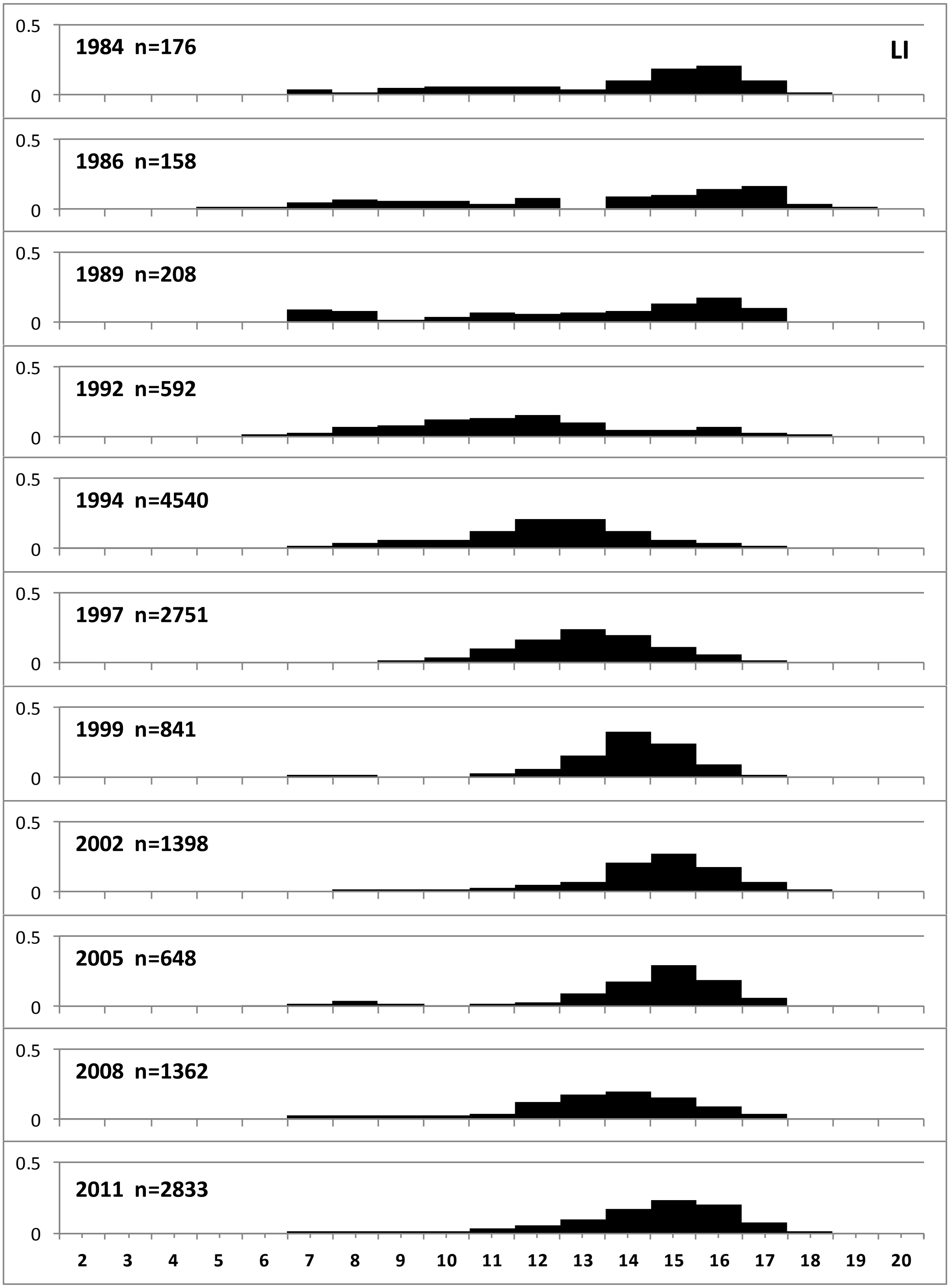
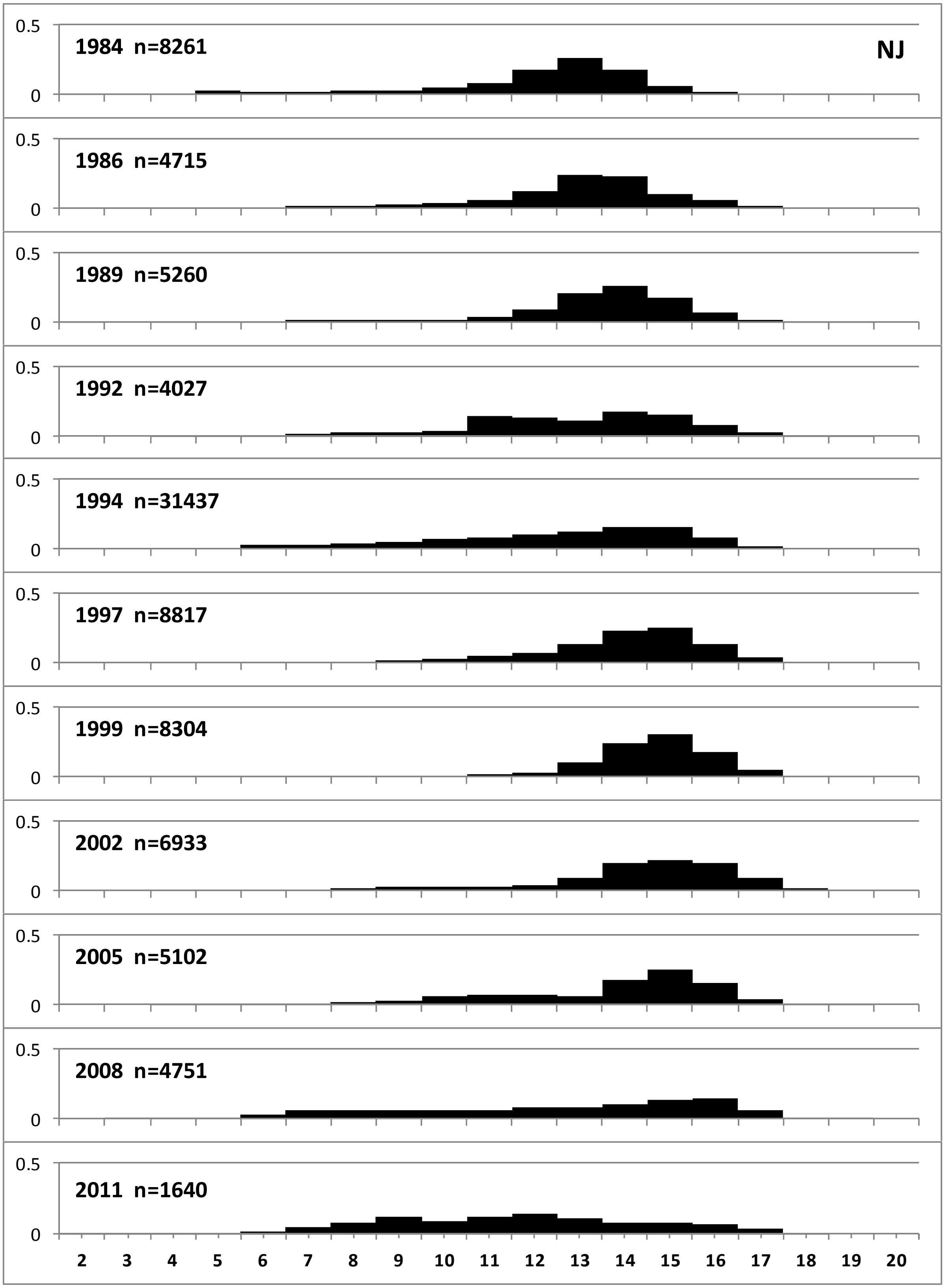


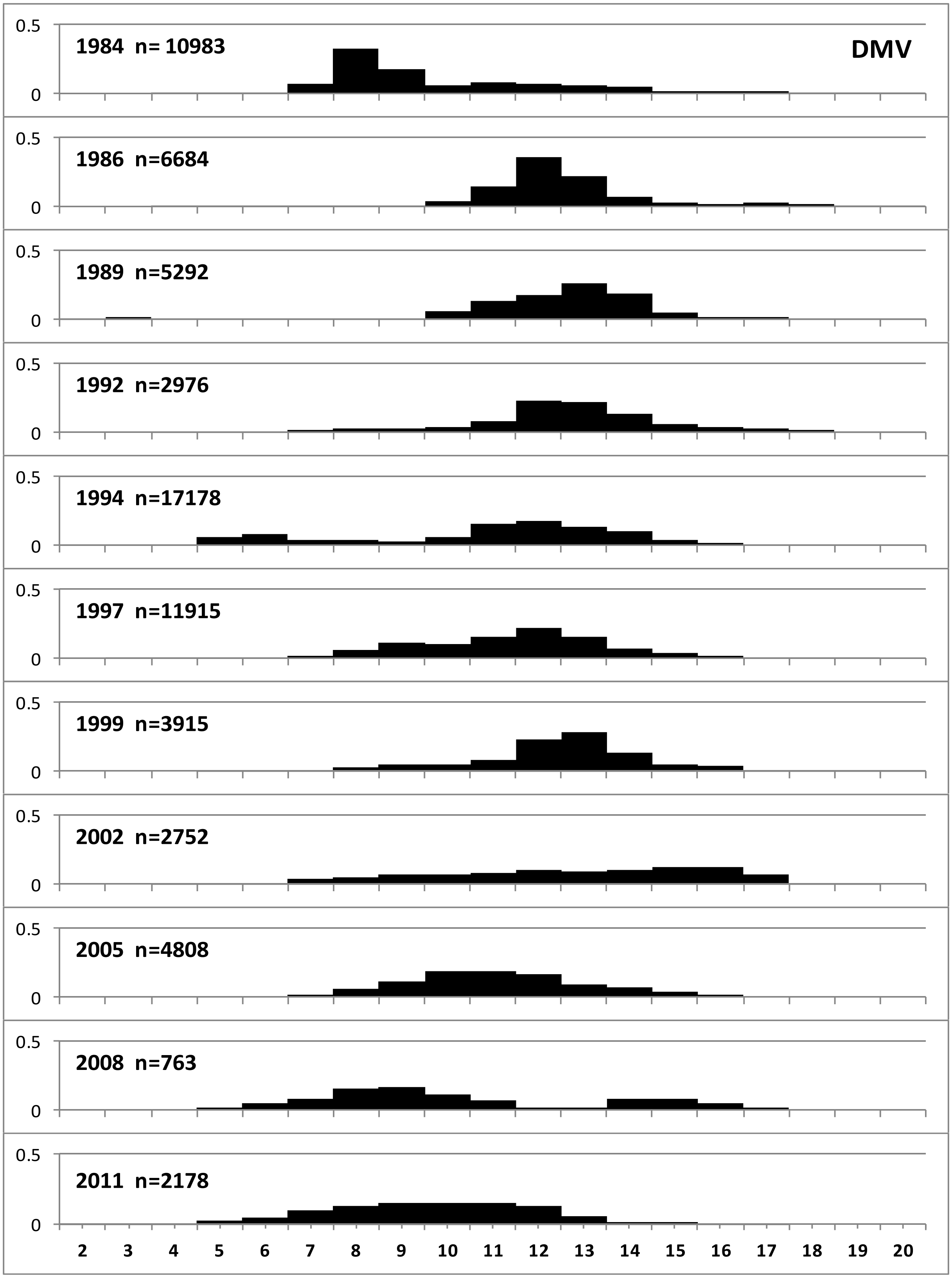
Figure A30. (Following pages) Survey length composition by region.


0.5
0
1984 n=5121
GBK
1986 n=2480
1989 n=1657
1992 n=2907
1994 n=6474
1997 n=6656
1999 n=2846
2002 n=626
2008 n=2379
2011 n=2150
2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20


0.5
0
1984 n=960
SNE
1986 n=210
1989 n=258
1992 n=143
1994 n=142
1997 n=601
1999 n=281
2002 n=158
2005 n=7
2008 n=415
2011 n=86
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20


0.5
0
LI
1984 n=176
1986 n=158
1989 n=208
1992 n=592
1994 n=4540
1997 n=2751
1999 n=841
2002 n=1398
2005 n=648
2008 n=1362
2011 n=2833
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20


0.5
0
1984 n=8261
NJ
1986 n=4715
1989 n=5260
1992 n=4027
1994 n=31437
1997 n=8817
1999 n=8304
2002 n=6933
2005 n=5102
2008 n=4751
2011 n=1640
2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20


0.5
0
1984 n= 10983
DMV
1986 n=6684
1989 n=5292
1992 n=2976
1994 n=17178
1997 n=11915
1999 n=3915
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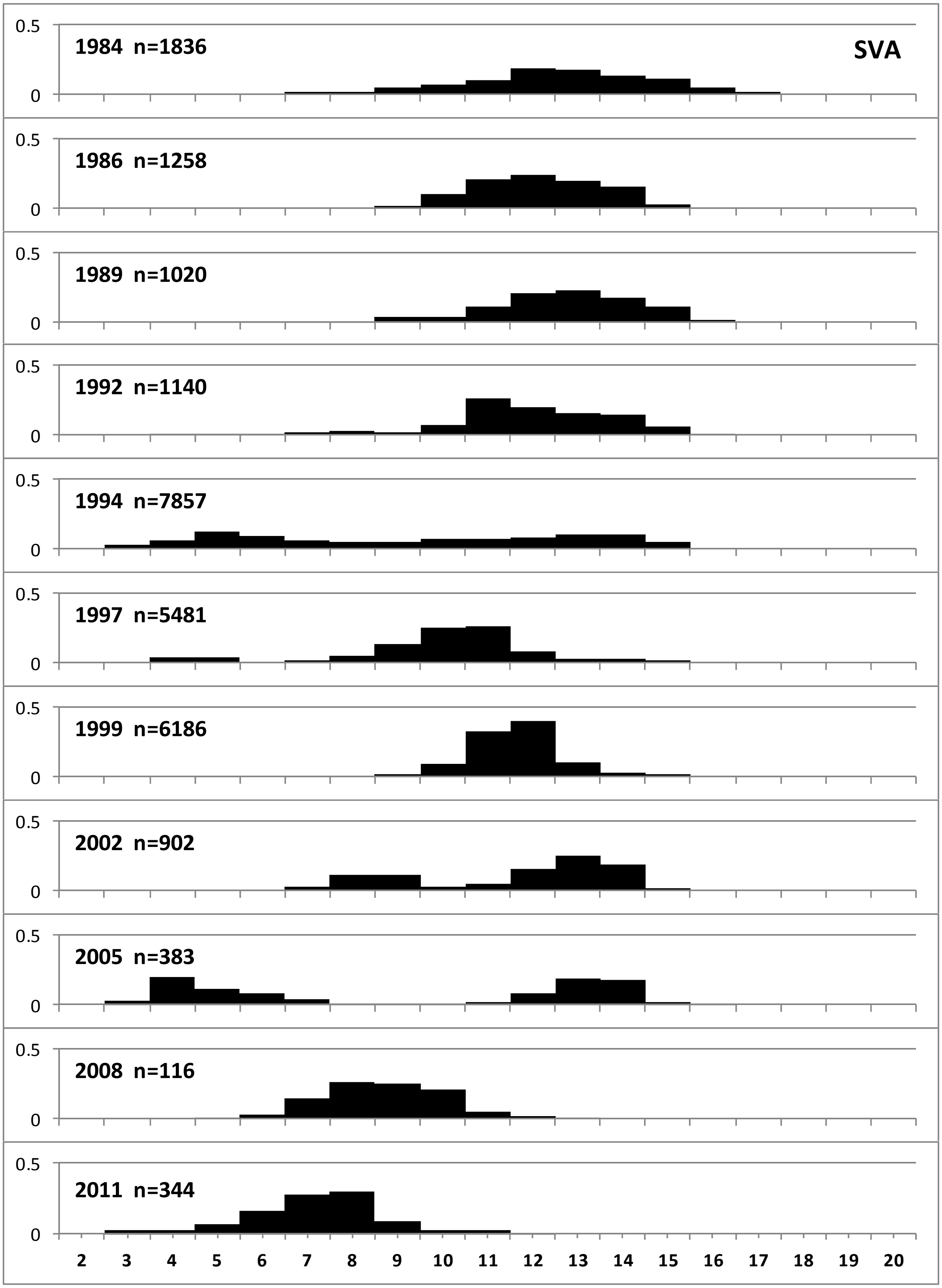


Figure A58.1. Age composition of NEFSC surveys in SVA.

Figure A31. Age composition of NEFSC surveys in SVA.

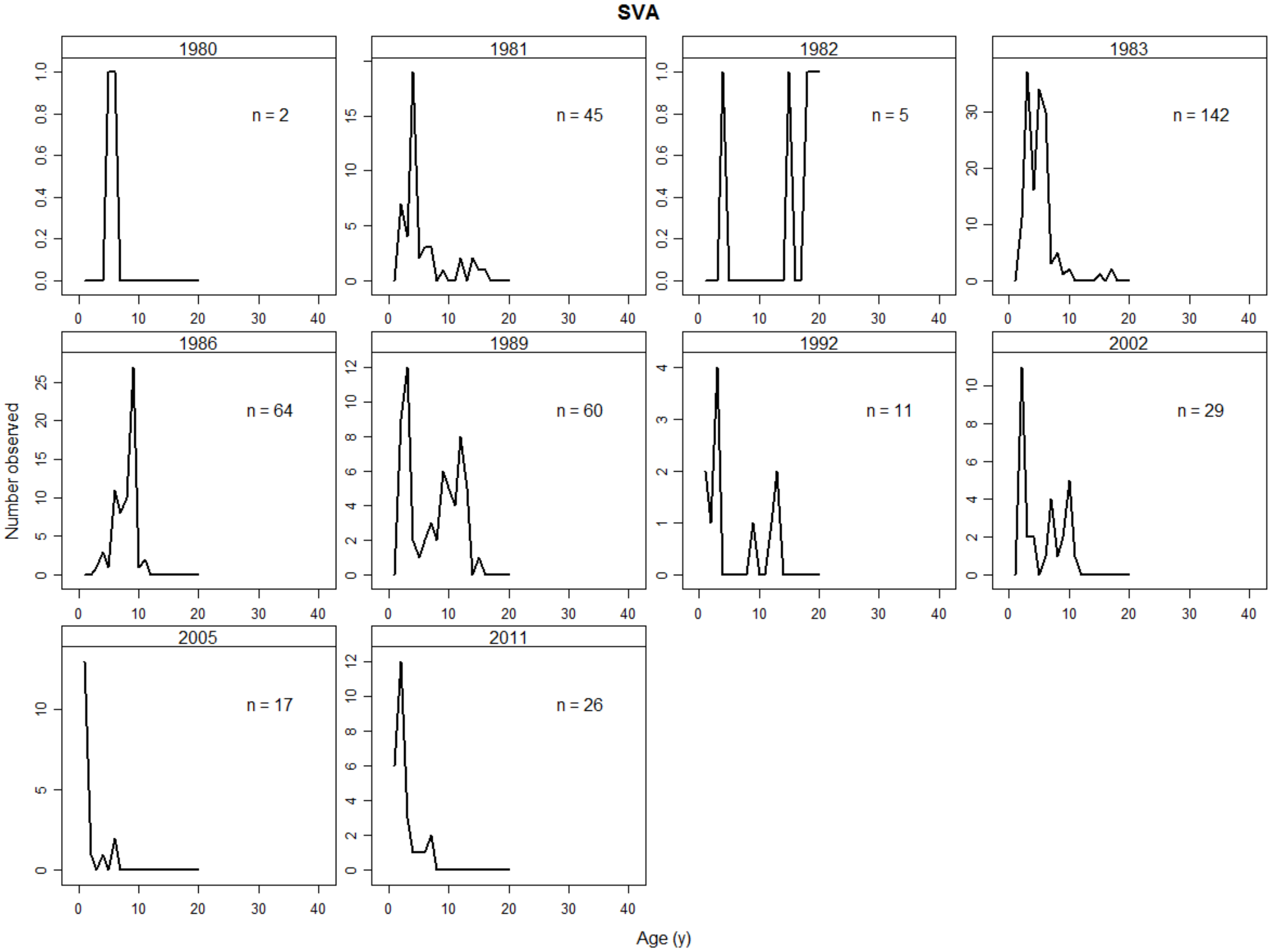


Figure A32. Age composition of NEFSC surveys in DMV.

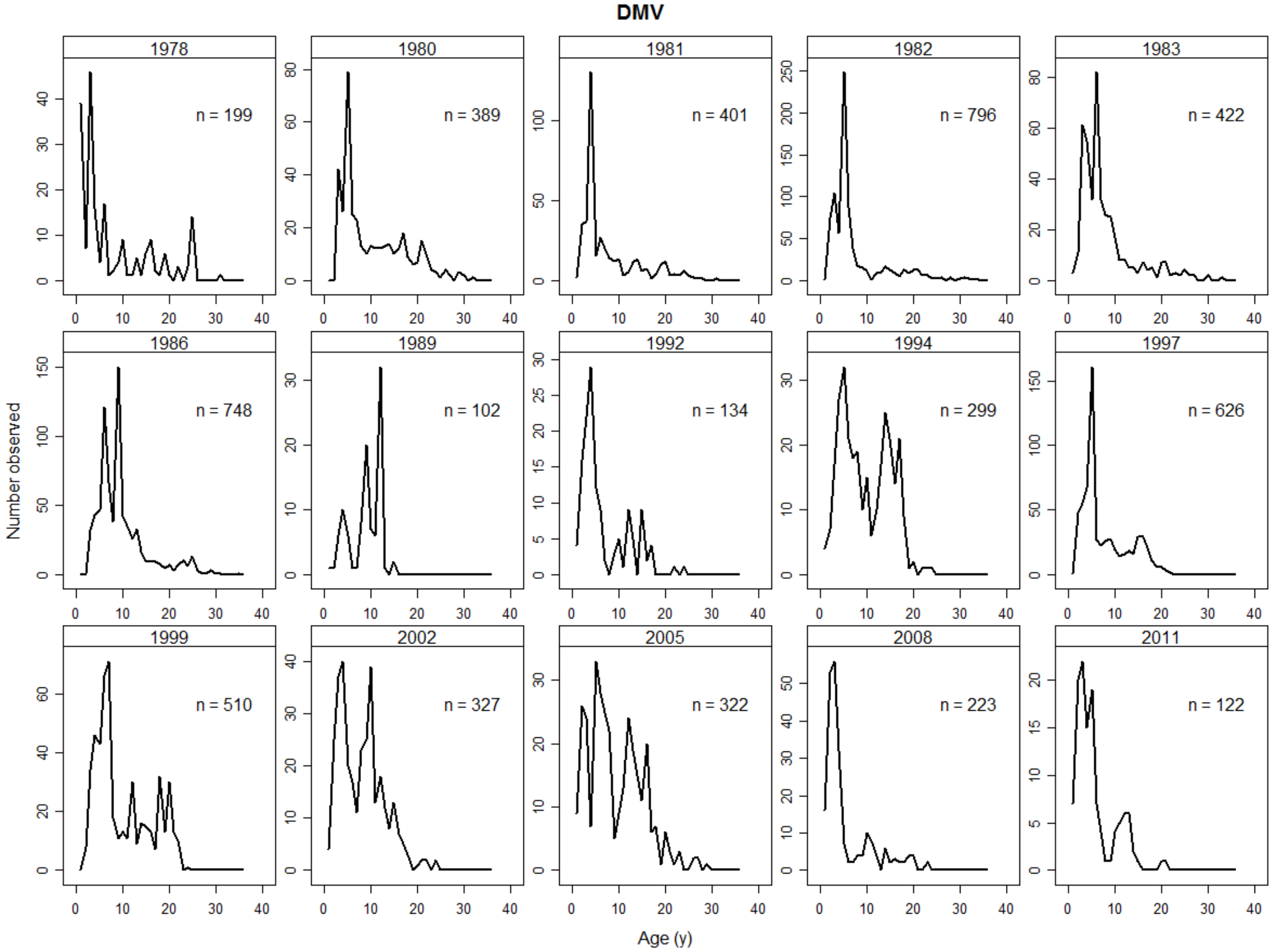


Figure A33. Age composition of NEFSC surveys in NJ.

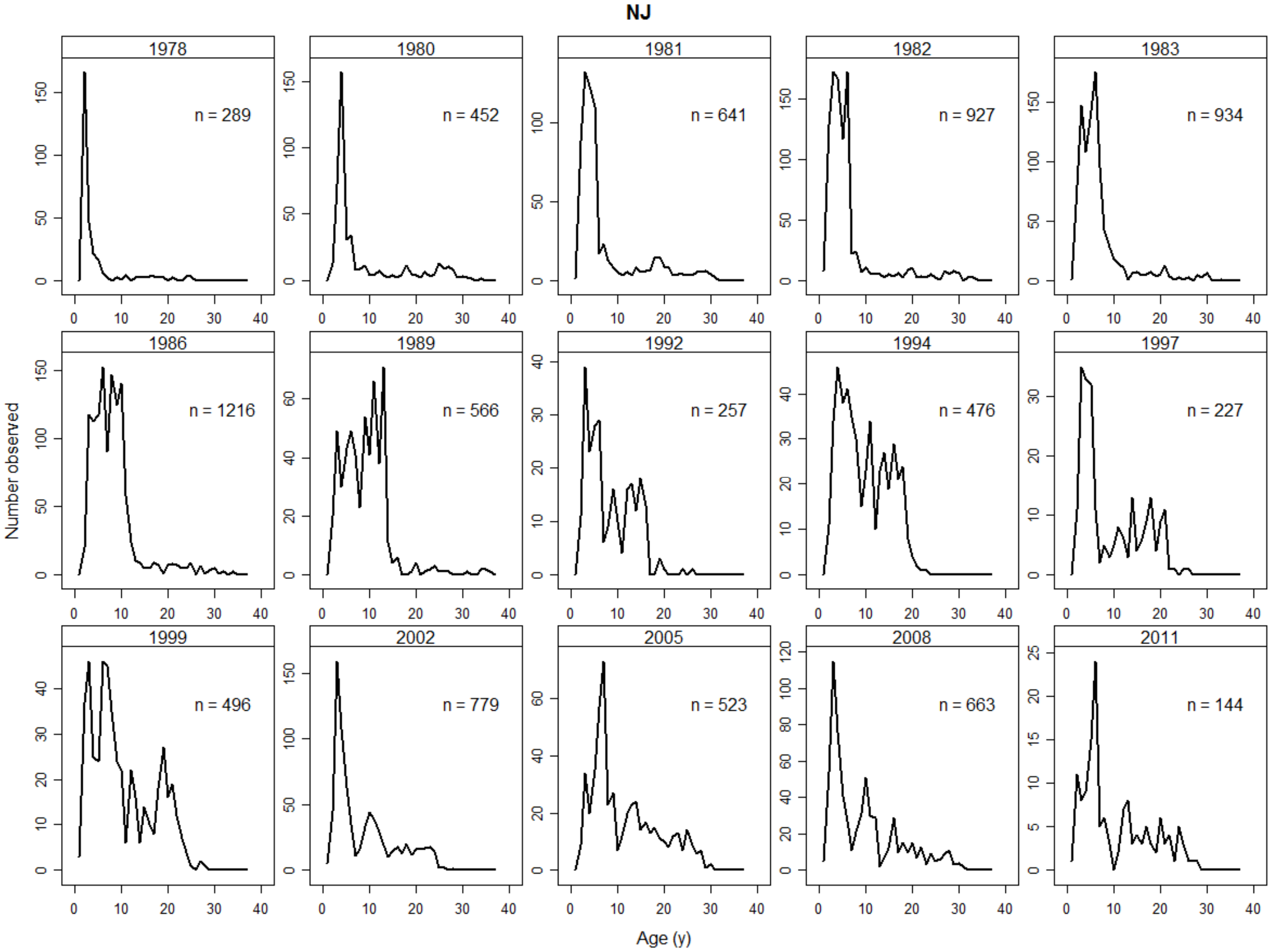


Figure A34. Age composition of NEFSC surveys in LI.

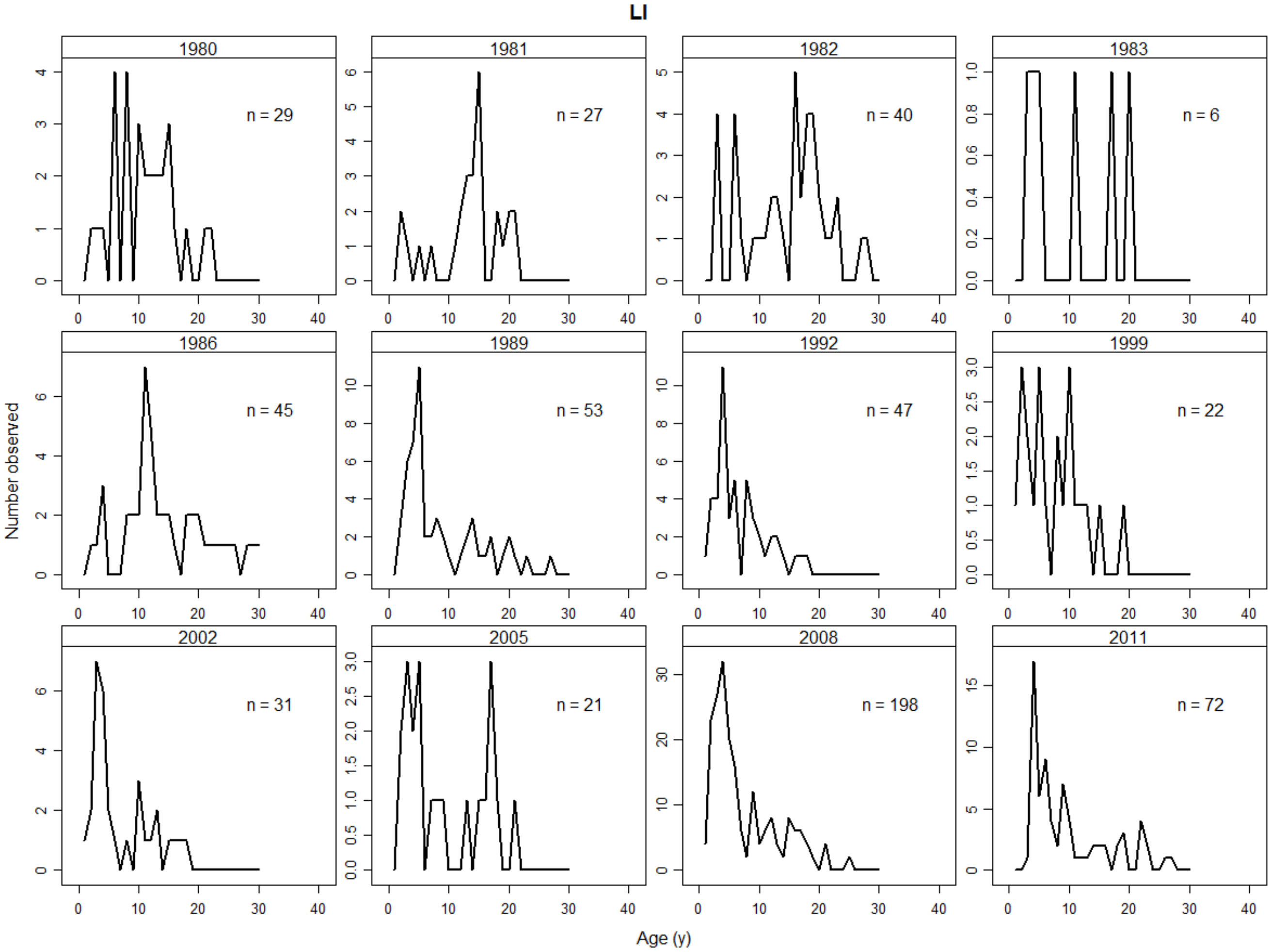


Figure A35. Age composition of NEFSC surveys in SNE.

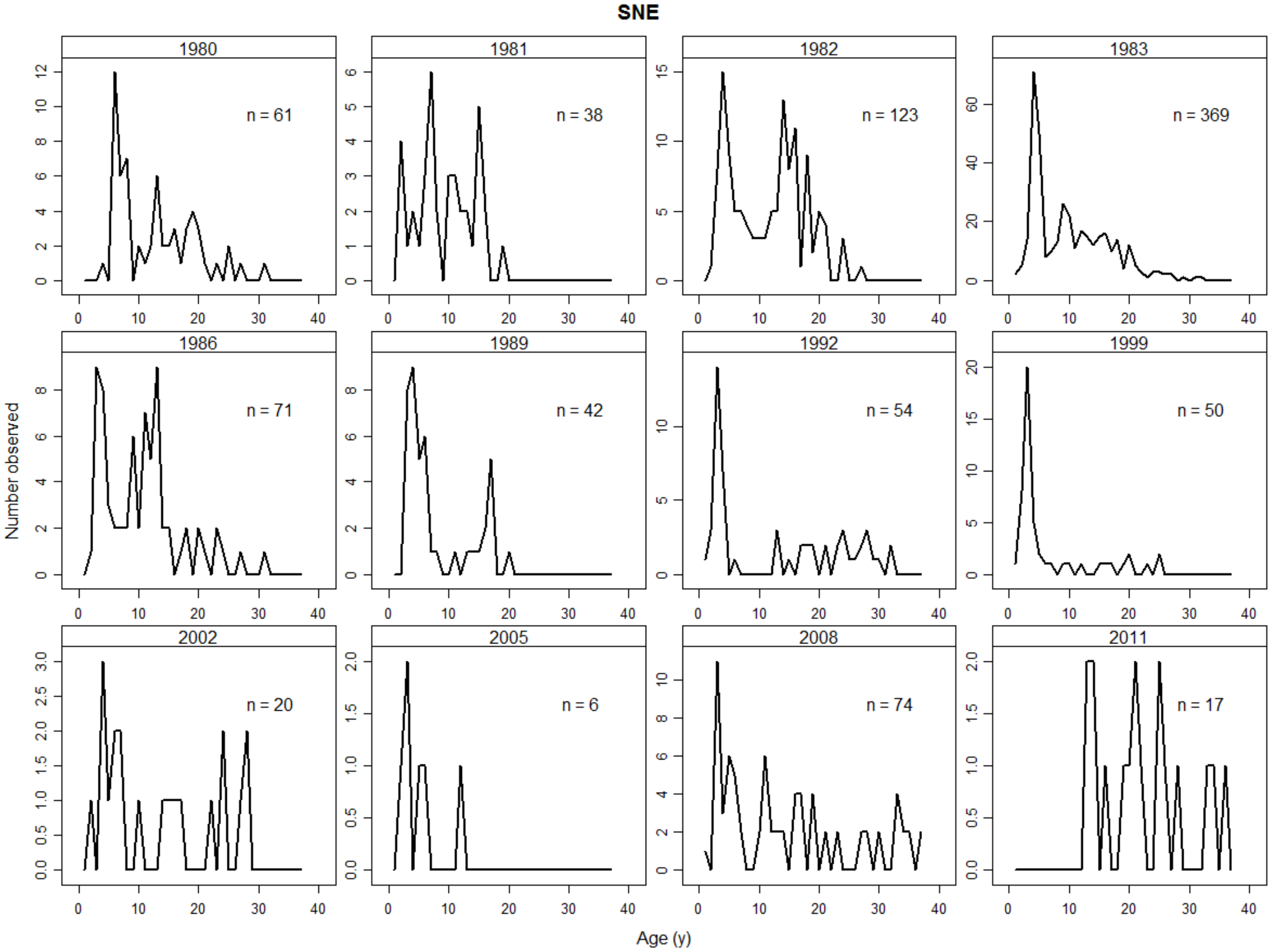


Figure A36. Age composition of NEFSC surveys in GBK.

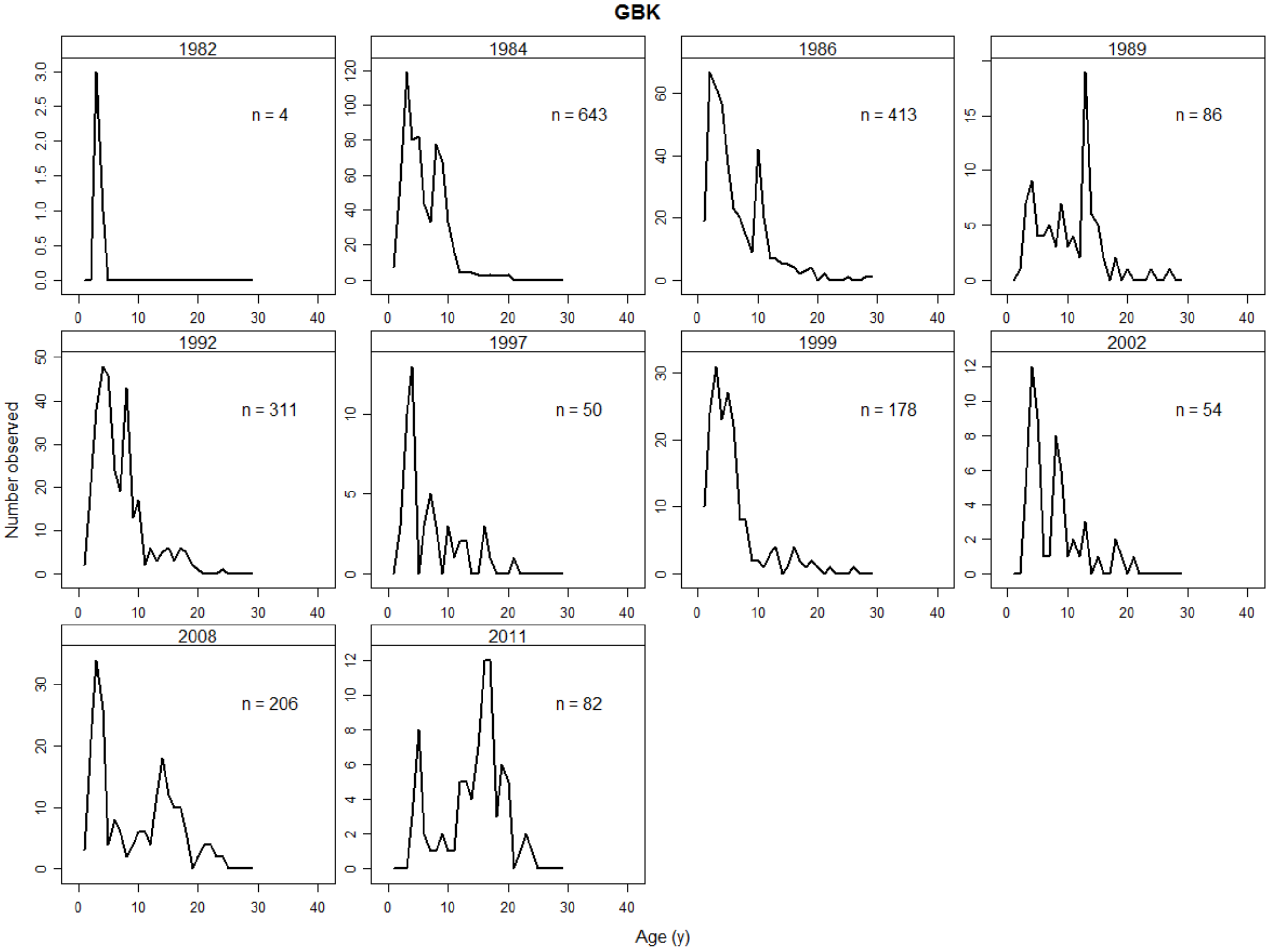
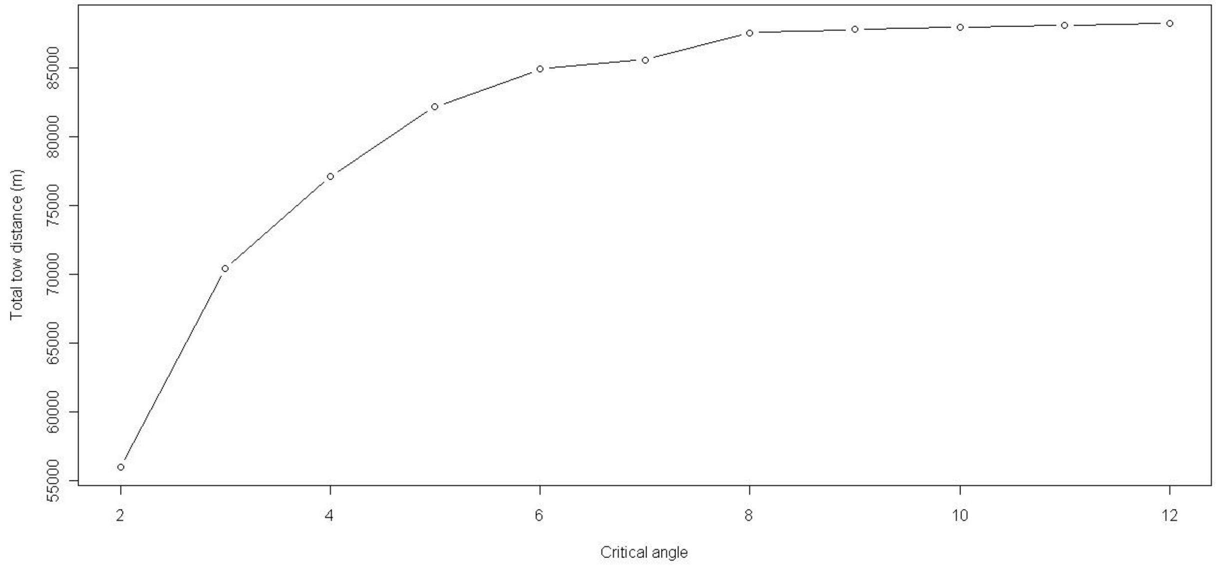


Figure A37. Total and average tow distance across all depletion experiments conducted in 2011 by the critical angle measured by the inclinometer and used to determine if the dredge was actively fishing. A larger critical angle results in more time fishing. The curve appears to asymptote at approximately 8 degrees and any critical angle between 8 and 12 degrees will produce approximately the same total and average tow distance.



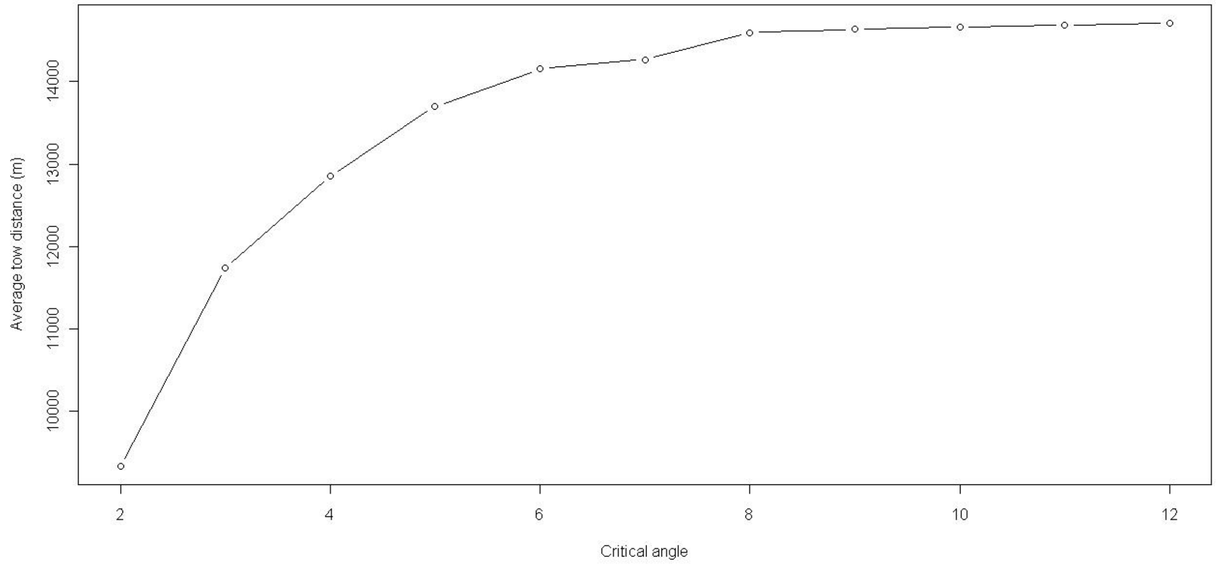


Figure A38. The total and average tow distance across all tows within each depletion experiment (including to Ocean quahog experiments) calculated using two common smoothing algorithms: loess and GAM splines. The choice of smoother did not appear to bias tow distance systematically.

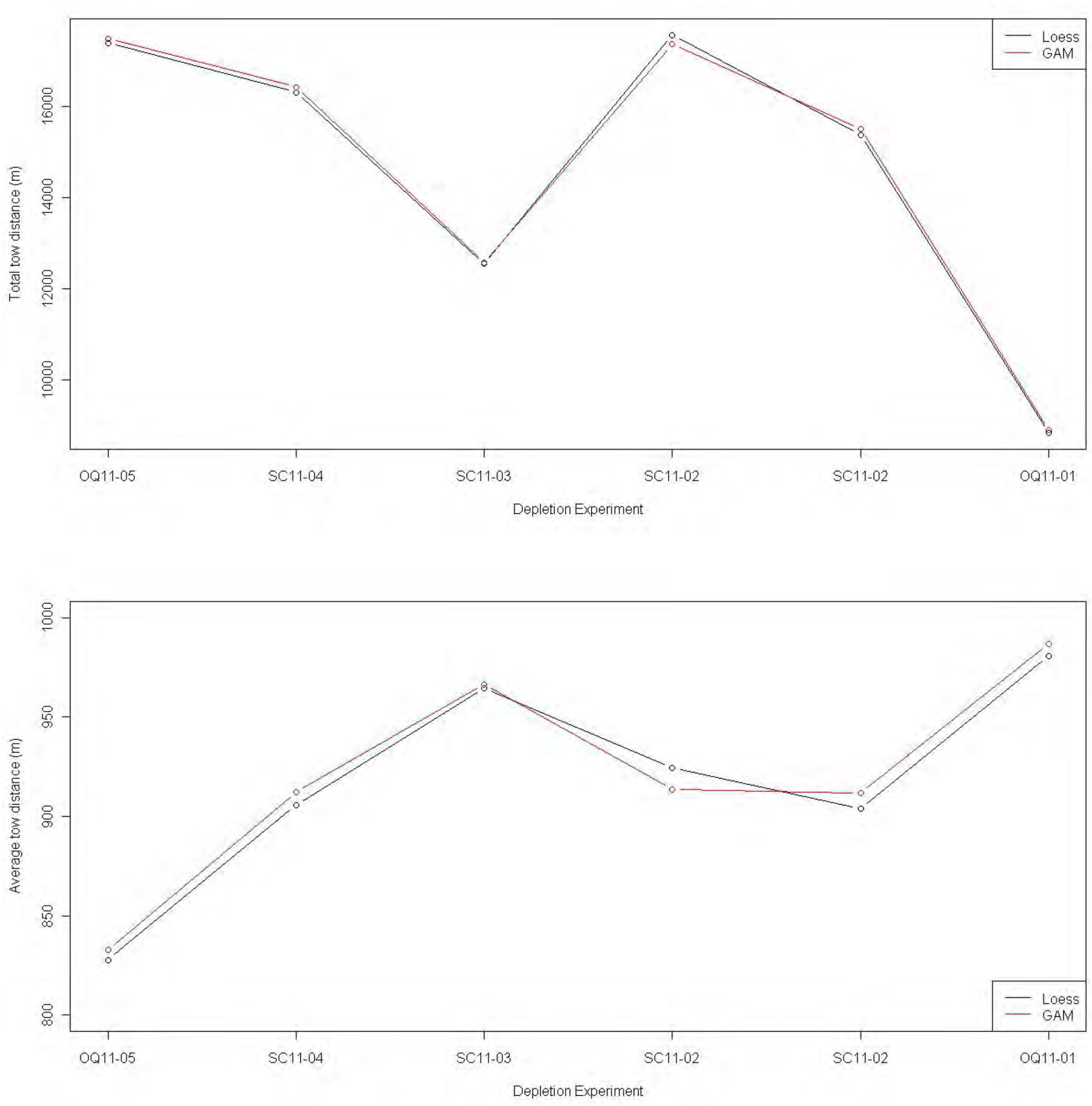


Figure A39. A comparison of the relative confidence in the components of the ratio used to estimate dredge efficiency. *D* is the density estimated in depletion experiments using the Patch model, while *d* is the density estimated using the set ups tows. The variability in *d* is relatively high compared to the variability in *D*. The dotted lines are for reference and represent a CV = 0.5 for each component.

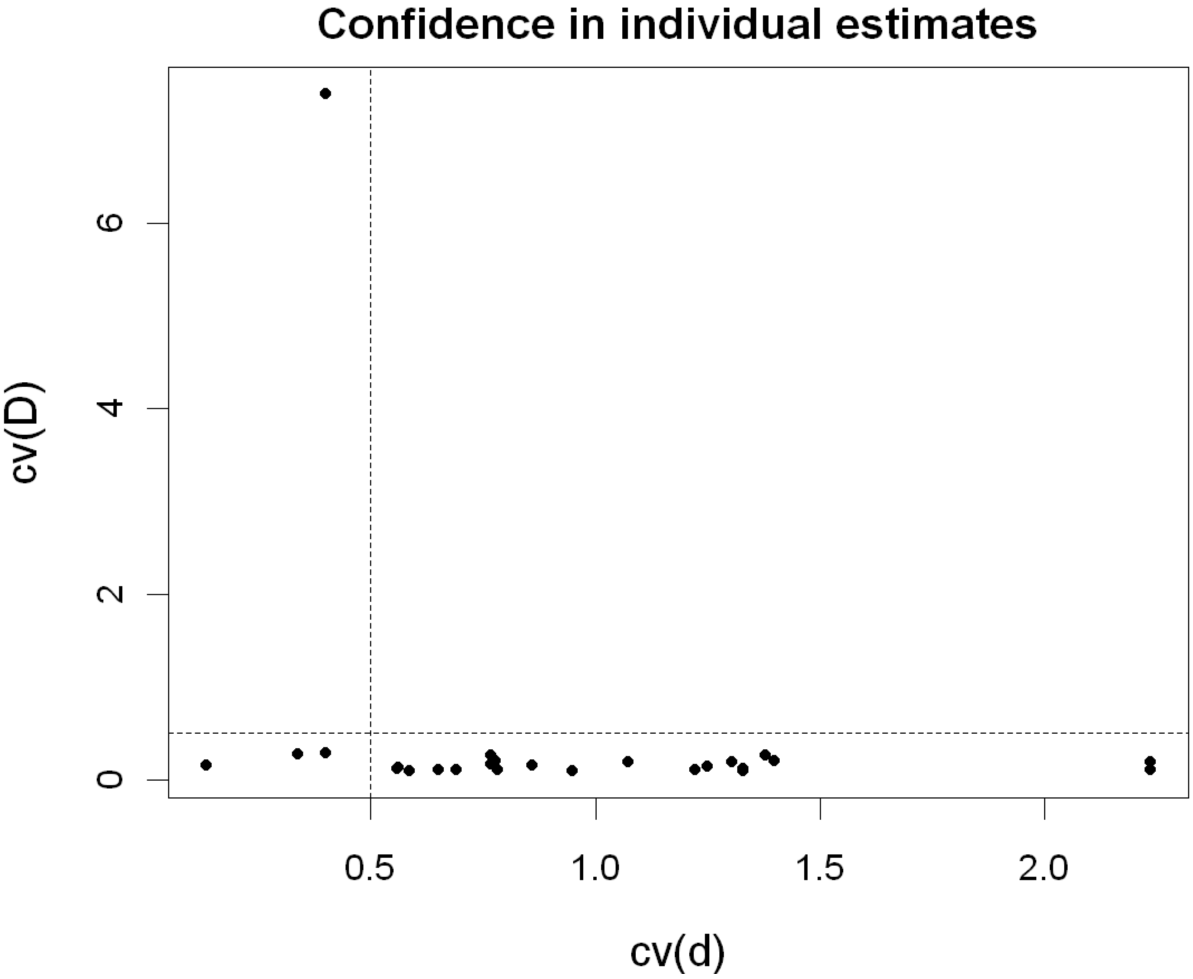


Figure A40. The set of prior knowledge for dredge efficiency estimates. Each individual estimate is shown with an error bar representing the magnitude of its CV.

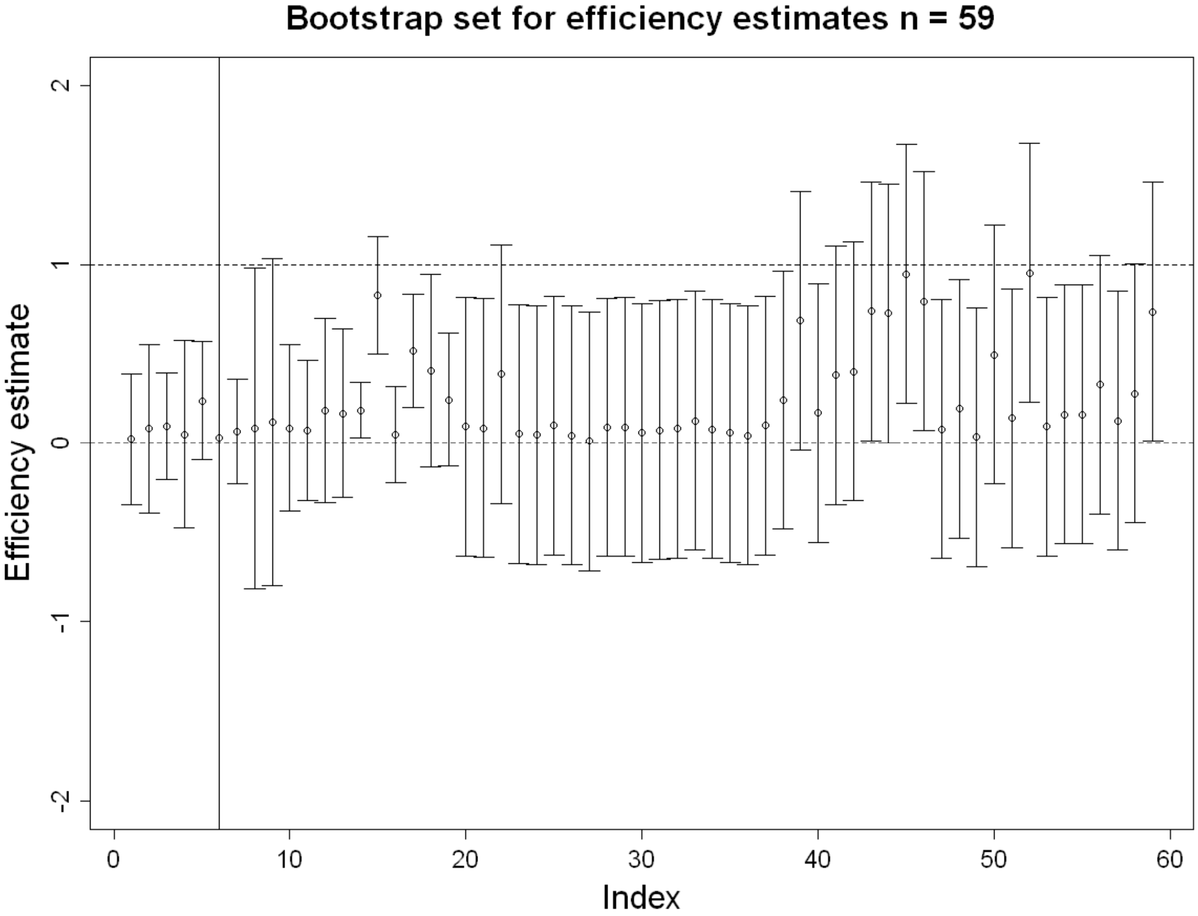


Figure A41. Bootstrapped data set and log normal fit. The distribution shown here is the prior distribution for survey dredge efficiency used in the assessment.

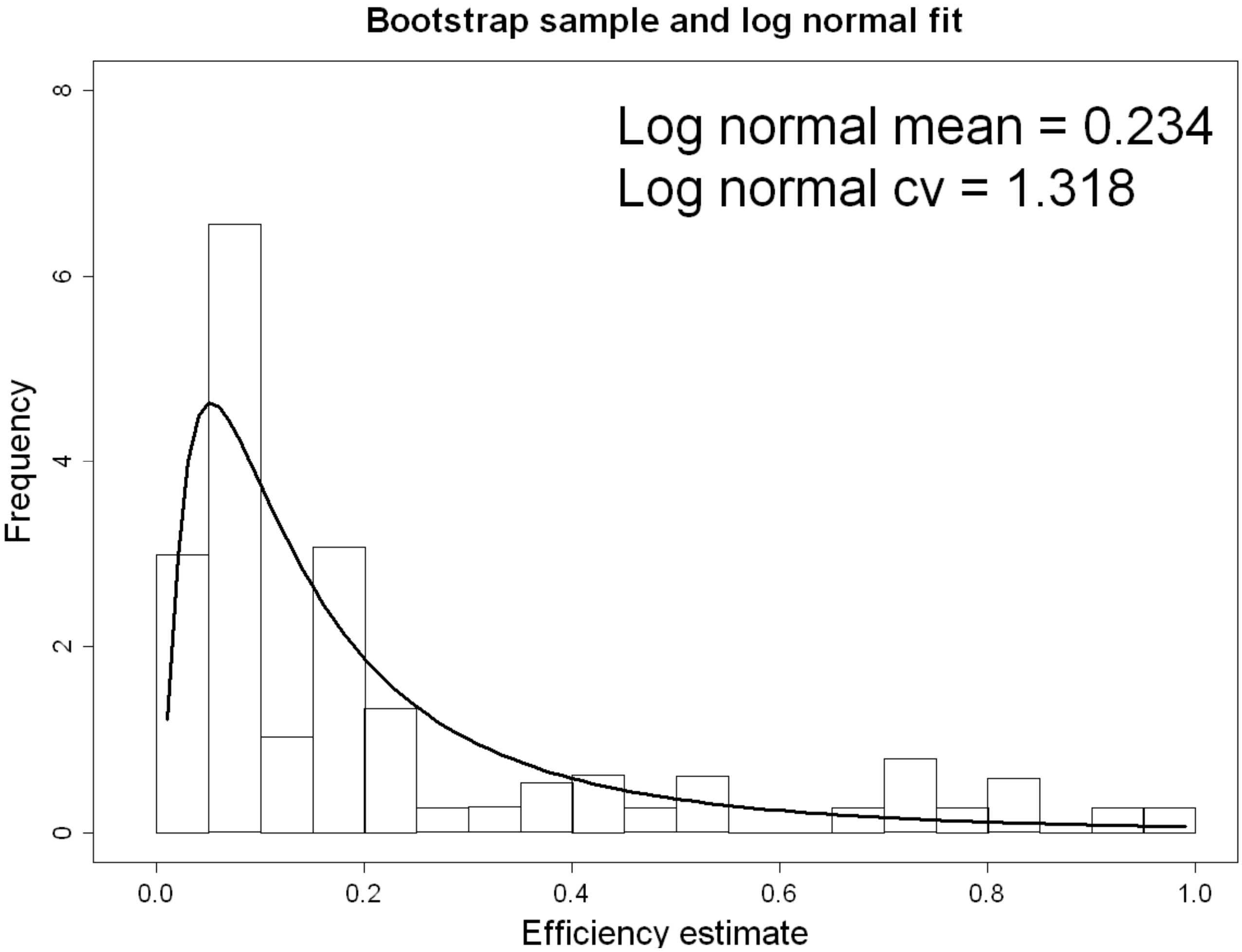


Figure A42. Maps of the tow sequence for all surfclam depletion experiments conducted in 2011.

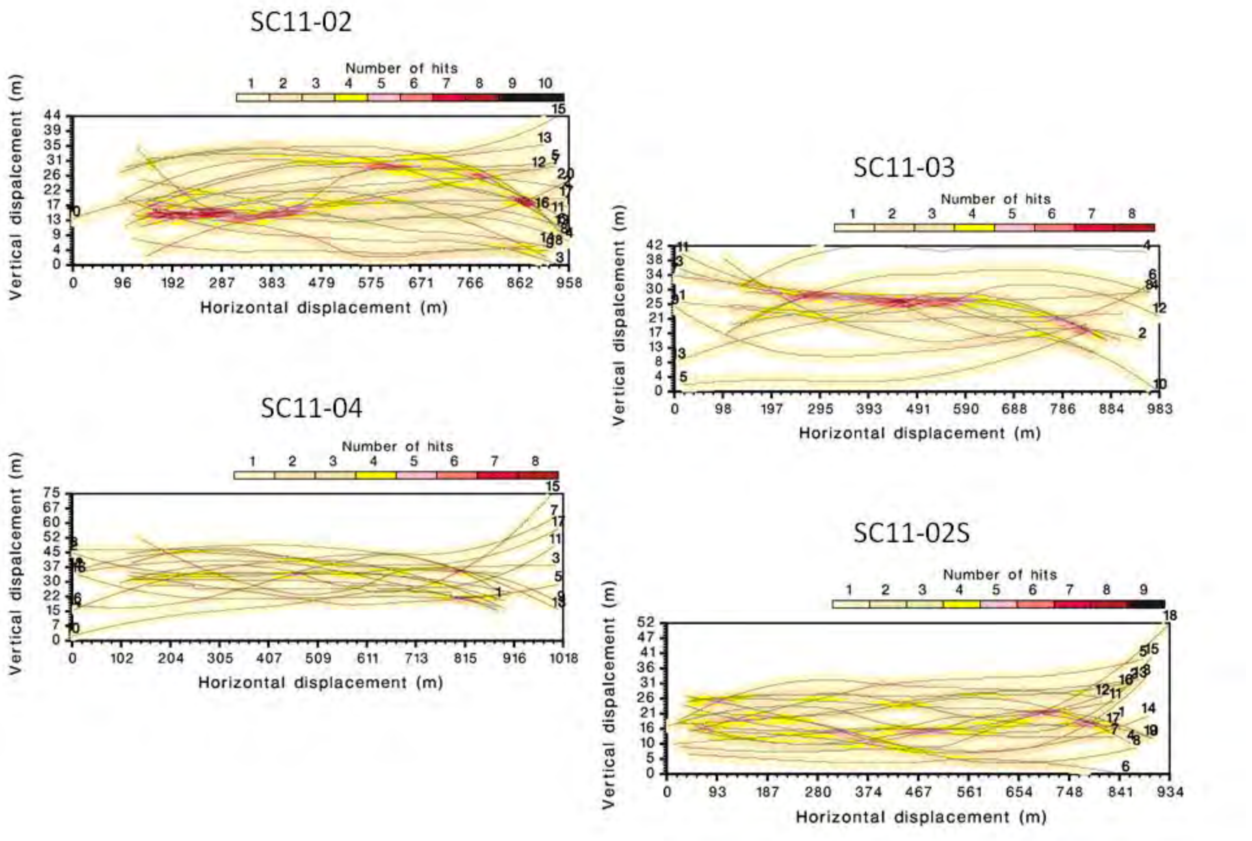


Figure A43. Patch model diagnostics for depletion experiment SC11-04. These include: catch by tow, catch per unit of effective area swept, catch vs. expected catch and the likelihood residuals from the patch model fit. Effective area swept accounts for the proportion of ground that is being repeatedly fished for the first, second, third, etc... overlapping tow. The expected catch is the catch predicted by the Patch model.

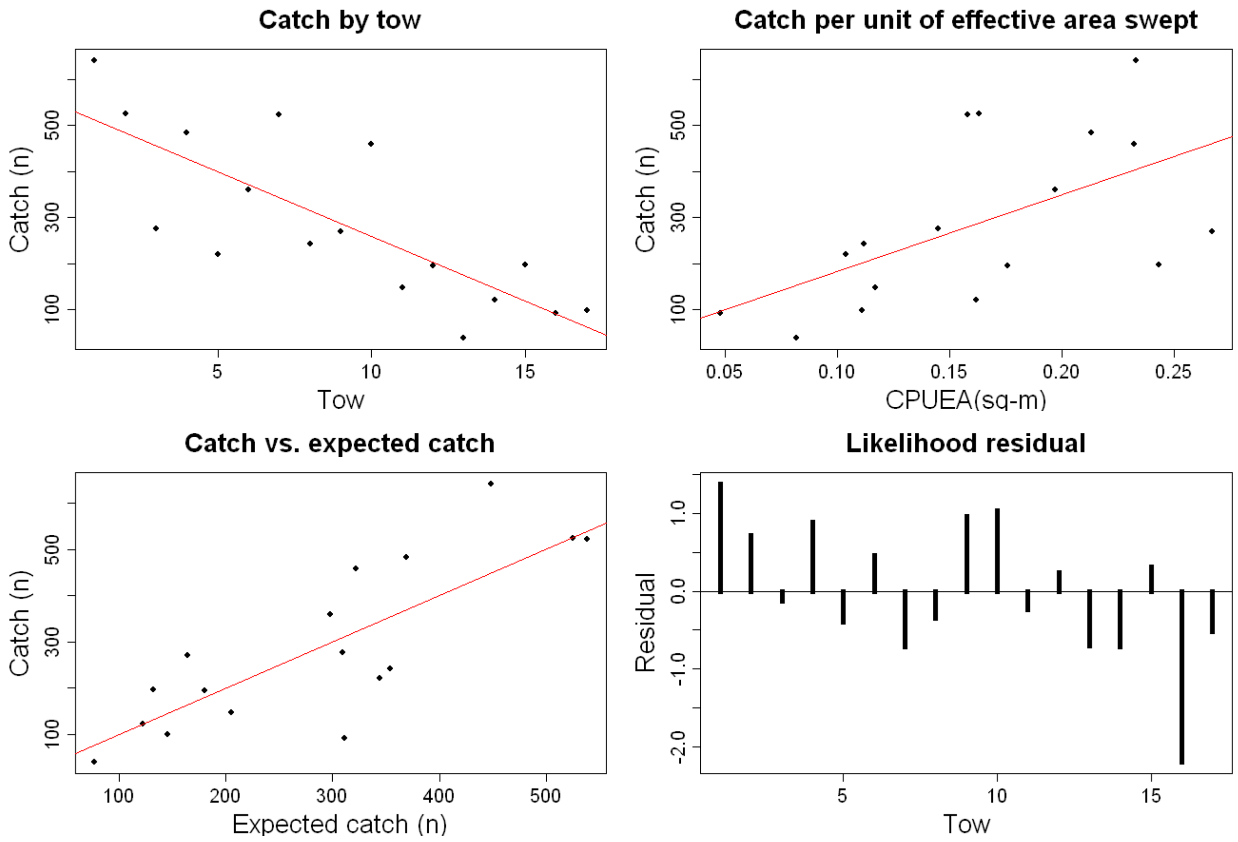


Figure A44. Patch model diagnostics for SC11-02.

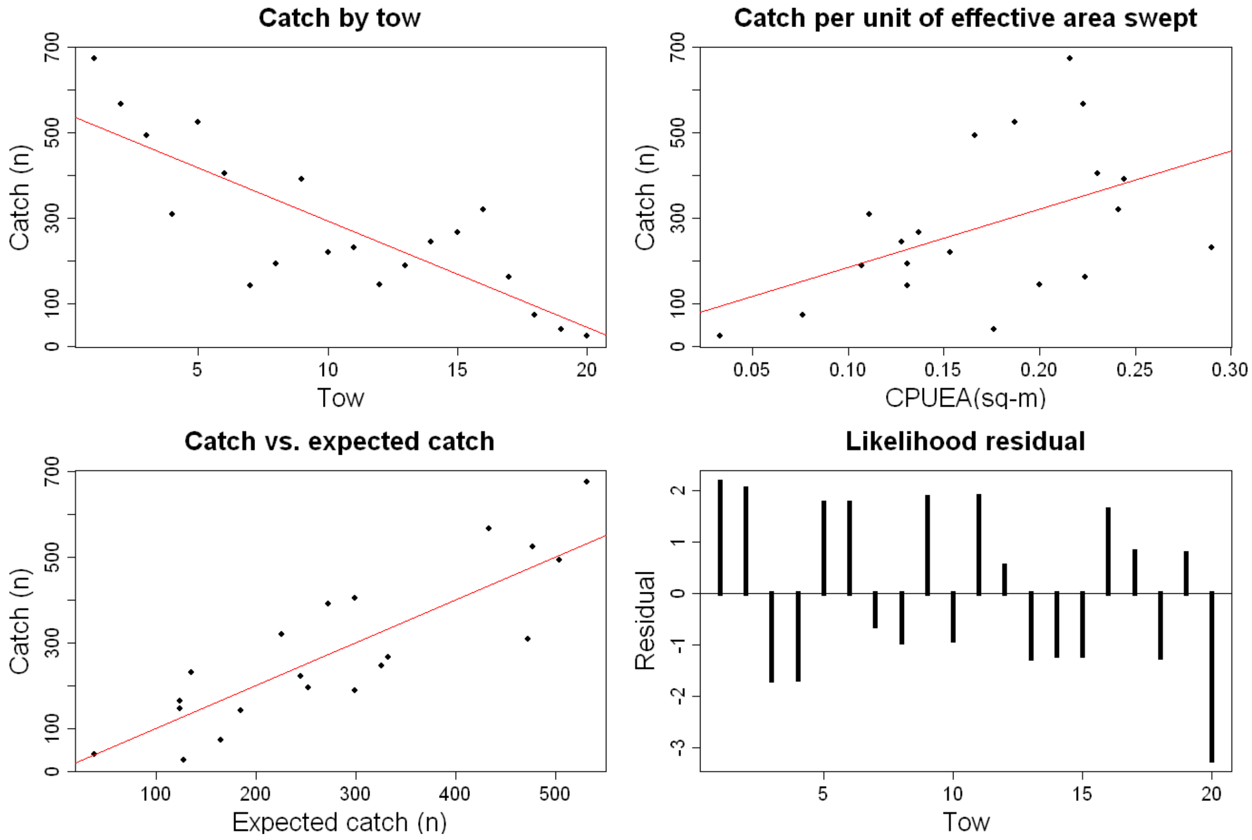


Figure A45. Patch model diagnostics for SC11-02S.

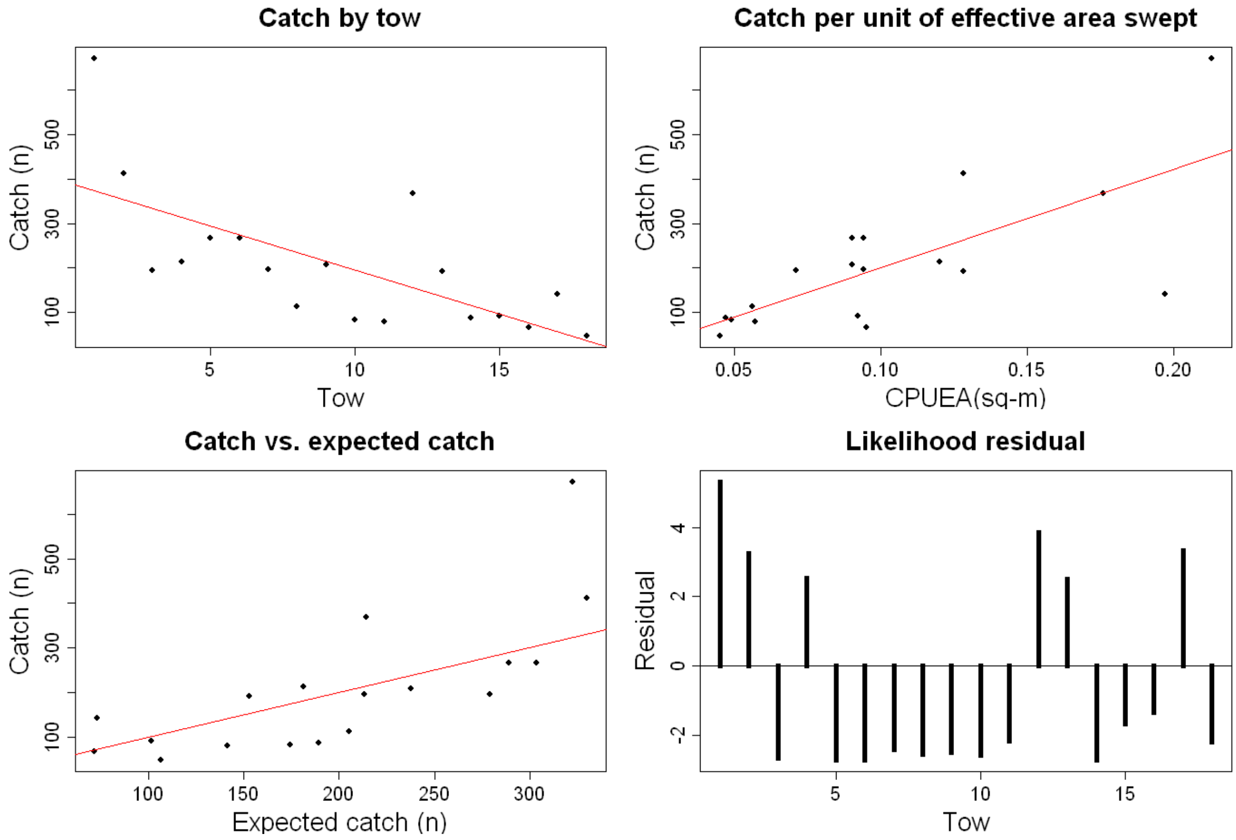


Figure A46. Patch model diagnostics for SC11-03.

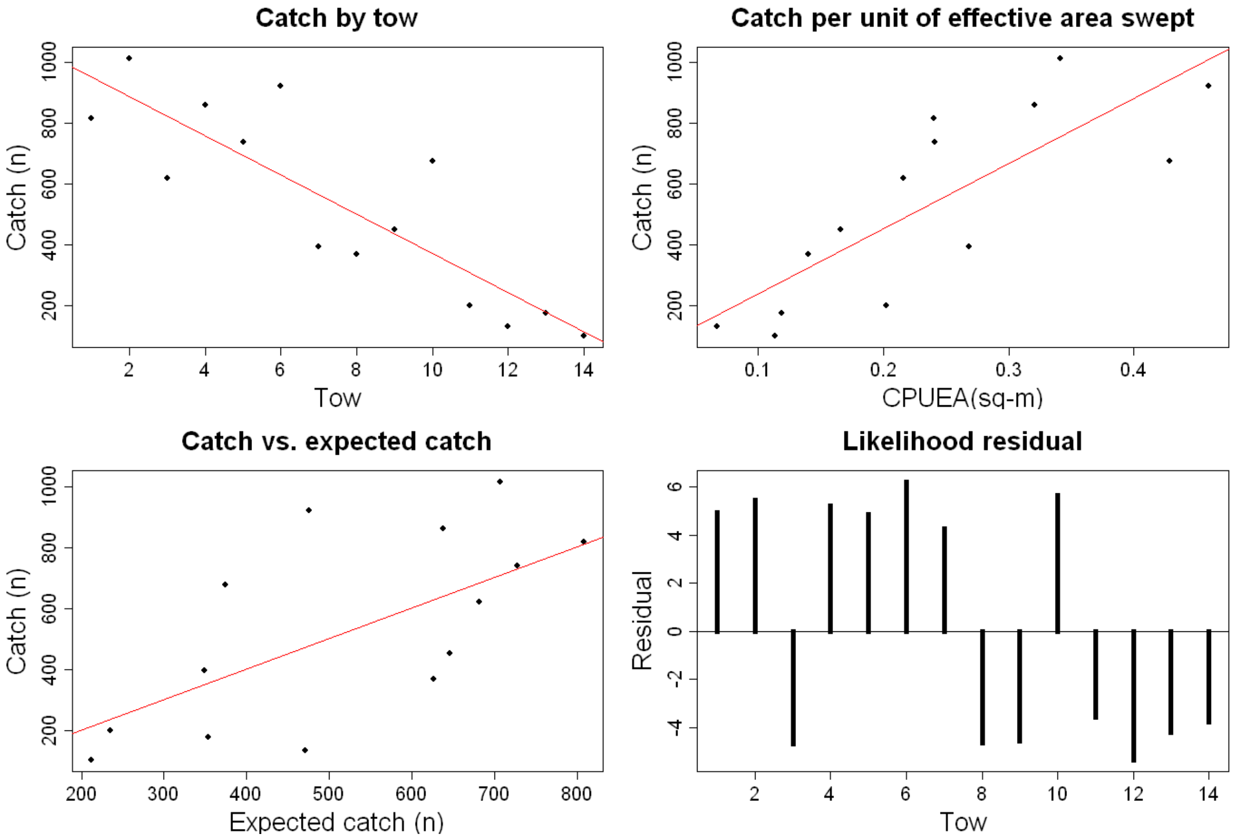


Figure A47. Likelihood profiles for SC11-02. The red lines are the estimates and delta method approximate 95% confidence intervals.

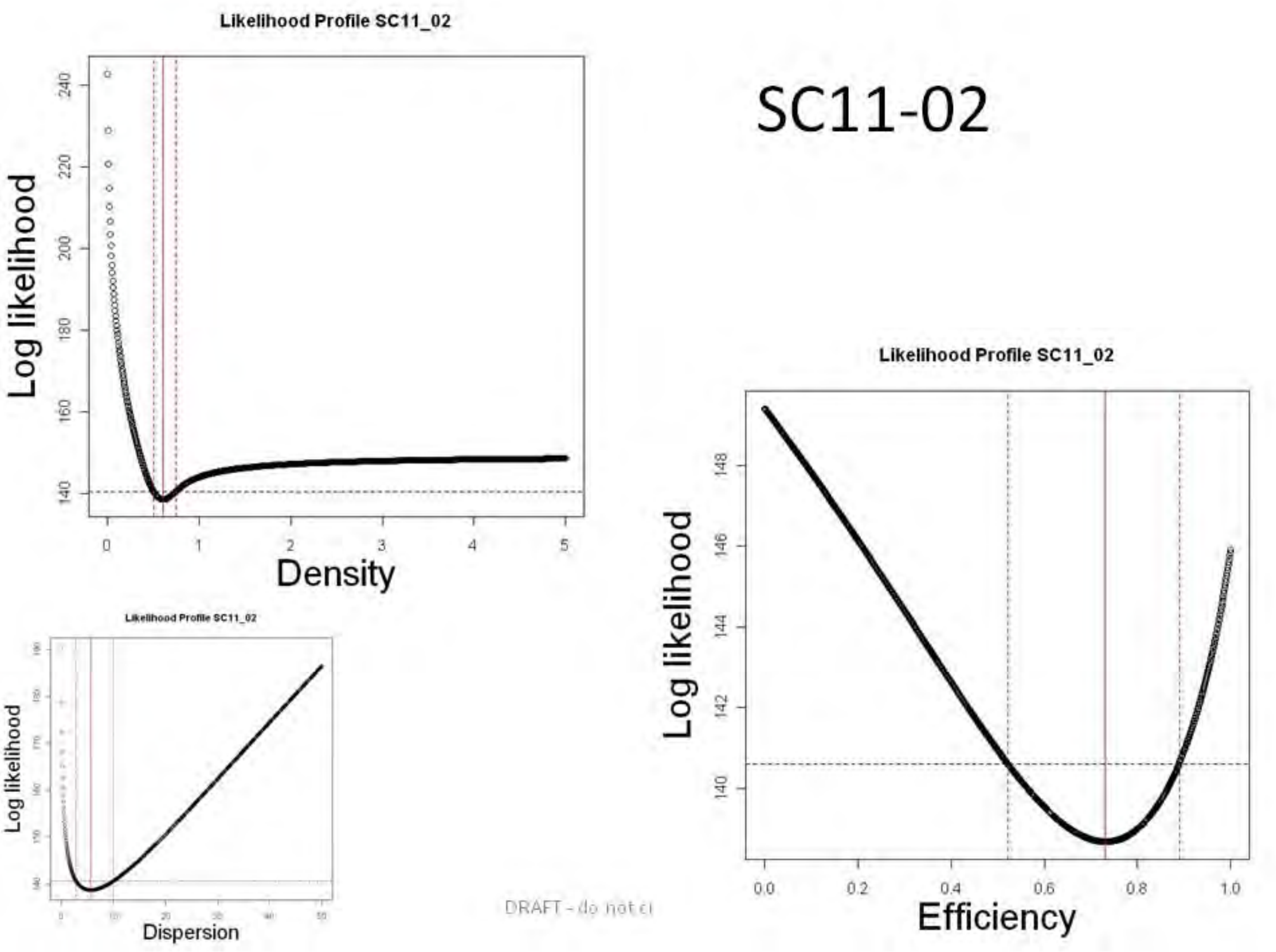


Figure A48. Likelihood profiles for SC11-02S.

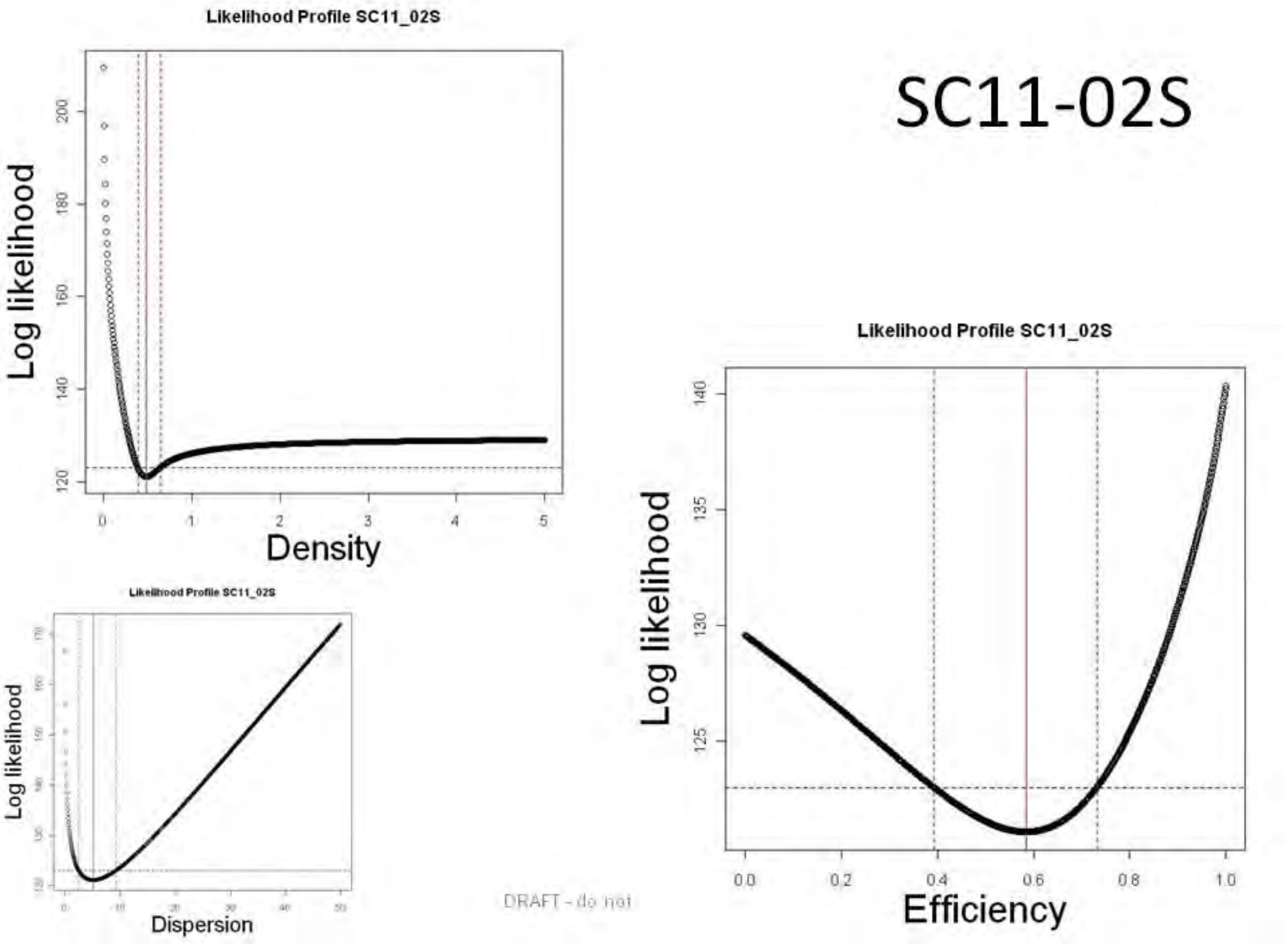
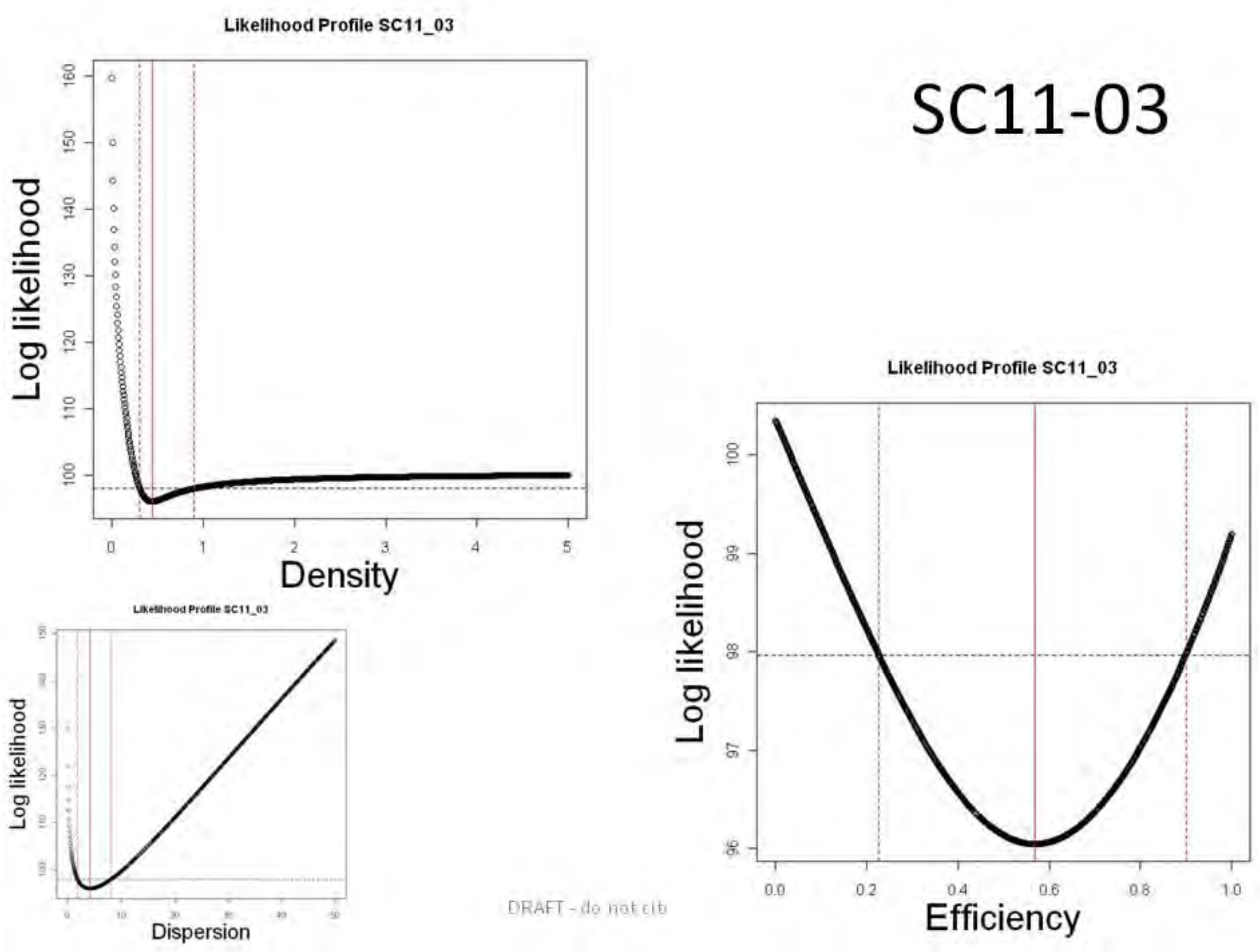


Figure A49. Likelihood profiles for SC11-03.



SC11-03

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Figure A50. Surfclam shell height composition data used to estimate selectivity of the NEFSC survey clam dredge. Summarized here using 1 cm bins.

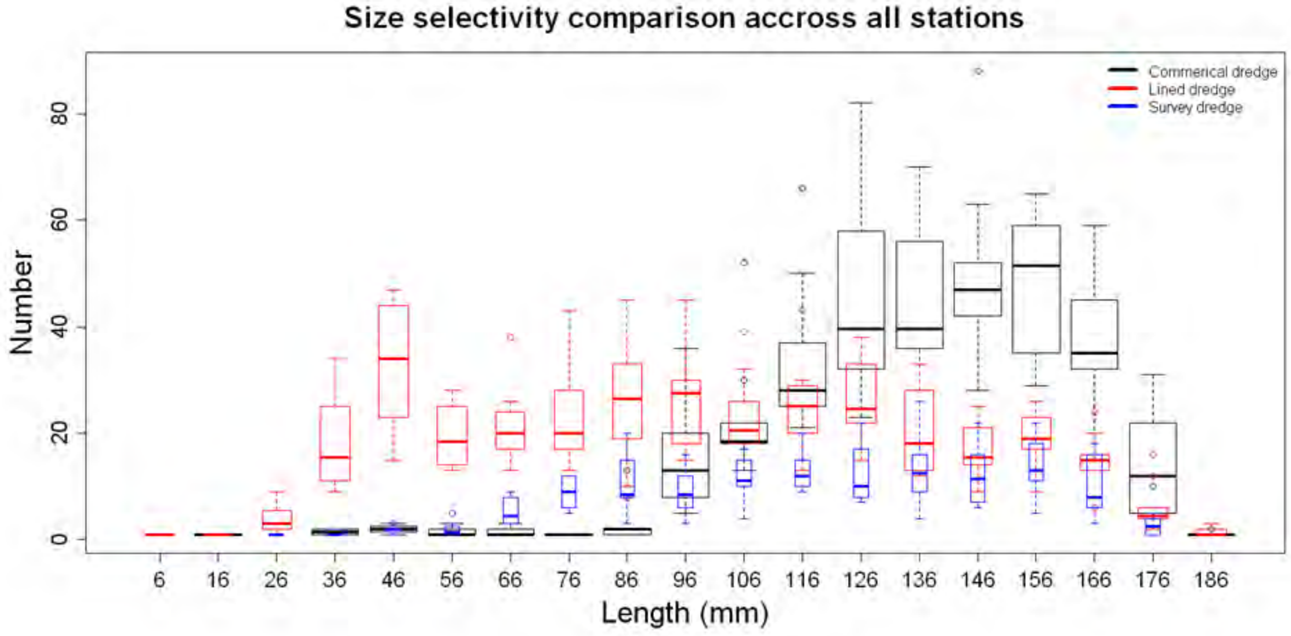


Figure A51. Swept area comparison at each station in survey selectivity experiments in 2008 and 2011.

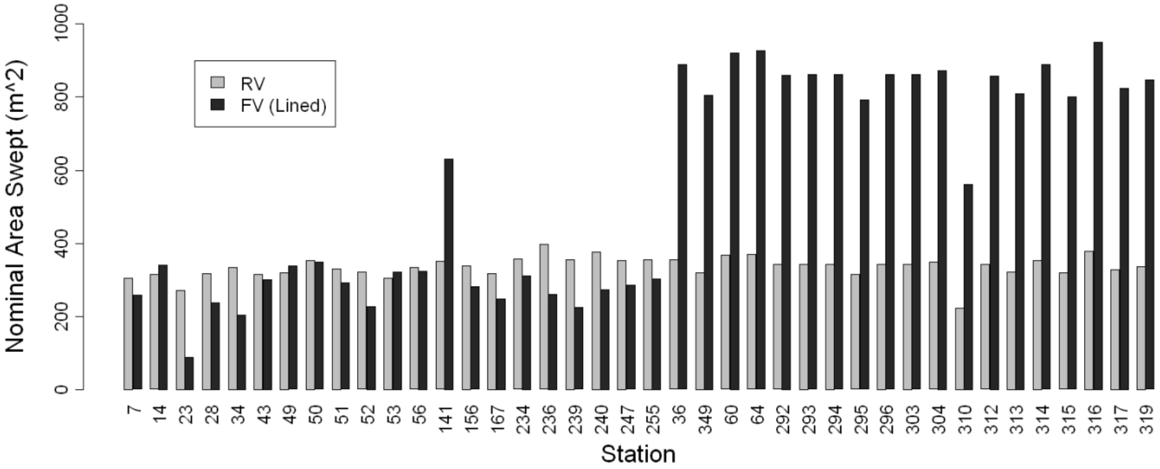


Figure A52. GAM model fit to selectivity data. The dots are the residuals, the gray band is the +/- 2 standard error confidence interval, and the rug plot above the x axis indicate data density (weights). Much of the variance shown is eliminated in modeling by the offset term which adjust for differences in area swept and the overall proportion of samples in the test gear.

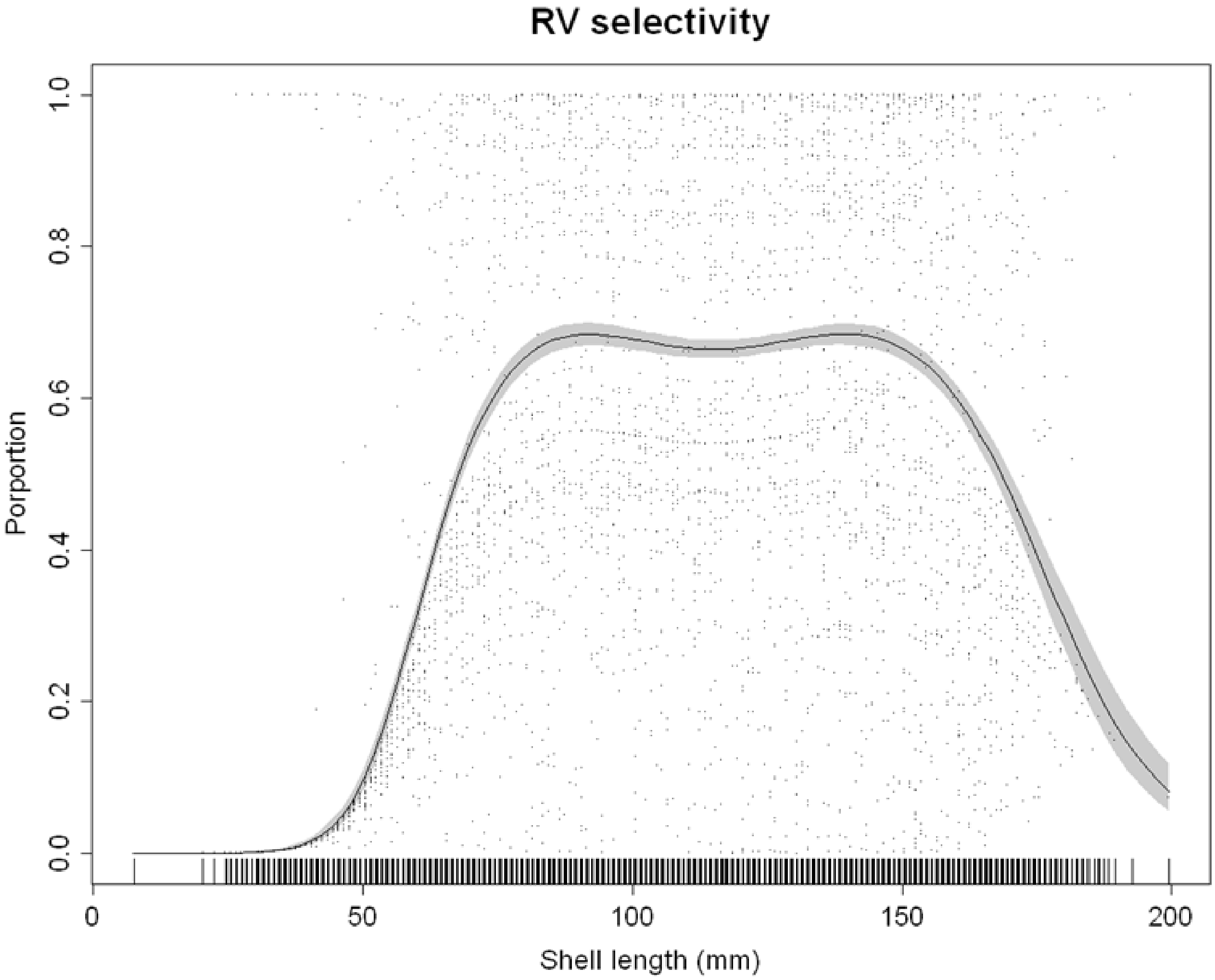


Figure A53. GAM fit at each station. This plot demonstrates that the domed shape is pervasive and not driven data from one or a few stations.

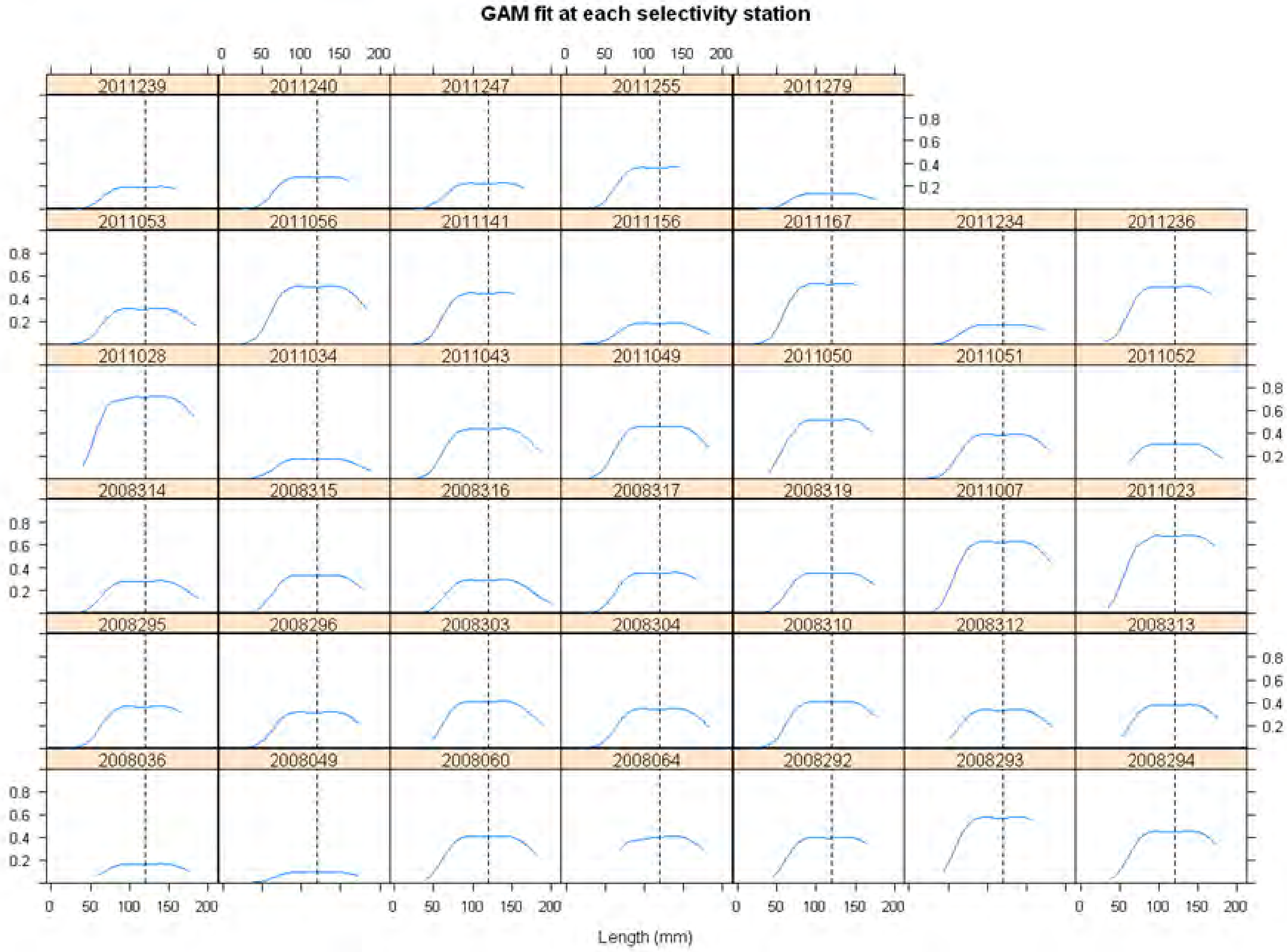


Figure A54. Rescaled selectivity fits for both survey and commercial dredges with +/- 2 standard errors.

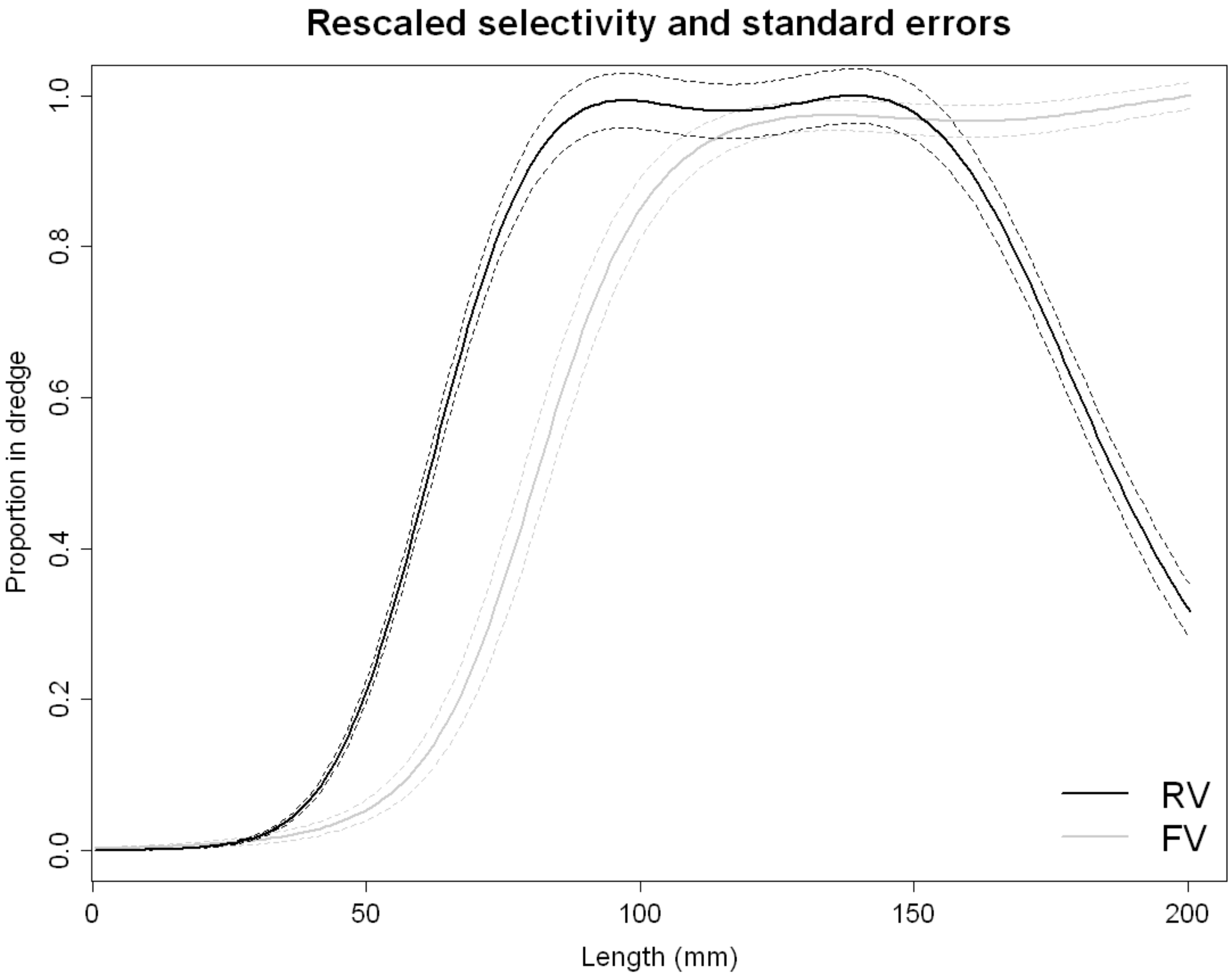


Figure A55. Swept area comparison at each station in commercial selectivity experiments in 2008 and 2011. Tow length for commercial station 314 is not available and station 314 was not used.

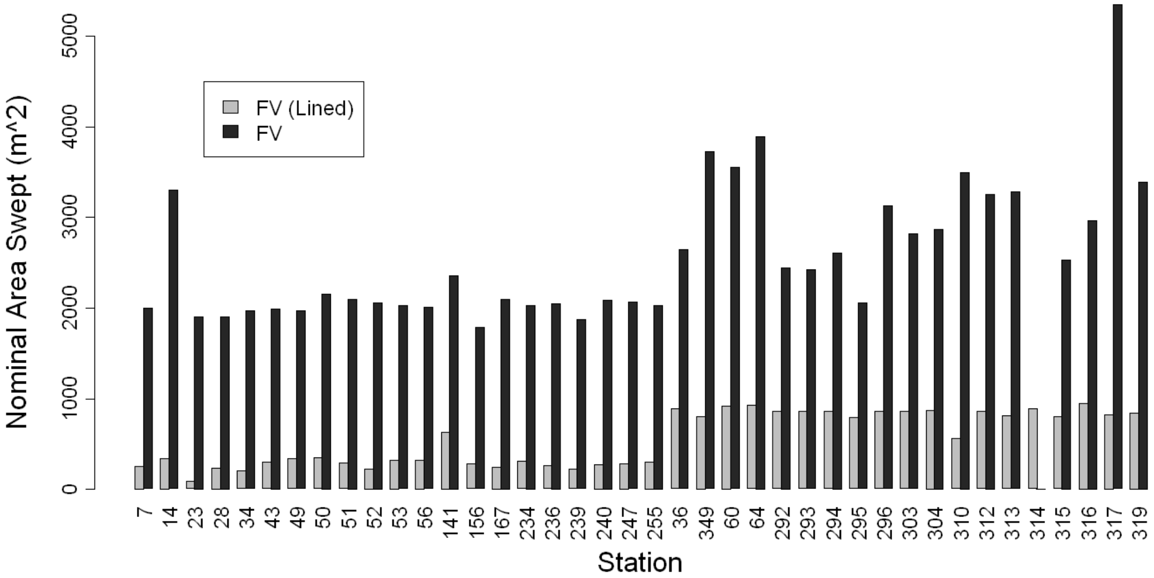


Figure A56. Length to meat weight curves from the last assessment and the current analysis. Both are based on general data, without regional or year effects. The average depth over all stations (33 m) was used to generate the curve for the current assessment in this figure.

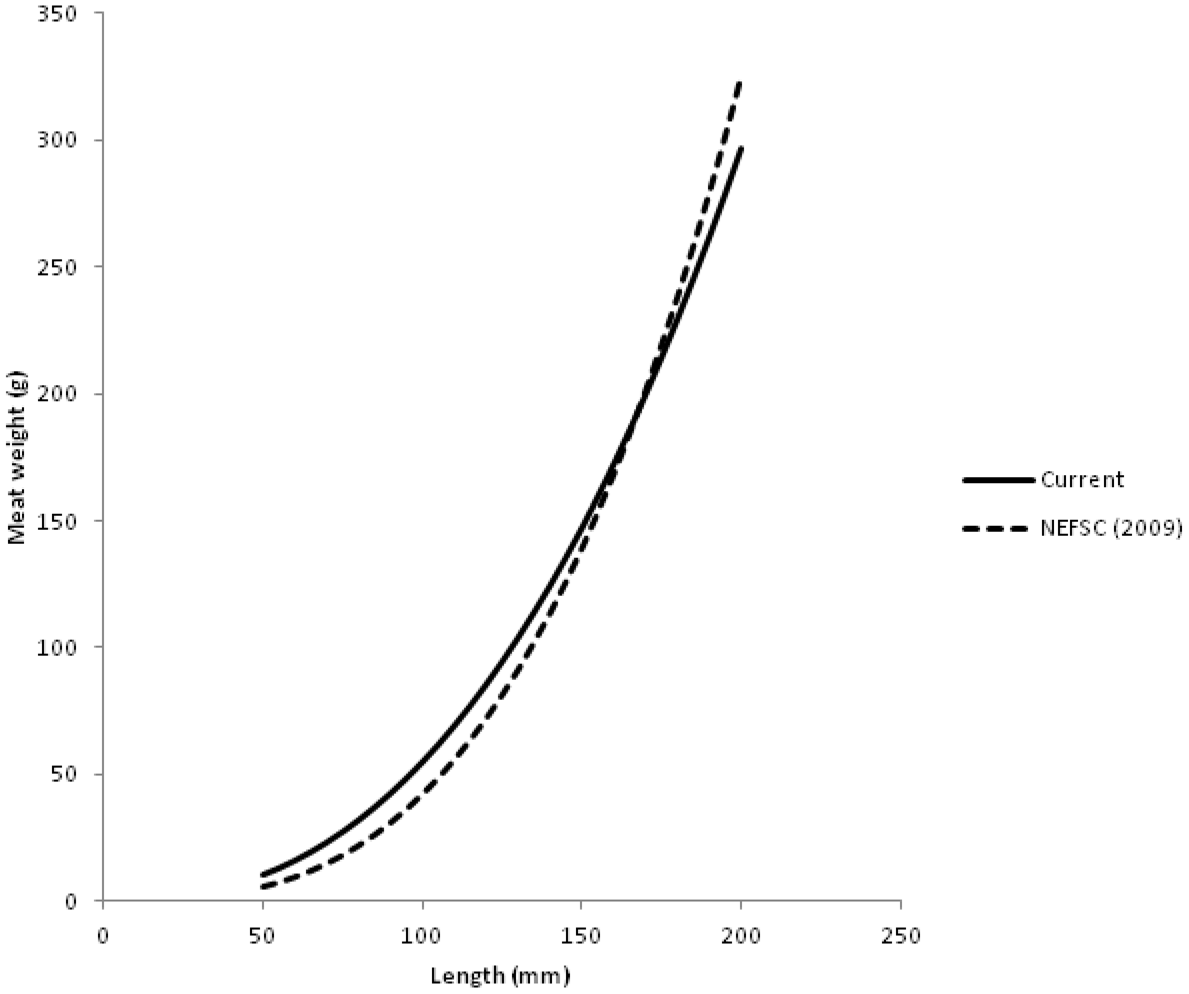
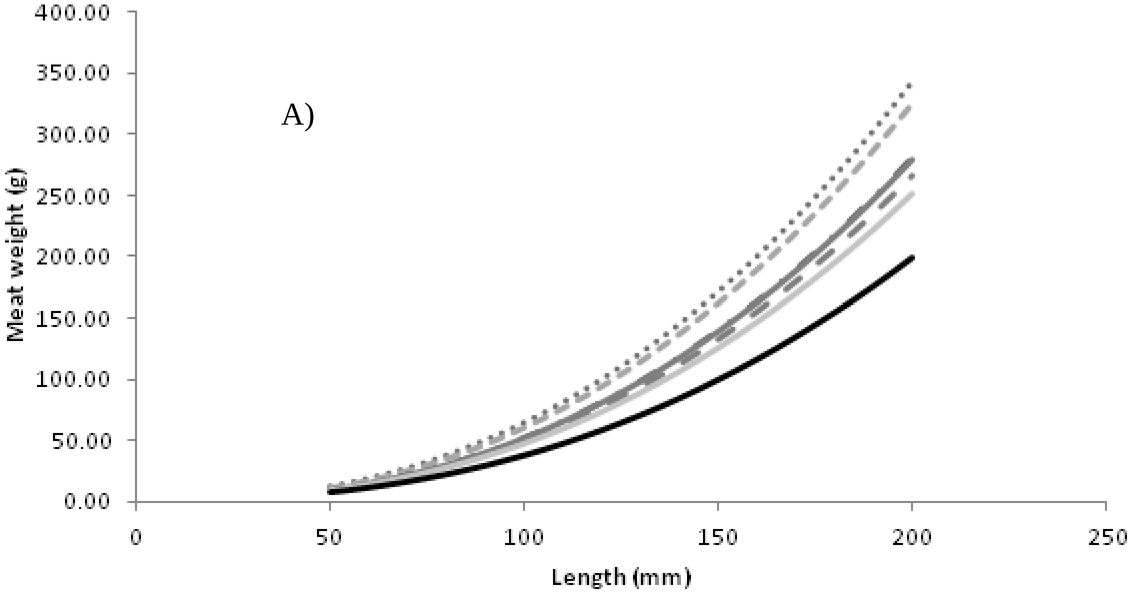


Figure A57. Regional differences in allometric relationships for surfclam. The same depth (33 m) was used to generate the curves for each region in A) and regional median depth was used to generate the curves in B).



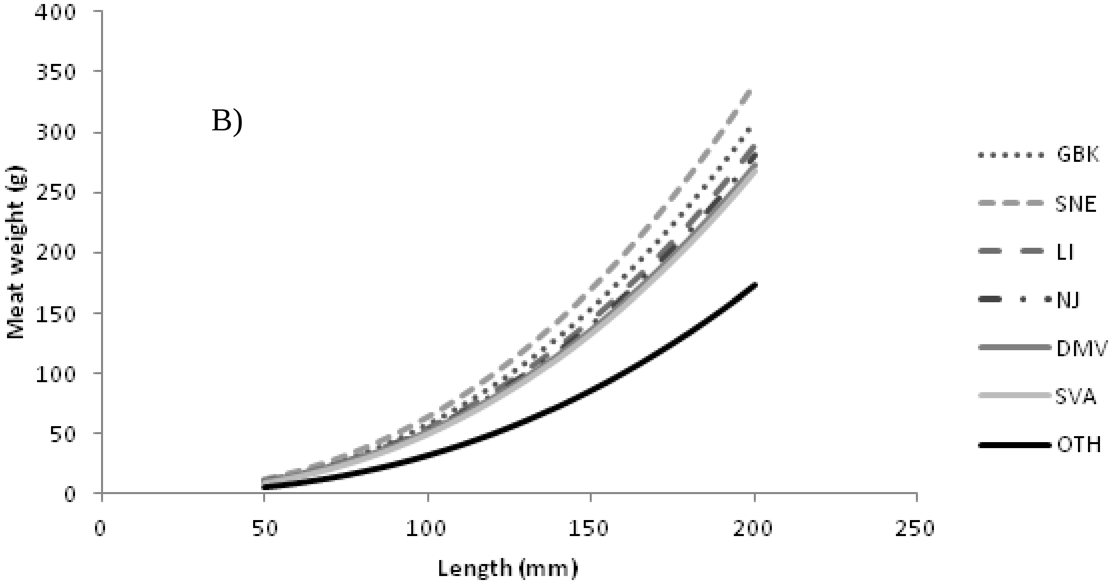


Figure A58. Age vs. length with fitted Von Bertalanffy growth curve for the DMV region in each survey year.

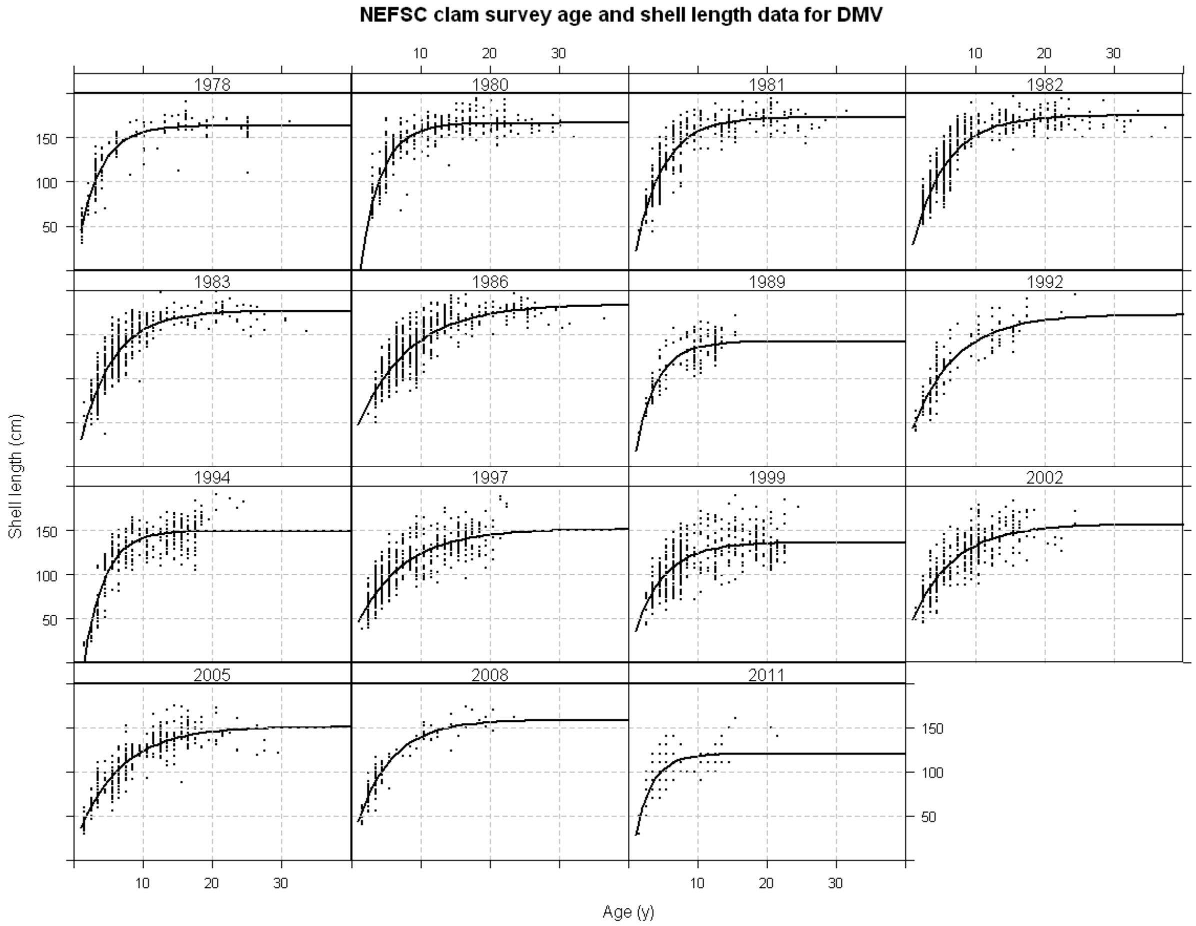


Figure A59. Age vs. length with fitted Von Bertalanffy growth curve for the NJ region in each survey year.

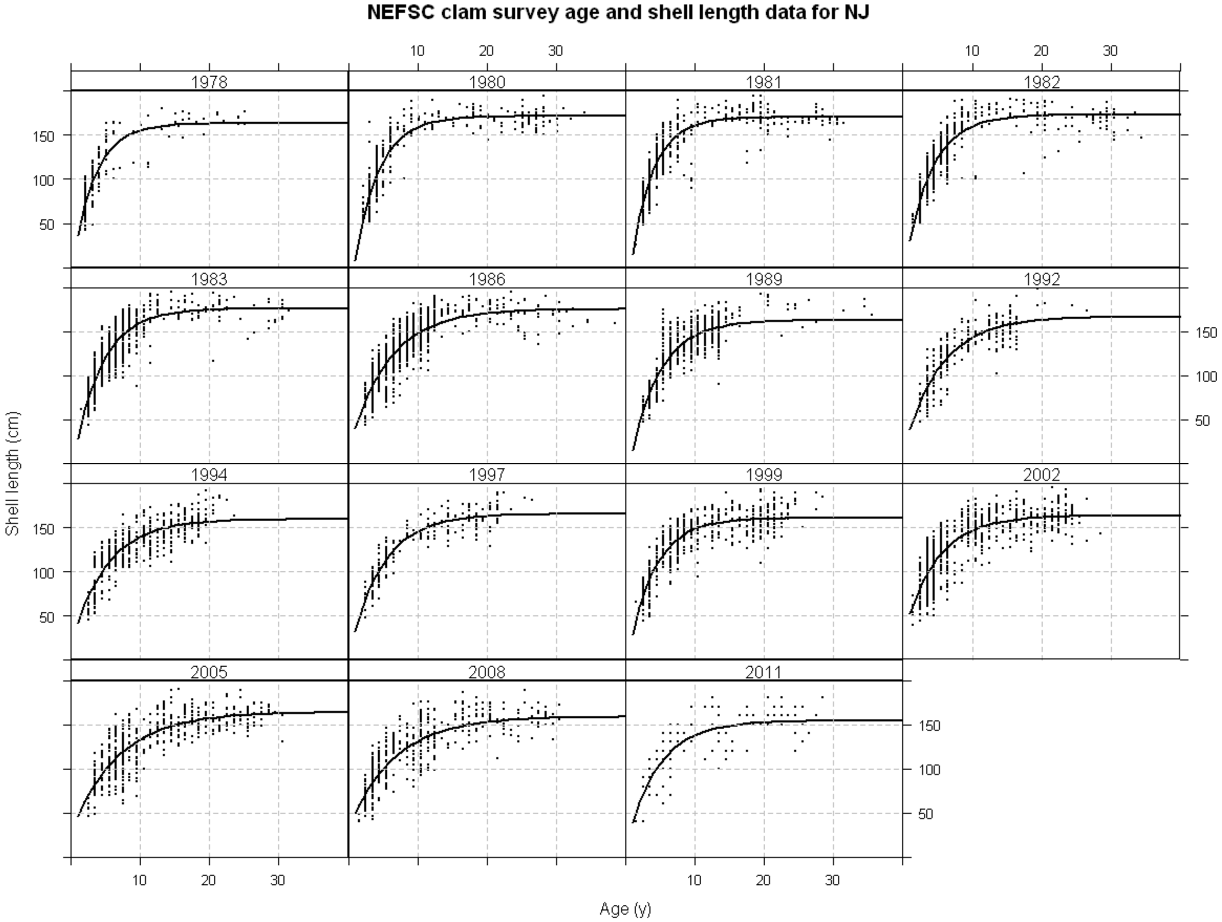


Figure A60. Age vs. length with fitted Von Bertalanffy growth curve for the LI region in each survey year.

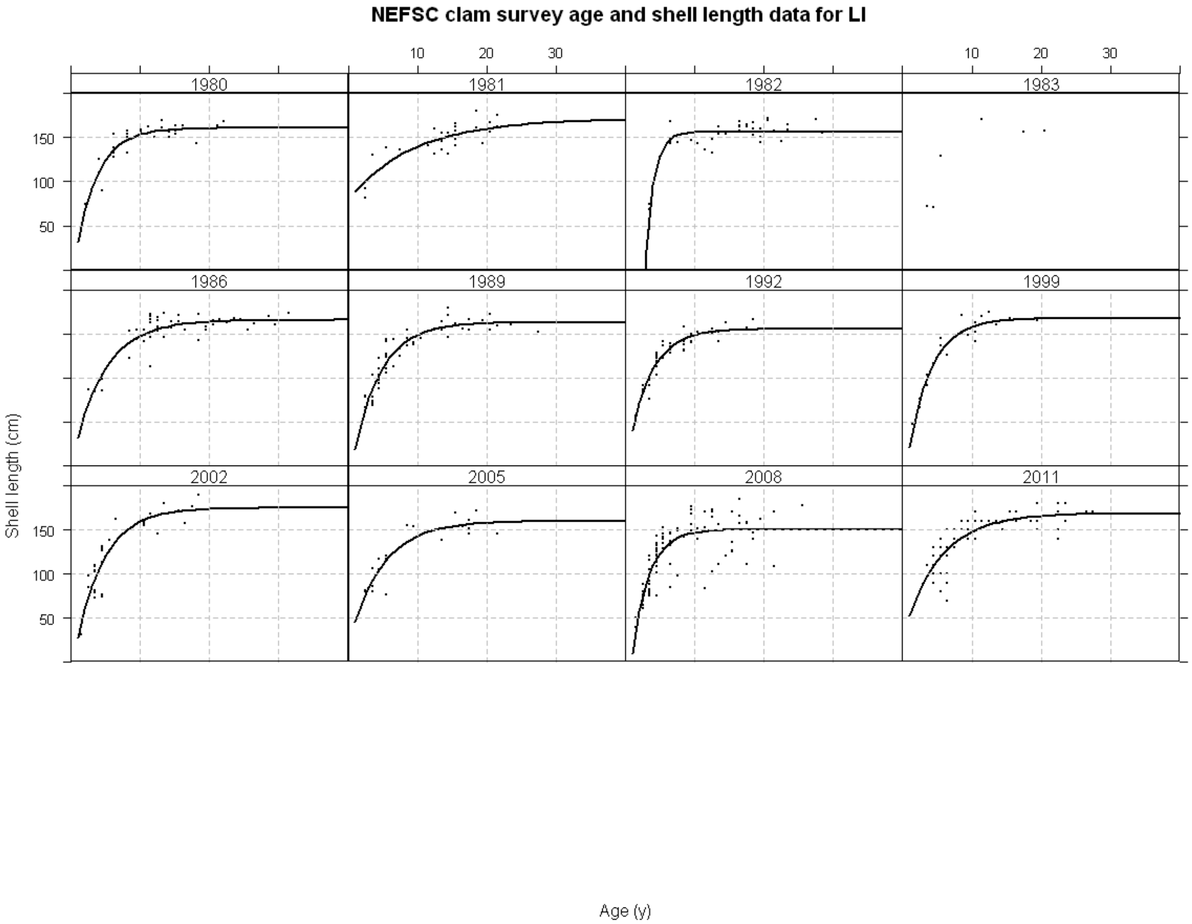


Figure A61. Age vs. length with fitted Von Bertalanffy growth curve for the SNE region in each survey year.

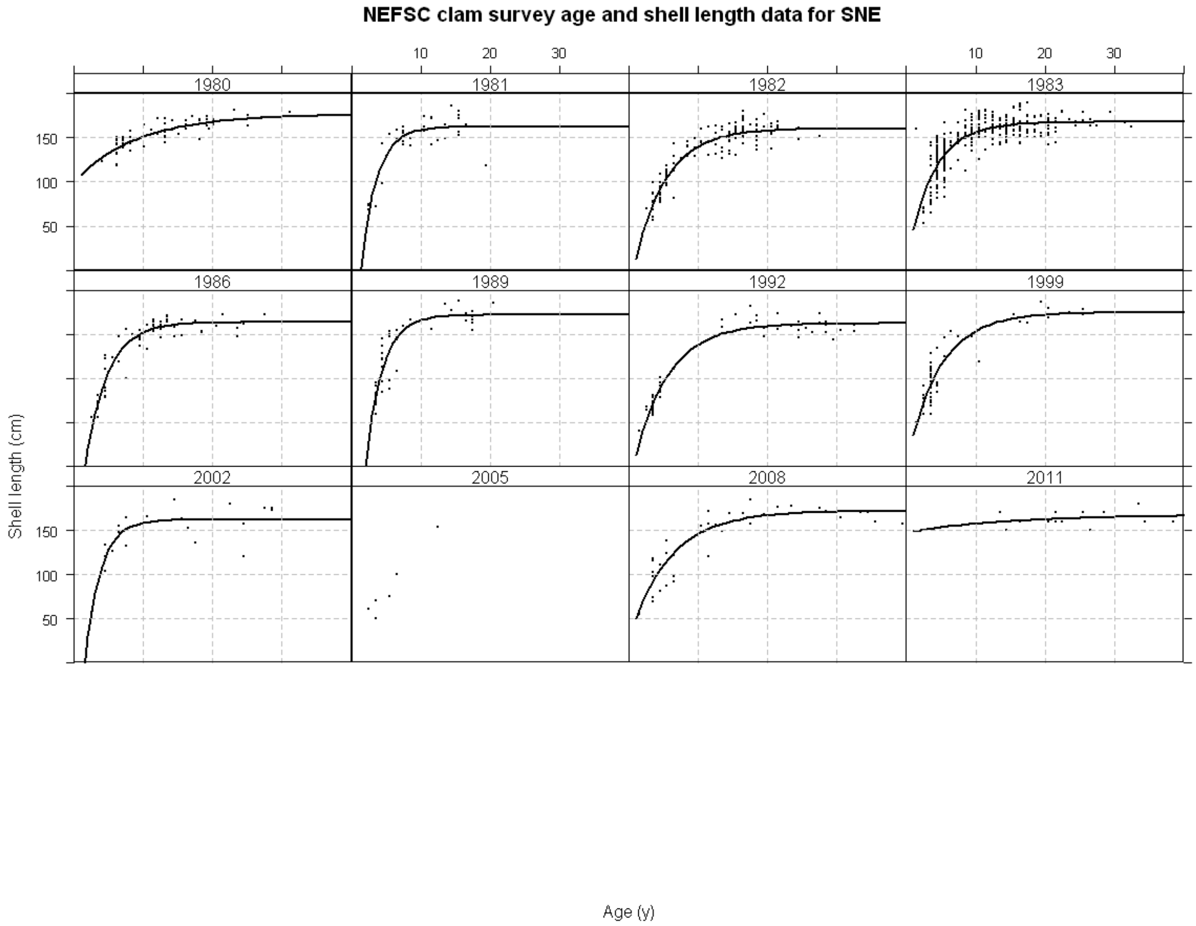


Figure A62. Age vs. length with fitted Von Bertalanffy growth curve for the GBK region in each survey year.

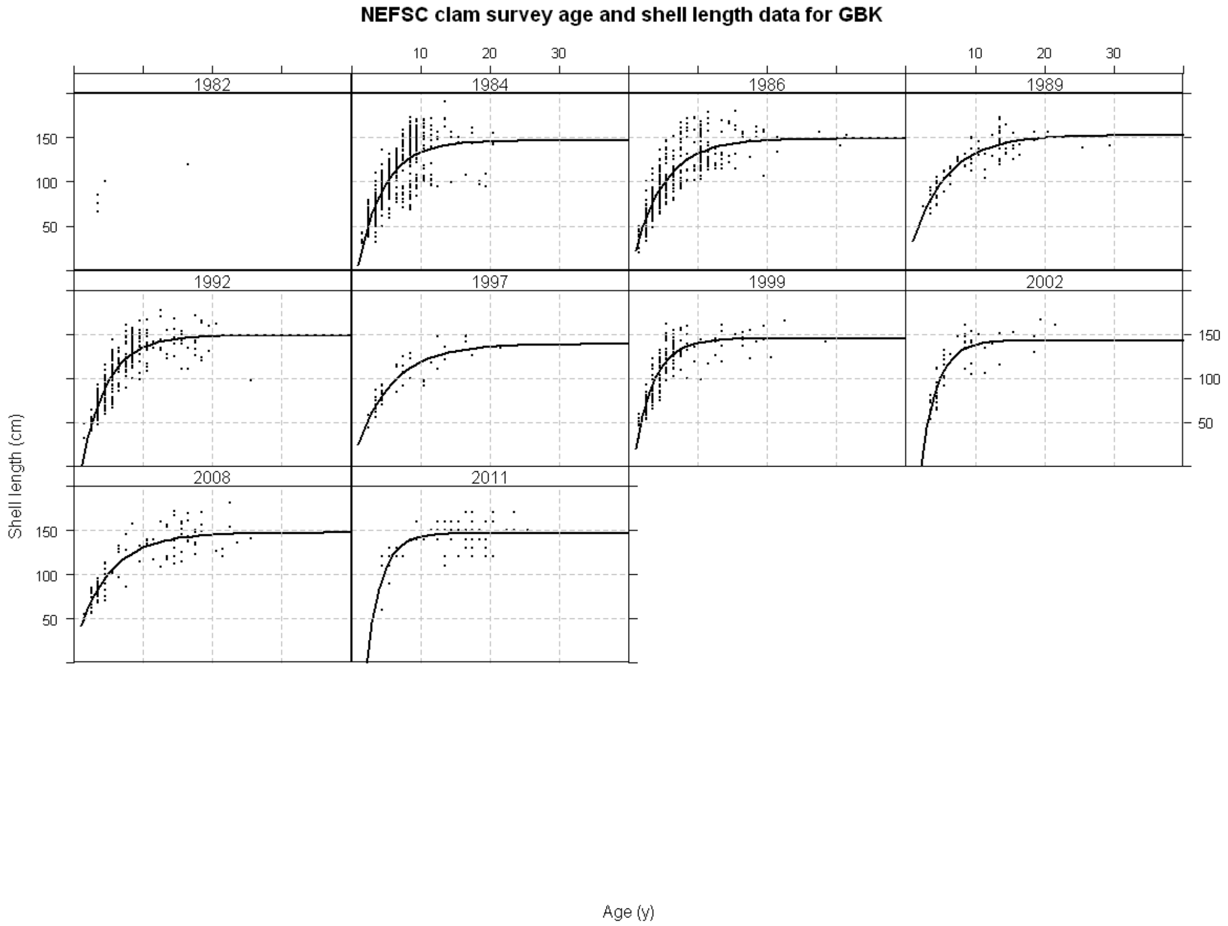


Figure A63. Weighted regression of estimated L_∞ in DMV over time.

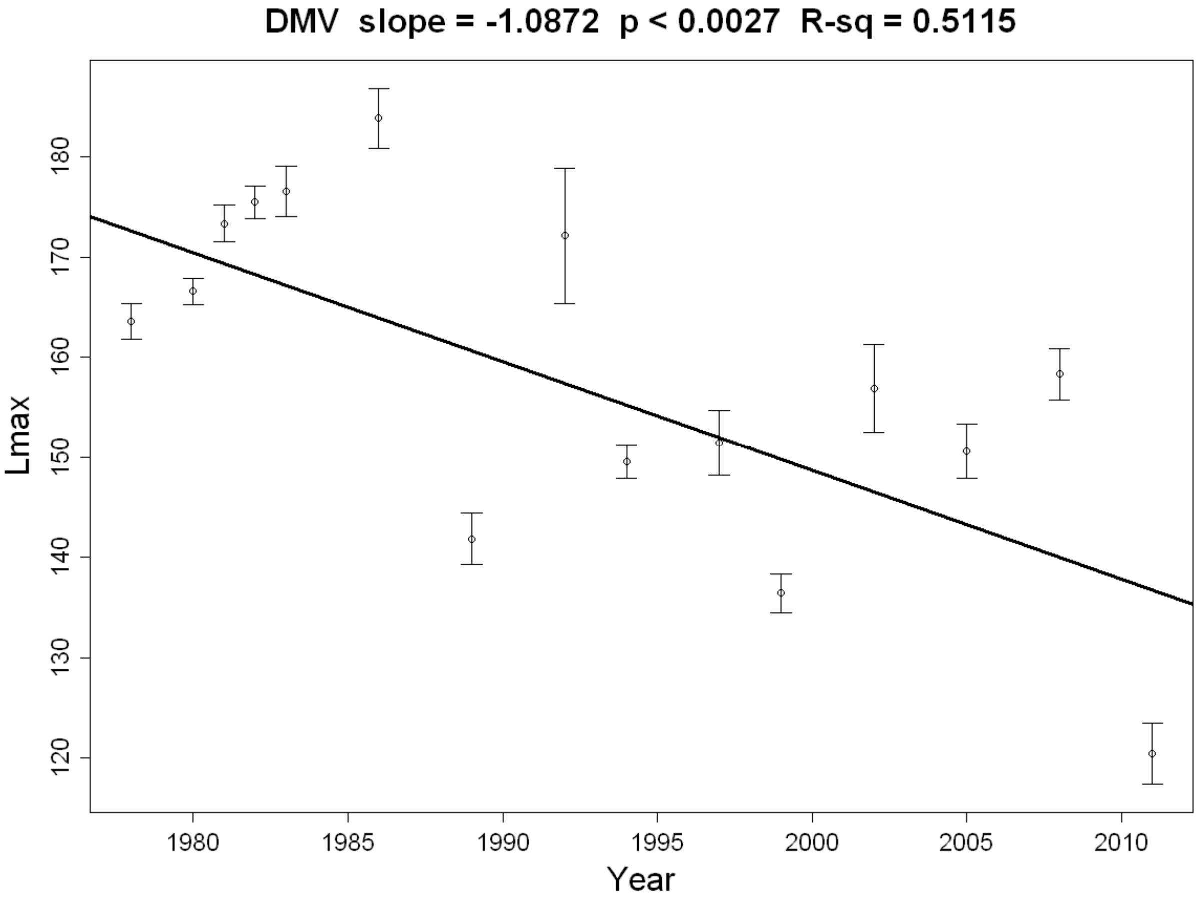


Figure A64. Weighted regression of L_∞ estimated in NJ over time.

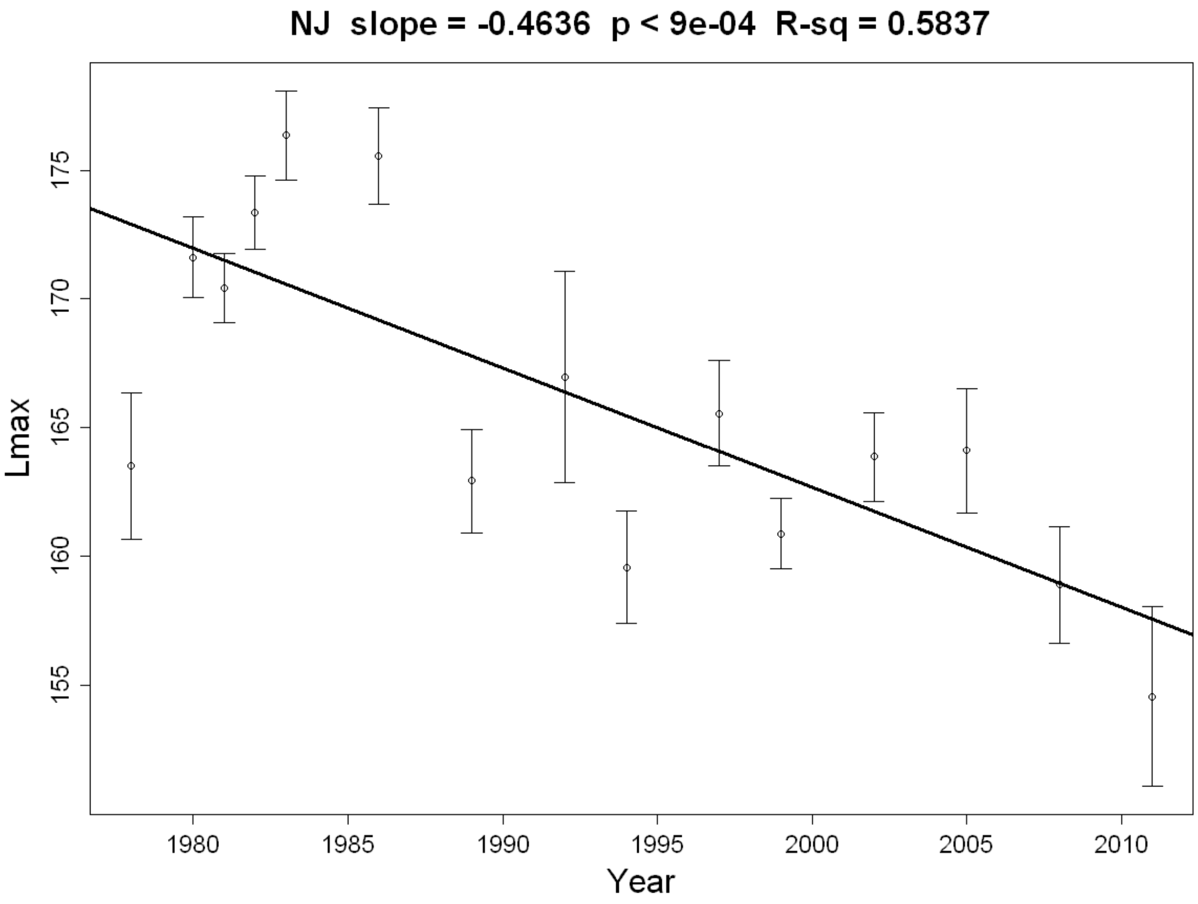


Figure A65. Weighted regression of *K* estimated in NJ over time.

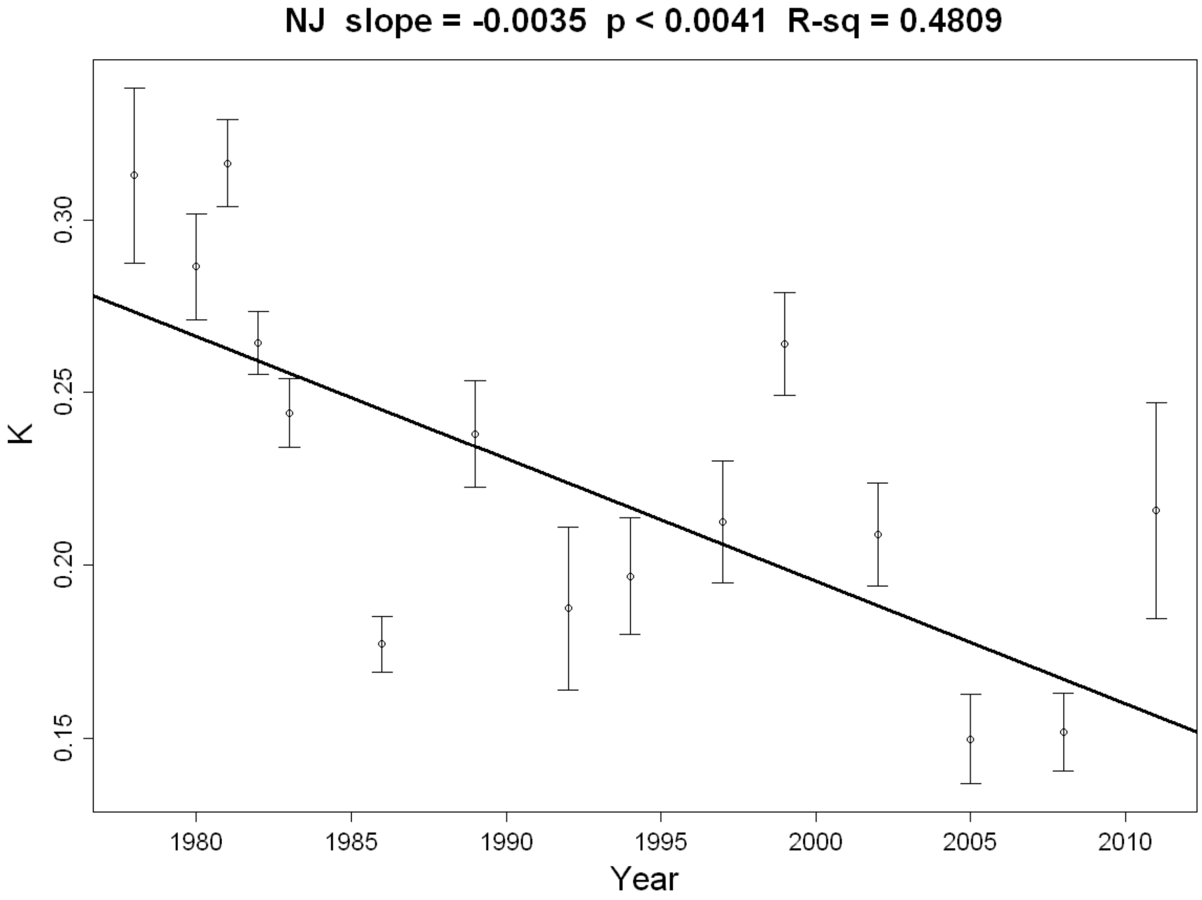


Figure A66. The proposed stock division. The northern area is GBK and the southern area is the remaining portion of the surfclam range in the US EEZ.

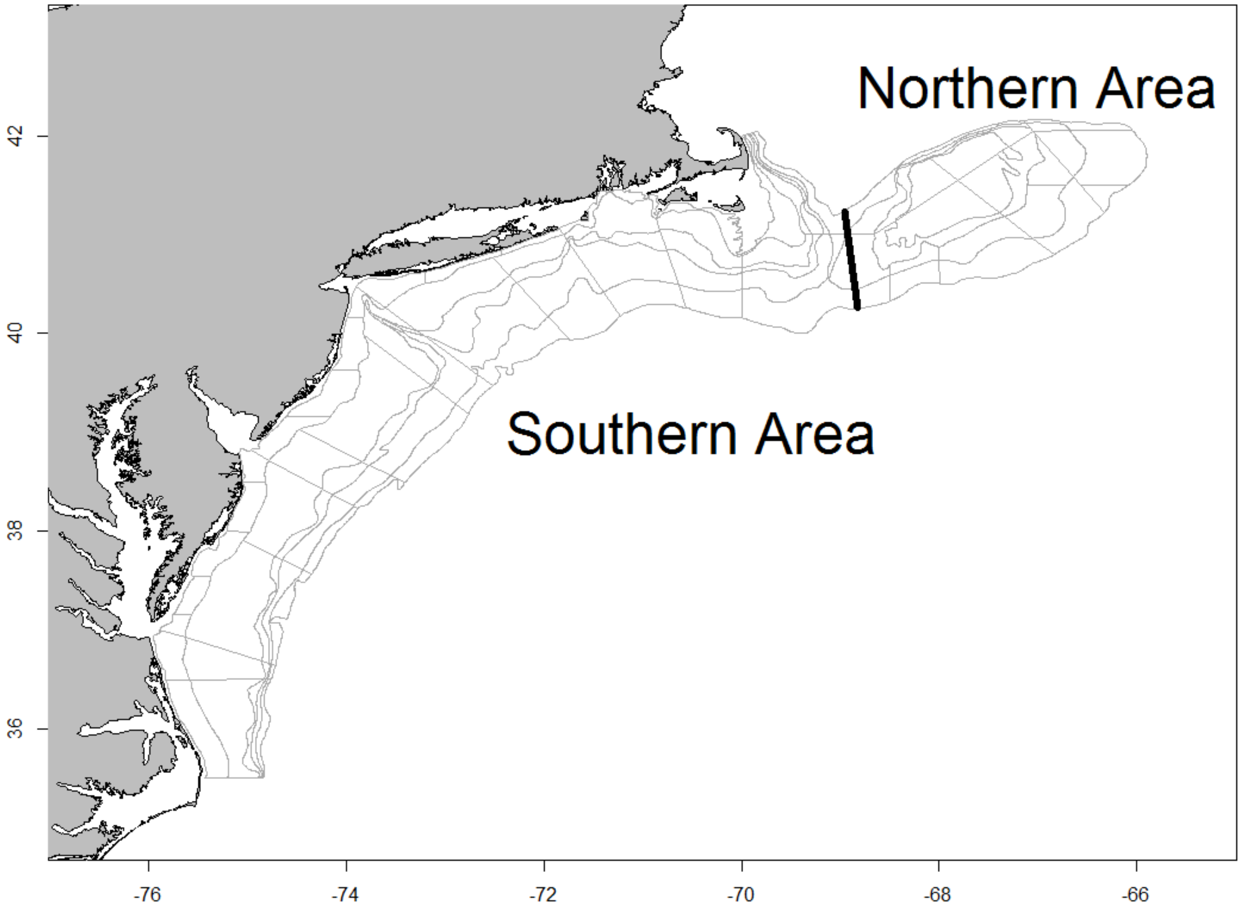


Figure A67. Survey age composition data for survey years and regions with at least 100 age samples. The first column, for example, shows the age composition of survey data for Georges Bank (GBK) in the north and New Jersey (NJ) and Delmarva (DMV) in the south during 1982.

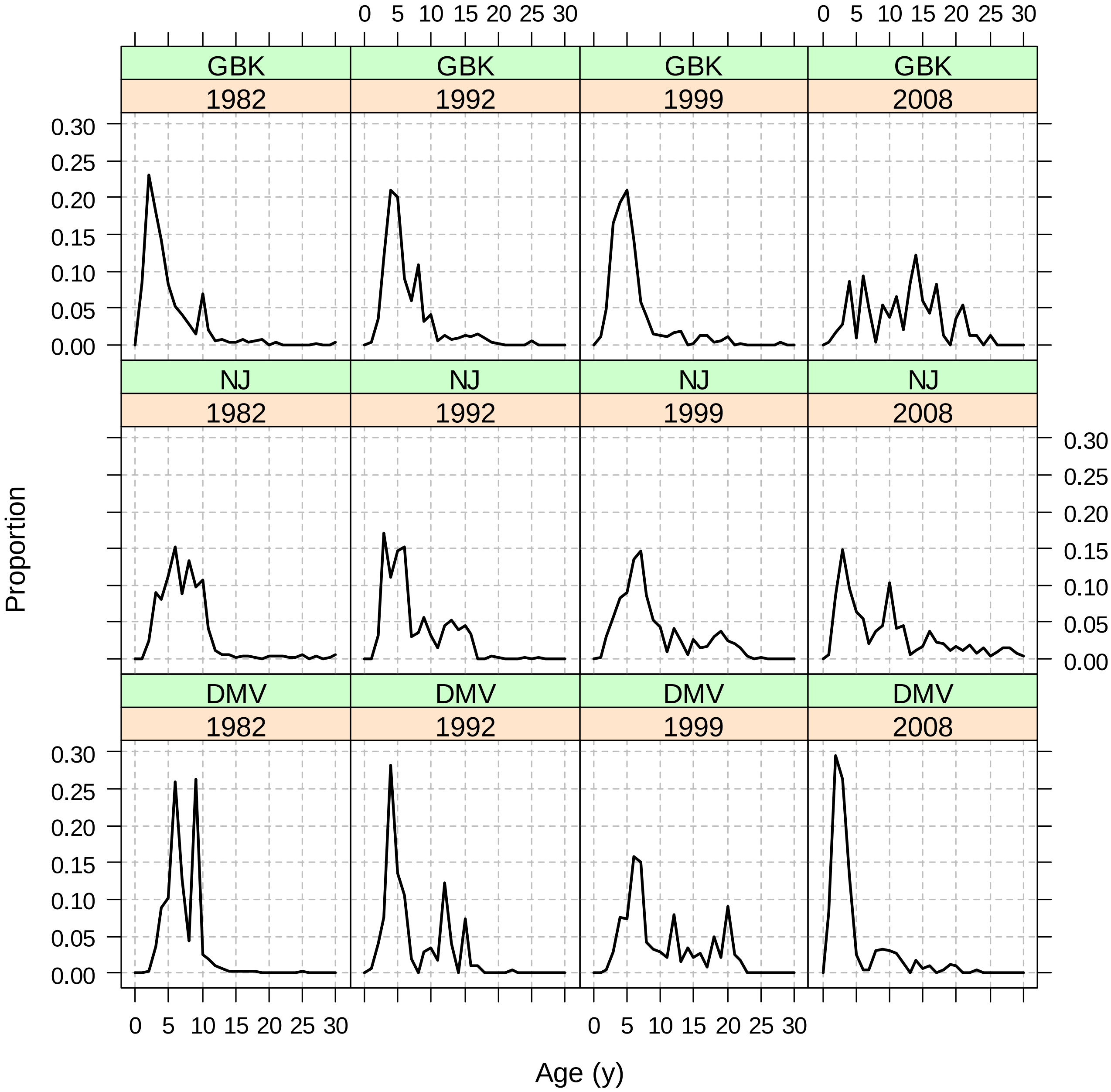


Figure A68. Data and availability by year in the SS3 model for surfclams in the southern area.

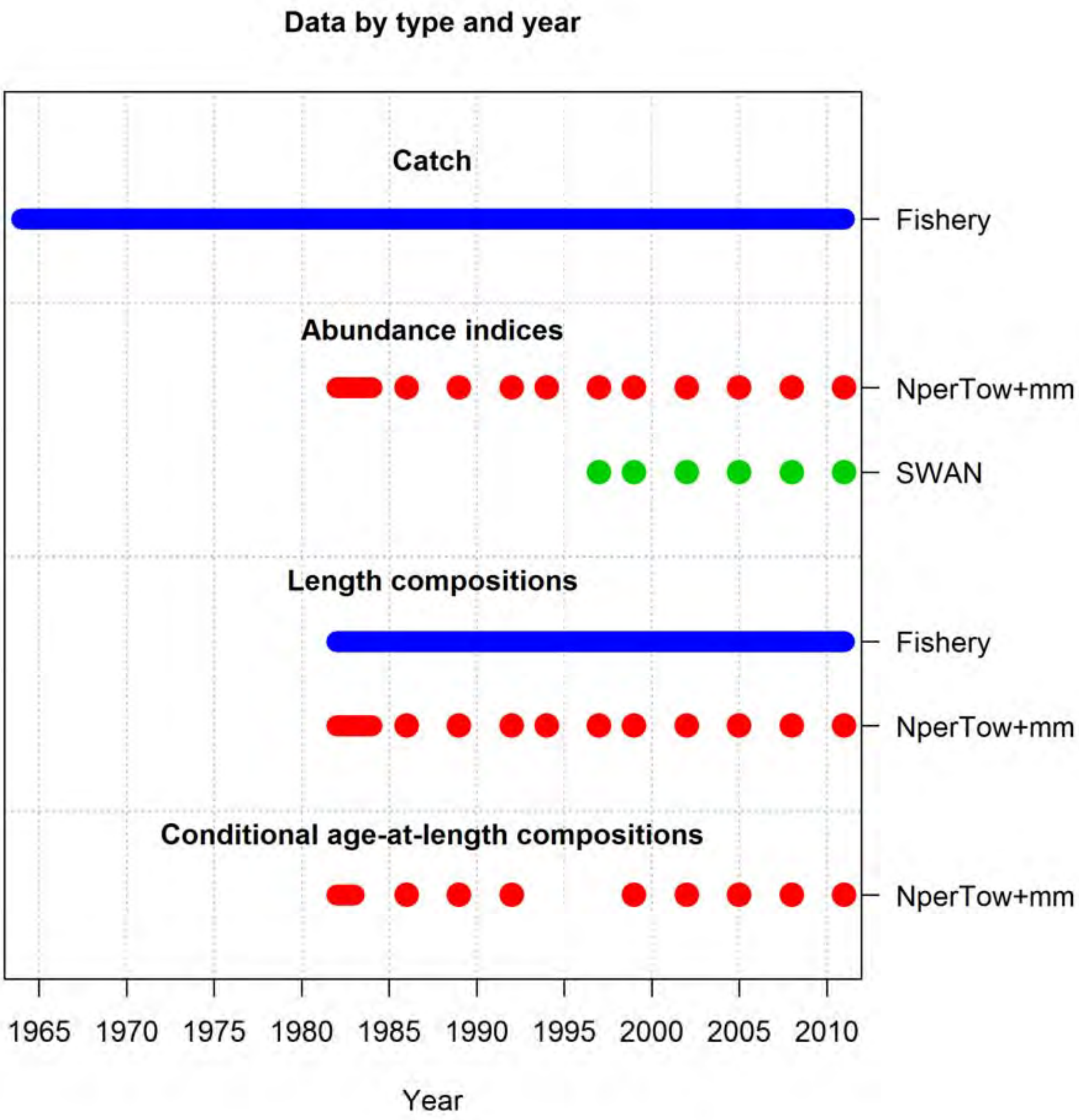


Figure A69. Data and availability by year in the SS3 model for surfclams in the GBK area.

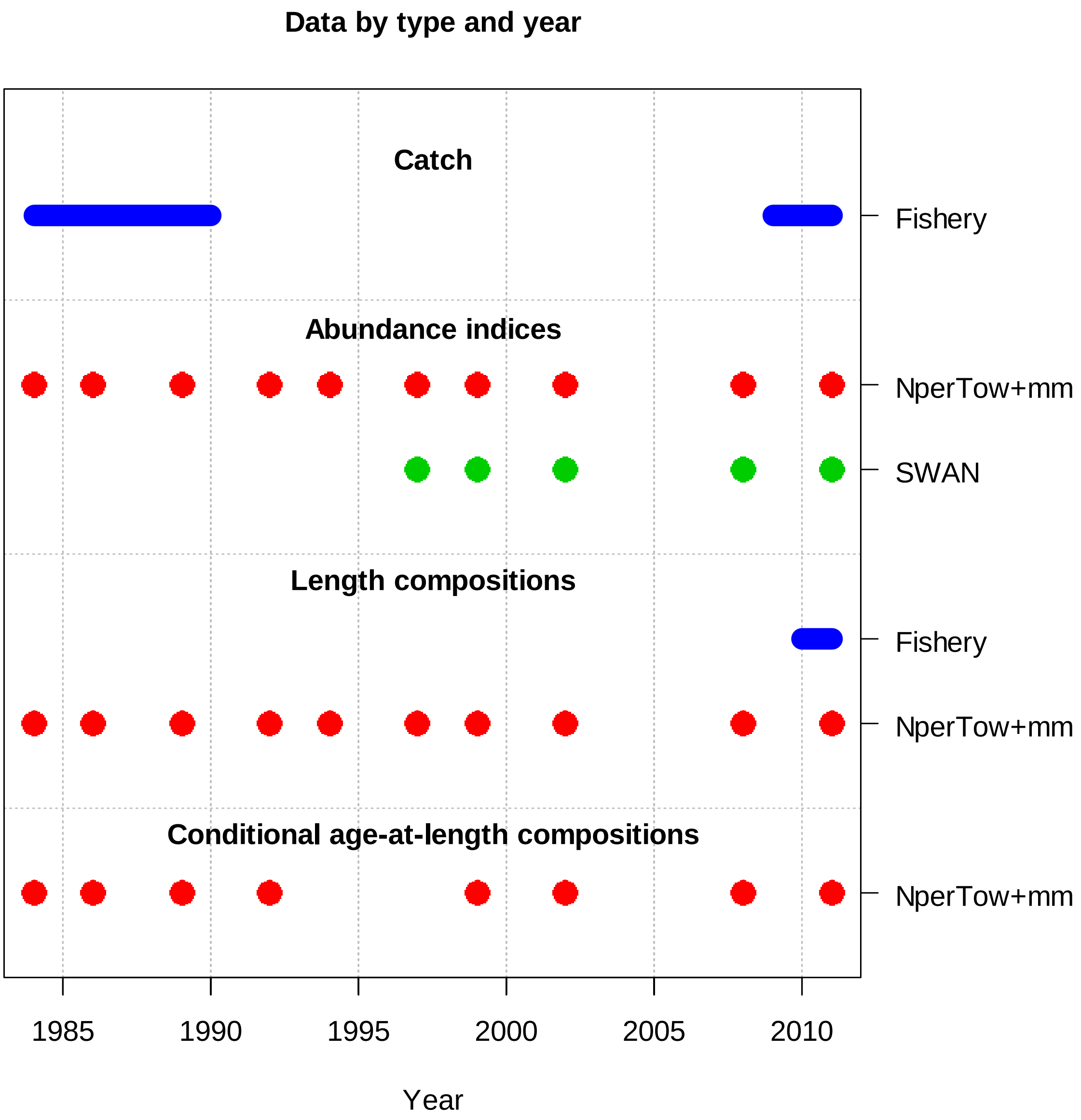


Figure A70. Results of sensitivity analyses in which growth parameters for surfclams in the southern area were estimated as random walks.

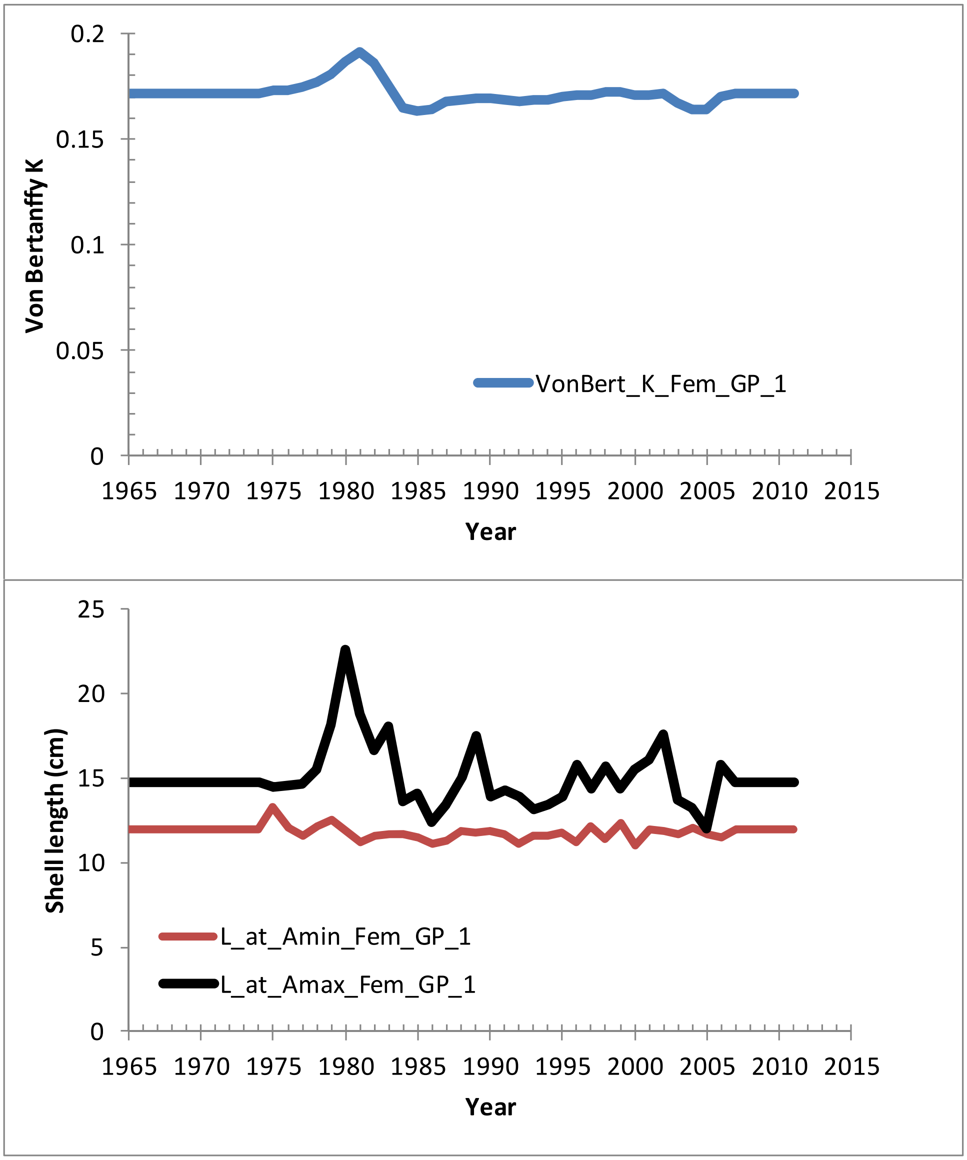


Figure A71. Growth curves estimated in preliminary SS3 model runs for surfclams in the south. The first curve listed in the legend is from external (initial) estimates of all growth parameter values that were fixed in SS3. The rest of the curves listed in the legend from top to bottom gave the best fit (lowest NLL) for the entire model and are listed in order of improving goodness of fit (decreasing NLL). The preferred growth model configuration was "Estimate Lmin and Lmax" (light blue line with open circle). In SS3, with Amin=4, growth at ages 0-4 is approximated by a linear term through zero so that the important of differences on the far left hand side are minimized.

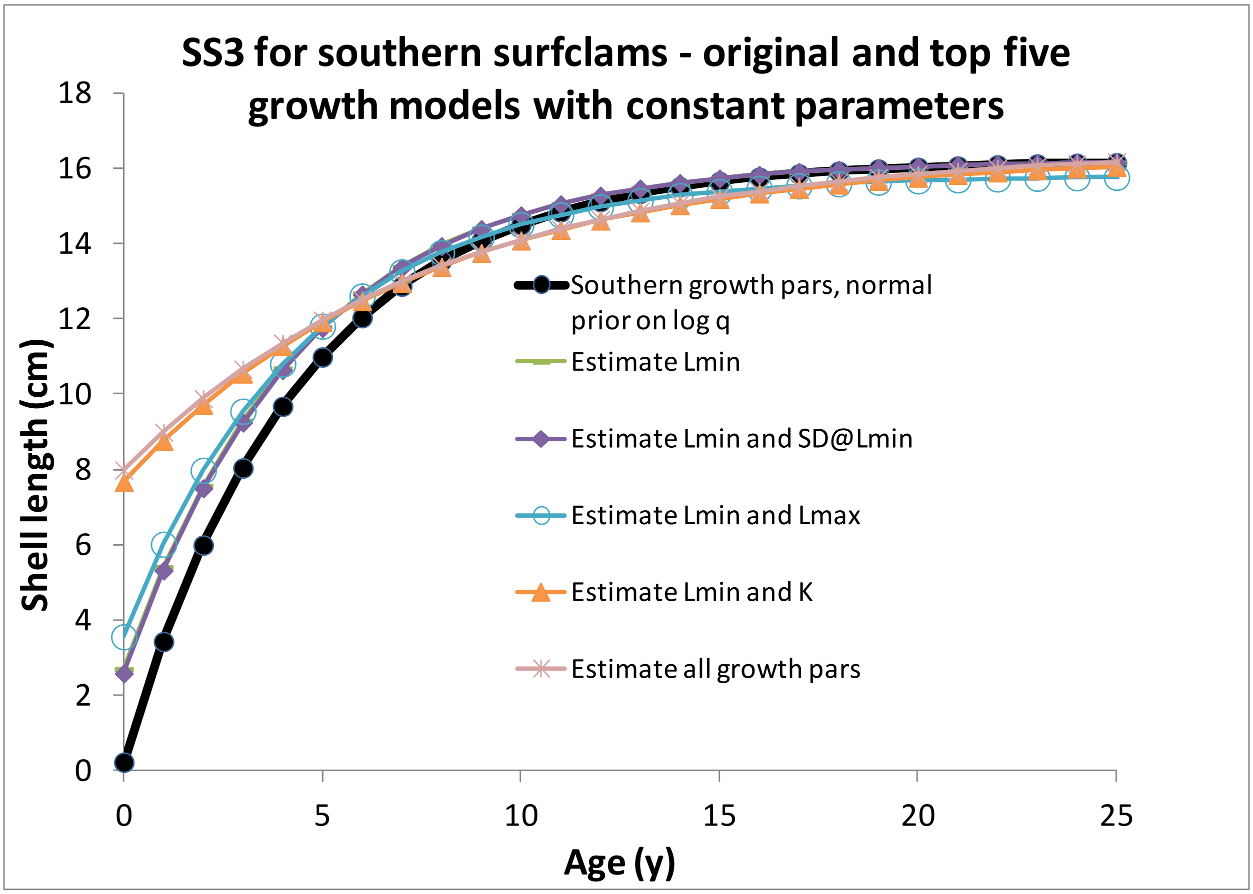


Figure A72. Observed survey data, predicted survey values and biomass estimates from two preliminary SS3 models with likelihood weights for survey trends lambda=1 and lambda=100.

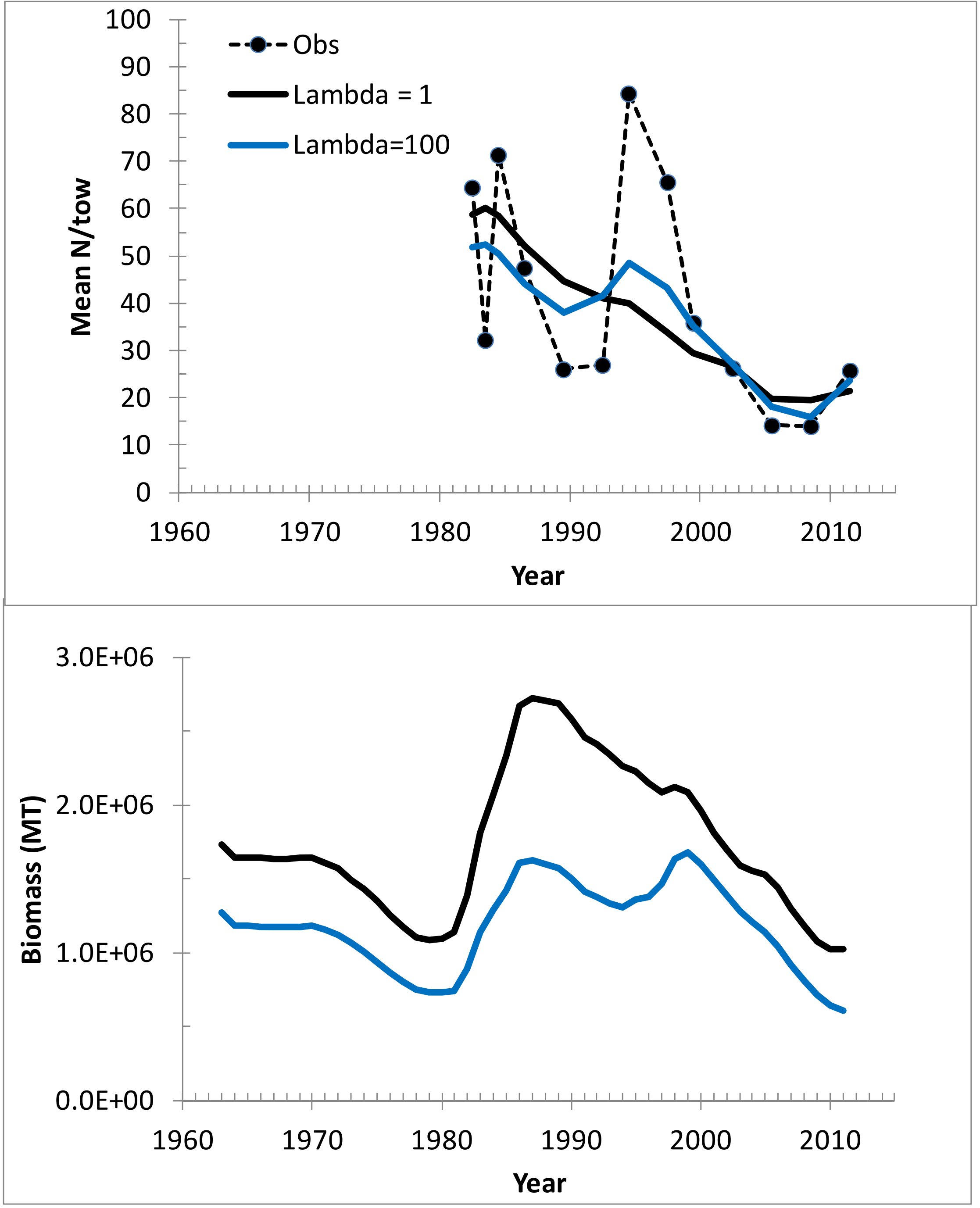


Figure A73. Biomass estimates from sensitivity analyses using a preliminary SS3 model for surfclams in the southern area to address lack of fit to survey size data for 1982, 1983 and 1986.

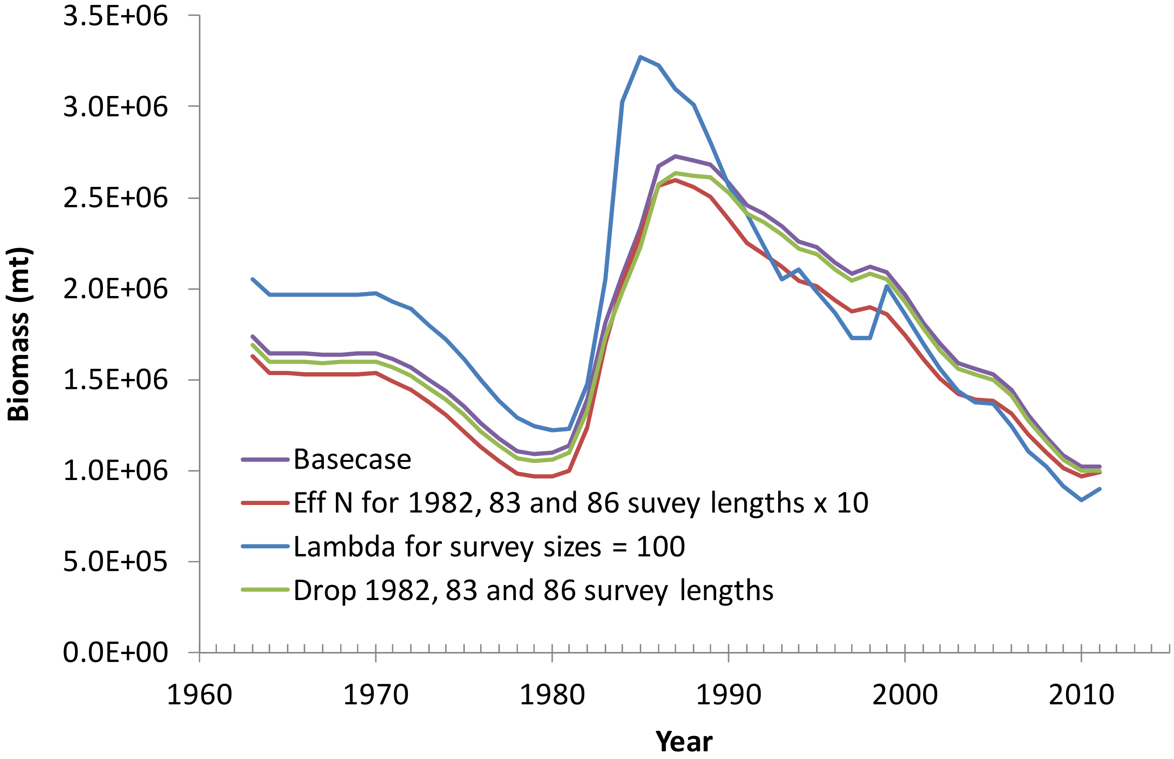


Figure A74. Biomass estimates for surfclams in the southern area from SS3, with 95% confidence intervals.

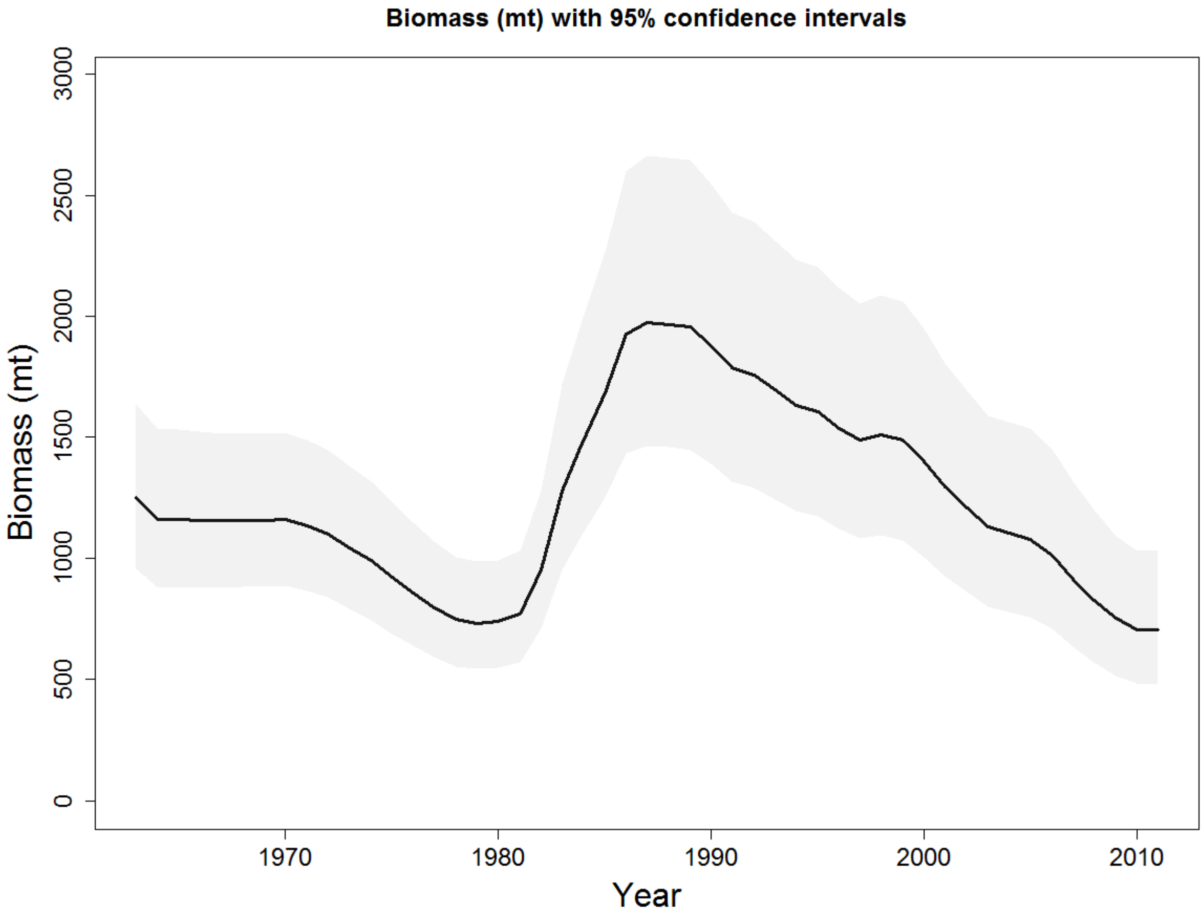


Figure A75. Recruitment estimates (thousands, age 0) for surfclams in the southern area from SS3, with 95% confidence intervals.

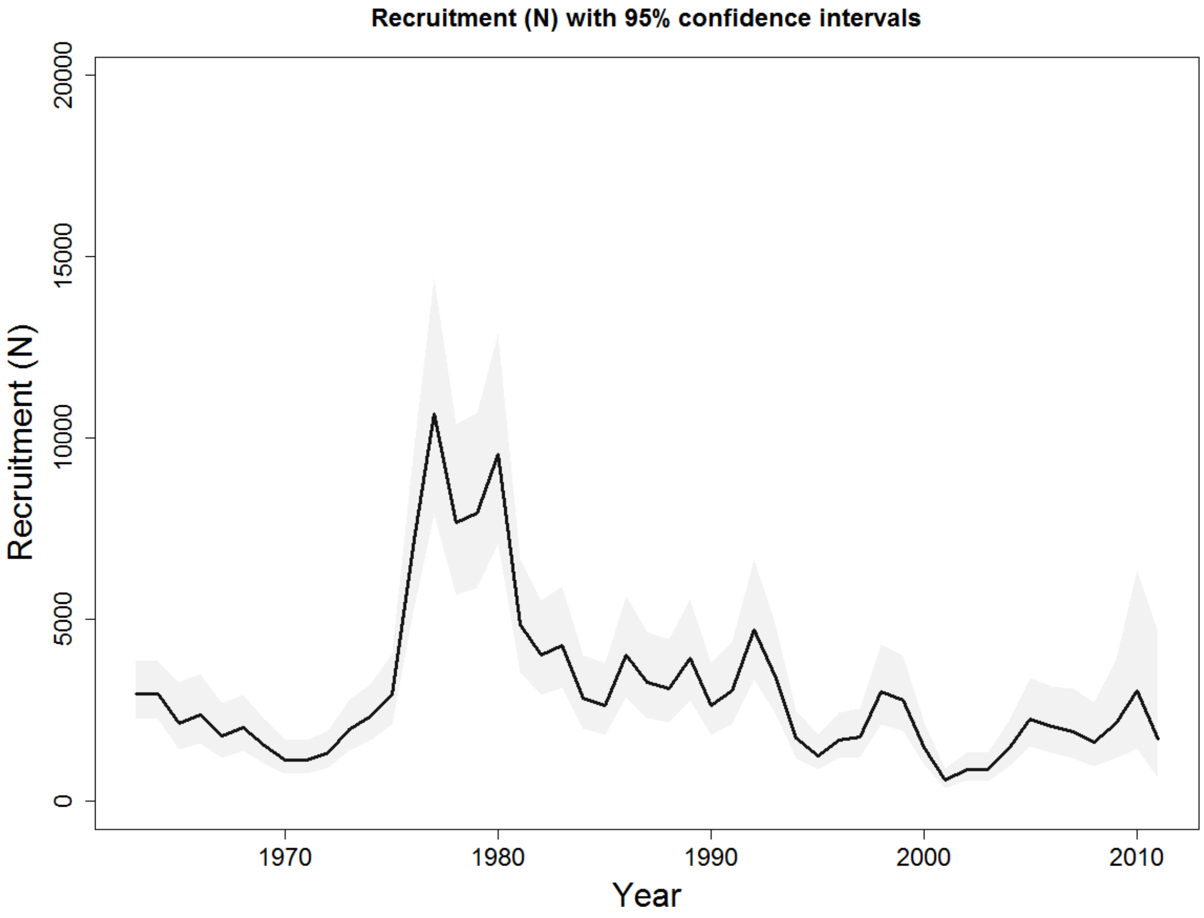


Figure A76. Fully recruited fishing mortality estimates for surfclams in the southern area from SS3, with 95% confidence intervals.

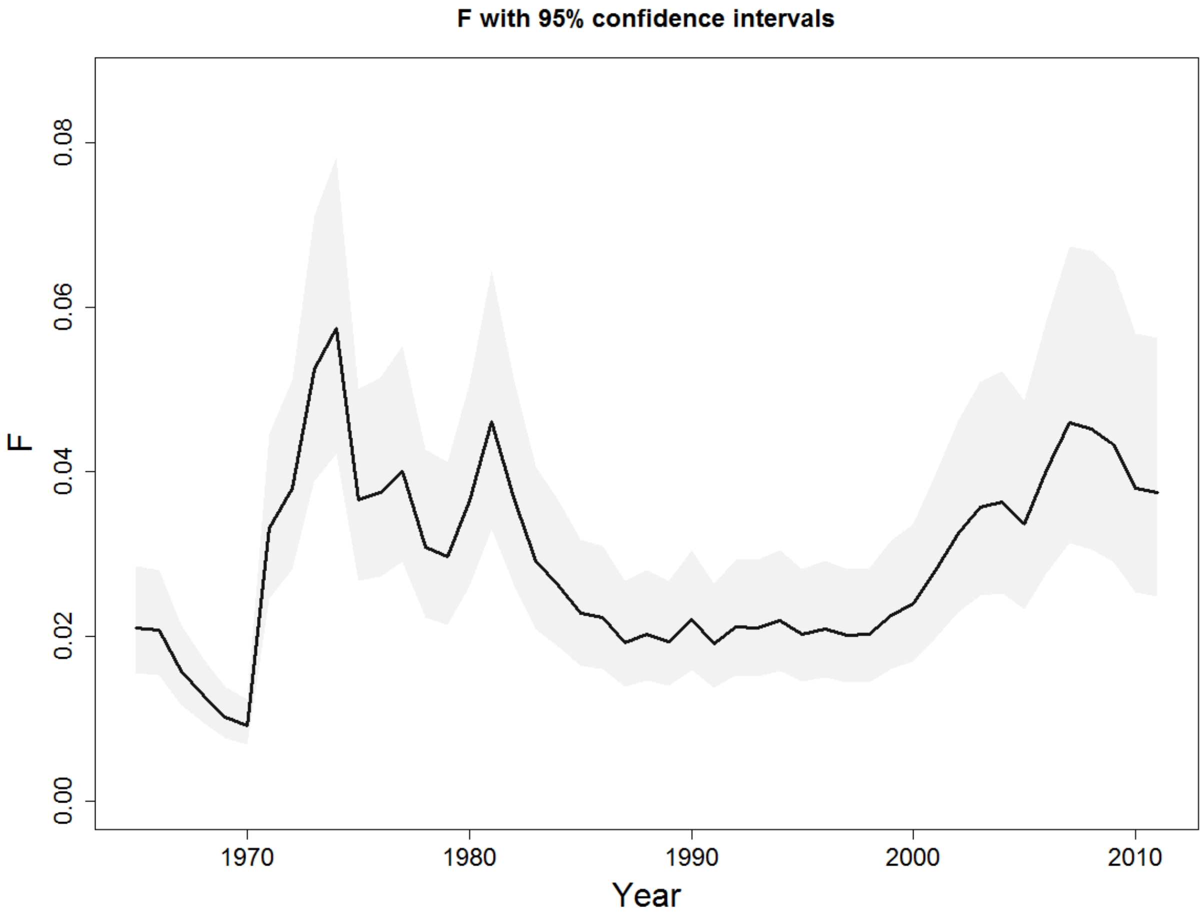


Figure A77. Biomass estimates for surfclams in the GBK area from SS3, with 95% confidence intervals.

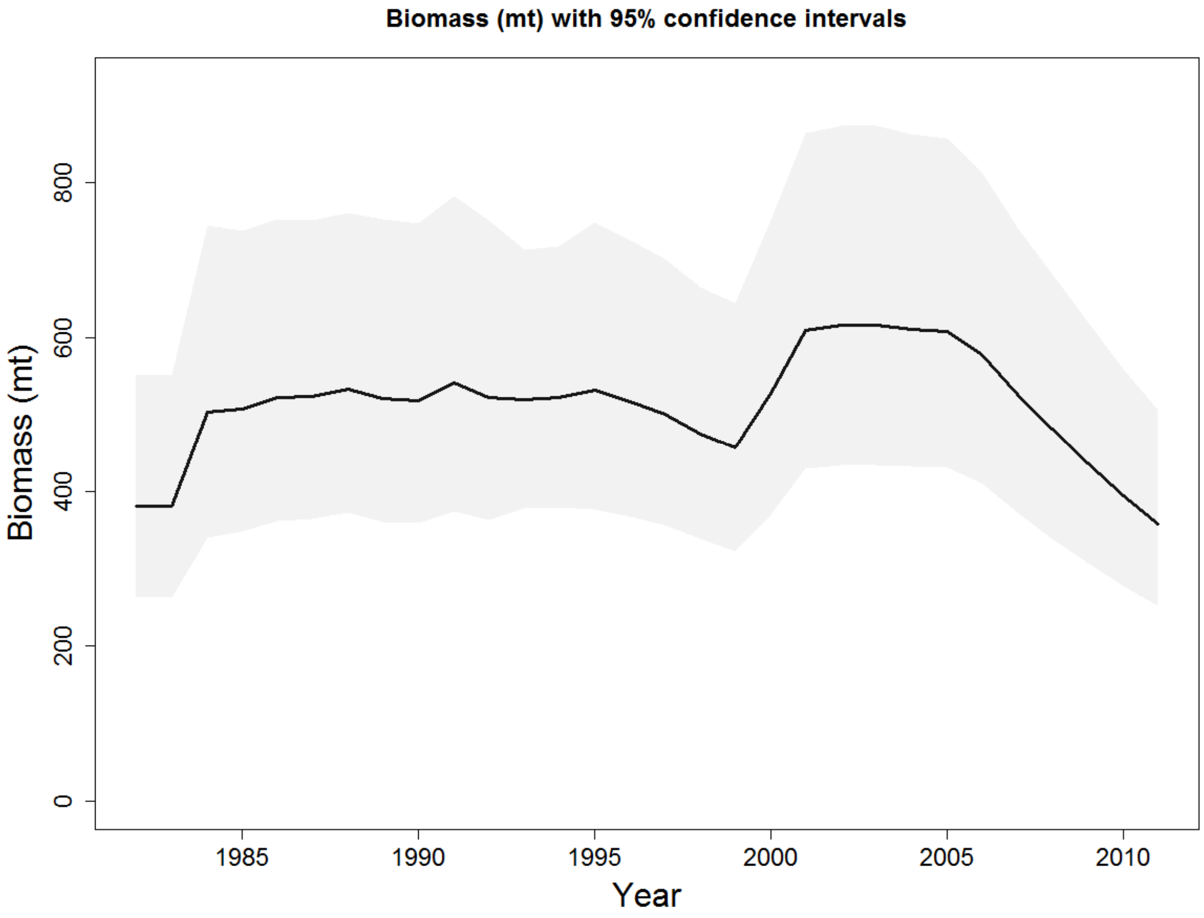


Figure A78. Recruitment estimates (thousands, age 0) from the northern area from SS3, with 95% asymptotic confidence intervals.

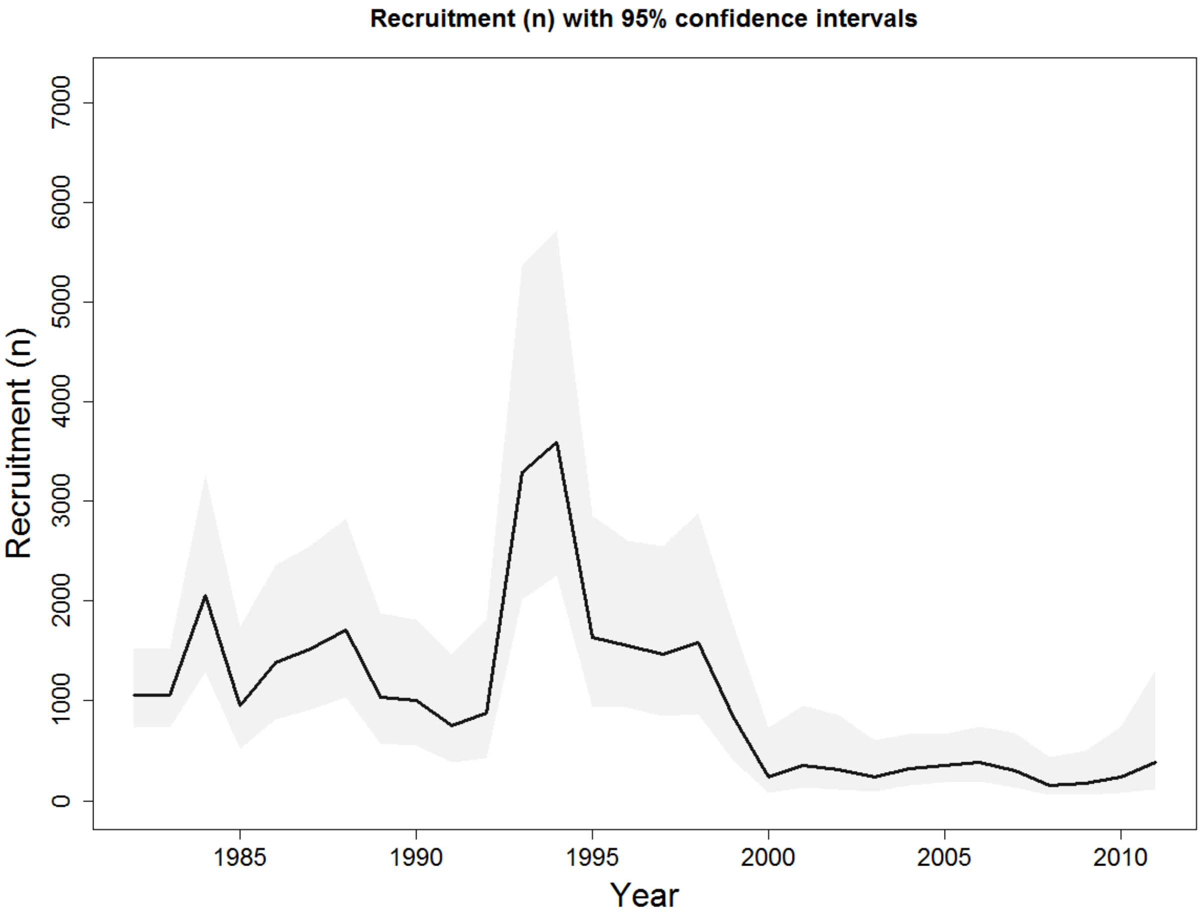


Figure A79. Fully recruited fishing mortality estimates from the GBK area, with 95% confidence intervals.

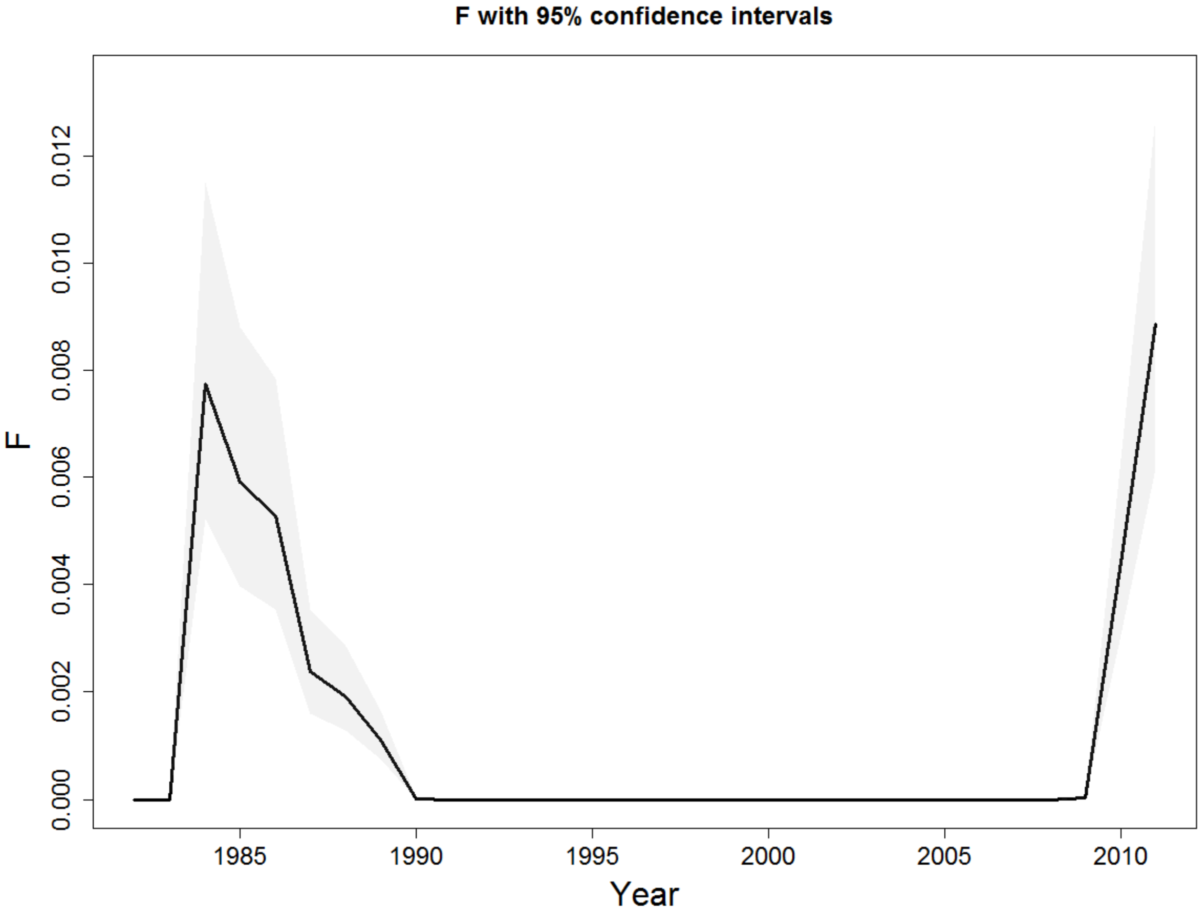


Figure A80. Likelihood profile analysis for survey dredge efficiency, 2011 biomass and the biomass status ratio (B_{2011}/B_{1999}) using the basecase SS3 model for surfclams in the southern area. The dashed line in panels A) and B) can be used to find bounds for approximate 95% confidence intervals. In particular, if two vertical lines are drawn through the intersection of the dashed black and blue likelihood lines, then the confidence interval bounds for dredge efficiency are found where the vertical lines intersect the x-axis and where the vertical lines intersect the red lines for biomass (A) and status ratio (B). Panel C) shows the effect on estimated biomass trend of fixing survey dredge efficiency at values between Q=0.18 and 0.49.

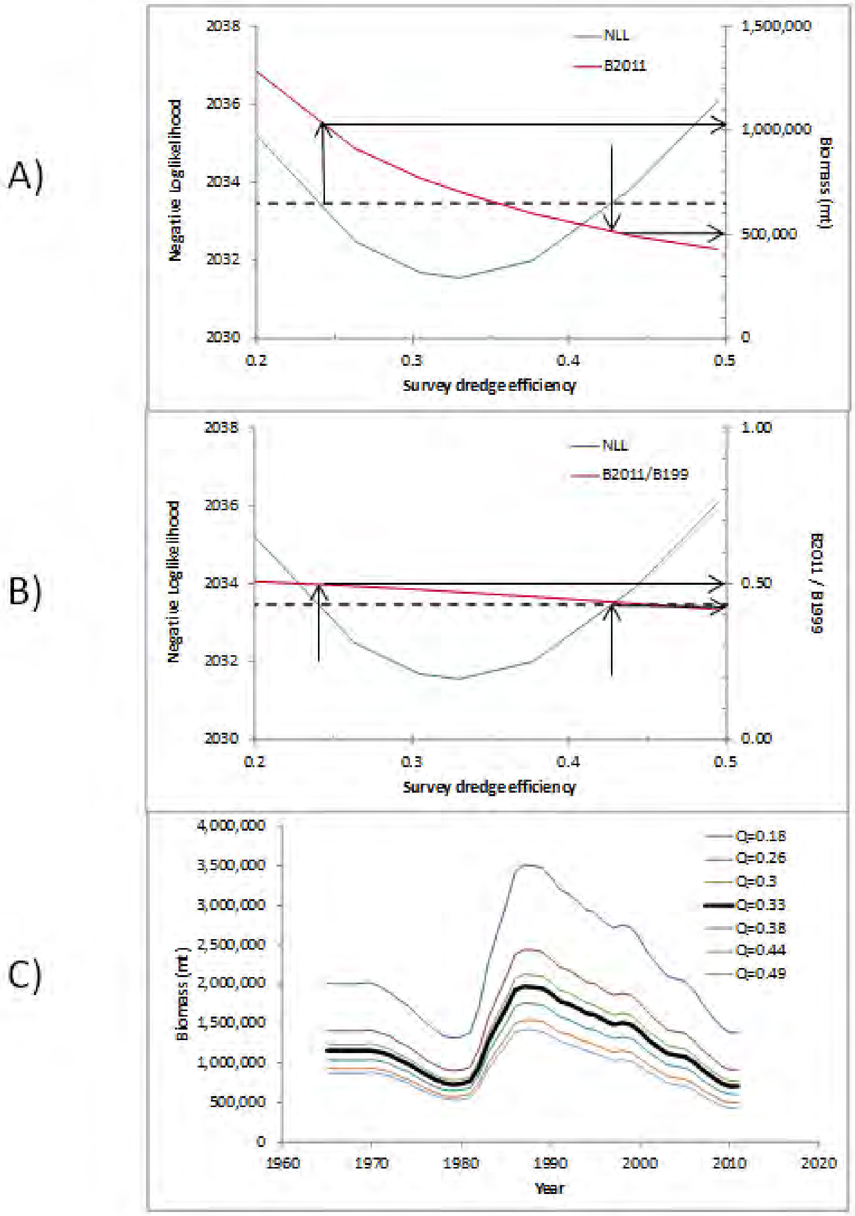


Figure A81. Internal retrospective pattern for biomass (ages 6+ y) from the southern area SS3 model. Mohn's $\rho = 0.02$ (9 year peel).

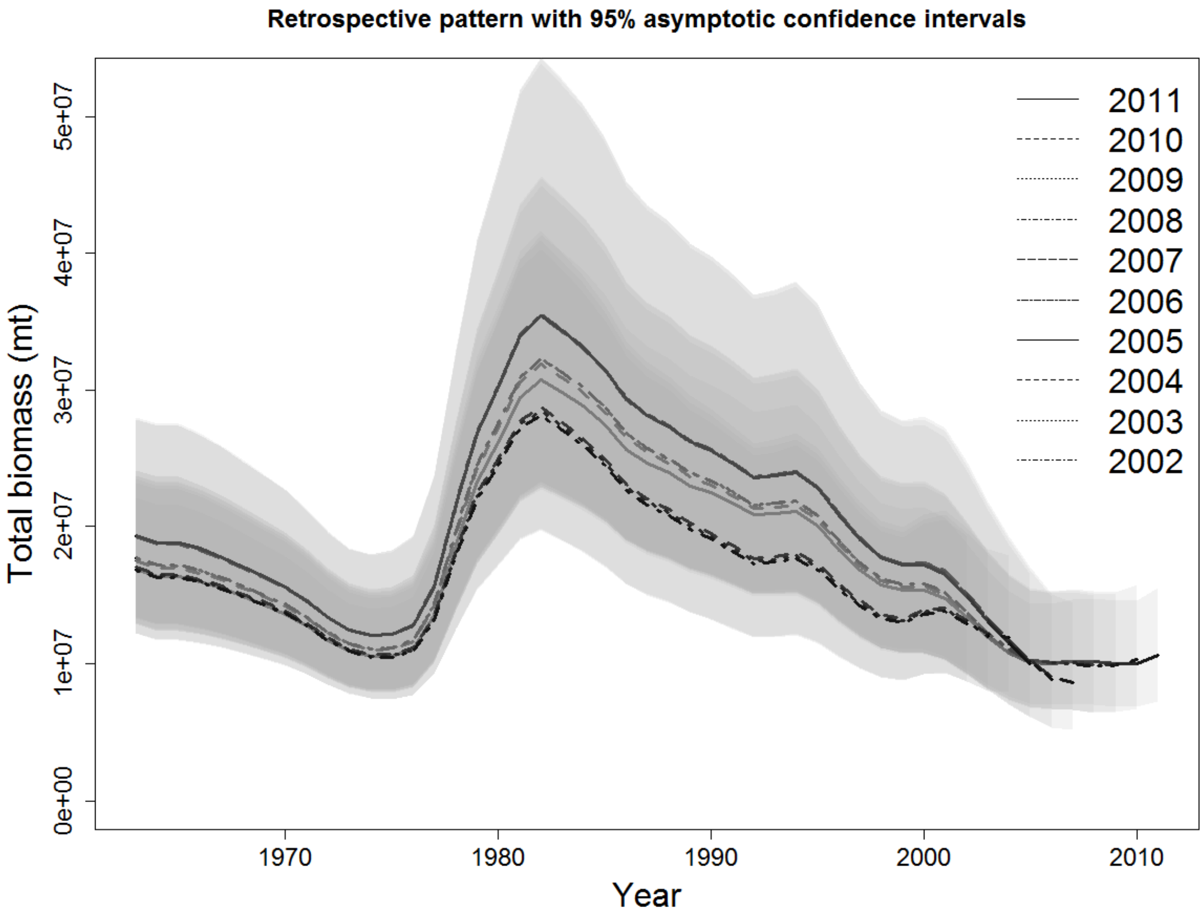


Figure A82. Internal retrospective pattern based on total biomass (ages 7+ y) from the GBK SS3 model. Mohn's $\rho = 0.30$ (9 year peel).

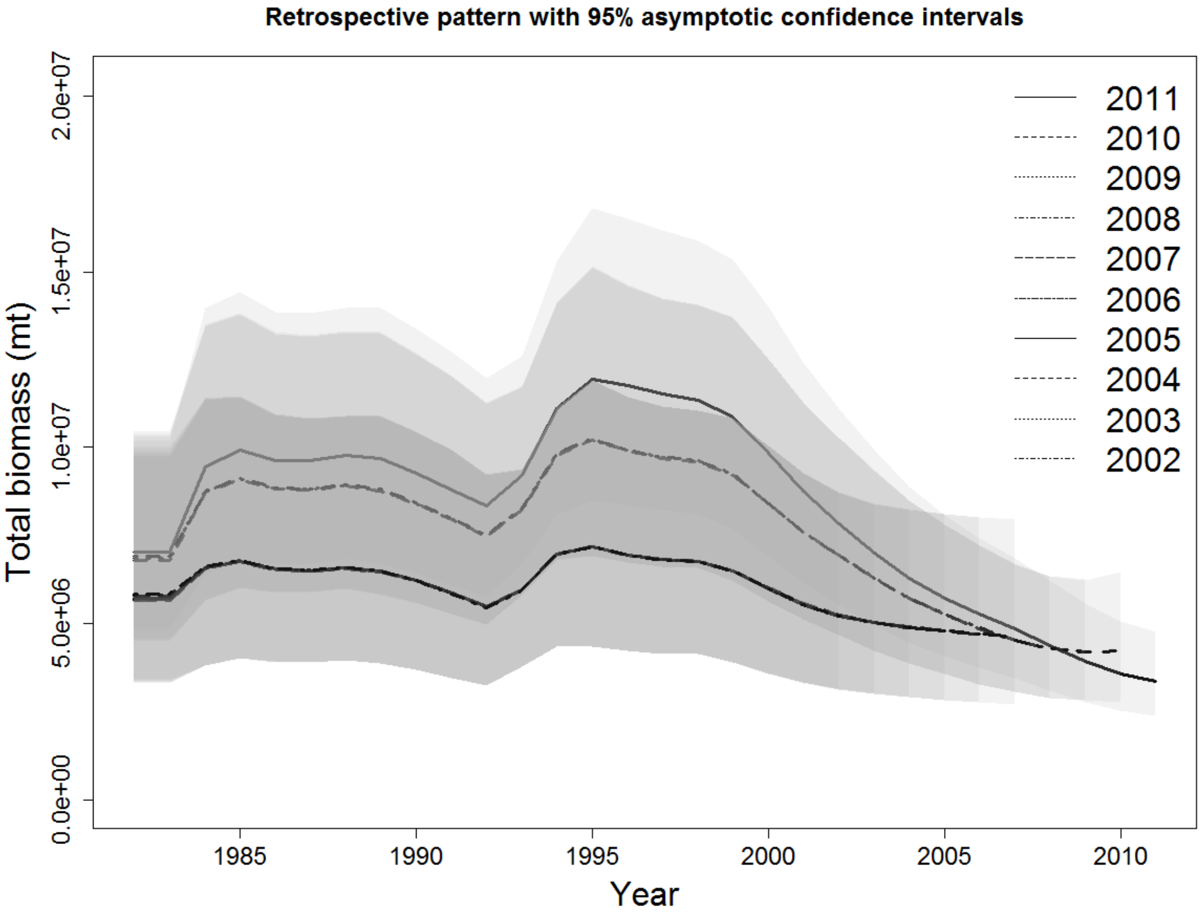


Figure A83. Historical retrospective comparing the biomass estimates for surfclams in the southern + GBK area from previous surfclam assessments.

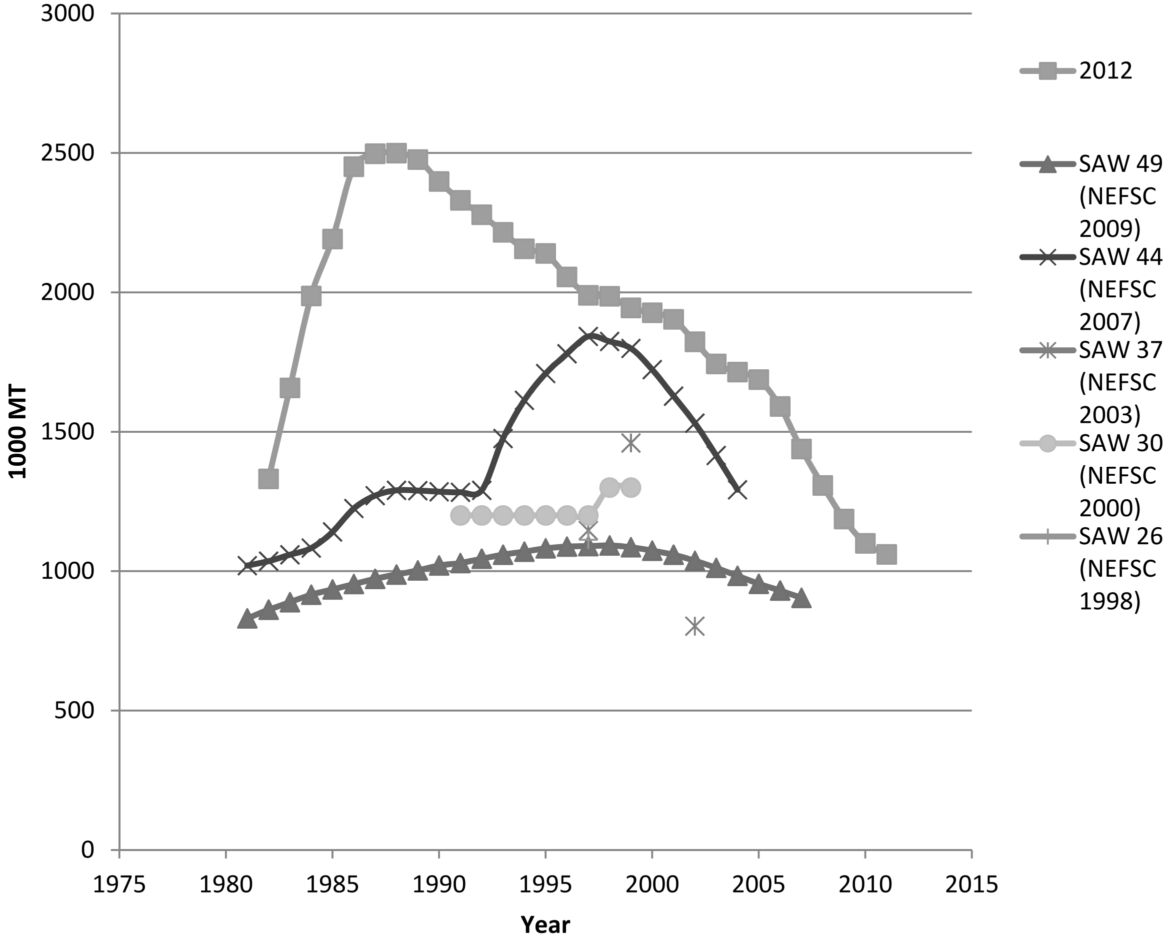


Figure A84. Whole stock biomass status estimates with cv and approximate 95% confidence intervals.

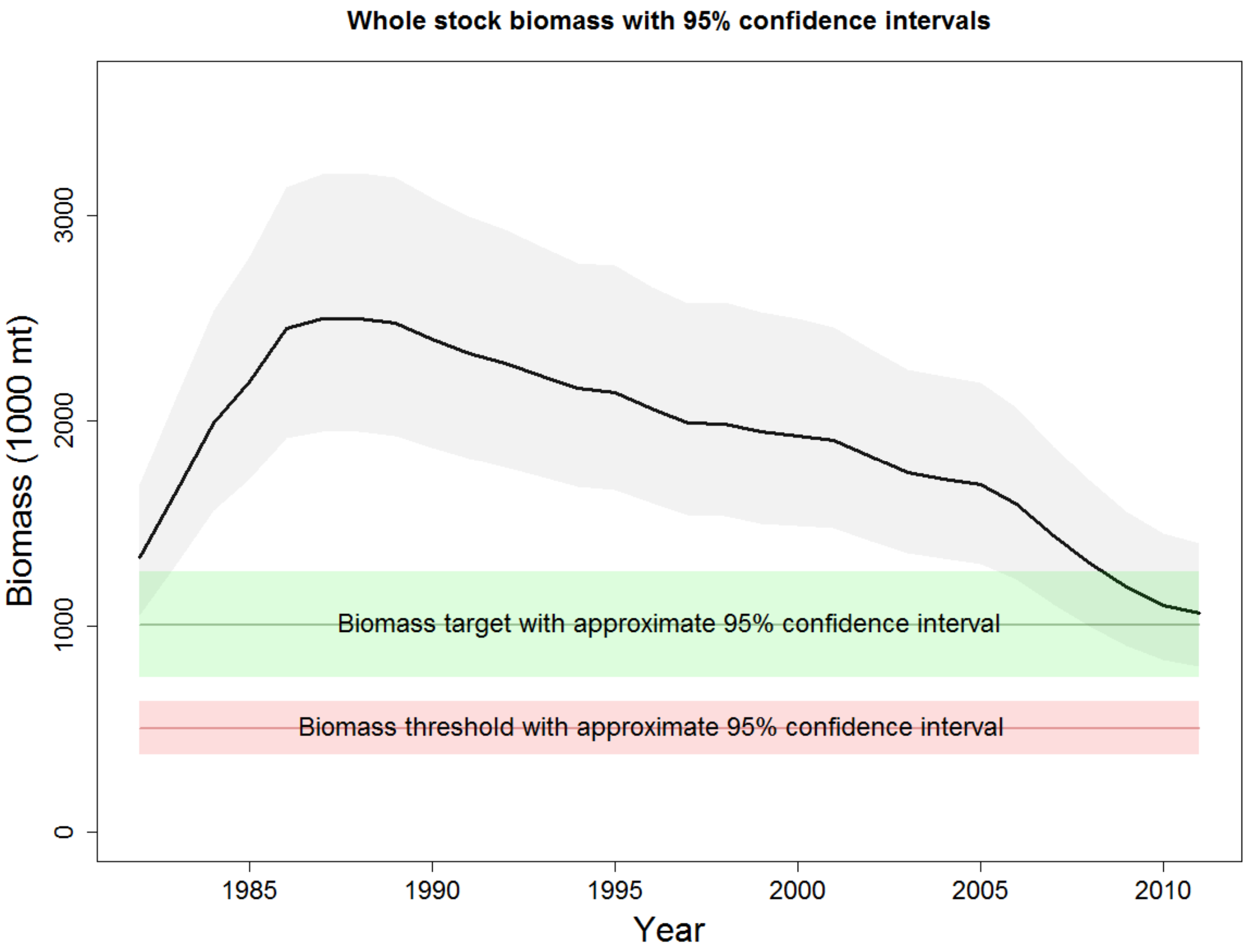


Figure A85. Whole stock F status estimates with cv and approximate 95% confidence intervals.

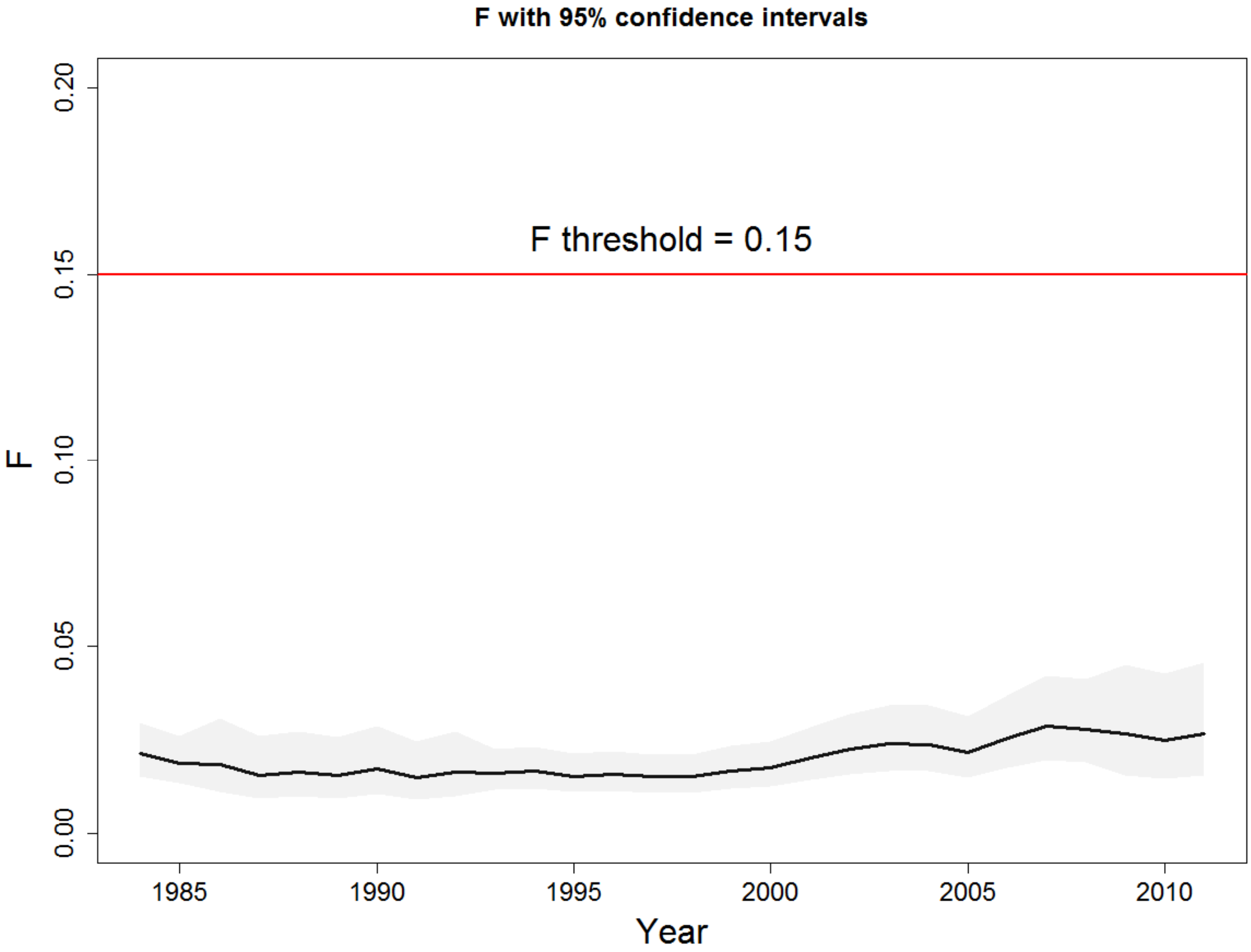


Figure A86. Southern area biomass status estimates with cv and approximate 95% confidence intervals.

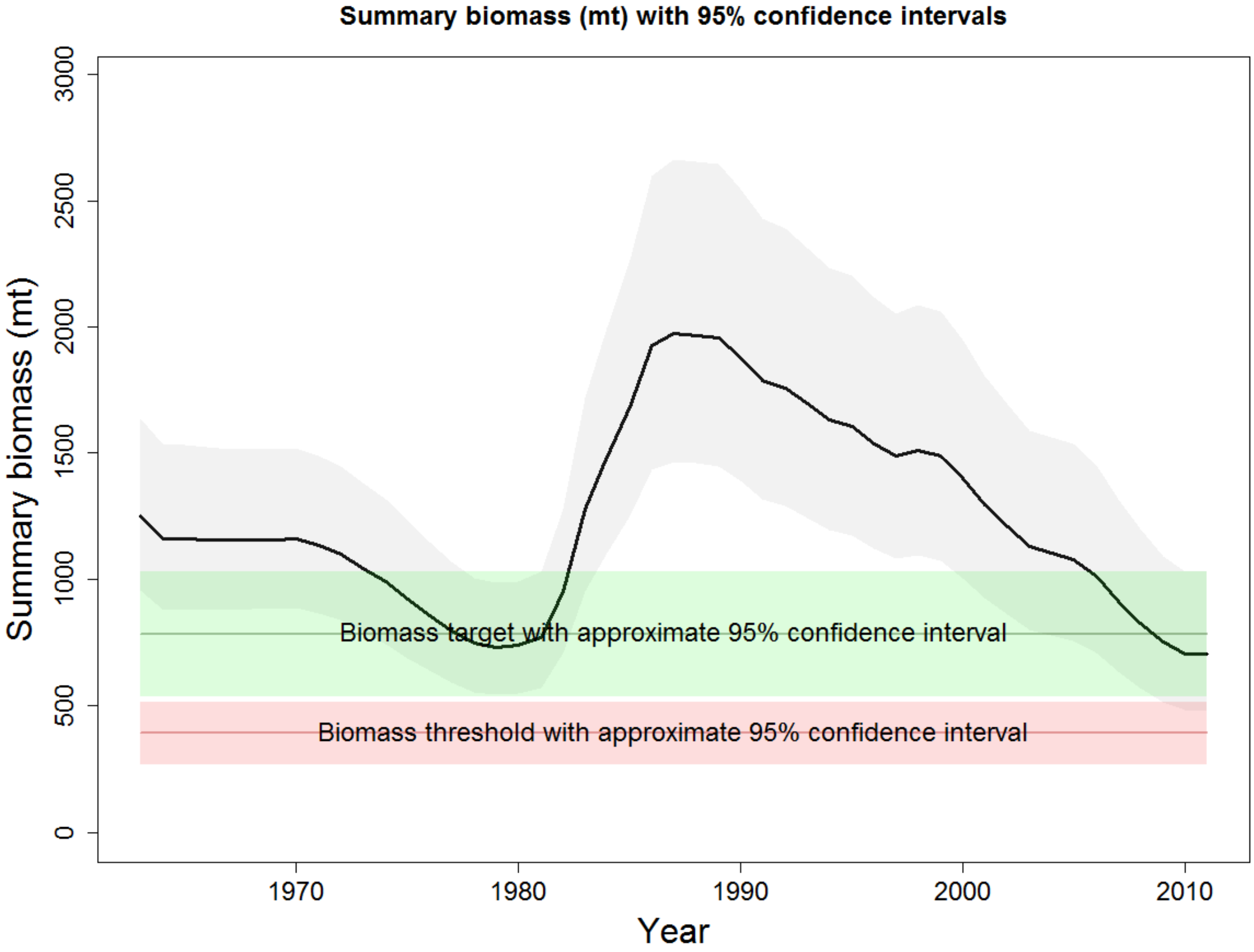


Figure A87. The distributions for B_{2011}~LogN(6.55,0.194) and $B_{THRESHOLD}$~LogN(5.92,0.167). The probability of being overfished is based on the methods of Shertzer et al. (2008).

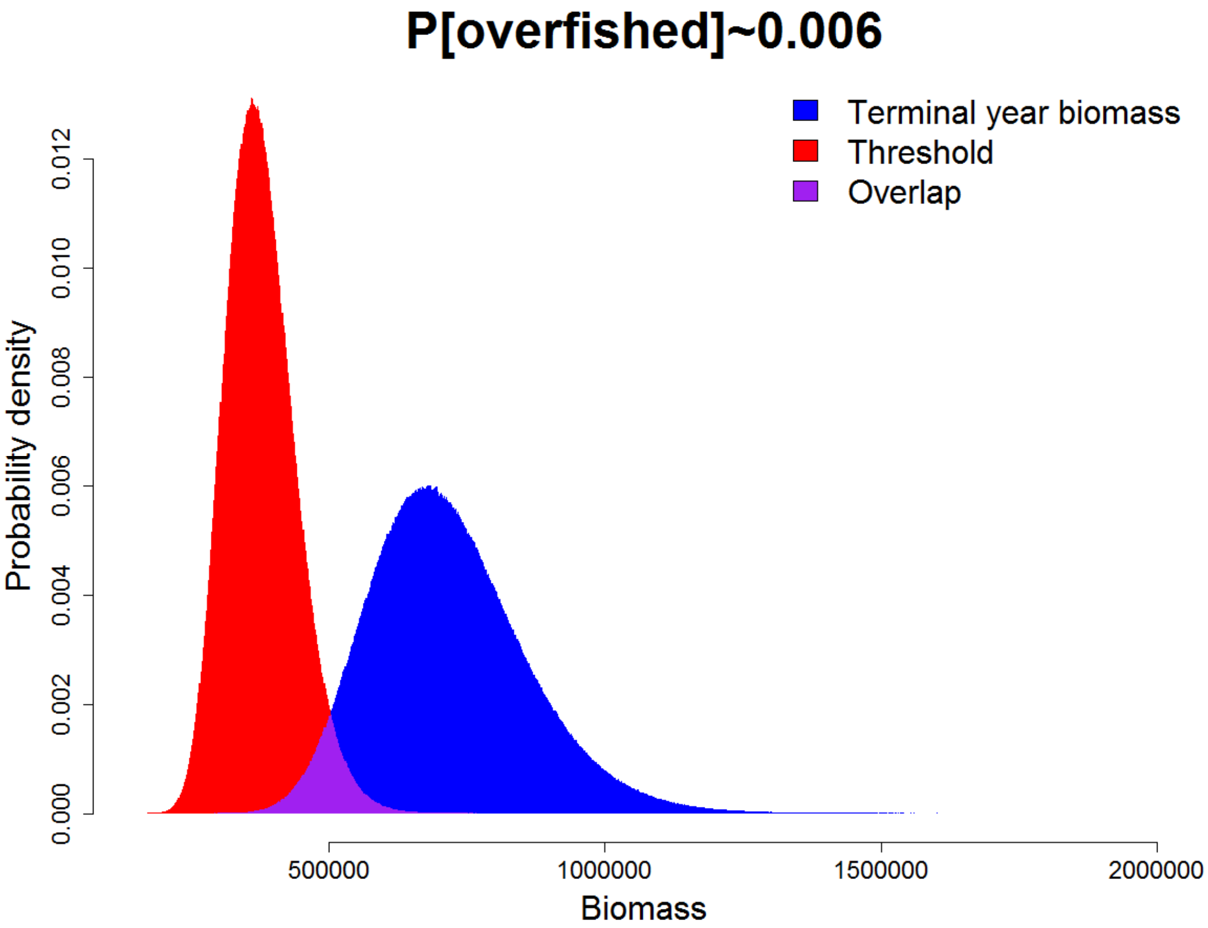


Figure A88. Southern area F status estimates with cv and approximate 95% confidence intervals.

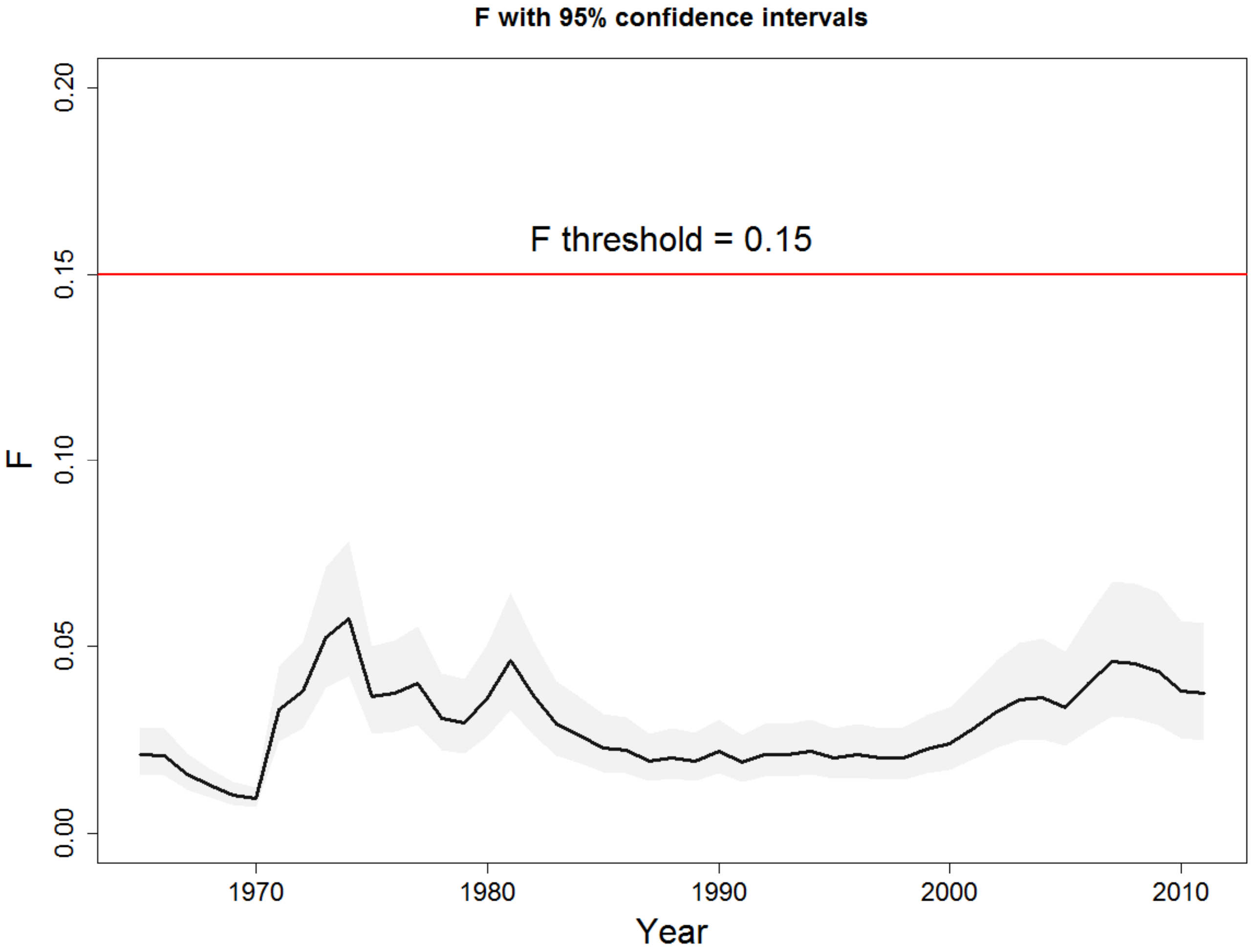


Figure A89. Projected biomass, landings and exploitation rates during 2012-2021 for surfclams in the southern, GBK and combined areas.

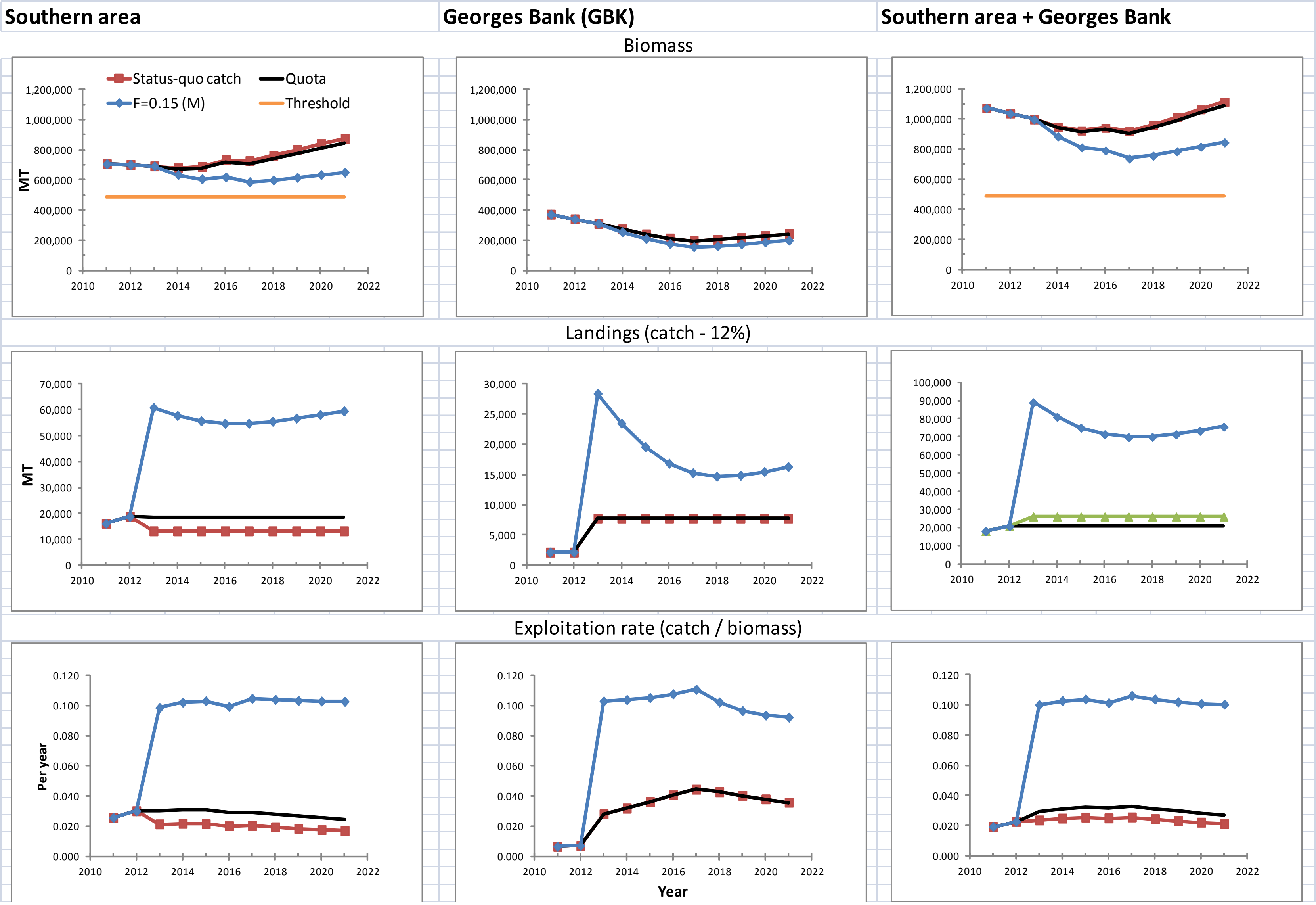


Figure A90. Summary biomass and 95% confidence intervals including projections for the whole stock, relative to biomass reference points. The dashed vertical line marks the terminal model year, 2011.

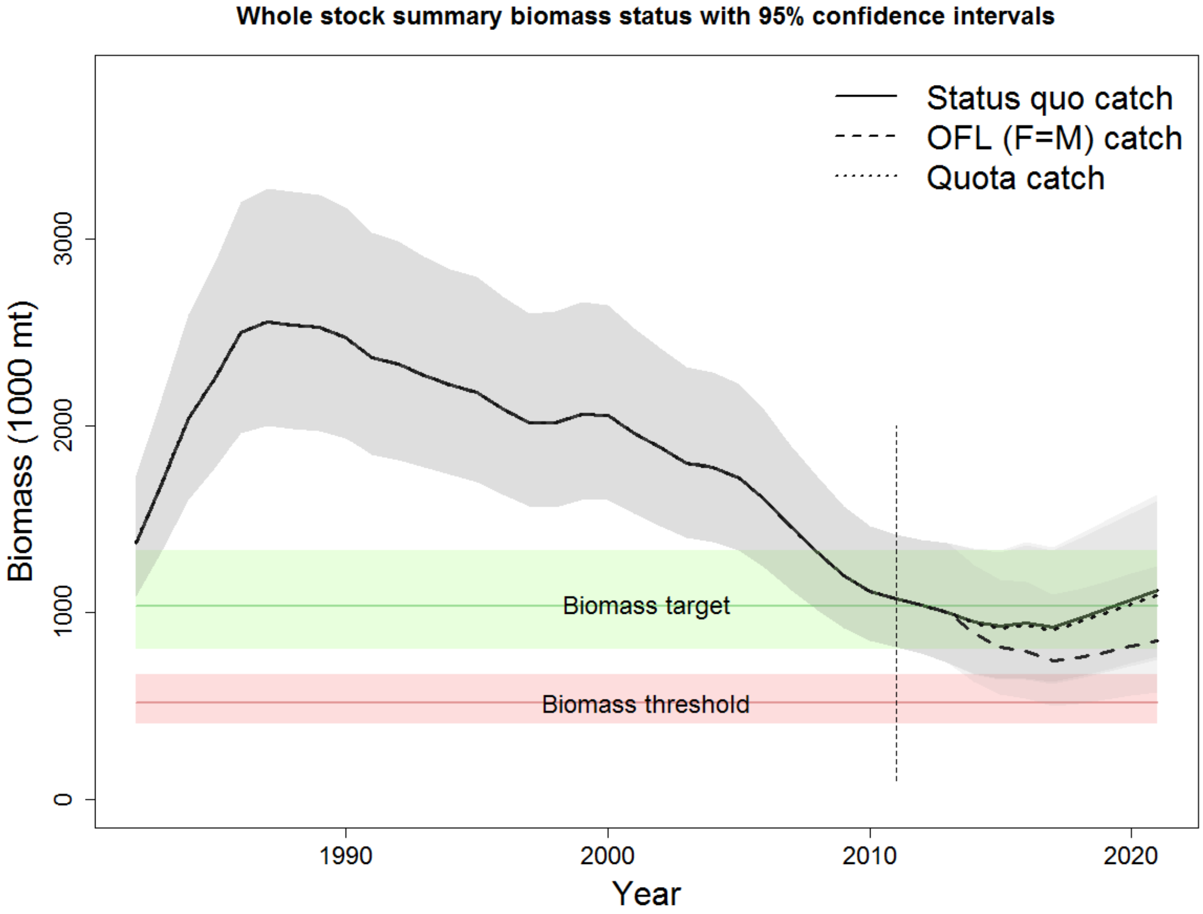


Figure A91. Annual fishing mortality and 95% confidence intervals including projections for the whole stock, relative to reference points.

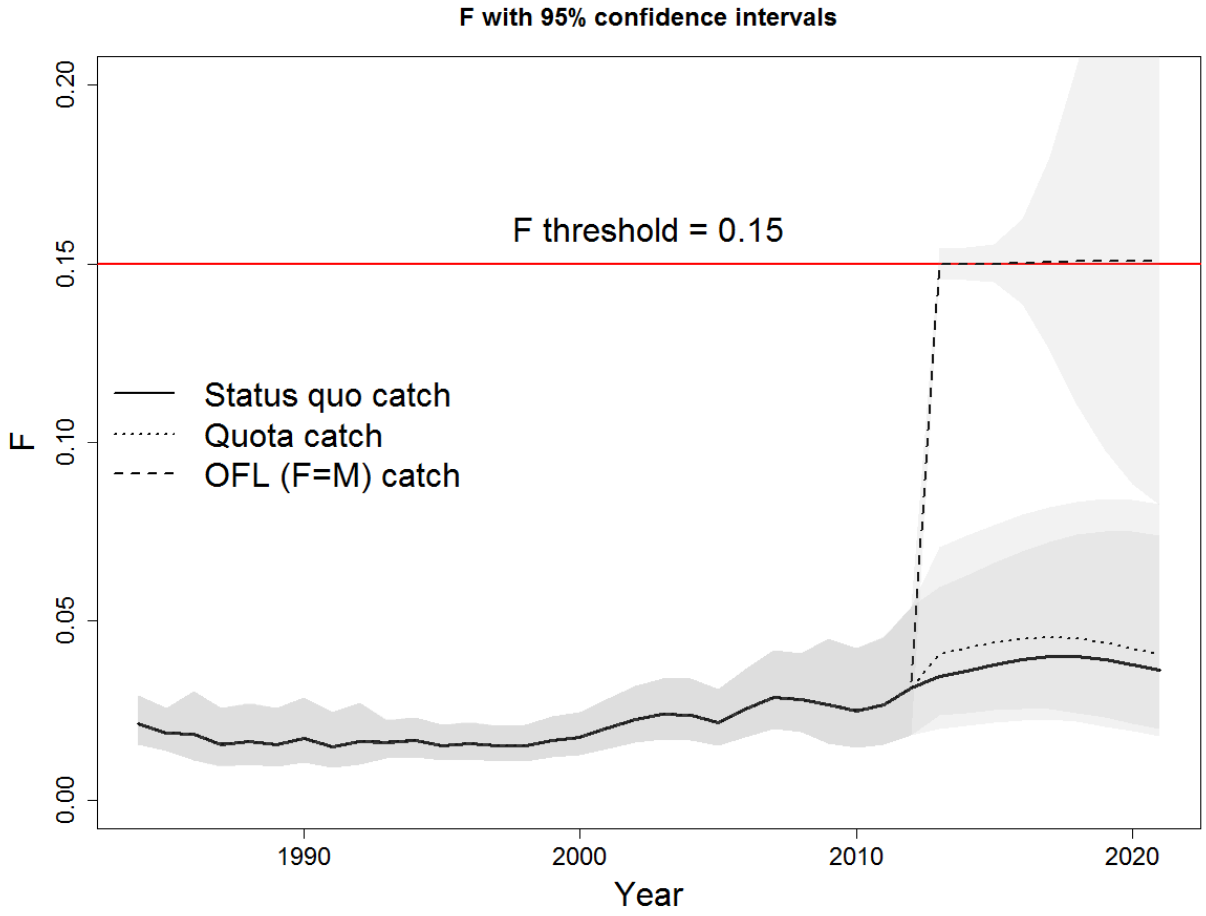


Figure A92. Summary biomass and 95% confidence intervals including projections for the southern area, relative to possible biomass reference points. The dashed vertical line marks the terminal model year, 2011.

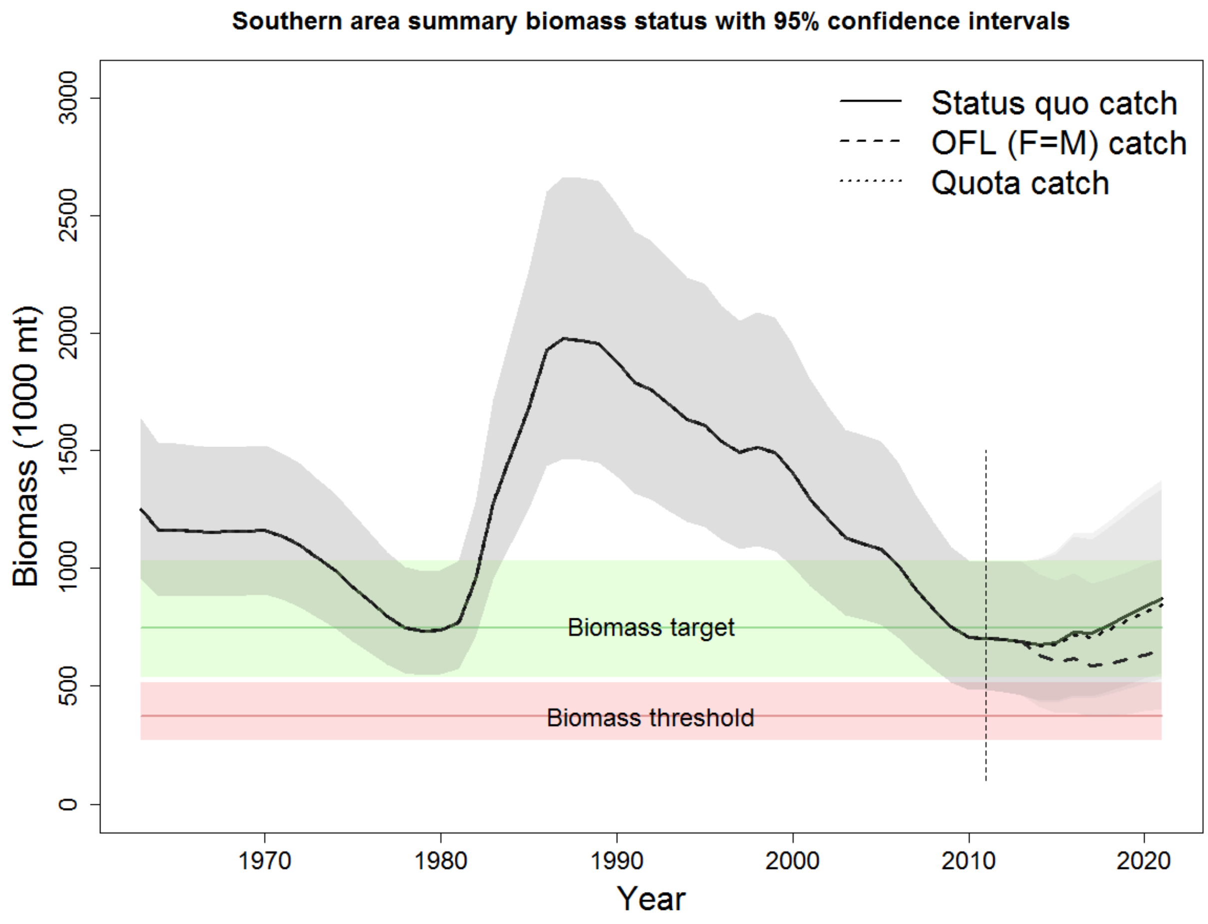


Figure A93. Annual fishing mortality and 95% confidence intervals including projections for the southern area, relative to reference points.

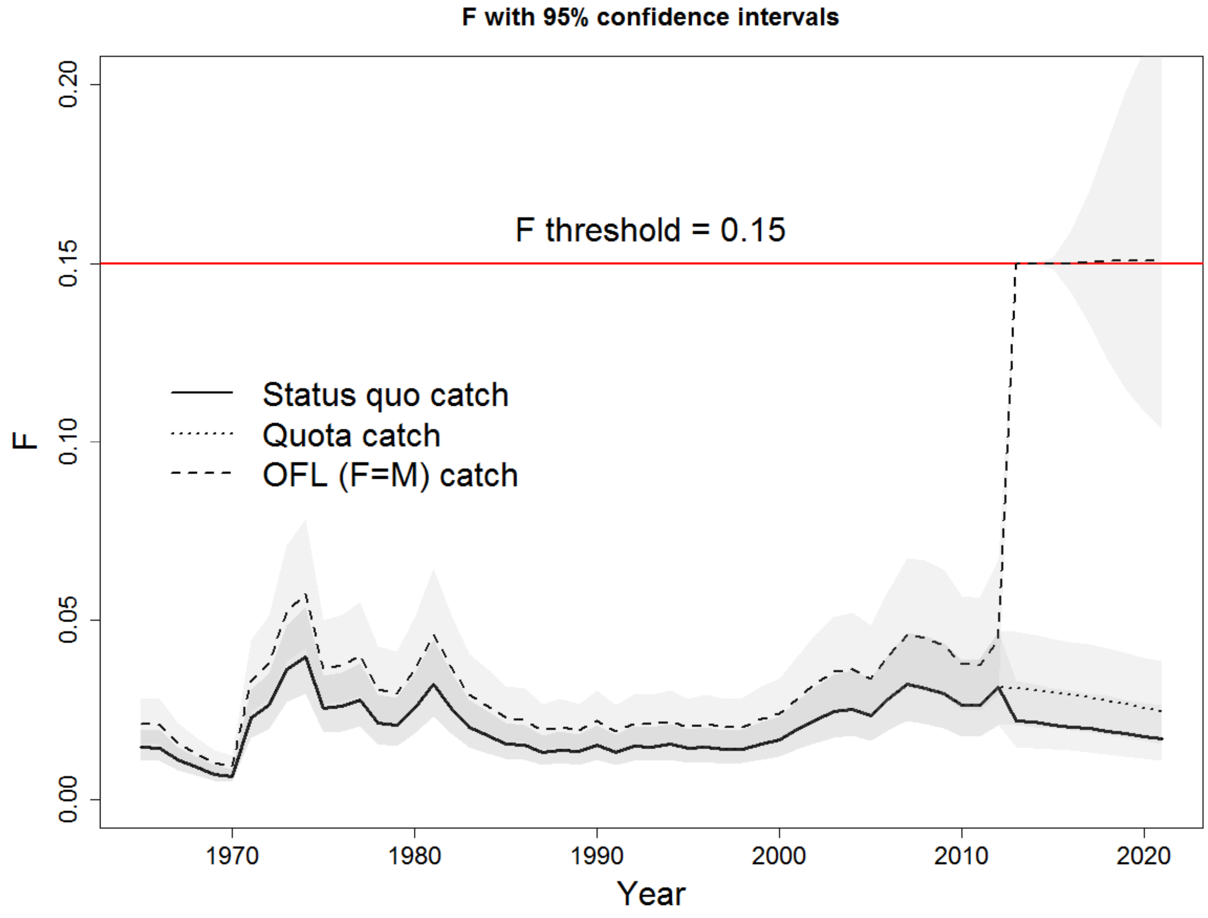


Figure A94. Summary biomass and 95% confidence intervals including projections for the northern area, relative to possible biomass reference points. The dashed vertical line marks the terminal model year, 2011.

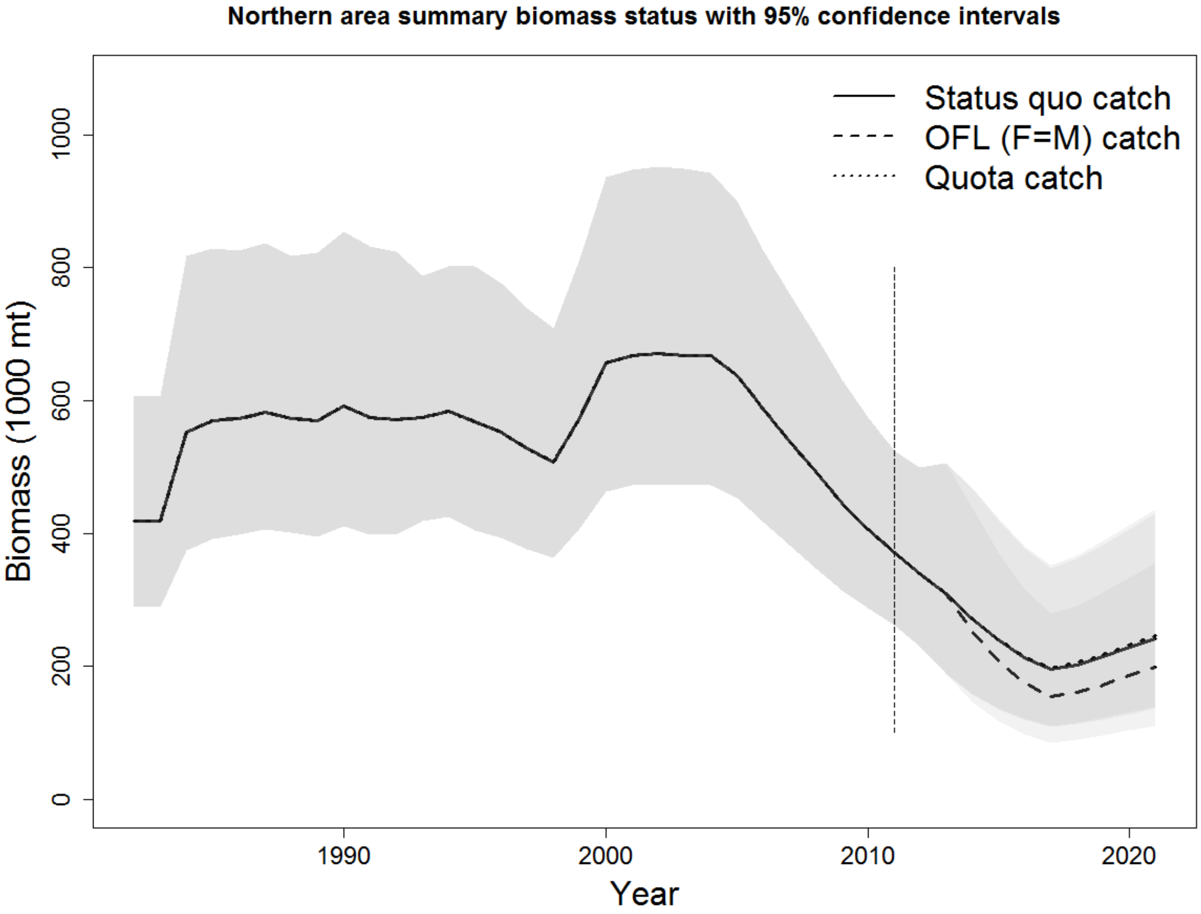


Figure A95. Annual fishing mortality and 95% confidence intervals including projections for the northern area, relative to reference points.

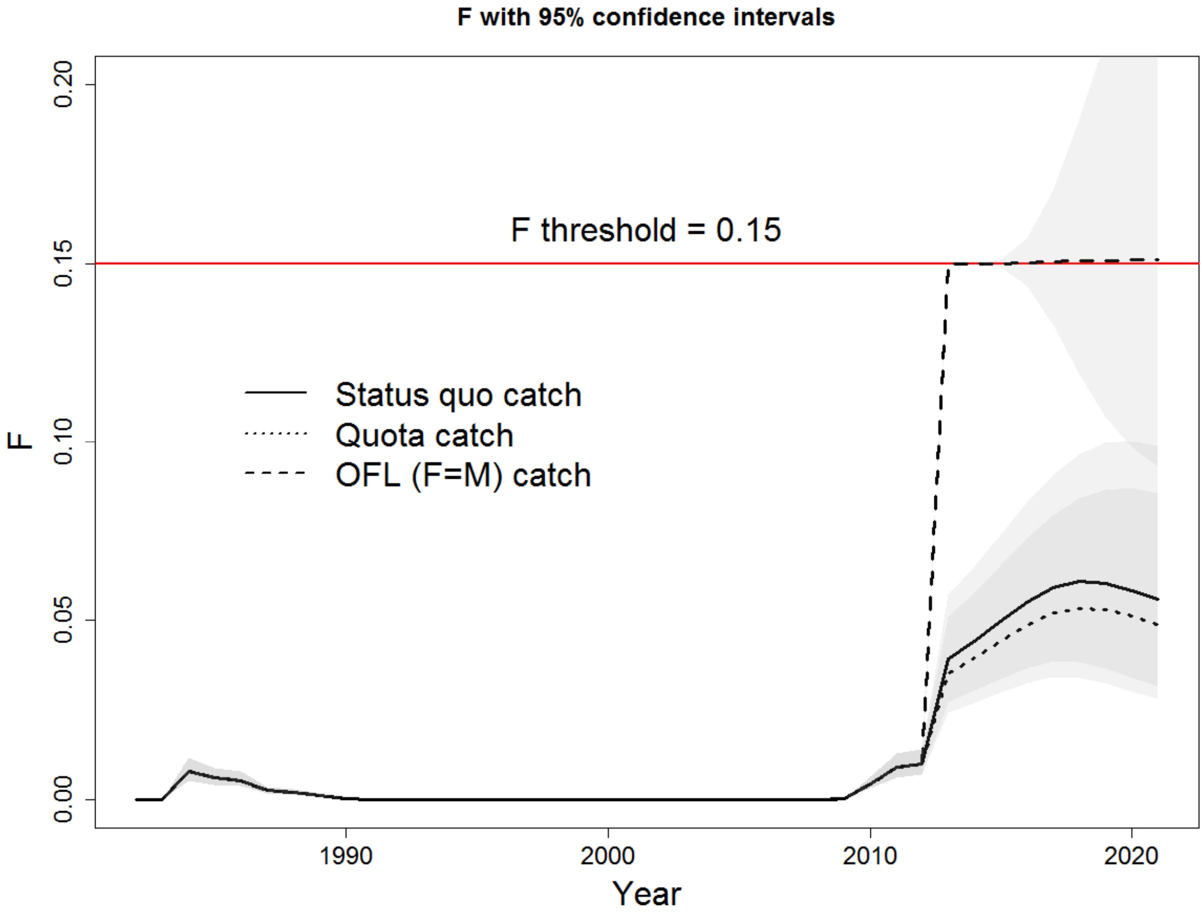


Figure A96. The maximum probability of the whole stock being overfished in any one of the next five years (2013 – 2017), given the three projection scenarios.

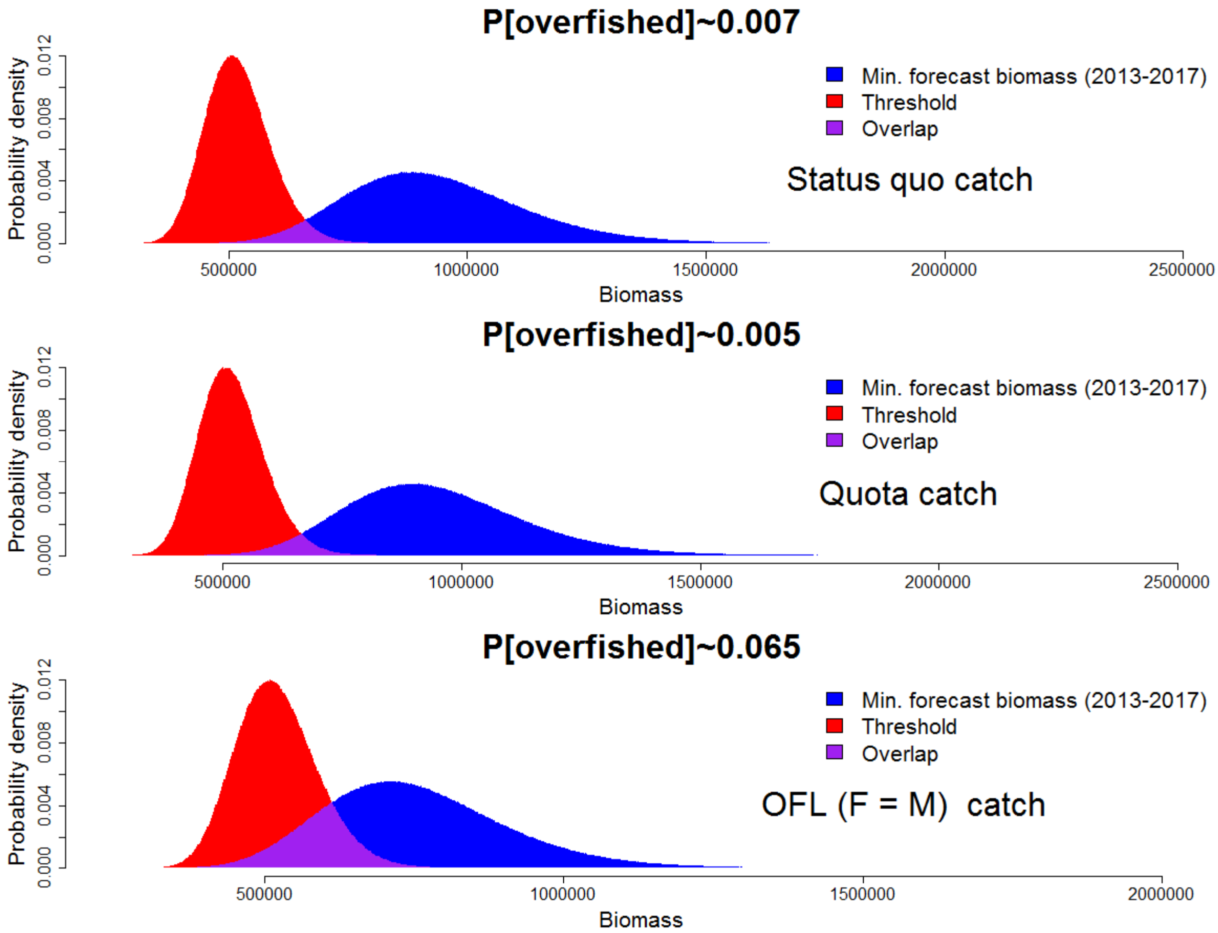


Figure A97. The maximum probability of the southern area being overfished in any one of the next five years (2013 – 2017), given the three projection scenarios.

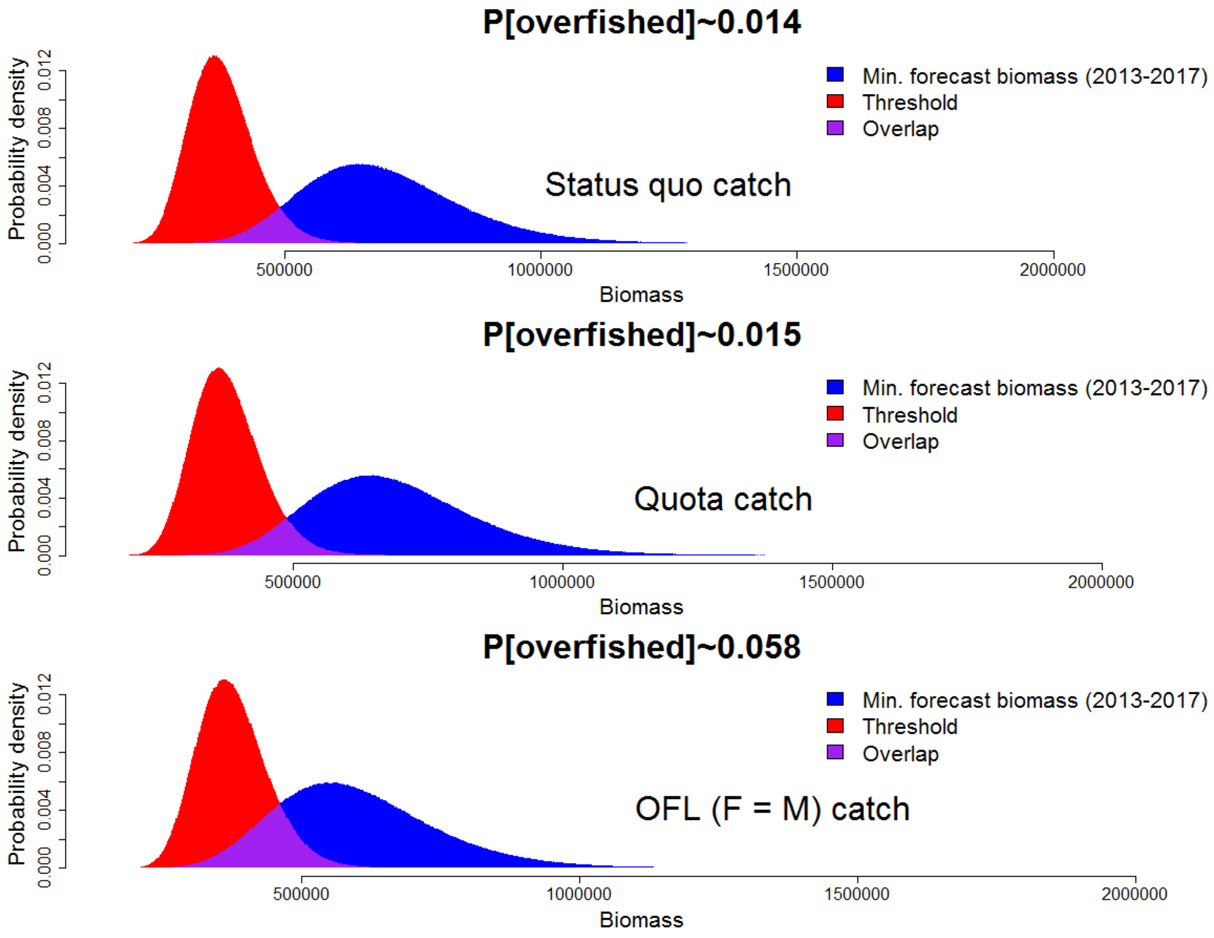


Figure A98. Probability distribution of the catch at the OFL for each of the next five years in projection for the whole stock.

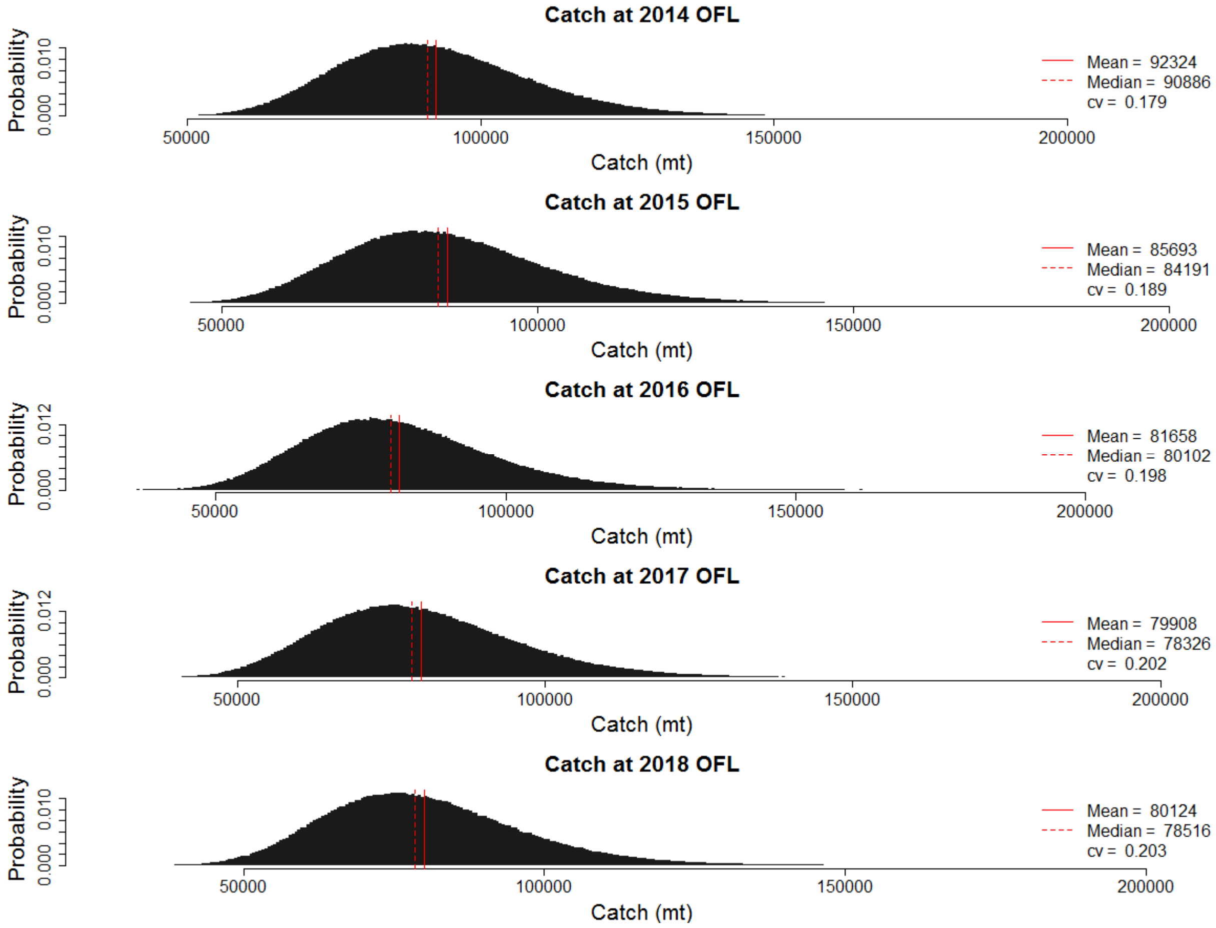


Figure A99. Probability distribution of the catch at the OFL for each of the next five years in projection for the southern area.

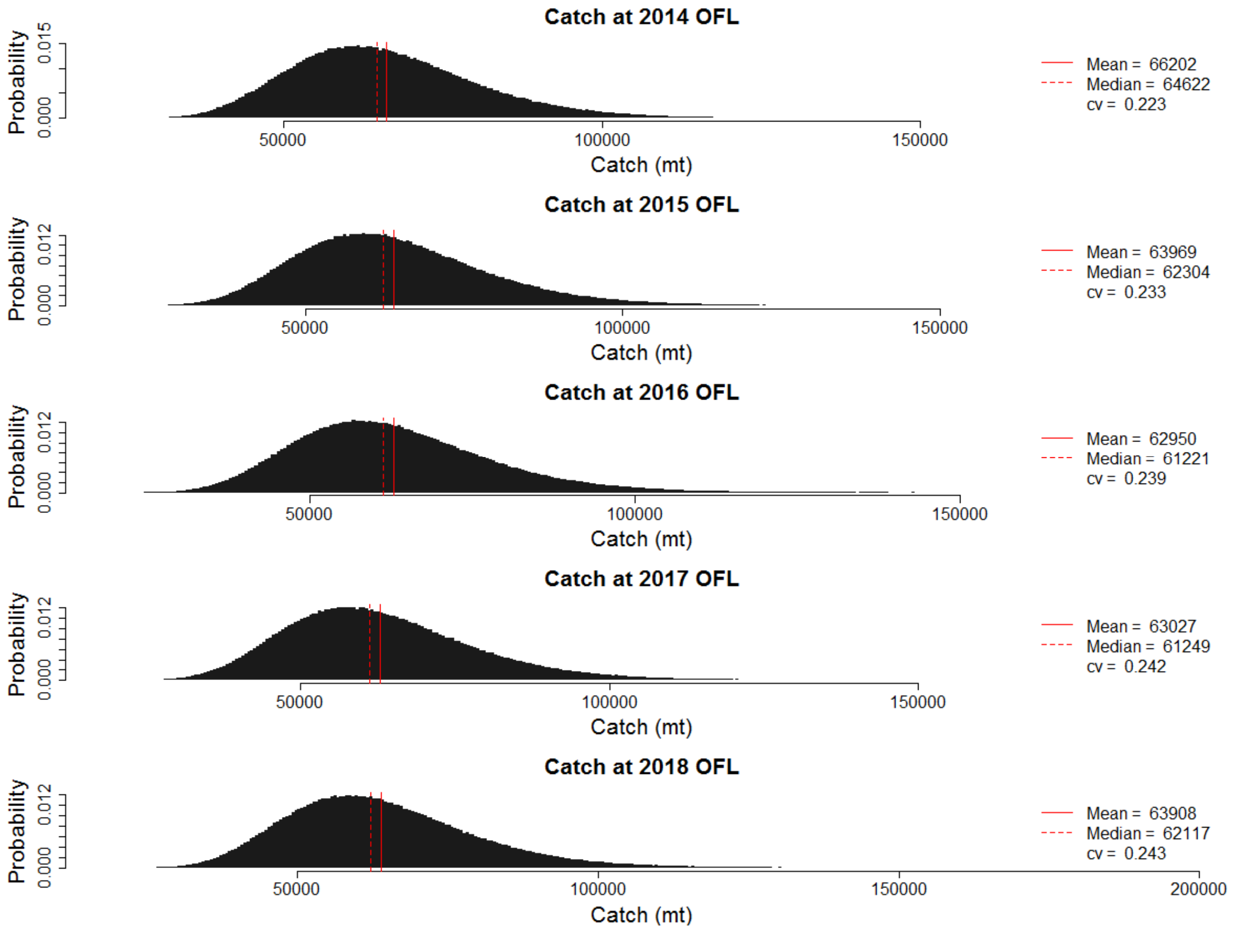
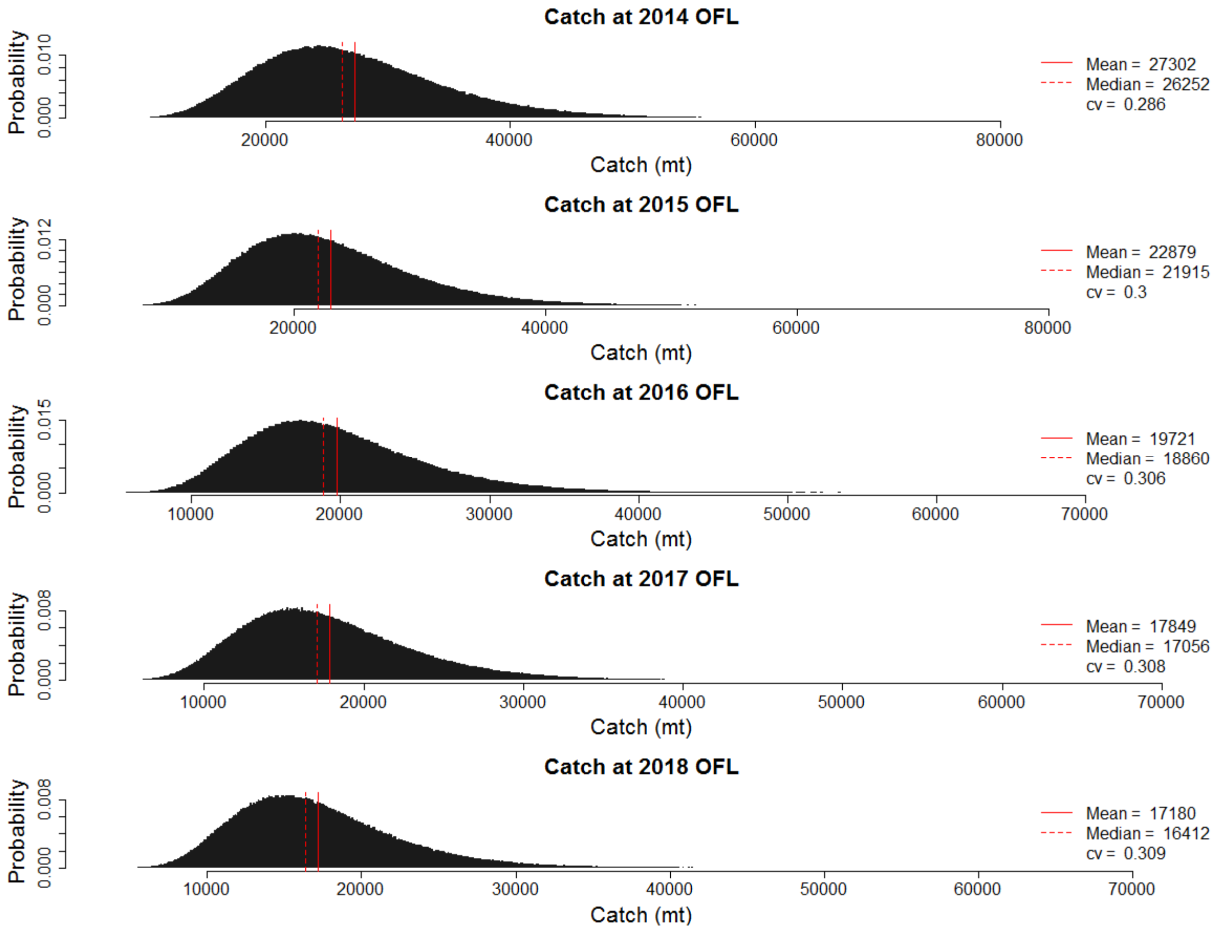


Figure A200. Probability distribution of the catch at the OFL for each of the next five years in projection for the GBK area.



A. Stock assessment appendices for Atlantic Surfclams in the US EEZ

Appendix A1: Surfclams in New York and New Jersey state waters[4]

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[1] Many thanks to Jeff Normant of the New Jersey Division of Fish and Wildlife and Debra Barnes and Jennifer O'Dwyer of the New York State Department of Environmental Conservation for data and assistance with this report.

The states of New York and New Jersey support surfclam fisheries in their territorial waters not covered by the NEFSC clam survey. The two states have carried out their own annual or semi-annual surveys of the resource since 1992 and 1988, respectively. Commercial and survey data from state waters are important in this assessment of the federally managed EEZ stock given the biological linkage between state waters and the EEZ, the productivity and importance of fisheries in state waters, and the possibility of environmental effects in southern surfclam habitat. New York and New Jersey state waters have historically been excellent habitat for surfclams, but there is evidence of declining recruitment to the population in both states. The percentage of landings harvested from state waters has been falling since 2001 (Figure 1).

The New York and New Jersey state surveys

The New Jersey State survey is conducted annually by the New Jersey Department of Environmental Protection from a commercial clam vessel with a commercial hydraulic dredge, most recently the F/V Ocean Bird. The survey has been conducted since 1988, and has followed a stratified random sampling protocol since 1994. The survey area is divided into regions covering the whole New Jersey coast, and each region has 3 one mile wide strata, parallel to the coast, covering surfclam habitat out to the 3-mile limit of state waters (Figure 2). Each survey does between 250 and 330 five minute tows, measuring the tow volume in bushels, then counting and measuring a known volume of surfclams for population estimates and length frequencies. Grab samples of the sediment are also taken.

Data from the state of New Jersey available for this appendix includes annual state surfclam survey numbers and lengths through 2012 and grab samples for juvenile surfclams through 2011. Surfclam landings from New Jersey state waters are available from 1989-2012.

The New York surfclam survey is conducted by the New York Department of Environmental Conservation approximately every three years. They use a commercial clam vessel, most recently the F/V Ocean Girl, with a hydraulic dredge. The survey area is divided into four regions which span the southern shore of Long Island. The three westernmost regions are subdivided into three mile wide strata (Figure 3). The most recent surveys have taken place in the summer or fall, had an average of 236 stations, and used a random stratified sampling technique. Tows are three minutes long, the total volume of each tow is measured in bushels, and half a bushel of surfclams from each tow is measured and counted for population estimates and length frequencies. A picture of the dredge used is shown in Figure 4.

Data from New York State are from the 2002, 2005, 2006, 2008 and 2012 state surfclam surveys. Total numbers, densities and length frequencies are available for all years and ages are available for all years except 2012. Surfclam landings from New York state waters are available through 2011.

Results

Both states have seen a significant decrease in the population of surfclams (Figure 5). The peak population of surfclams in New Jersey in recent years seems to have occurred in 1996, a few years before the peak in biomass in the EEZ in 1998-1999. The data available to us from New York do not go back far enough to see evidence of a concurrent population peak.

Despite the decline in numbers of clams in surveys since 2002, landings in New York stayed remained relatively high through 2006 (Figure 6). There was a very large harvest limit set in 2004 (930,000 bushels) and it was almost reached, making the landings from New York from that year almost double what they had been in years before. In 2010 and 2011, landings were around 200,000 bushels annually.

Surfclam landings for human consumption from New Jersey state waters have fallen from a high of about 700,000 bushels in 2003 to less than 100,000 in 2005 and to near zero levels since 2006. Since the early 2000s, a few tens of thousands of bushels of surfclams have been harvested annually from “prohibited waters” (where they are not allowed to be sold for human consumption due to contamination) to be sold as bait (Figure 7). About a third of the surfclam standing stock in New Jersey is in prohibited waters (Figure 8).

In the 2000s, the length composition of surfclams in New Jersey was narrow and composed of only larger surfclams, indicating a lack of new recruitment. However, recent survey data shows some smaller clams recruiting to the population (Figure 9). The 2011 NEFSC clam survey also showed evidence of some recruitment off New Jersey and New York.

Surfclams from the New York surveys conducted in 2005 and 2006 were larger on average than those collected in 2002, yet some smaller clams were seen in the 2008 and 2012 surveys, mirroring the bump in recruitment seen in the New Jersey and NEFSC surveys (Figure 10).

Surfclam densities have historically been high in the inshore areas surveyed by New Jersey and New York states compared to offshore areas south of Georges Bank surveyed by NEFSC (Figure 12). However, inshore densities appear to be falling recently towards levels typical of more unproductive offshore areas (Figure 11). However, the comparisons in Figure 11 are approximate due to differences in dredge design, capture efficiency and size selectivity. Numbers have been falling in all strata in New Jersey (Figure 13).

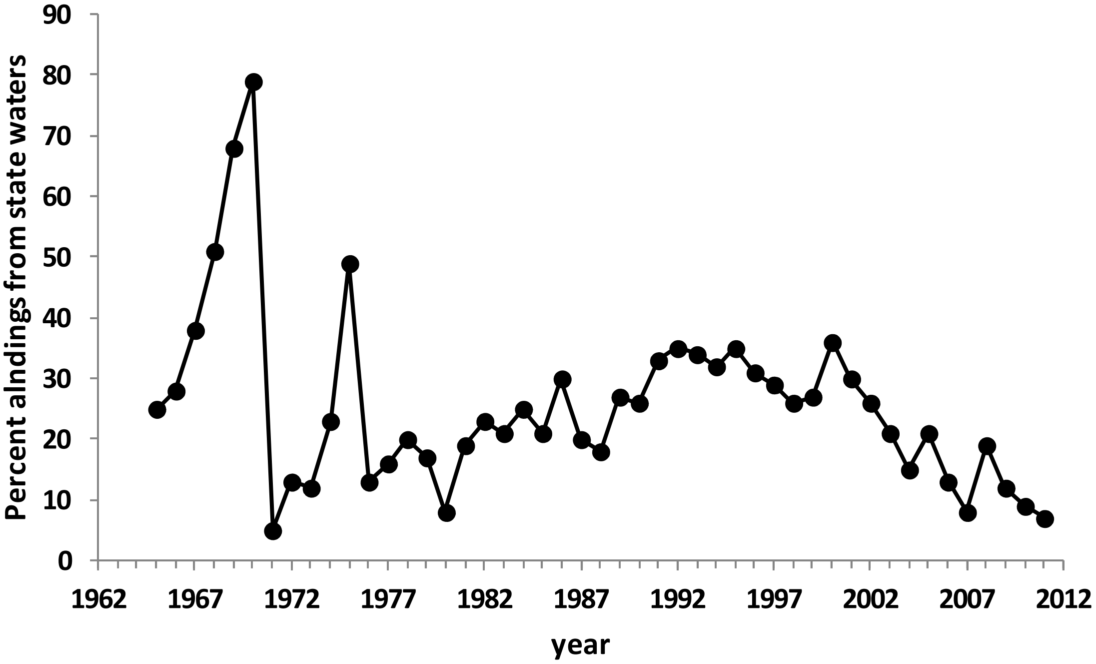
Recently it appears surfclams in New York and New Jersey have been unable to resupply their aging populations with new recruits. This could be happening because there is not enough successful spawning occurring and the supply of larvae is not there, or because smaller surfclams are dying before they are available to a survey or commercial dredge.

In New Jersey, grab sample data collected regularly since 1994 from the area of the survey show that juvenile surfclams are setting successfully out of the plankton (Figure 14). Some years have been better than others with occasional larger sets such as the ones seen in 2005 and 2009, a typical pattern for bivalve recruitment. This data does not show any downward trend in juvenile surfclams that might explain the decline in older surfclams of fishable size.

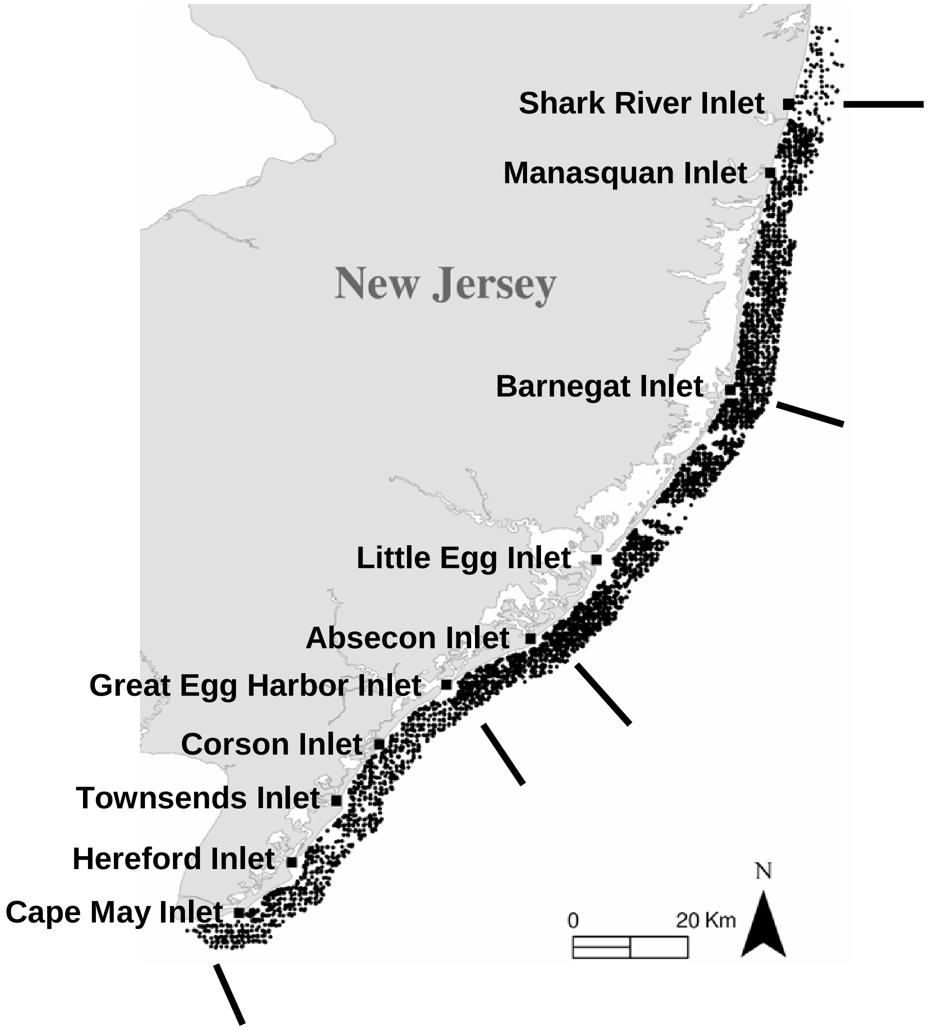
Surfclam age frequencies from the New York surveys in 2002, 2005, 2006 and 2008 (Figure 15) show that surfclams of all ages are present with recognizable ~1996, ~1991 and ~1988 year classes which can be followed. The 2008 data also reflect the recent recruitment seen in the survey size frequencies in both New York and New Jersey. Age data from the Long Island region of the NEFSC survey are not available, but recognizable year classes seen in the New Jersey region included one in 1992.

Length-at-age data from the New York surveys (figure 16) indicate there was no significant change in growth rate from 2002 through 2008, but all regions and strata were lumped together so spatial changes may be masked.

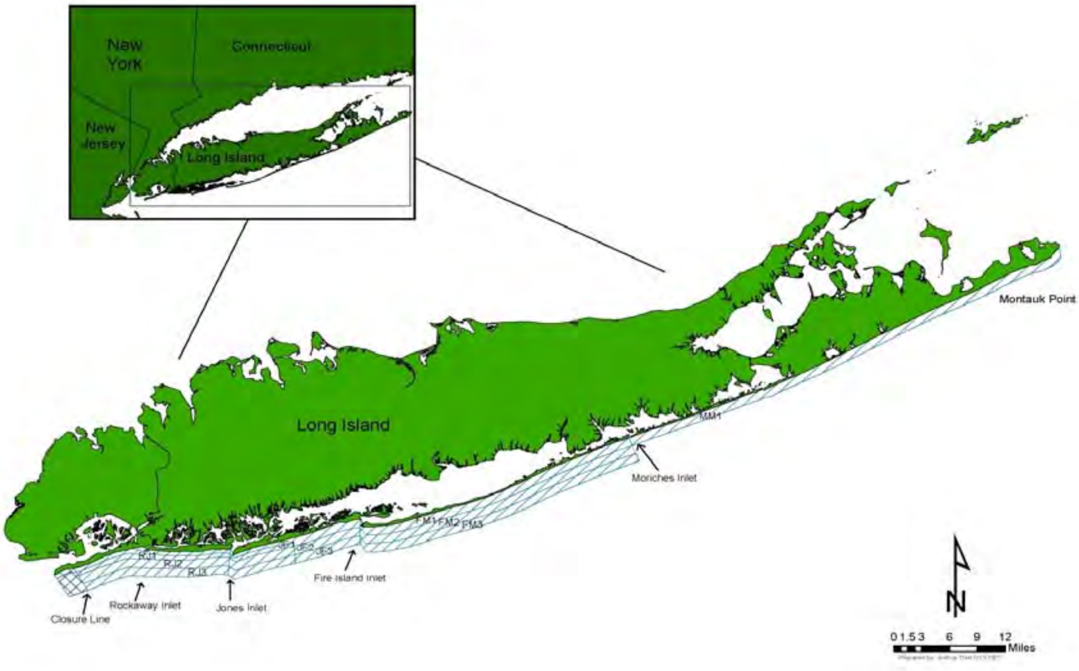
Exploitation rates (landings / survey abundance) were calculated for surfclams in both NJ and NY state waters (Figure 17). The data suggest that exploitation rates in NJ waters decreased from about 4% in 1996 to 2% in 1997-1998 then increased to about 6% in 2002 before falling to zero by 2005 as the fishery for human consumption all but ceased. The limited data for NY indicate that exploitation increased from 2002 to 2008 (landings data were not available for NY in 2012). These simple exploitation rates provide useful information about relative trends in fishing mortality, but they assume all the surfclams in the path of the survey dredge are captured, which is almost never true. The capture efficiency of a clam dredge is almost always less than one, so exploitation rates calculated here for surfclams in state waters are probably overestimated. NJ landings for use as bait were excluded because surfclams for bait are harvested in contaminated areas outside of the survey region.



Appendix A1, Figure 1. Percentage of total surfclam landings that came from state waters, which are mostly New Jersey and New York with small amounts from New England.

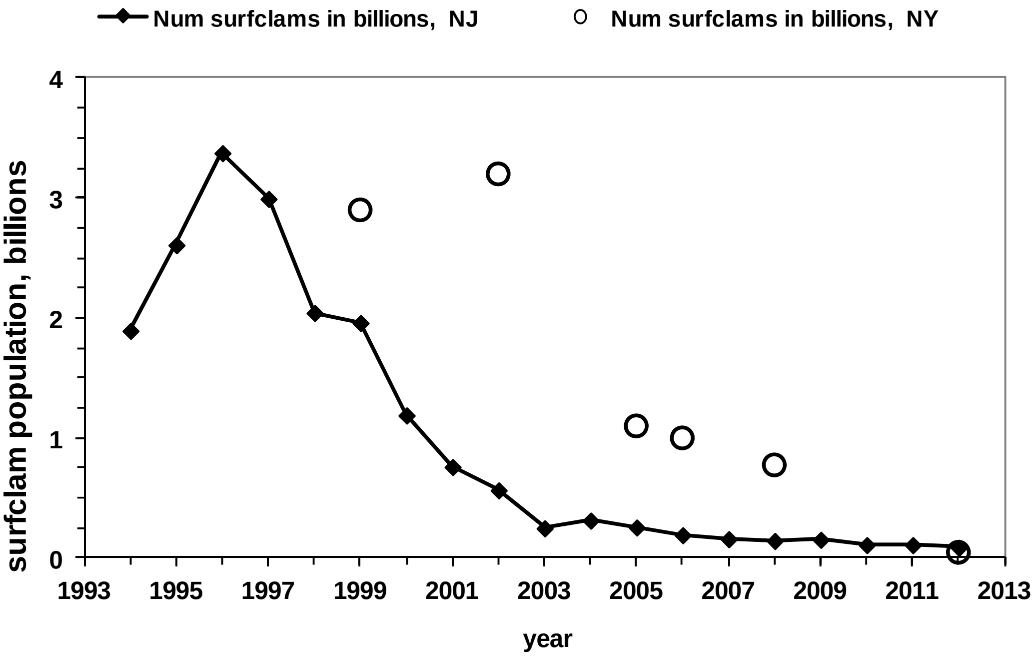


Appendix A1, Figure 2. Map showing the sampling regions for the NJ state survey, and station locations 1988-2008. Within each region there are three along-shore depth strata one mile wide. Map courtesy of Jeff Normant, NJDEP.

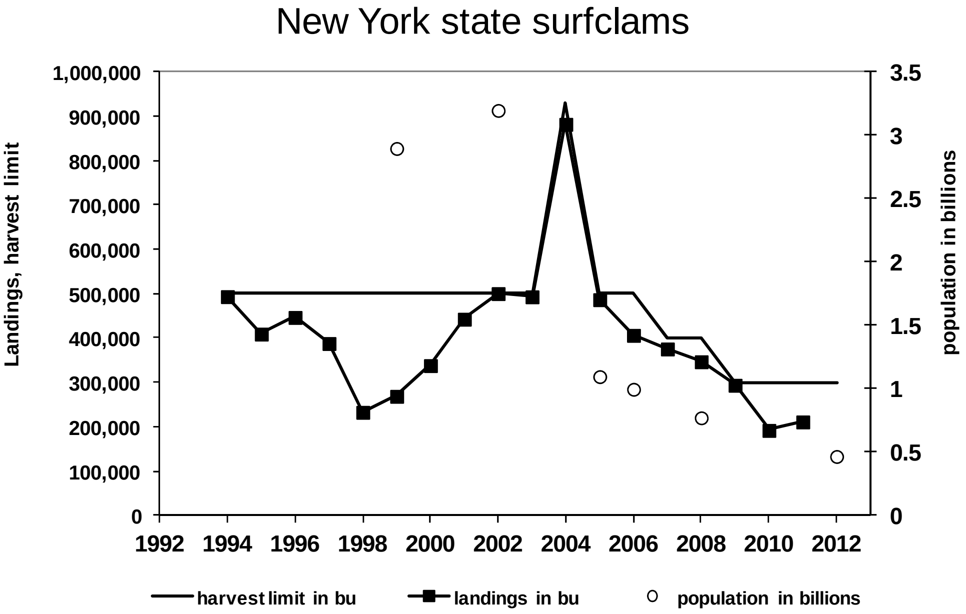


Appendix A1, Figure 3. Map showing New York state sampling regions from west to east: RJ, JF and FM, which each have 3 depth strata, and MM which has one depth stratum. Map courtesy of Wade Carden, NYSDEC.

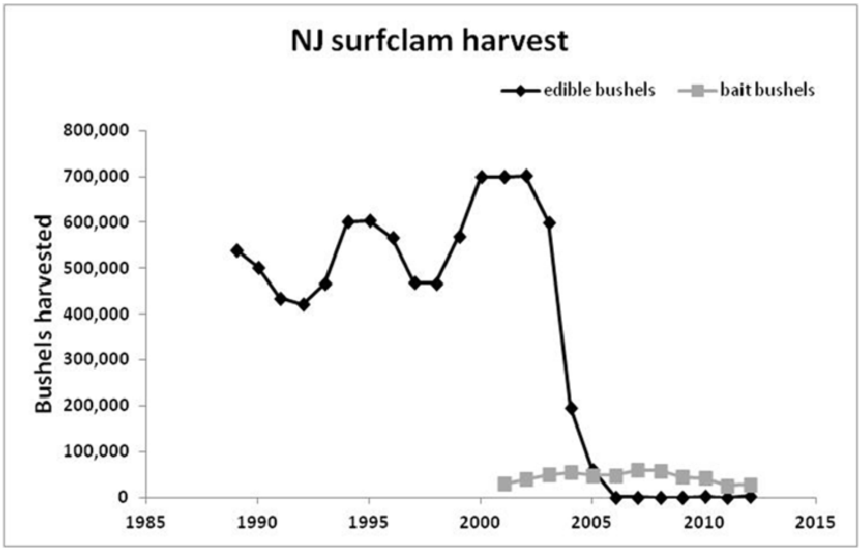
Appendix A1, Figure 4. The inshore commercial clam dredge used for the New York surveys. Photo courtesy of Jeff Normant, NJDEP; William Burton, Versar, Inc.; and Beth Brandreth, USACE.



Appendix A1, Figure 5. Survey-based population estimates for surfclams in New Jersey and New York from years when there was random stratified sampling.

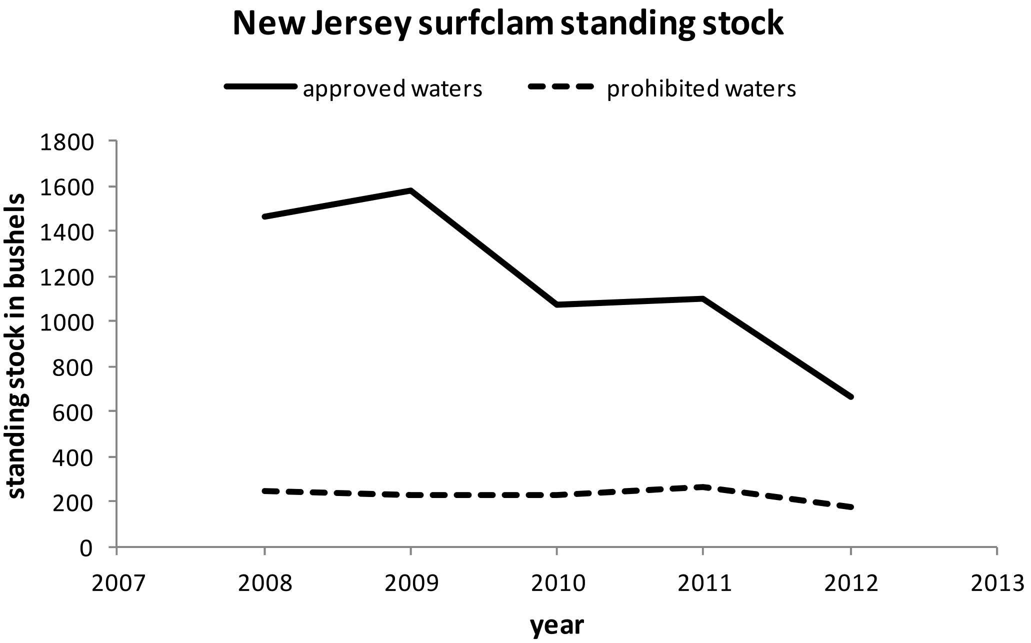


Appendix A1, Figure 6. Landings, harvest limit and population of surfclams in New York state waters. Landings and harvest limit are scaled to the left axis and population is scaled to the right axis. The harvest limit was raised to 890,000 bushels for one year in 2004.

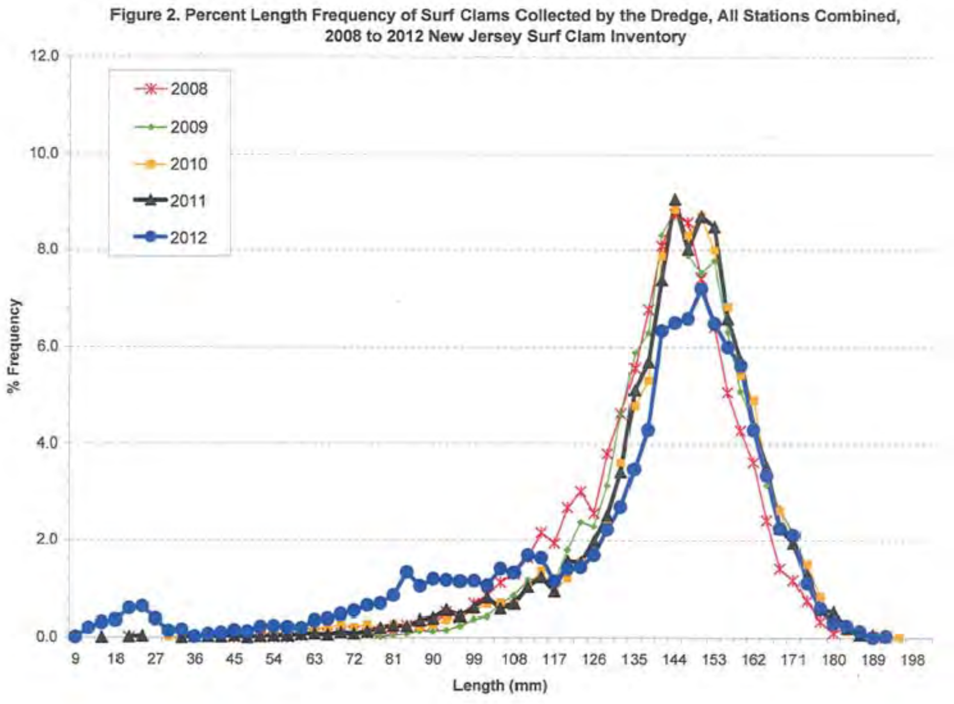


Quota for 2010-2011 season: 55,296 bushels (season OCT –MAY)
Quota for 2011-2012 season: 49,152 bushels

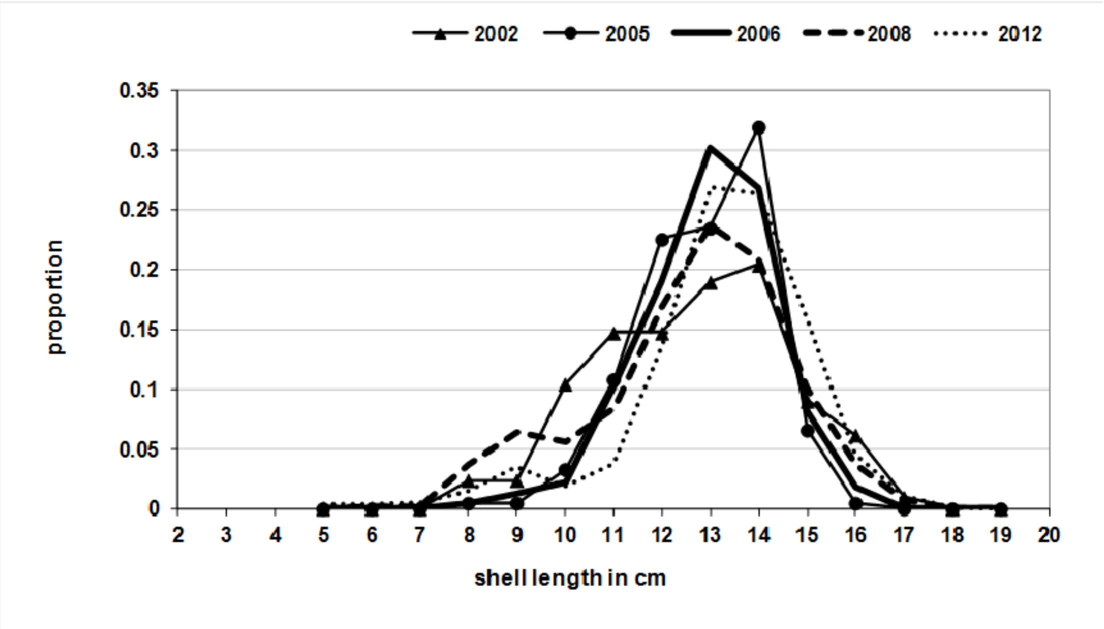
Appendix A1, Figure State - 7. Bushels of surfclams harvested from New Jersey “approved” (surfclams for human consumption) and “prohibited” (surfclams for bait only) waters.



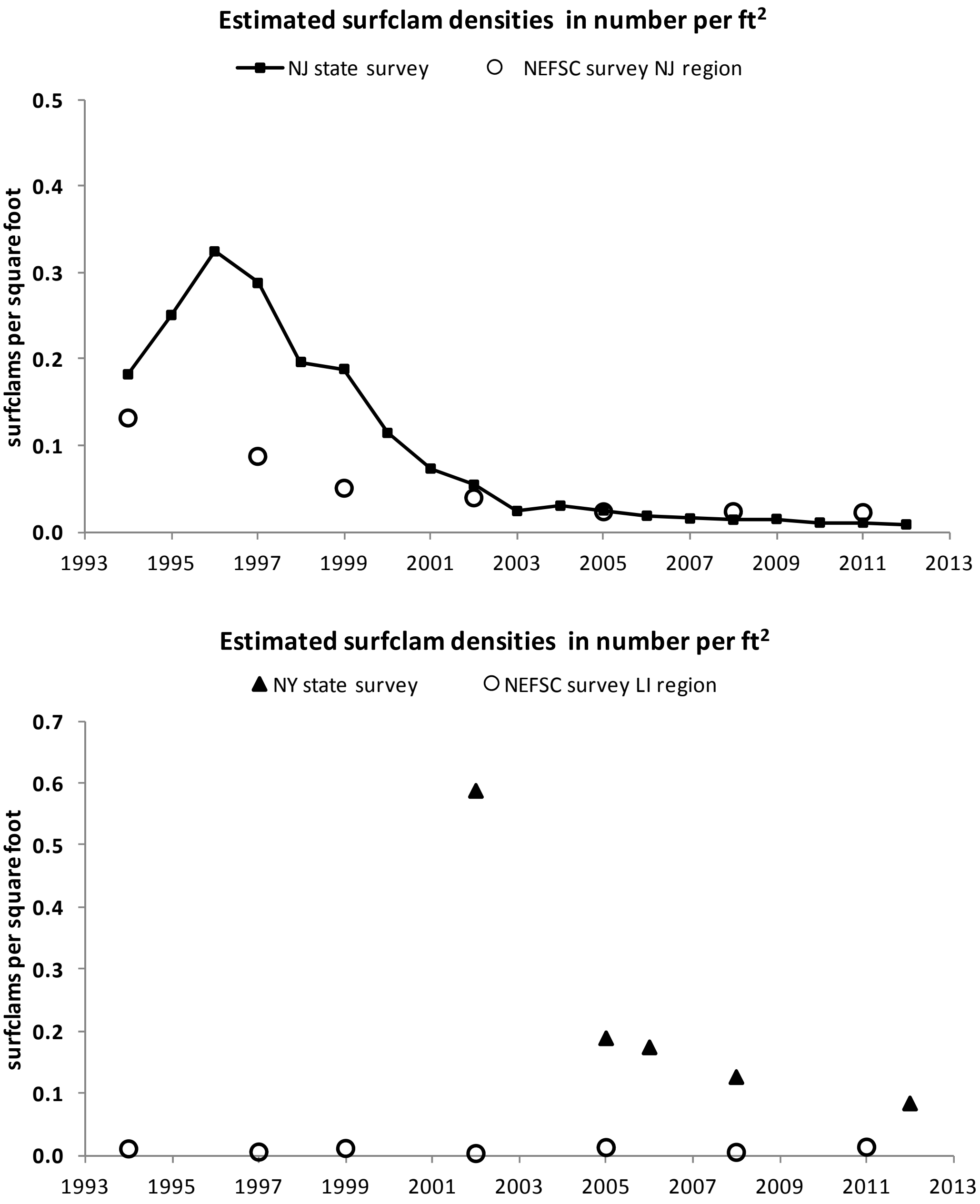
Appendix A1, Figure 8. Standing stock in industry bushels from New Jersey state waters. Clams from approved waters can be sold for human consumption, while clams from prohibited waters are sold for bait only.



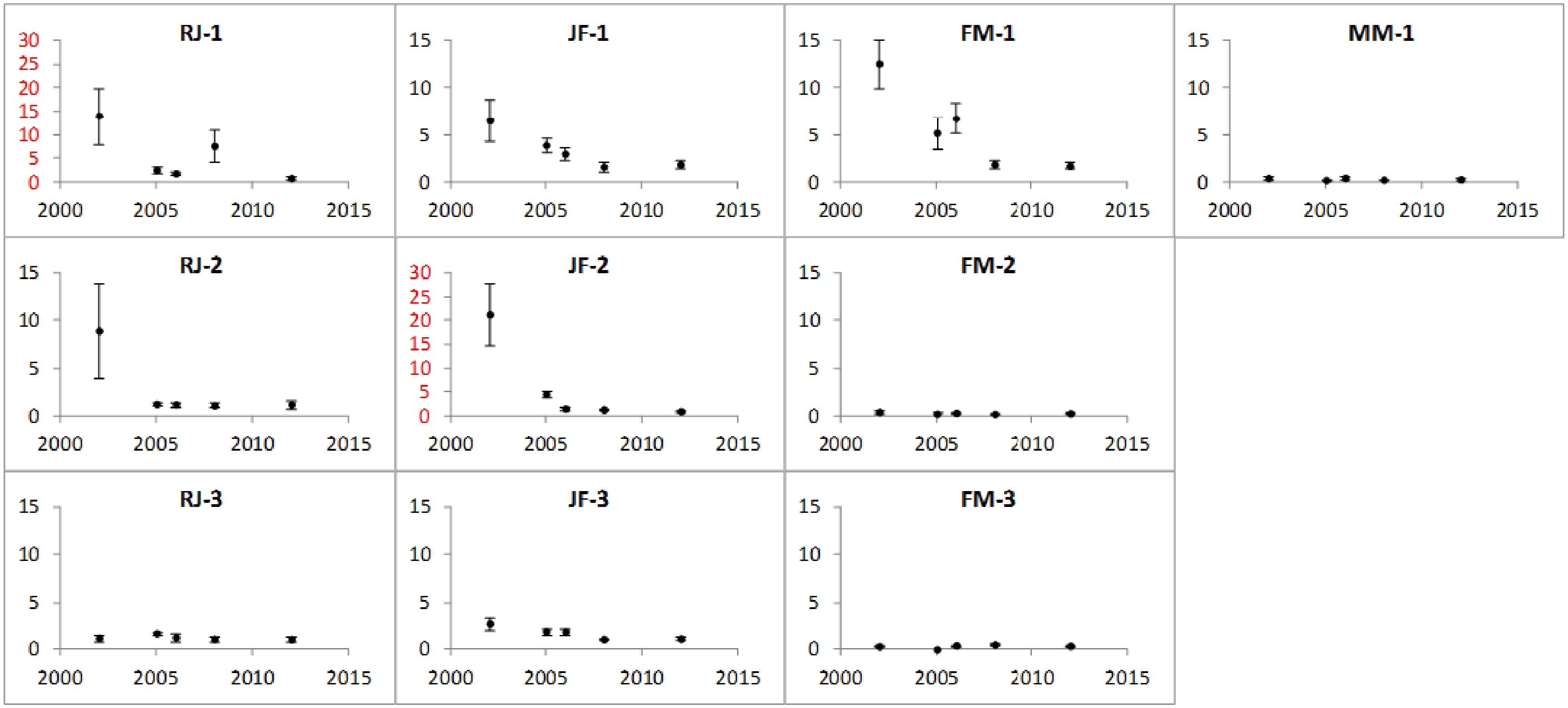
Appendix A1, Figure 9. Length frequencies from the 2008-2012 annual New Jersey state surfclam surveys. Figure courtesy of Jeff Normant, NJDEP.



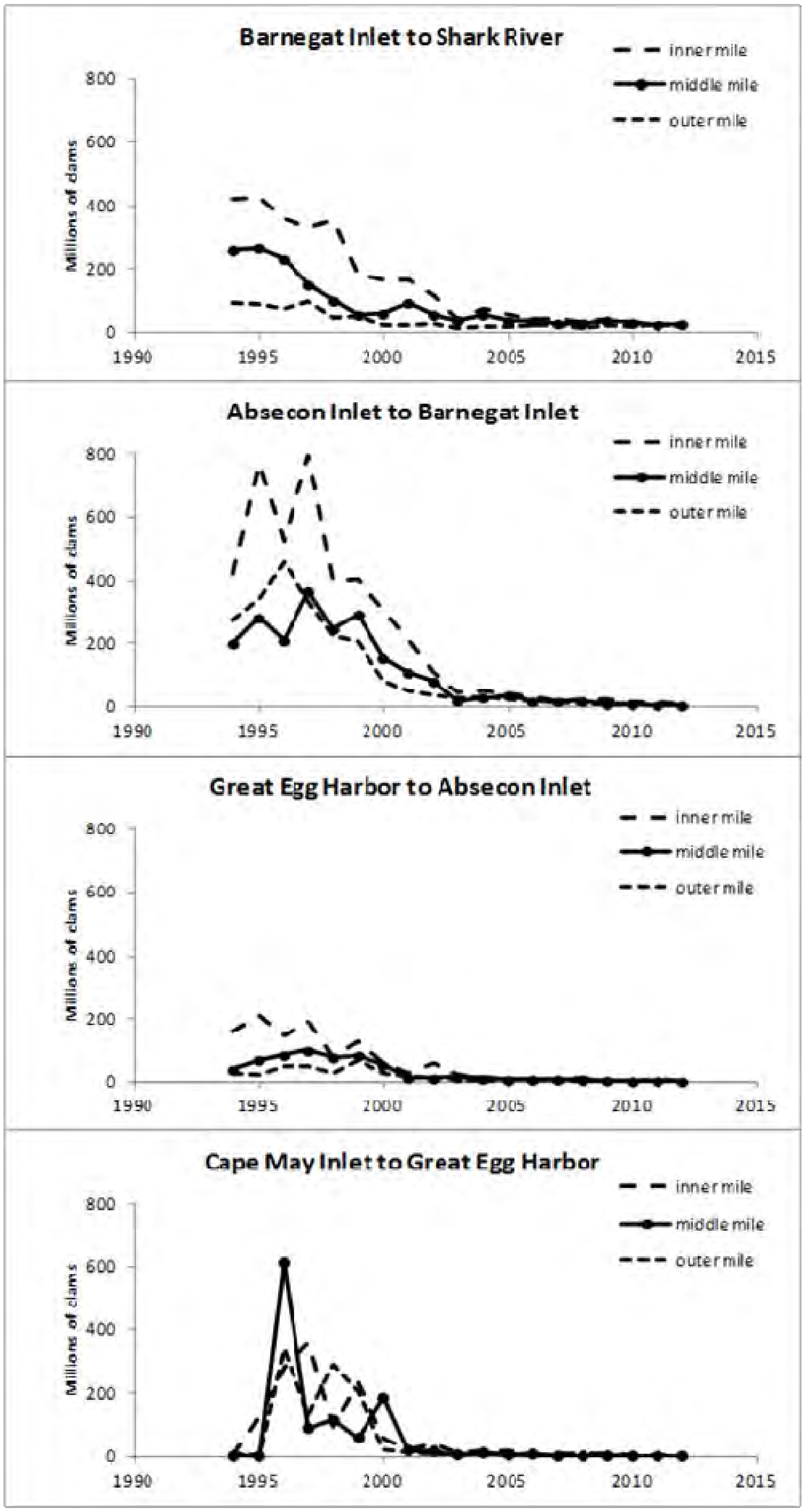
Appendix A1, Figure 10. Length frequencies from the 2002, 2005, 2006, 2008 and 2012 New York state surfclam surveys.



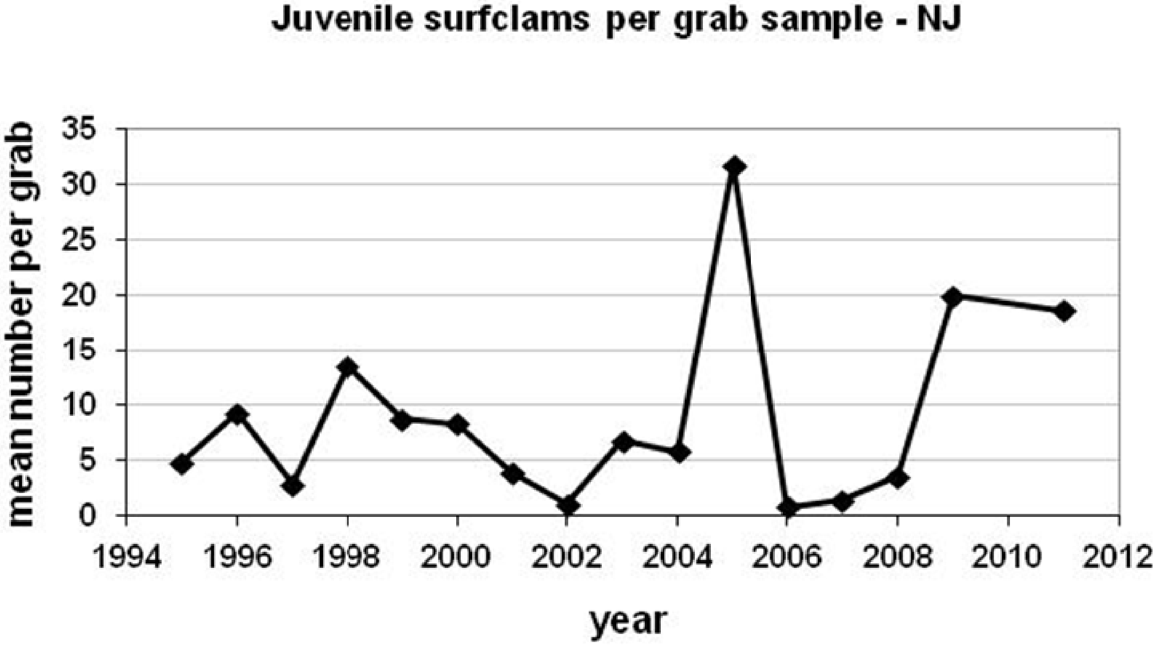
Appendix A1, Figure 11. A rough comparison of surfclam density estimates (total estimated number of clams over the area surveyed in square feet) from the NJ State survey and the NJ region of the NEFSC survey in federal waters (top) and the NY state survey and LI region of the NEFSC survey in federal waters (top). All sizes of clams were included, and an adjustment was made to the NEFSC data to account for a dredge efficiency of 0.256. No adjustments were made to the NY or NJ data. The comparisons are approximate due to differences in dredge design, capture efficiency and size selectivity



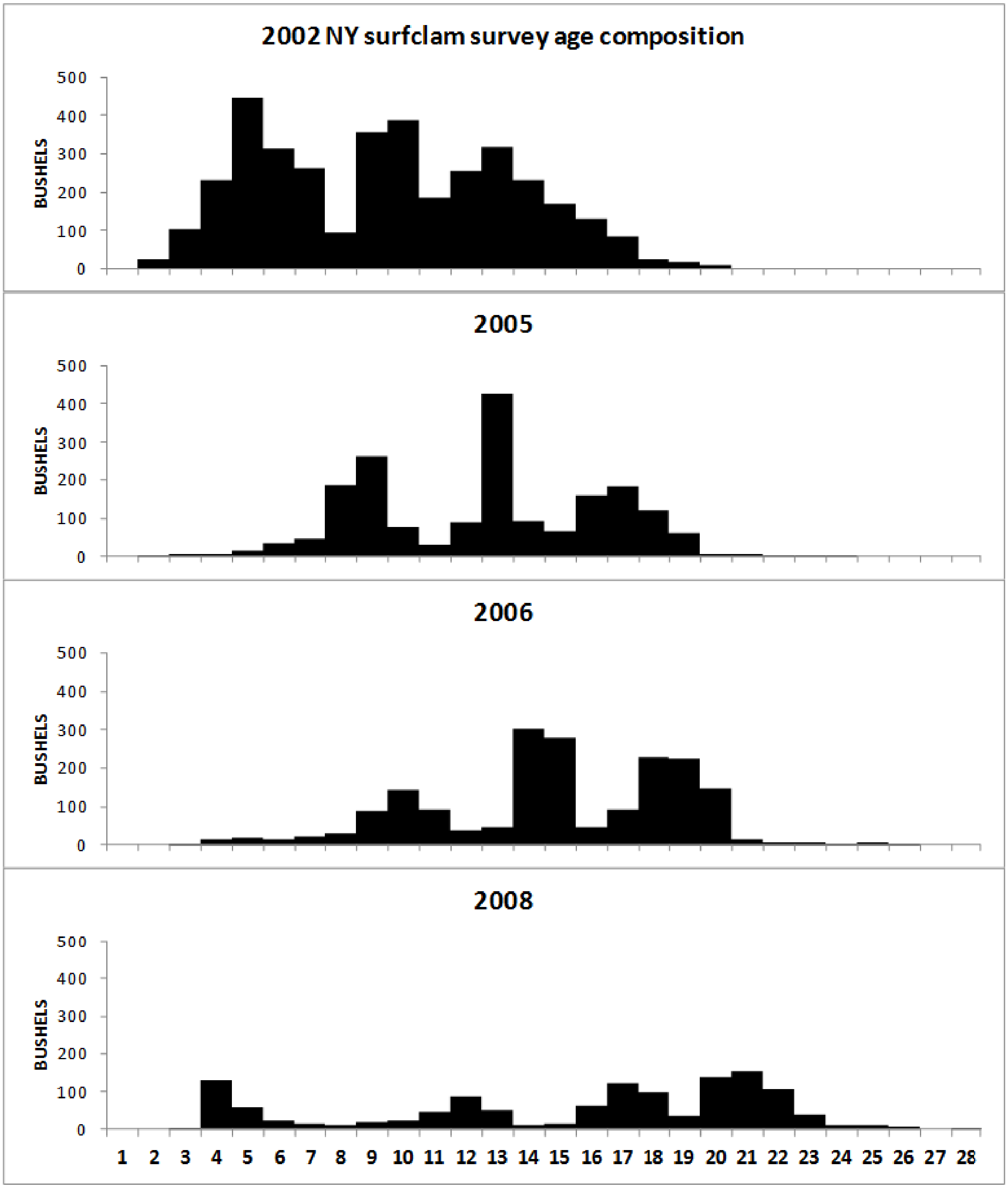
Appendix A1, Figure 12. New York State Surfclam Survey - Estimated density of clams, in individuals per m^2, per stratum by survey year. Strata cover the waters off the south side of Long Island. Plots are laid out in order with the left plots representing the westernmost strata, which are broken down into inner, middle and outer miles (numbers 1-3), covering the three-mile limit of State waters. The easternmost stratum has only the inner substratum. RJ = Rockaway Inlet to Jones Inlet, JF = Jones Inlet to Fire Island Inlet, FM = Fire Island Inlet to Moriches Inlet, MM = Moriches Inlet to Montauk Point.



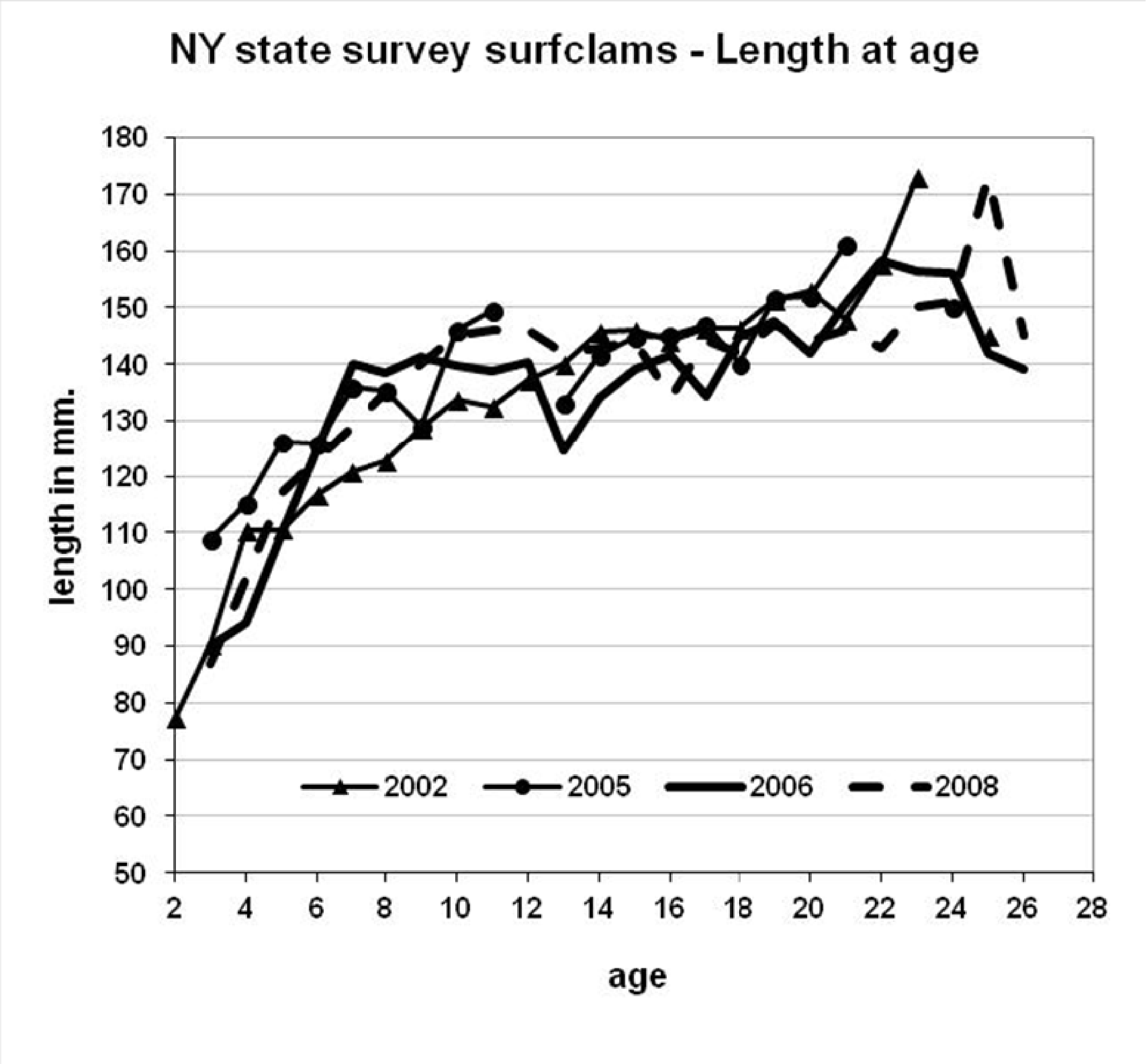
Appendix A1, Figure 13. New Jersey State survey - estimated number of clams per stratum by survey year. Plots are laid out in order with the top plot representing the northernmost stratum. Strata are further broken down into inner, middle and outer miles, covering the three-mile limit of State waters.



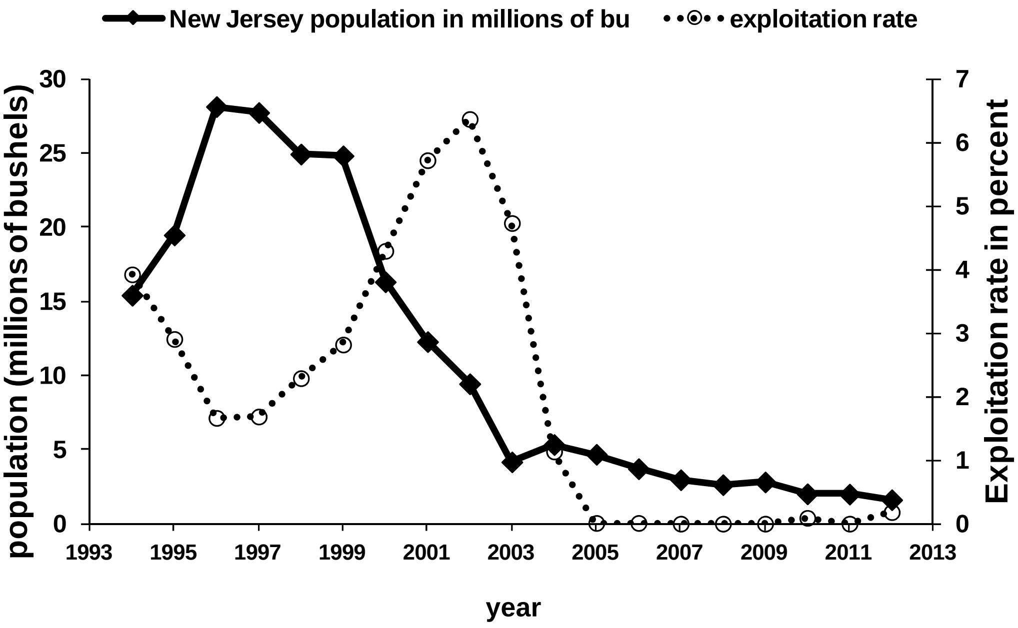
Appendix A1, Figure 14. As part of the annual survey, the state of New Jersey takes sediment grab samples, which contain recently settled juvenile surfclams. The clams are generally less than 10mm. About 300 grabs are taken every survey, and the area sampled is 1/10 of a square meter.

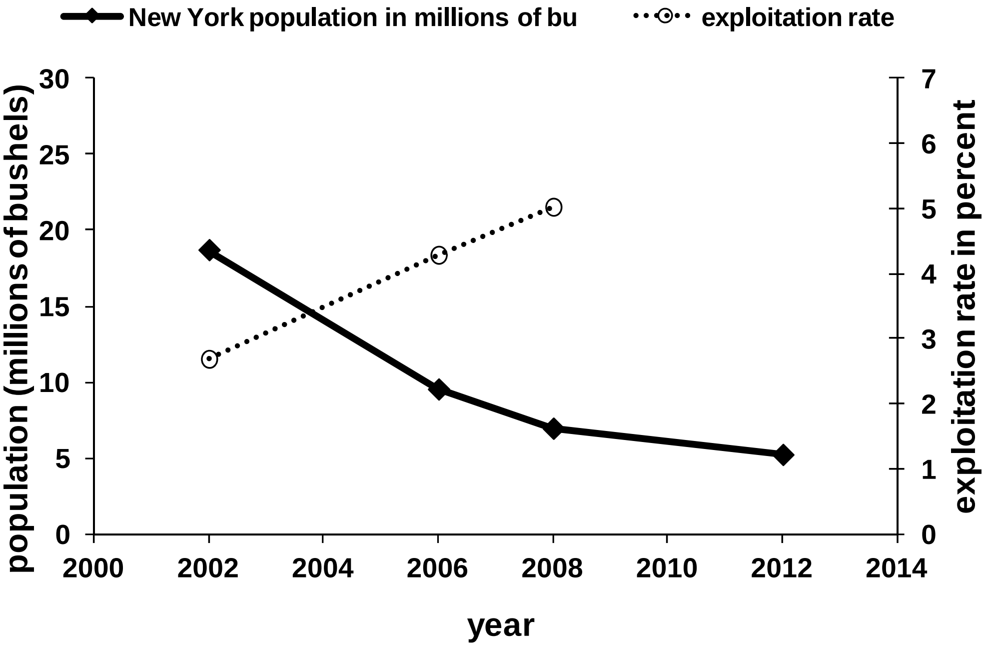


Appendix A1, Figure 15. Age compositions from the 2002, 2005, 2006 and 2008 New York State surfclam surveys, in bushels at age.



Appendix A1, Figure 16. Surfclam length at age from the 2002, 2005, 2006 and 2008 New York State surveys.

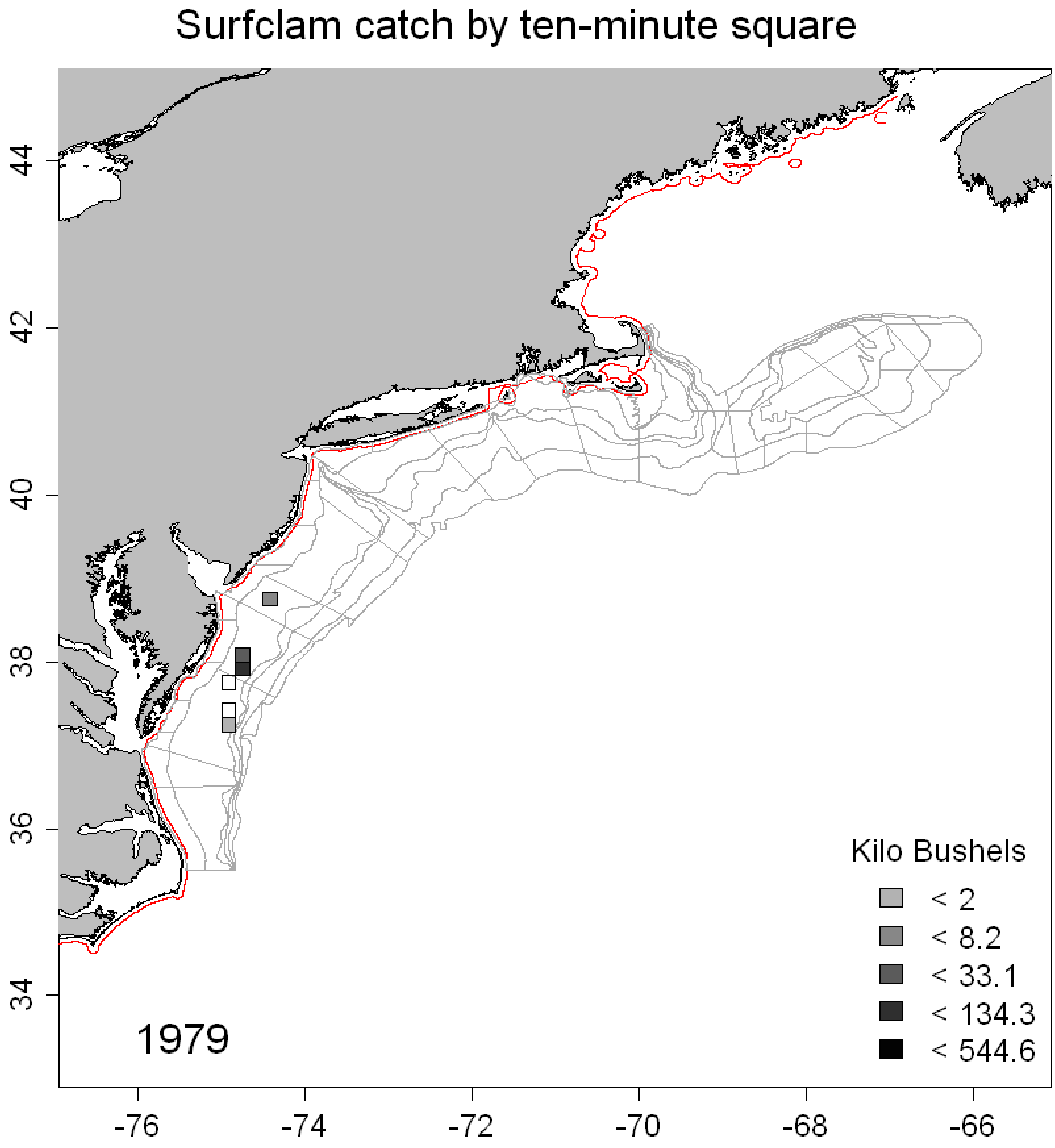


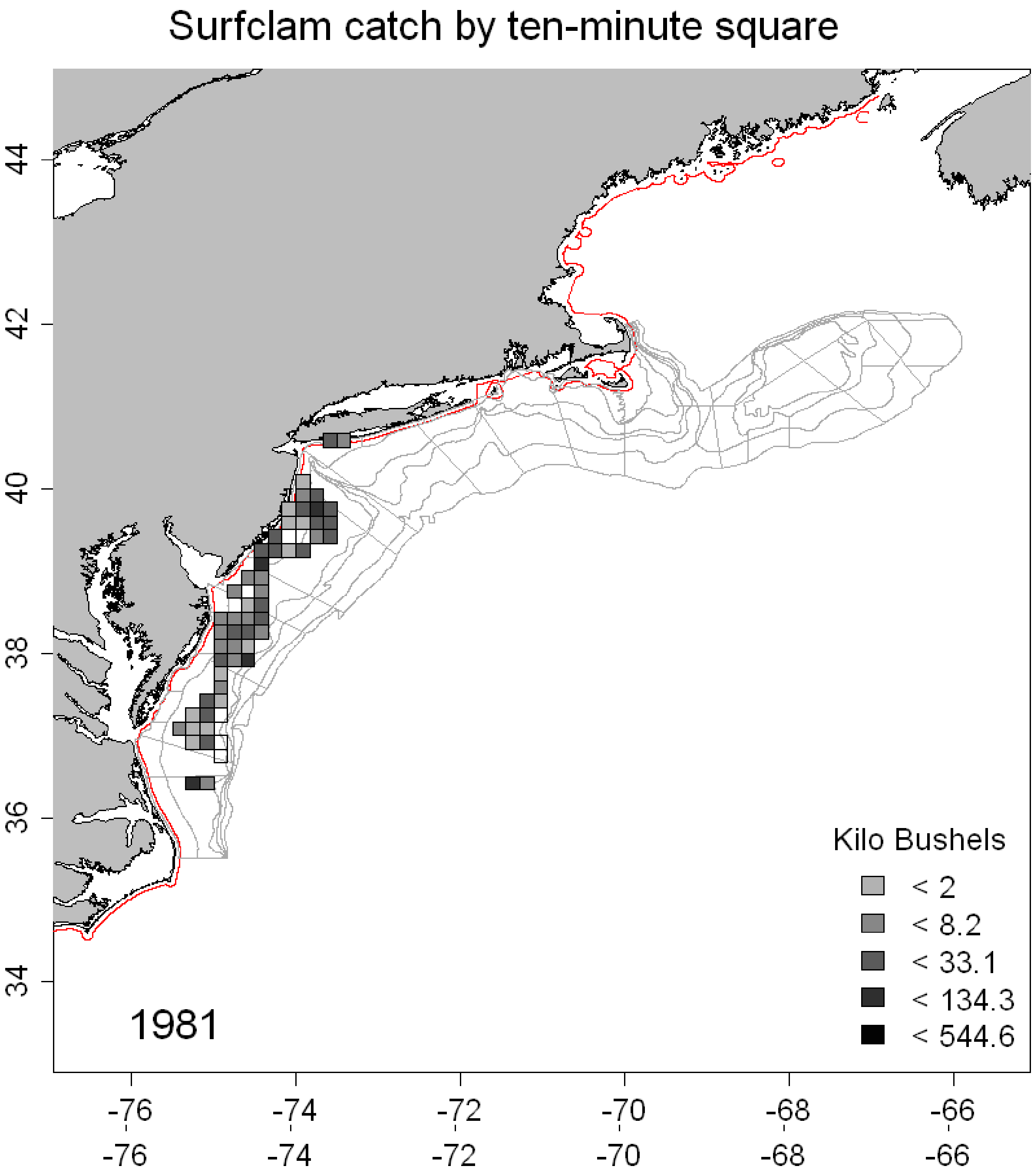


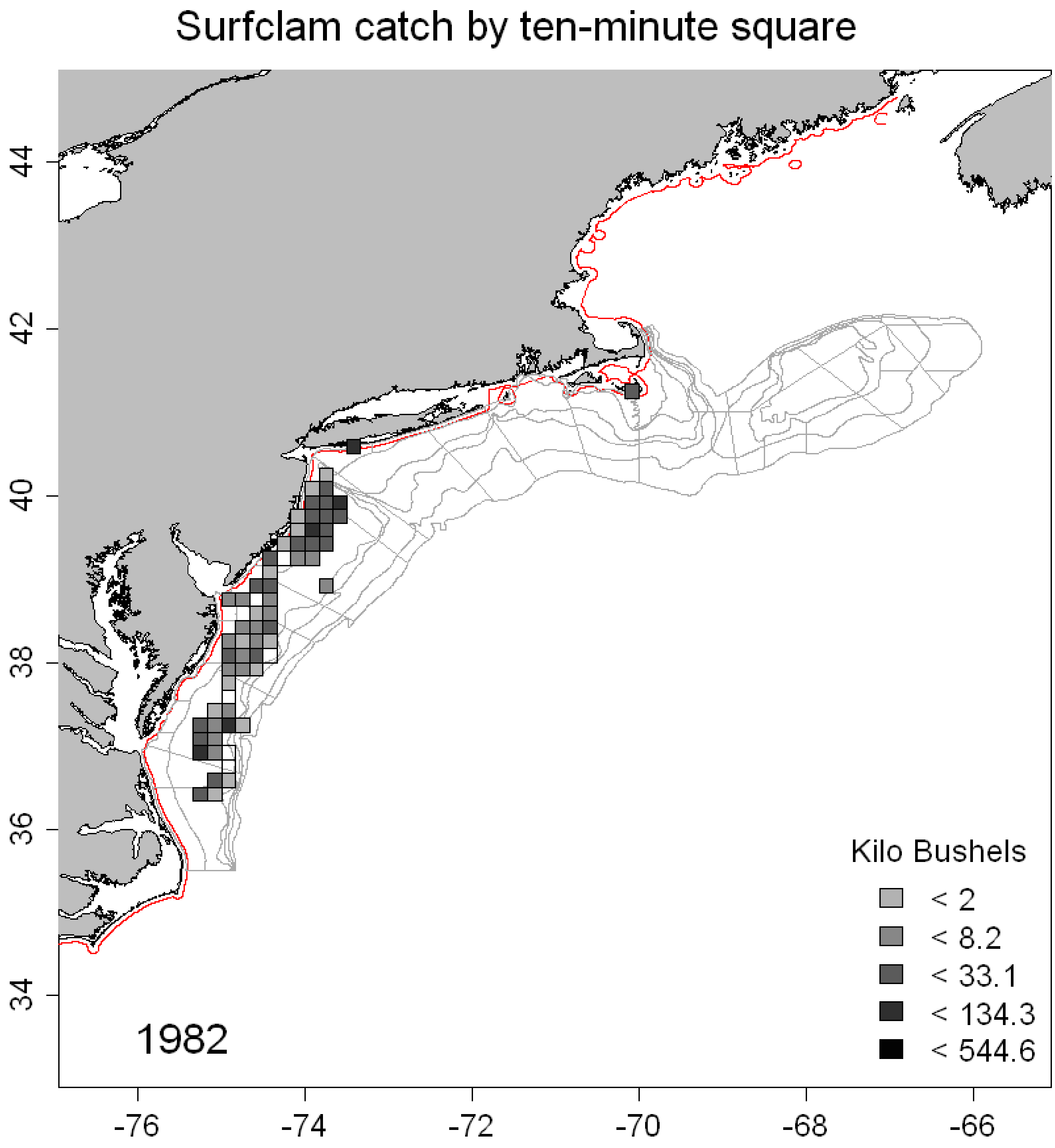
Appendix A1, Figure 17. Exploitation rates (expressed as landings as a percentage of estimated biomass) and population biomass for New Jersey (top) and New York state surfclams.

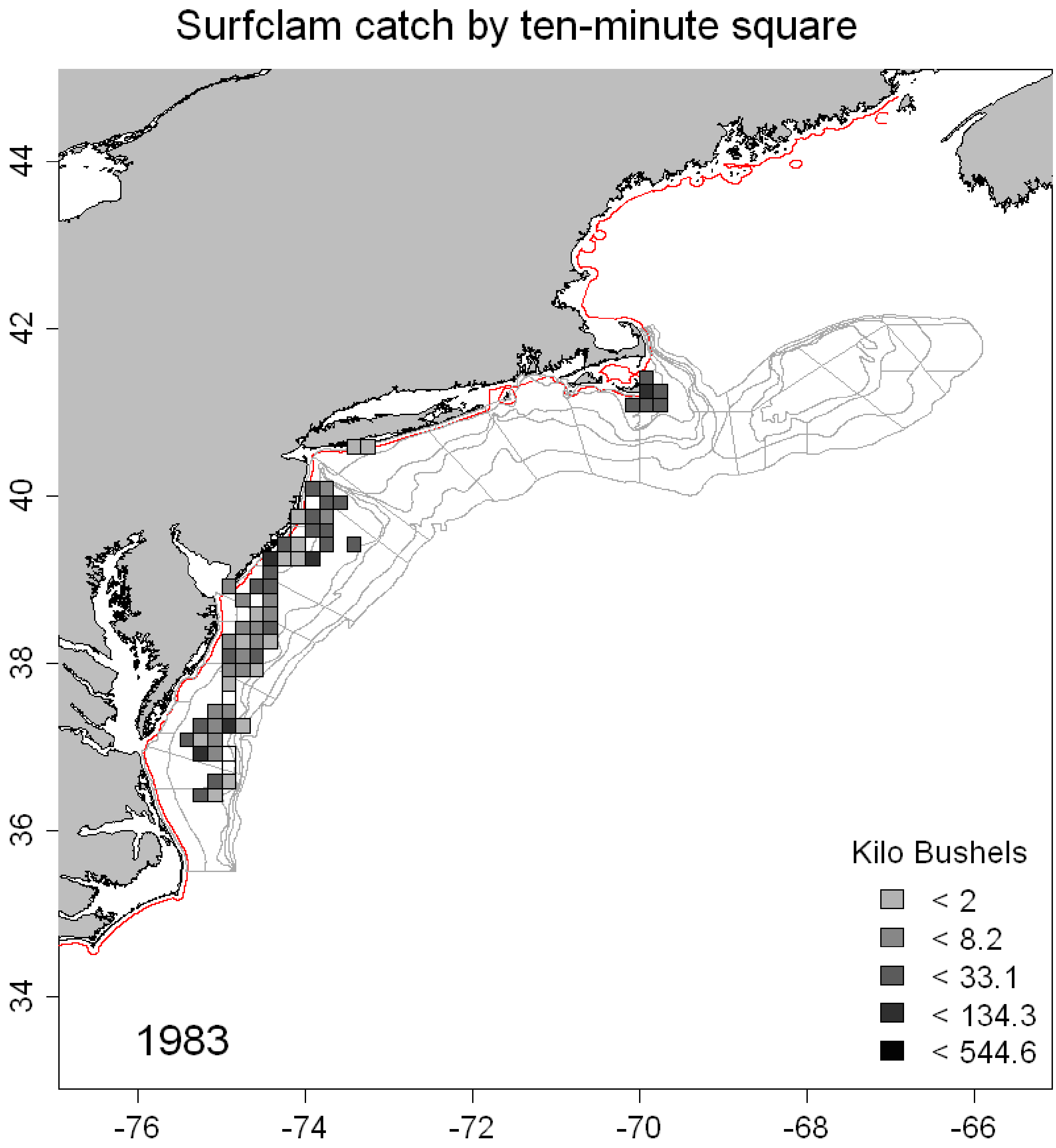
Appendix A2: Maps of commercial harvest through time

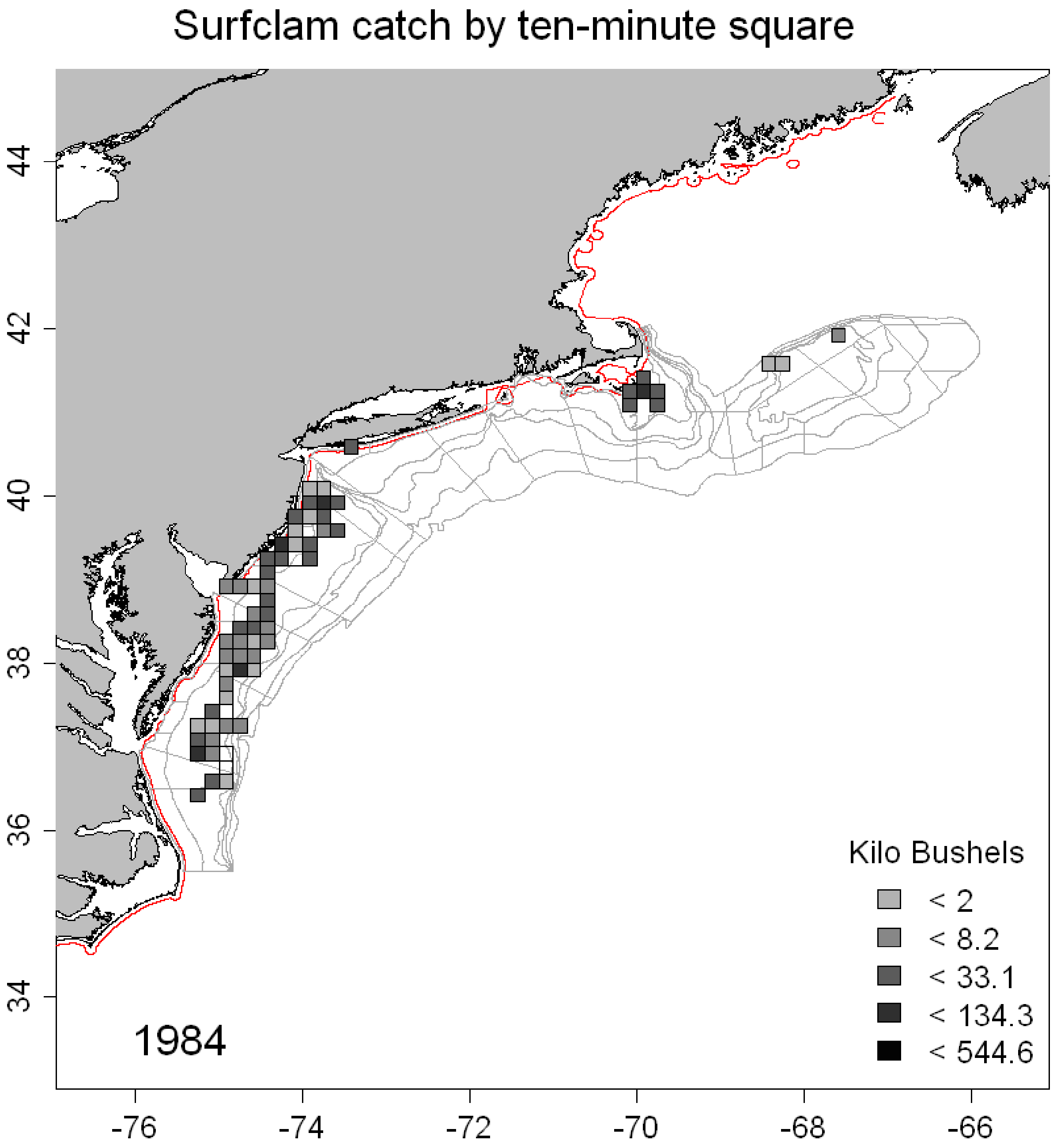
Appendix A2, Figure 1. Landings, time fished and LPUE by ten-minute square from 1979 – 2011 (Following pages).

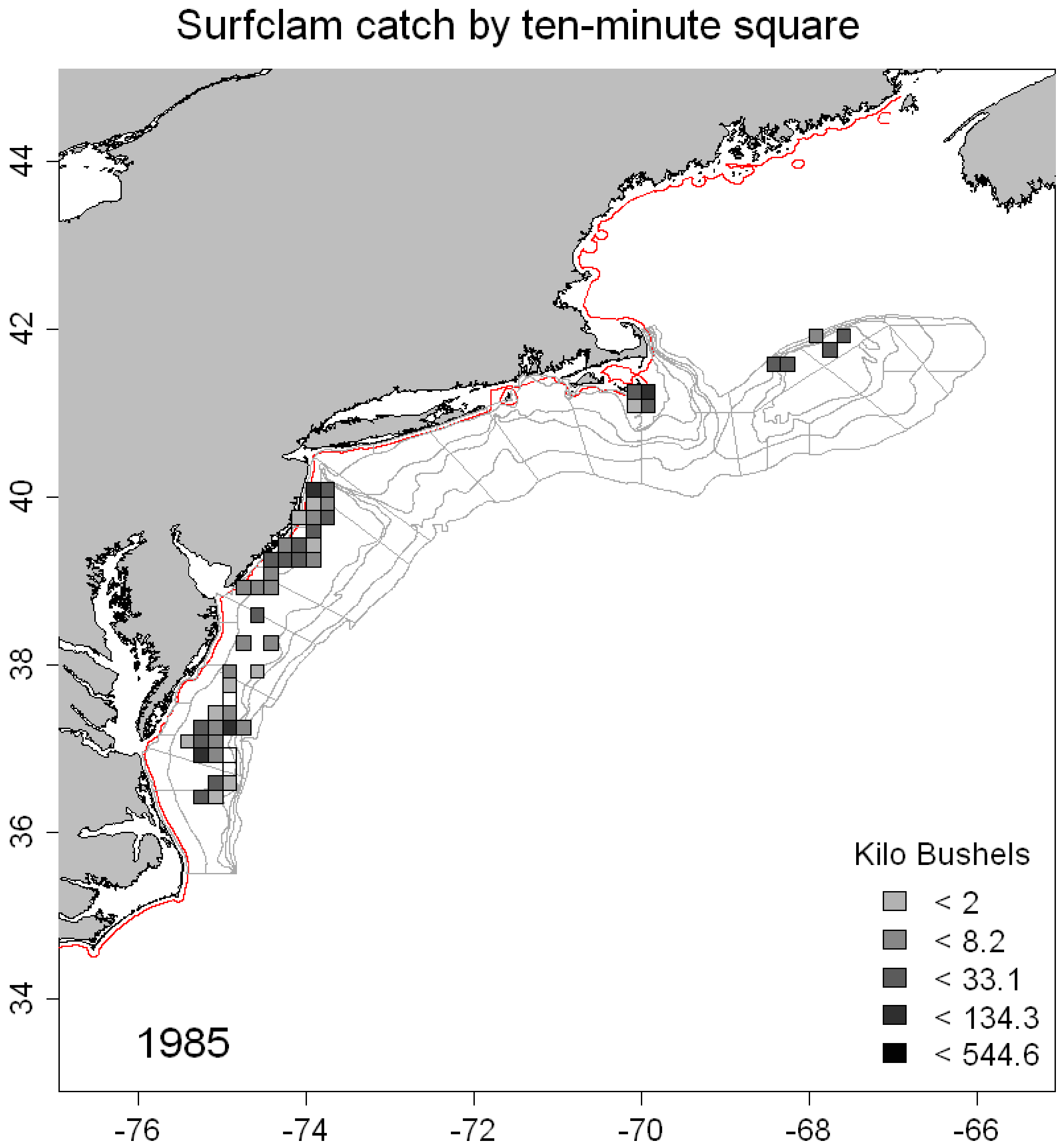



Surfclam catch by ten-minute square
44
42
40
38
36
34
1981
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
-76
-74
-72
-70
-68
-66
-76
-74
-72
-70
-68
-66

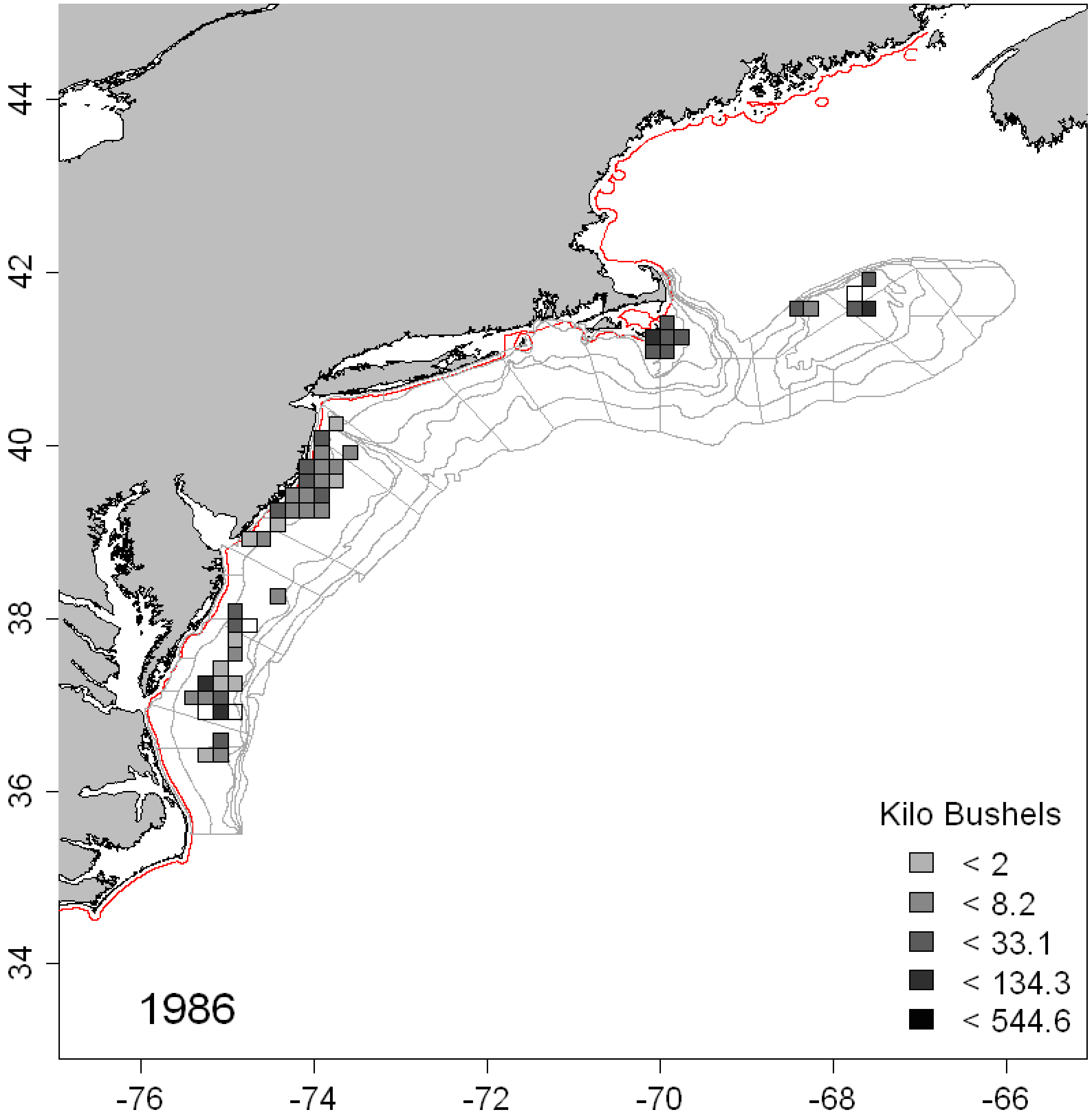

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1982


Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1983

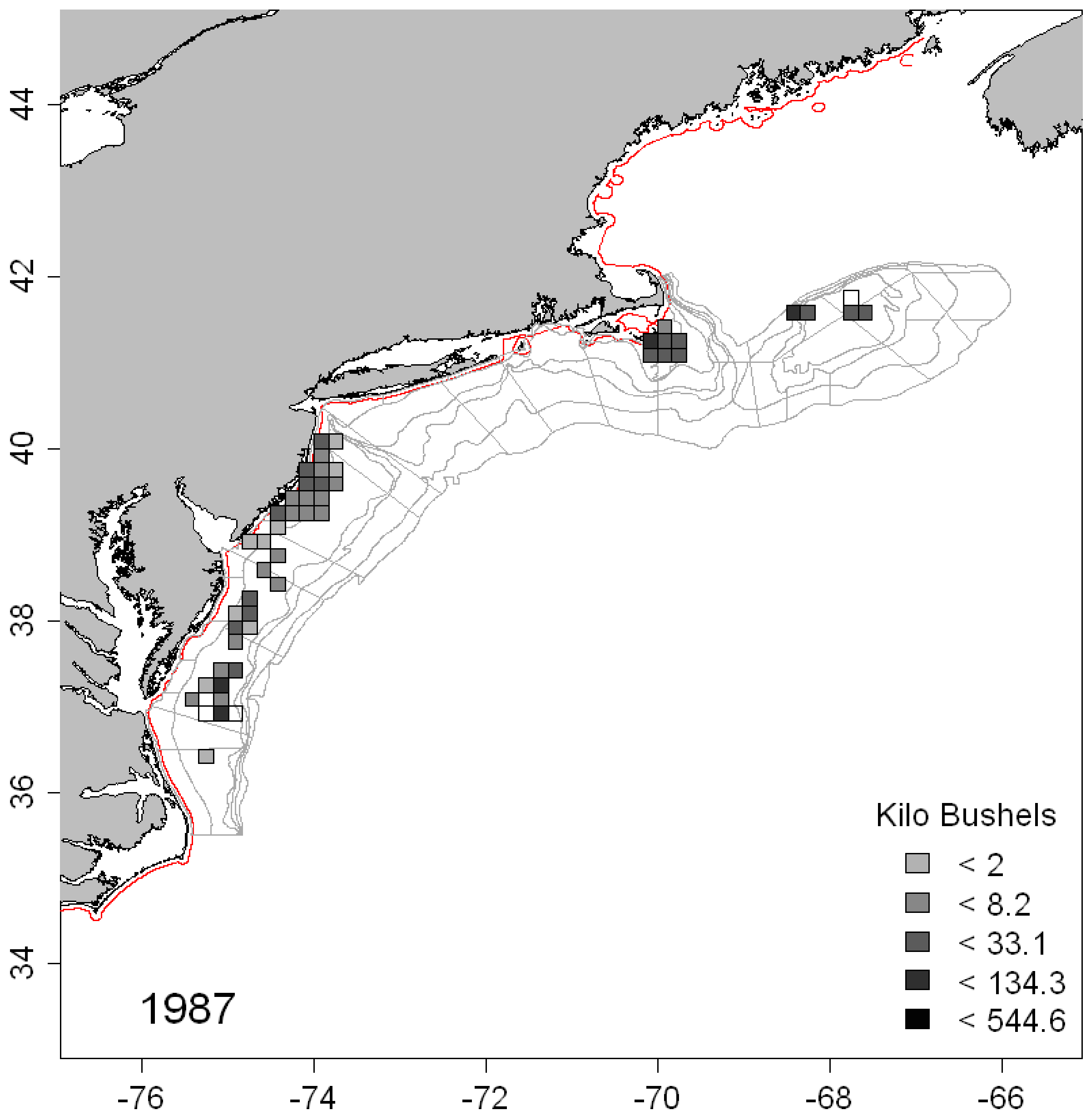

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1984

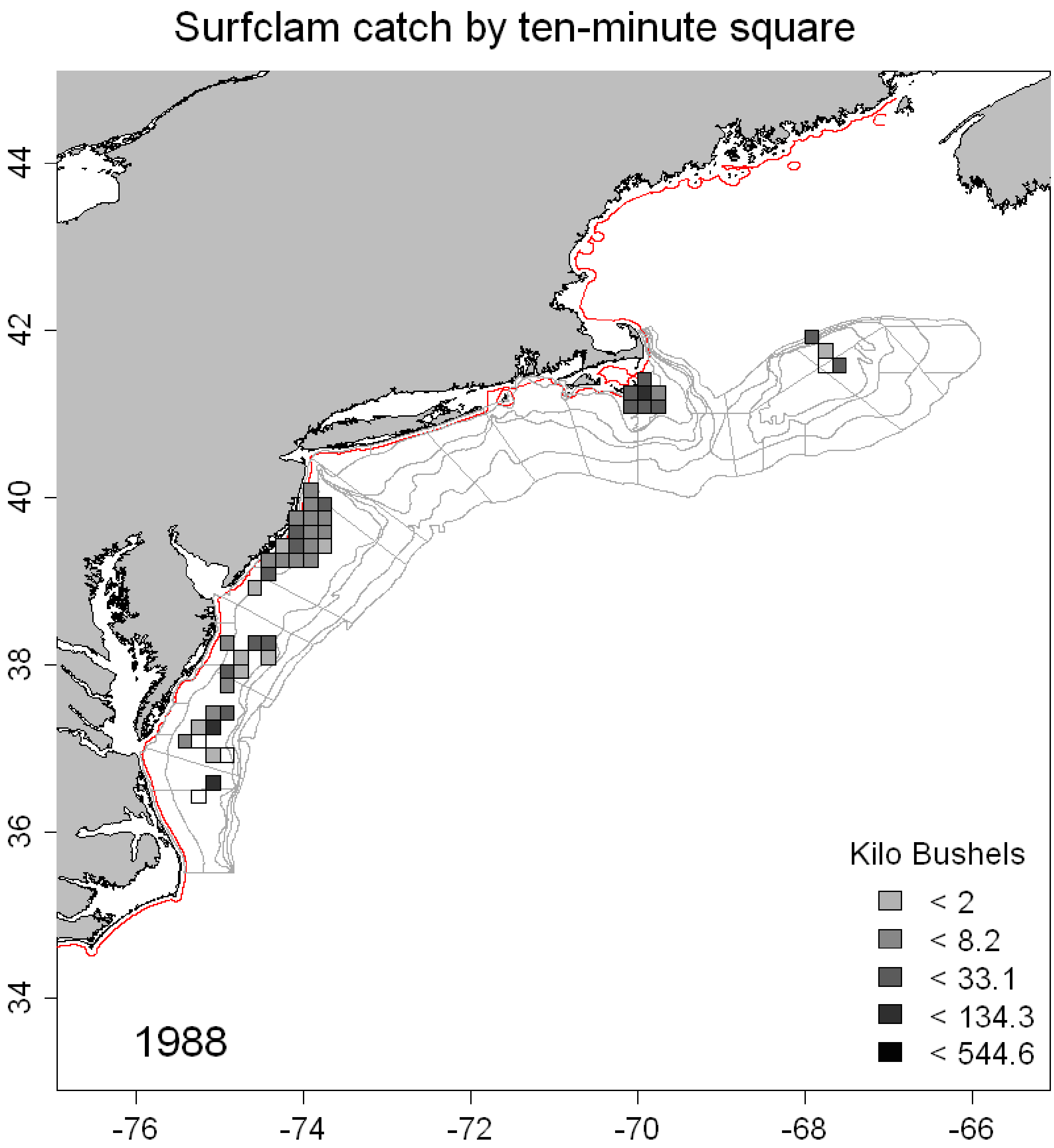

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
1985
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

Surfclam catch by ten-minute square

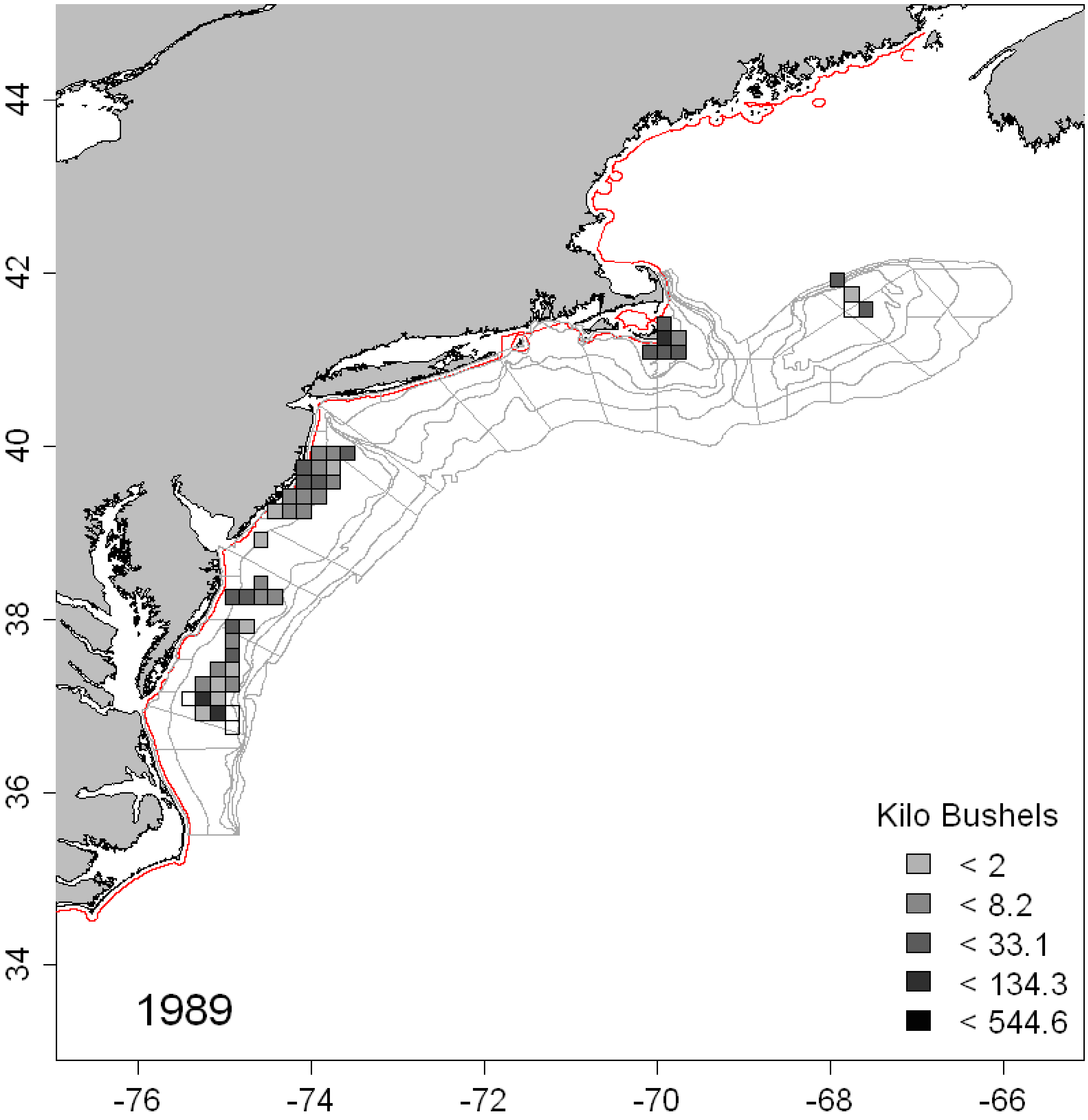


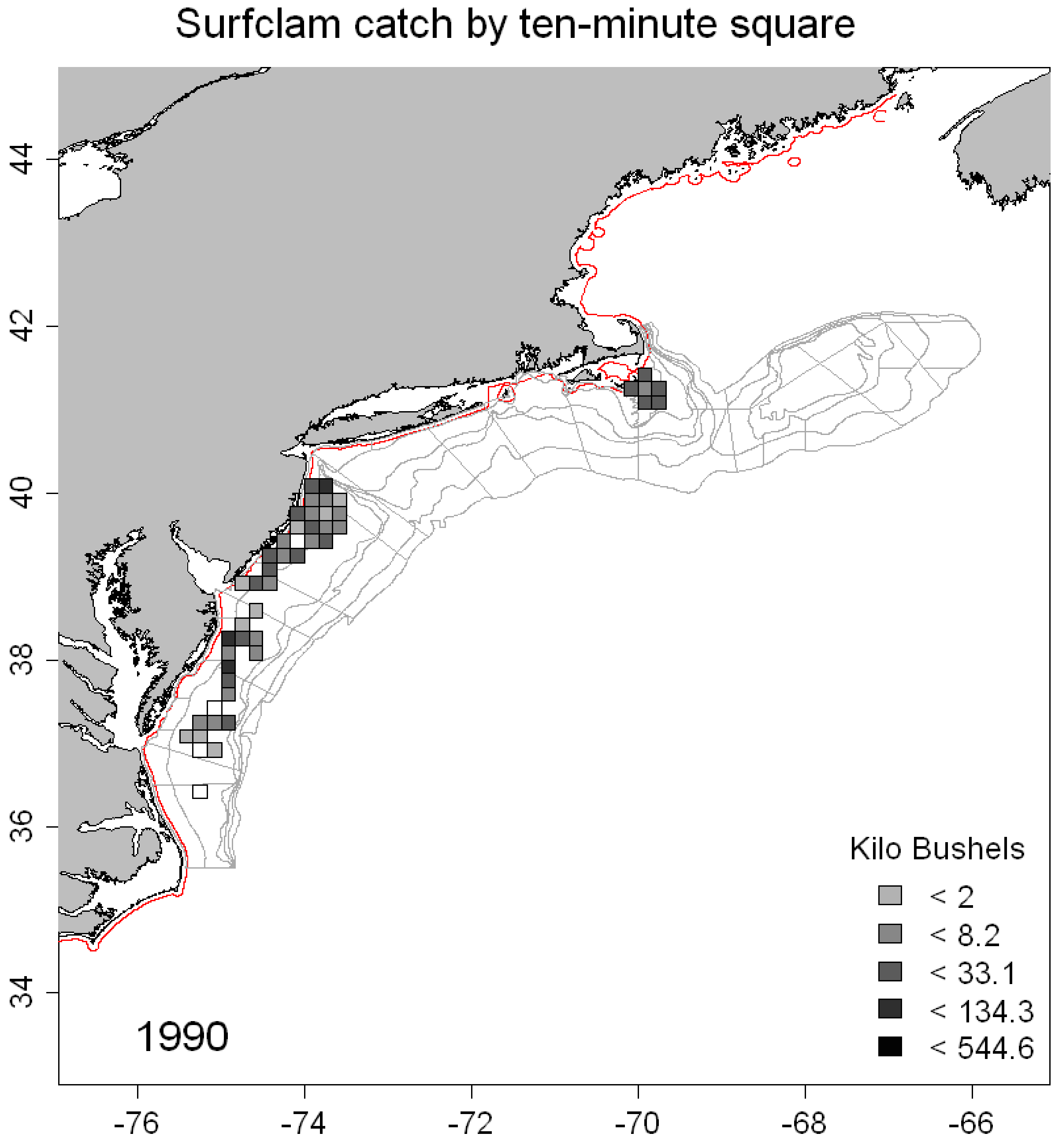
Surfclam catch by ten-minute square

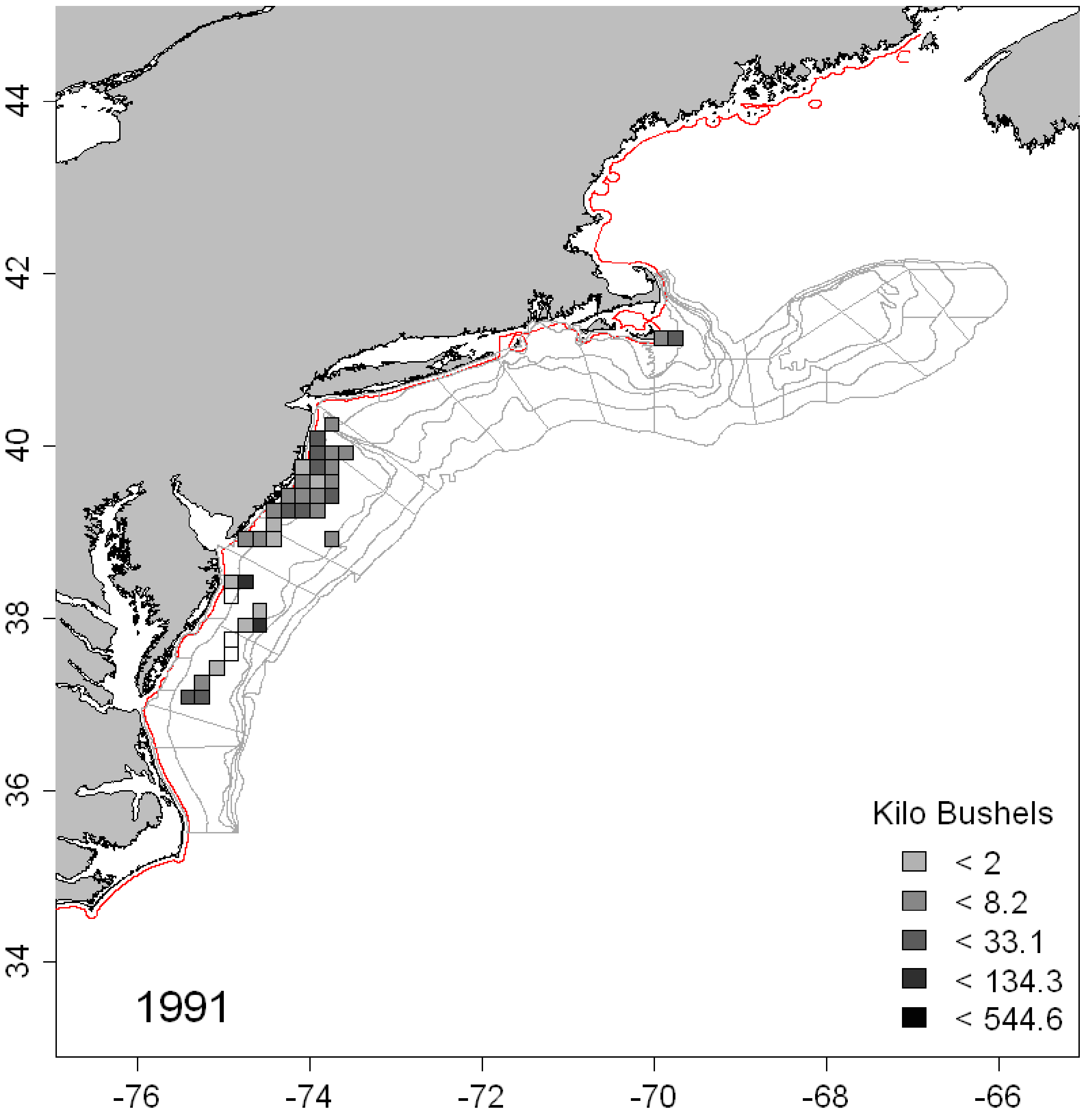



Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1988

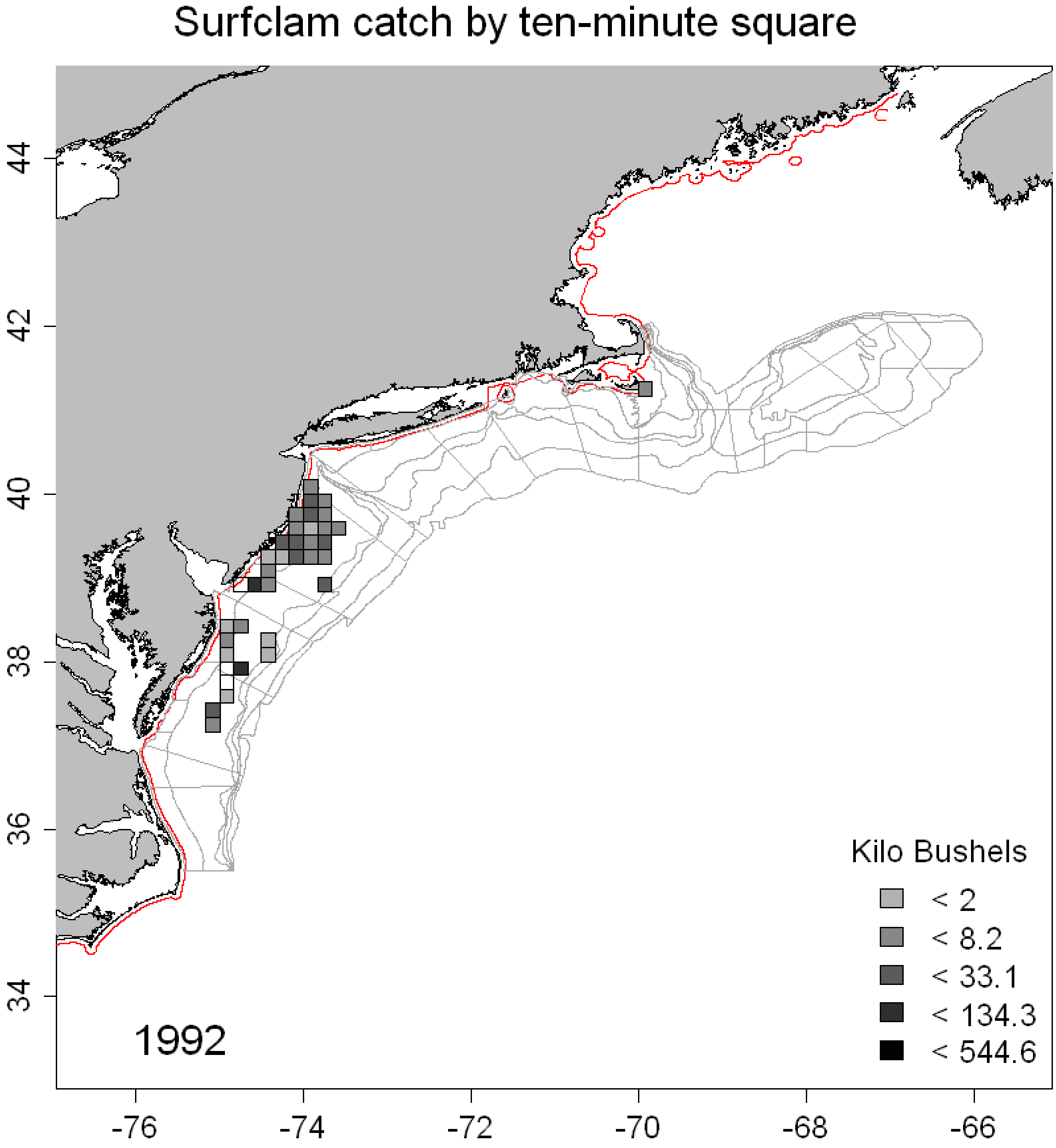
Surfclam catch by ten-minute square



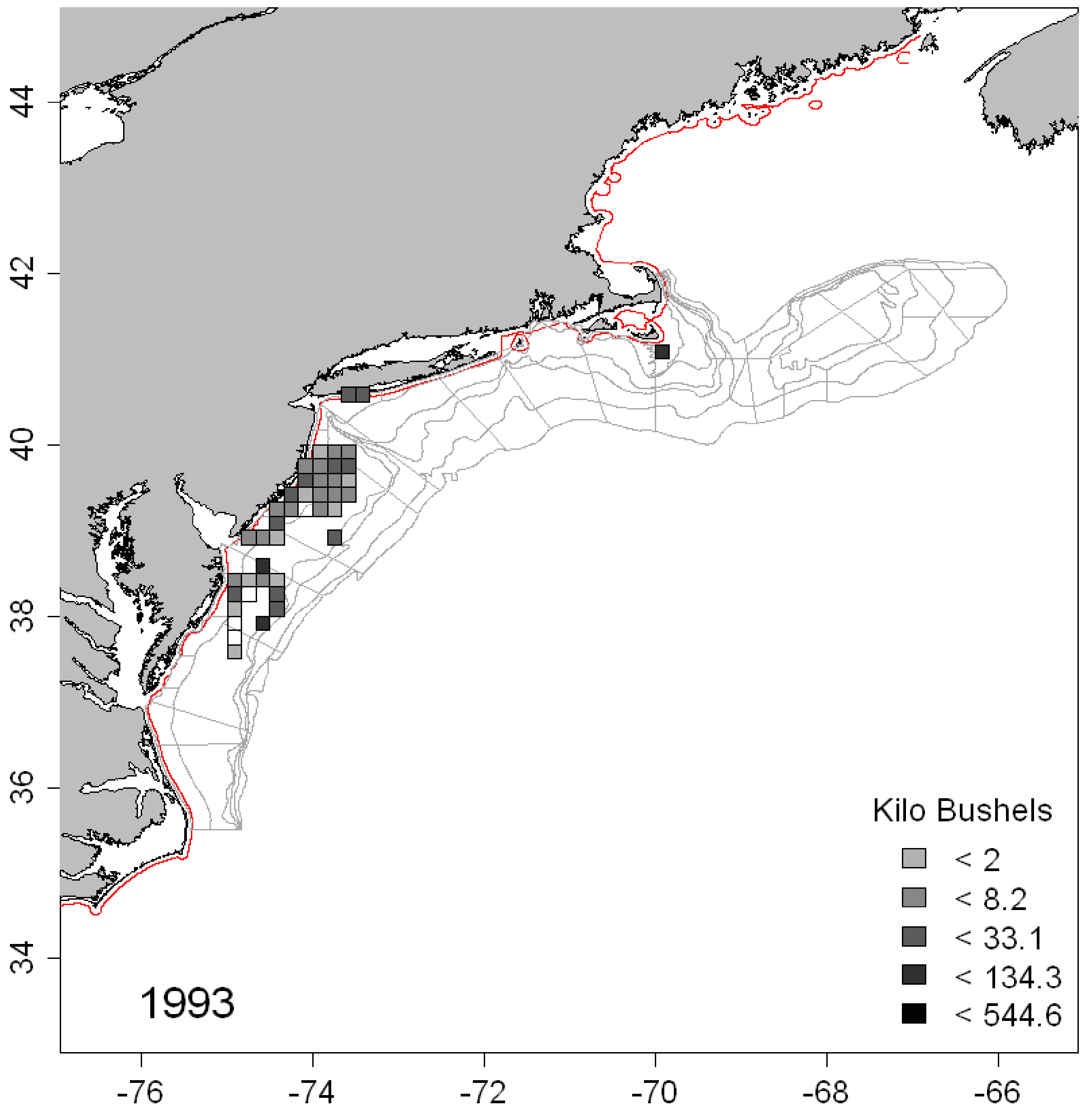

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1990



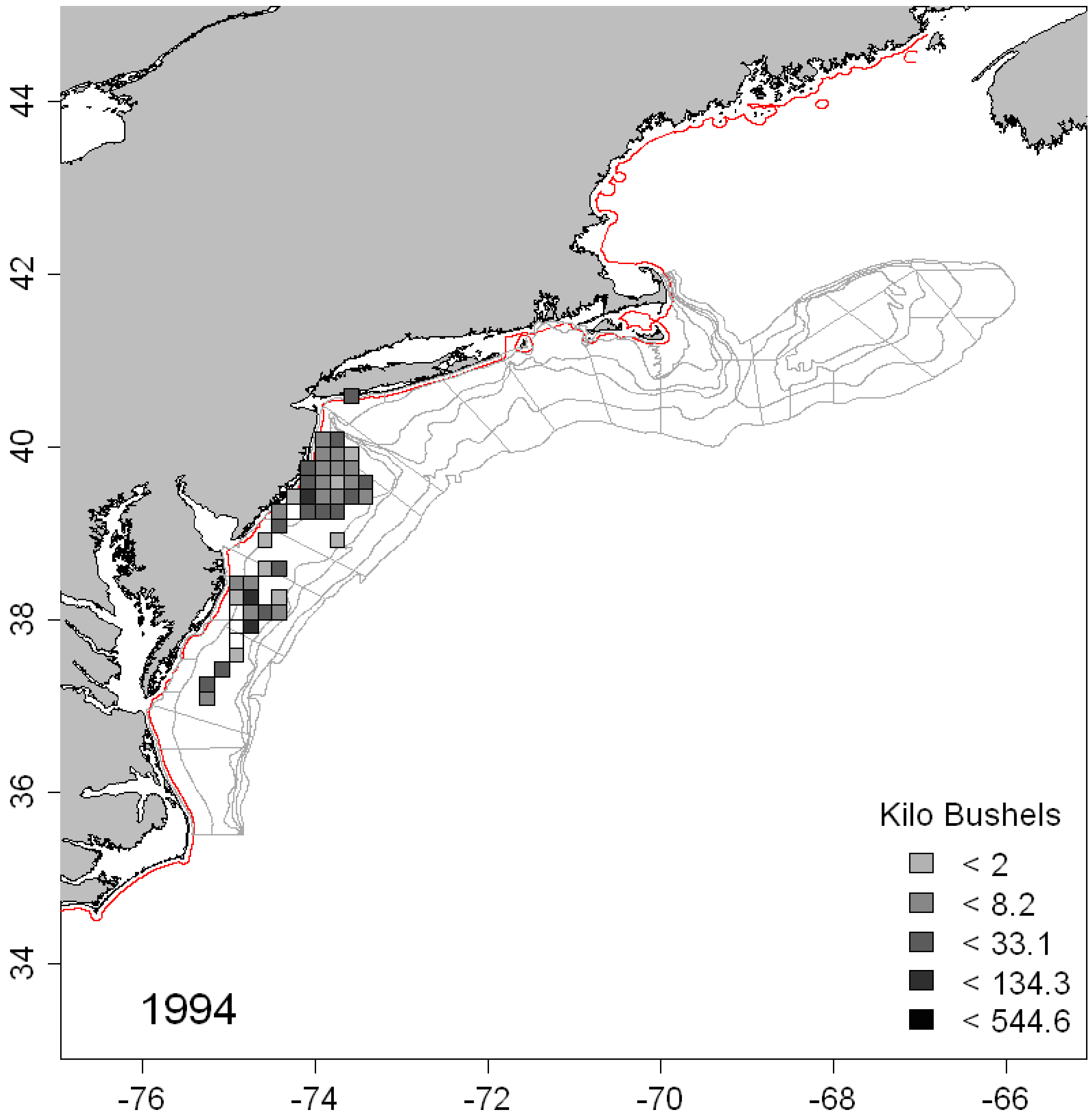
Surfclam catch by ten-minute square

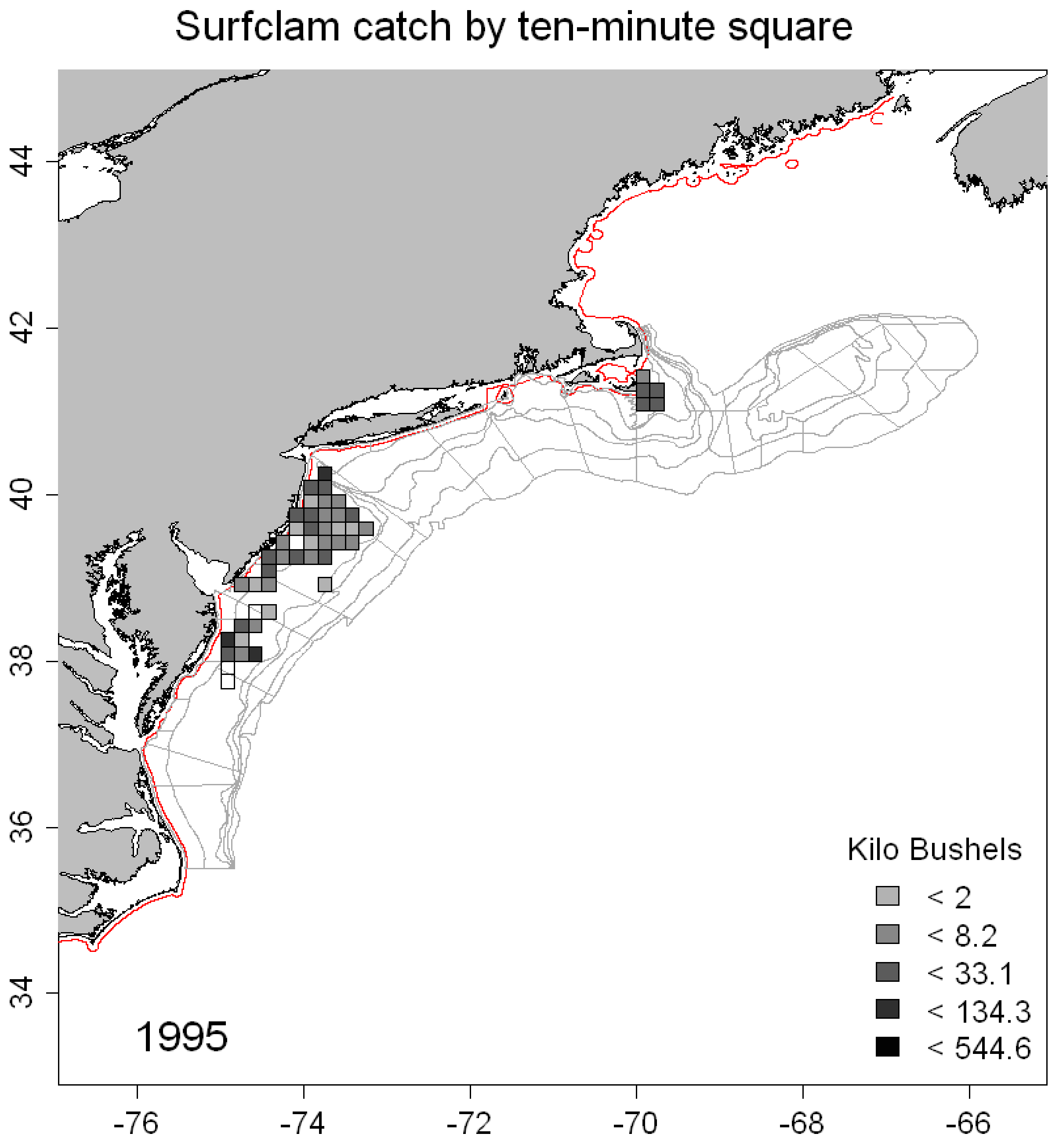

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1992

Surfclam catch by ten-minute square

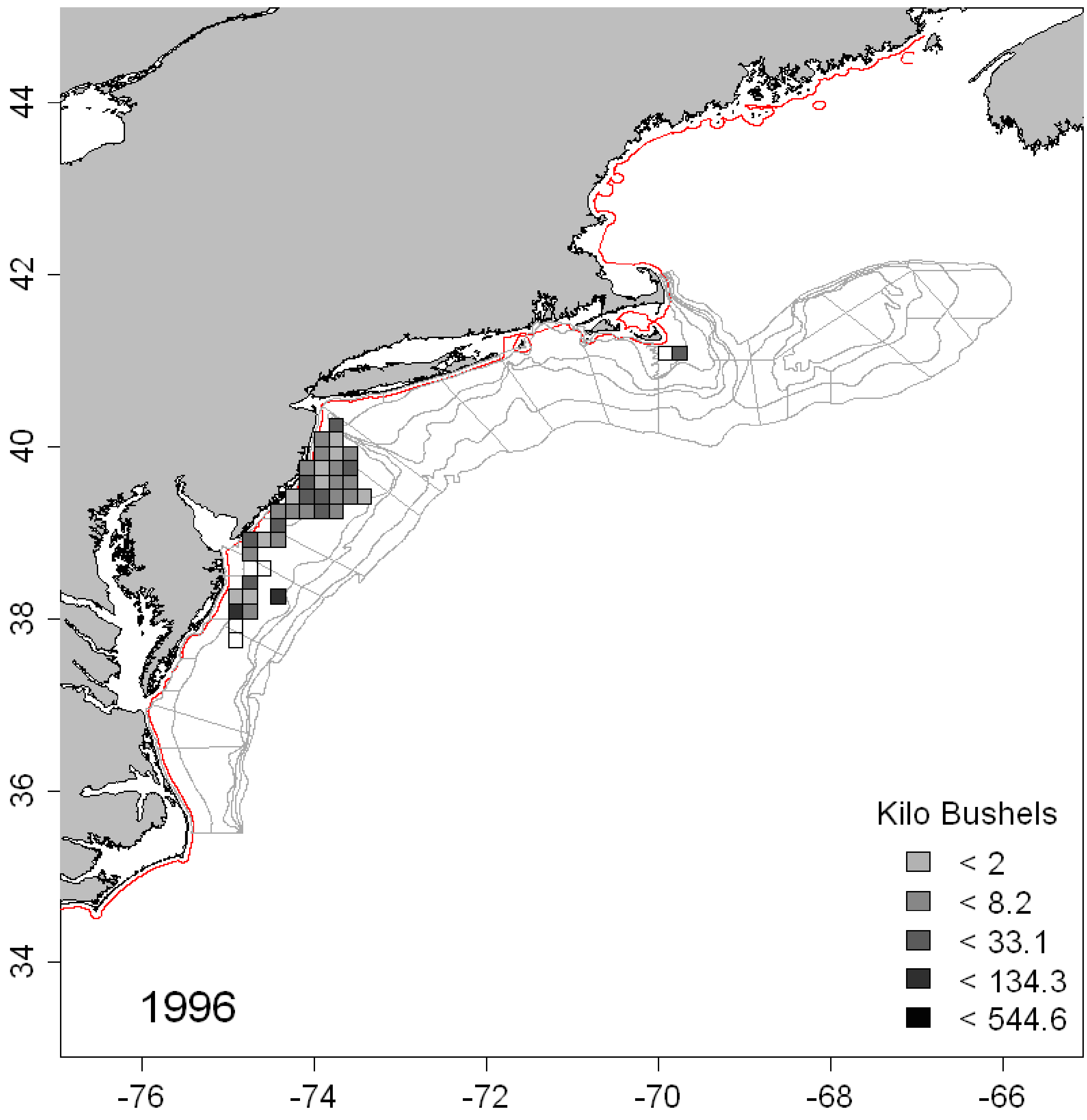


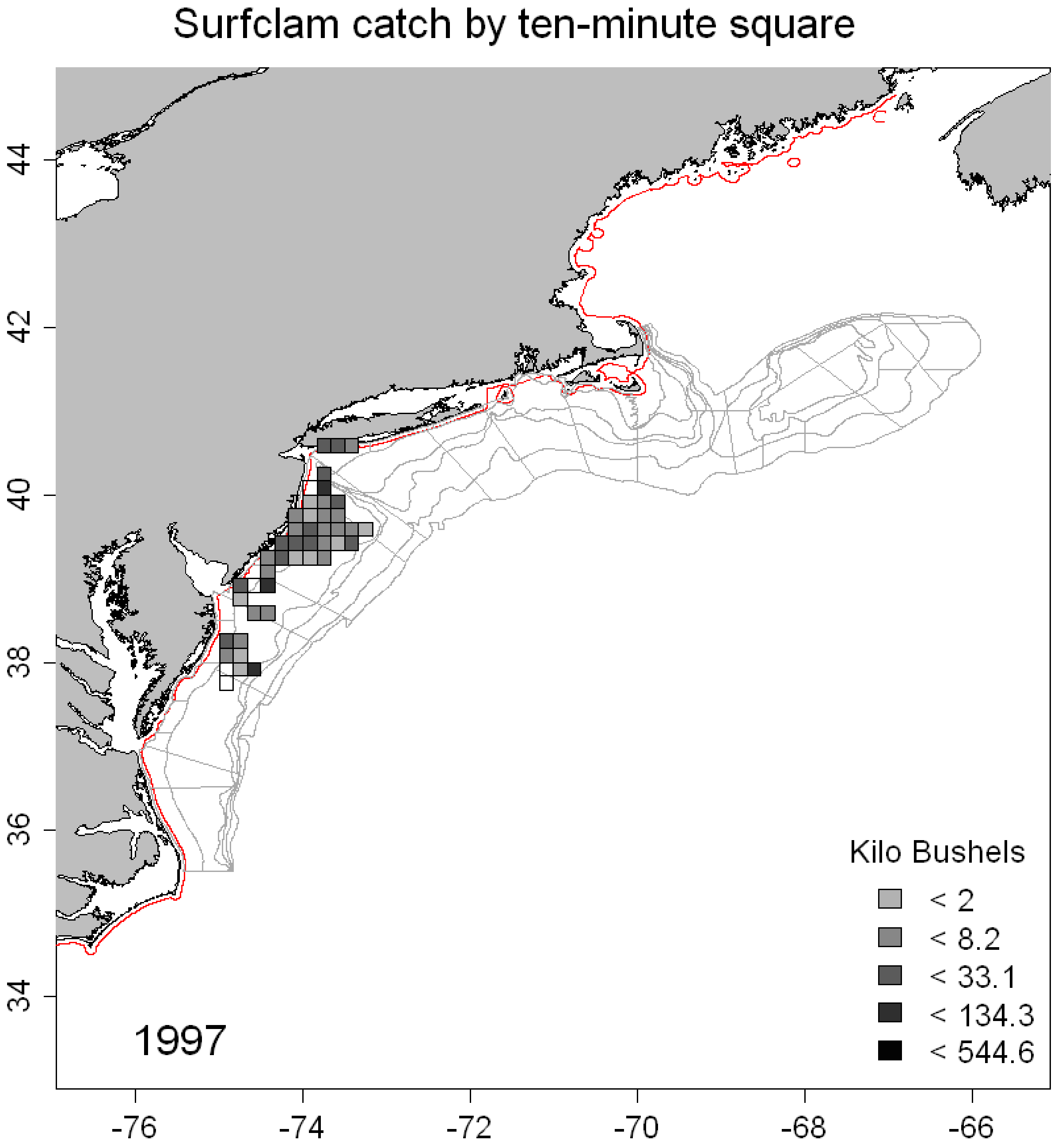
Surfclam catch by ten-minute square

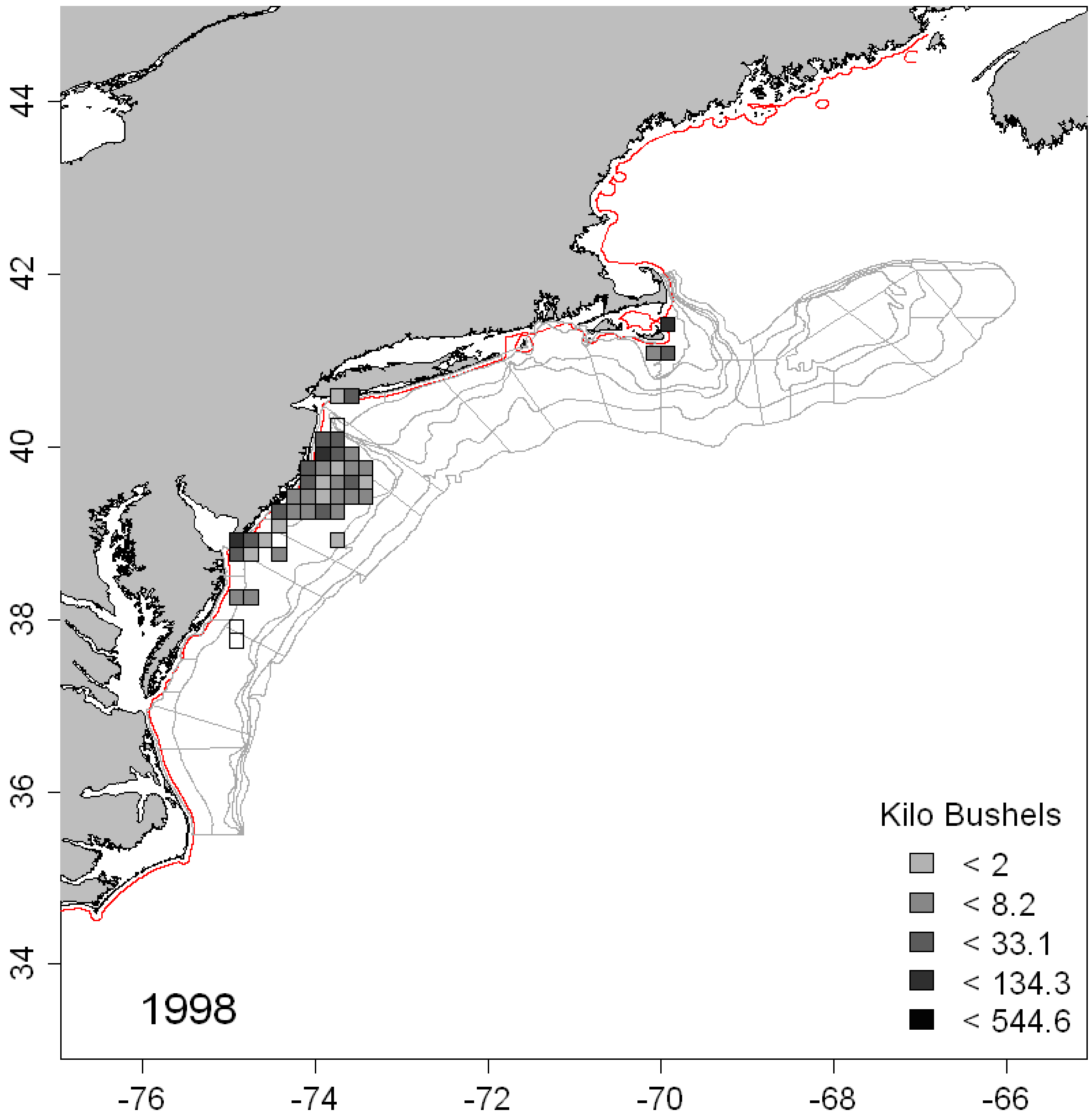



Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
1995
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

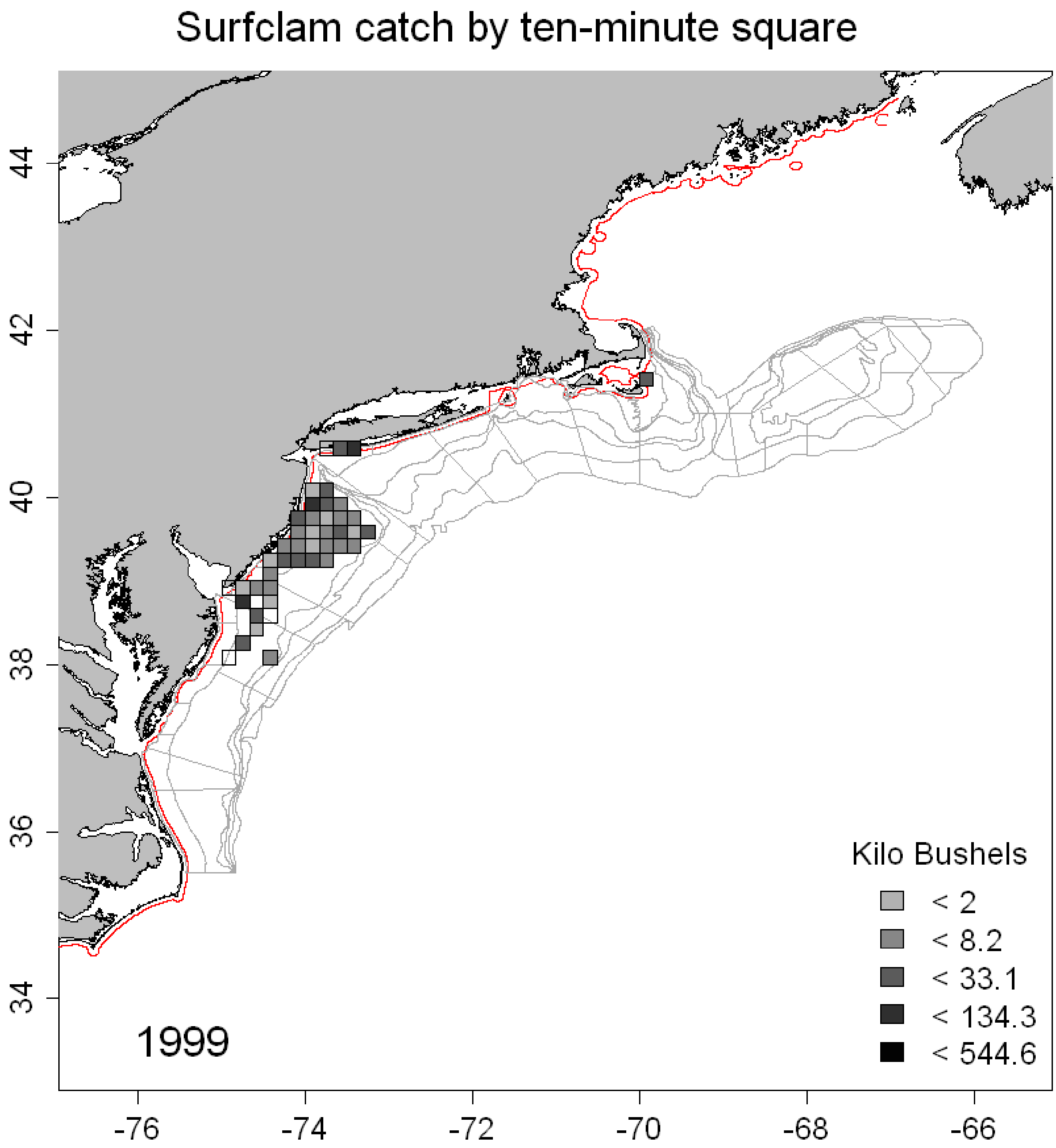
Surfclam catch by ten-minute square

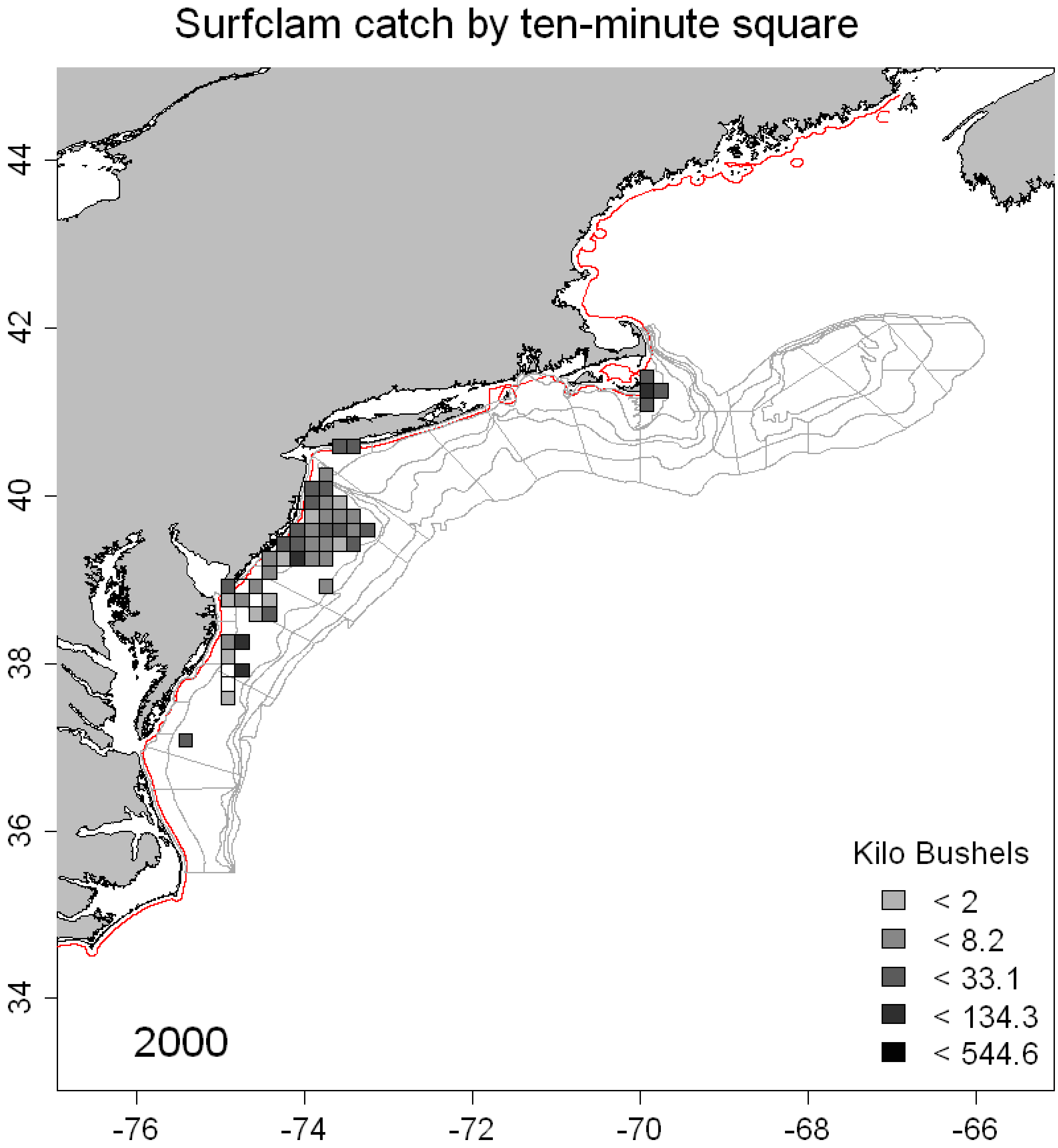


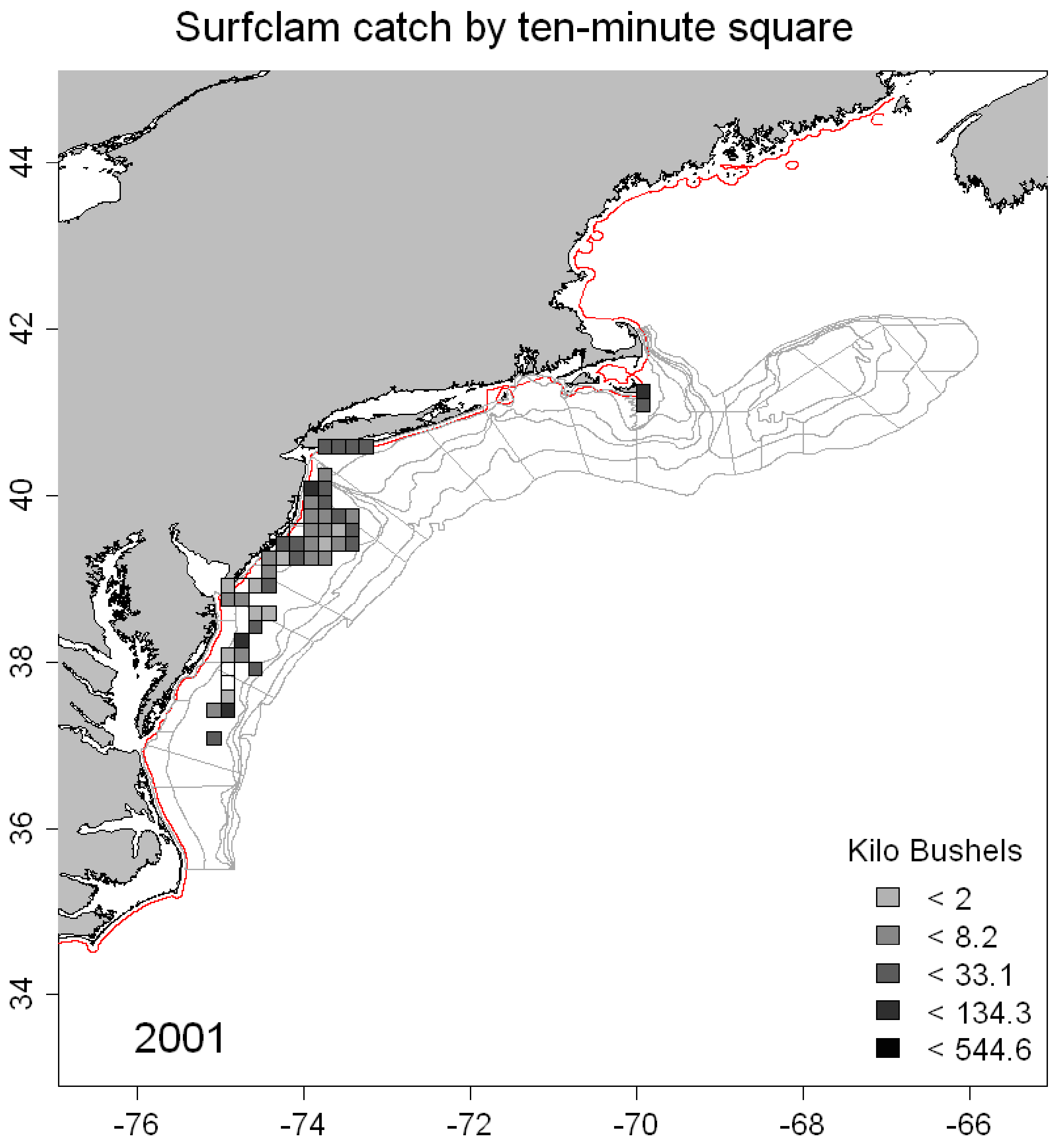

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1997



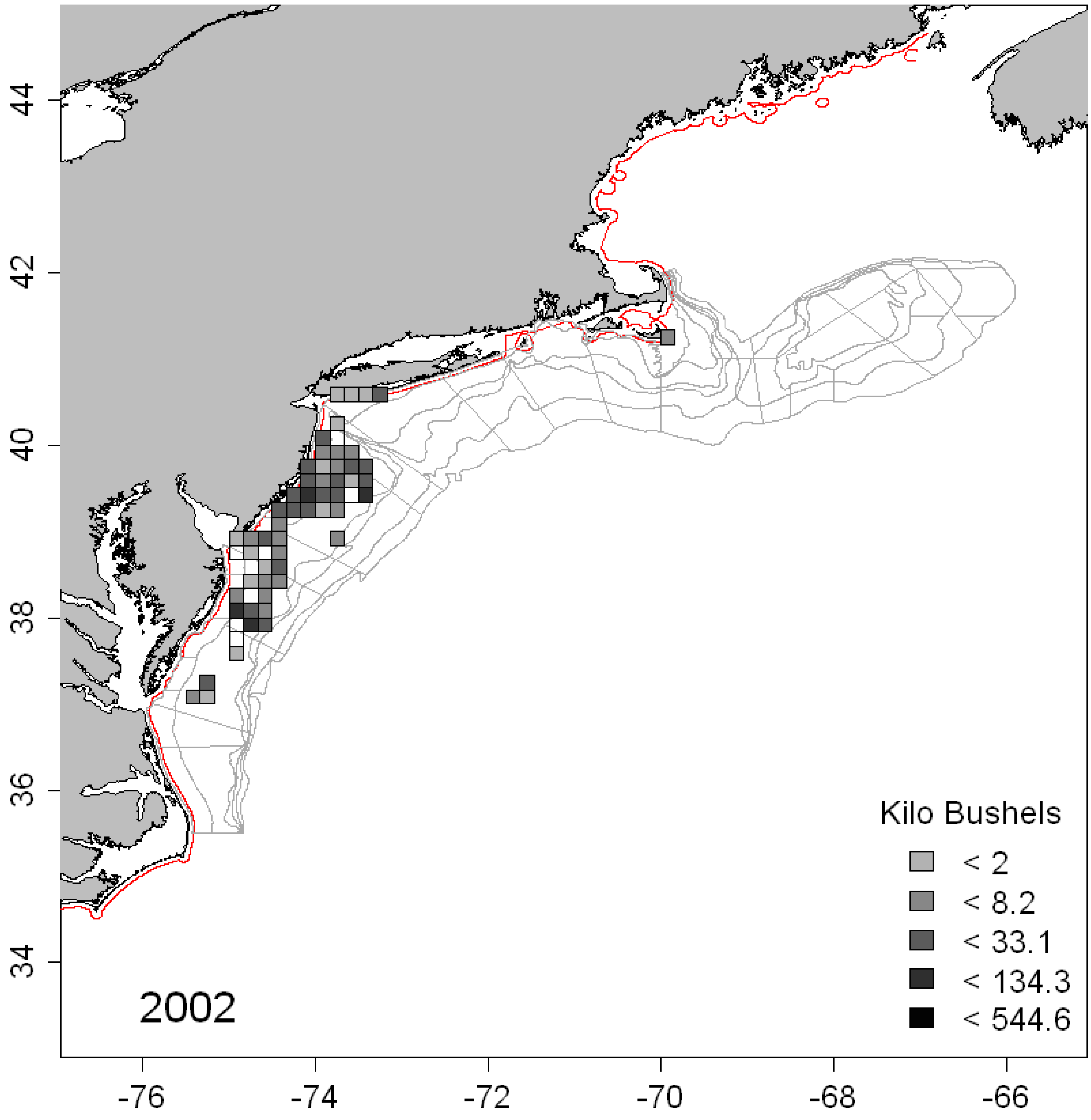
Surfclam catch by ten-minute square

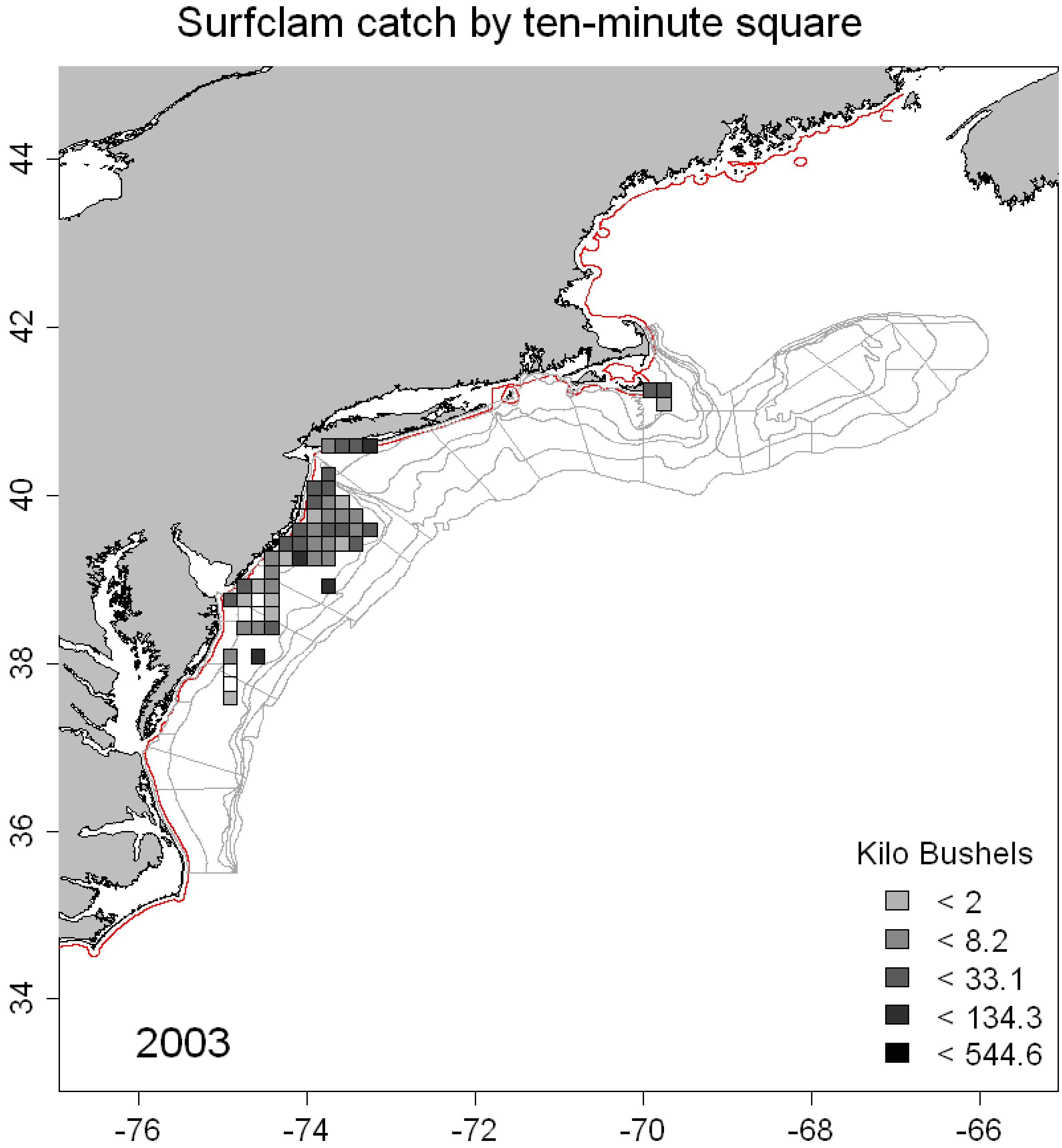

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
1999


Surfclam catch by ten-minute square
44
42
40
38
36
34
2000
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

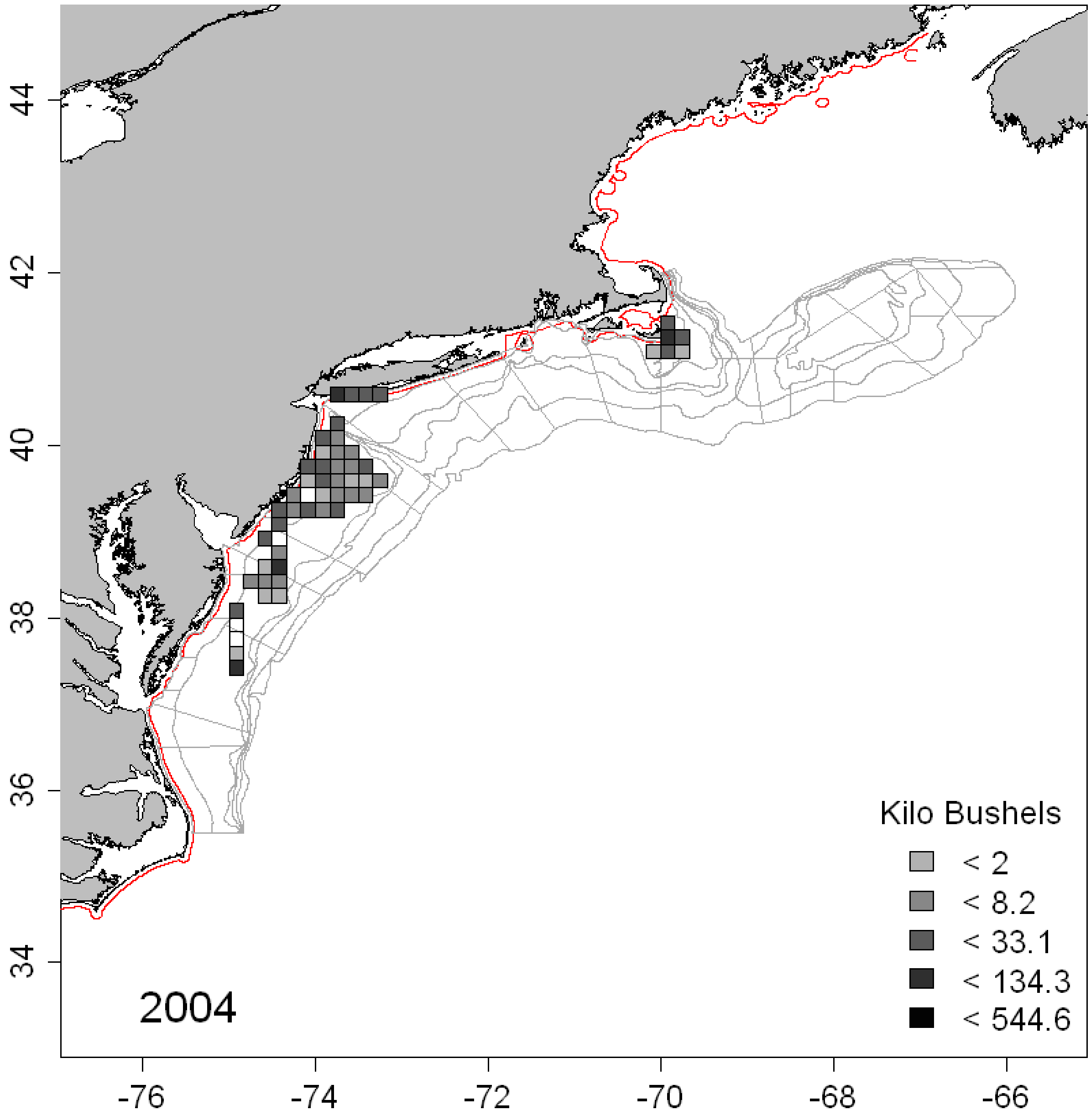

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
2001
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

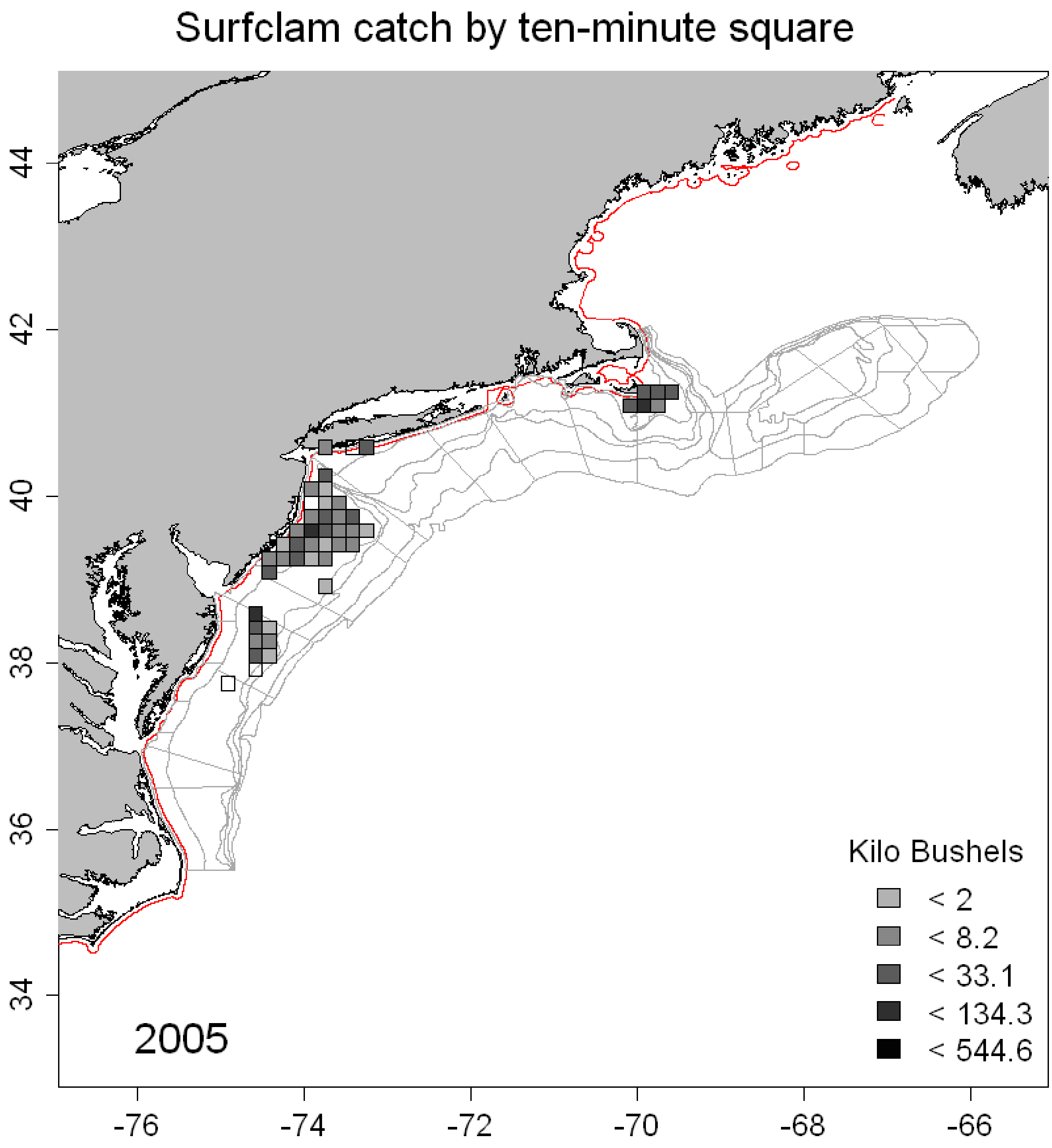
Surfclam catch by ten-minute square

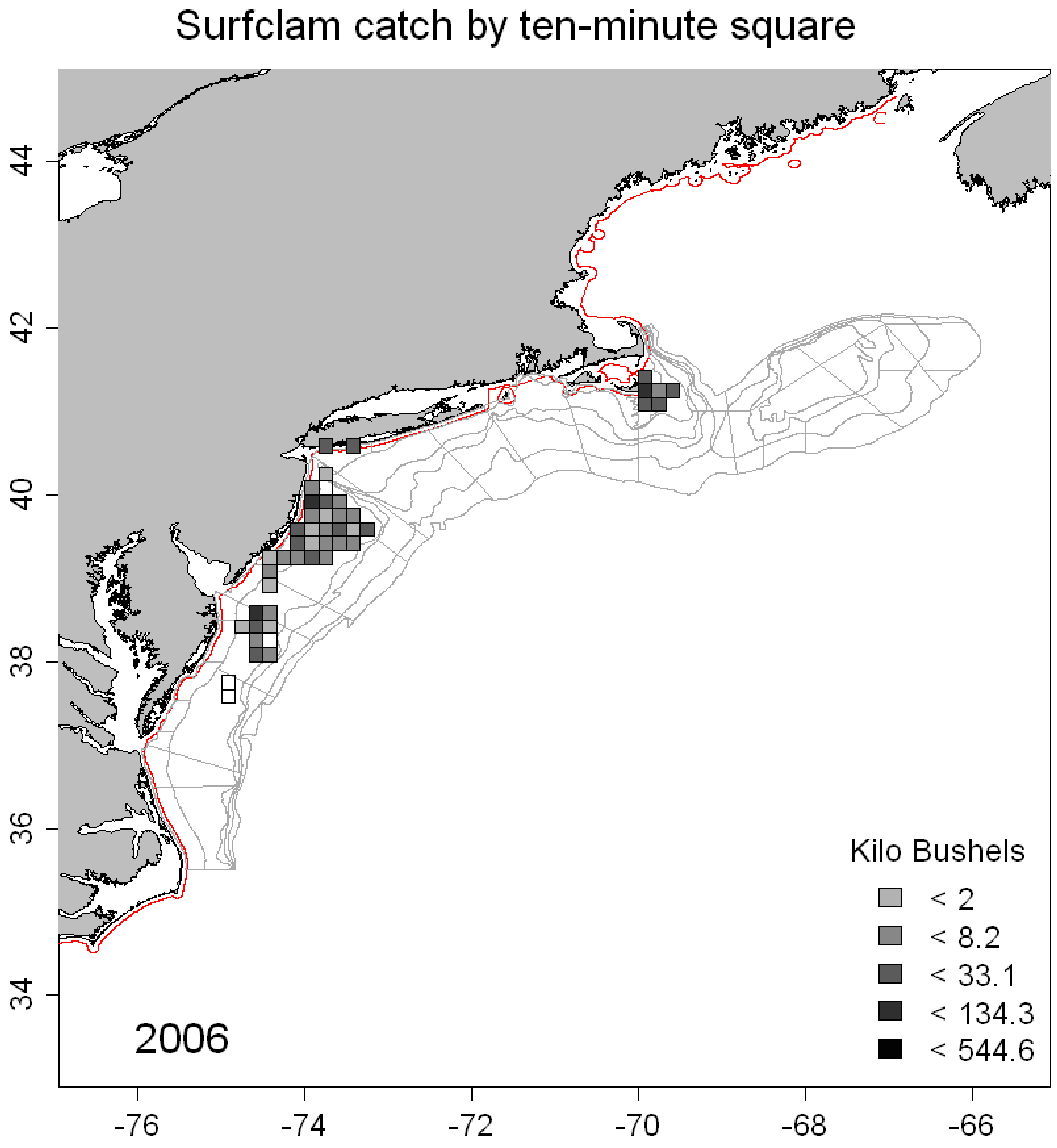



Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
2003
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

Surfclam catch by ten-minute square

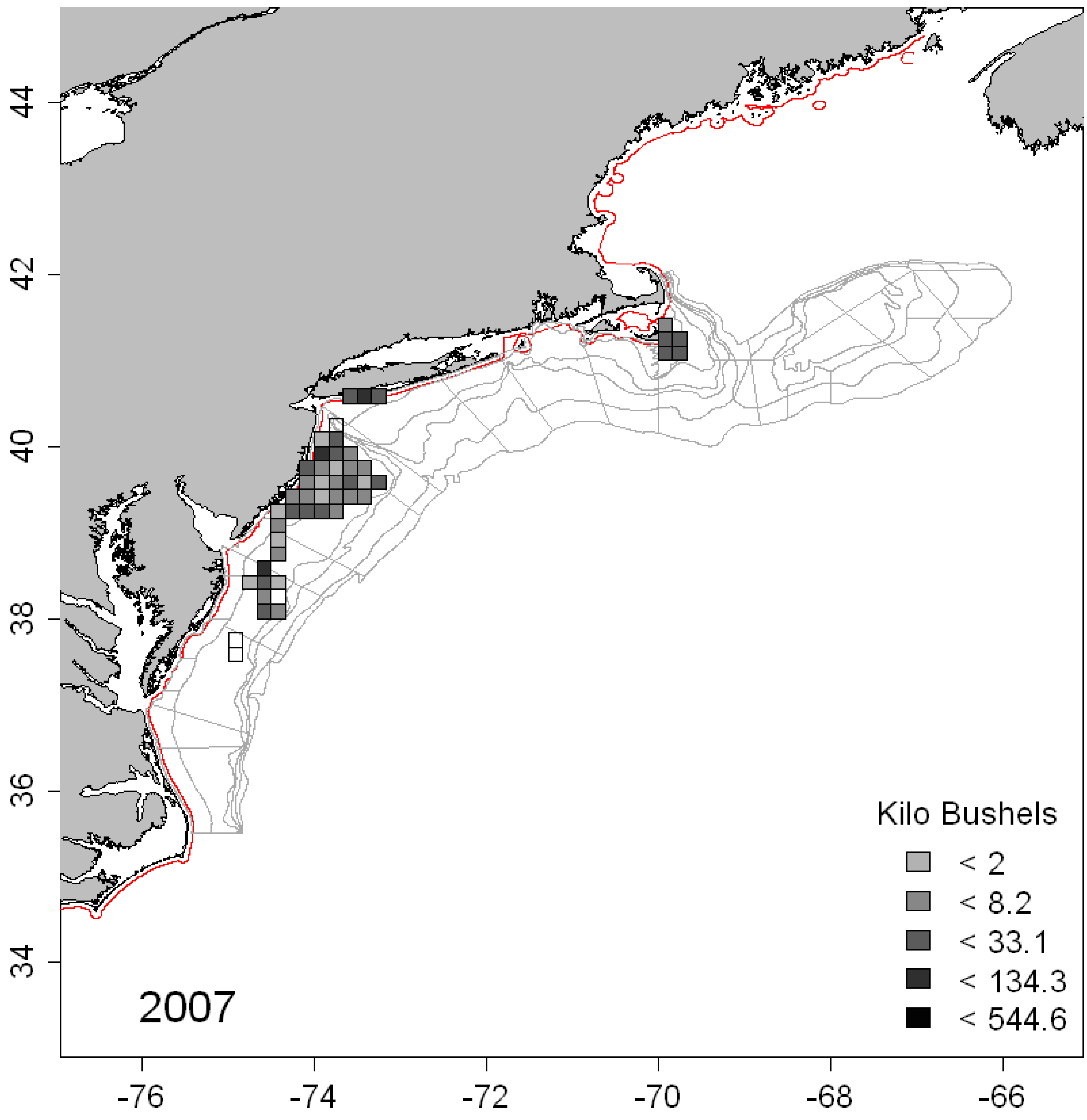



Surfclam catch by ten-minute square

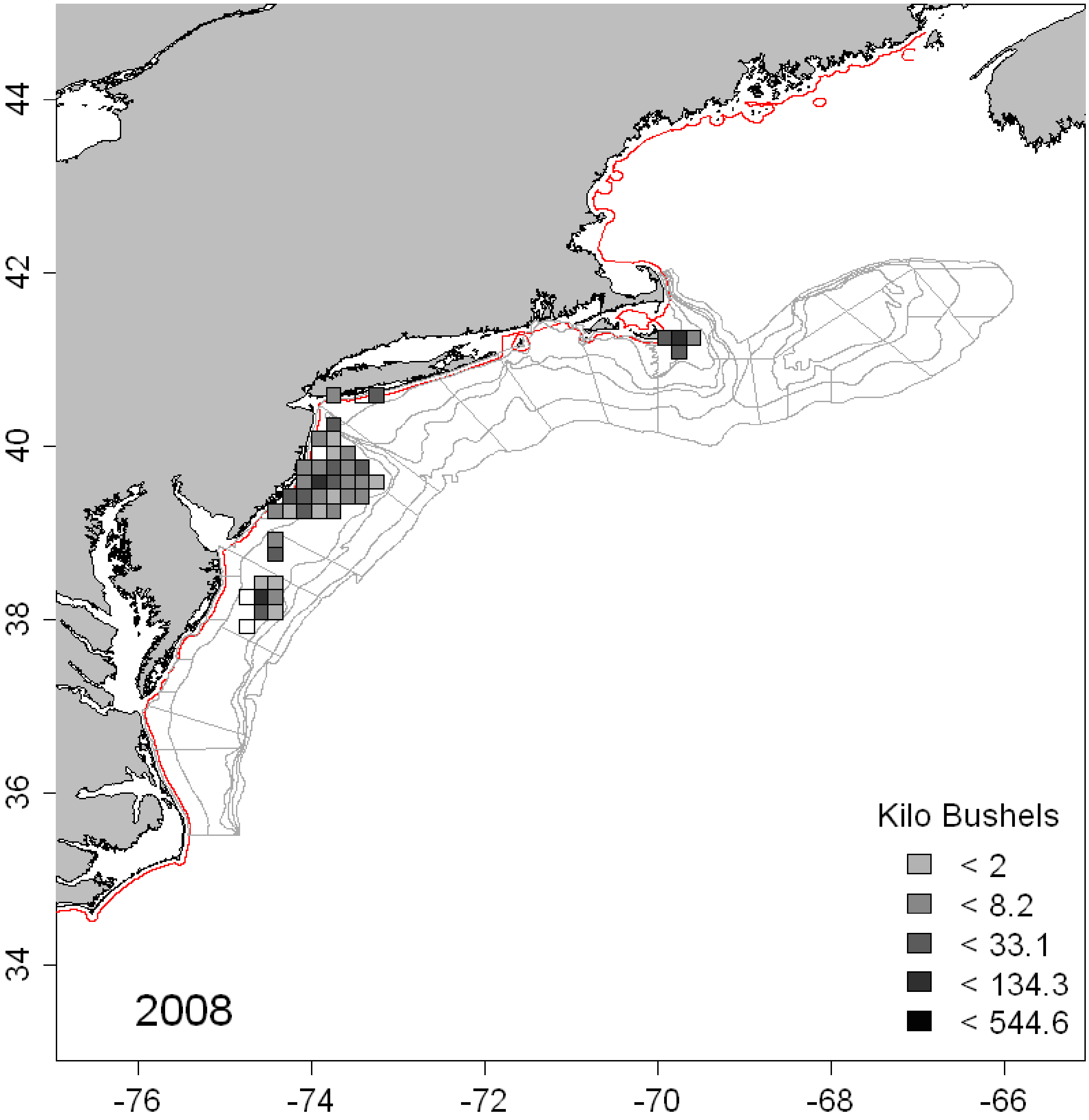


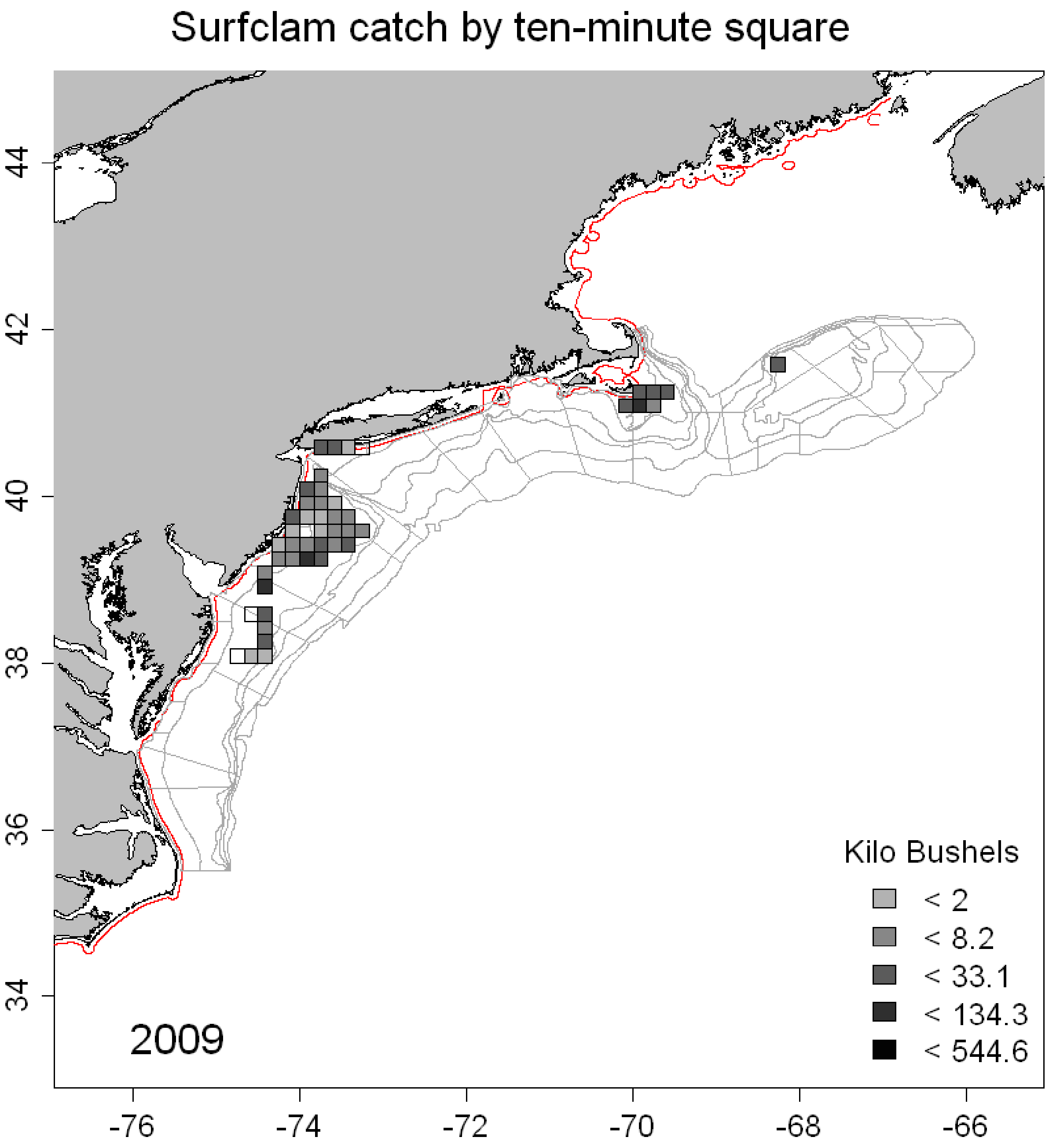
Surfclam catch by ten-minute square

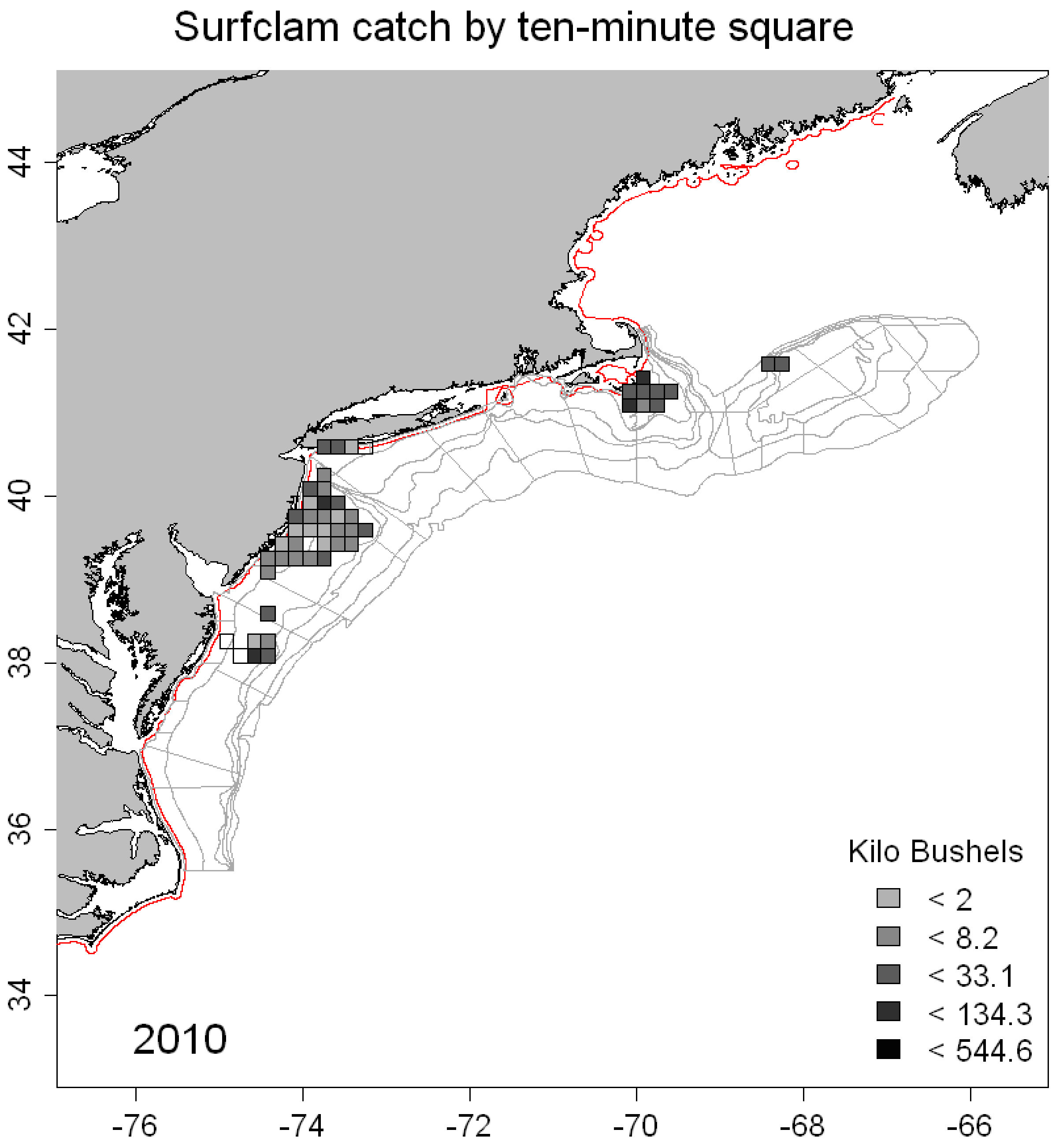
Surfclam catch by ten-minute square

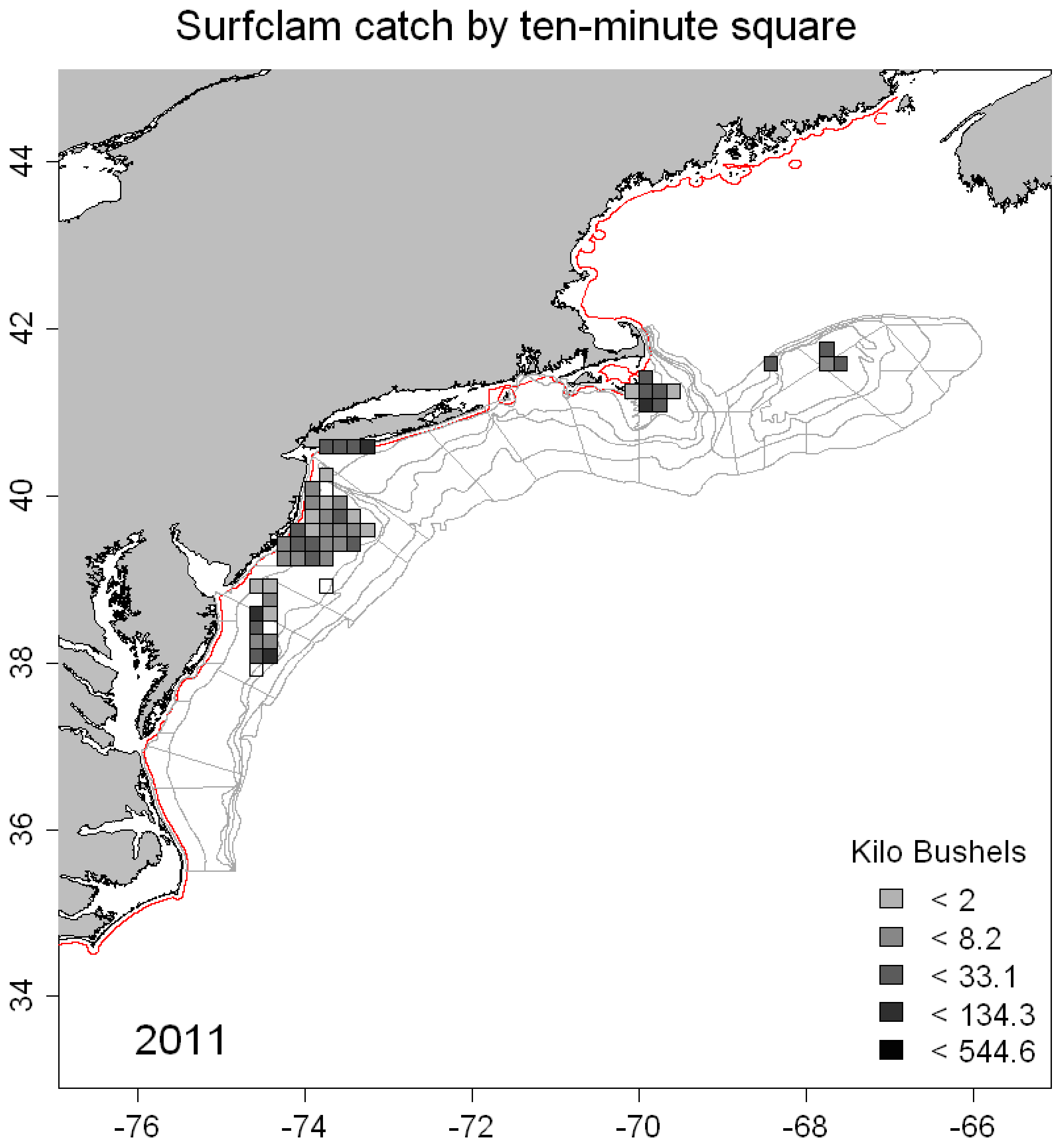


Surfclam catch by ten-minute square



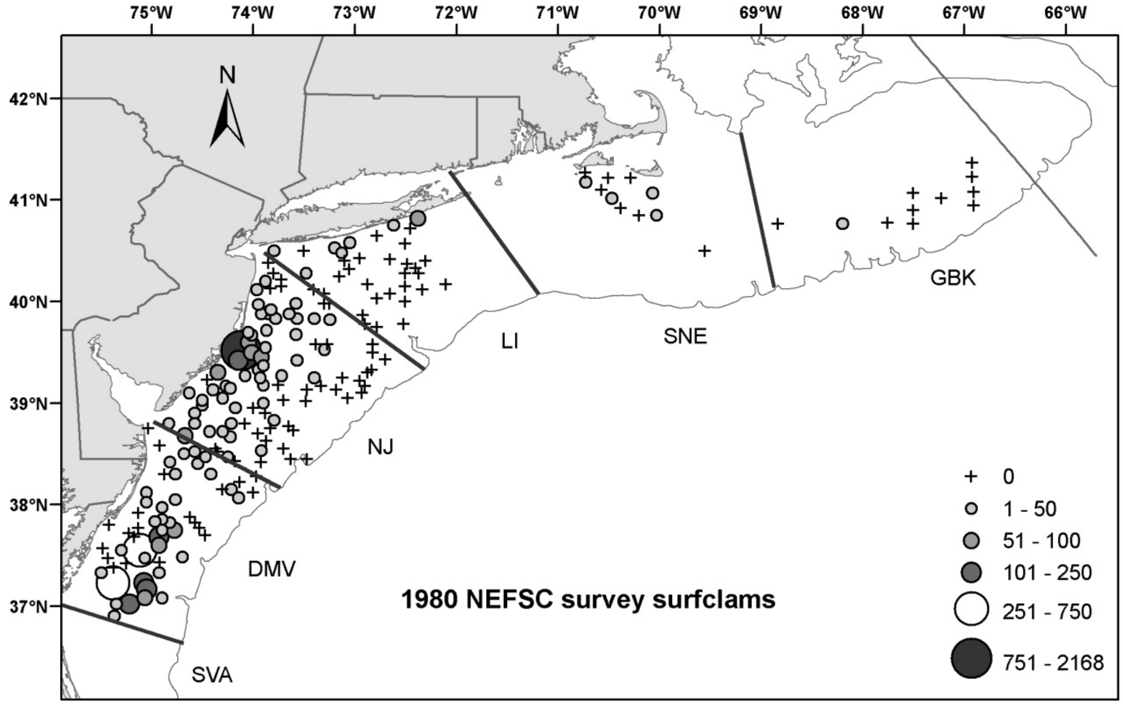

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6
2009

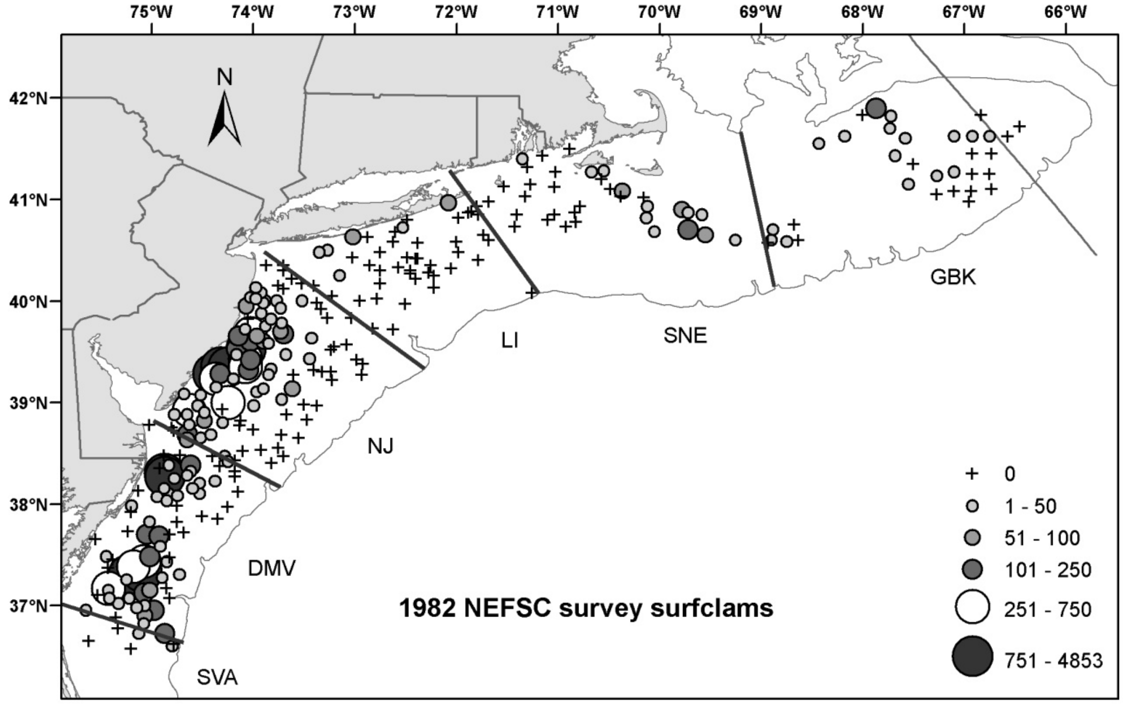

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
2010
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

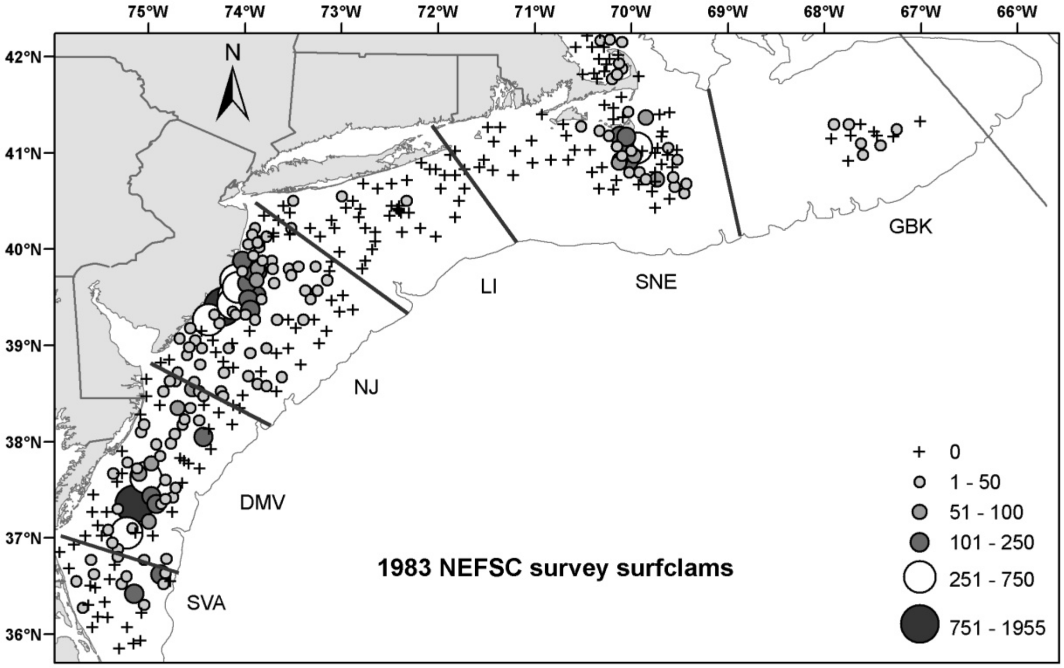

Surfclam catch by ten-minute square
44
42
40
38
36
34
-76
-74
-72
-70
-68
-66
2011
Kilo Bushels
< 2
< 8.2
< 33.1
< 134.3
< 544.6

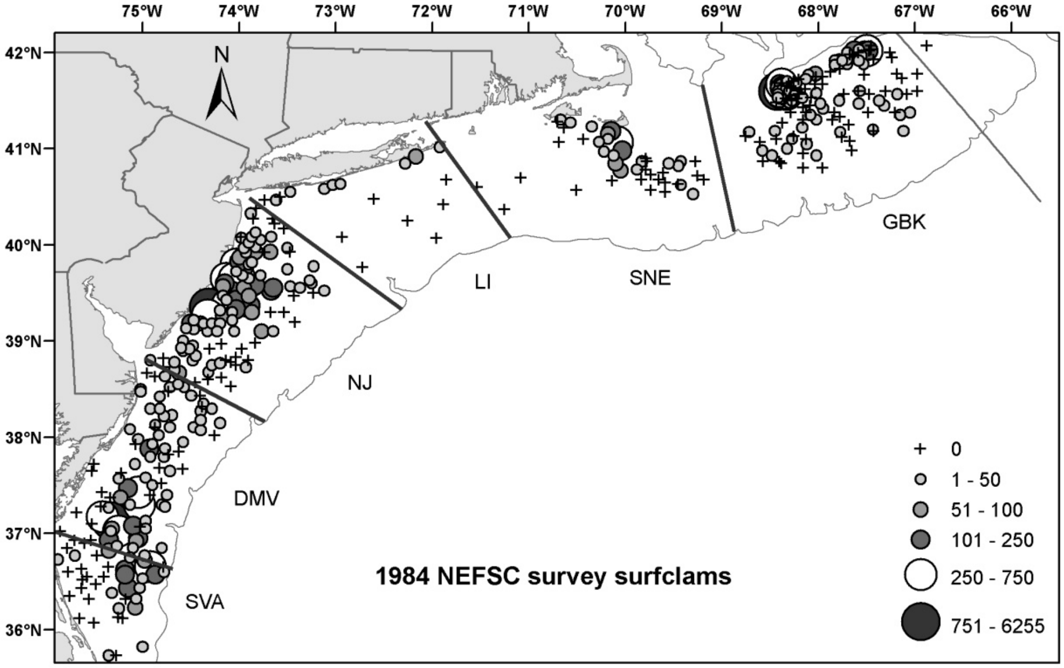
Appendix A3: Maps of NEFSC clam surveys

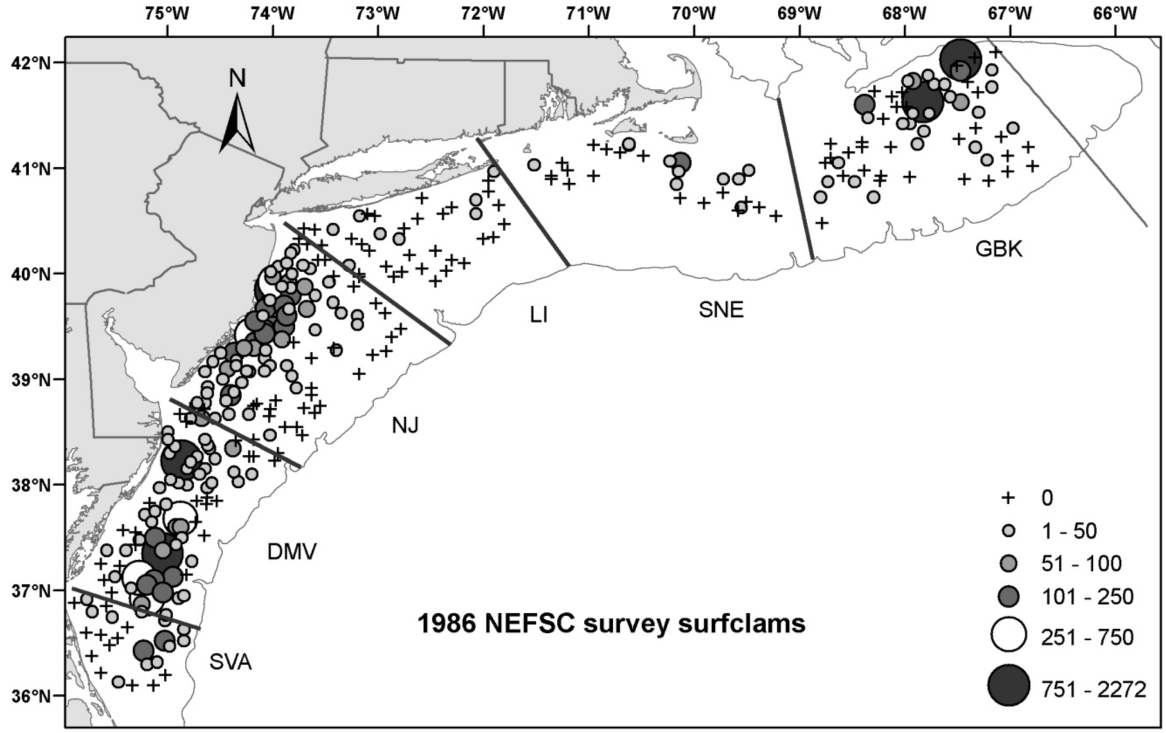
(Following pages) Maps of NEFSC clam survey surfclam catches since 1980. Symbols represent number per tow of clams of all sizes. The maximum number of clams caught in a tow is the highest number in the legend.

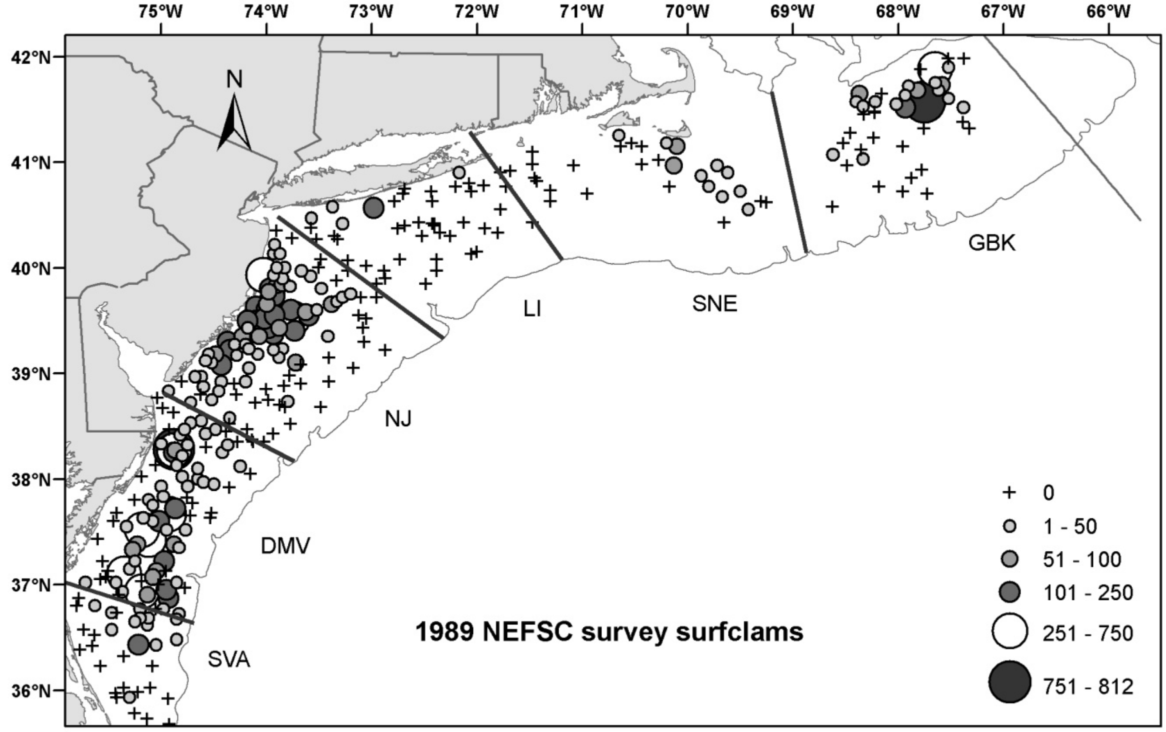

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
SNE
LI
NJ
DMV
SVA
1980 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 2168

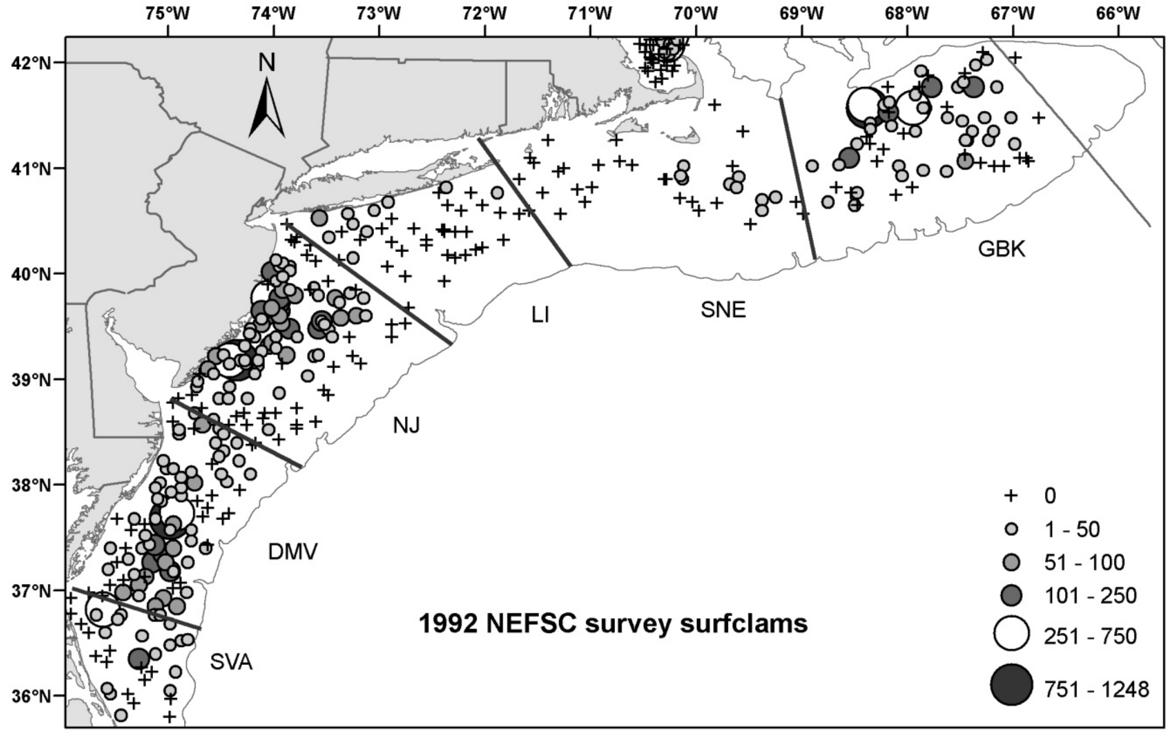

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
SNE
LI
NJ
DMV
SVA
1982 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 4853

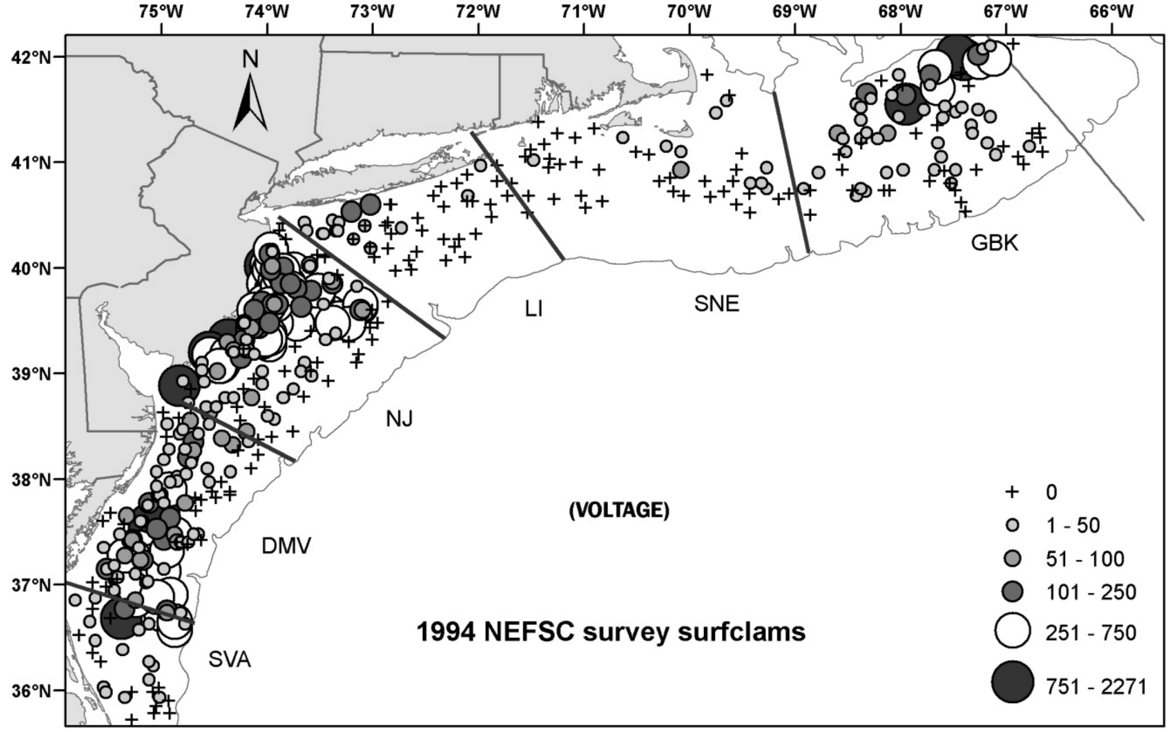

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
1983 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 1955

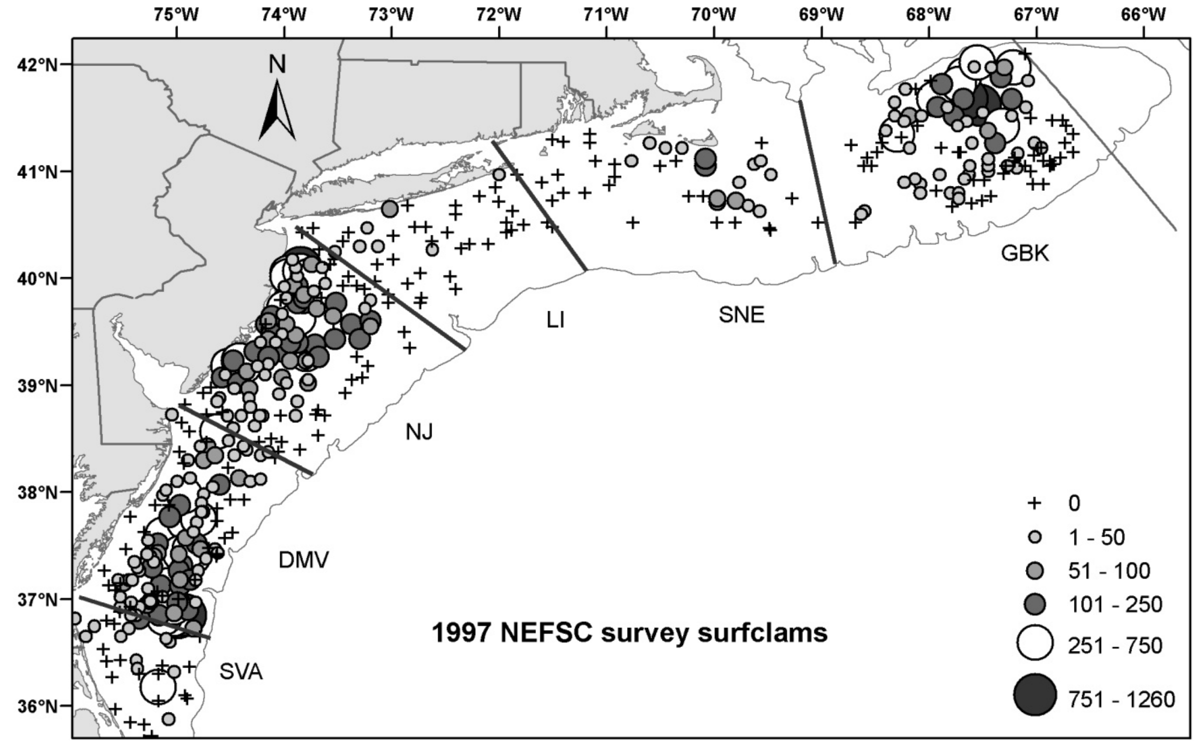

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
1984 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
250 - 750
751 - 6255

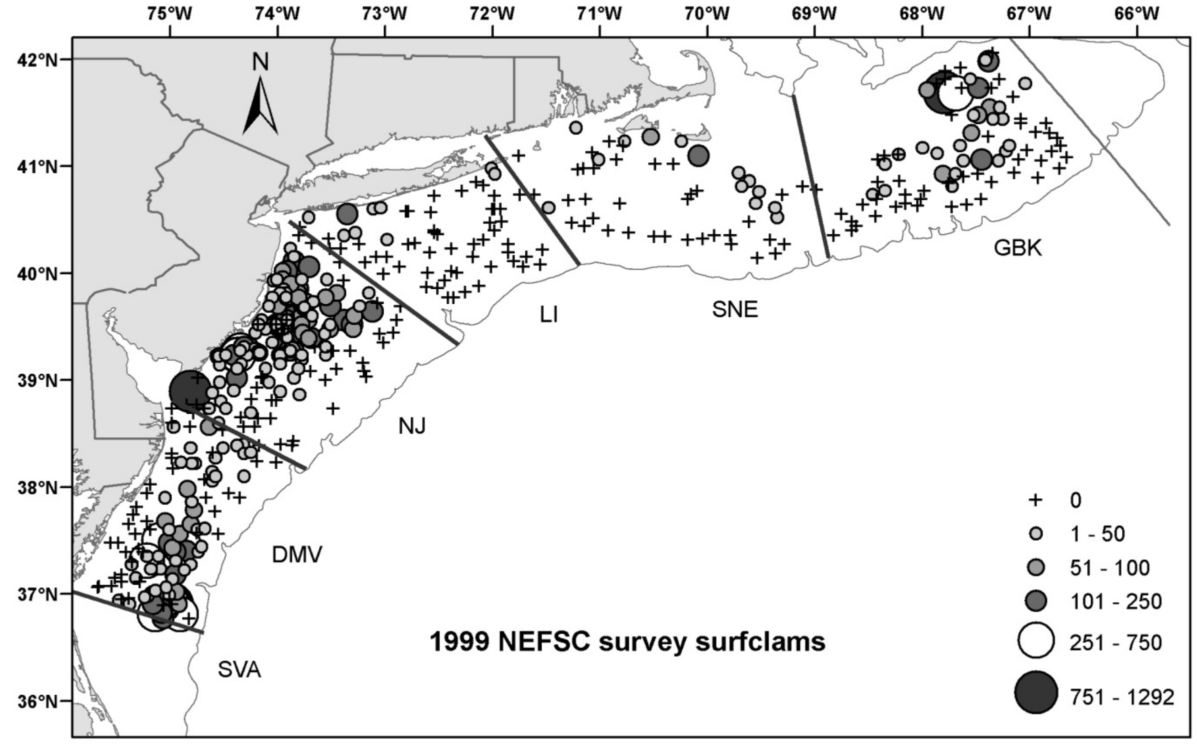

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
1986 NEFSC survey surfclams
0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 2272

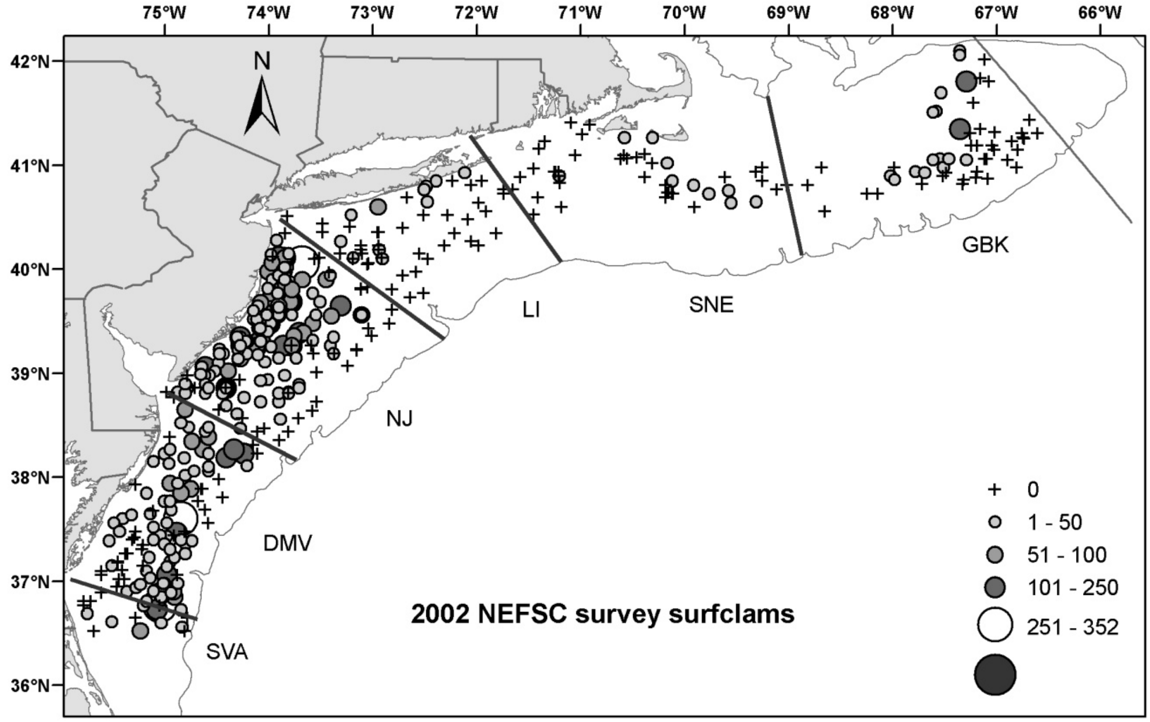

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
1989 NEFSC survey surfclams
0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 812


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 1248
1992 NEFSC survey surfclams


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
GBK
LI
SNE
NJ
DMV
SVA
(VOLTAGE)
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 2271
1994 NEFSC survey surfclams


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
LI
SNE
GBK
NJ
DMV
SVA
1997 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 1260


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
36°N
N
LI
SNE
GBK
NJ
DMV
SVA
1999 NEFSC survey surfclams
+ 0
1 - 50
51 - 100
101 - 250
251 - 750
751 - 1292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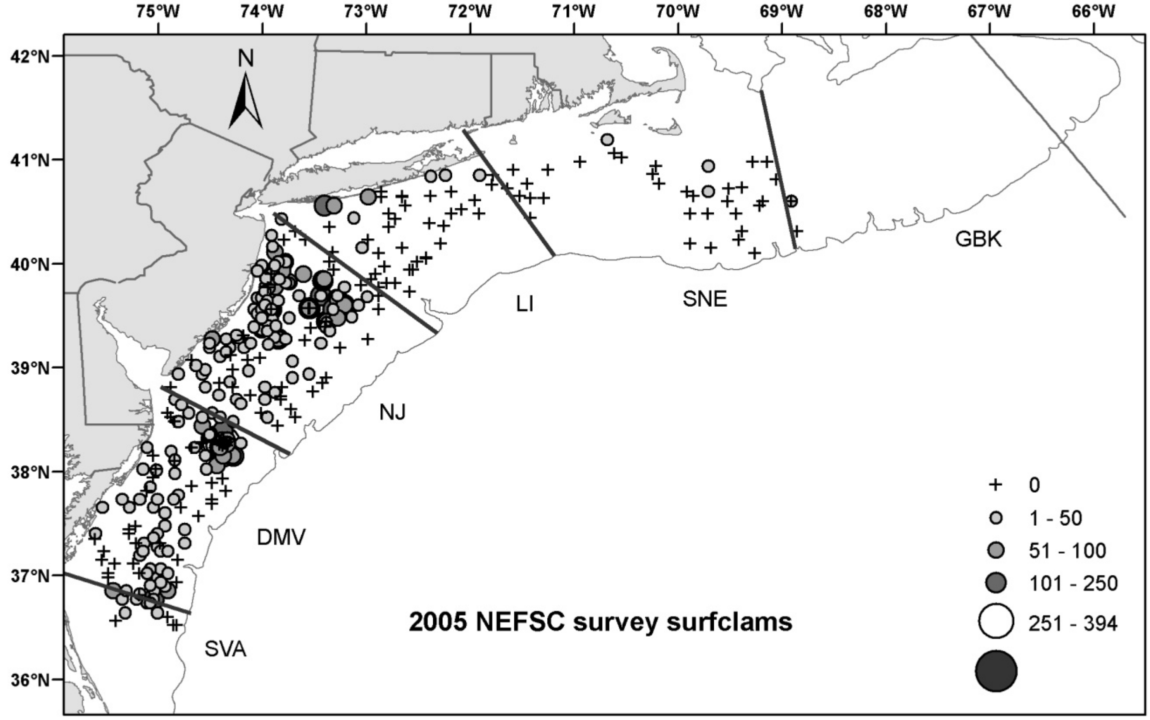
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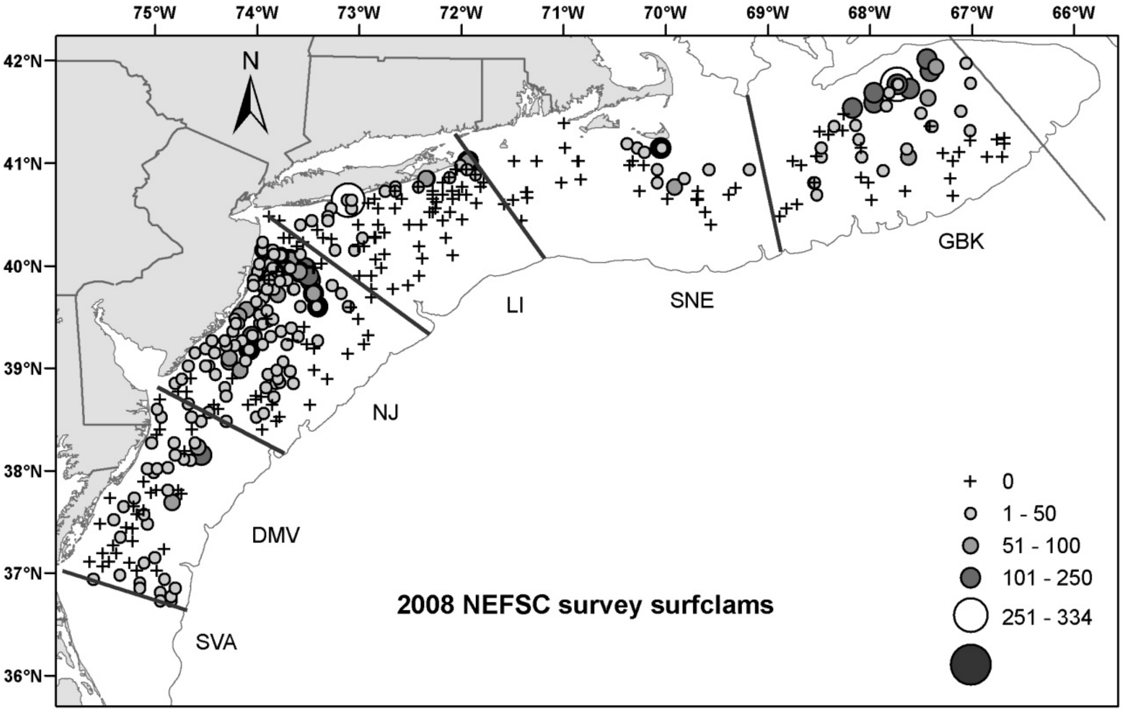
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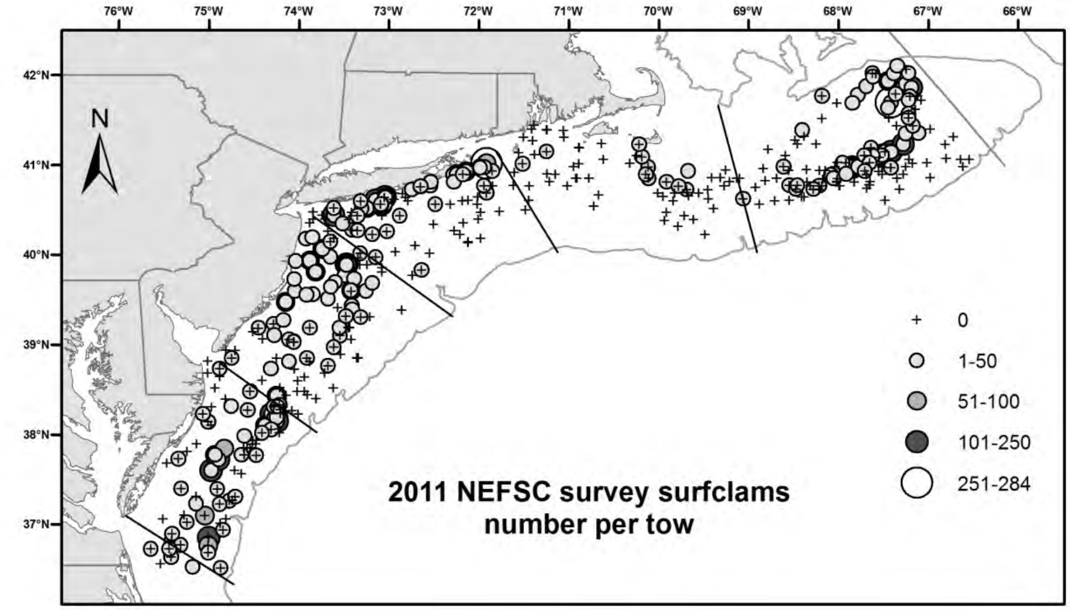
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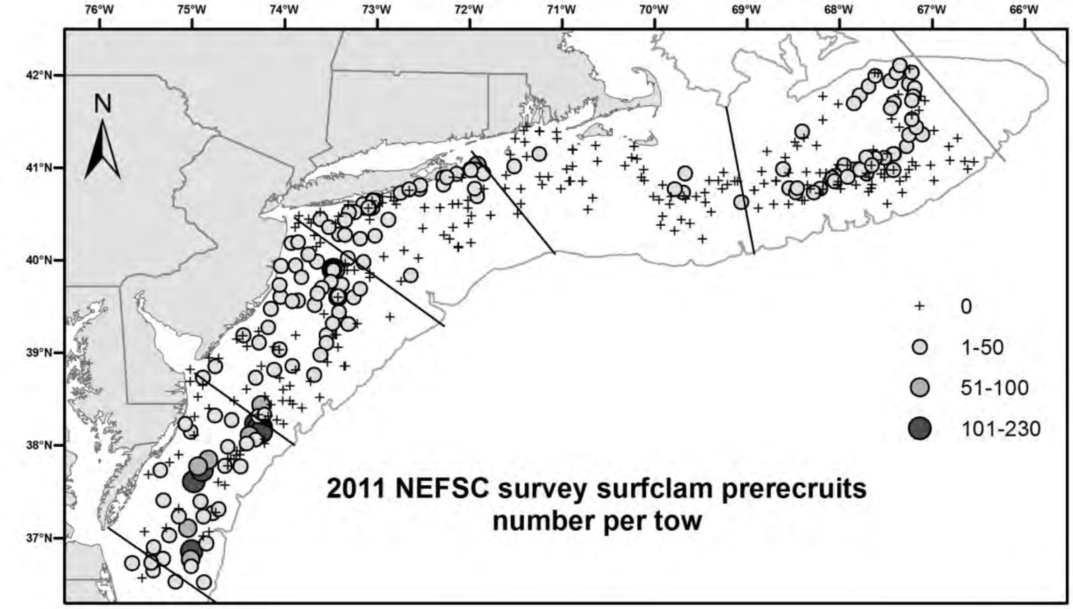
2011 NEFSC survey surfclams
number per tow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2011 NEFSC survey surfclam prerecruits
number per tow

Appendix A4: KLAMZ methods

KLAMZ Assessment Model – Technical Documentation

The KLAMZ assessment model is based on the Deriso-Schnute delay-difference equation (Deriso 1980; Schnute 1985; Quinn and Deriso 1999). The delay-difference equation is a relatively simple and implicitly age structured approach to counting fish in either numerical or biomass units. It gives the same results as explicitly age-structured models (e.g. Leslie matrix model) if fishery selectivity is "knife-edged", if somatic growth follows the von Bertalanffy equation, and if natural mortality is the same for all age groups in each year. Knife-edge selectivity means that all individuals alive in the model during the same year experience the same fishing mortality rate.[5] Natural and fishing mortality rates, growth parameters and recruitment may change from year to year, but delay-difference calculations assume that all individuals share the same mortality and growth parameters within each year. The KLAMZ model includes simple numerical models (e.g. Conser 1995) as special cases because growth can be turned off so that all calculations are in numerical units (see below).

As in many other simple models, the delay difference equation explicitly distinguishes between two age groups. In KLAMZ, the two age groups are called "new" recruits (R_t in biomass or numerical units at the beginning of year t) and "old" recruits (S_t) that together comprise the whole stock (B_t). New recruits are individuals that recruited at the beginning of the current year (at nominal age k).[6] Old recruits are all older individuals in the stock (nominal ages k+1 and older, survivors from the previous year). As described above, KLAMZ assumes that new and old recruits are fully vulnerable to the fishery. The most important differences between the delay-difference and other simple models (e.g. Prager 1994; Conser 1995; Jacobson et al. 1994) are that von Bertalanffy growth is used to calculate biomass dynamics and that the delay-difference model captures transient age structure effects due to variation in recruitment, growth and mortality exactly. Transient effects on population dynamics are captured exactly because, as described above, the delay-difference equation is algebraically equivalent to an explicitly age-structured model with von Bertalanffy growth.

The KLAMZ model incorporates a few extensions to Schnute's (1985) revision of Deriso's (1980) original delay difference model. Most of the extensions facilitate tuning to a wider variety of data that anticipated in Schnute (1985). The KLAMZ model is programmed in

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[5] In applications, assumptions about knife-edge selectivity can be relaxed by assuming the model tracks "fishable", rather that total, biomass (NEFSC 2000a; 2000b). An analogous approach assigns pseudo-ages based on recruitment to the fishery so that new recruits in the model are all pseudo-age k. The synthetic cohort of fish pseudo-age k may consist of more than one biological cohort. The first pseudo-age (k) can be the predicted age at first, 50% or full recruitment based a von Bertalanffy curve and size composition data (Butler et al. 2002). The "incomplete recruitment" approach (Deriso 1980) calculates recruitment to the model in each year R_t as the weighted sum of contributions from two or more biological cohorts (year-classes) from spawning during successive years (i.e. $R_t = \sum_{a=1}^{k} r_a \Pi_{t-a}$ where k is the age at full recruitment to the fishery, r_a is the contribution of fish age k-a to the fishable stock, and Π_{t-a} is the number or biomass of fish age k-a during year t).

[6] In some applications, and more generally, new recruits might be defined as individuals recruiting at the beginning or at any time during the current time step (e.g. NEFSC 1996). [6]
Otter Research Ltd., Box 2040, Sydney, BC, Canada V8L 3S3 (otter@otter-rsch.com).

both Excel and in C++ using AD Model Builder[7] libraries. The AD Model Builder version is faster, more reliable and probably better for producing "official" stock assessment results. The Excel version is slower and implements fewer features, but the Excel version remains useful in developing prototype assessment models, teaching and for checking calculations.

The most significant disadvantage in using the KLAMZ model and other delay-difference approaches, beyond the assumption of knife-edge selectivity, is that age and length composition data are not used in tuning. However, one can argue that age composition data are used indirectly to the extent they are used to estimate growth parameters or if survey survival ratios (e.g. based on the Heinke method) are used in tuning (see below).

Population dynamics

The assumed birth date and first day of the year are assumed the same in derivation of the delay-difference equation. It is therefore natural (but not strictly necessary) to tabulate catch and other data using annual accounting periods that start on the assumed biological birthday of cohorts.

Biomass dynamics

As implemented in the KLAMZ model, Schnute's (1985) delay-difference equation is:

$$B_{t+1} = (1+\rho)\,\tau_t\, B_t - \rho\,\tau_t\,\tau_{t-1}\, B_{t-1} + R_{t+1} - \rho\,\tau_t\, J_t\, R_t$$

where B_t is total biomass of individuals at the beginning of year t; ρ is Ford's growth coefficient (see below); $\tau_t = exp(-Z_t) = exp[-(F_t + M_t)]$ is the fraction of the stock that survived in year t, Z_t, F_t, and M_t are instantaneous rates for total, fishing and natural mortality; and R_t is the biomass of new recruits (at age k) at the beginning of the year. The natural mortality rate M_t may vary over time. Instantaneous mortality rates in KLAMZ model calculations are biomass-weighted averages if von Bertalanffy growth is turned on in the model. However, biomass-weighted mortality estimates in KLAMZ are the same as rates for numerical estimates under the assumption of knife-edge selectivity because all individuals are fully recruited. The growth parameter $J_t = w_{t-1,k-1} / w_{t,k}$ is the ratio of mean weight one year before recruitment (age k-1 in year t-1) and mean weight at recruitment (age k in year t).

It is not necessary to specify body weights at and prior to recruitment in the KLAMZ model (parameters v_{t-1} and V_t in Schnute 1985) because the ratio J_t and recruitment biomass contain the same information. Schnute's (1985) original delay difference equation is:

$$B_{t+1} = (1+\rho)\,\tau_t\, B_t - \rho\,\tau_t\tau_{t-1}\, B_{t-1} + w_{t+1,k}N_{t+1} - \rho\tau_t w_{t-1,k-1}\, N_t$$

To derive the equation used in KLAMZ, substitute recruitment biomass R_{t+1} for the product $w_{t+1,k}\,N_{t+1,k}$ and adjusted recruitment biomass $J_t\,R_t = (w_{t-1,k-1}/w_{t,k})\; w_{t,k}\; N_{t,k} =$ $w_{t-1,k-1}\; N_t$ in the last term on the right hand side. The advantage in using the alternate parameterization for biomass dynamic calculations in KLAMZ is that recruitment is estimated directly in units of biomass and the number of growth parameters is reduced. The disadvantage is that numbers of recruits are not estimated directly by the model. When required, numerical

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recruitments must be calculated externally as the ratio of estimated recruitment biomass and the average body weight for new recruits.

Numerical population dynamics

Growth can be turned on off so that abundance, rather than biomass, is tracked in the KLAMZ model. Set J_t=1 and ρ=0 in the delay difference equation, and use N_t (for numbers) in place of B_t to get:

$$N_{t+1} = \tau_t N_t + R_{t+1}$$

Mathematically, the assumption J_t=1 means that no growth occurs the assumption ρ=0 means that the von Bertalanffy *K* parameter is infinitely large (Schnute 1985). All tuning and population dynamics calculations in KLAMZ for biomass dynamics are also valid for numerical dynamics.

Growth

As described in Schnute (1985), biomass calculations in the KLAMZ model are based on Schnute and Fournier's (1980) re-parameterization of the von Bertalanffy growth model:

$$w_a = w_{k-1} + (w_k - w_{k-1})(1 + \rho^{1+a-k}) / (1 - \rho)$$

where $w_k=V$ and $w_{k-1}=v$. Schnute and Fournier's (1980) growth model is the same as the traditional von Bertalanffy growth model {$W_a = W_{max} [1 - exp(-K(a-t_{zero})]$ where W_{max}, *K* and t_{zero} are parameters}. The two growth models are the same because $W_{max} = (w_k - \rho\, w_{k-1})/(1-\rho)$, $K = -ln(\rho)$ and $t_{zero} = ln[(w_k - w_{k-1})/(w_k - \rho\, w_{k-1})] / ln(\rho)$.

In the KLAMZ model, the growth parameters J_t can vary with time but ρ is constant. Use of time-variable J_t values with ρ is constant is the same as assuming that the von Bertalanffy parameters W_{max} and t_{zero} change over time. Many growth patterns can be mimicked by changing W_{max} and t_{zero} (Overholtz et al., 2003). *K* is a parameter in the C++ version and, in principal, estimable. However, in most cases it is necessary to use external estimates of growth parameters as constants in KLAMZ.

Instantaneous growth rates

Instantaneous growth rate (IGR) calculations in the KLAMZ model are an extension to the original Deriso-Schnute delay difference model. IGRs are used extensively in KLAMZ for calculating catch biomass and projecting stock biomass forward to the time at which surveys occur. The IGR for new recruits depends only on growth parameters:

$$G_t^{New} = \ln\left(\frac{w_{k+1,t+1}}{w_{k,t}}\right) = \ln(1 + \rho - \rho J_t)$$

IGR for old recruits is a biomass-weighted average that depends on the current age structure and growth parameters. It can be calculated easily by projecting biomass of old recruits $S_t=B_t-R_t$ (escapement) forward one year with no mortality:

$$S_t^* = (1 + \rho)S_t - \rho\tau_{t-1}B_{t-1}$$

where the asterisk (*) means just prior to the start of the subsequent year *t*+1. By definition, the IGR for old recruits in year *t* is $G_t^{Old} = \ln\left(S_t^* / S_t\right)$. Dividing by S_t gives:

$$G_t^{Old} = \ln\left[(1+\rho) - \rho\tau_{t-1}\frac{B_{t-1}}{S_t}\right]$$

IGR for the entire stock is the biomass weighted average of the IGR values for new and old recruits:

$$G_t = \frac{R_t G_t^{New} + S_t G_t^{Old}}{B_t}$$

All IGR values are zero if growth is turned off.

Recruitment

In the Excel version of the KLAMZ model, annual recruitments are calculated $R_t = e^{\Omega_t}$ where Ω_t is a log transformed annual recruitment parameter, which is estimated in the model. In the C++ version, recruitments are calculated based on two log geometric mean recruitment parameters (μ, ι_t), and a set of annual log scale deviation parameters (ω_t):

$$\Omega_t = \mu + \iota_t + \omega_t$$

The parameter ι_t is an offset for a step function that may be zero for all years or zero for years up to a user-specified "change year" and any value (usually estimated) afterward. The user must specify the change year, which cannot be estimated. The change year might be chosen based on auxiliary information outside the model, preliminary model fits or by carrying out a set of runs using sequential change year values and to choosing the change year that provides the best fit to the data.

The deviations ω_t are constrained to average zero.[8] With the constraint, for example, estimation of μ and the set of ω_t values (1+ *n* years parameters) is equivalent to estimation of the smaller set (*n* years) of Ω_t values.

Recruitment as a rate

Recruitment is assumed in the KLAMZ model to occur at the beginning of the year. However, it is often useful to calculate recruitment biomass as an instantaneous rate for comparison to instantaneous rates for natural mortality, fishing mortality and growth. If recruitment were a continuous process, then the instantaneous rate for year *t* could be calculated:

$$r_t = \ln\left(\frac{B_{t+1}}{B_t}\right) + M_t + F_t - G_t$$

The recruitment rate can not be calculated for the last year in the model because St is not available. The KLAMZ model calculates recruitment rates for all other years automatically.

Natural mortality

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[8] The constraint is implemented by adding $L = \lambda\overline{\varpi}^2$ (where $\overline{\varpi}$ is the average deviation) to the objective function, generally with a high weighting factor ($\lambda = 1000$) so that the constraint is binding.

Natural mortality rates (M_t) are assumed constant in the Excel version of the KLAMZ model. In the C++ version, natural mortality rates may be estimated as a constant value or as a set of values that vary with time. In the model:

$$M_t = me^{\varpi_t}$$

where $m=exp(\pi)$ is the geometric mean natural mortality rate, π is a model parameter that may be estimated (in principal but not in practical terms), and ϖ_t is the log scale year-specific deviation. Deviations may be zero (turned off) so that M_t is constant, may vary in a random fashion due to autocorrelated or independent process errors, or may based on a covariate.[9] Model scenarios with zero recruitment may be initializing the parameter π to a small value (e.g. 10^{-16}) and not estimating it.

Random natural mortality process errors are effects due to predation, disease, parasitism, ocean conditions or other factors that may vary over time but are not included in the model. Calculations are basically the same as for survey process errors (see below).

Natural mortality rate covariate calculations are similar to survey covariate calculations (see below) except that the *user should standardize covariates to average zero over the time period included in the model*:

$$\kappa_t = K_t - \overline{K}$$

where κ_t is the standardized covariate, K_t is the original value, and $\overline{K}$ is the mean of the original covariate for the years in the model. *Standardization to mean zero is important because otherwise m is not the geometric mean natural mortality rate* (the convention is important in some calculations, see text).

Log scale deviations that represent variability around the geometric mean are calculated:

$$\varpi_t = \sum_{j=1}^{n} p_j \kappa_t$$

where n is the number of covariates and p_j is the parameter for covariate j. These conventions mean that the units for the covariate parameter p_j are 1/units of the original covariate, the parameter p_j measures the log scale effect of changing the covariate by one unit, and the parameter m is the log scale geometric mean.

Fishing mortality and catch

Fishing mortality rates (F_t) are calculated so that predicted and observed catch data (landings plus estimated discards in units of weight) “agree” to the extent specified by the user. It is not necessary, however, to assume that catches are measured accurately (see “Observed and predicted catch”).

Fishing mortality rate calculations in Schnute (1985) are exact but relating fishing mortality to catch in weight is complicated by continuous somatic growth throughout the year as

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[9] Another approach to using time dependent natural mortality rates is to treat estimates of predator consumption as discarded catch (see “Predator consumption as discard data”). In addition, estimates of predator abundance can be used in fishing effort calculations (see “Predator data as fishing effort”).

fishing occurs. The KLAMZ model uses a generalized catch equation that incorporates continuous growth through the fishing season. By the definition of instantaneous rates, the catch equation expresses catch as the product:

$$\hat{C}_t = F_t \overline{B}_t$$

where $\hat{C}_t$ is predicted catch weight (landings plus discard) and $\overline{B}_t$ is average biomass.

Following Chapman (1971) and Zhang and Sullivan (1988), let $X_t=G_t-F_t-M_t$ be the net instantaneous rate of change for biomass.[10] If the rates for growth and mortality are equal, then $X_t=0$, $\overline{B}_t = B_t$ and $C_t = F_t B_t$. If the growth rate G_t exceeds the combined rates of natural and fishing mortality ($F_t + M_t$), then $X_t > 0$. If mortality exceeds growth, then $X_t < 0$. In either case, with $X_t \neq 0$, average biomass is computed:

$$\overline{B}_t \approx -\frac{\left(1-e^{X_t}\right)B_t}{X_t}$$

When $X_t \neq 0$, the expression for $\overline{B}_t$ is an approximation because G_t approximates the rate of change in mean body weight due to von Bertalanffy growth. However, the approximation is reasonably accurate and preferable to calculating catch biomass in the delay-difference model with the traditional catch equation that ignores growth during the fishing season.[11] Average biomass can be calculated for new recruits, old recruits or for the whole stock by using either G_t^{New}, G_t^{Old} or G_t.

In the KLAMZ model, the modified catch equation may be solved analytically for F_t given C_t, B_t, G_t and M_t (see the "Calculating F_t" section below). Alternatively, fishing mortality rates can be calculated using a log geometric mean parameter (Φ) and a set of annual log scale deviation parameters (ψ_t):

$$F_t = e^{\Phi+\psi_t}$$

where the deviations ψ_t are constrained to average zero. When the catch equation is solved analytically, catches must be assumed known without error but the analytical option is useful when catch is zero or very near zero, or the range of fishing mortality rates is so large (e.g. minimum F=0.000001 to maximum F=3) that numerical problems occur with the alternative approach. The analytical approach is also useful if the user wants to reduce the number of parameters estimated by nonlinear optimization. In any case, the two methods should give the same results for catches known without error.

<u>Surplus production</u>

Annual surplus production is calculated "exactly" by projecting biomass at the beginning of each year forward with no fishing mortality:

$$\mathrm{B}_t^* = (1+\rho)\,\mathrm{e}^{-\mathrm{M}}\,\mathrm{B}_t - \rho\,\mathrm{e}^{-2\mathrm{M}}\mathrm{B}_{t-2} - \rho\,\mathrm{e}^{-\mathrm{M}}\,\mathrm{J}_{t-1}\,\mathrm{R}_{t-1} + R_t$$

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[10] By convention, the instantaneous rates G_t, F_t and M_t are always expressed as numbers $\geq$ 0.

[11] The traditional catch equation $C_t = F_t(1-e^{-Z_t})B_t / Z_t$ where $Z_t=F_t+M_t$ underestimates catch biomass for a given level of fishing mortality F_t and overestimates F_t for a given level of catch biomass. The errors can be substantial for fast growing fish, particularly if recent recruitments were strong.

By definition, surplus production $P_t=B^*_t-B_t$ (Jacobson et al. 2002).

Per recruit modeling

Per recruit model calculations in the Excel version of the KLAMZ simulate the life of a hypothetical cohort of arbitrary size (e.g. R=1000) starting at age k with constant M_t, F (survival) and growth (ρ and average J ($\bar{J}$)) in a population initially at zero biomass. In the first year:

$$B_1 = R$$

In the second year:

$$B_2 = (1+\rho)\,\tau\, B_1 - \rho\,\tau\,\bar{J}\,R_1$$

In the third and subsequent years:

$$B_{t+1} = (1+\rho)\,\tau\, B_t - \rho\,\tau^2\, B_{t-1}$$

This iterative calculation is carried out until the sum of lifetime cohort biomass from one iteration to the next changes by less than a small amount (0.0001). Total lifetime biomass, spawning biomass and yield in weight are calculated by summing biomass, spawning biomass and yield over the lifetime of the cohort. Lifetime biomass, spawning biomass and yield per recruit are calculated by dividing totals by initial recruitment (R).

Status determination variables

The user may specify a range of years (e.g. the last three years) to use in calculating recent average fishing mortality $\bar{F}_{\mathrm{Re}\,cent}$ and biomass $\bar{B}_{\mathrm{Re}\,cent}$ levels. These status determination variables are used in calculation of status ratios such as $\bar{F}_{\mathrm{Re}\,cent} / F_{MSY}$ and $\bar{B}_{\mathrm{Re}\,cent}$ /B_{MSY}.

Goodness of Fit and Parameter Estimation

Parameters estimated in the KLAMZ model are chosen to minimize an objective function based on a sum of weighted negative log likelihood (NLL) components:

$$\Xi = \sum_{v=1}^{N_\Xi} \lambda_v L_v$$

where N_Ξ is the number of NLL components (L_v) and the λ_v are emphasis factors used as weights. The objective function Ξ may be viewed as a NLL or a negative log posterior (NLP) distribution, depending on the nature of the individual L_v components and modeling approach. Except during sensitivity analyses, weighting factors for objective function components (λ_v) are usually set to one. An arbitrarily large weighting factor (e.g. λ_v =1000) is used for “hard” constraints that must be satisfied in the model. Arbitrarily small weighting factors (e.g. λ_v =0.0001) can be used for “soft” model-based constraints. For example, an internally estimated spawner-recruit curve or surplus production curve might be estimated with a small weighting factor to summarize stock-recruit or surplus production results with minimal influence on biomass, fishing mortality and other estimates from the model. Use of a small weighting factor for an internally estimated surplus production or stock-recruit curve is equivalent to fitting a curve to model estimates of biomass and recruitment or surplus production in the output file, after the model is fit (Jacobson et al. 2002).

Likelihood component weights vs. observation-specific weights

Likelihood component weights (λ_v) apply to entire NLL components. Entire components are often computed as the sum of a number of individual NLL terms. The NLL for an entire survey, for example, is composed of NLL terms for each of the annual survey observations. In KLAMZ, observation-specific (for data) or instance-specific (for constraints or prior information) weights (usually w_j for observation or instance j) can be specified as well. Observation-specific weights for a survey, for example, might be use to increase or decrease the importance of one or more observations in calculating goodness of fit.

NLL kernels

NLL components in KLAMZ are generally programmed as "concentrated likelihoods" to avoid calculation of values that do not affect derivatives of the objective function.[12] For $x \sim N(\mu, \sigma^2)$, the complete NLL for one observation is:

$$L = \ln(\sigma) + \ln\left(\sqrt{2\pi}\right) + 0.5\left(\frac{x-u}{\sigma}\right)^2$$

The constant $\ln\left(\sqrt{2\pi}\right)$ can always be omitted because it does not affect derivatives. If the standard deviation is known or assumed known, then $\ln(\sigma)$ can be omitted as well because it is a constant that does not affect derivatives. In such cases, the concentrated negative log likelihood is:

$$L = 0.5\left(\frac{x-\mu}{\sigma}\right)^2$$

If there are N observations with possible different variances (known or assumed known) and possibly different expected values:

$$L = 0.5\sum_{i=1}^{N}\left(\frac{x_i-\mu_i}{\sigma_i}\right)^2$$

If the standard deviation for a normally distributed quantity is not known and is (in effect) estimated by the model, then one of two equivalent calculations is used. Both approaches assume that all observations have the same variance and standard deviation. The first approach is used when all observations have the same weight in the likelihood:

$$L = 0.5N\ln\left[\sum_{i=1}^{N}\left(x_i - u\right)^2\right]$$

where N is the number of observations. The second approach is equivalent but used when the weights for each observation (w_i) may differ:

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[12] Unfortunately, concentrated likelihood calculations cannot be used with MCMC and other Bayesian approaches to characterizing posterior distributions. Therefore, in the near future, concentrated NLL calculations will be replaced by calculations for the entire NLL. At present, MCMC calculations in KLAMZ are not useful.

$$L = \sum_{i=1}^{N} w_i \left[\ln(\sigma) + 0.5 \left(\frac{x_i - u}{\sigma} \right)^2 \right]$$

In the latter case, the maximum likelihood estimator:

$$\hat{\sigma} = \sqrt{\frac{\sum_{i=1}^{N} (x_i - \hat{x})^2}{N}}$$

(where $\hat{x}$ is the average or predicted value from the model) is used for σ. The maximum likelihood estimator is biased by $N/(N-d_f)$ where d_f is degrees of freedom for the model. The bias may be significant for small sample sizes but d_f is usually unknown.

Landings, discards, catch

Discards are from external estimates (d_t) supplied by the user. If $d_t \geq 0$, then the data are used as the ratio of discard to landed catch so that:

$$D_t = L_t \Delta_t$$

where $\Delta_t = D_t / L_t$ is the discard ratio. If $d_t < 0$ then the data are treated as discard in units of weight:

$$D_t = abs(d_t).$$

In either case, total catch is the sum of discards and landed catch ($C_t = L_t + D_t$). It is possible to use discards in weight $d_t < 0$ for some years and discard as proportions $d_t > 0$ for other years in the same model run. If catches are estimated (see below) so that the estimated catch $\hat{C}_t$ does not necessarily equal observed landings plus discard, then estimated landings are computed:

$$\hat{L}_t = \frac{\hat{C}_t}{1 + \Delta_t}$$

and estimated discards are:

$$\hat{D}_t = \Delta_t \hat{L}_t.$$

Calculating F_t

As described above, fishing mortality rates may be estimated based on the parameters Φ and ψ_t to satisfy a NLL for observed and predicted catches:

$$L = 0.5 \sum_{t=0}^{N} w_t \left(\frac{\hat{C}_t - C_t}{\kappa_t} \right)^2$$

where the standard error $\kappa_t = CV_{catch} \hat{C}_t$ with CV_{catch} and weights are w_t supplied by the user. The weights can be used, for example, if catch data in some years are less precise than in others. Using observation specific weights, any or every catch in the time series can potentially be estimated.

The other approach to calculating F_t values is by solving the generalized catch equation (see above) iteratively. Subtracting predicted catch from the generalized catch equation gives:

$$g\left(F_t\right) = C_t + \frac{F_t\left(1 - e^{X_t}\right)}{X_t} B_t = 0$$

where $X_t=G_t-M_t-F_t$. If $X_t=0$, then $\overline{B}_t = B_t$ and $F_t=C_t/B_t$.

If $X_t \neq 0$, then the Newton-Raphson algorithm is used to solve for F_t (Kennedy and Gentle 1980). At each iteration of the algorithm, the current estimate F_t^i is updated using:

$$F_t^{i+1} = F_t^i - \frac{g\left(F_t^i\right)}{g'\left(F_t^i\right)}$$

where $g'\left(F_t^i\right)$ is the derivative F_t^i. Omitting subscripts, the derivative is:

$$g^{'}\left(F\right) = -\frac{Be^{-F}\left[\left(e^F - e^\gamma\right)\gamma + e^\gamma F\gamma - e^\gamma F^2\right]}{X^2}$$

where $\gamma=G-M_t$. Iterations continue until $g\left(F_t^i\right)$ and $abs\left[g\left(F_t^{i+1}\right) - g\left(F_t^{i+1}\right)\right]$ are both less than a small number (e.g. ≤ 0.00001).

Initial values are important in algorithms that solve the catch equation numerically (Sims 1982). If $M_t+F_t > G_t$ so that $X_t < 0$, then the initial value F_t^0 is calculated according to Sims (1982). If $M_t+F_t < G_t$ so that $X_t > 0$, then initial values are calculated based on a generalized version of Pope's cohort analysis (Zhang and Sullivan 1988):

$$F_t^0 = \gamma_t - \ln\left[\frac{\left(B_t e^{0.5\gamma_t} - C_t\right)e^{0.5\gamma_t}}{B_t}\right]$$

F for landings versus F for discards

The total fishing mortality rate for each year can be partitioned into a component due to landed catch ${}^{L}F_t = \frac{D_t}{C_t} F_t$, and a component due to discard ${}^{D}F_t = \frac{L_t}{C_t} F_t$.

Predator consumption as discard data

In modeling population dynamics of prey species, estimates of predator consumption can be treated like discard in the KLAMZ model as a means for introducing time dependent natural mortality. Consider a hypothetical example with consumption data (mt y^{-1}) for three important predators. If the aggregate consumption data are included in the model as "discards", then the fishing mortality rate for discards ${}^{d}F_t$ (see above) would be an estimate of the component of natural mortality due to the three predators. In using this approach, the average level of natural mortality m would normally be reduced (e.g. so that $m_{new} + {}^{d}\overline{F} = m_{old}$) or estimated to account for the portion of natural mortality attributed to bycatch.

Surplus production calculations are harder to interpret if predator consumption is treated as discard data because surplus production calculations assume that $F_t=0$ (see above) and because surplus production is defined as the change in biomass from one year to the next in the absence of fishing (i.e. no landings or bycatch). However, it may be useful to compare surplus production at a given level of biomass from runs with and without consumption data as a means of estimating maximum changes in potential fishery yield if the selected predators were eliminated (assuming no change in disease, growth rates, predation by other predators, etc.).

Effort calculations

Fishing mortality rates can be tuned to fishing effort data for the “landed” catch (i.e. excluding discards). Years with non-zero fishing effort used in the model must also have landings greater than zero. Assuming that effort data are lognormally distributed, the NLL for fishing effort is:

$$NLL = 0.5\sum_{y=1}^{n_{eff}} w_y \left[\frac{\ln\left(E_y / \hat{E}_y\right)}{\sigma} \right]^2$$

where w_y is an observation-specific weight, n_{eff} is the number of active effort observations (i.e. with $w_y > 0$), E_y and $\bar{E}_y$ are observed and predicted fishing effort data, and the log scale variance σ is a constant calculated from a user-specified CV.

Predicted fishing effort data are calculated:

$$\hat{E}_y = \zeta F_y^{\vartheta}$$

where $\zeta = e^u$, $\vartheta = e^b$, and u and b are parameters estimated by the model. If the parameter b is not estimated, then $\vartheta=1$ so that the relationship between fishing effort and fishing mortality is linear. If the parameter b is estimated, then $\vartheta \neq 1$ and the relationship is a power function.

Predator data as fishing effort

As described under “Predator consumption as discard data”, predator consumption data can be treated as discard. If predator abundance data are available as well, and assuming that mortality due predators is a linear function of the predator-prey ratio, then both types of data may be used together to estimate natural mortality. The trick is to: 1) enter the predator abundance data as fishing effort; 2) enter the actual fishery landings as “discard”; 3) enter predator consumption estimates of the prey species as “landings” so that the fishing effort data refer to the predator consumption data; 4) use an option in the model to calculate the predator-prey ratio for use in place of the original predator abundance “fishing effort” data; and 5) tune fishing mortality rates for landings (a.k.a. predator consumption) to fishing effort (a.k.a. predator-prey ratio).

Given the predator abundance data κ_y, the model calculates the predator-prey ratio used in place of fishing effort data (E_y) as:

$$E_y = \frac{\kappa_y}{B_y}$$

where B_y is the model’s current estimate of total (a.k.a “prey”) biomass. Subsequent calculations with E_y and the model’s estimates of “fishing mortality” (F_y, really a measure of natural mortality) are exactly as described above for effort data. In using this approach, it is probably advisable to reduce *m* (the estimate of average mortality in the model) to account for the proportion of natural mortality due to predators included in the calculation. Based on experience to date, natural mortality due to consumption by the suite of predators can be estimated but only if *m* is assumed known.

Initial population age structure

In the KLAMZ model, old and new recruit biomass during the first year (R_1 and $S_1 = B_1 - R_1$) and biomass prior to the first year (B_0) are estimated as log scale parameters. Survival in the year prior to the first year ("year 0") is $\tau_0 = e^{-F_0 - M_1}$ with F_0 chosen to obtain catch C_0 (specified as data) from the estimated biomass B_0. IGRs during year 0 and year 1 are assumed equal ($G_0 = G_1$) in catch calculations.

Biomass in the second year of as series of delay-difference calculations depends on biomass (B_0) and survival (τ_0) in year 0:

$$\mathrm{B}_2 = (1+\rho)\,\tau_1\,\mathrm{B}_1 - \rho\,\tau_1\,\tau_0\,\mathrm{B}_0 + \mathrm{R}_2 - \rho\,\tau_1\,\mathrm{J}_1\,\mathrm{R}_1$$

There is, however, there is no direct linkage between B_0 and escapement biomass ($S_1 = B_1 - R_1$) at the beginning of the first year.

The missing link between B_0, S_1 and B_1 means that the parameter for B_0 tends to be relatively free and unconstrained by the underlying population dynamics model. In some cases, B_0 can be estimated to give good fit to survey and other data, while implying unreasonable initial age composition and surplus production levels. In other cases, B_0 estimates can be unrealistically high or low implying, for example, unreasonably high or low recruitment in the first year of the model (R_1). Problems arise because many different combinations of values for R_1, S_1 and B_0 give similar results in terms of goodness of fit. This issue is common in stock assessment models that use forward simulation calculations because initial age composition is difficult to estimate. It may be exacerbated in delay-difference models because age composition data are not used.

The KLAMZ model uses two constraints to help estimate initial population biomass and initial age structure.[13] The first constraint links IGRs for escapement (G^{Old}) in the first years to a subsequent value. The purpose of the constraint is to ensure consistency in average growth rates (and implicit age structure) during the first few years. For example, if IGRs for the first n_G years are constrained[14], then the NLL for the penalty is:

$$L_G = 0.5 \sum_{t=1}^{n_G} \left[\frac{\ln\left(G_t^{Old} \big/ G_{n_G+1}^{Old}\right)}{\sigma_G} \right]^2$$

where the standard deviation σ_G is supplied by the user. It is usually possible to use the standard deviation of Q_t^{Old} for later years from a preliminary run to estimate σ_G for the first few years. The constraint on initial IGRs should probably be "soft" and non-binding ($\lambda \approx 1$) because there is substantial natural variation in somatic growth rates due to variation in age composition.

The second constraint links B_0 to S_1 and ensures conservation of mass in population dynamics between years 0 and 1. In other words, the parameter for escapement biomass in year 1 is constrained to match an approximate projection of the biomass in year 0, accounting for growth, and natural and fishing mortality. The constraint is intended to be binding and satisfied

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[13] Quinn and Deriso (1999) describe another approach attributed to a manuscript by C. Walters.

[14] Normally, $n_G \leq 2$.

exactly (e.g. λ=1000) because incompatible values of S_1 and B_0 are biologically impossible. In calculations:

$$S_1^p = B_0 e^{G_1 - F_0 - M_1}$$

where S_1^p is the projected escapement in year 1 and B_0 is the model's estimate of total biomass in year 0. The instantaneous rates for growth and natural mortality from year 1 (G_1 and M_1) are used in place of G_0 and M_0 because the latter are unavailable. The NLL for the constraint:

$$L = \left[\ln\left(\frac{S_1^p}{S_1}\right)^2 \right]^2 + \left(S_1^p - S_1\right)^2$$

uses a log scale sum of squares and an arithmetic sum of squares. The former is effective when S_1 is small while the latter is effective when S_1 is large. Constants and details in calculation of NLL for the constraint are not important because the constraint is binding (e.g. λ=1000).

Equilibrium pristine biomass

It may be useful to constrain the biomass estimate for the first year in a model run towards an estimate of equilibrium pristine biomass if, for example, stock dynamics tend to be stable and catch data are available for the first years of the fishery, or as an alternative to the approach described above for initializing the age structure of the simulated population in the model. Equilibrium pristine biomass $\tilde{B}_0$ is calculated based on the model's estimate of average recruitment and with no fishing mortality (calculations are similar to those described under "Per-recruit modeling" except that average recruitment is assumed in each year).[15] The NLL term for the constraint is:

$$L = \ln\left(\frac{\tilde{B}_0}{B_0}\right)^2$$

Pristine equilibrium biomass is used as a hard constraint with a high emphasis factor (λ) so that the variance and constants normally used in NLL calculations are not important.

Estimating natural mortality

As described above, natural mortality calculations involve a parameter for the geometric mean value (m) and time dependent deviations (ϖ_t, which may or may not be turned on). Constraints on natural mortality process errors and natural mortality covariates can be used to help estimate the time dependent deviations and overall trend. The geometric mean natural mortality rate is usually difficult to estimate and best treated as a known constant. However, in the C++ version of the KLAMZ model, $m=e^{\pi}$ (where π is an estimable parameter in the model) and estimates of m can be conditioned on the constraint:

$$L = 0.5\left[\frac{\ln\left(w/w_{Target}\right)}{\sigma_{\varpi}}\right]^2$$

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[15] Future versions of the KLAMZ model will allow equilibrium initial biomass to be calculated based on other recruitment values and for a user-specified level of F (Butler et al. 2003).

where w_{Target} is a user supplied mean or target value and σ_{ϖ} is a log scale standard deviation. The standard deviation is calculated from an arithmetic scale CV supplied by the user. Upper and lower bounds for *m* may be specified as well.

Goodness of fit for trend data

Assuming lognormal errors[16], the NLL used to measure goodness-of-fit to "survey" data that measure trends in abundance or biomass (or survival, see below) is:

$$L = 0.5 \sum_{j=1}^{N_v} \left[\frac{\ln\left(I_{v,j} \big/ \hat{I}_{v,j} \right)}{\sigma_{v,j}} \right]^2$$

where $I_{v,t}$ is an index datum from survey *v*, hats "^" denote model estimates, $\sigma_{v,j}$ is a log scale standard error (see below), and N_v is the number of observations. There are two approaches to calculating standard errors for log normal abundance index data in KLAMZ and it is possible to use different approaches for different types of abundance index data in the same model (see below).

Standard errors for goodness of fit

In the first approach, all observations for one type of abundance index share the same standard error, which is calculated based on overall goodness of fit. This approach implicitly estimates the standard error based on goodness of fit, along with the rest of the parameters in the model (see "NLL kernels" above).

In the second approach, each observation has a potentially unique standard error that is calculated based on its CV. The second approach calculates log scale standard errors from arithmetic CVs supplied as data by the user (Jacobson et al. 1994):

$$\sigma_{v,t} = \sqrt{\ln\left(1 + CV_{v,t}^2\right)}$$

Arithmetic CV's are usually available for abundance data. It may be convenient to use $CV_{v,t}$=1.31 to get $\sigma_{v,t}$=1.

There are advantages and disadvantages to both approaches. CV's carry information about the relative precision of abundance index observations. However, CV's usually overstate the precision of data as a measure of fish abundance[17] and may be misleading in comparing the precision of one sort of data to another as a measure of trends in abundance (e.g. in contrasting standardized LPUE that measure fishing success, but not abundance, precisely with survey data that measure trends in fish abundance directly, but not precisely). Standard errors estimated

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[16] Abundance indices with statistical distributions other than log normal may be used as well, but are not currently programmed in the KLAMZ model. For example, Butler et al. (2003) used abundance indices with binomial distributions in a delay-difference model for cowcod rockfish. The next version of KLAMZ will accommodate presence-absence data with binomial distributions.

[17] The relationship between data and fish populations is affected by factors (process errors) that are not accounted for in CV calculations.

implicitly are often larger and more realistic, but assume that all observations in the same survey are equally reliable.

Predicted values for abundance indices

Predicted values for abundance indices are calculated:

$$\hat{I}_{v,t} = Q_v A_{v,t}$$

where Q_v is a survey scaling parameter (constant here but see below) that converts units of biomass to units of the abundance index. $A_{v,t}$ is available biomass at the time of the survey.

In the simplest case, available biomass is:

$$A_{v,t} = s_{v,New} R_t e^{-X_t^{New} \Delta_{v,t}} + s_{v,Old} S_t e^{-X_t^{Old} \Delta_{v,t}}$$

where $s_{v,New}$ and $s_{v,Old}$ are survey selectivity parameters for new recruits (R_t) and old recruits (S_t); $X_t^{New} = G_t^{New} - F_t - M_t$ and $X_t^{Old} = G_t^{Old} - F_t - M_t$; $j_{v,t}$ is the Julian date at the time of the survey, and $\Delta_{v,t}=j_{v,t}/365$ is the fraction of the year elapsed at the time of the survey.

Survey selectivity parameter values ($s_{v,New}$ and $s_{v,Old}$) are specified by the user and must be set between zero and one. For example, a survey for new recruits would have $s_{v,New}$=1 and $s_{v,Old}$=0. A survey that measured abundance of the entire stock would have $s_{v,New}$=1 and $s_{v,Old}$=1.

Terms involving $\Delta_{v,t}$ are used to project beginning of year biomass forward to the time of the survey, making adjustments for mortality and somatic growth.[18] As described below, available biomass $A_{v,t}$ is adjusted further for nonlinear surveys, surveys with covariates and surveys with time variable $Q_{v,t}$.

Scaling parameters (Q) for log normal abundance data

Scaling parameters for surveys with lognormal statistical errors were computed using the maximum likelihood estimator:

$$Q_v = e^{\frac{\sum_{i=1}^{N_v} \left[\ln\left(\frac{I_{v,i}}{A_{v,i}} \right) \Big/ \sigma_{v,j}^2 \right]}{\sum_{j=1}^{N_j} \left(1 \big/ \sigma_{v,j}^2 \right)}}$$

where N_v is the number of observations with individual weights greater than zero. The closed form maximum likelihood estimator gives the same answer as if scaling parameters are estimated as free parameters in the assessment model assuming lognormal survey measurement errors.

Survey covariates

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[18] It may be important to project biomass forward if an absolute estimate of biomass is available (e.g. from a hydroacoustic or daily egg production survey), if fishing mortality rates or high or if the timing of the survey varies considerably from year to year.

Survey scaling parameters may vary over time based on covariates in the KLAMZ model. The survey scaling parameter that measures the relationship between available biomass and survey data becomes time dependent:

$$\hat{I}_{v,t} = Q_{v,t} A_{v,t}$$

and

$$Q_{v,t} = Q_v e^{\sum_{r=1}^{n_v} d_{r,t} \theta_r}$$

with n_v covariates for the survey and parameters θ_r estimated in the model. Covariate effects and available biomass are multiplied to compute an adjusted available biomass:

$$A'_{v,t} = A_{v,t} e^{\sum_{r=1}^{n_v} d_{r,t} \theta_r}$$

The adjusted available biomass $A'_{v,t}$ is used instead of the original value $A_{v,t}$ in the closed form maximum likelihood estimator described above.

Covariates might include, for example, a dummy variable that represents changes in survey bottom trawl doors or a continuous variable like average temperature data if environmental factors affect distribution and catchability of fish schools. Dummy variables are usually either 0 or 1, depending on whether the effect is present in a particular year. With dummy variables, Q_v is the value of the survey scaling parameter with no intervention ($d_{r,t}$=0).

For ease in interpretation of parameter estimates for continuous covariates (e.g. temperature data), it is useful to center covariate data around the mean:

$$d_{r,t} = d'_{r,t} - \overline{d'_r}$$

where $d'_{r,t}$ is the original covariate. When covariates are continuous and mean-centered, Q_v is the value of the survey scaling parameter under average conditions ($d_{r,t}$=0) and units for the covariate parameter are easy to interpret (for example, units for the parameter are 1/ ^{o}C if the covariate is mean centered temperature in ^{o}C).

It is possible to use a survey covariate to adjust for differences in relative stock size from year to year due to changes in the timing of a survey. However, this adjustment may be made more precisely by letting the model calculate $\Delta_{v,t}$ as described above, based on the actual timing data for the survey during each year.

Nonlinear abundance indices

With nonlinear abundance indices, and following Methot (1990), the survey scaling parameter is a function of available biomass:

$$Q_{v,t} = Q_v A_{v,t}^{\Gamma}$$

so that:

$$\hat{I}_{v,t} = \left(Q_v A_{v,t}^{\Gamma}\right) A_{v,t}$$

Substituting $e^{\gamma}=\Gamma+1$ gives the equivalent expression:

$$\hat{I}_{v,t} = Q_v A_{v,t}^{e^{\gamma}}$$

where γ is a parameter estimated by the model and the survey scaling parameter is no longer time dependent. In calculations with nonlinear abundance indices, the adjusted available biomass:

$$A'_{v,t} = A_{v,t}^{e^{\gamma}}$$

is computed first and used in the closed form maximum likelihood estimator described above to calculate the survey scaling parameter. In cases where survey covariates are also applied to a nonlinear index, the adjustment for nonlinearity is carried out first.

Survey Q process errors

The C++ version of the KLAMZ model can be used to allow survey scaling parameters to change in a controlled fashion from year to year (NEFSC 2002):

$$Q_{v,t} = Q_v e^{\varepsilon_{v,t}}$$

where the deviations $\varepsilon_{v,t}$ are constrained to average zero. Variation in survey Q values is controlled by the NLL penalty:

$$L = 0.5 \sum_{j=1}^{N_v} \left[\frac{\varepsilon_{v,j}}{\sigma_v} \right]^2$$

where the log scale standard deviation σ_v based on an arithmetic CV supplied by the user (e.g. see NEFSC 2002). In practice, the user increases or decreases the amount of variability in Q by decreasing or increasing the assumed CV.

Survival ratios as surveys

In the C++ version of KLAMZ, it is possible to use time series of survival data as "surveys". For example, an index of survival might be calculated using survey data and the Heinke method (Ricker 1975) as:

$$A_t = \frac{I_{k+1,t+1}}{I_{k,t}}$$

so that the time series of A_t estimates are data that may potentially contain information about scale or trends in survival. Predicted values for an a survival index are calculated:

$$\hat{A}_t = e^{-Z_t}$$

After predicted values are calculated, survival ratio data are treated in the same way as abundance data (in particular, measurement errors are assumed to be lognormal). Selectivity parameters are ignored for survival data but all other features (e.g. covariates, nonlinear scaling relationships and constraints on *Q*) are available.

Recruitment models

Recruitment parameters in KLAMZ may be freely estimated or estimated around an internal recruitment model, possibly involving spawning biomass. An internally estimated recruitment model can be used to reduce variability in recruitment estimates (often necessary if data are limited), to summarize stock-recruit relationships, or to make use of information about recruitment in similar stocks. There are four types of internally estimated recruitment models in KLAMZ: 1) random (white noise) variation around a constant or time dependent mean modeled as a step function; 2) random walk (autocorrelated) variation around a constant or time dependent mean modeled as a step function; 3) random variation around a Beverton-Holt recruitment model; and 4) random variation around a Ricker recruitment model. The user must specify a type of recruitment model but the model is not active unless the likelihood component for the recruitment model is turned on ($\lambda > 0$).

The first step in recruit modeling is to calculate the expected log recruitment level *E[ln(R$_t$)]* given the recruitment model. For random variation around a constant mean, the expected log recruitment level is the log geometric mean recruitment:

$$E\left[\ln\left(R_t\right)\right] = \sum_{j=1}^{N} \ln\left(R_j\right) \Big/ N$$

For a random walk around a constant mean recruitment, the expected log recruitment level is the logarithm of recruitment during the previous year:

$$E\left[\ln\left(R_t\right)\right] = \ln\left(R_{t-1}\right)$$

with no constraint on recruitment during the first year R_1.

For the Beverton-Holt recruitment model, the expected log recruitment level is:

$$E\left[\ln\left(R_t\right)\right] = \ln\left[e^a T_{t-\ell} \big/ \left(e^b + T_{t-\ell}\right)\right]$$

where $a=e^{\alpha}$ and $b=e^{\beta}$, the parameters α and β are estimated in the model, T_t is spawning biomass, and ℓ is the lag between spawning and recruitment. Spawner-recruit parameters are estimated as log transformed values (e^{α} and e^{β}) to enhance model stability and ensure the correct sign of values used in calculations. Spawning biomass is:

$$T_t = m_{new} R_t + m_{old} S_t$$

where m_{new} and m_{old} are maturity parameters for new and old recruits specified by the user. For the Ricker recruitment model, the expected log recruitment level is:

$$E\left[\ln\left(R_t\right)\right] = \ln\left(S_{t-\ell}\, e^{a - bS_{t-\ell}}\right)$$

where $a=e^{\alpha}$ and $b=e^{\beta}$, and the parameters α and β are estimated in the model.

Given the expected log recruitment level, log scale residuals for the recruitment model are calculated:

$$r_t = \ln\left(R_t\right) - E\left[\ln\left(R_t\right)\right]$$

Assuming that residuals are log normal, the NLL for recruitment residuals is:

$$L = \sum_{t=t_{first}}^{N} w_t \left[\ln\left(\sigma_r\right) + 0.5 \left(r_t \big/ \sigma_r \right)^2 \right]$$

where λ_t is an instance-specific weight usually set equal one. The additional term in the NLL [$ln(\sigma_r)$] is necessary because the variance σ_r^2 is estimated internally, rather than specified by the user.

The log scale variance for residuals is calculated using the maximum likelihood estimator:

$$\sigma_r^2 = \frac{\sum_{j=t_{first}}^{N} r_j}{N}$$

where N is the number of residuals. For the recruitment model with constant variation around a mean value, t_{first}=1. For the random walk recruitment model, t_{first}=2. For the Beverton-Holt and Ricker models, $t_{first}=\ell+1$ and the recruit model imposes no constraint on variability of recruitment during years 1 to ℓ (see below). The biased maximum likelihood estimate for σ^2 (with N in the divisor instead of the degrees of freedom) is used because actual degrees of freedom are unknown. The variance term σ^2 is calculated explicitly and stored because it is used below.

Constraining the first few recruitments

It may be useful to constrain the first ℓ years of recruitments when using either the Beverton-Holt or Ricker models if the unconstrained estimates for early years are erratic. In the KLAMZ model, this constraint is calculated:

$$NLL = \sum_{t=1}^{t_{first}-1} w_t \left\{ \ln(\sigma_r + 0.5\left[\frac{\ln\left(R_t / E\left(R_{t_{first}}\right)\right)}{\sigma_r}\right]^2 \right\}$$

where t_{first} is the first year for which expected recruitment $E(R_l)$ can be calculated with the spawner-recruit model. In effect, recruitments that not included in spawner-recruit calculations are constrained towards the first spawner-recruit prediction. The standard deviation is the same as used in calculating the NLL for the recruitment model.

Prior information about the absolute value abundance index scaling parameters (Q)

A constraint on the absolute value one or more scaling parameters (Q_v) for abundance or survival indices may be useful if prior information is available (e.g. NEFSC 2000; NEFSC 2001; NEFSC 2002). In the Excel version, it is easy to program these (and other) constraints in an *ad-hoc* fashion as they are needed. In the AD Model Builder version, log normal and beta distributions are preprogrammed for use in specifying prior information about Q_v for any abundance or survival index.

The user must specify which surveys have prior distributions, minimum and maximum legal bounds (q_{min} and q_{max}), the arithmetic mean $(\bar{q})$ and the arithmetic CV for the prior the distribution. Goodness of fit for Q_v values outside the bounds (q_{min}, q_{max}) are calculated:

$$L = \left| \begin{array}{l} 10000\,(Q_v - q_{\max})^2 \ if \ Q_v \geq q_{\max} \\ 10000\,(q_{\min} - Q_v)^2 \ if \ Q_v \leq q_{\min} \end{array} \right.$$

Goodness of fit for Q_v values inside the legal bounds depend on whether the distribution of potential values is log normal or follows a beta distribution.

Lognormal case

Goodness of fit for lognormal Q_v values within legal bounds is:

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

where the log scale standard deviation $\varphi = \sqrt{\ln(1 + CV)}$ and $\tau = \ln(\overline{q}) - \frac{\varphi^2}{2}$ is the mean of the corresponding log normal distribution.

Beta distribution case

The first step in calculation goodness of fit for Q_v values with beta distributions is to calculate the mean and variance of the corresponding "standardized" beta distribution:

$$\overline{q}' = \frac{\overline{q} - q_{\min}}{D}$$

and

$$Var(q') = \left(\frac{\overline{q}\ CV}{D}\right)^2$$

where the range of the standardized beta distribution is $D=q_{max}-q_{min}$. Equating the mean and variance to the estimators for the mean and variance for the standardized beta distribution (the "method of moments") gives the simultaneous equations:

$$\overline{q}' = \frac{a}{a+b}$$

and

$$Var(q') = \frac{ab}{(a+b)^2(a+b+1)}$$

where a and b are parameters of the standardized beta distribution.[19] Solving the simultaneous equations gives:

$$b = \frac{(\overline{q}' - 1)\left[Var(q') + (\overline{q}' - 1)\overline{q}'\right]}{Var(q')}$$

and:

$$a = \frac{b\overline{q}'}{1 - \overline{q}'}$$

Goodness of fit for beta Q_v values within legal bounds is calculated with the NLL:

$$L = (a-1)\ln(Q'_v) + (b-1)\ln(1 - Q'_v)$$

where $Q'_v = Q_v / (Q_v - q_{\min})$ is the standardized value of the survey scaling parameter Q_v.

Prior information about relative abundance index scaling parameters (*Q-ratios*)

Constraints on "Q-ratios" can be used in fitting models if some information about the relative values of scaling parameters for two abundance indices is available. For example, ASMFC

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[19] If x has a standardized beta distribution with parameters a and b, then the probability of x is

$$P(x) = \frac{x^{a-1}(1-x)^{b-1}}{\Gamma(a,b)}.$$

(2001, p. 46-47) assumed that the relative scaling parameters for recruit and post-recruit lobsters taken in the same survey was either 0.5 or 1. If both indices are from the same survey cruise (e.g. one index for new recruits and one index for old recruits in the same survey), then assumptions about q-ratios are analogous to assumptions about the average selectivity of the survey of the survey for new and old recruits.

Q-ratio constraints tend to stabilize and have strong effects on model estimates. ASMFC (2001, p. 274) found, for example, that goodness of fit to survey data, abundance and fishing mortality estimates for lobster changed dramatically over a range of assumed q-ratio values.

To use q-ratio information in the KLAMZ model, the user must identify two surveys, a target value for the ratio of their Q values, and a CV for differences between the models estimated q-ratio and the target value. For example, if the user believes that the scaling parameters for abundance index 1 and abundance index 3 is 0.5, with a CV=0.25 for uncertainty in the prior information then the model's estimate of the q-ratio is $\rho=Q_1/Q_3$. The goodness of fit calculation is:

$$L = 0.5\left(\frac{\ln(\rho/\tau)}{\sigma}\right)^2$$

where τ is the target value and the log scale standard deviation σ is calculated from the arithmetic CV supplied by the user.

Normally, a single q-ratio constraint would be used for the ratio of new and old recruits taken during the same survey operation. However, in KLAMZ any number of q-ratio constraints can be used simultaneously and the scaling parameters can be for any two indices in the model.

Surplus production modeling

Surplus production models can be fit internally to biomass and surplus production estimates in the model (Jacobson et al. 2002). Models fit internally can be used to constrain estimates of biomass and recruitment, to summarize results in terms of surplus production, or as a source of information in tuning the model. The NLL for goodness of fit assumes normally distributed process errors in the surplus production process:

$$L = 0.5\sum_{j=1}^{N_P}\left(\frac{\tilde{P}_j - P_j}{\sigma}\right)^2$$

where N_p is the number of surplus production estimates (number of years less one), $\tilde{P}_t$ is a predicted value from the surplus production curve, P_t is the assessment model estimate, and the standard deviation σ is supplied by the user based, for example, on preliminary variances for surplus production estimates.[20] Either the symmetrical Schaefer (1957) or asymmetric Fox (1970) surplus production curve may be used to calculate $\tilde{P}_t$ (Quinn and Deriso 1999).

It may be important to use a surplus production curve that is compatible with recruitment patterns or assumptions about the underlying spawner-recruit relationship. More research is

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[20] Variances in NLL for surplus production-biomass models are a subject of ongoing research. The advantage in assuming normal errors is that negative production values (which occur in many stocks, e.g. Jacobson et al. 2001) are accommodated. In addition, production models can be fit easily by linear regression of P_t on B_t and B_t^2 with no intercept term. However, variance of production estimate residuals increases with predicted surplus production. Therefore, the current approach to fitting production curves in KLAMZ is not completely satisfactory.

required, but the asymmetric shape of the Fox surplus production curve appears reasonably compatible with the assumption that recruitment follows a Beverton-Holt spawner-recruit curve (Mohn and Black 1998). In contrast, the symmetric Schaefer surplus production model appears reasonably compatible with the assumption that recruitment follows a Ricker spawner-recruit curve.

The Schaefer model has two log transformed parameters that are estimated in KLAMZ:

$$\tilde{P}_t = e^{\alpha} B_t - e^{\beta} B_t^2$$

The Fox model also has two log transformed parameters:

$$\tilde{P}_t = -e\left(e^{e^{\alpha}}\right)\frac{B_t}{e^{\beta}}\log\left(\frac{B_t}{e^{\beta}}\right)$$

See Quinn and Deriso (1999) for formulas used to calculate reference points (F_{MSY}, B_{MSY}, MSY, and K) for both surplus production models.

Catch/biomass

Forward simulation models like KLAMZ may tend to estimate absurdly high fishing mortality rates, particularly if data are limited. The likelihood constraint used to prevent this potential problem is:

$$L = 0.5\sum_{t=0}^{N}\left(d_t^2 + q^2\right)$$

where:

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

and

with the threshold value κ normally set by the user to about 0.95. Values for κ can be linked to maximum F values using the modified catch equation described above. For example, to use a maximum fishing mortality rate of about $F \approx 4$ with M=0.2 and G=0.1 (maximum X=4+0.2-0.1=4.1), set $\kappa \approx F/X(1-e^{-X})$=4 / 4.1 (1-$e^{-4}$)=0.96.

Uncertainty

The AD Model Builder version of the KLAMZ model automatically calculates variances for parameters and quantities of interest (e.g. R_t, F_t, B_t, F_{MSY}, B_{MSY}, $\overline{F}_{\mathrm{Re}\,cent}$, $\overline{B}_{\mathrm{Re}\,cent}$, $\overline{F}_{\mathrm{Re}\,cent} / F_{MSY}$, $\overline{B}_{\mathrm{Re}\,cent} / B_{MSY}$, etc.) by the delta method using exact derivatives. If the objective function is the log of a proper posterior distribution, then Markov Chain Monte Carlo (MCMC) techniques implemented in AD Model Builder libraries can be used estimate posterior distributions representing uncertainty in the same parameters and quantities.[21]

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[21] MCMC calculations are not available in the current version because objective function calculations use concentrated likelihood formulas. However, the C++ version of KLAMZ is programmed in other respects to accommodate Bayesian estimation.

Bootstrapping

A FORTRAN program called BootADM can be used to bootstrap survey and survival index data in the KLAMZ model. Based on output files from a "basecase" model run, BootADM extracts standardized residuals:

$$r_{v,j} = \frac{\ln\left(I_{v,j} \Big/ \hat{I}_{v,j} \right)}{\sigma_{v,j}}$$

along with log scale standard deviations *(* $\sigma_{v,j}$, originally from survey CV's or estimated from goodness of fit), and predicted values $\left(\hat{I}_{v,j}\right)$ for all active abundance and survival observations. The original standardized residuals are pooled and then resampled (with replacement) to form new sets of bootstrapped survey "data":

$$^{x}I_{v,j} = \hat{I}_{v,j} e^{r\sigma_{v.j}}$$

where *r* is a resampled residual. Residuals for abundance and survival data are combined in bootstrap calculations. BootADM builds new KLAMZ data files and runs the KLAMZ model repetitively, collecting the bootstrapped parameter and other estimates at each iteration and writing them to a comma separated text file that can be processed in Excel to calculate bootstrap variances, confidence intervals, bias estimates, etc. for all parameters and quantities of interest (Efron 1982).

Projections

Stochastic projections can be carried out using another FORTRAN program called SPROJDDF based on bootstrap output from BootADM. Basically, bootstrap estimates of biomass, recruitment, spawning biomass, natural and fishing mortality during the terminal years are used with recruit model parameters from each bootstrap run to start and carryout projections.[22] Given a user-specified level of catch or fishing mortality, the delay-difference equation is used to project stock status for a user-specified number of years. Recruitment during each projected year is based on simulated spawning biomass, log normal random numbers, and spawner-recruit parameters (including the residual variance) estimated in the bootstrap run. This approach is similar to carrying out projections based on parameters and state variables sampled from a posterior distribution for the basecase model fit. It differs from most current approaches because the spawner-recruit parameters vary from projection to projection.

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[22] At present, only Beverton-Holt recruitment calculations are available in SPROJDDF.

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Appendix A5: KLAMZ model results

KLAMZ modeling

The KLAMZ model for the entire surfclam stock during was the main modeling approach and primary basis for providing management advice in the last assessment (NEFSC 2010). KLAMZ model results are provided here to build a bridge between the previous assessment and the current one. KLAMZ results are also provided for the Northern and Southern areas.

The KLAMZ assessment model is based on the Deriso-Schnute delay-difference equation (Deriso 1980; Schnute 1985; see complete technical documentation in Appendix A4). The delay-difference equation is a relatively simple and implicitly age structured approach. It gives the same results as explicitly age-structured models (e.g. Leslie matrix model) if fishery selectivity is "knife-edged", if somatic growth follows the von Bertalanffy equation, and if natural mortality is the same for all age groups in each year. Natural and fishing mortality rates, growth parameters and recruitment may change from year to year.

There are two age or size groups in KLAMZ, "new" and "old" recruits that, together, comprise the whole stock. New recruits are individuals that recruited at the beginning of the current year. Old recruits are all older individuals in the stock that recruited at the beginning of previous years.

KLAMZ delay-difference models in this assessment were for surfclam biomass dynamics during 1981-2011 and were generally similar to models used in the last surfclam assessment (NEFSC 2010). The first year with survey data was 1982, however, the model has an estimable parameter for biomass in 1981 that defines the initial age structure. Landings data are available for earlier years. A number of changes, primarily to input data, for this assessment are described below under "Building a bridge". As in the last assessment, the natural mortality rate is M=0.15 y^{-1} (Appendix A4).

Growth patterns were assumed to vary over time in all models because of recent slow growth in the DMV and NJ regions and because of changes in the distribution of the stock among regions which have different SLMWT and von Bertalanffy growth patterns. In the KLAMZ model, the growth parameter $J_t=w_{t-1k-1}/w_{t,k}$ (where $w_{t,k}$ is the mean body weight of a surfclam at the age of recruitment k in year t) may vary from year to year. The growth parameter J_t represents the combined effects of the traditional von Bertalanffy growth parameters W_∞ and t_0. This approach was adequate for surfclams because much of the variation in growth appeared to be in maximum size W_∞ (Table A16 Assessment report).

Model configuration

NEFSC clam survey data in the KLAMZ model were for new and old recruits. Surveys were assumed to occur in the middle of the year because the NEFSC clam survey is carried out during late May-early July. As in the previous assessment, survey data used in the KLAMZ model were trends, after holes (unsampled survey strata in some years) were filled to the extent possible by borrowing data from the previous and successive surveys. Some years were not used in whole stock or Northern area modeling because GBK was undersampled (Figure 1). For example, GBK was not sampled at all in 2005.

Survey trend data (stratified mean kg/tow) for surfclams 120-129 mm SL were assumed to track trends in biomass of new recruits. Survey data for surfclams 130+ mm were assumed to track trends in the entire stock (old recruits).

Following NEFSC (2009), swept area biomass estimates were included in the assessment model to measure scale, but not trends, in biomass. Swept area biomass estimates were not efficiency corrected in this case because the prior on survey efficiency (see TOR 2) was intended to carry forward model uncertainty in scale. Goodness of fit to the swept area biomass data was given nil weight in the overall objective function. However, the likelihood of the estimated scaling parameter for swept area biomass was calculated based on a log normal prior distribution with mean 0.234 and arithmetic CV = 1.32 and the likelihood was added to the objective function used in fitting the model. The CV was estimated by bootstrapping all available data on survey dredge efficiency (see TOR 2). The CV is relatively broad and the prior information had a little effect in determining the overall scale of surfclam biomass and fishing mortality estimates. Experience has shown that surfclam stock assessment data, aside from the swept are area biomass estimates, are uninformative about the overall scale of biomass but do provide information about trends. Thus, the model tended to be uncertain regarding overall scale, for which there was limited data beyond the somewhat uninformative (high CV) prior distribution on survey dredge efficiency.

Following NEFSC (2003) surfclam recruits were estimated in the KLAMZ model as a random walk with steps constrained by a variance parameter. A smooth, random walk process is probably not ideal from a biological perspective because of the evidence in survey age composition data for strong year classes, but the approach was necessary because of the lack of annual recruitment data. The random walk approach keeps the recruitment estimate in year *t* at the same level as in year *t-1*, unless there is a good reason, in terms of goodness of fit, to change it. For surfclams in the KLAMZ model, the random walk approach helped avoid excessive variation in recruitment, enhanced model convergence, and ensured that some recruitment was estimated for each year.

In modeling surfclam population dynamics with random walk recruitment, it is important to tune the “random walk recruitment variance” σ_R^2 which measures variability in the size of successive steps taken during the random walk (i.e. variance in [$\ln(R_1/R_2)$, $\ln(R_2/R_3)$, $\ln(R_3/R_4)$, etc.], where R_t is the recruitment estimate for year *t*). As σ_R^2 approaches zero, recruitment estimates become smooth and tend towards a constant value with no changes from year to year. As σ_R^2 becomes large, estimated recruitments will change randomly and more widely from one year to next.

Following NEFSC (2007), initial KLAMZ model runs assumed high CV for steps in the random walk. The assumed CV was gradually decreased in subsequent runs until the model was just able to fit the survey data without pattern in residuals and the model was able to fully converge (the Hessian matrix was invertible). In addition, the CV for fit to the survey data (residual CV) was compared to CV for the actual survey data to determine if the model was fitting the survey data more closely than should be expected based on the precision of the survey data (implying that σ_R^2 was too large). Finally, it was determined that the fit to the "old" recruit time series should be better than the fit to the new recruit time series as the older recruits were based on a broader set of size classes and thus more data. The goal was basically to find the model that would adequately explain the survey data for surfclams, but not over fit the new recruit time series.

Recruitment estimates for surfclam from the KLAMZ model are complicated to interpret because of the constraints on variability and limited survey data. Under these conditions, recruitment estimates for surfclam from the KLAMZ model should probably be regarded as "nuisance" parameters of less interest than biomass and fishing mortality estimates. Recruitment estimates for surfclams at best reflect long term average trends. However, recruitment estimates in the KLAMZ model are aliased with model misspecification, survey noise, survey year effects, natural mortality and variability in growth.

Results-whole stock

The KLAMZ model fit survey biomass trend data reasonably well (Figure 2). The model fit the whole stock survey data index better than the index for new recruits, as expected based on the CV for the two sets of survey data (CV for the recruit index are higher).

The survey scaling parameter for efficiency corrected swept area biomass was Q=0.16, which is close to the mode of the prior distribution of survey dredge efficiency. This indicates that the trend data, landings and model estimates did not provide sufficient information on scale to shift the model away from the relatively uninformative prior information about Q for swept area biomass estimates.

Model results (Figure 3 - 4) suggest that surplus production was high before the late 1990's and steadily declined afterwards to negative levels during 2001-2011 as somatic growth and recruitment rates declined. Biomass increased until the late 1990s when surplus production was less than catch.

Bootstrap and delta method CV for biomass, and recruitment estimates were < 25% indicating that estimates were reasonably precise (Table 1). The bootstrap CV for fishing mortality were high because the denominator, the estimated fishing mortality values, were often close to zero. Delta method CV are probably the better measure of uncertainty in this case.

Internal retrospective analysis

Retrospective analyses were carried out with the base case KLAMZ model for terminal years 2000-2011 (Figure 5). There was little evidence of a retrospective problem in either biomass or fishing mortality estimates. The model tends to fluctuate somewhat in scale because the scale of the model is uncertain, but the trend is consistent through time. Changes in scale tended to occur when data from an additional NEFSC clam survey (as in the case of 2002, 2008 and 2011) was dropped.

Historical retrospective analysis

Biomass and fishing mortality estimates from surfclam stock assessments carried out since 1998 were compared to determine the stability of stock estimates used to provide management advice (Figure 6). The scale of the model fit is considerably higher than in past assessments. This is primarily due to changes in the way survey efficiency was estimated and the increased variance in the prior distribution for survey Q. The most important aspect of the historical retrospective analysis is the substantial differences between base case biomass and fishing mortality estimates and estimates from the previous assessment. The factors responsible for these changes are explained below.

Performance of historical projections

The current model differed from historical projections. Comparisons in trend were used because the scale of the model in the last assessment was much lower (Figure 6). In the last assessment the projected biomass in 2011 was approximately 6% lower than biomass in 2008. Using the current whole stock KLAMZ model, biomass in 2011 was approximately 14% lower than biomass in 2008 (Table 2). The discrepancy can be explained by differences in estimated trend between the models, caused by differences in the fit to the survey data (see below).

Building a bridge

Differences between estimates in the base case model in this assessment and the last assessment due to modifications to data and modeling procedures. These are discussed below, one step at a time (Figure 7). The most important factors contributing to differences between the base case model biomass estimates in this assessment and estimates in the previous assessment are: additional variance in the prior distribution for survey Q (Step 3), and additional variance allowed in the fit to the recruit time series (Step 2, Step 13).

Step 1 was to run the KLAMZ model using updated data from the last assessment to determine if any new bugs had crept into the model code. The model was able to estimate parameters, but produced steep gradients and did not converge. Step 2 was to allow more freedom in the variance of the random walk recruitment parameter, σ_R^2 , which allowed a better fit to the survey data for both old and new recruits. This step reduced the magnitude of the gradients, but still did not produce an invertible hessian matrix. Step 3 was to incorporate the new prior distribution for survey Q, which increased the variance in the prior by an order of magnitude from the last assessment. Step 4 was to include the new selectivity estimates for the survey dredge. The fifth step was to incorporate new SLMWT relationships. Step 6 was to add the updated growth estimates. The model converged for the first time after this step. The seventh step was to decouple the surveys (in previous estimates there was overlap in size classes between the old and new recruits). The eighth step was to include discards in the fishery data being used (a correction to an oversight). The ninth step was to remove data from 1983 from the whole stock model due to poor coverage on GBK. Step 10 was to incorporate changes in sensor data criteria used to identify and discard “bad” survey tows for use in estimating efficiency corrected swept area biomass. The eleventh step was to fix a bug in the routine to borrow data from adjacent years to fill holes in the survey time series. Step 12 was to fix a bug in the growth estimates added in step 6. Finally step 13 was to adjust the σ_R^2 parameter to minimize the overall Likelihood function. Convergence was generally tenuous throughout this process. The model was sensitive to starting conditions and generally produced large gradients even when the hessian matrix was invertible.

Results-Southern Area

The KLAMZ model for the southern area (SVA to SNE) incorporated all of the data available. All survey years were included for new (120 – 129 mm SL) and old (130+ mm SL) recruits. Swept area biomass for all years in which dredge sensors were deployed (1997 and after; Figure 8) were included as well. Catch data between 1982 and 2011 were used.

Other model parameters were selected according to the methodology established in the whole stock model. Growth parameters and juvenile ratios (see above) were calculated for the appropriate subset of the data for the whole stock (animals from SVA to SNE). The σ_R^2 parameter (see above) was chosen to minimize a concentrated Likelihood function that ignored the recruitment model component. The recruitment model component is always minimized by a σ_R^2 equal to zero because it prefers a recruitment model with fewer parameters (see Appendix A4).

Changing the σ_R^2 parameter had a substantial affect on the overall model (Figure 9). The trend of the model fit was relatively unaffected, but the scale changed by as much as a factor of three depending on the value of σ_R^2 chosen.

The model fit the survey data reasonably well (Figure 10). Trends in the overall fit were similar to the fit for the whole stock, indicating that the population biomass peaked in the late 1990's. The southern area, however, indicates a steeper decline since then (Figure 11).

Surplus production (Figure 12) was positive until the mid 1990's and has been negative since then, until 2011. The upward trend in surplus production over the last six years has been driven by strong recruitment.

The scale parameter for the KLAMZ model, survey *Q*, was 0.55. This value is considerably higher than the survey *Q* estimated for the whole stock (0.16). The discrepancy is a result of uncertainty in our extra-model estimates of survey dredge efficiency (see above) and is reflected in the prior distribution which has a CV of 134%. The KLAMZ model is therefore given very little information about scale and that uncertainty is evident in the trouble KLAMZ has in establishing a consistent scale.

Bootstrap runs (n=500) for the southern area KLAMZ model runs were fairly consistent though there were a few extreme outliers (Figure 13). This is reflected in the bootstrap CV which were generally high (Table 3) and driven by outliers which tended to be unconverged cases (~3%). Delta method CV were generally below 20%.

Internal Retrospective

Retrospective analysis indicates a shift in scale, but not trend, as survey years are removed from the model (Figure 14). The model tends to fluctuate somewhat in scale because the scale of the model is uncertain, but the trend is consistent through time. Changes in scale tended to occur when data from an additional NEFSC clam survey (as in the case of 2002, 2008 and 2011) were dropped.

Results-Northern Area

The KLAMZ model for the northern area (GBK) incorporated a subset of the data available. There were some years where coverage on GBK was poor (1982, 1983) and other years where GBK was not sampled (2005). Swept area biomass for all years in which dredge sensors were deployed and GBK was sampled (1997 and after, excluding 2005; Figure 15) were included as well. Catch data was sparse, as GBK was not fished for 20 years between 1989 and 2008.

Other model parameters were selected according to the methodology established in the whole stock model. Growth parameters and juvenile ratios were calculated for the appropriate subset of the data for the whole stock (animals from GBK). The σ_R^2 parameter (see above) was chosen to minimize a concentrated likelihood function, that ignored the recruitment model component. The recruitment model component is minimized by a σ_R^2 equal to zero, because it prefers a recruitment model with fewer parameters (see Appendix A4). This choice could not be made naively however, as it is possible to overfit the recruitment index at the expense of other data. In this case the minimum of the concentrated likelihood occurred at $\ln(\sigma_R^2) = -4$, which would have resulted in the goodness of fit to the recruitment time series being less than the goodness of fit implied by the CV of the index itself. The σ_R^2 parameter was gradually increased until the goodness of fit to the index was greater than the goodness of fit implied by the survey CV ($\ln(\sigma_R^2) = -4.65$; Figure 16). Changing the σ_R^2 parameter had little effect on the overall model (Figure 17).

The model fit the survey data reasonably well (Figure 16). Based on the fit to the survey data, the northern area has been growing since the cessation of fishing there in 1989. The upward trend in growth seems to be tapering off and has been essentially flat for approximately the last 5 years (Figure 18).

Surplus production (Figure 19) was positive from the late 1980's until 2010. The decline in surplus production is probably due to declining recruitment since 1995 (Figure 19).

The scale parameter for the KLAMZ model, survey *Q*, which is analogous to survey dredge efficiency in efficiency corrected swept are biomass calculations was 0.14. This value was comparable to the survey *Q* estimated for the whole stock (0.16). The estimated Q was close to the mean of the prior distribution and indicated that the data provided to the KLAMZ model for the Northern area probably provided very little information about scale. The prior distribution we used was highly uninformative and (CV = 134% see TOR 2 above) and was not likely to influence the estimate of survey Q very much in the presence of data that informed scale. The fact the estimated survey Q did not differ from mean of the prior probably means that the data were not informative regarding scale.

Bootstrap runs (n=500) for the Northern area KLAMZ model runs were fairly consistent (Figure 20). This is reflected in the bootstrap CV which were generally tight (Table 4). Delta method CV were generally very high (~100%). The discrepancy between delta method CV based on the Hessian matrix and the bootstrap CV is probably due to differences between the two methods. The delta method uncertainty reveals a flat likelihood and thus a wide CV in the area immediately around the converged solution. If however the “flatness” of the likelihood surface is confined to a relatively small parameter space, the bootstrap solutions might all arrive at nearly the same solution and thus produce a relatively narrow CV. Some evidence for this is provided by the high rate of convergence in the bootstrap runs (100% converged) and by the fact that profiles over various values of σ_R^2 (Figure 17) and survey Q (Figure 21) indicate that the solution is fairly stable over these parameters. There is simply not enough information in these data to provide a strongly peaked likelihood surface.

Internal Retrospective

Retrospective analysis indicates a shift in scale, but not trend as survey years are removed from the model (Figure 22). There are no indications of retrospective problems in the Northern area KLAMZ model.

Appendix A5. Table 1. Bootstrap and delta method CV for whole stock KLAMZ runs.

	Biomass		F		Recruitment	
Year	Bootstrap cv	Delta cv	Bootstrap cv	Delta cv	Bootstrap cv	Delta cv
1981	27.58	28.27	50.62	28.40	24.45	46.92
1982	25.43	19.80	51.56	19.88	22.57	41.23
1983	23.79	14.73	53.04	14.81	22.82	27.38
1984	22.60	13.31	54.64	13.39	21.47	28.36
1985	21.74	13.57	56.53	13.64	20.58	26.08
1986	21.01	14.40	58.40	14.48	20.53	27.24
1987	20.57	15.31	59.28	15.38	20.62	25.93
1988	20.23	15.98	59.53	16.06	20.76	21.73
1989	19.91	16.27	59.44	16.34	21.25	23.75
1990	19.78	16.33	58.92	16.41	21.13	23.80
1991	19.71	16.31	57.99	16.38	19.89	22.66
1992	19.42	16.27	56.90	16.34	18.26	21.67
1993	18.80	16.44	57.21	16.50	19.44	19.49
1994	18.54	16.36	57.44	16.41	17.34	22.45
1995	18.05	16.05	57.04	16.09	17.15	22.85
1996	17.58	15.92	56.69	15.96	19.28	20.31
1997	17.30	15.99	56.86	16.02	19.02	23.32
1998	17.15	16.09	56.15	16.12	19.53	22.66
1999	17.07	16.20	55.91	16.24	19.90	25.74
2000	17.07	16.30	55.70	16.34	19.89	26.17
2001	17.09	16.41	55.72	16.46	19.21	24.45
2002	17.12	16.54	56.11	16.60	19.84	27.88
2003	17.20	16.64	57.09	16.70	20.79	29.18
2004	17.33	16.76	58.46	16.83	21.33	29.29
2005	17.49	16.91	59.91	16.97	21.21	28.56
2006	17.63	17.05	61.53	17.13	20.67	26.88
2007	17.75	17.22	63.41	17.30	20.78	23.39
2008	17.79	17.34	64.94	17.42	20.33	28.27
2009	17.82	17.52	66.30	17.59	21.00	28.79
2010	17.84	17.82	67.19	17.89	22.59	25.45
2011	17.88	18.12	67.41	18.19	NA	NA
mean	19.23	16.72	58.32	16.78	20.45	26.40

Appendix A5. Table 2. Mean, median and quantiles of relative biomass change from 2008 to 2011, comparing projections from the last assessment to the current KLAMZ model results.

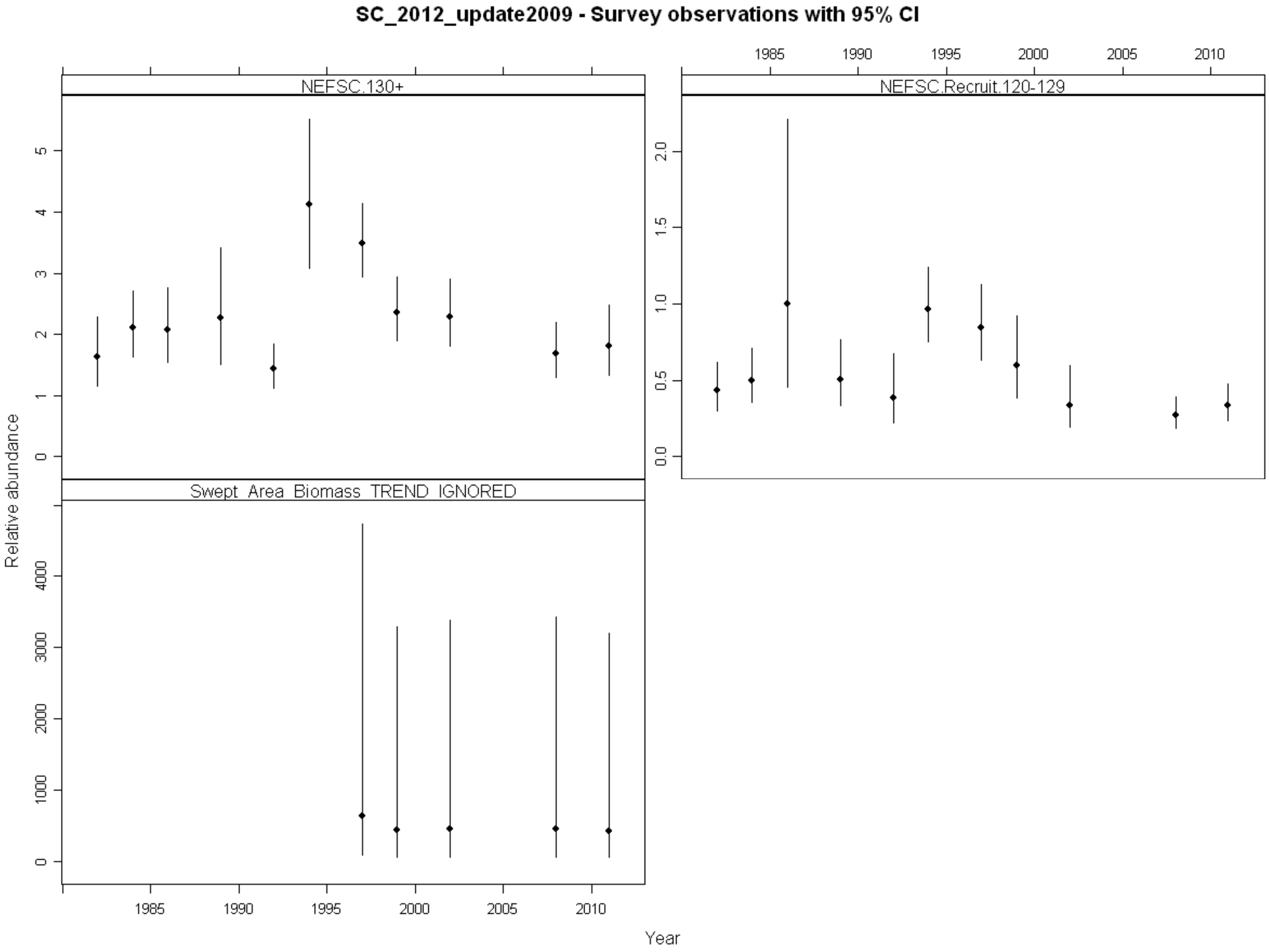
change from 2008 to 2011		
Statistic	Proj 2009	This Assessment
Q10%	-7.54%	-14.63%
Mean	-5.72%	-13.55%
Median	-5.63%	-13.50%
Q90%	-3.80%	-12.50%

Appendix A5. Table 3. Bootstrap and delta method CV for southern area KLAMZ runs.

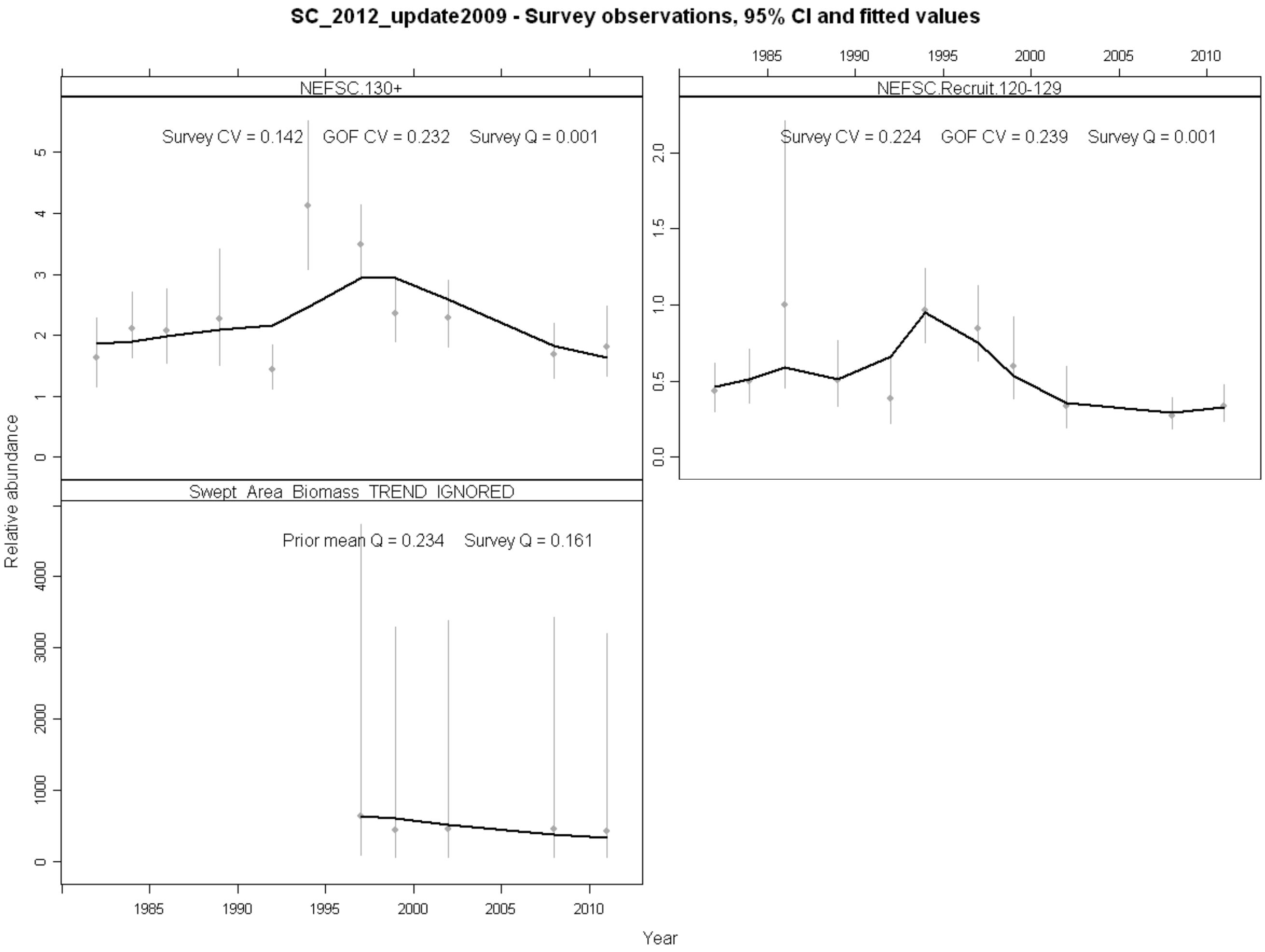
	Biomass		Fishing Mortality		Recruitment	
Year	Bootstrap CV	Delta CV	Bootstrap CV	Delta CV	Bootstrap CV	Delta CV
1981	56.48	5.46	25.60	5.56	59.88	16.53
1982	57.17	6.30	24.28	6.42	55.42	15.85
1983	57.74	7.78	23.75	7.91	54.17	15.11
1984	58.08	9.10	23.61	9.24	53.81	14.71
1985	58.59	10.15	23.68	10.32	53.84	14.26
1986	59.07	11.00	23.87	11.17	57.68	13.82
1987	60.19	11.61	24.04	11.82	60.74	13.37
1988	61.47	12.10	24.16	12.33	62.41	12.86
1989	62.89	12.47	24.19	12.72	56.66	12.61
1990	63.19	12.72	24.10	12.96	51.71	12.26
1991	62.69	12.82	23.90	13.03	47.89	11.84
1992	61.13	12.75	23.63	12.97	43.65	11.31
1993	58.90	12.60	23.42	12.82	45.27	10.88
1994	57.26	12.41	23.30	12.59	41.87	11.00
1995	55.59	12.24	23.12	12.39	40.87	10.97
1996	54.10	12.06	22.91	12.19	42.47	10.90
1997	53.12	11.87	22.70	11.99	47.17	11.21
1998	52.97	11.79	22.53	11.93	51.52	11.27
1999	53.34	11.77	22.57	11.92	54.75	11.36
2000	54.14	11.83	22.67	11.99	56.99	11.38
2001	55.16	11.93	22.82	12.13	58.42	11.32
2002	56.43	12.11	23.08	12.36	55.56	11.37
2003	57.89	12.38	23.44	12.67	52.08	11.36
2004	59.41	12.71	23.87	13.04	48.71	11.06
2005	60.83	13.12	24.26	13.46	49.87	11.70
2006	62.18	13.45	24.75	13.89	51.36	11.98
2007	64.03	13.92	25.43	14.46	53.19	12.00
2008	66.27	14.55	26.14	15.14	51.26	12.98
2009	68.06	15.09	27.00	15.70	50.15	13.63
2010	69.15	15.57	27.88	16.18	50.43	14.33
2011	69.29	15.97	28.85	16.66	NA	NA
mean	59.57	11.99	24.18	12.26	51.99	12.51

Appendix A5. Table 4. Bootstrap and delta method CV for GBK area KLAMZ runs.

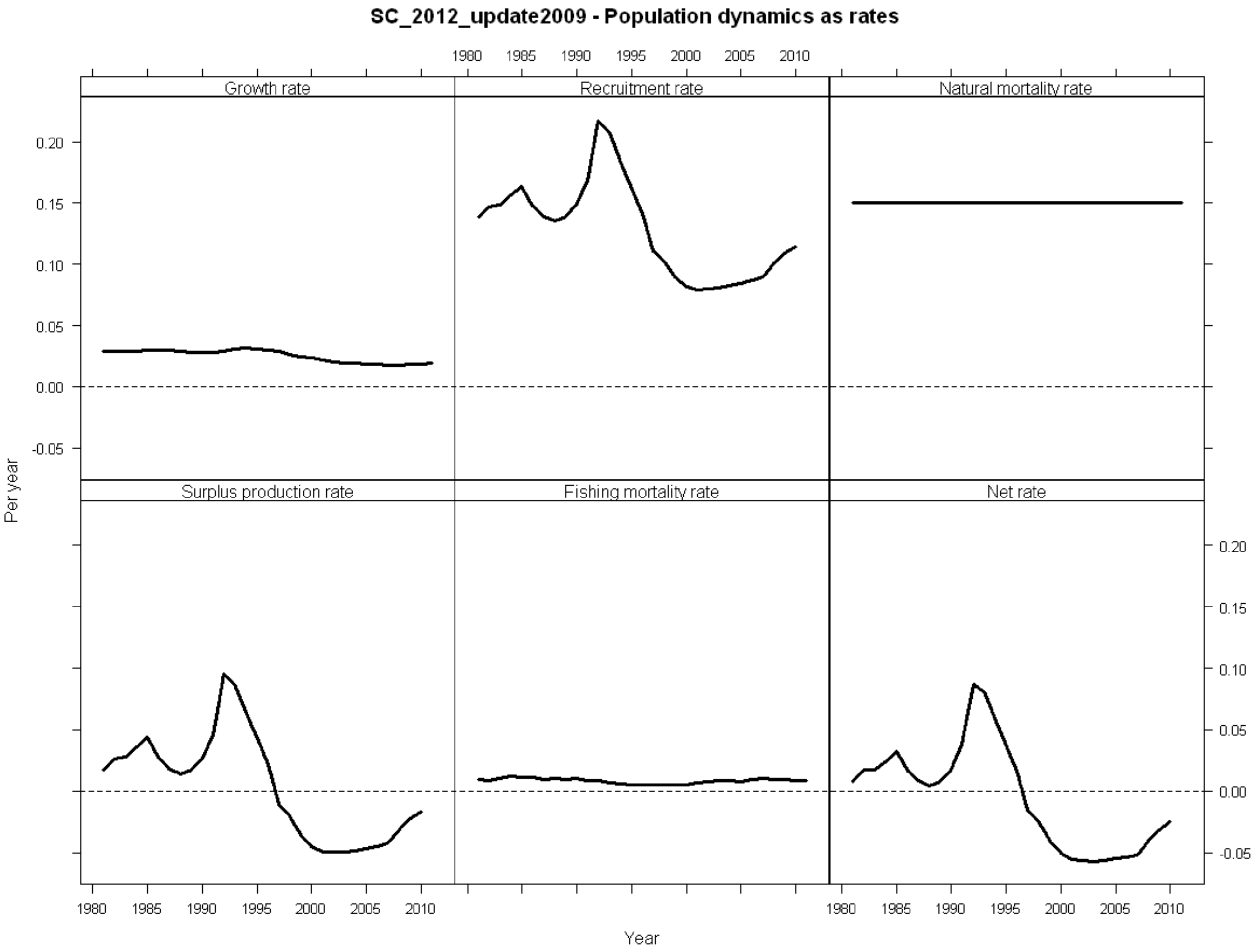
	Biomass		Fishing Mortality		Recruitment	
Year	Bootstrap CV	Delta CV	Bootstrap CV	Delta CV	Bootstrap CV	Delta CV
1981	70.64	99.01	NA	NA	27.70	97.13
1982	65.04	99.13	NA	NA	27.76	97.14
1983	59.55	99.15	NA	NA	27.69	97.43
1984	54.31	99.16	46.48	97.38	25.06	97.97
1985	49.38	99.14	41.49	96.97	23.96	97.70
1986	44.58	99.14	37.18	96.54	24.20	97.53
1987	39.84	99.16	33.47	96.08	24.57	97.44
1988	35.41	99.18	30.24	95.70	24.62	97.44
1989	31.50	99.21	27.50	95.45	24.61	97.55
1990	28.19	99.23	25.27	95.27	24.41	97.81
1991	25.57	99.24	NA	NA	24.70	97.83
1992	23.53	99.22	NA	NA	22.19	98.03
1993	21.99	99.19	NA	NA	21.33	98.45
1994	20.72	99.12	NA	NA	19.37	98.45
1995	19.62	99.01	NA	NA	17.95	98.76
1996	18.40	98.87	NA	NA	18.18	98.43
1997	16.99	98.72	NA	NA	14.43	98.30
1998	15.49	98.55	NA	NA	15.30	98.41
1999	14.03	98.35	NA	NA	14.53	98.02
2000	12.70	98.10	NA	NA	15.37	98.22
2001	11.65	97.76	NA	NA	16.78	97.74
2002	10.93	97.38	NA	NA	18.34	97.42
2003	10.65	97.02	NA	NA	20.15	97.26
2004	10.82	96.63	NA	NA	21.50	97.11
2005	11.36	96.18	NA	NA	22.32	97.25
2006	12.13	95.92	NA	NA	23.11	97.72
2007	12.98	95.69	NA	NA	25.04	97.79
2008	13.84	95.55	NA	NA	25.17	98.13
2009	14.67	94.86	14.67	98.91	26.83	96.86
2010	15.46	94.10	15.45	99.08	30.11	95.66
2011	16.28	93.27	16.23	99.16	NA	NA
mean	26.07	97.88	28.80	97.05	22.24	97.70



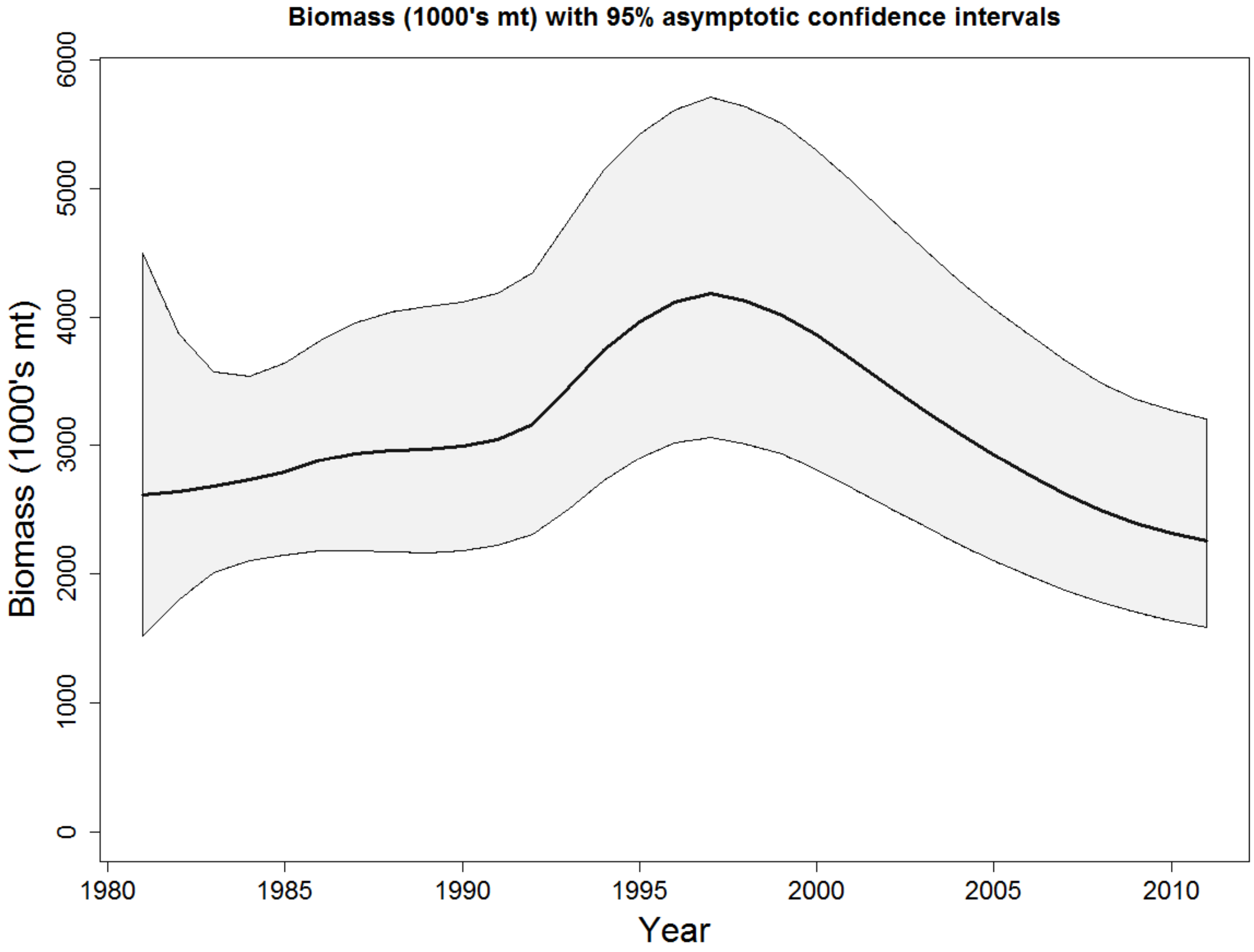
Appendix A5. Figure 1. Whole stock survey data and swept area biomass estimates with approximate 95% confidence intervals.



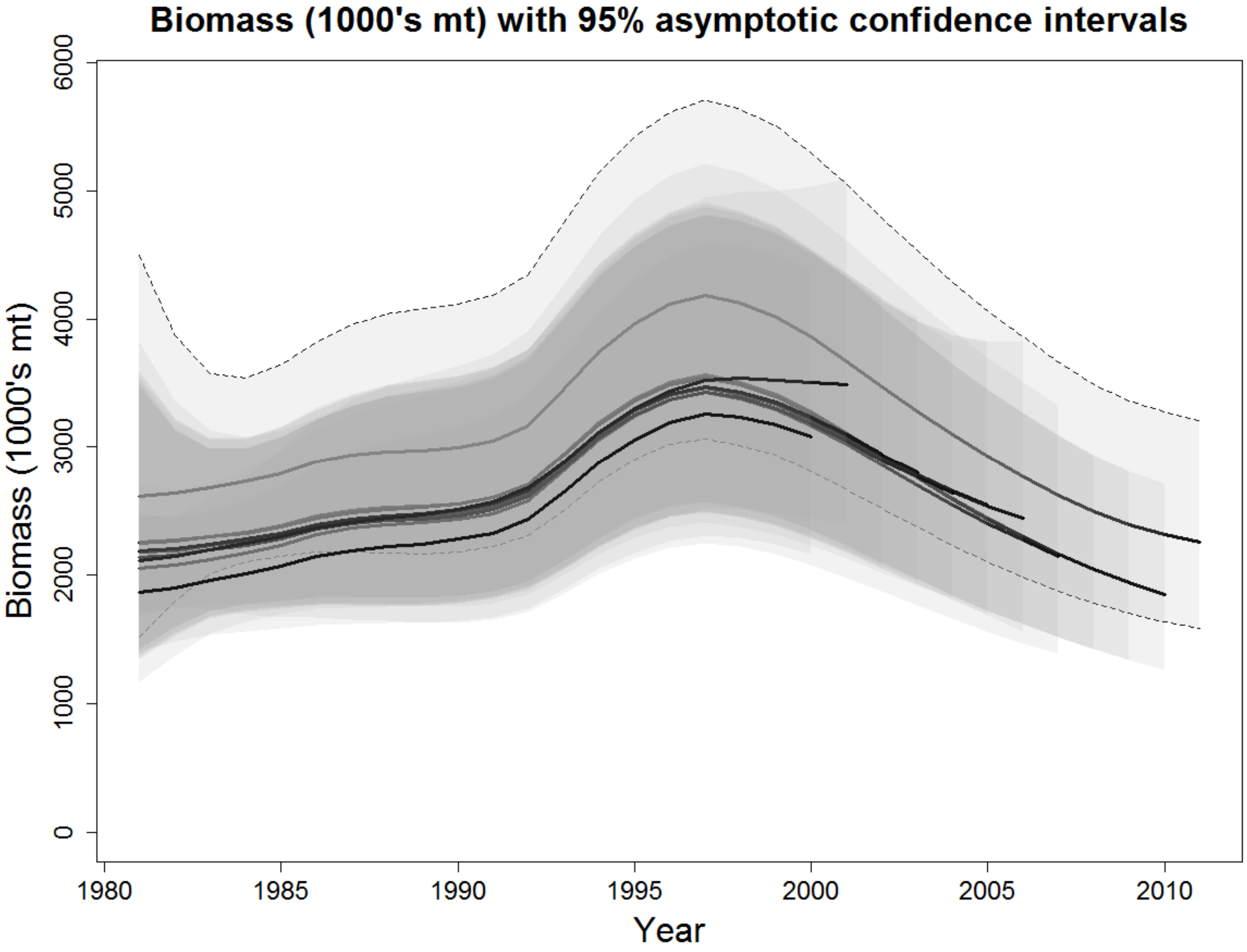
Appendix A5. Figure 2. Whole stock survey data and swept area biomass estimates with approximate 95% confidence intervals and KLAMZ model fits with goodness of fit statistics and estimated catchability parameters.



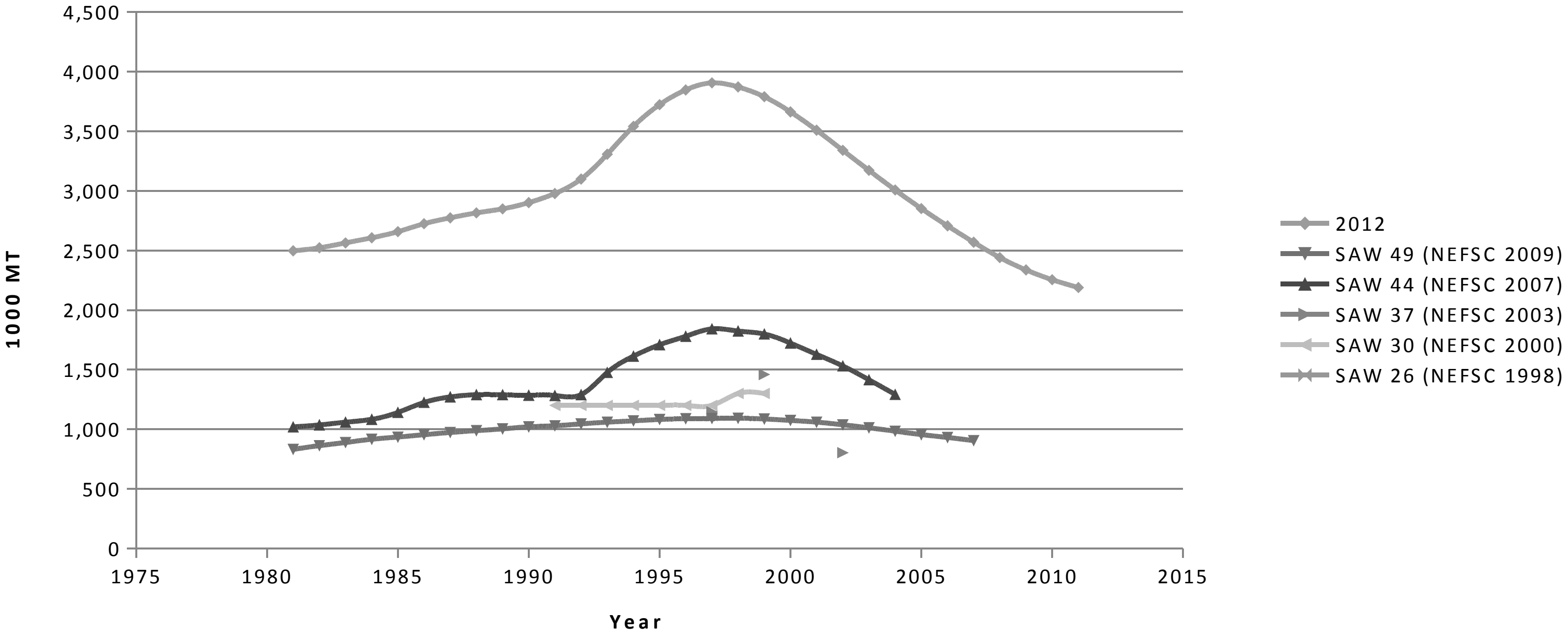
Appendix A5. Figure 3. Some population dynamics, shown as rates, estimated in KLAMZ for the whole stock.



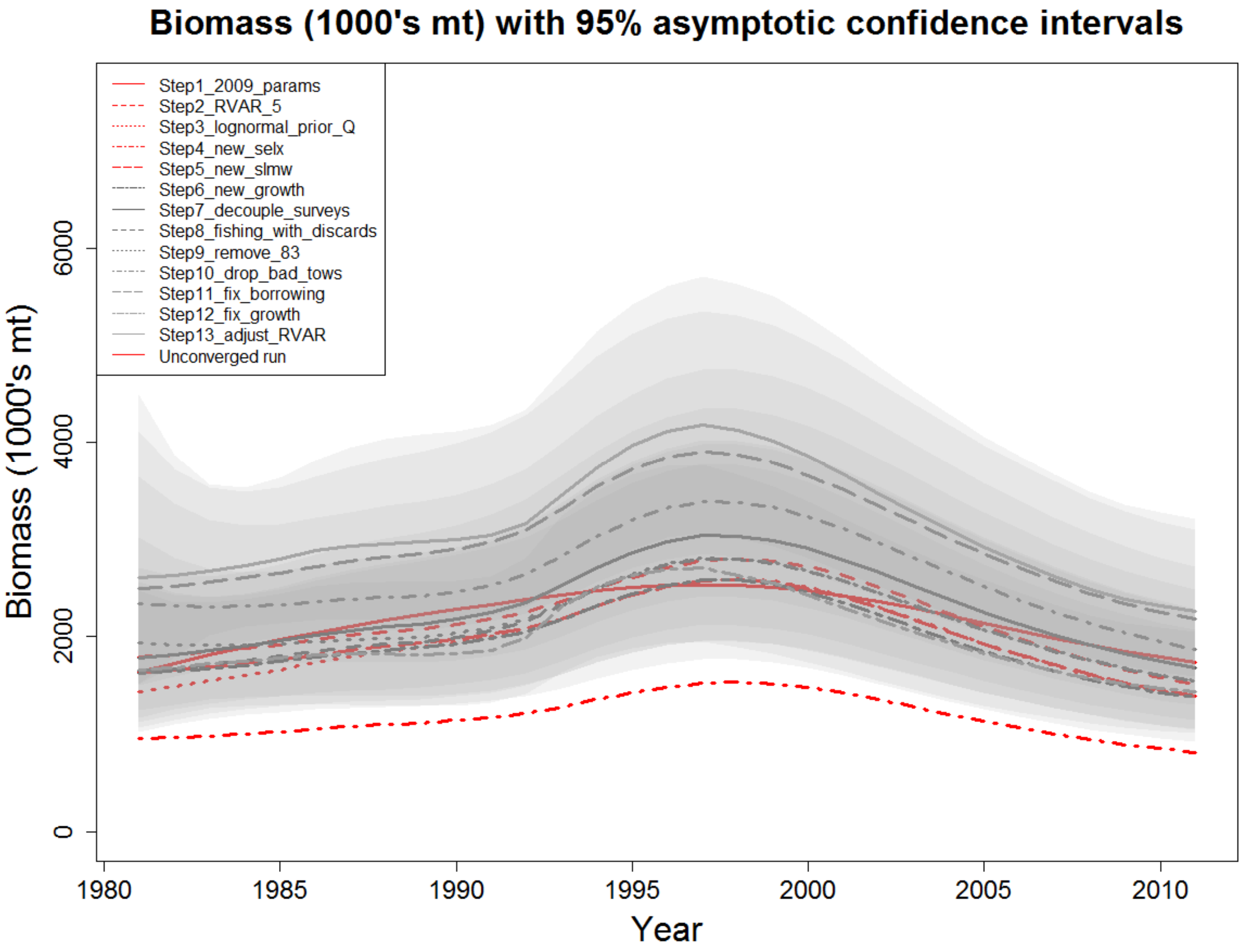
Appendix A5. Figure 4. Total biomass (1000 mt) estimated for the whole stock.



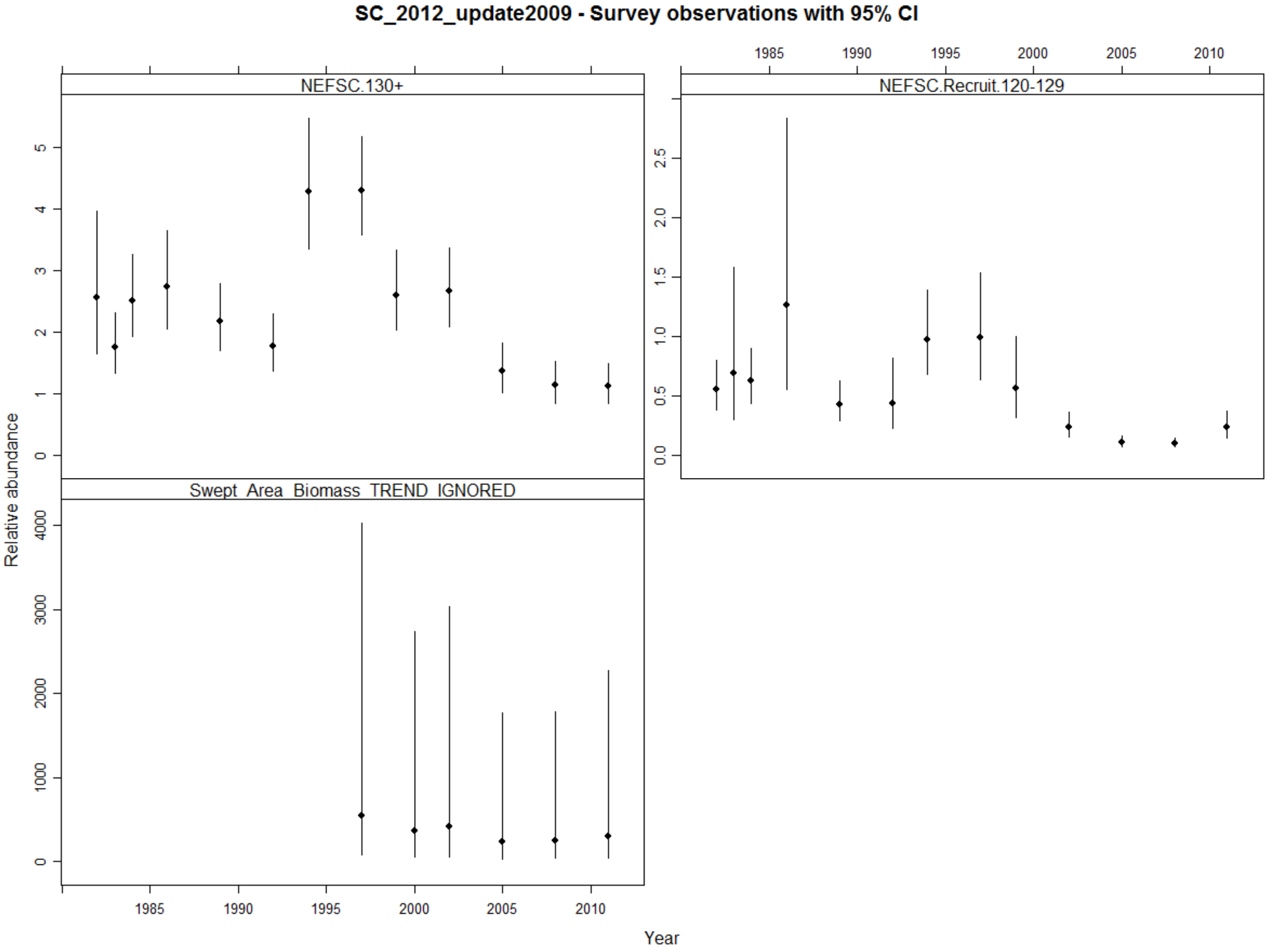
Appendix A5. Figure 5. Retrospective patterns in total biomass for the years 2000-2011 using the base case whole stock KLAMZ model.



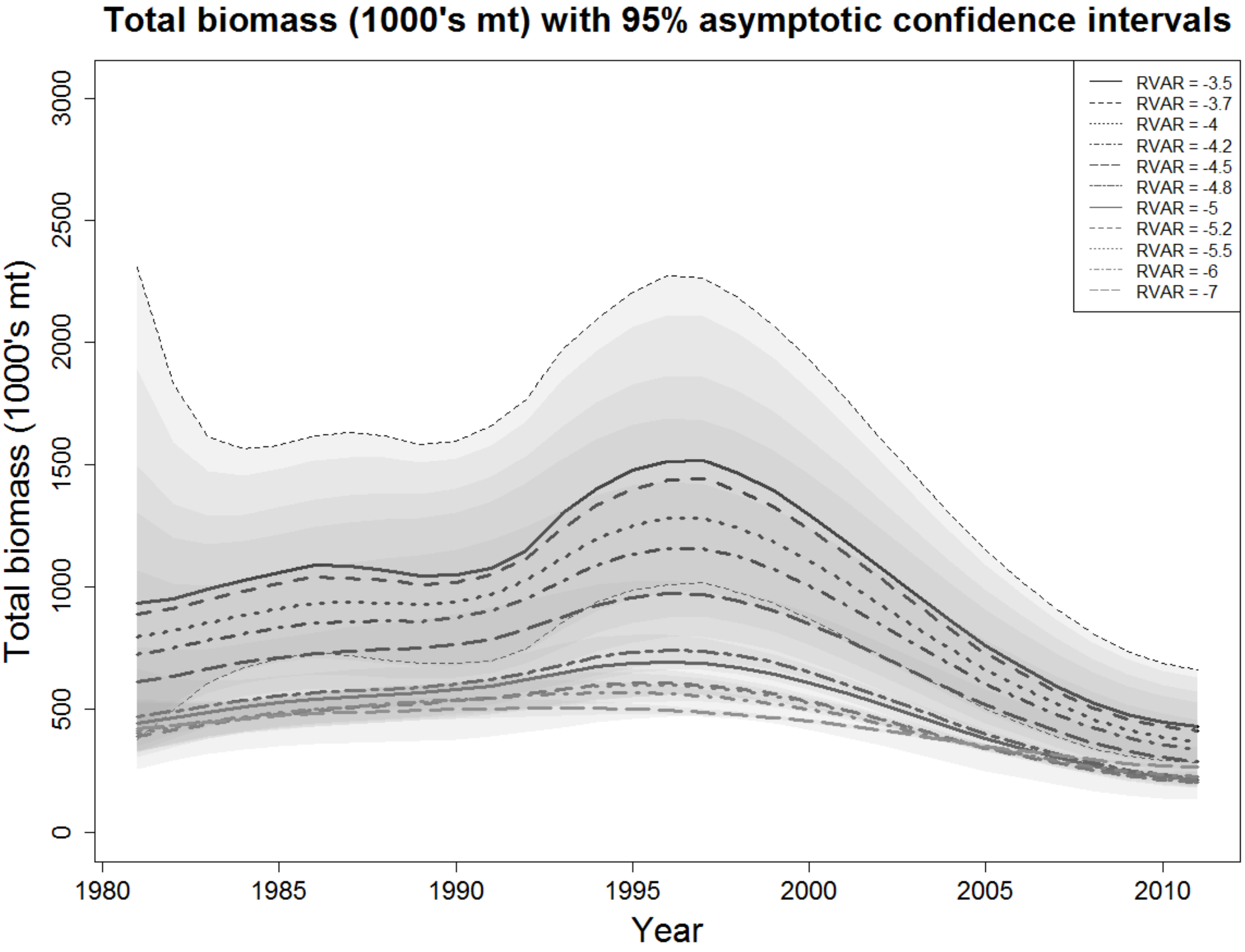
Appendix A5. Figure 6. Historical retrospective pattern in basecase whole stock KLAMZ models.



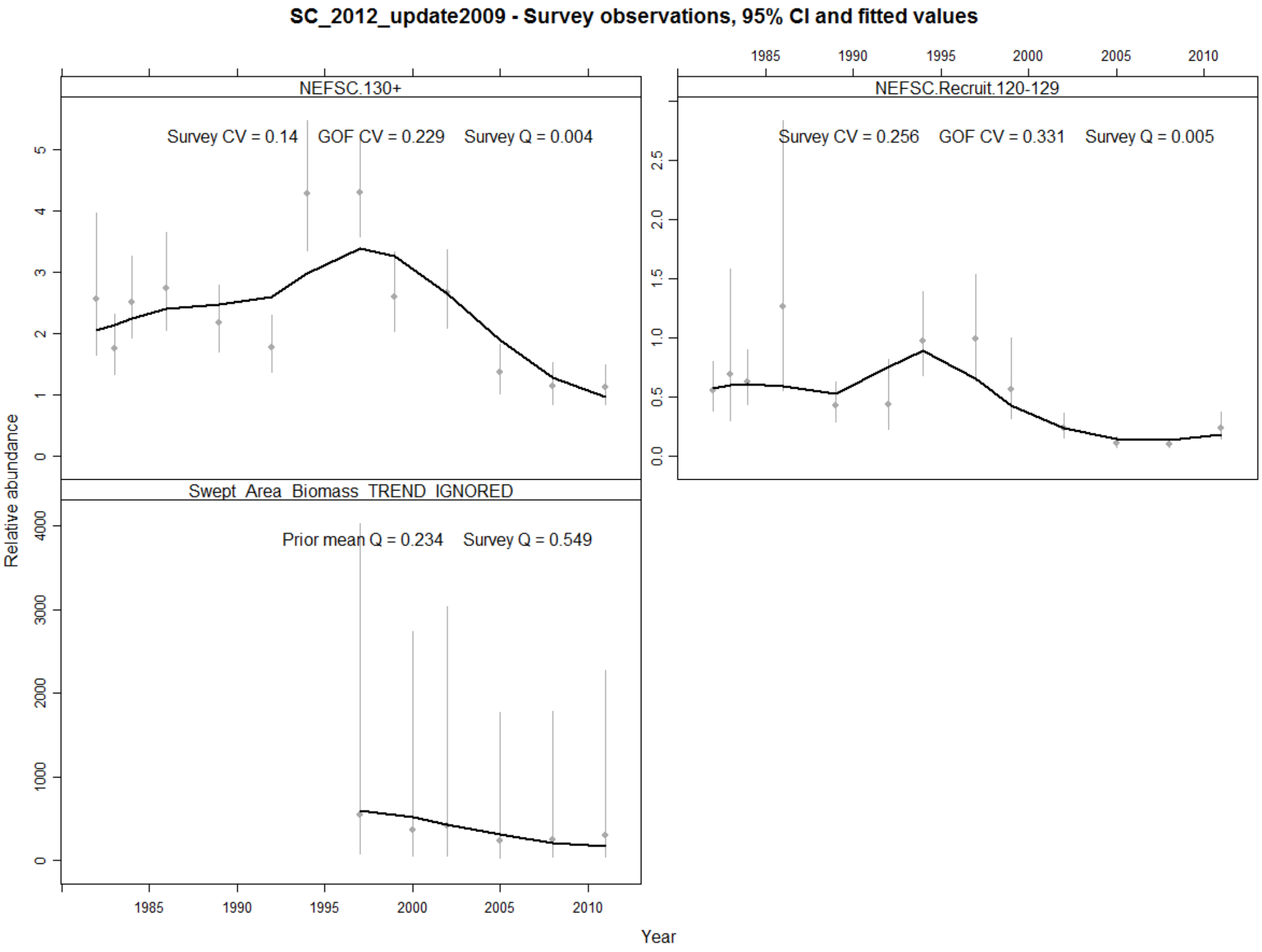
Appendix A5. Figure 7. Build a bridge. The steps involved in updating the KLAMZ model from the 2009 assessment to the current base case whole stock KLAMZ version. Not all runs converged (red lines) and so asymptotic confidence intervals based on the delta method were not always available.



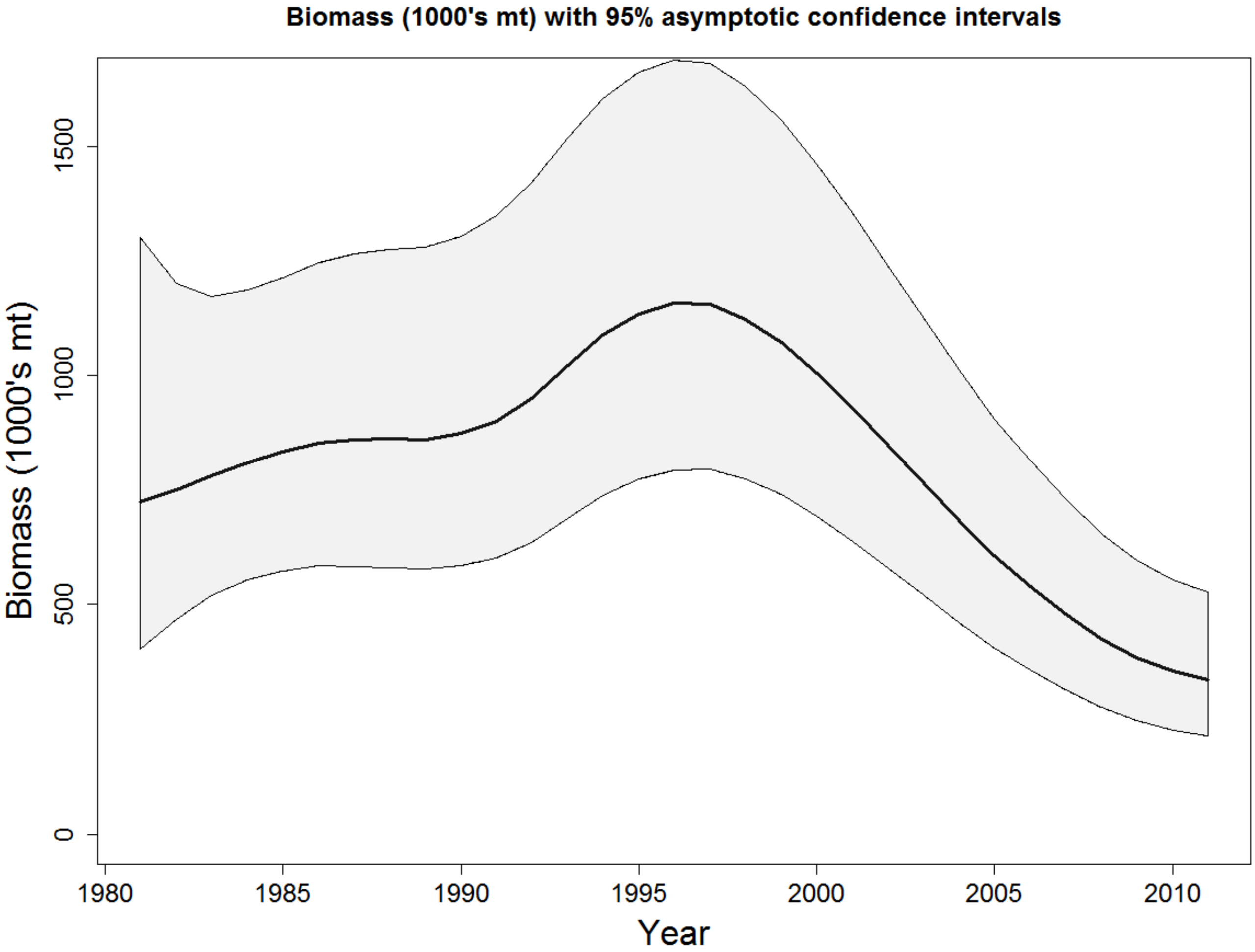
Appendix A5. Figure 8. The data with approximate 95% confidence intervals used to model the southern area (SVA to SNE) with KLAMZ.



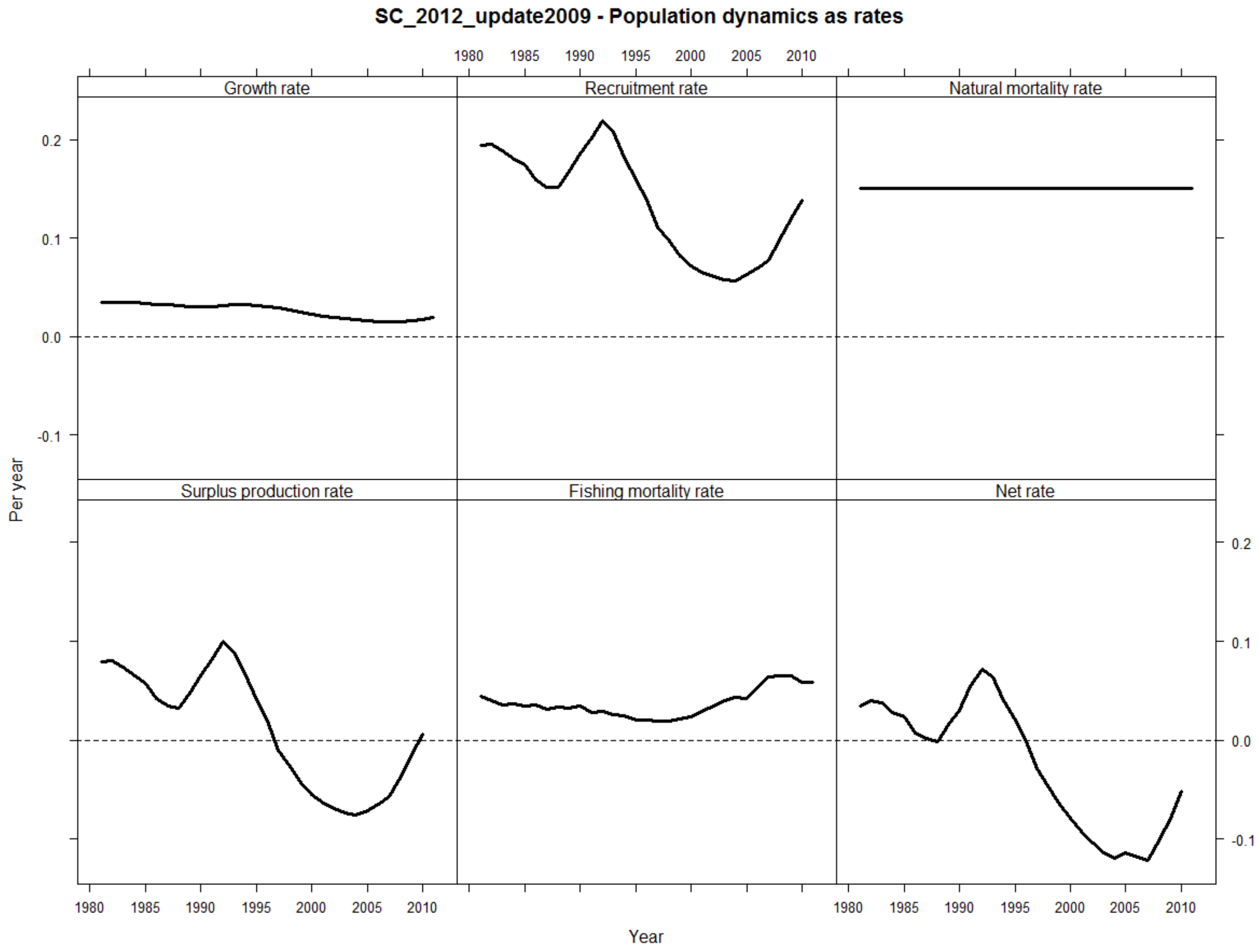
Appendix A5. Figure 9. Sensitivity to σ_R^2 the variance in the random walk recruitment parameter (RVAR).



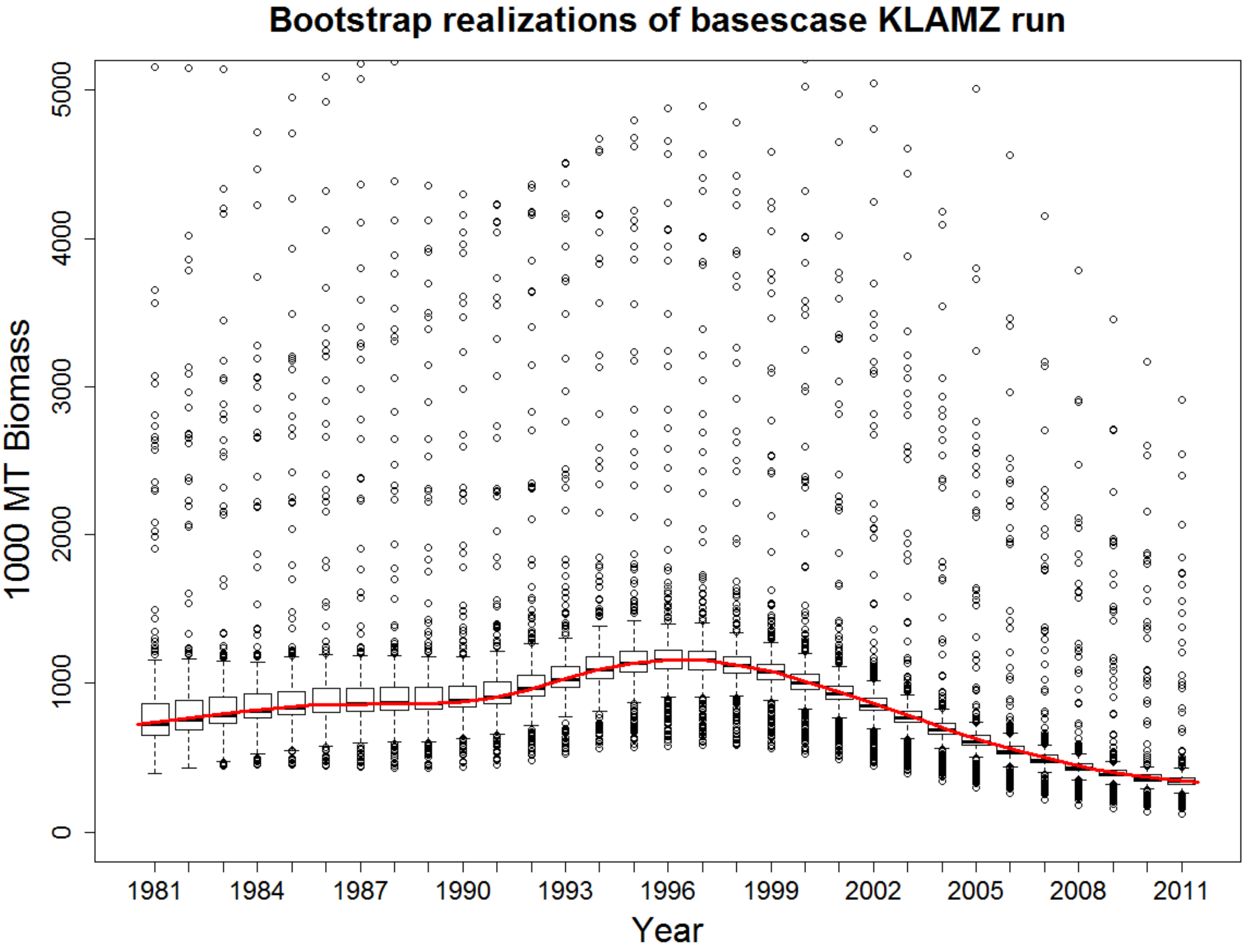
Appendix A5. Figure 10. KLAMZ model fit to the southern area.



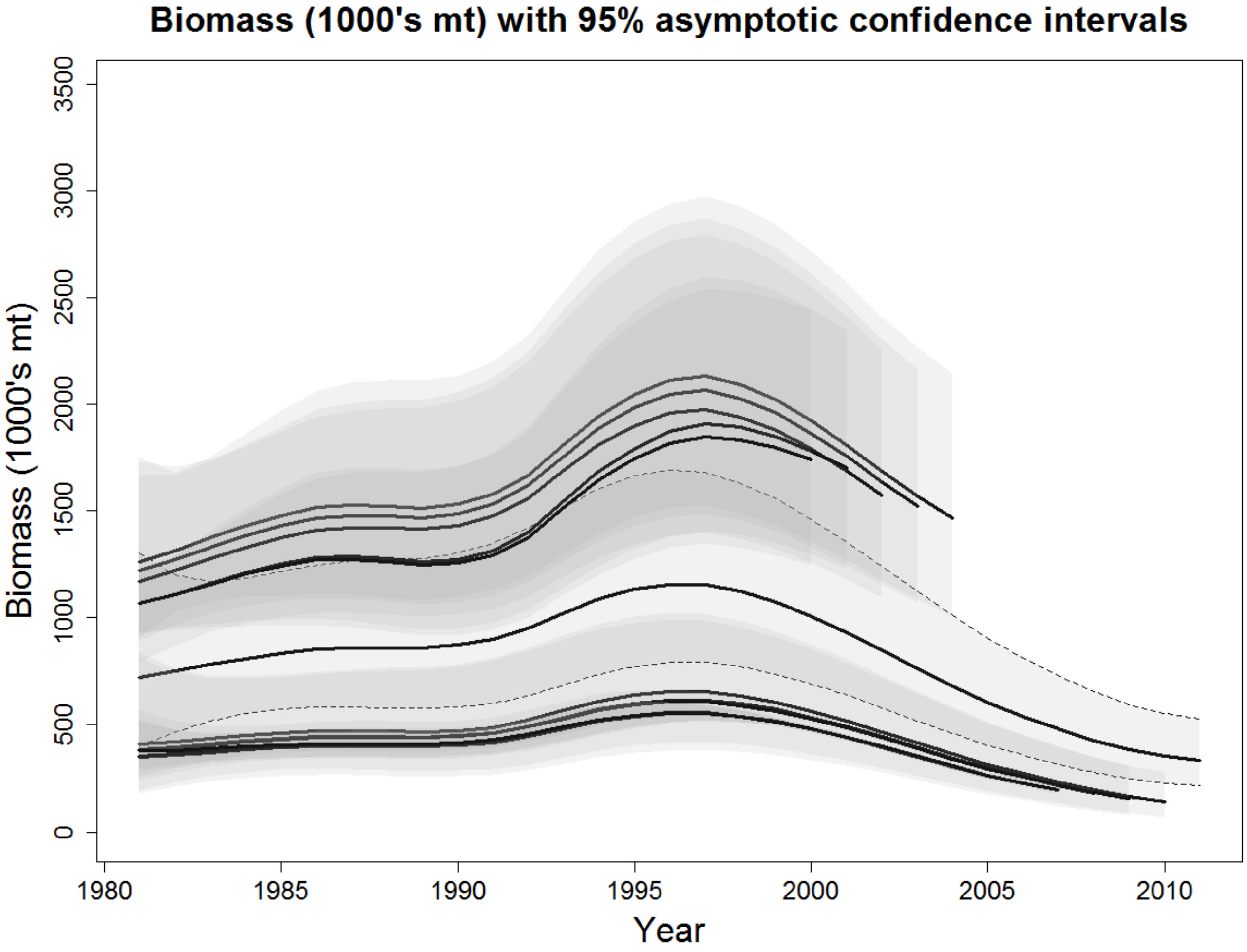
Appendix A5. Figure 11. Biomass (1000 mt) estimated using KLAMZ for the southern area.



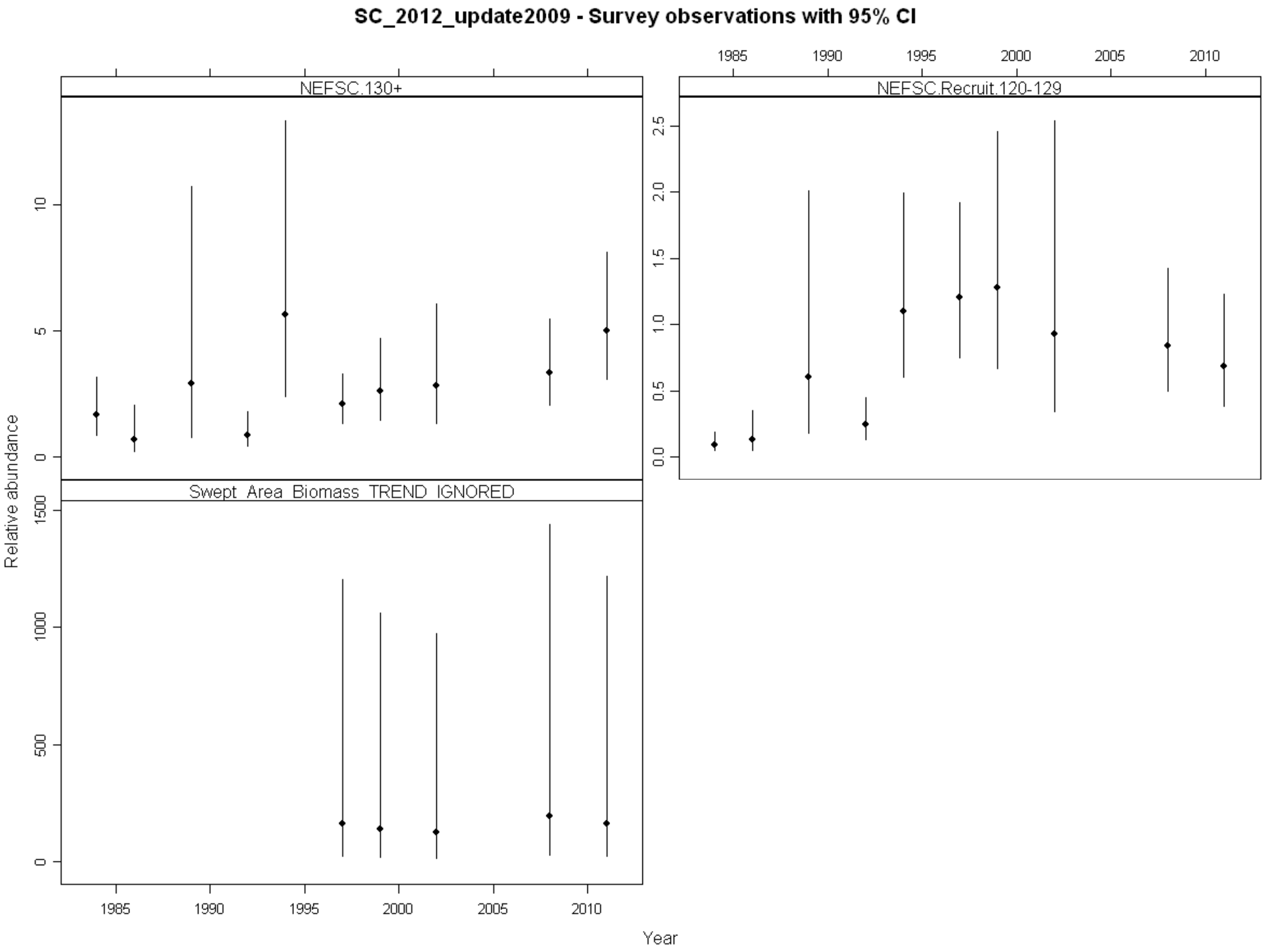
Appendix A5. Figure 12. Population dynamics as rates over time for the southern area.



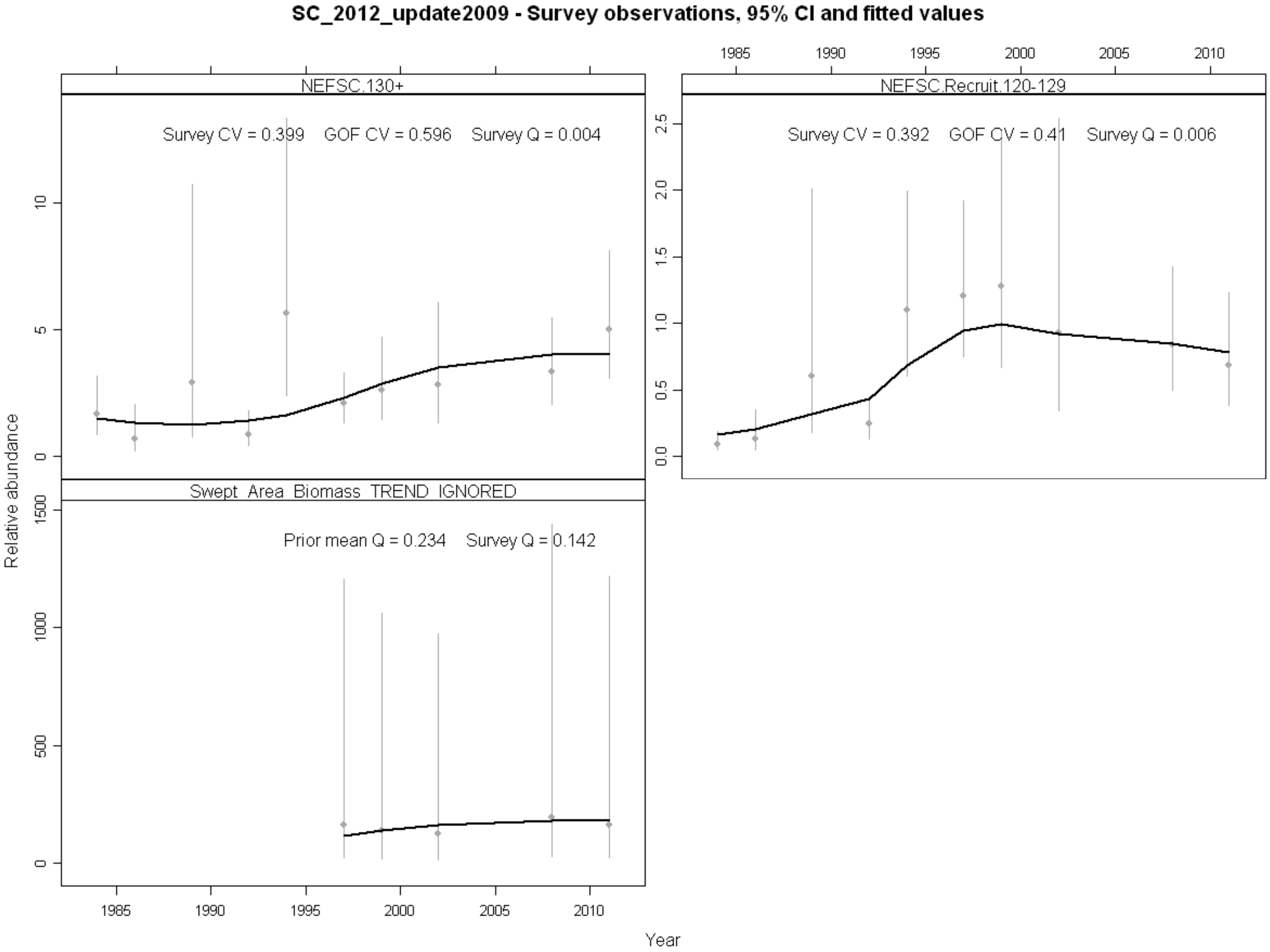
Appendix A5. Figure 13. Bootstrap iterations of the KLAMZ model biomass estimates for the southern area. The base case is shown in red.



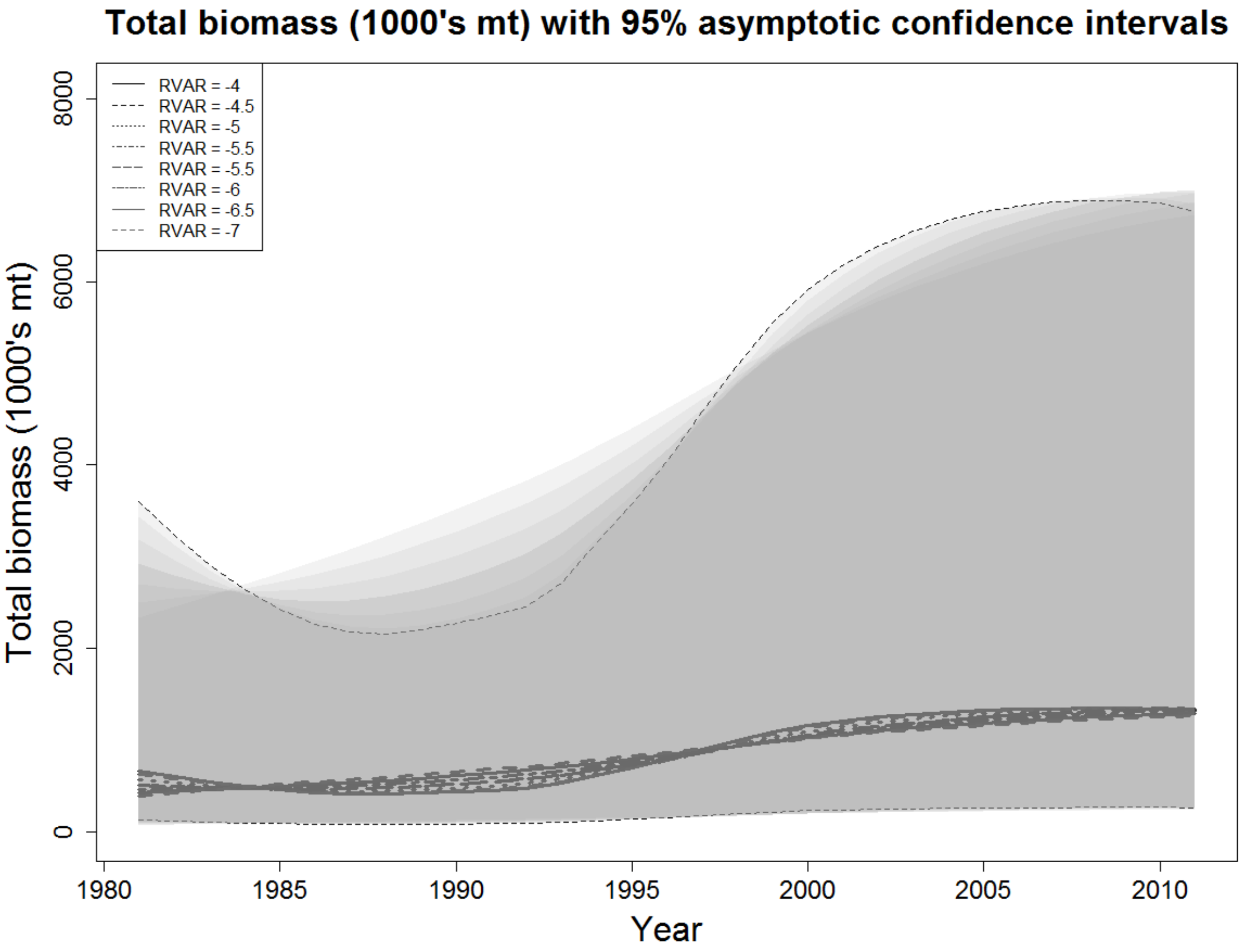
Appendix A5. Figure 14. Retrospective patterns in total biomass for the years 2000-2011 using the base case southern area KLAMZ model.



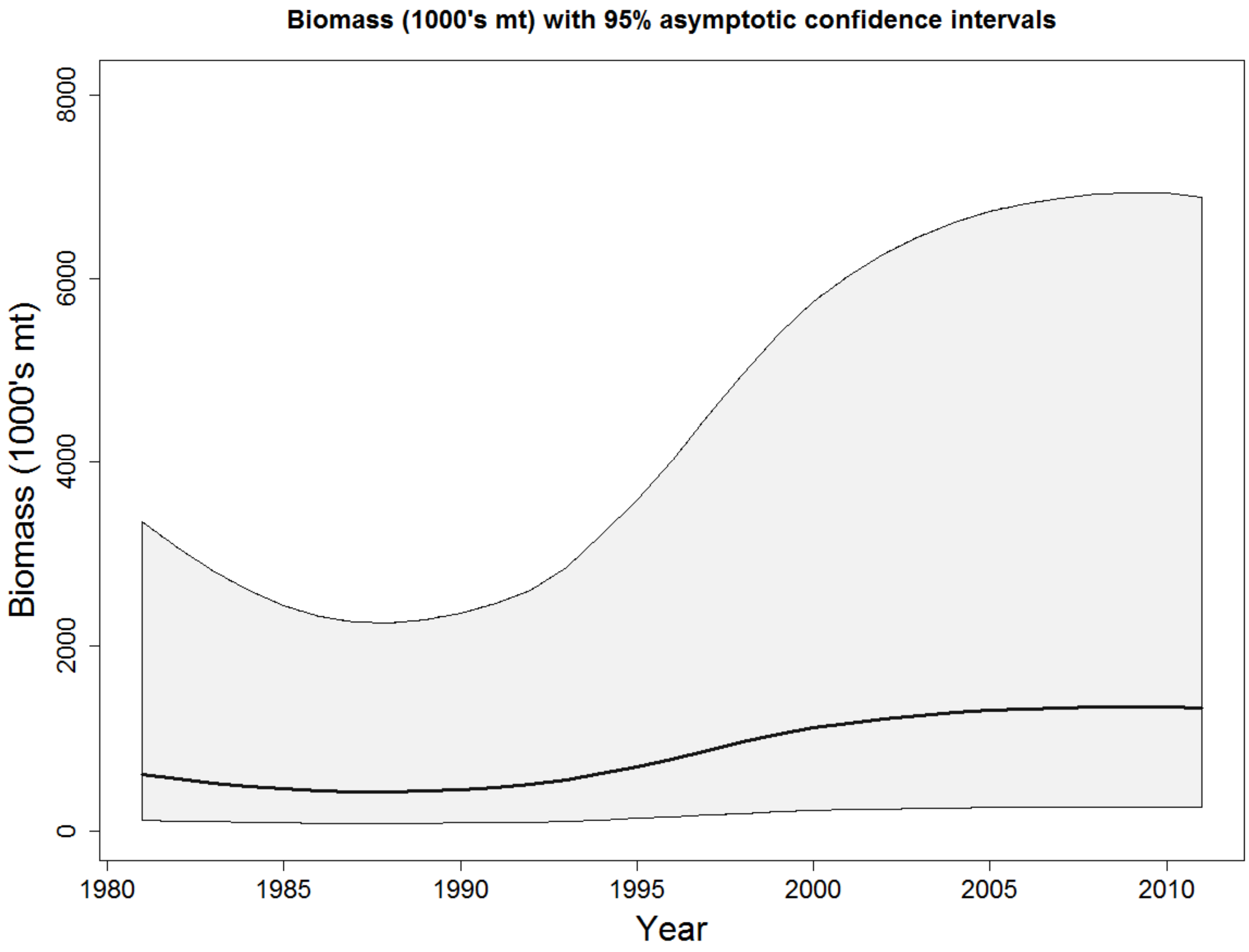
Appendix A5. Figure 15. The data with approximate 95% confidence intervals used to model the northern area (GBK) with KLAMZ.



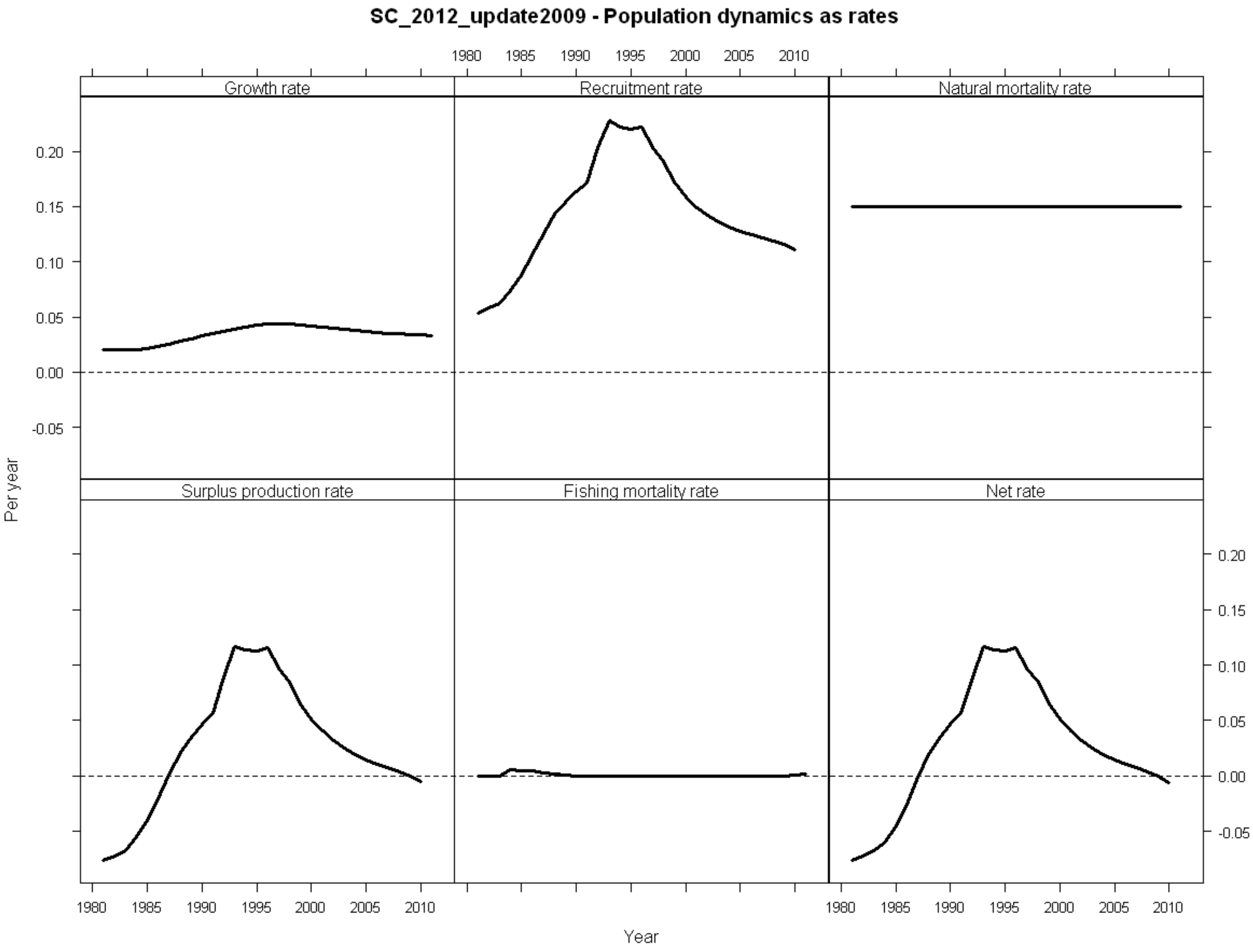
Appendix A5. Figure 16. KLAMZ model fit to the northern area (GBK).



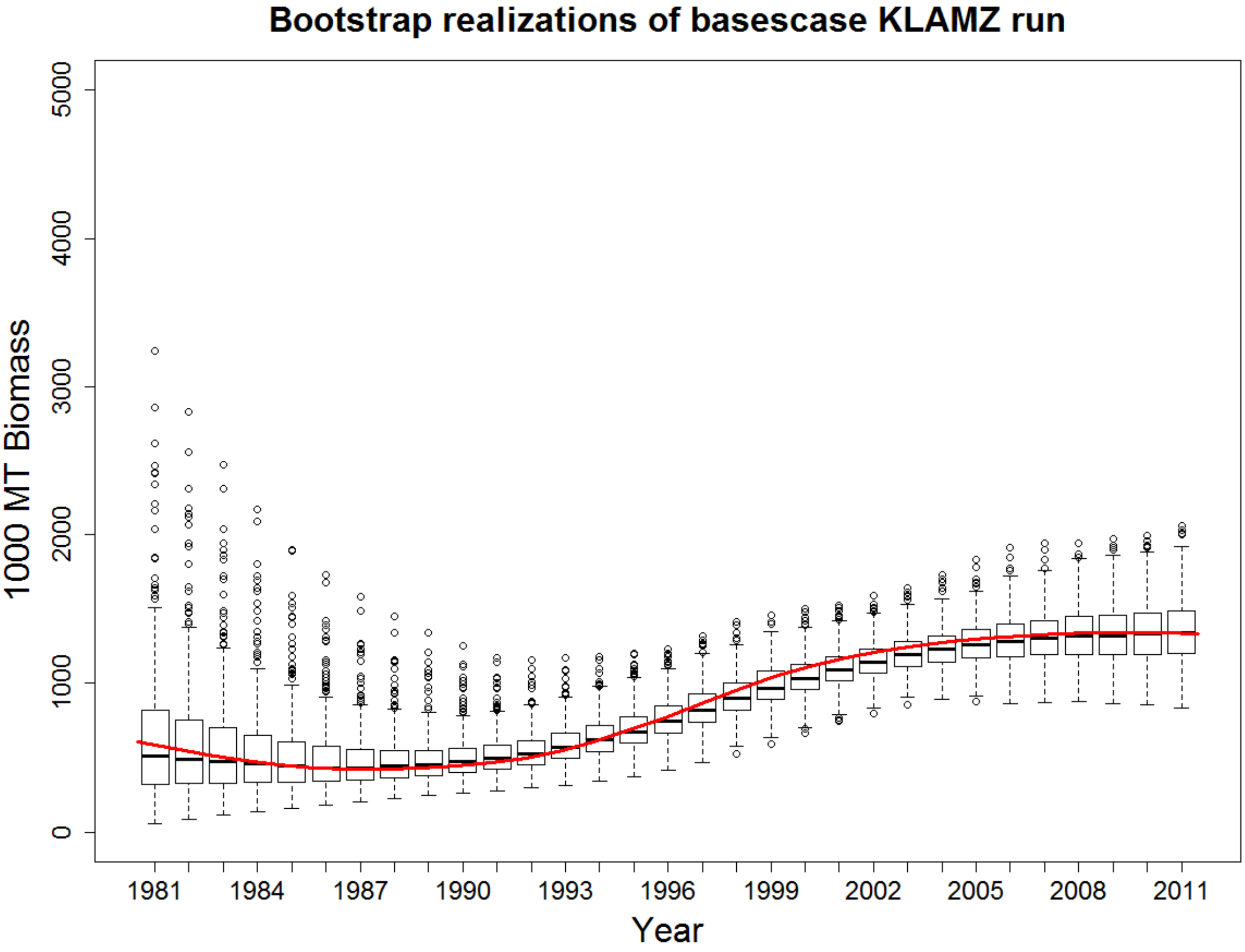
Appendix A5. Figure 17. Sensitivity to σ_R^2 in total biomass for northern area KLAMZ model fit.



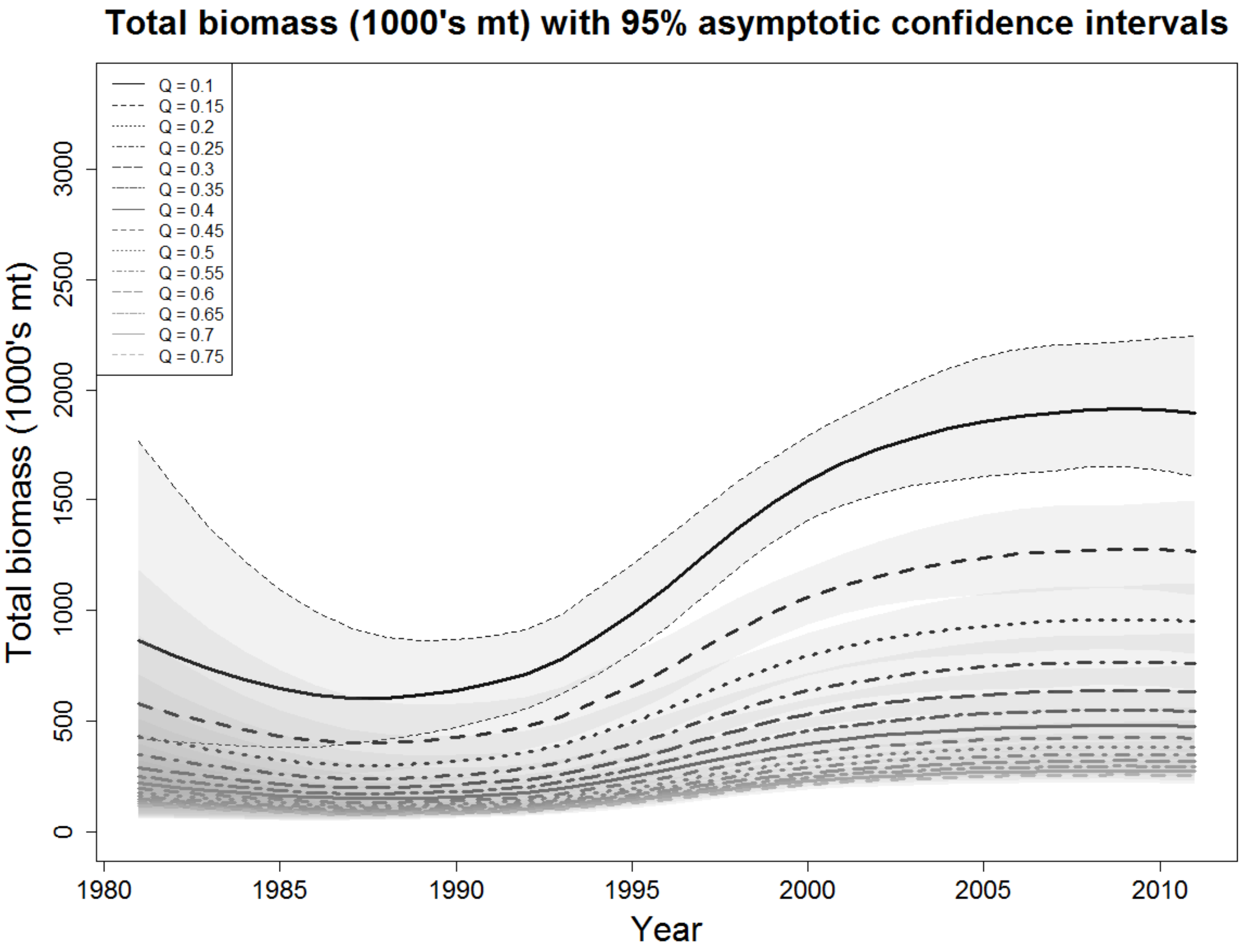
Appendix A5. Figure 18. Trend in biomass in the northern area.



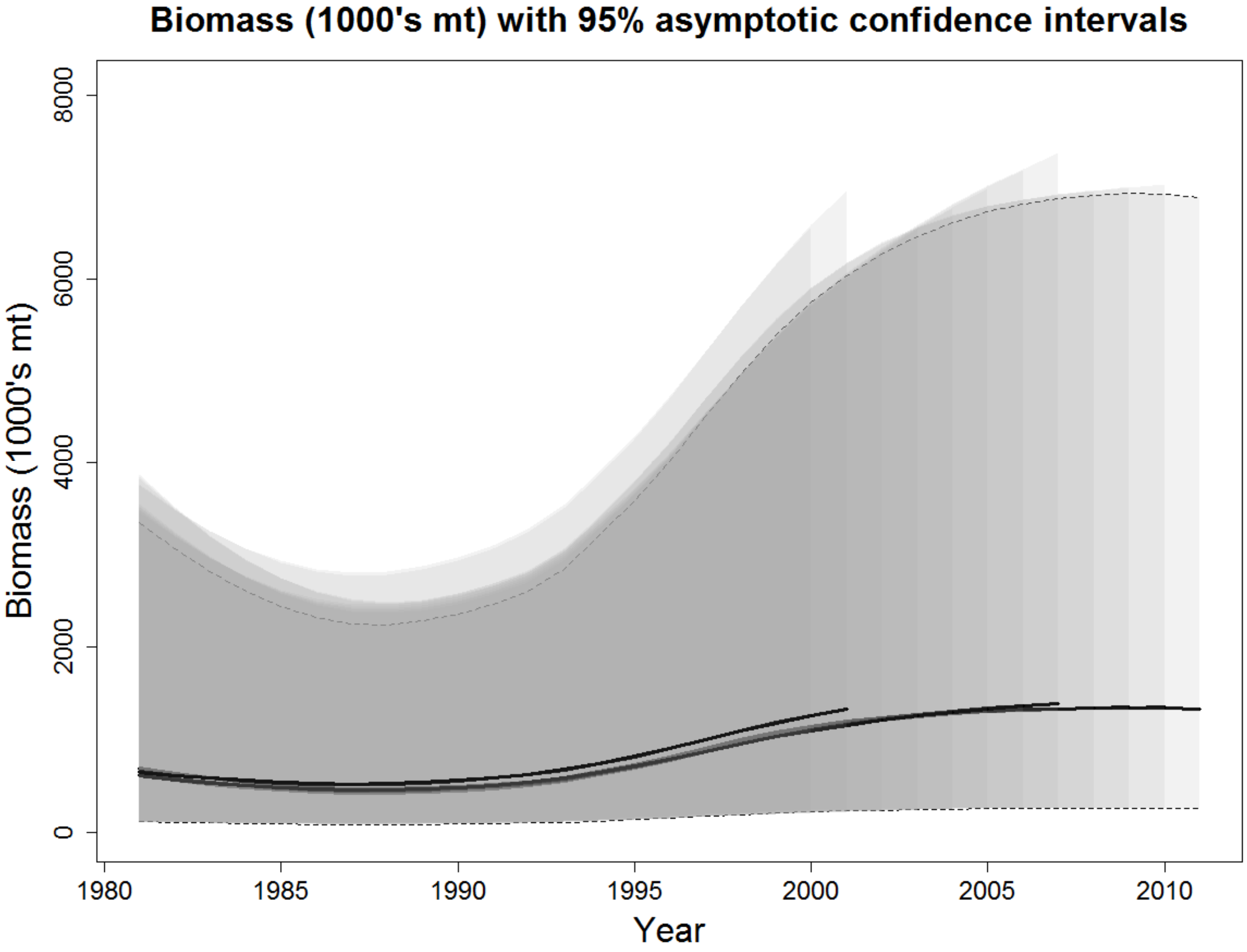
Appendix A5. Figure 19. Population dynamics as rates from KLAMZ model on northern area.



Appendix A5. Figure 20. Bootstrap iterations of the KLAMZ model biomass estimates for the northern area. The base case is shown in red.



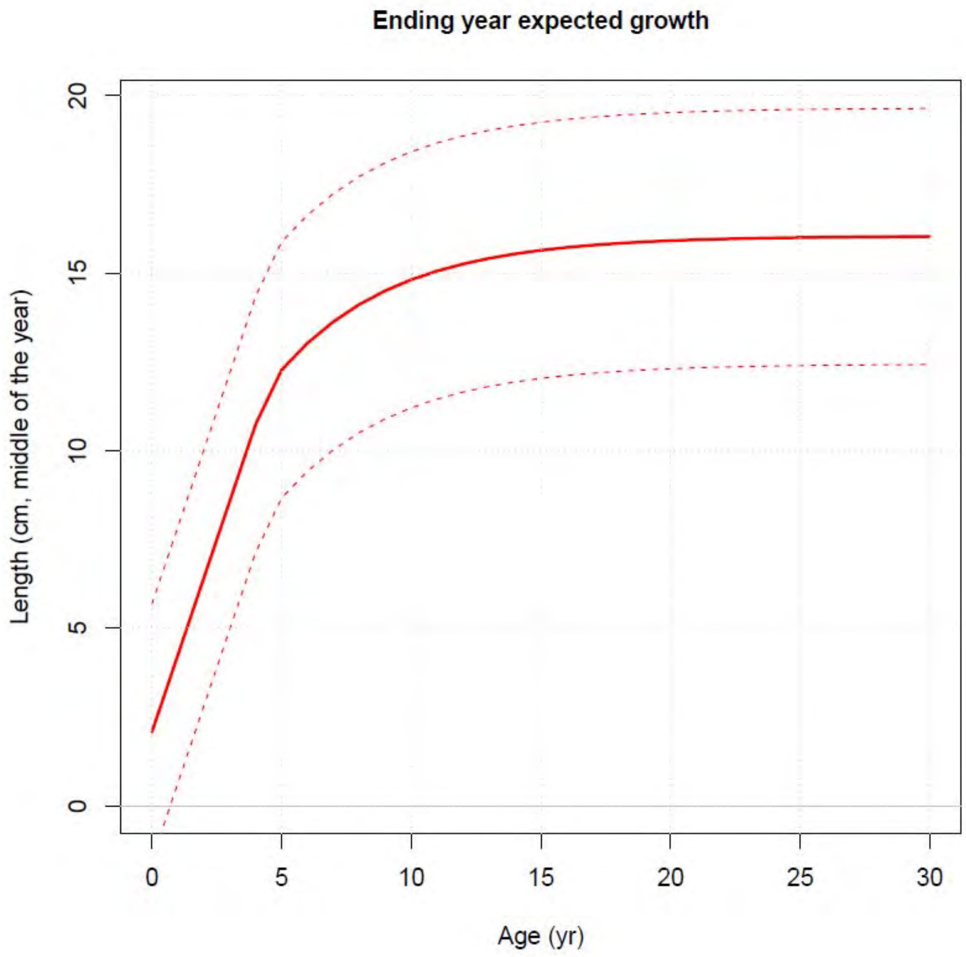
Appendix A5. Figure 21. Profile over survey Q for the northern area.

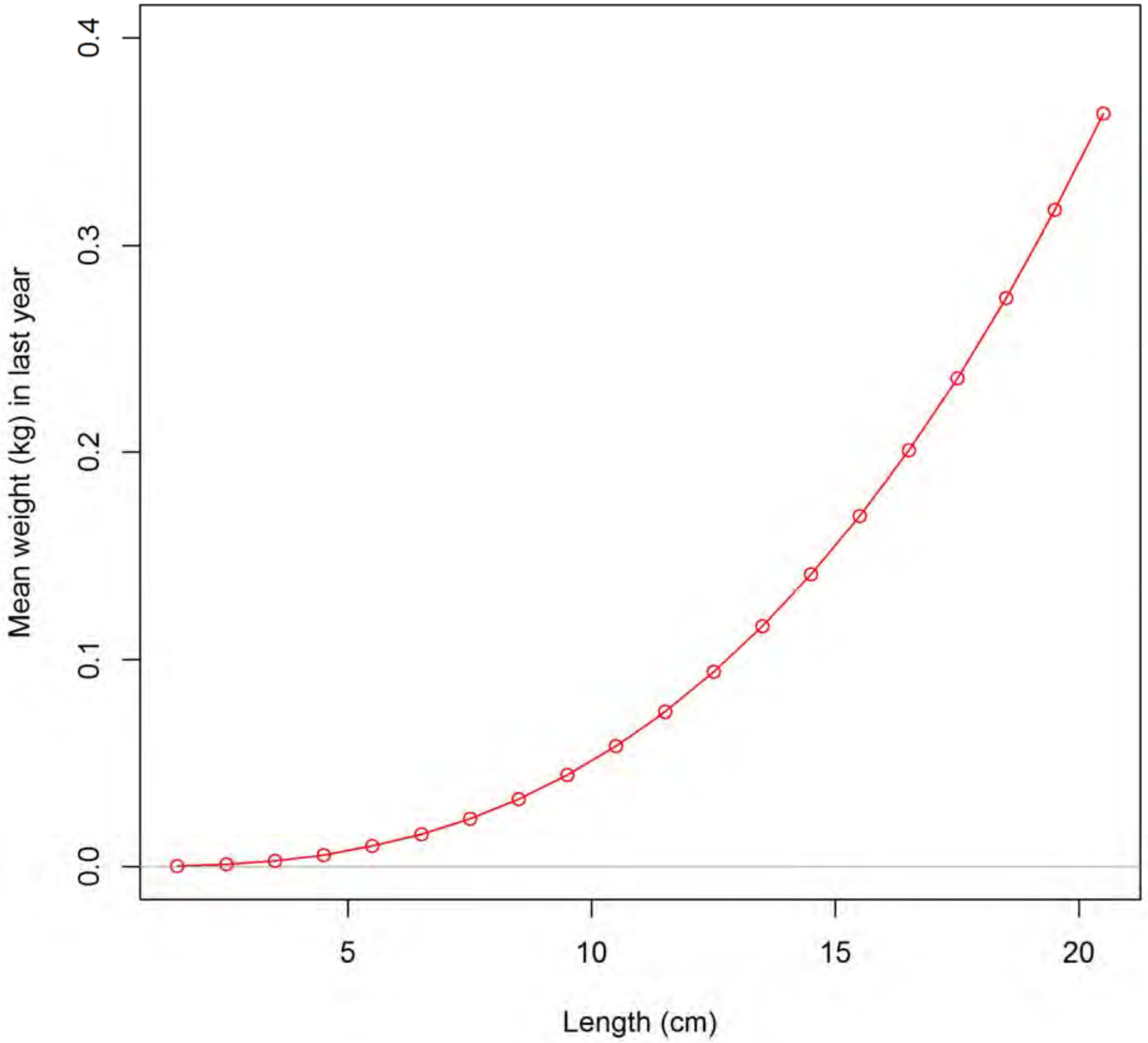


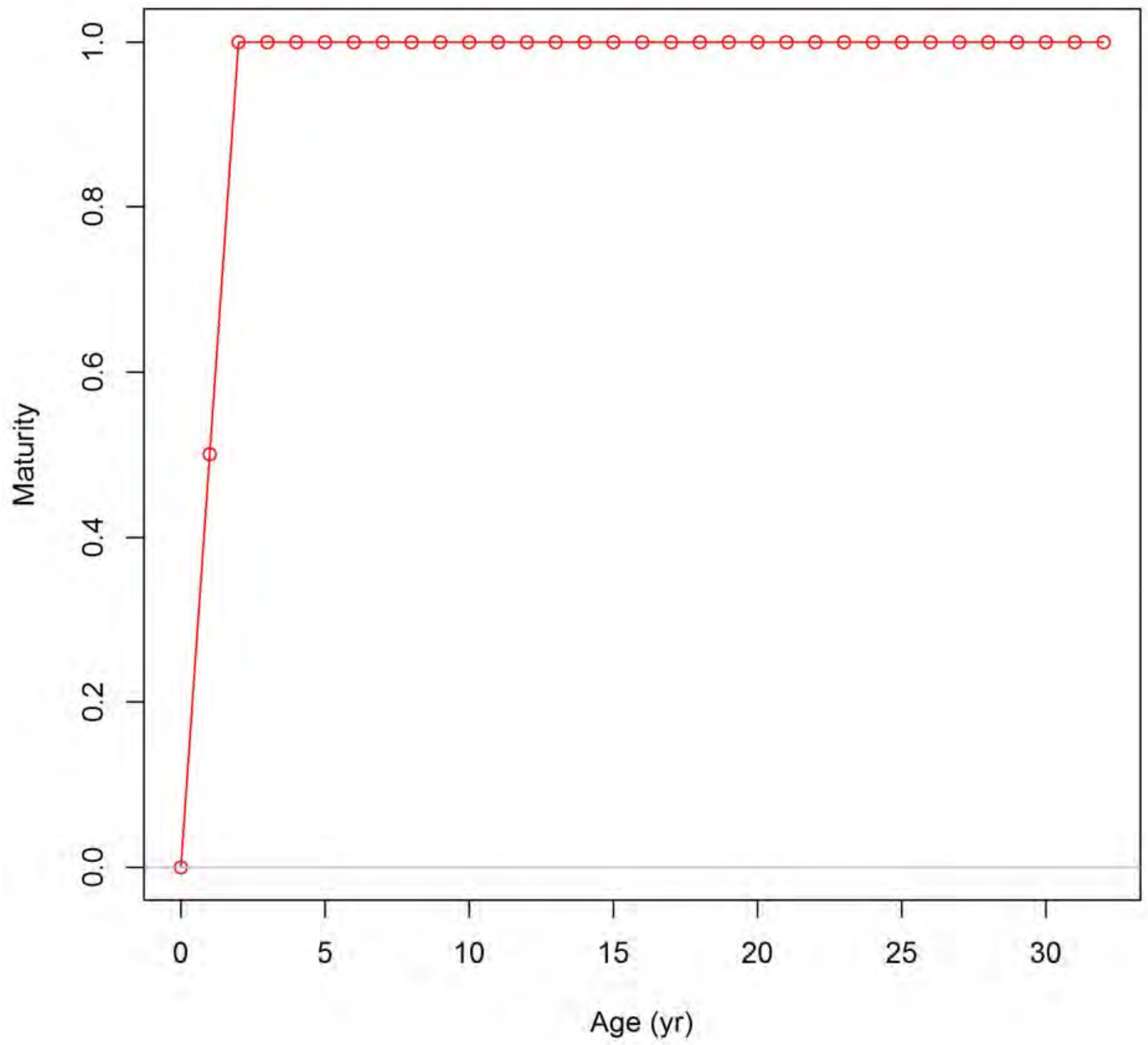
Appendix A5. Figure 22. Retrospective patterns in total biomass for the years 2000-2011 using the base case northern area KLAMZ model.

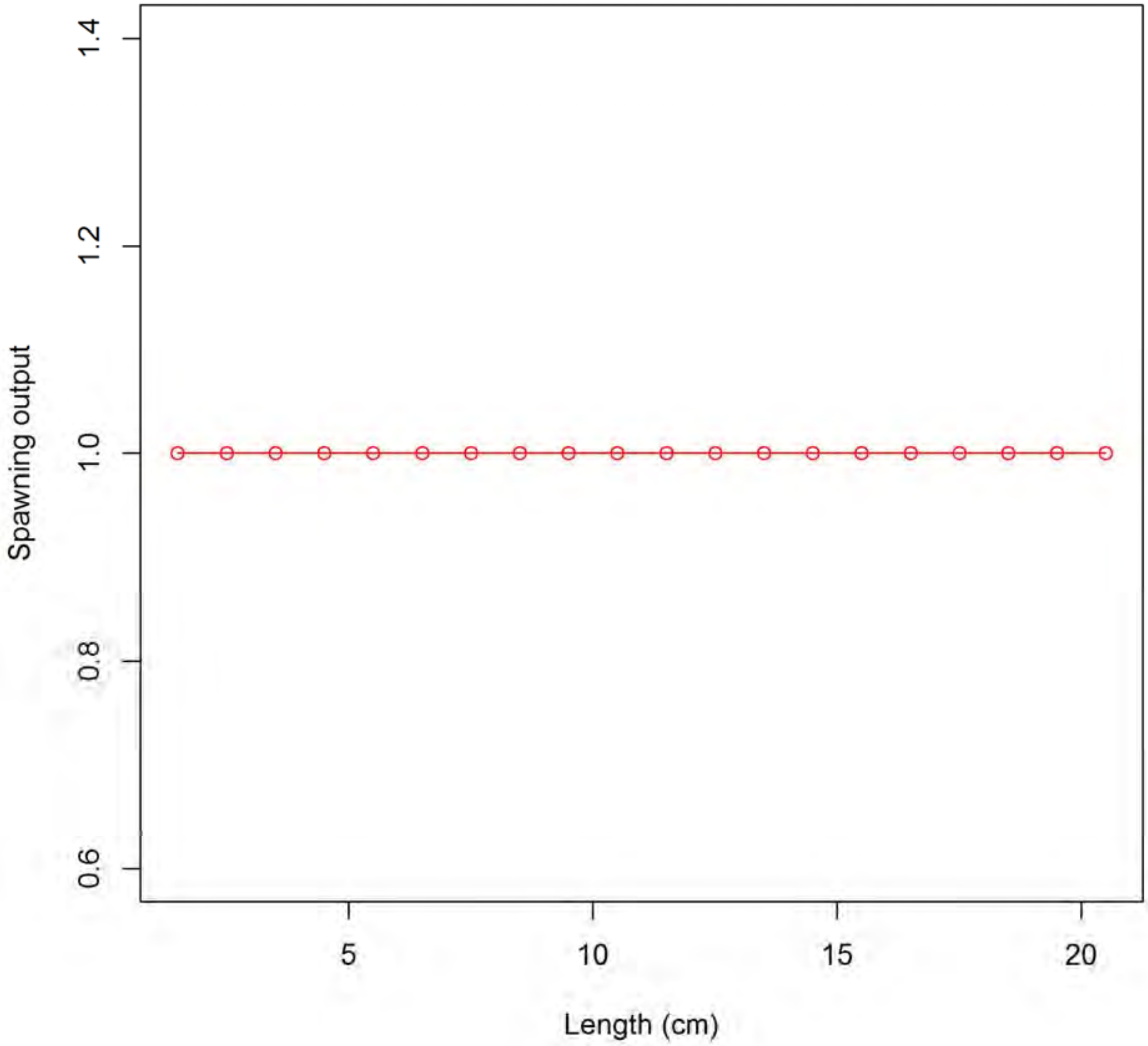
Appendix A6: SS3 diagnostics for the southern area

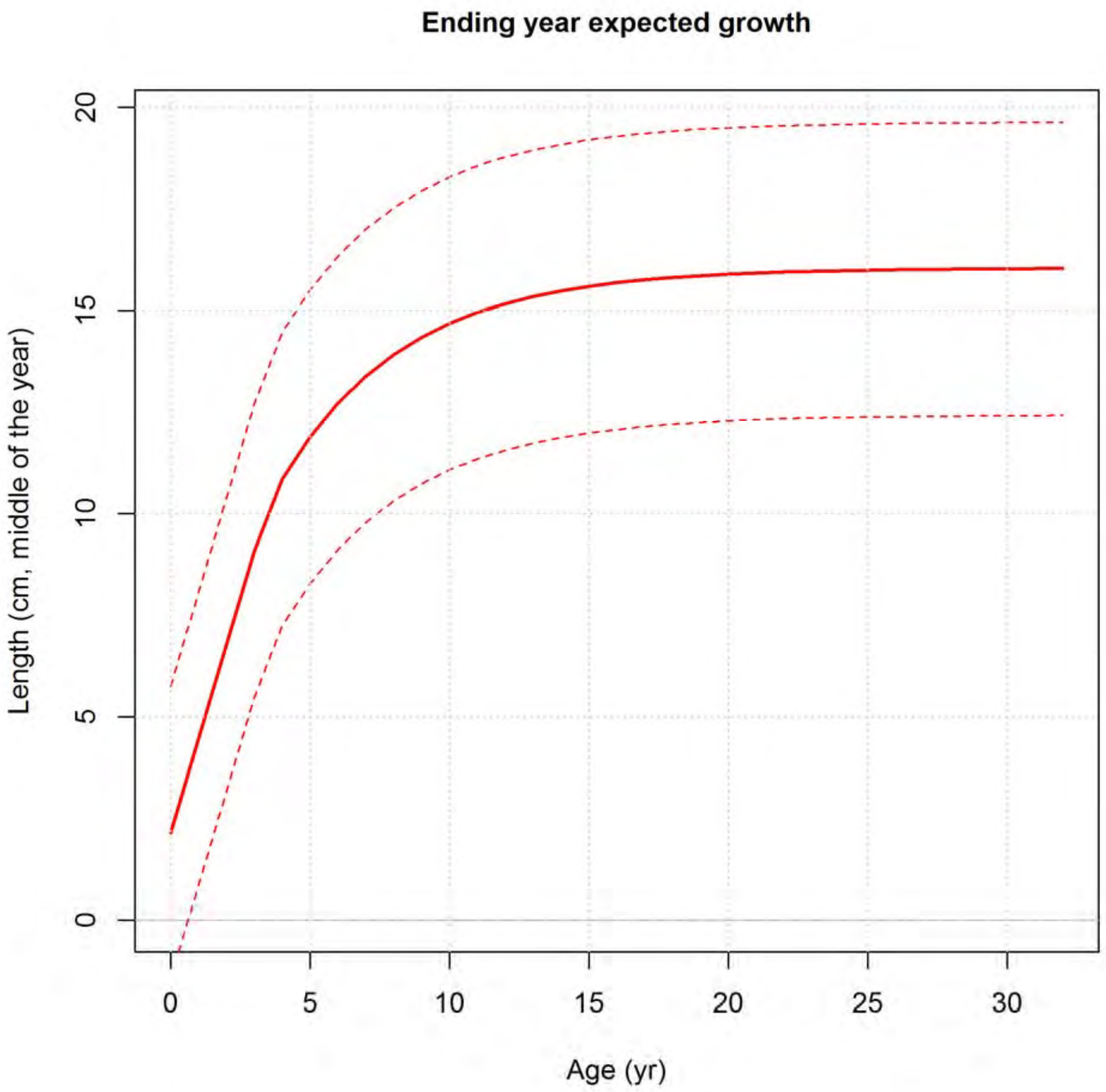
Plots created using the 'r4ss' package in R
Stock Synthesis version: SS−V3.24f
StartTime: Thu Dec 6 12:28:02 2012
Data_File: Surfclam_South−1.dat
Control_File: Surfclam_South−1.ctl

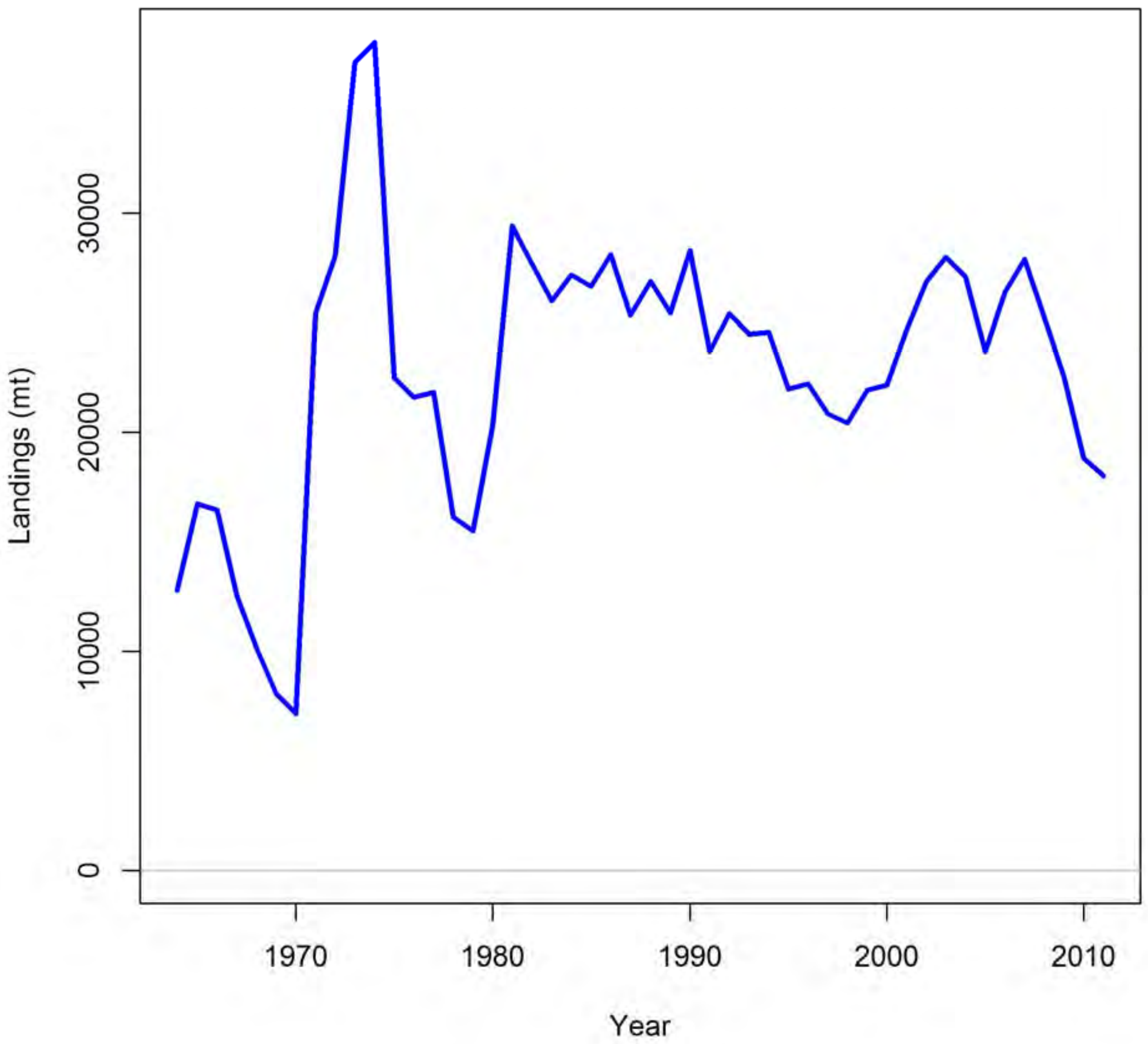

Ending year expected growth
Length (cm, middle of the year)
0
5
10
15
20
0
5
10
15
20
25
30
Age (yr)

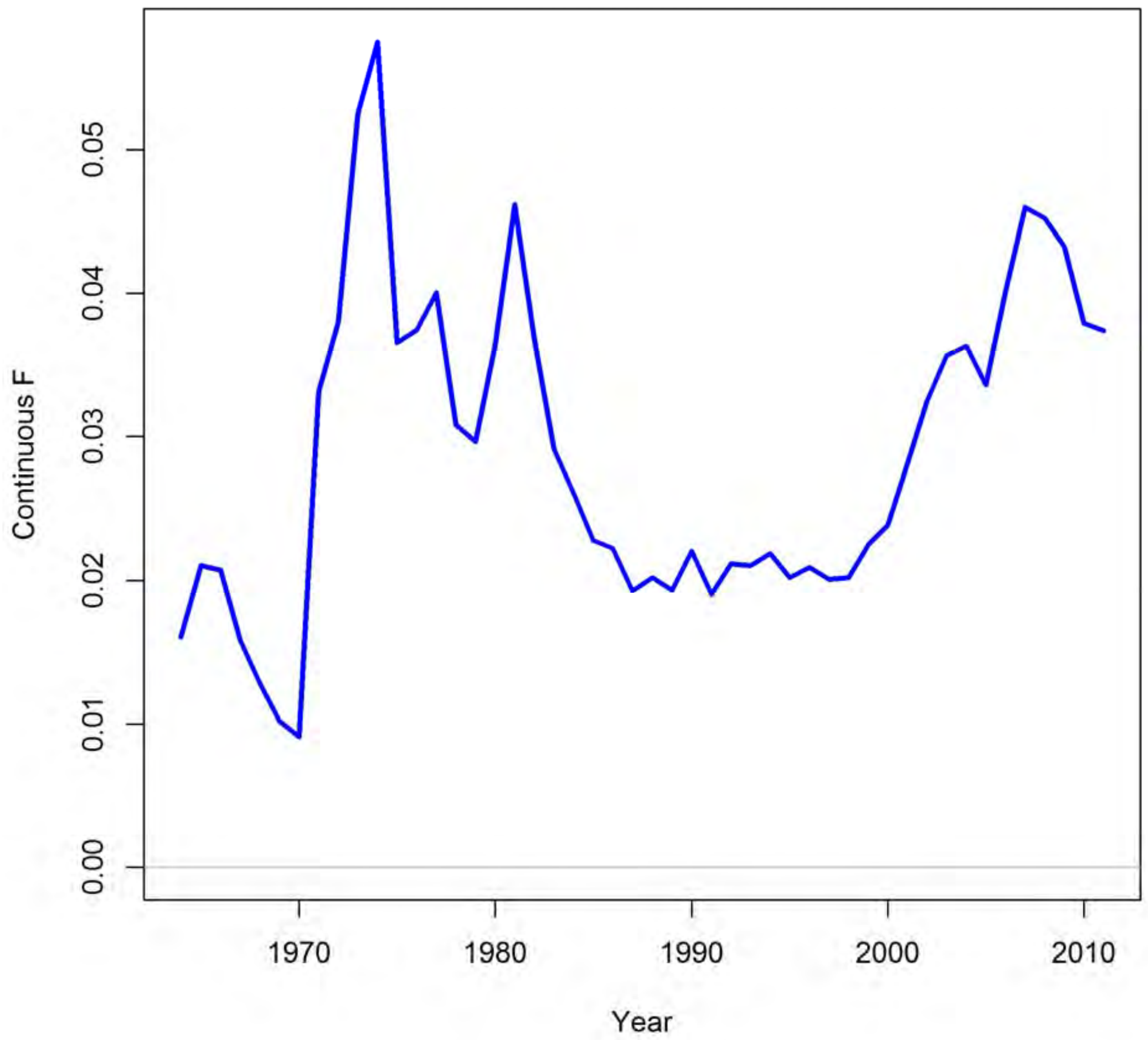

0.4
0.3
0.2
0.1
0.0
Mean weight (kg) in last year
5
10
15
20
Length (cm)


Maturity
0.0
0.2
0.4
0.6
0.8
1.0
0
5
10
15
20
25
30
Age (yr)

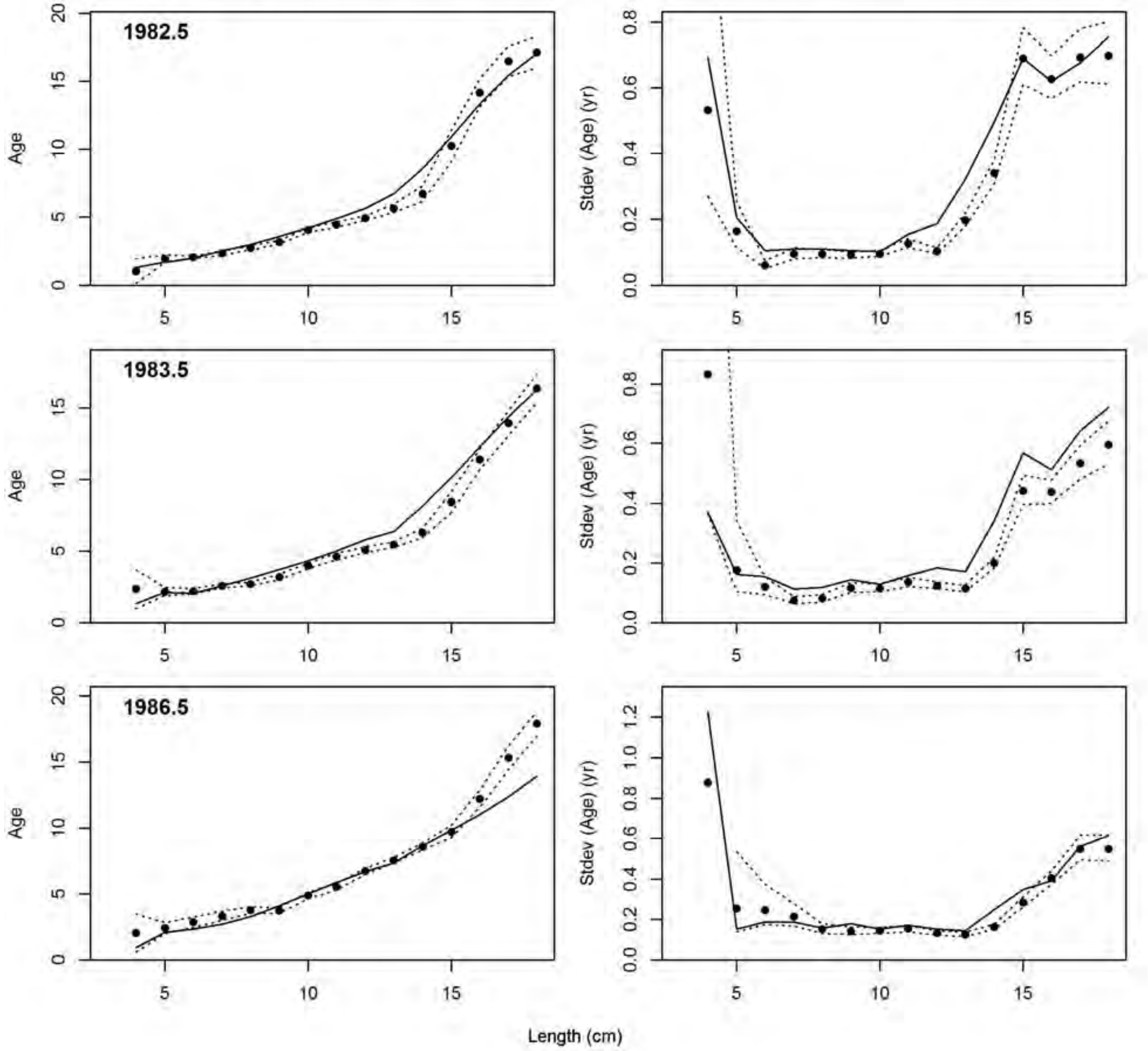

Spawning output
1.4
1.2
1.0
0.8
0.6
5
10
15
20
Length (cm)


Ending year expected growth
Length (cm, middle of the year)
0
5
10
15
20
Age (yr)
0
5
10
15
20
25
30

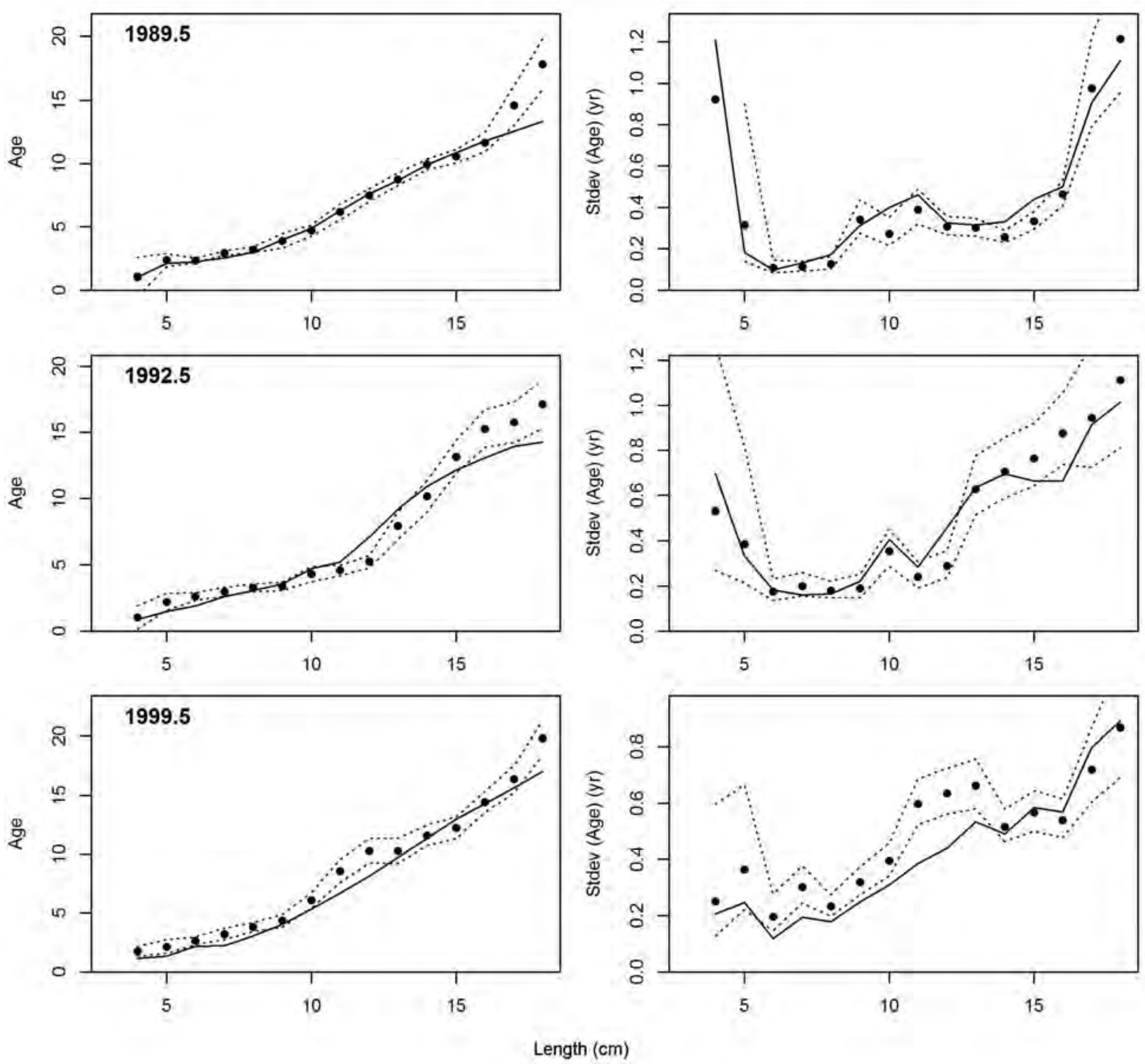

Landings (mt)
0
10000
20000
30000
1970
1980
1990
2000
2010
Year


Continuous F
0.00
0.01
0.02
0.03
0.04
0.05
1970
1980
1990
2000
2010
Year

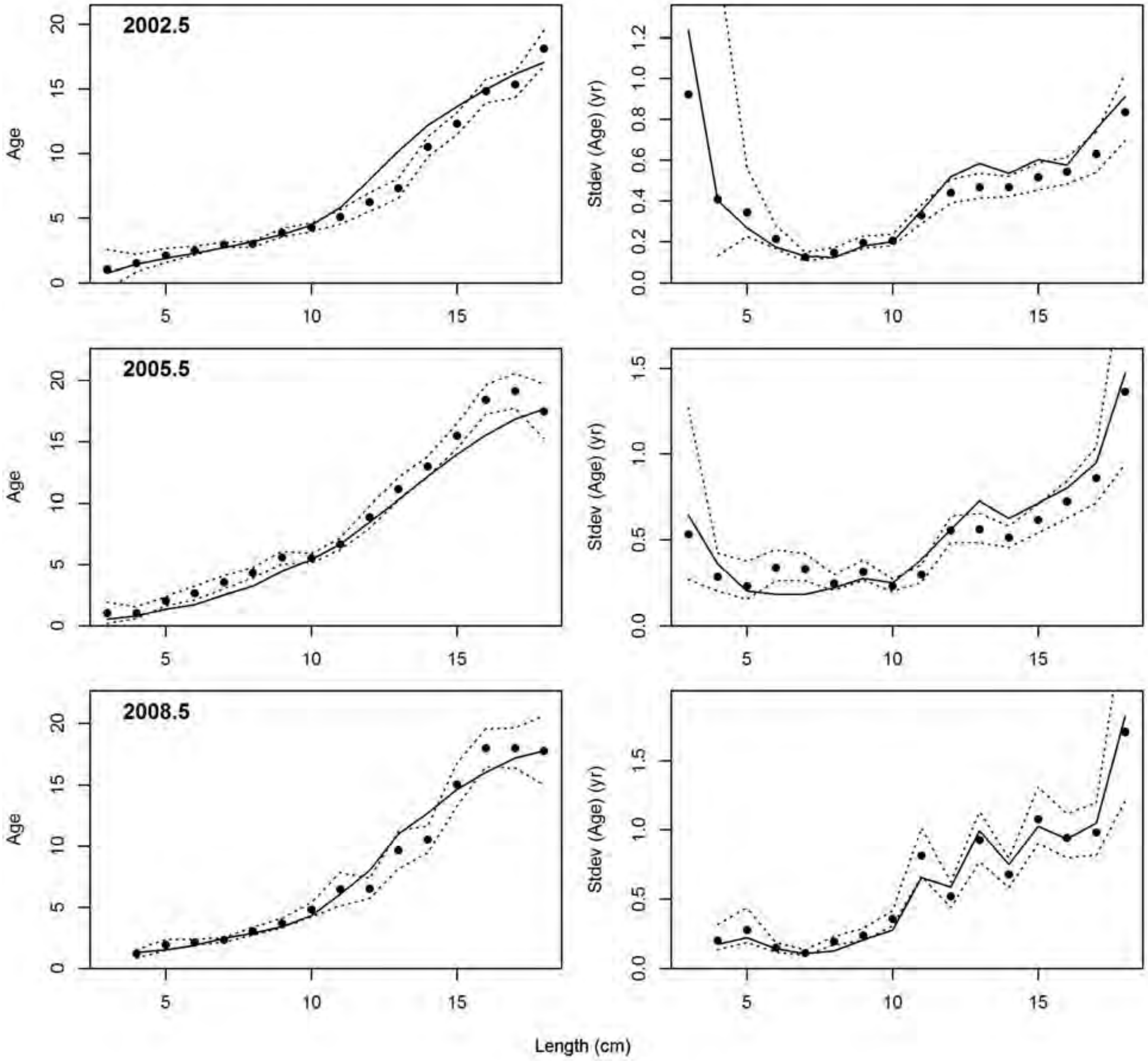
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm

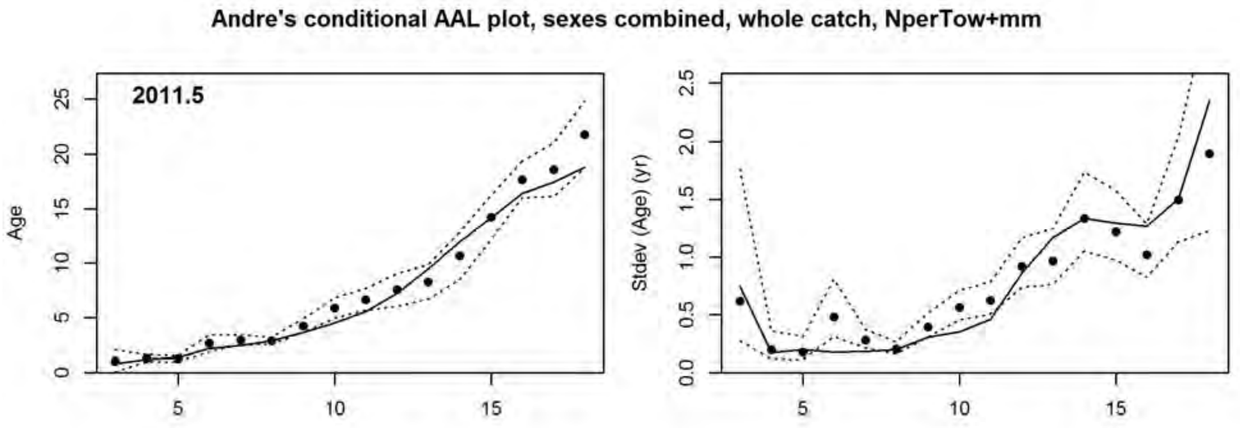


Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm



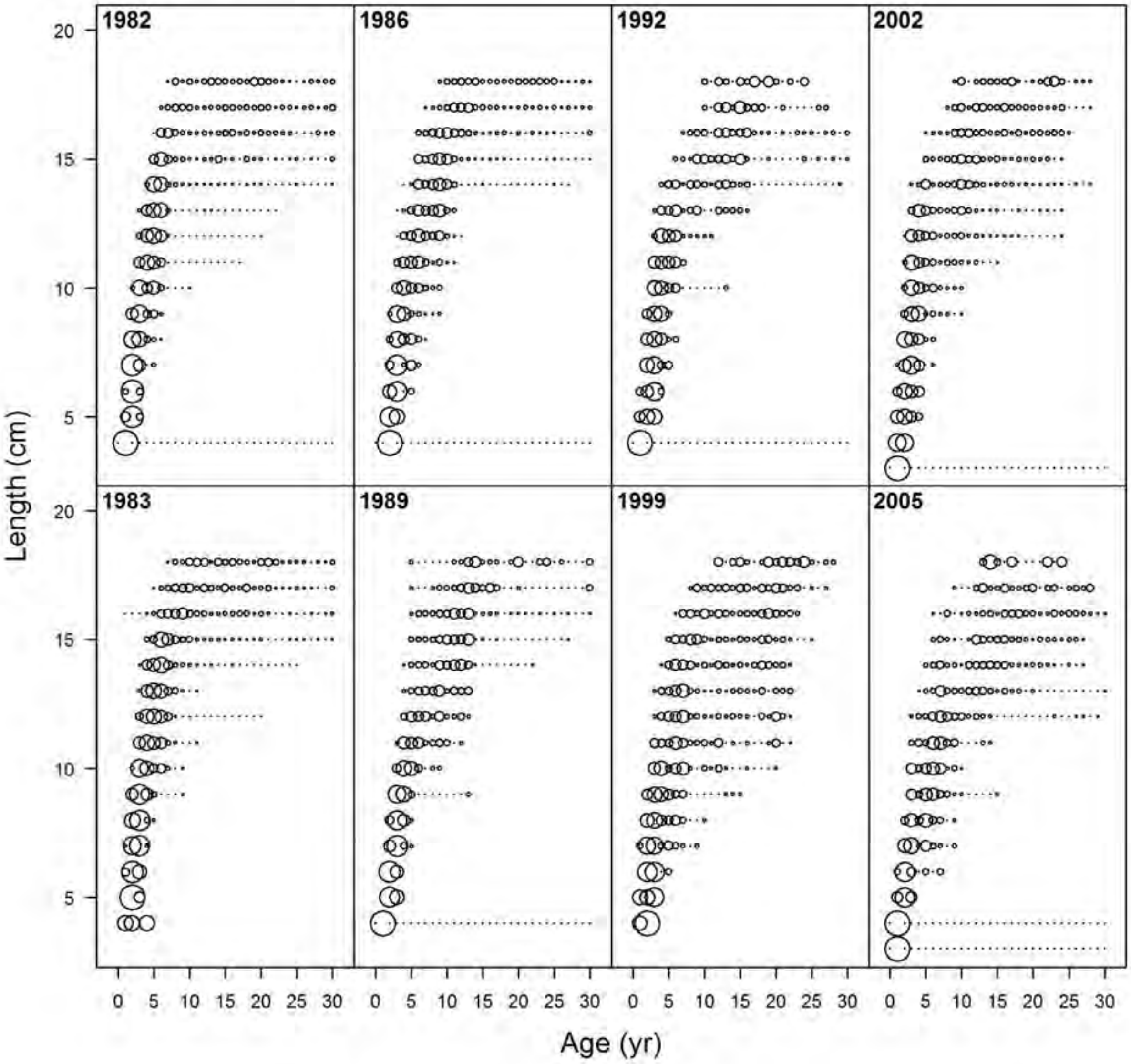
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm



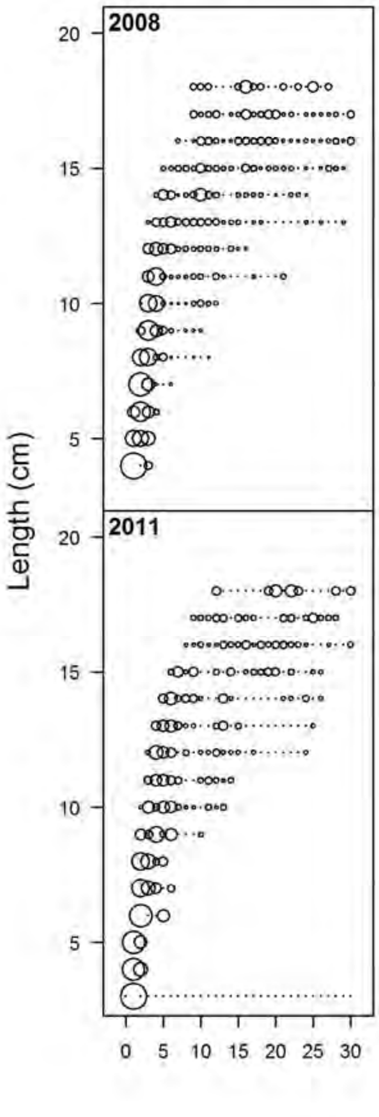


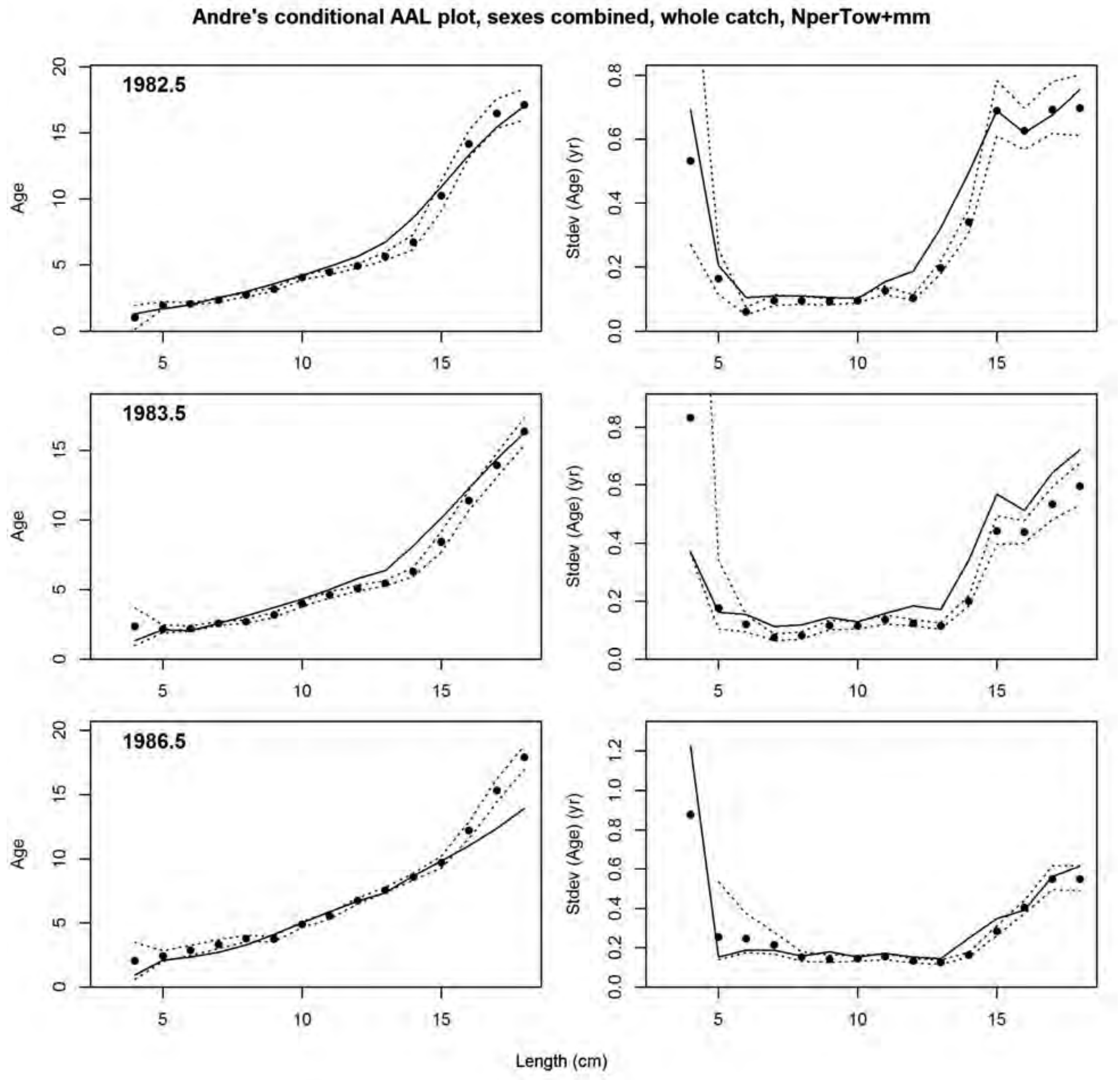
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm

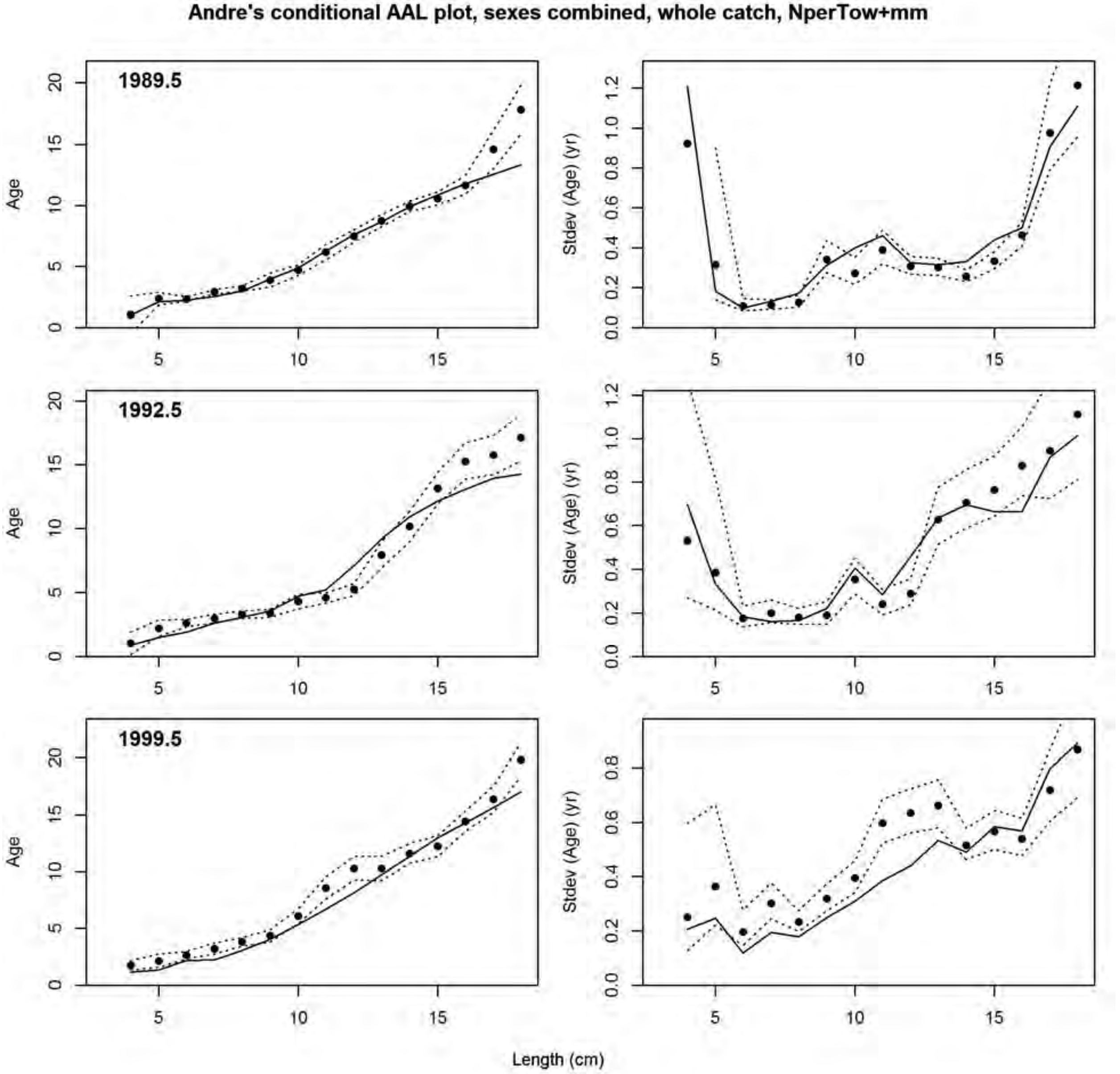
conditional age-at-length data, sexes combined, whole catch, NperTow+mm (max=1



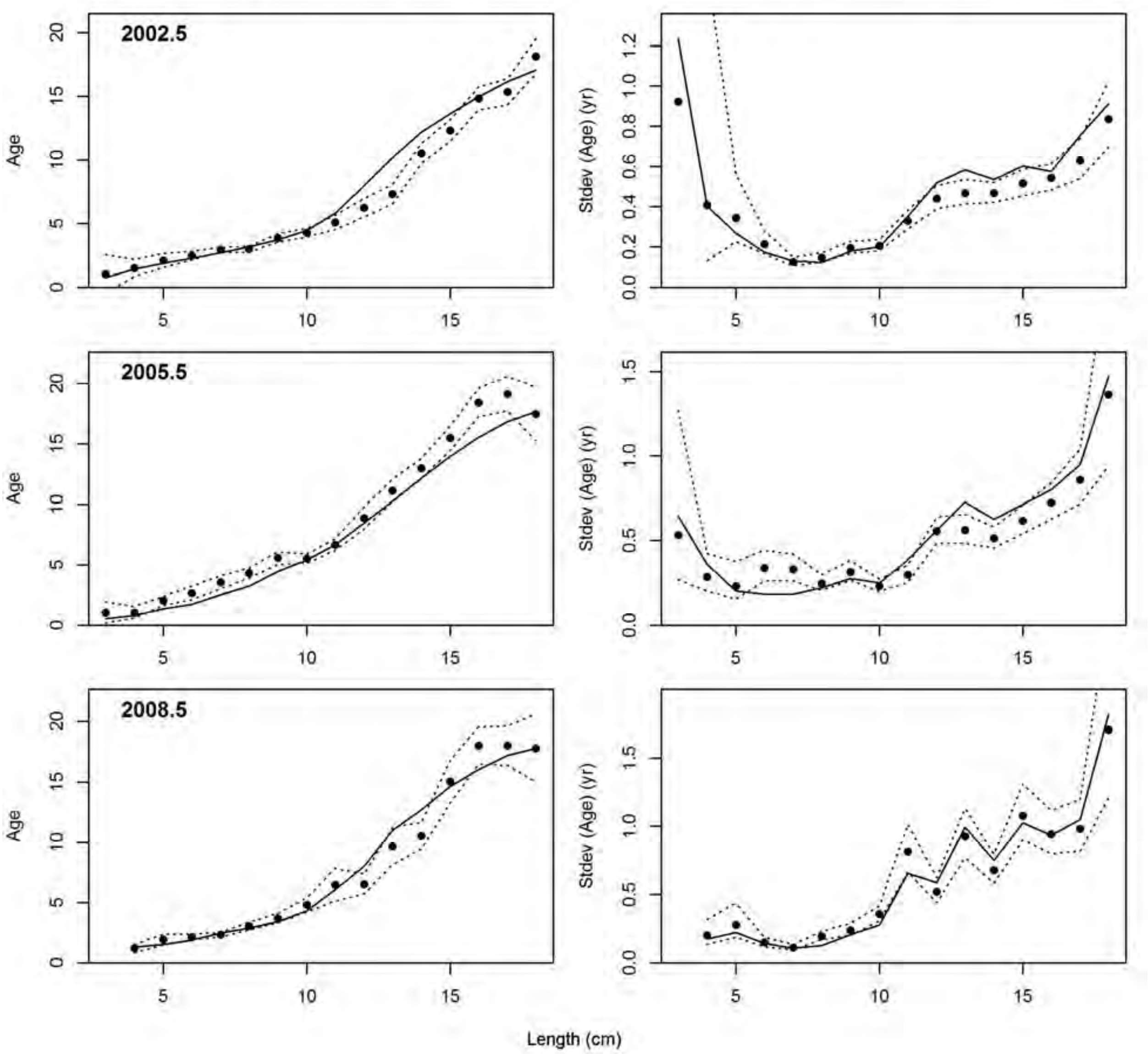
conditional age-at-length data, sexes combined, whole catch, NperTow+mm (max=1

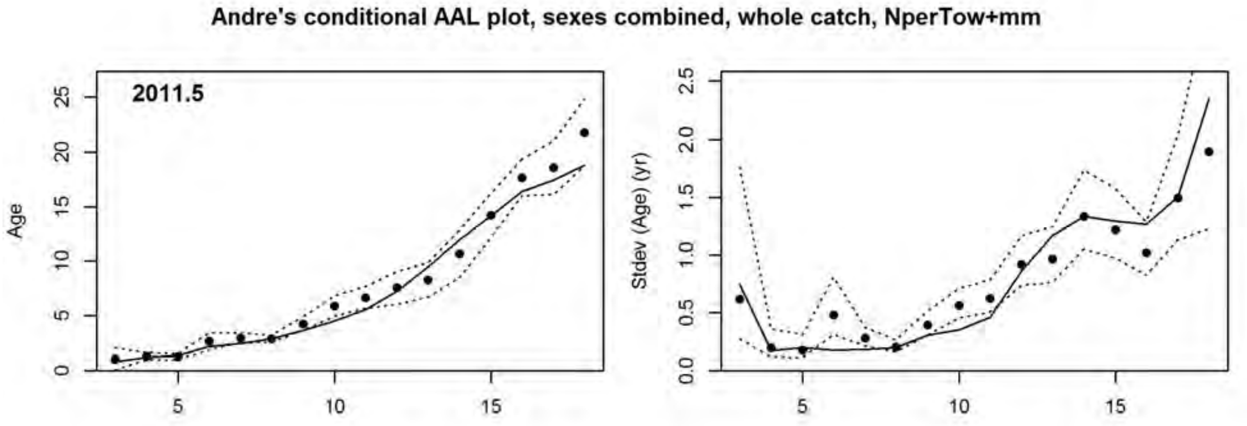



Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm
1982.5
1983.5
1986.5
Age
Stdev (Age) (yr)
0
5
10
15
20
0.0
0.2
0.4
0.6
0.8
1.0
1.2
Length (cm)

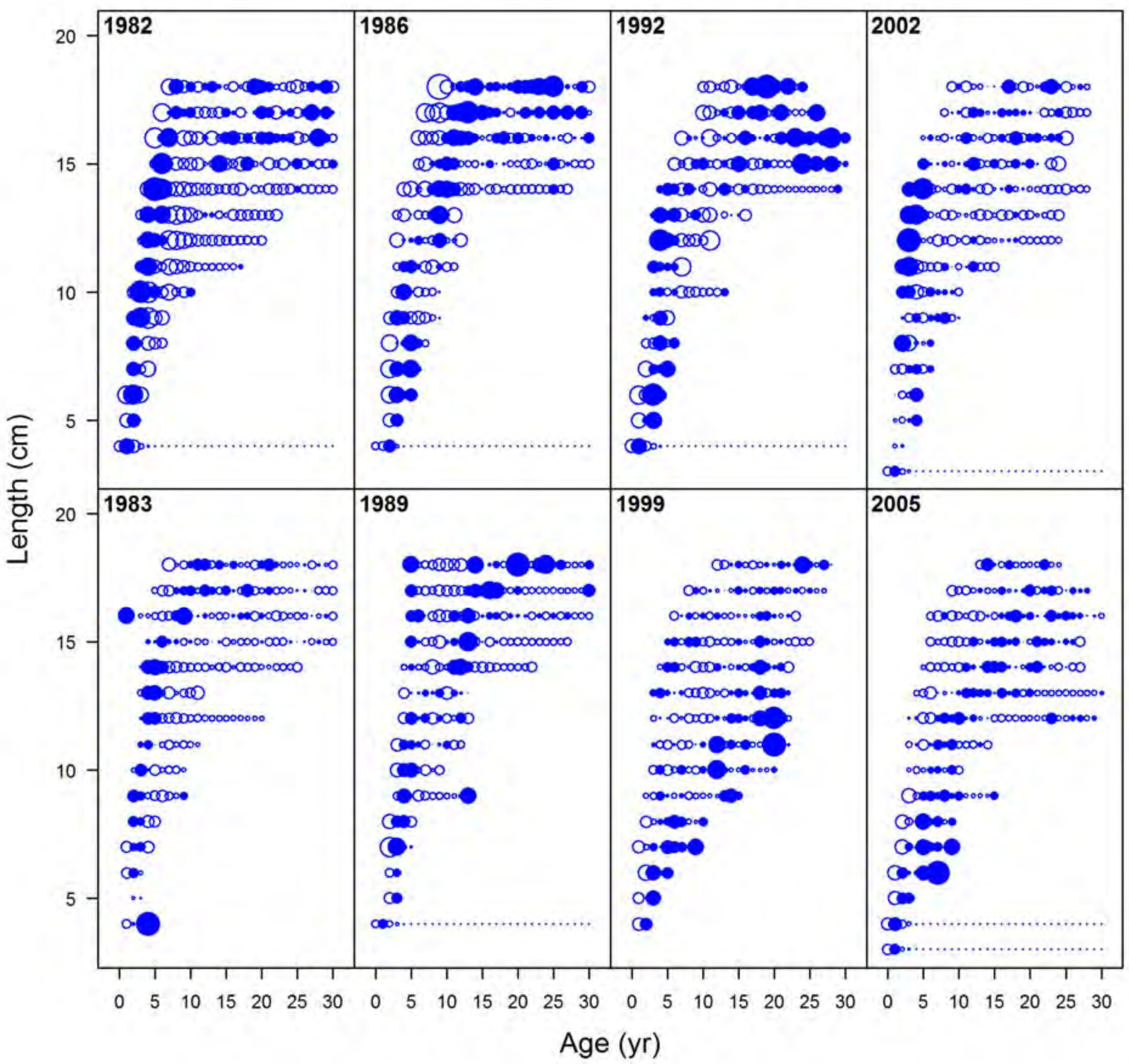

Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm
1989.5
1992.5
1999.5
Age
Stdev (Age) (yr)
0
5
10
15
20
0.0
0.2
0.4
0.6
0.8
1.0
1.2
Length (cm)

Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm

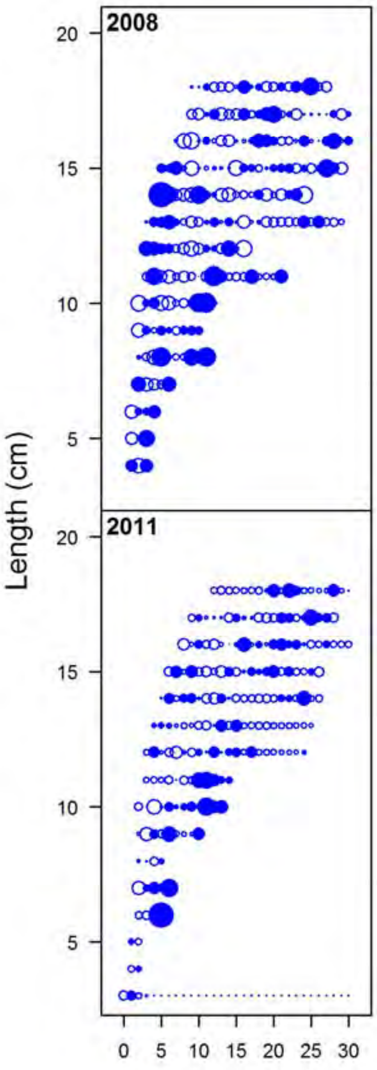



Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm
2011.5
Age
0
5
10
15
20
25
5
10
15
Stdev (Age) (yr)
0.0
0.5
1.0
1.5
2.0
2.5
5
10
15
Length (cm)

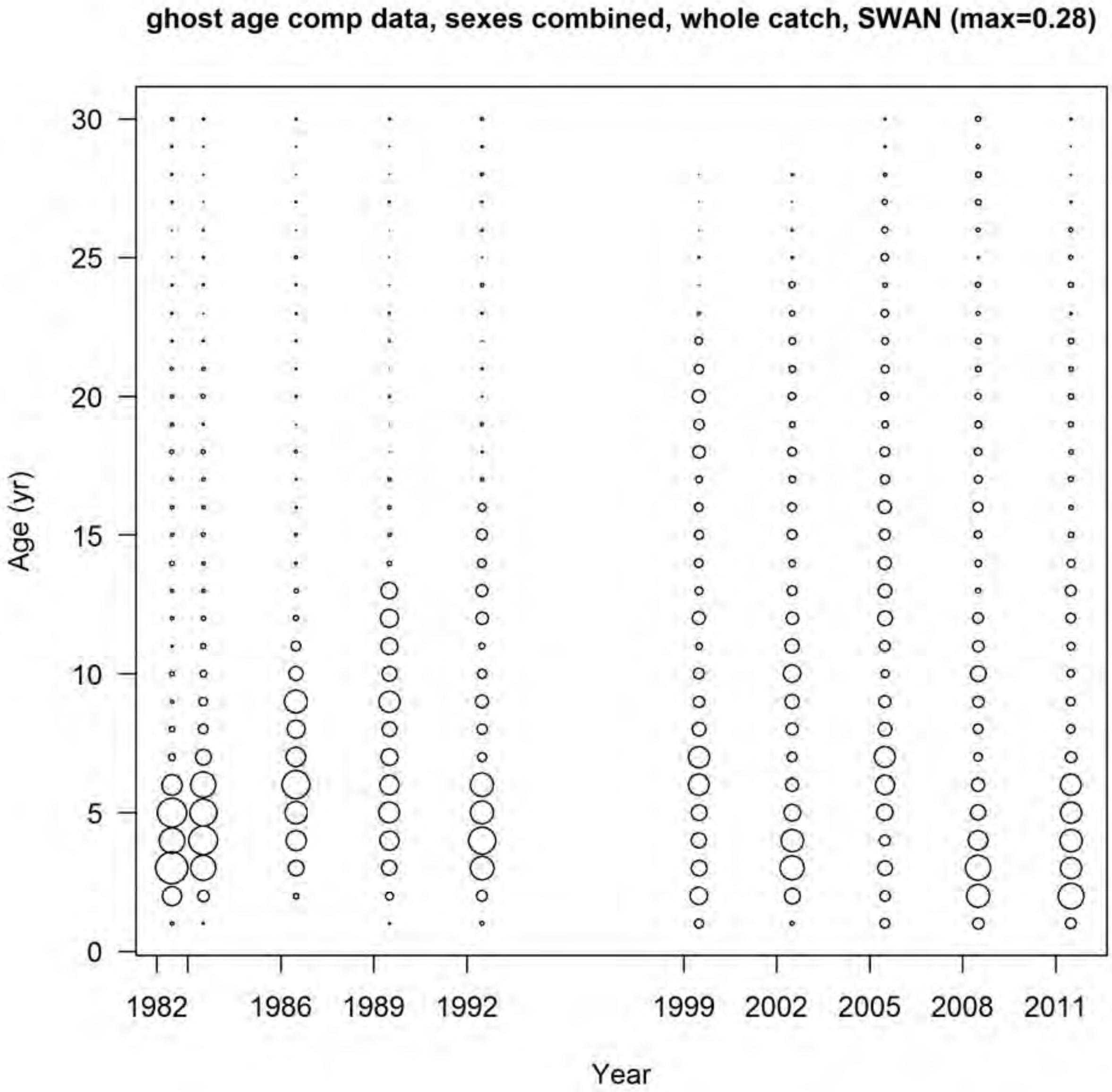
Pearson residuals, sexes combined, whole catch, NperTow+mm (max=10.83)



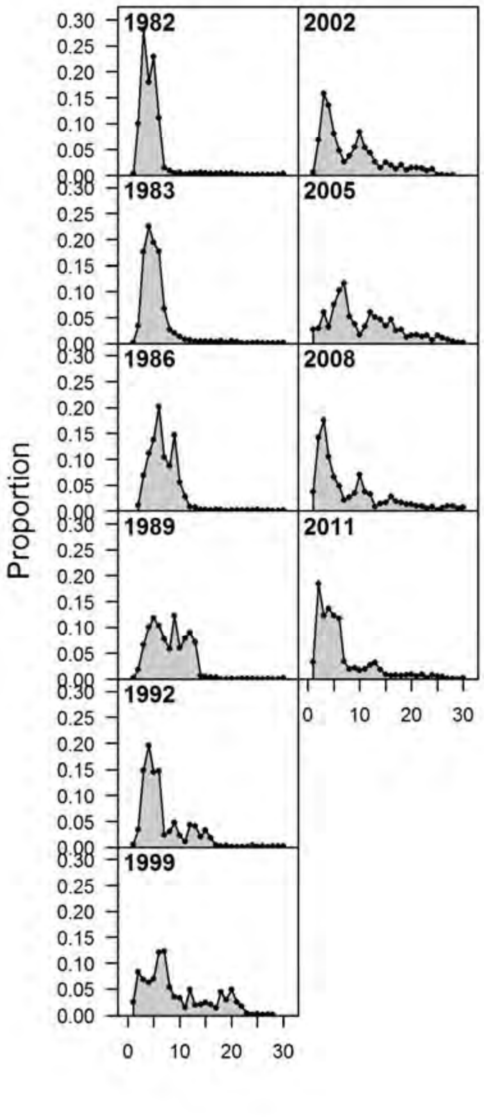
Pearson residuals, sexes combined, whole catch, NperTow+mm (max=10.83)

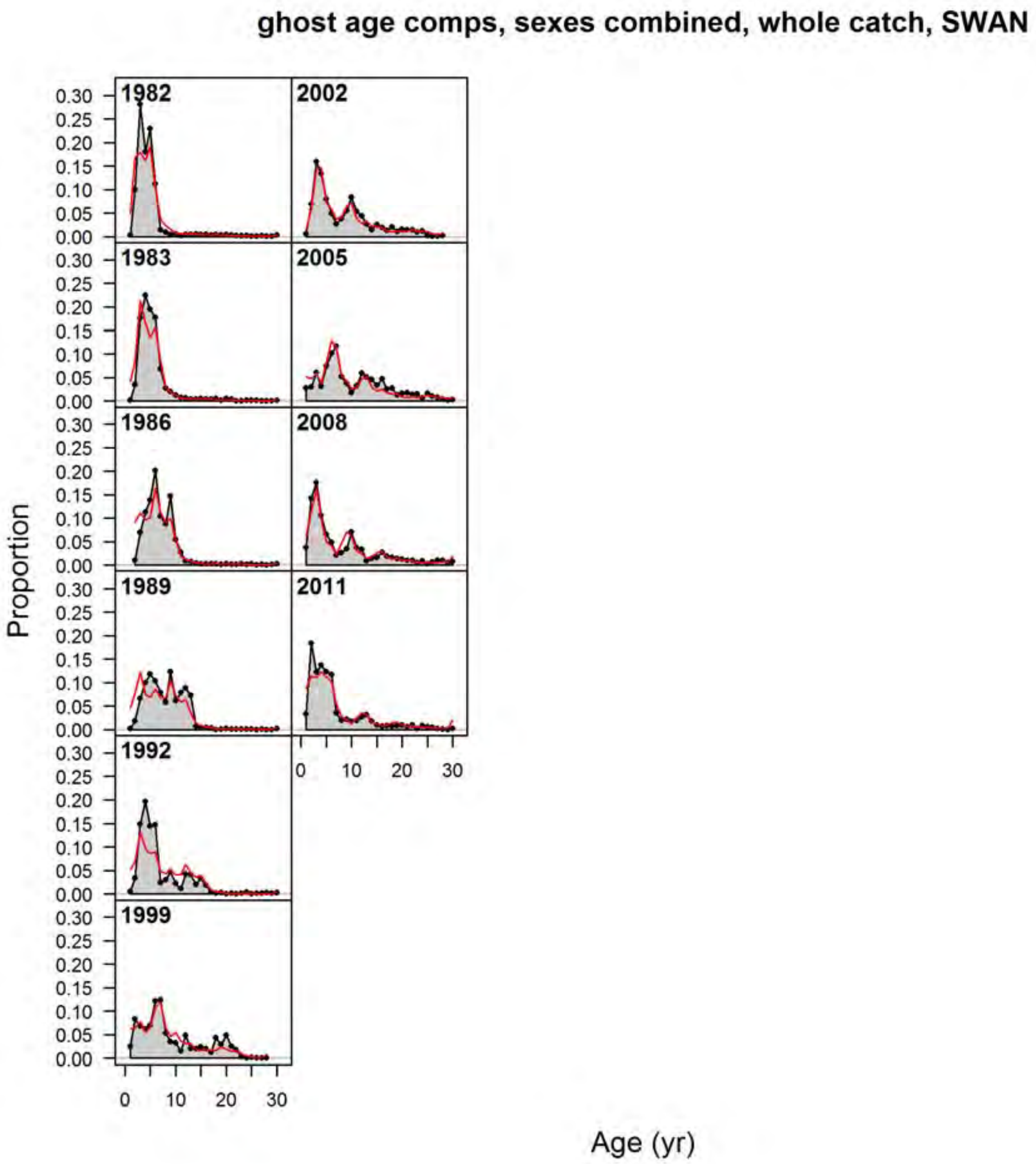


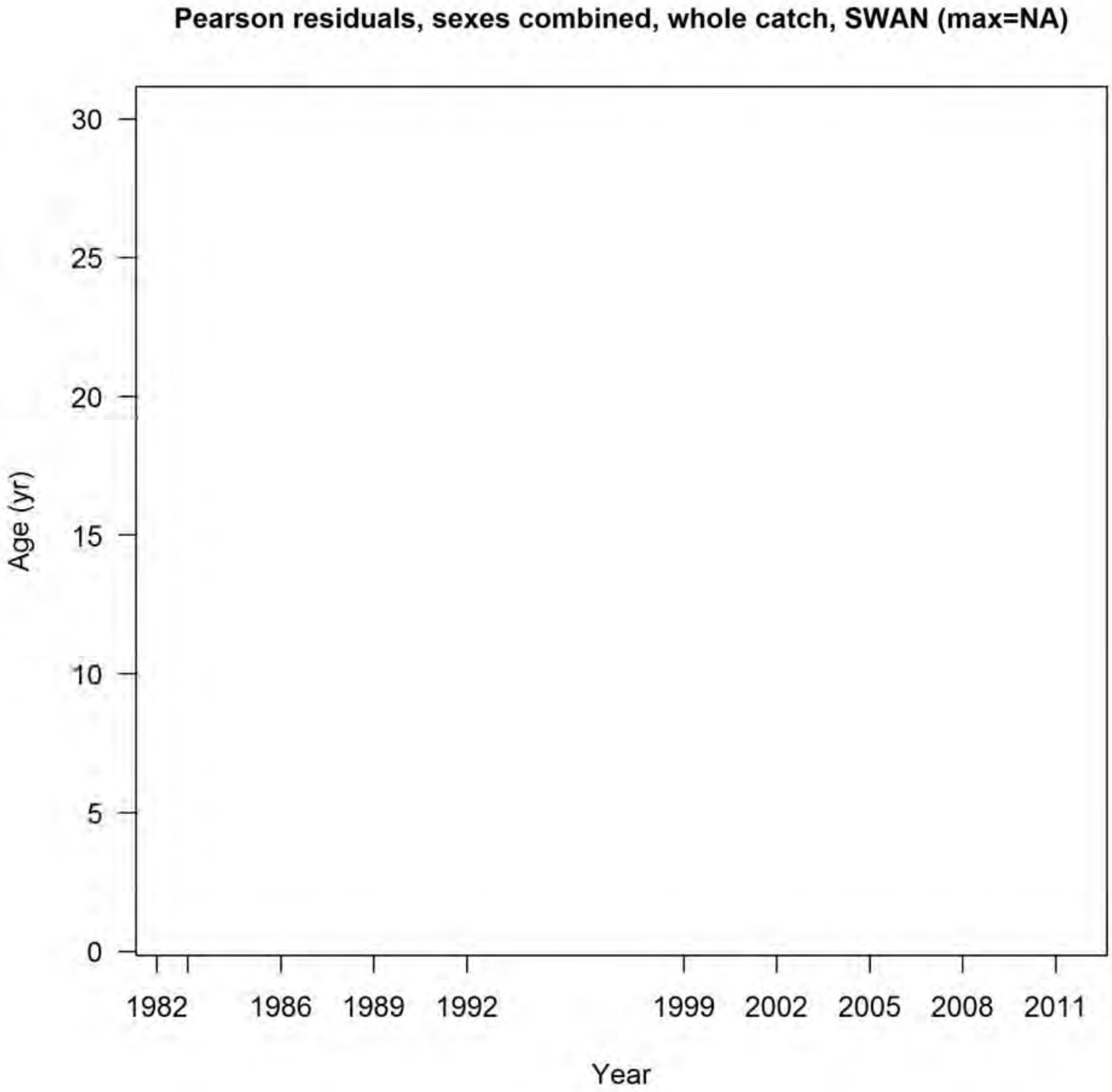
Age (yr)

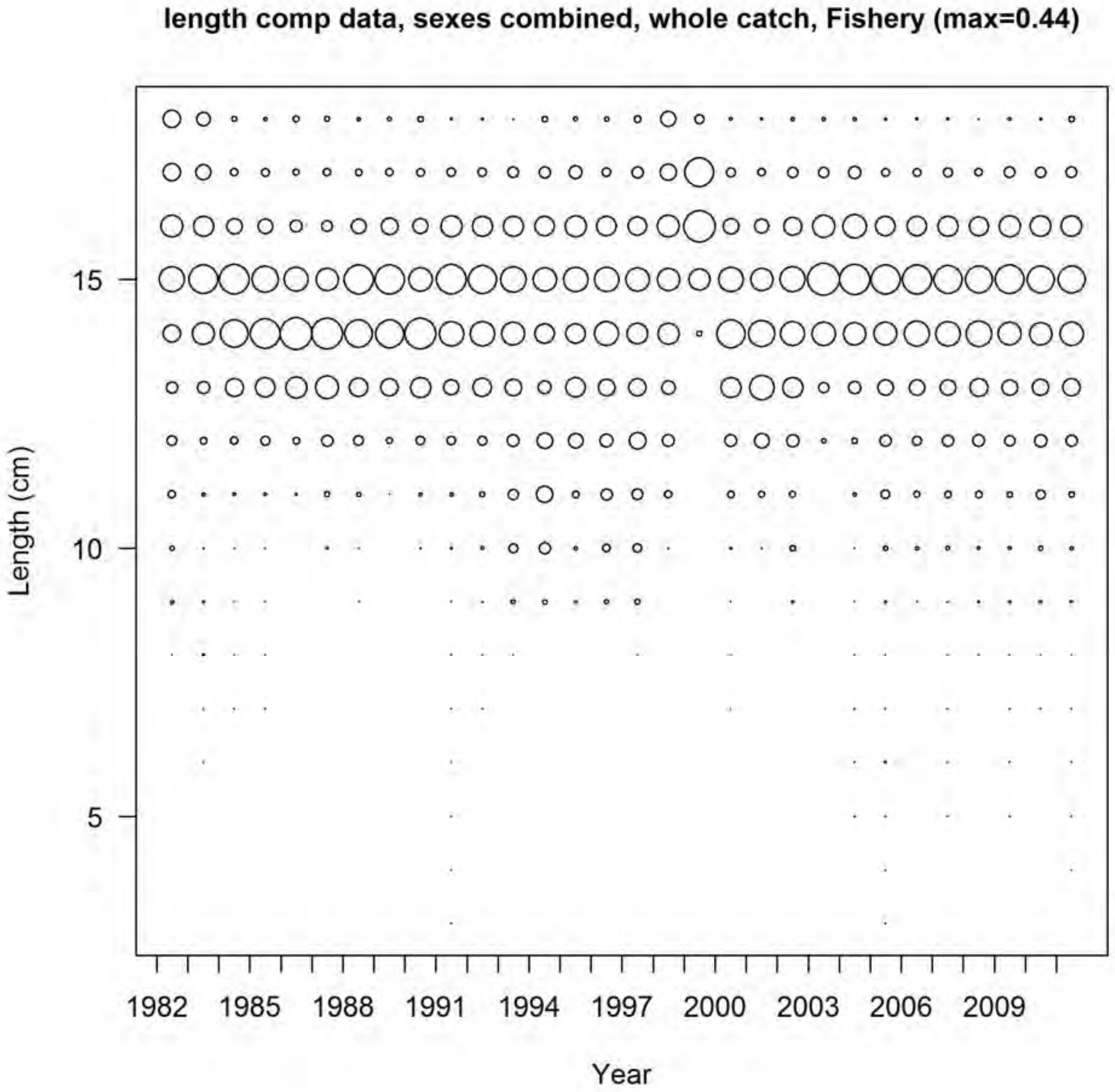

ghost age comp data, sexes combined, whole catch, SWAN (max=0.28)
Age (yr)
0
5
10
15
20
25
30
1982
1986
1989
1992
1999
2002
2005
2008
2011
Year

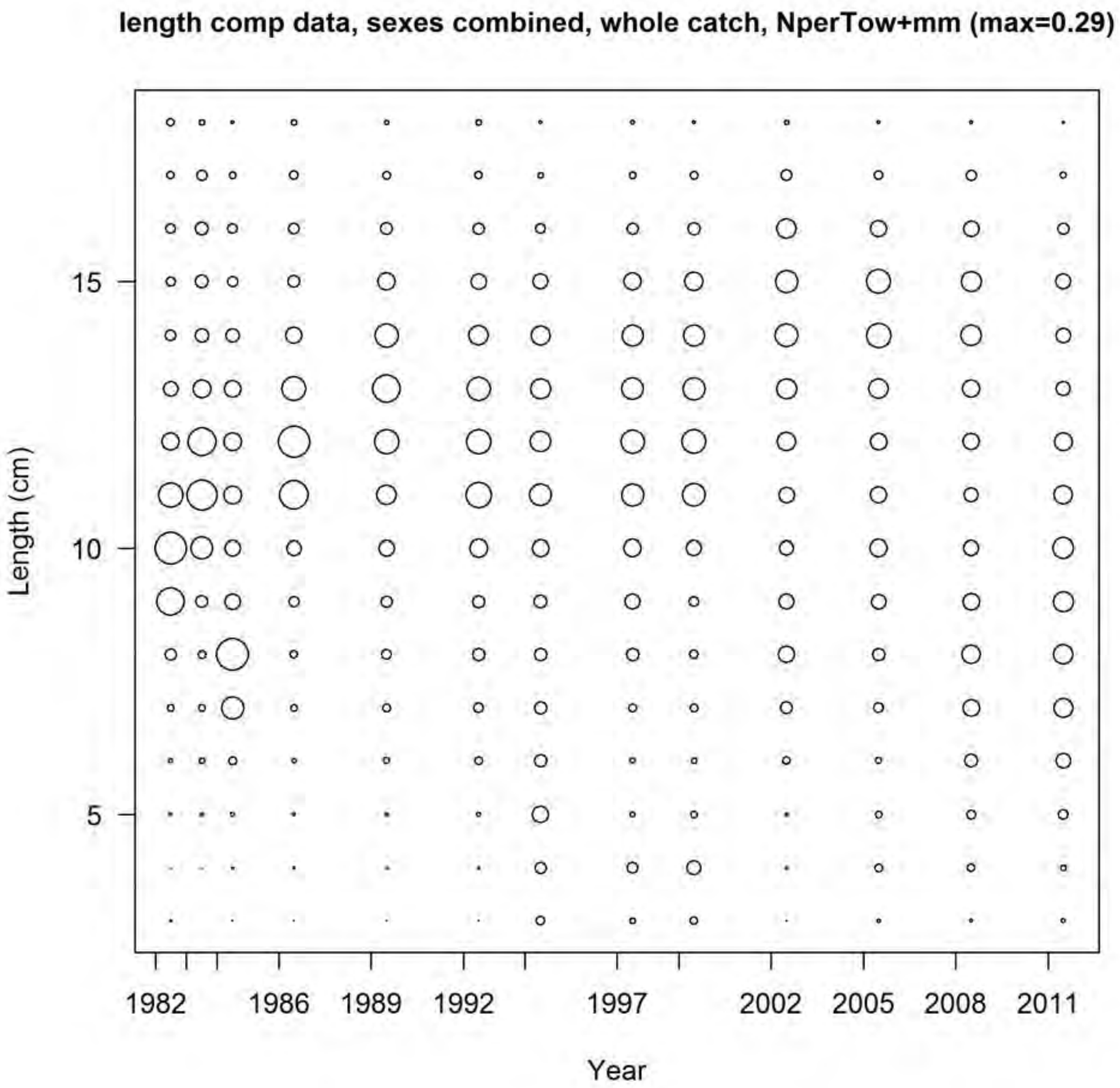
ghost age comp data, sexes combined, whole catch, SWAN

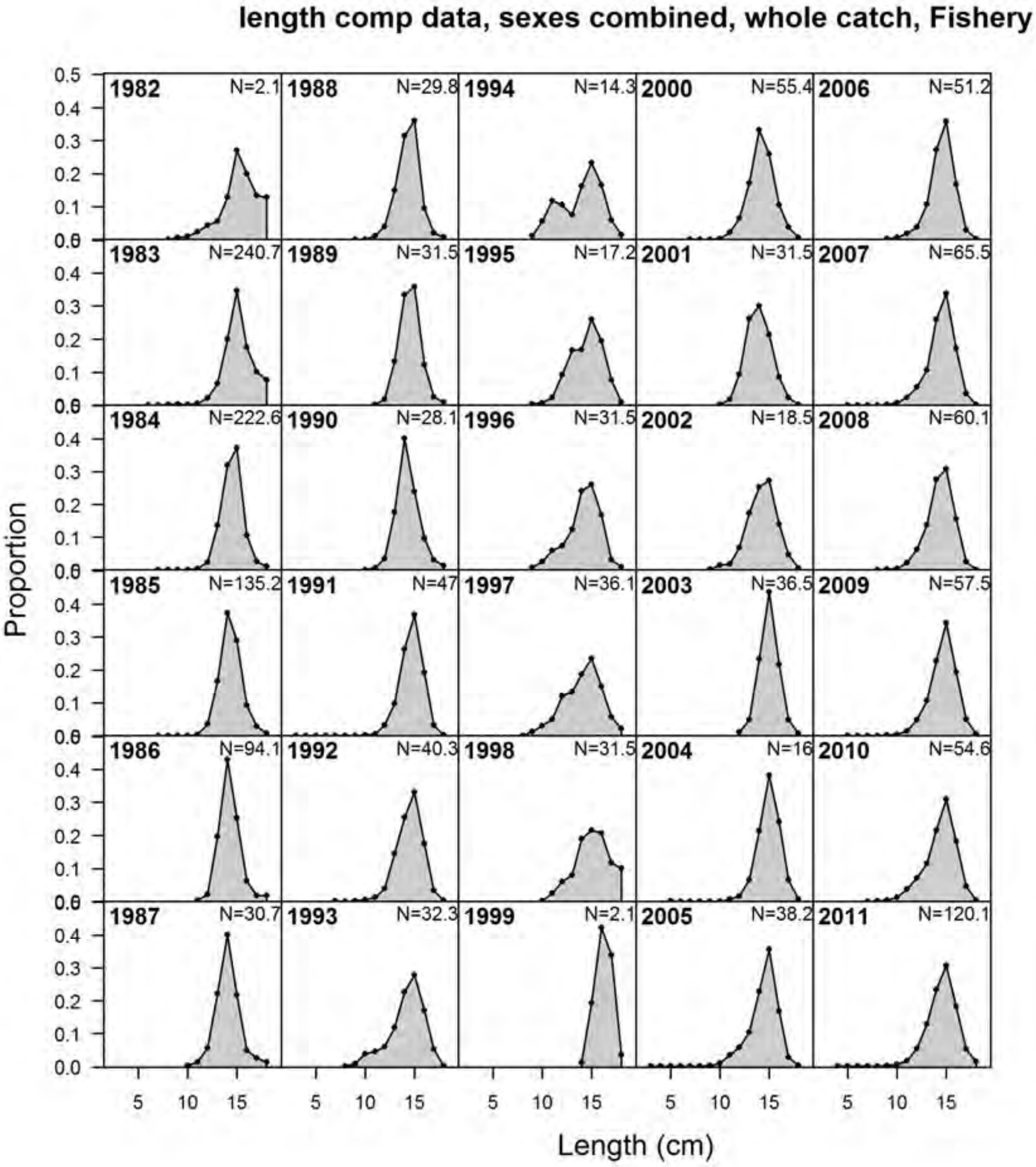


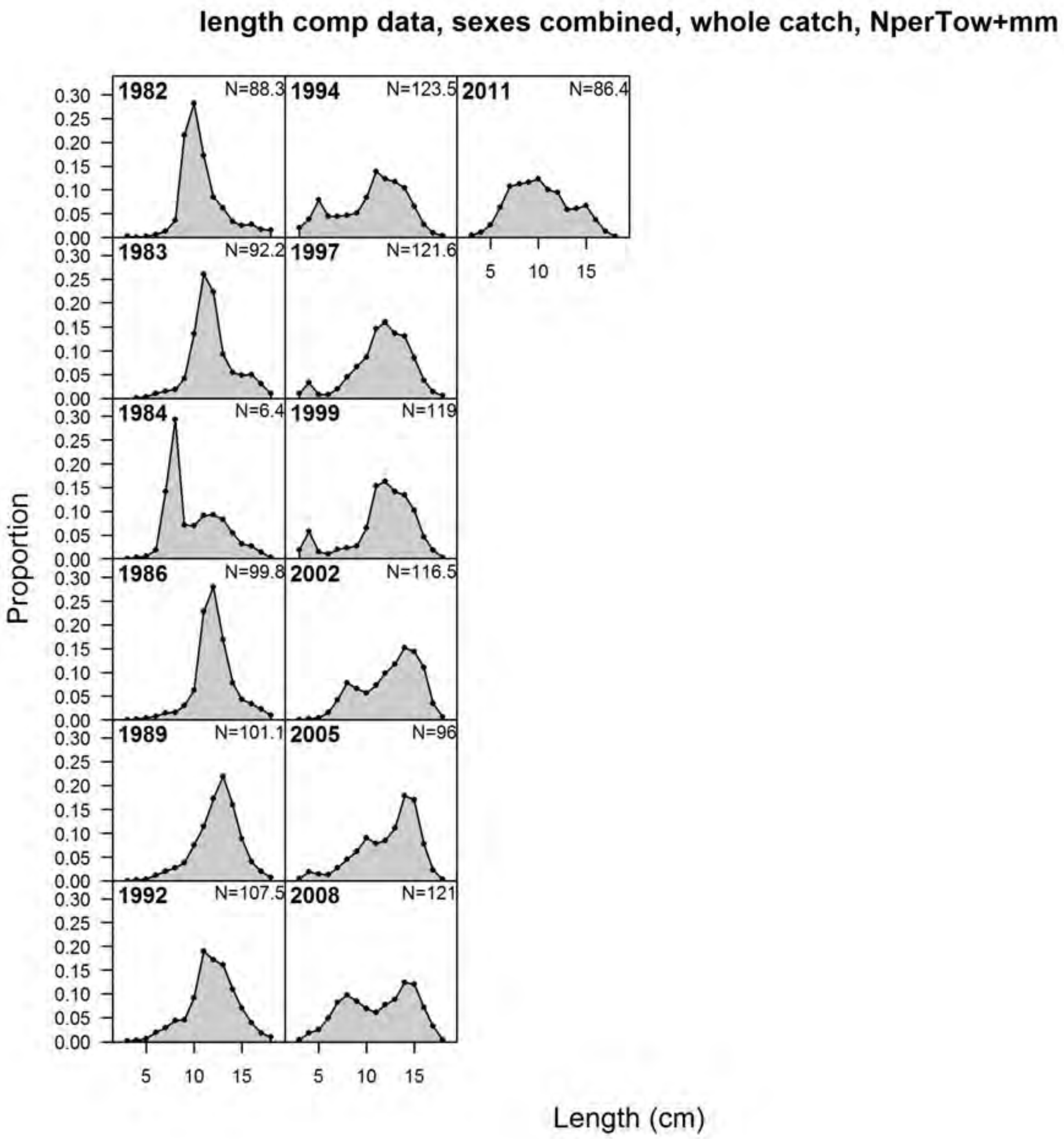

ghost age comps, sexes combined, whole catch, SWAN
1982
2002
1983
2005
1986
2008
1989
2011
1992
1999
Proportion
0.00
0.05
0.10
0.15
0.20
0.25
0.30
0
10
20
30
Age (yr)

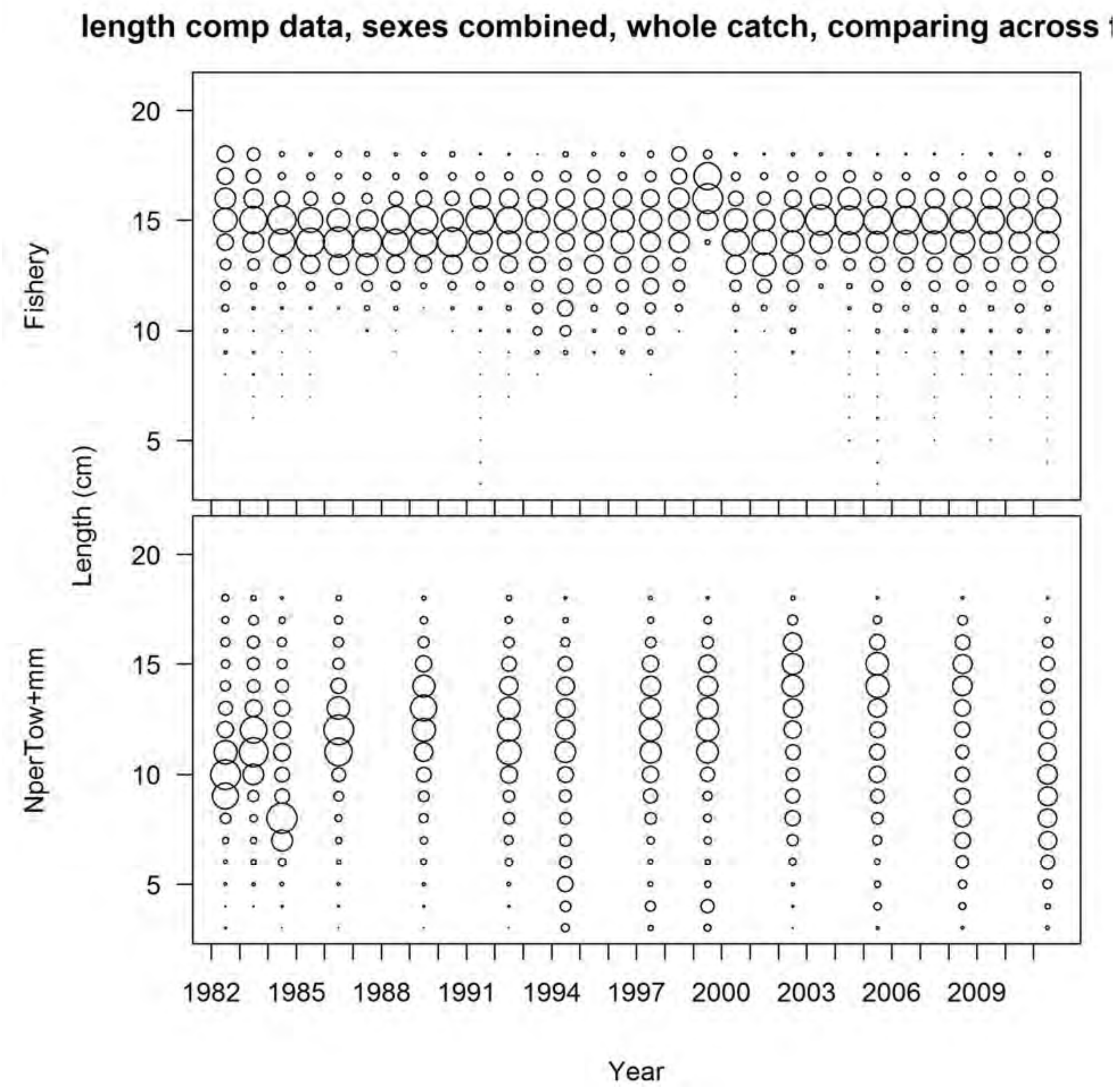

Pearson residuals, sexes combined, whole catch, SWAN (max=NA)
Age (yr)
30
25
20
15
10
5
0
1982
1986
1989
1992
1999
2002
2005
2008
2011
Year

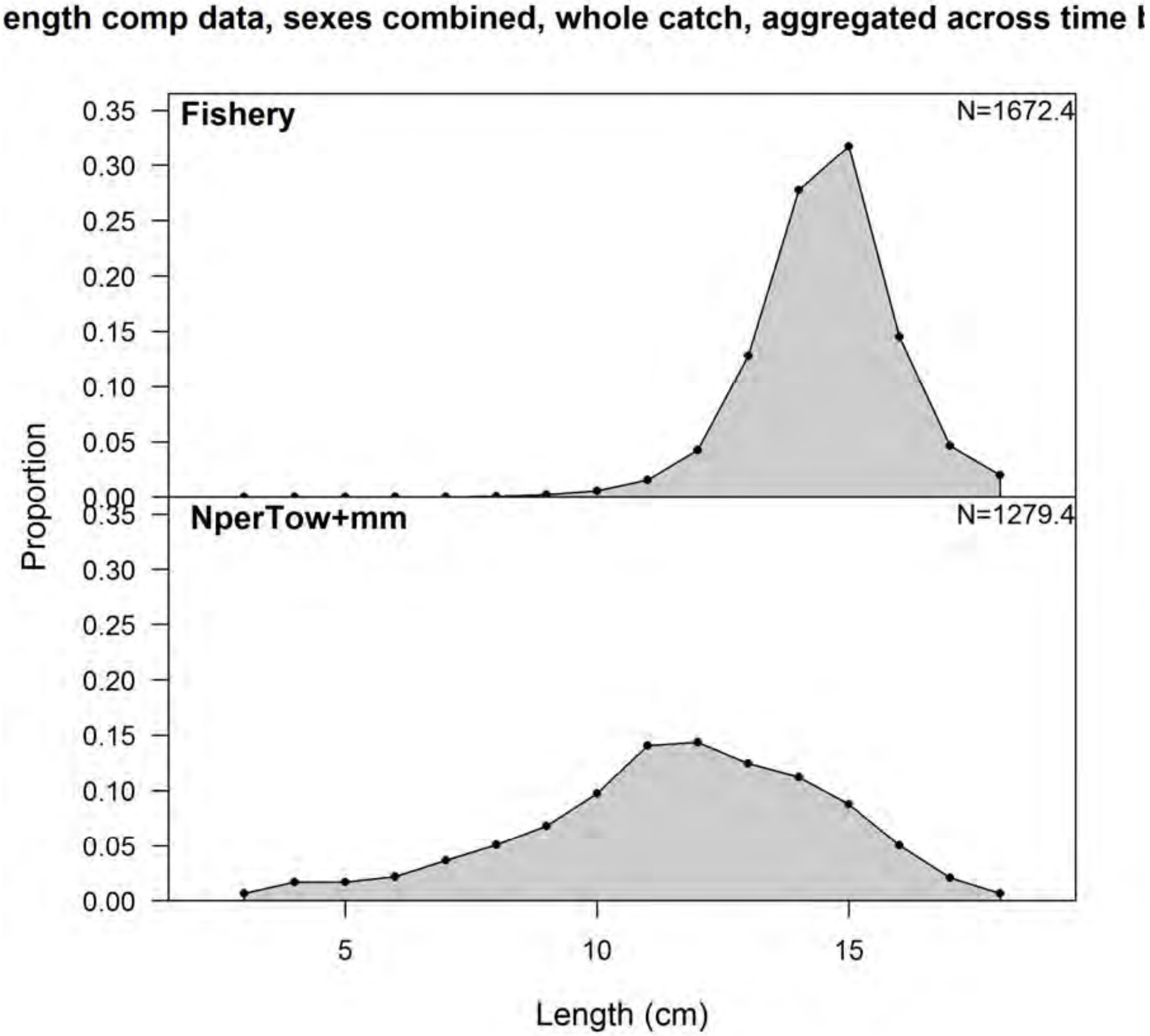

length comp data, sexes combined, whole catch, Fishery (max=0.44)
Length (cm)
15
10
5
1982
1985
1988
1991
1994
1997
2000
2003
2006
2009
Year

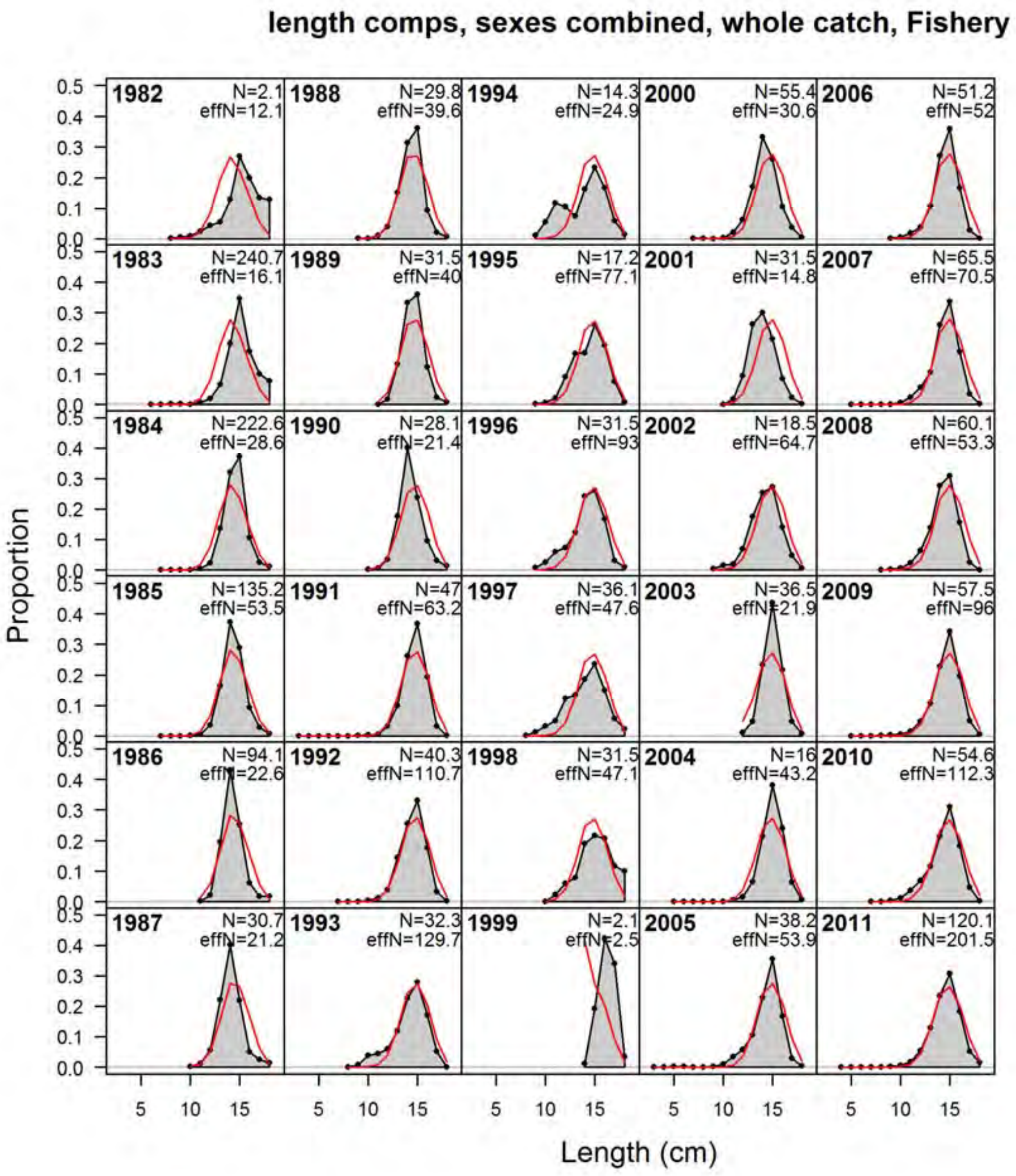

length comp data, sexes combined, whole catch, NperTow+mm (max=0.29)
Length (cm)
15
10
5
1982
1986
1989
1992
1997
2002
2005
2008
2011
Year

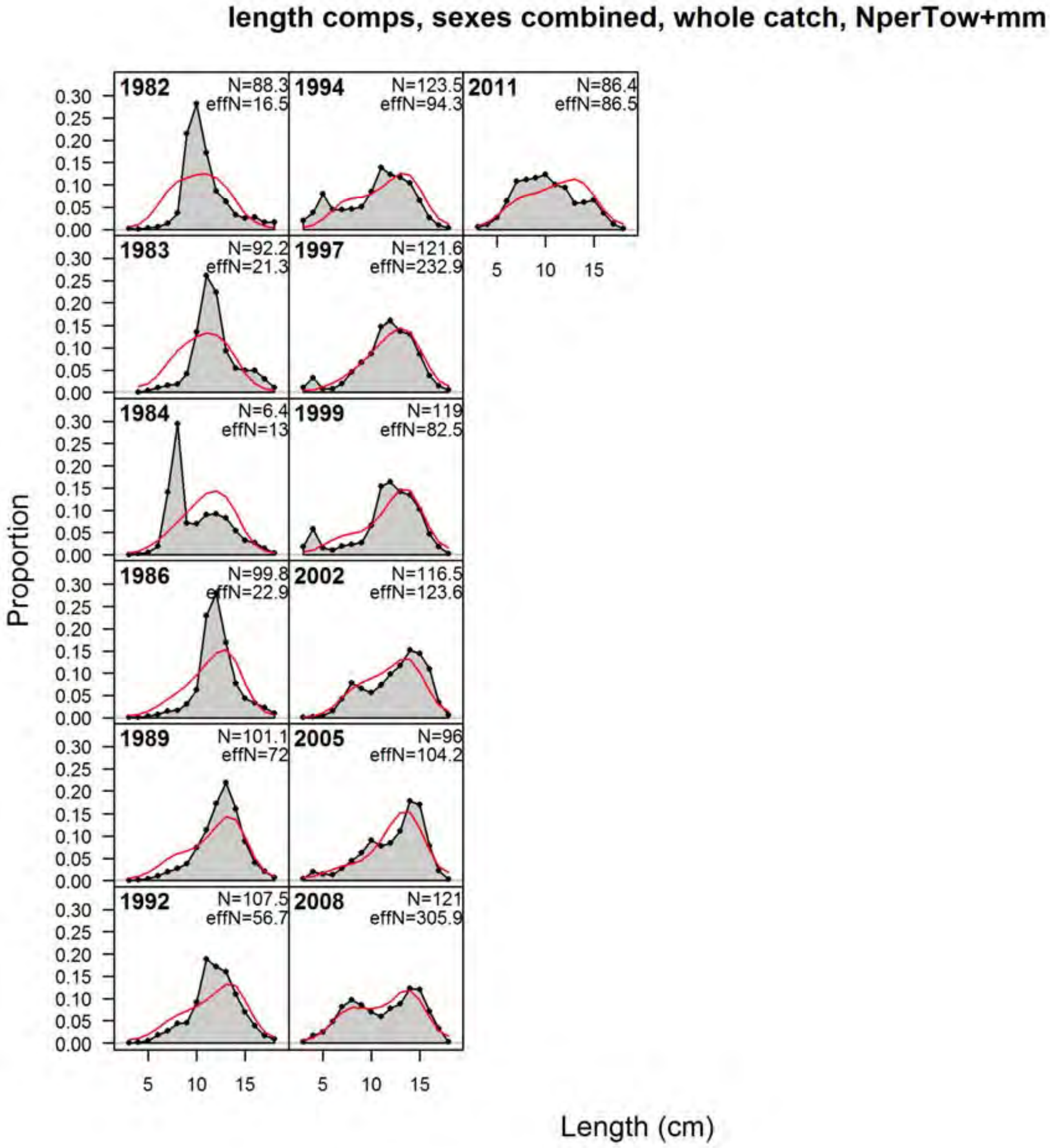

length comp data, sexes combined, whole catch, Fishery
Proportion
Length (cm)
1982 N=2.1
1988 N=29.8
1994 N=14.3
2000 N=55.4
2006 N=51.2
1983 N=240.7
1989 N=31.5
1995 N=17.2
2001 N=31.5
2007 N=65.5
1984 N=222.6
1990 N=28.1
1996 N=31.5
2002 N=18.5
2008 N=60.1
1985 N=135.2
1991 N=47
1997 N=36.1
2003 N=36.5
2009 N=57.5
1986 N=94.1
1992 N=40.3
1998 N=31.5
2004 N=16
2010 N=54.6
1987 N=30.7
1993 N=32.3
1999 N=2.1
2005 N=38.2
2011 N=120.1
0.0
0.1
0.2
0.3
0.4
0.5
5
10
15


length comp data, sexes combined, whole catch, NperTow+mm
1982 N=88.3
1994 N=123.5
2011 N=86.4
1983 N=92.2
1997 N=121.6
1984 N=6.4
1999 N=119
1986 N=99.8
2002 N=116.5
1989 N=101.1
2005 N=96
1992 N=107.5
2008 N=121
0.00
0.05
0.10
0.15
0.20
0.25
0.30
Proportion
5
10
15
Length (cm)

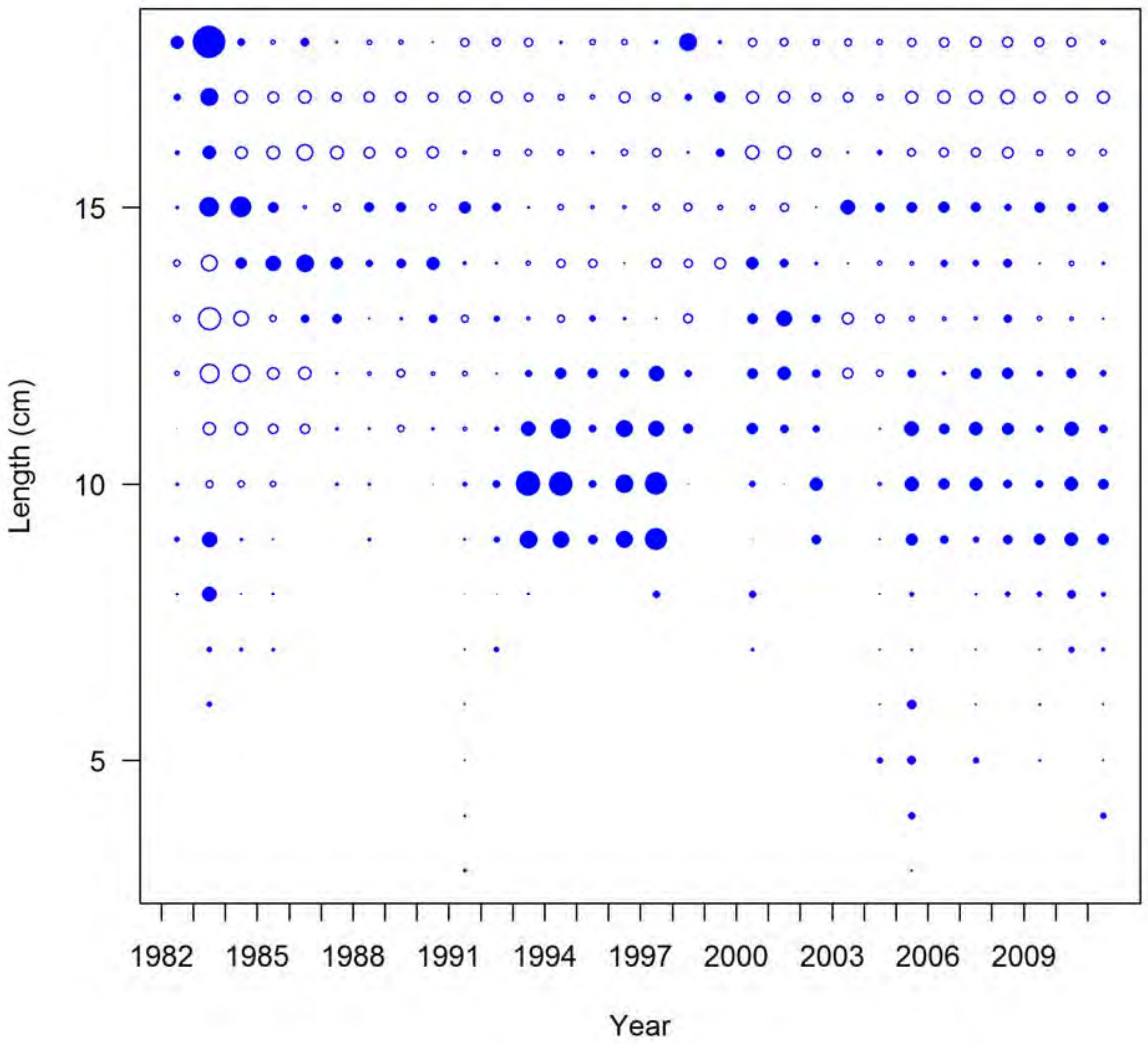

length comp data, sexes combined, whole catch, comparing across
Fishery
NperTow+mm
Length (cm)
20
15
10
5
1982
1985
1988
1991
1994
1997
2000
2003
2006
2009
Year

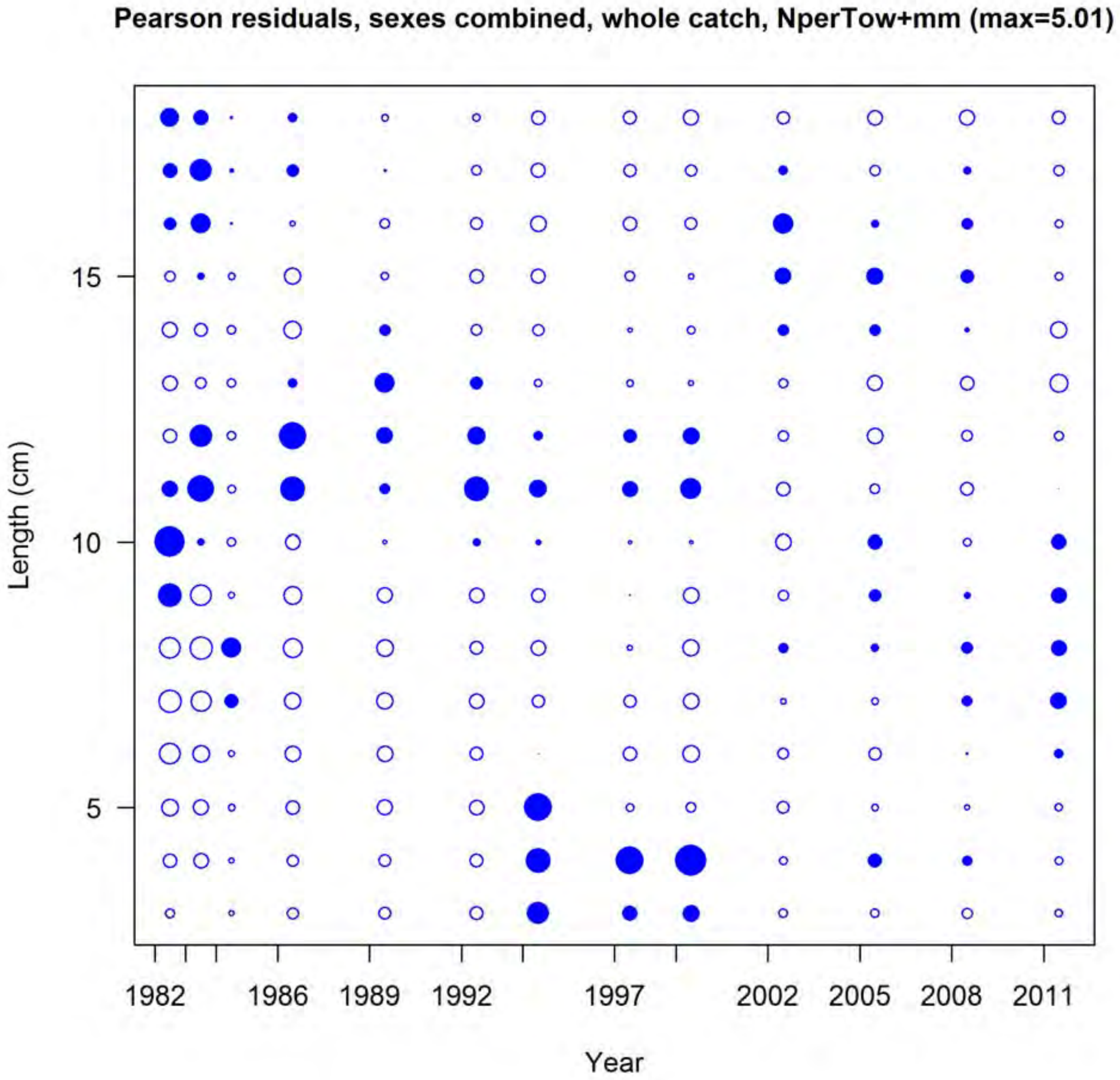

ength comp data, sexes combined, whole catch, aggregated across time l
Fishery
N=1672.4
NperTow+mm
N=1279.4
Proportion
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
5
10
15
Length (cm)

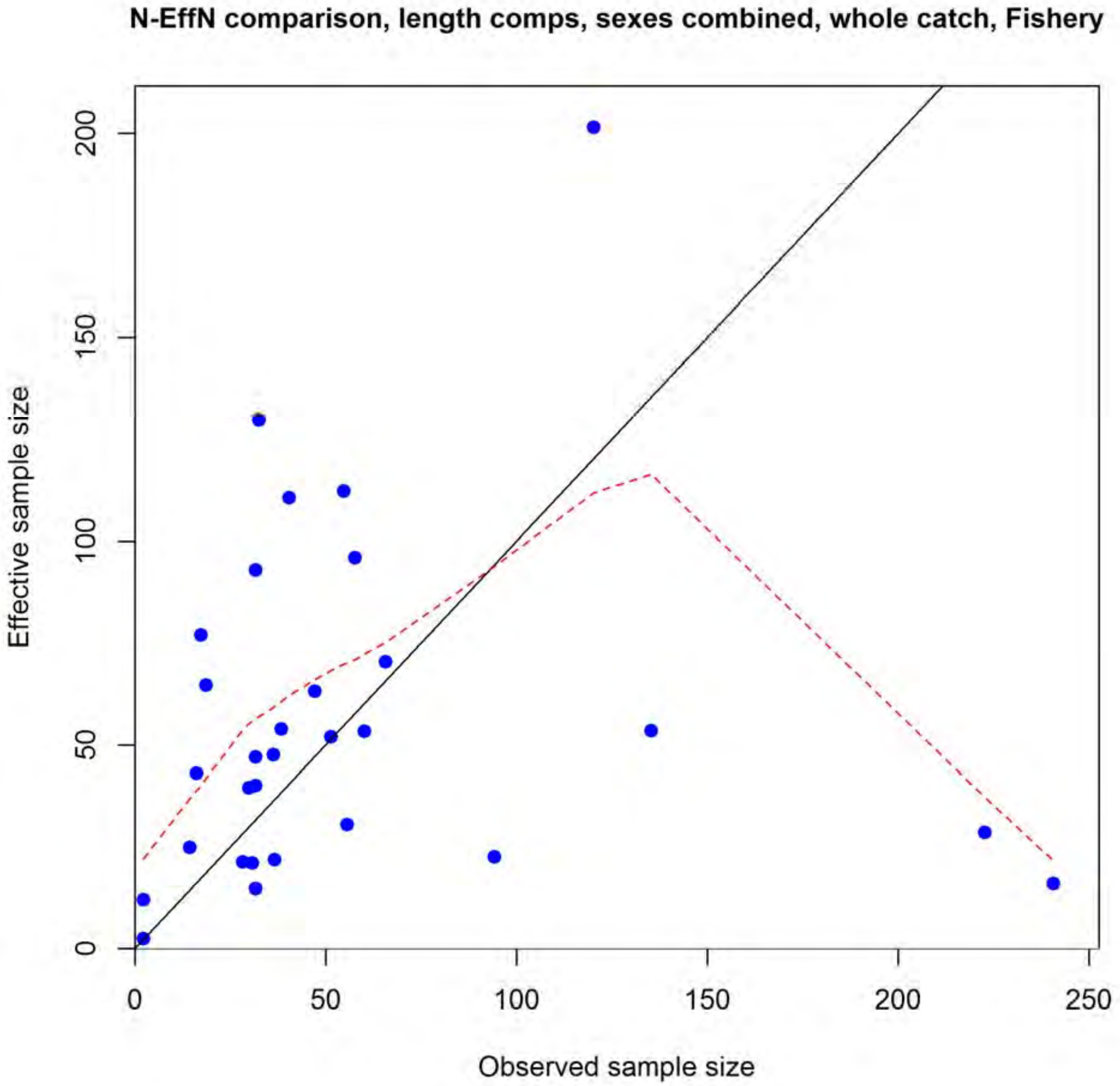

length comps, sexes combined, whole catch, Fishery
Proportion
Length (cm)
1982 N=2.1 effN=12.1
1983 N=240.7 effN=16.1
1984 N=222.6 effN=28.6
1985 N=135.2 effN=53.5
1986 N=94.1 effN=22.6
1987 N=30.7 effN=21.2
1988 N=29.8 effN=39.6
1989 N=31.5 effN=40
1990 N=28.1 effN=21.4
1991 N=47 effN=63.2
1992 N=40.3 effN=110.7
1993 N=32.3 effN=129.7
1994 N=14.3 effN=24.9
1995 N=17.2 effN=77.1
1996 N=31.5 effN=93
1997 N=36.1 effN=47.6
1998 N=31.5 effN=47.1
1999 N=2.1 effN=2.5
2000 N=55.4 effN=30.6
2001 N=31.5 effN=14.8
2002 N=18.5 effN=64.7
2003 N=36.5 effN=21.9
2004 N=16 effN=43.2
2005 N=38.2 effN=53.9
2006 N=51.2 effN=52
2007 N=65.5 effN=70.5
2008 N=60.1 effN=53.3
2009 N=57.5 effN=96
2010 N=54.6 effN=112.3
2011 N=120.1 effN=201.5
0.0
0.1
0.2
0.3
0.4
0.5
5
10
15


length comps, sexes combined, whole catch, NperTow+mm
1982 N=88.3 effN=16.5
1994 N=123.5 effN=94.3
2011 N=86.4 effN=86.5
1983 N=92.2 effN=21.3
1997 N=121.6 effN=232.9
1984 N=6.4 effN=13
1999 N=119 effN=82.5
1986 N=99.8 effN=22.9
2002 N=116.5 effN=123.6
1989 N=101.1 effN=72
2005 N=96 effN=104.2
1992 N=107.5 effN=56.7
2008 N=121 effN=305.9
Proportion
0.00
0.05
0.10
0.15
0.20
0.25
0.30
5
10
15
Length (cm)

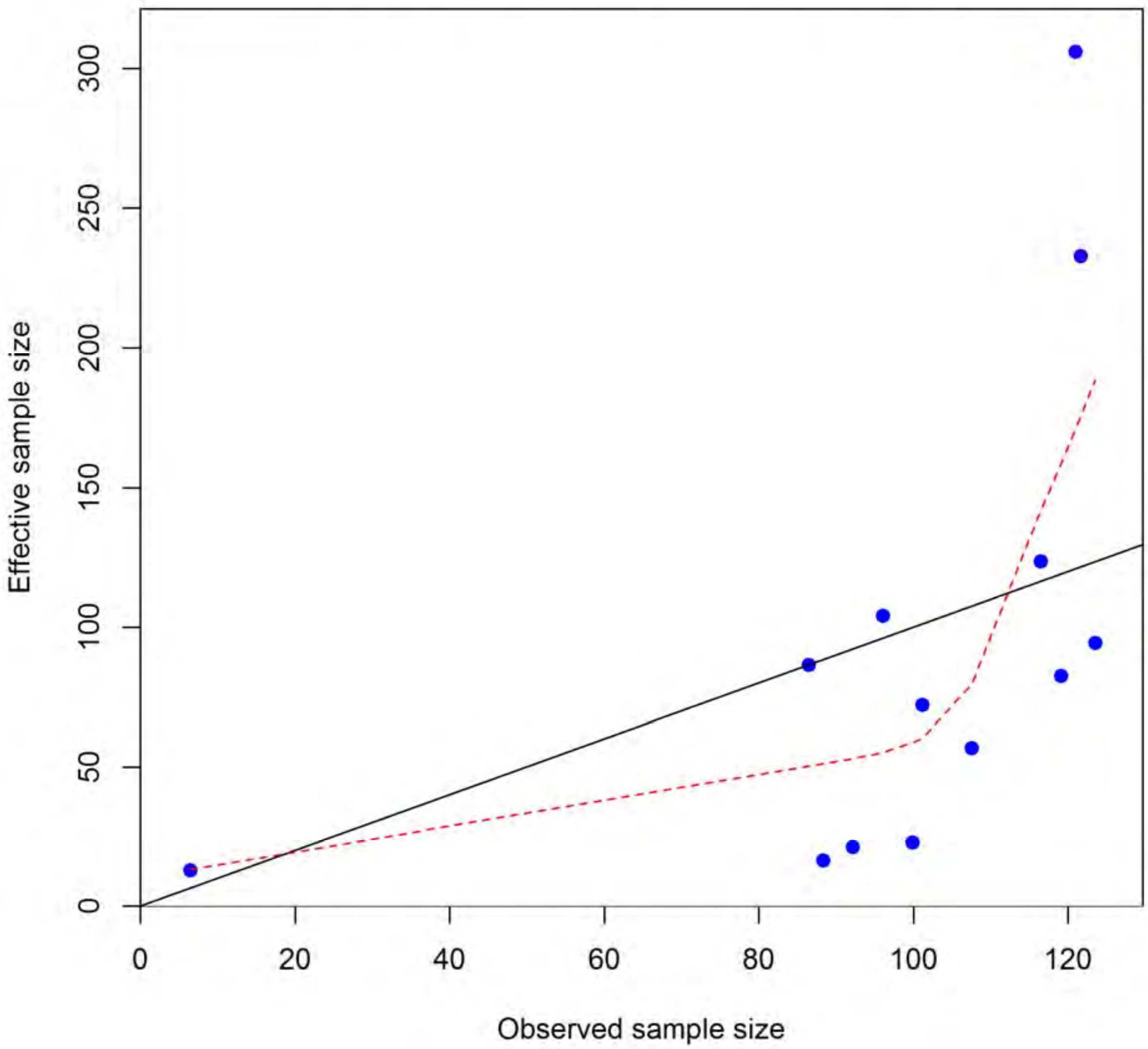
Pearson residuals, sexes combined, whole catch, Fishery (max=11.27)

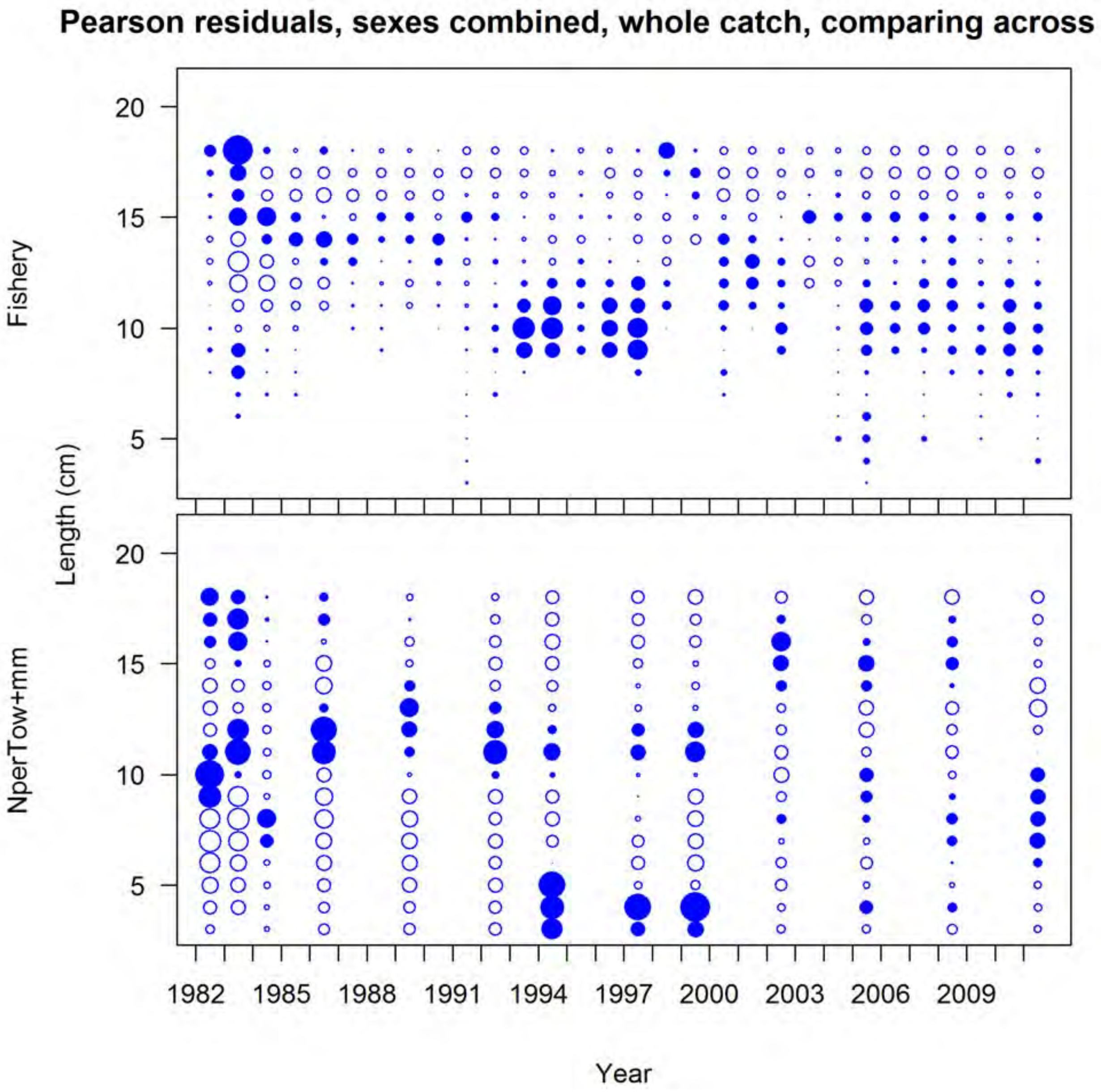



Pearson residuals, sexes combined, whole catch, NperTow+mm (max=5.01)
Length (cm)
15
10
5
1982
1986
1989
1992
1997
2002
2005
2008
2011
Year

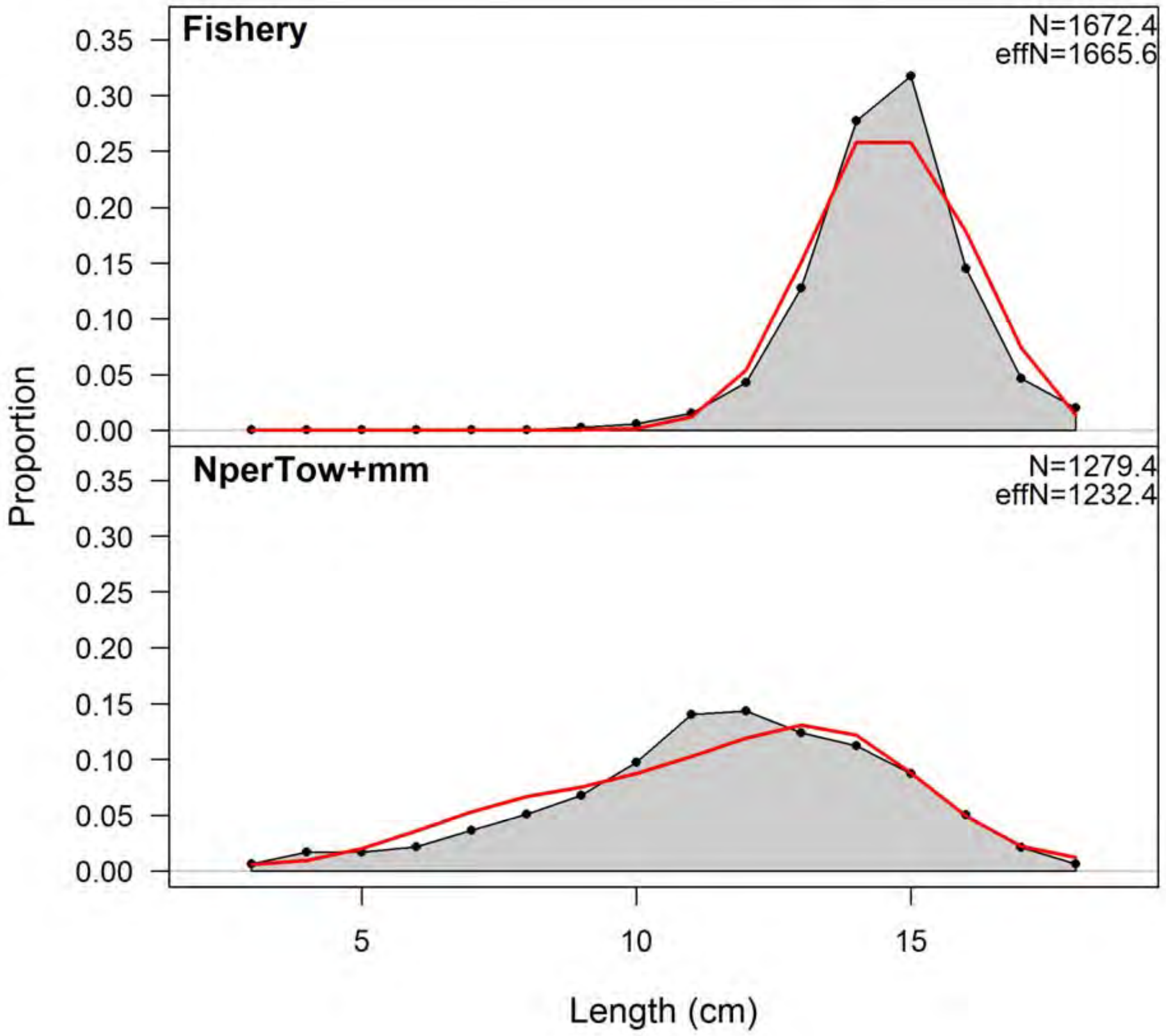

N-EffN comparison, length comps, sexes combined, whole catch, Fishery
Effective sample size
0
50
100
150
200
Observed sample size
0
50
100
150
200
250

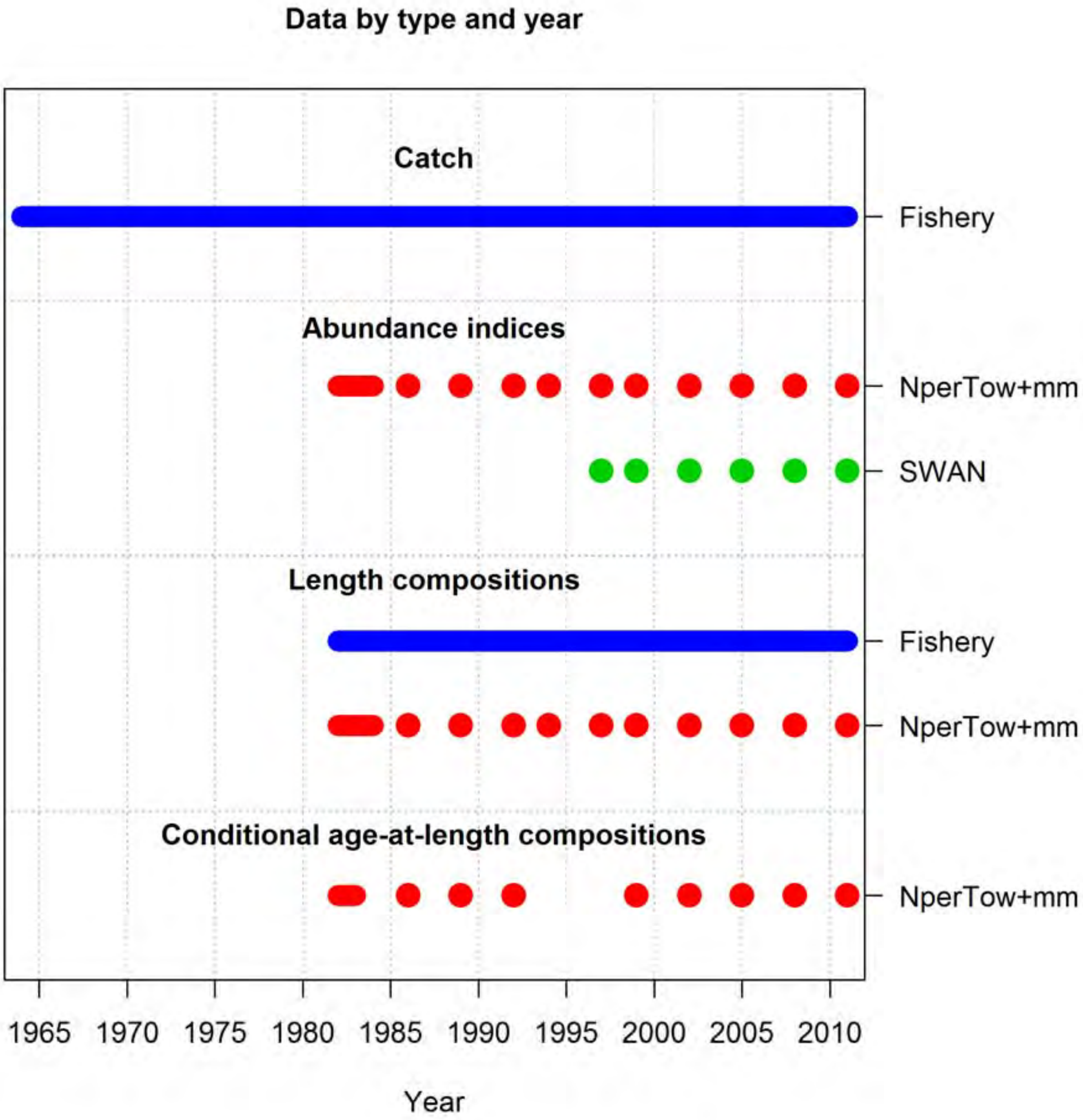
N-EffN comparison, length comps, sexes combined, whole catch, NperTow+mm

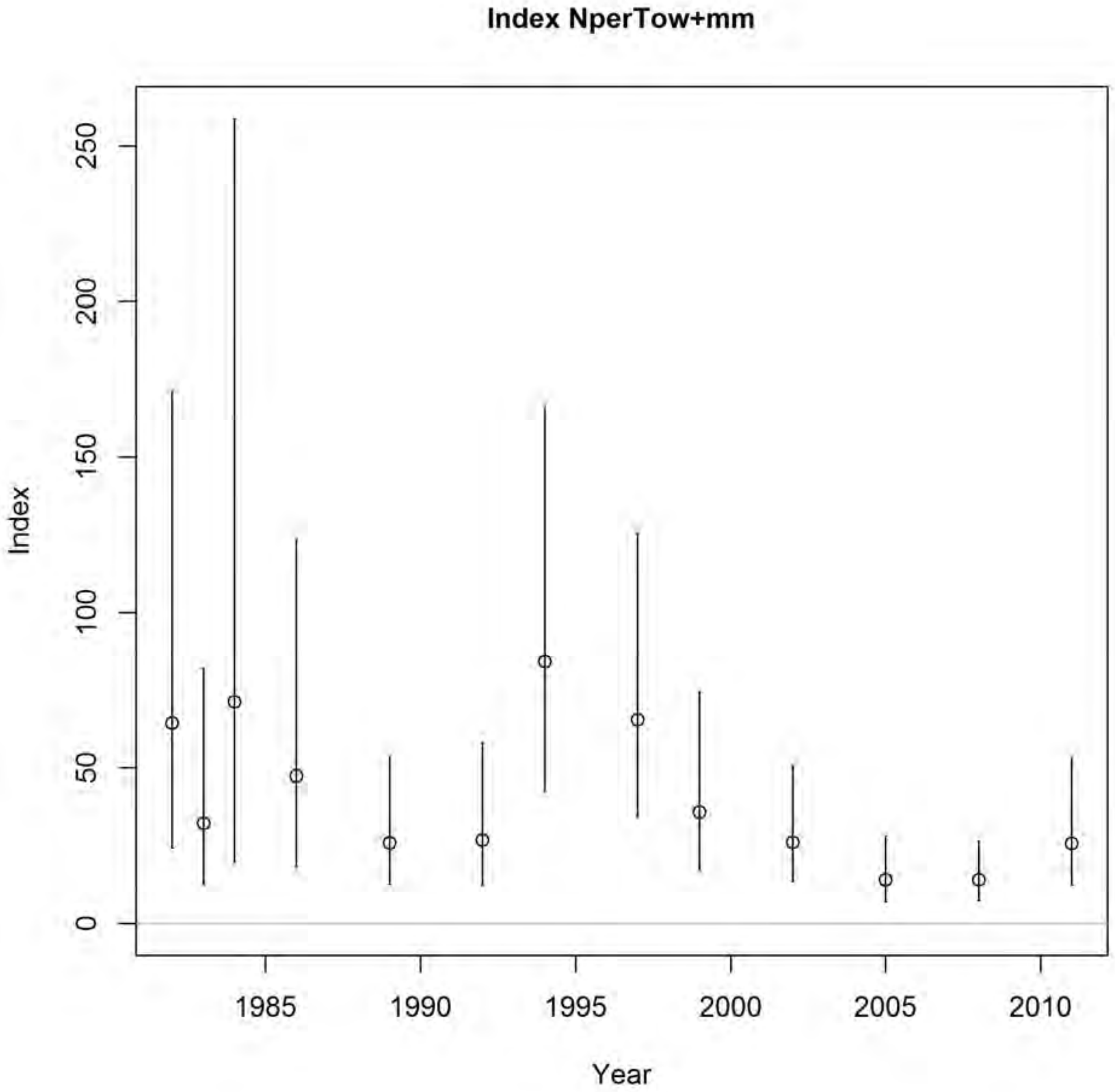


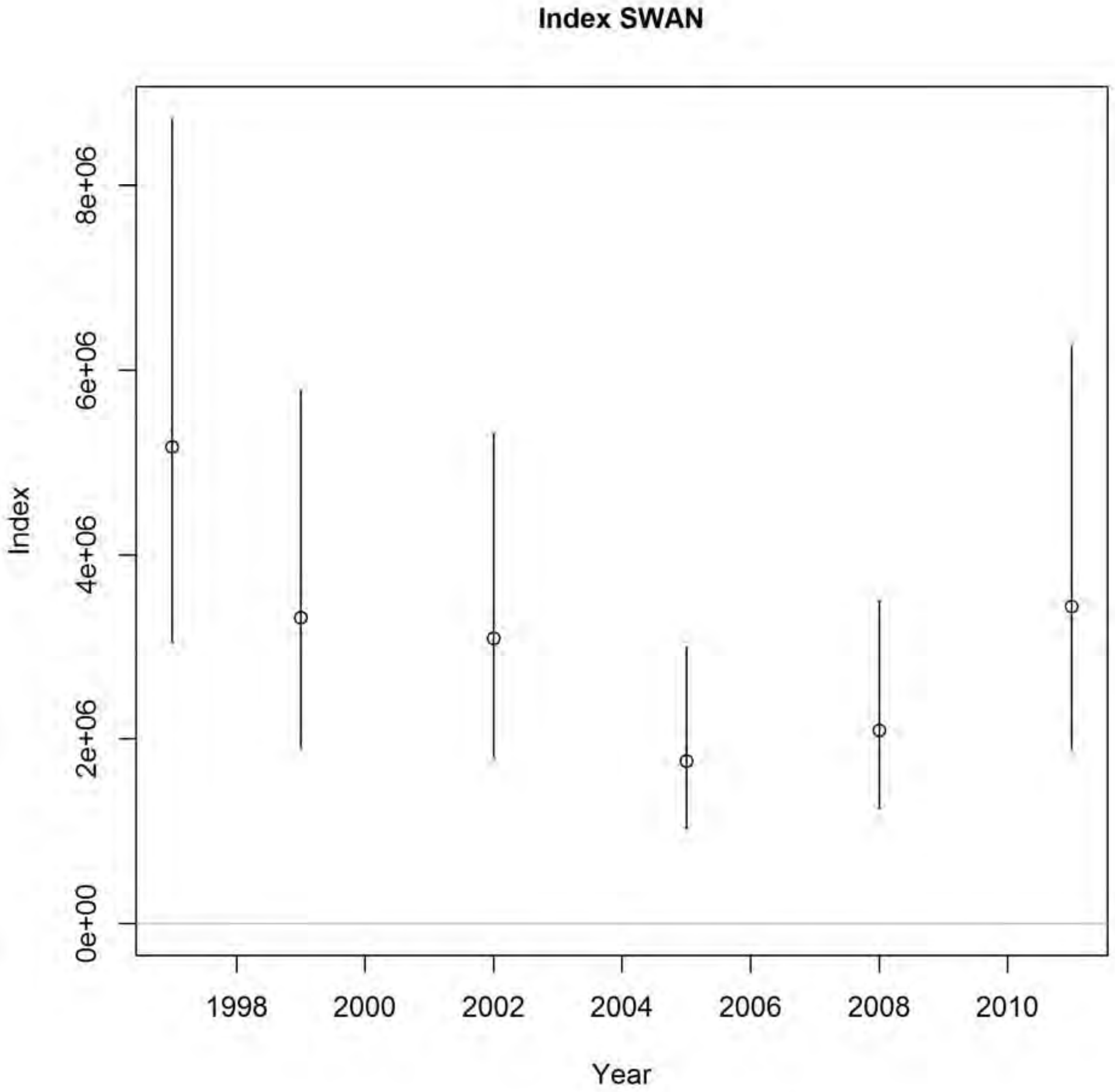

Pearson residuals, sexes combined, whole catch, comparing across
Fishery
NperTow+mm
Length (cm)
20
15
10
5
1982
1985
1988
1991
1994
1997
2000
2003
2006
2009
Year

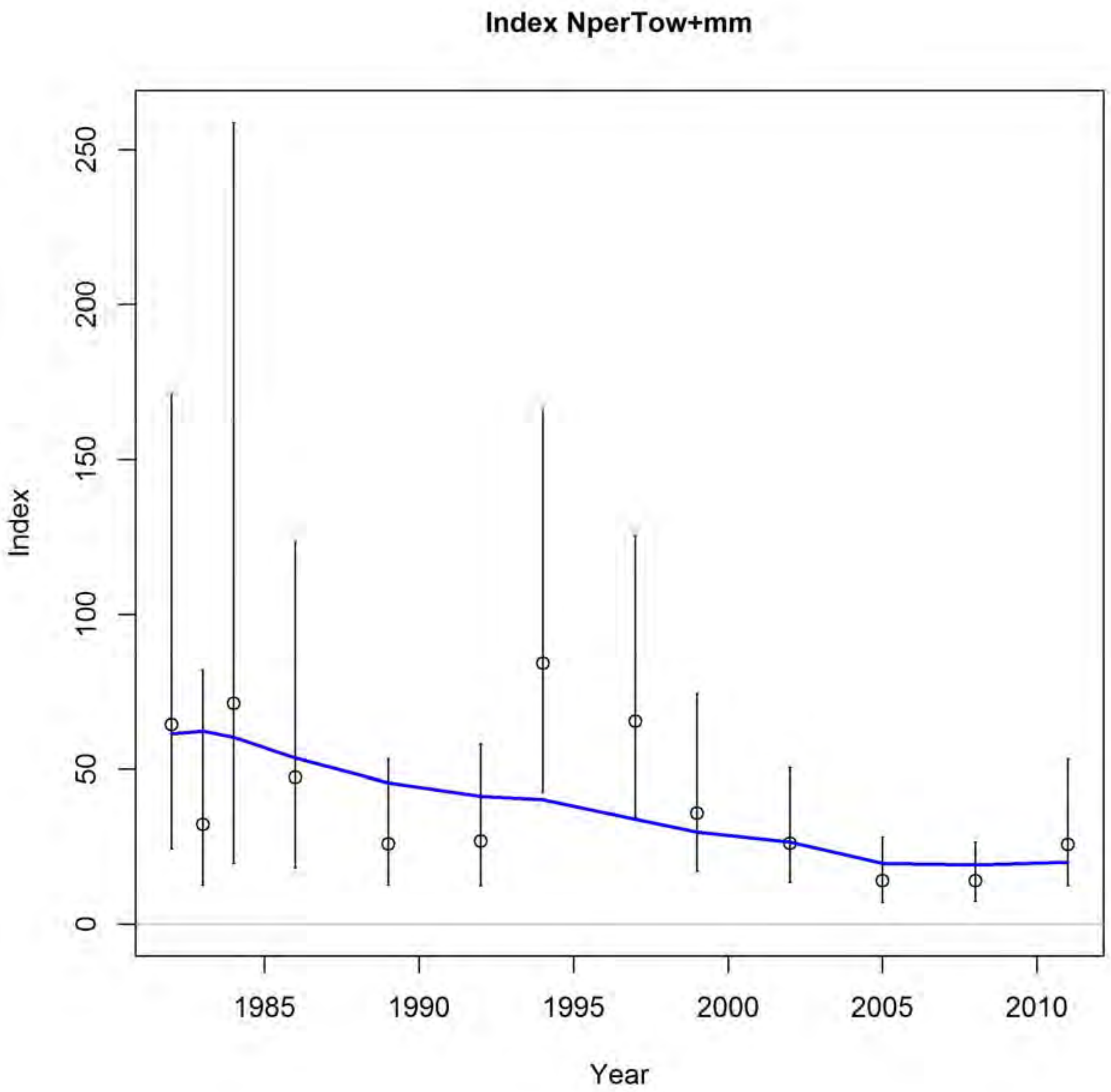
length comps, sexes combined, whole catch, aggregated across time by

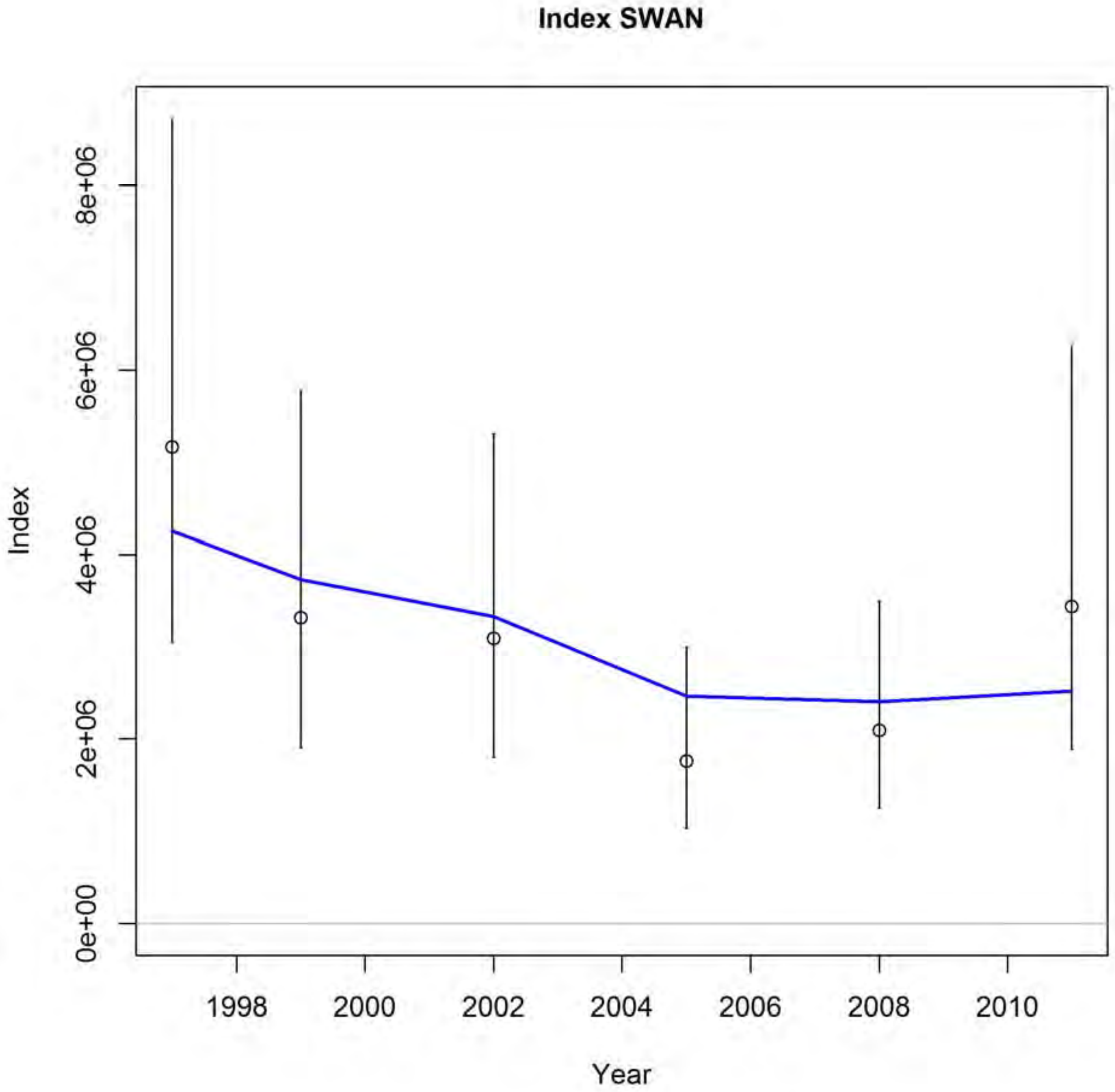


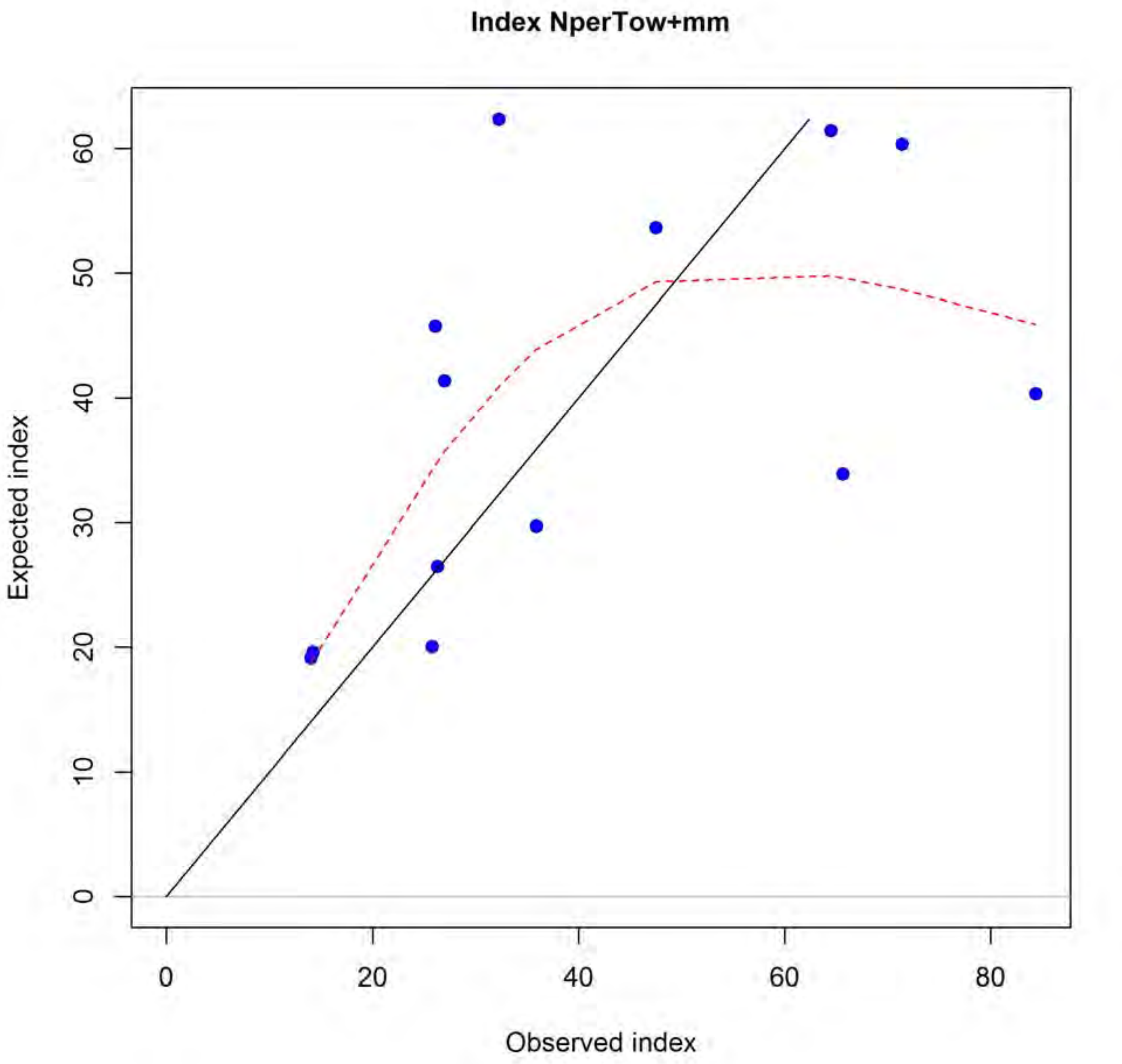

Data by type and year
Catch
Fishery
Abundance indices
NperTow+mm
SWAN
Length compositions
Fishery
NperTow+mm
Conditional age-at-length compositions
NperTow+mm
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
Year

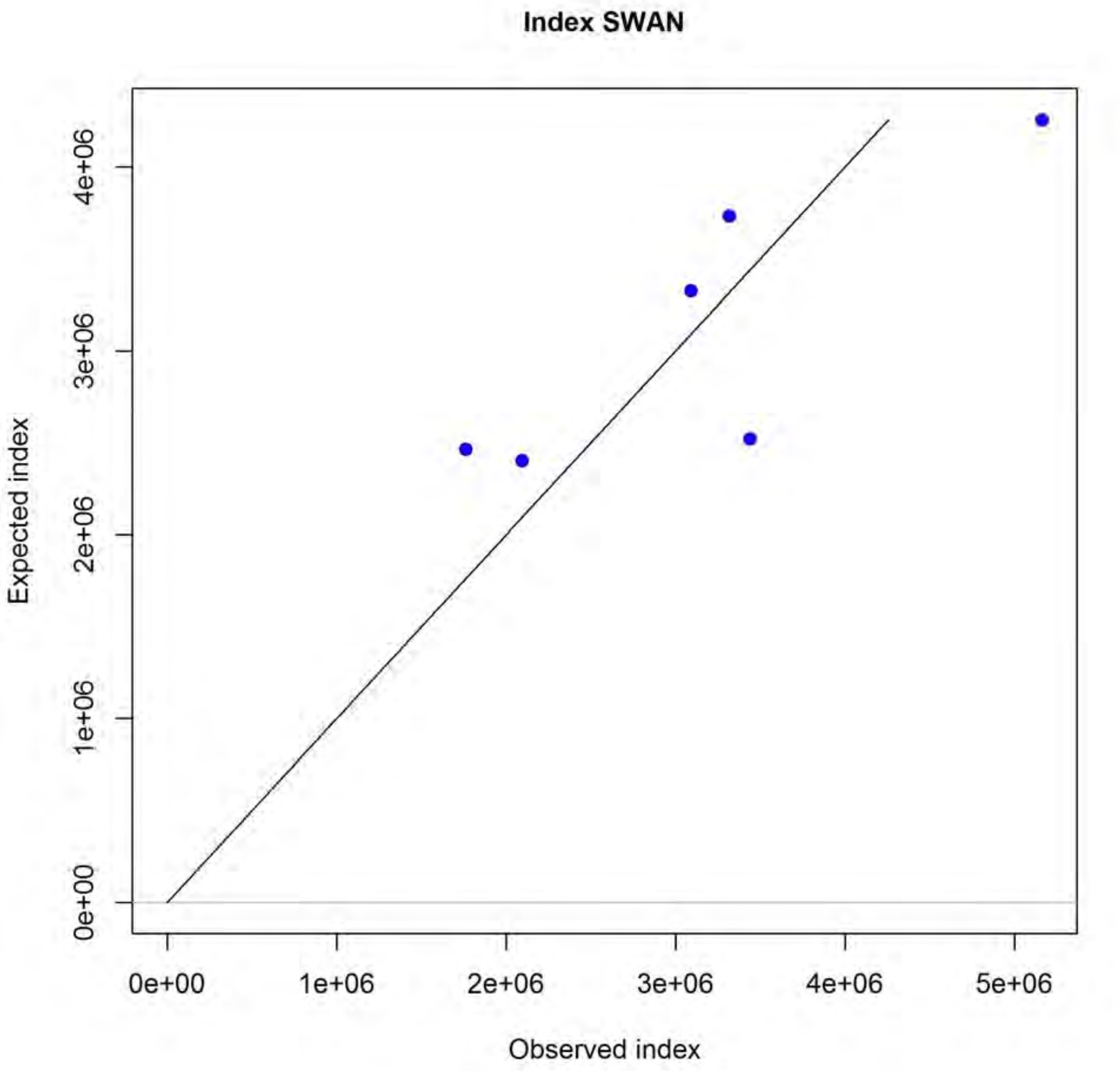

Index NperTow+mm
Index
0
50
100
150
200
250
1985
1990
1995
2000
2005
2010
Year

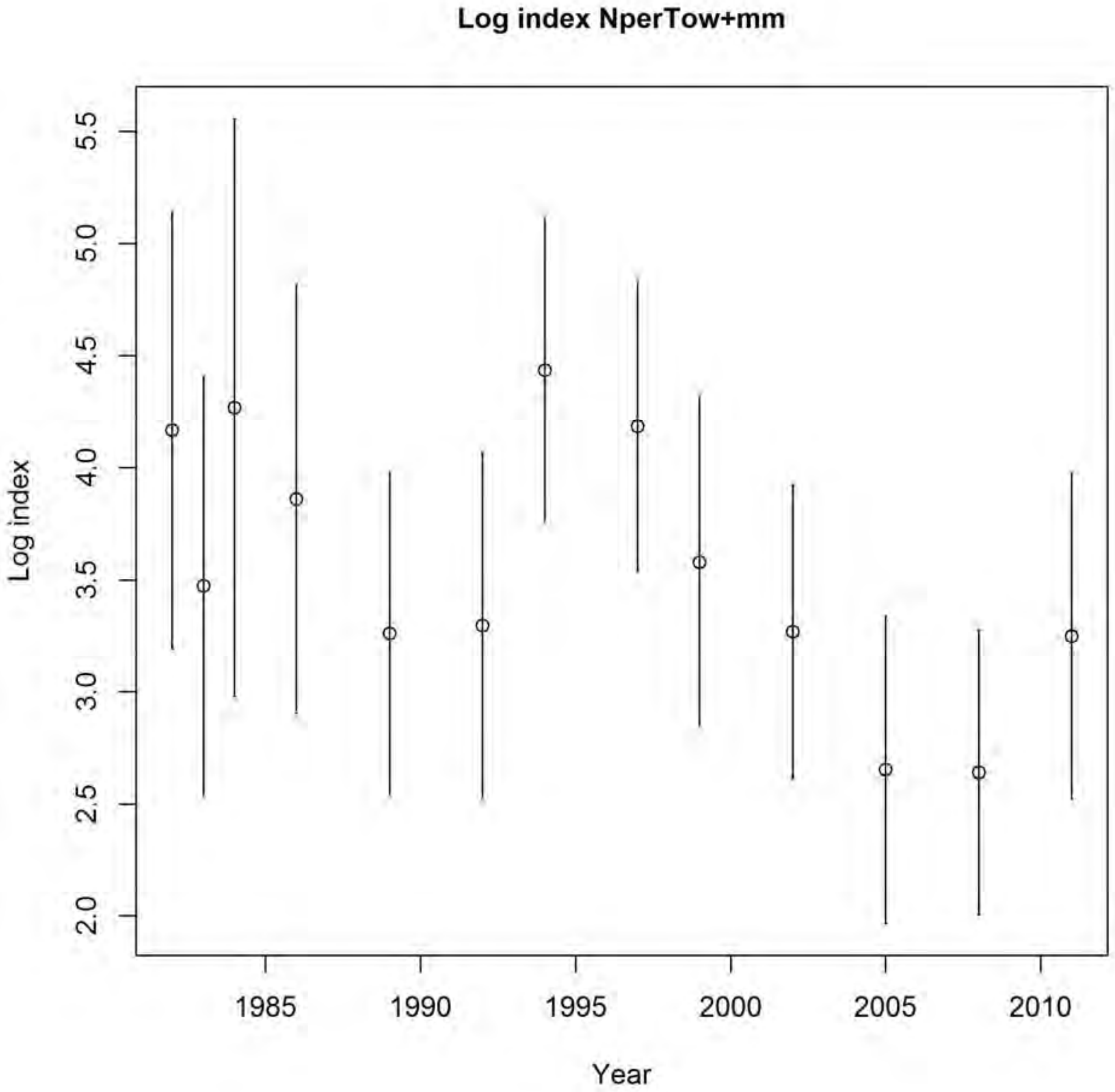

Index SWAN
Index
0e+00
2e+06
4e+06
6e+06
8e+06
1998
2000
2002
2004
2006
2008
2010
Year


Index NperTow+mm
Index
0
50
100
150
200
250
1985
1990
1995
2000
2005
2010
Year

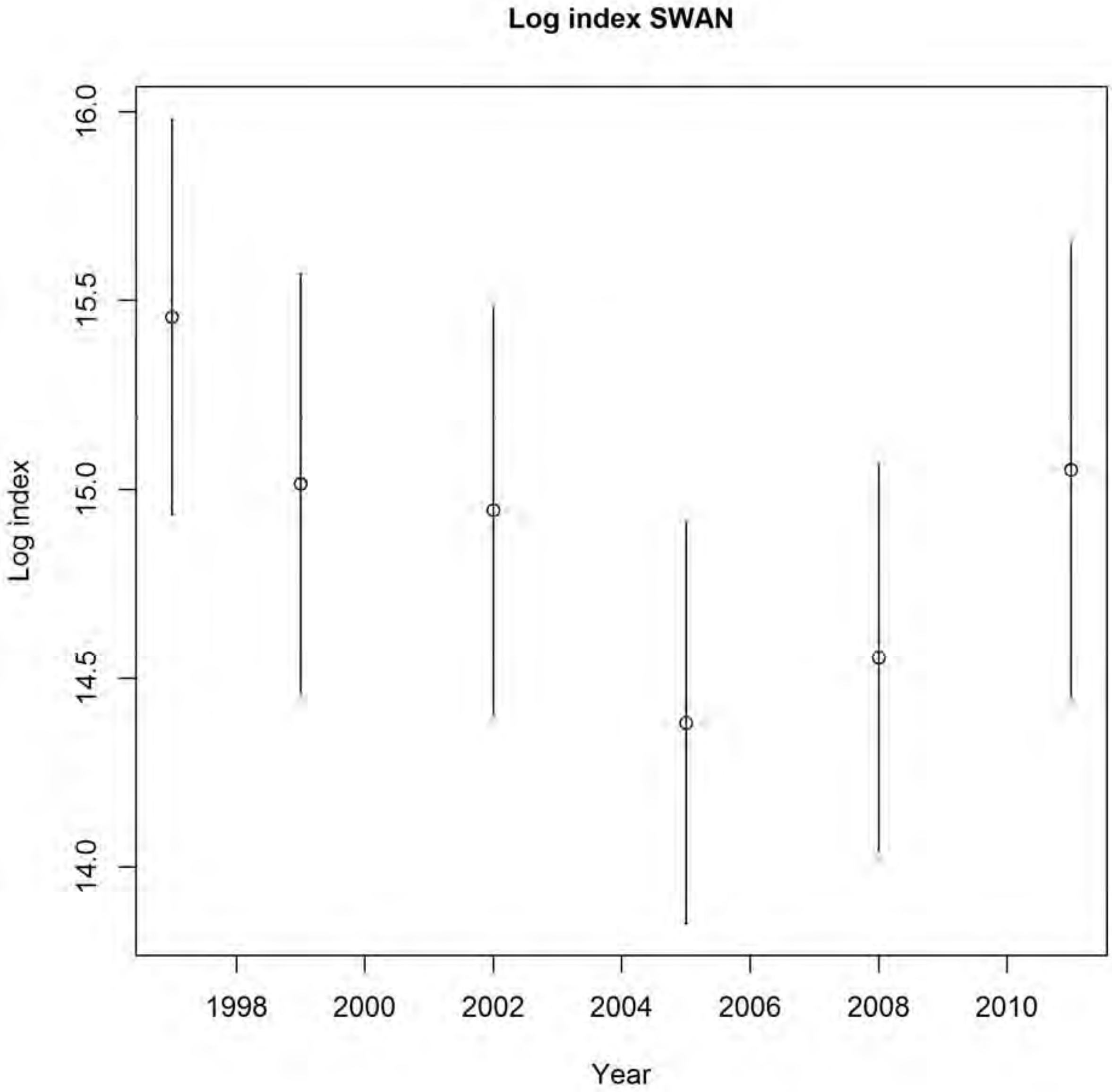

Index SWAN
Index
0e+00
2e+06
4e+06
6e+06
8e+06
1998
2000
2002
2004
2006
2008
2010
Year

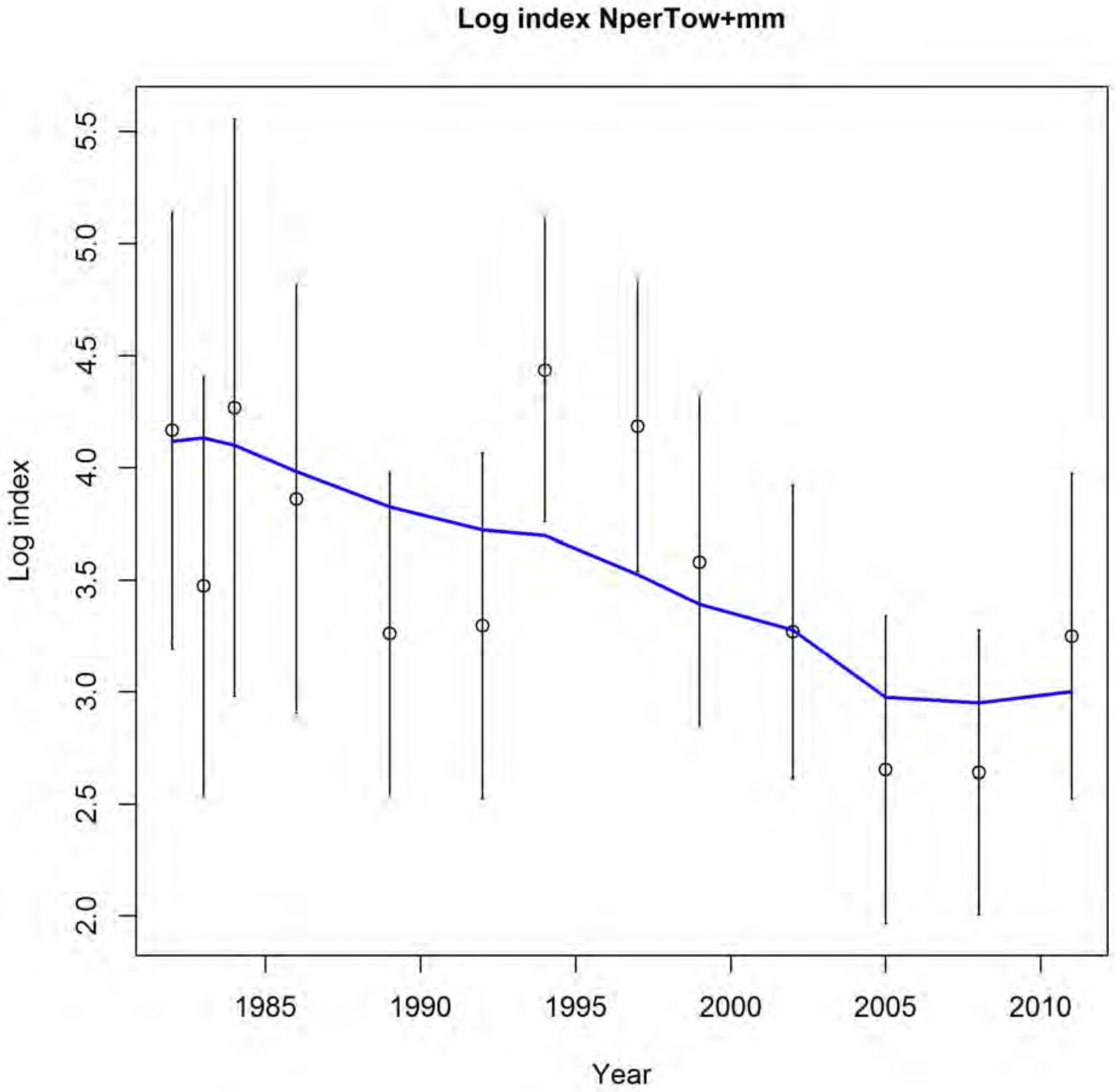

Index NperTow+mm
Expected index
0
10
20
30
40
50
60
0
20
40
60
80
Observed index

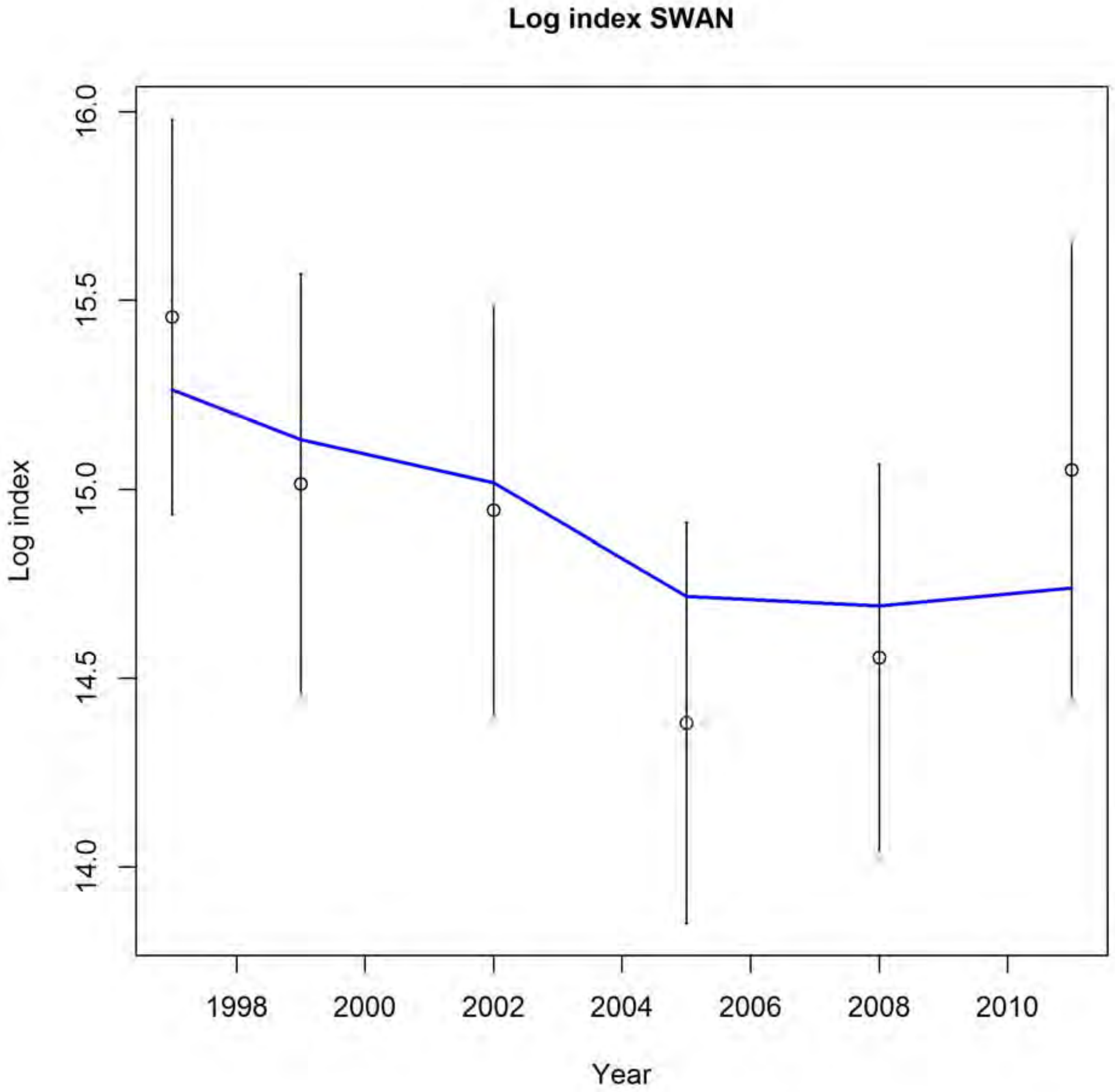

Index SWAN
Expected index
Observed index
0e+00
1e+06
2e+06
3e+06
4e+06
5e+06



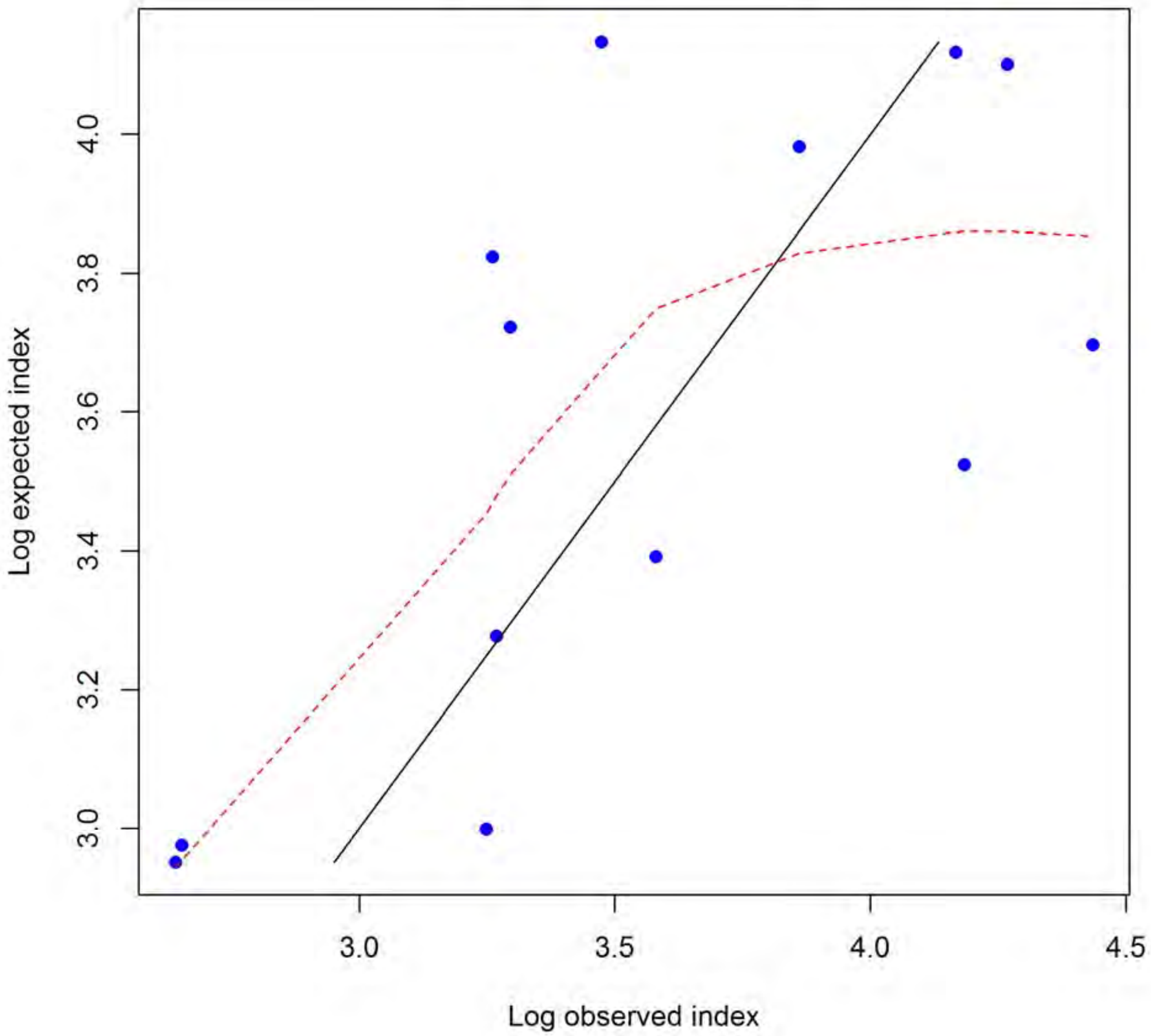
Log index NperTow+mm
Log index
5.5
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1985
1990
1995
2000
2005
2010
Year

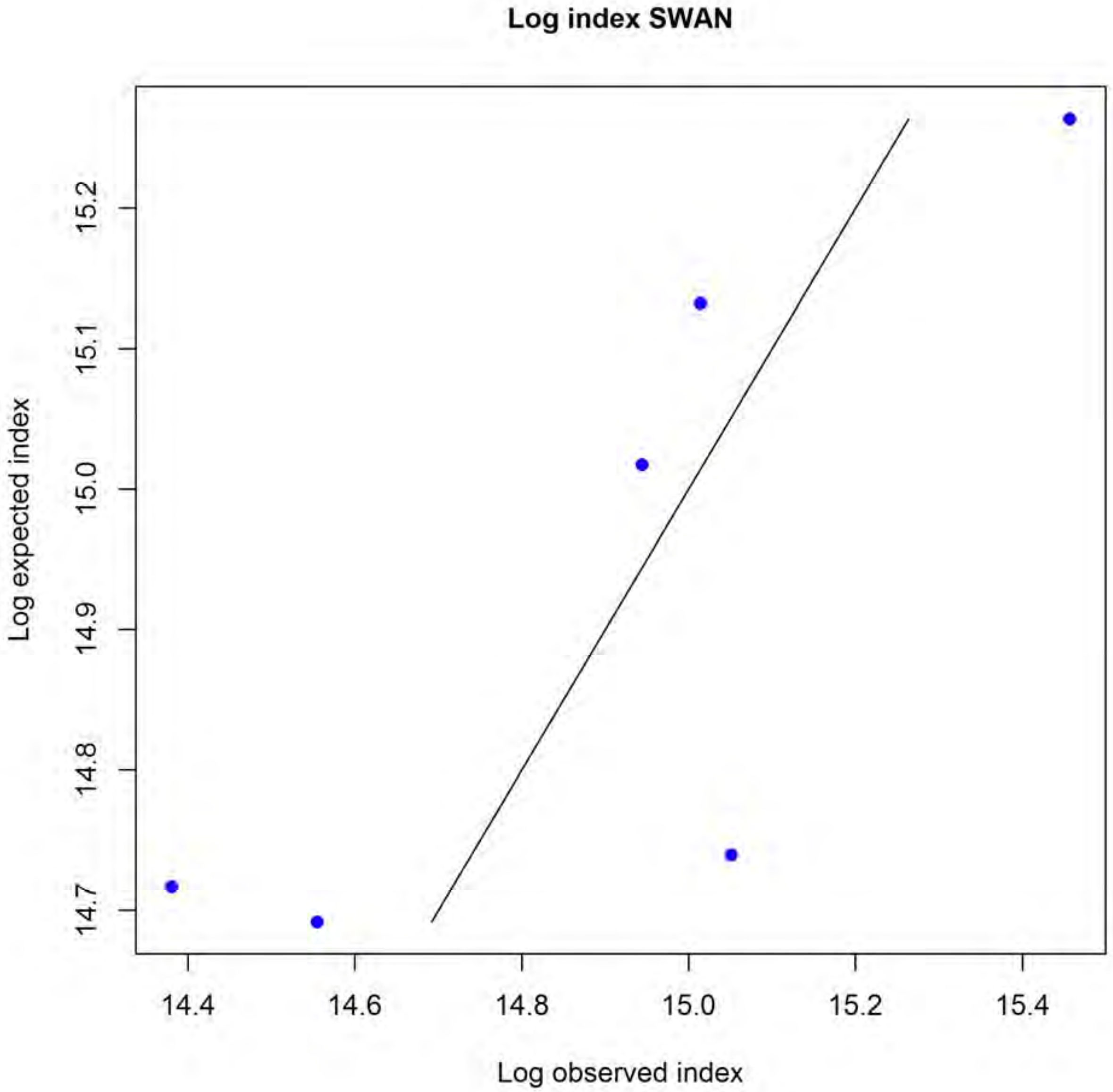

Log index SWAN
Log index
16.0
15.5
15.0
14.5
14.0
1998
2000
2002
2004
2006
2008
2010
Year

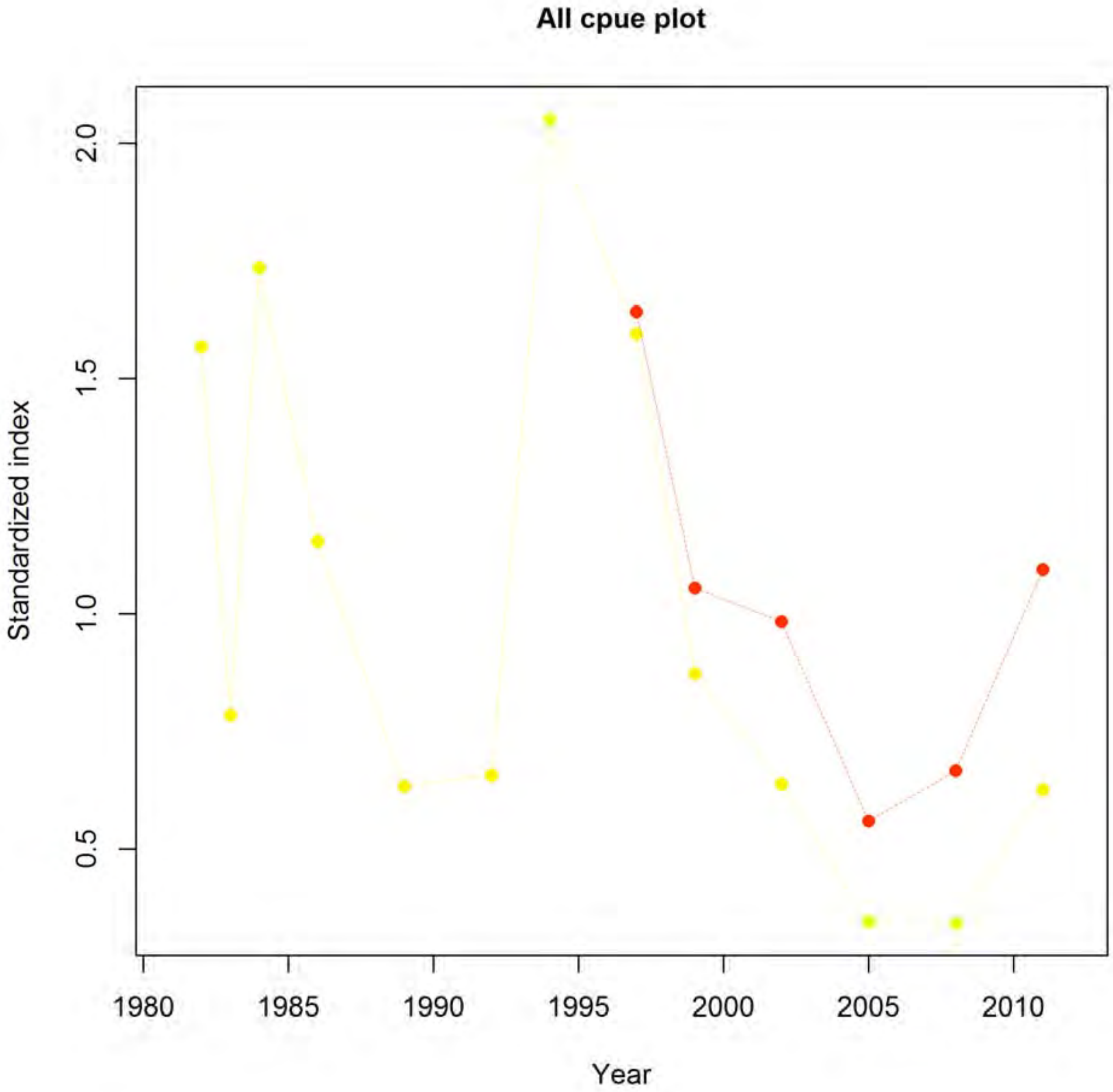

Log index NperTow+mm
Log index
5.5
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1985
1990
1995
2000
2005
2010
Year


Log index SWAN
Log index
Year
16.0
15.5
15.0
14.5
14.0
1998
2000
2002
2004
2006
2008
2010

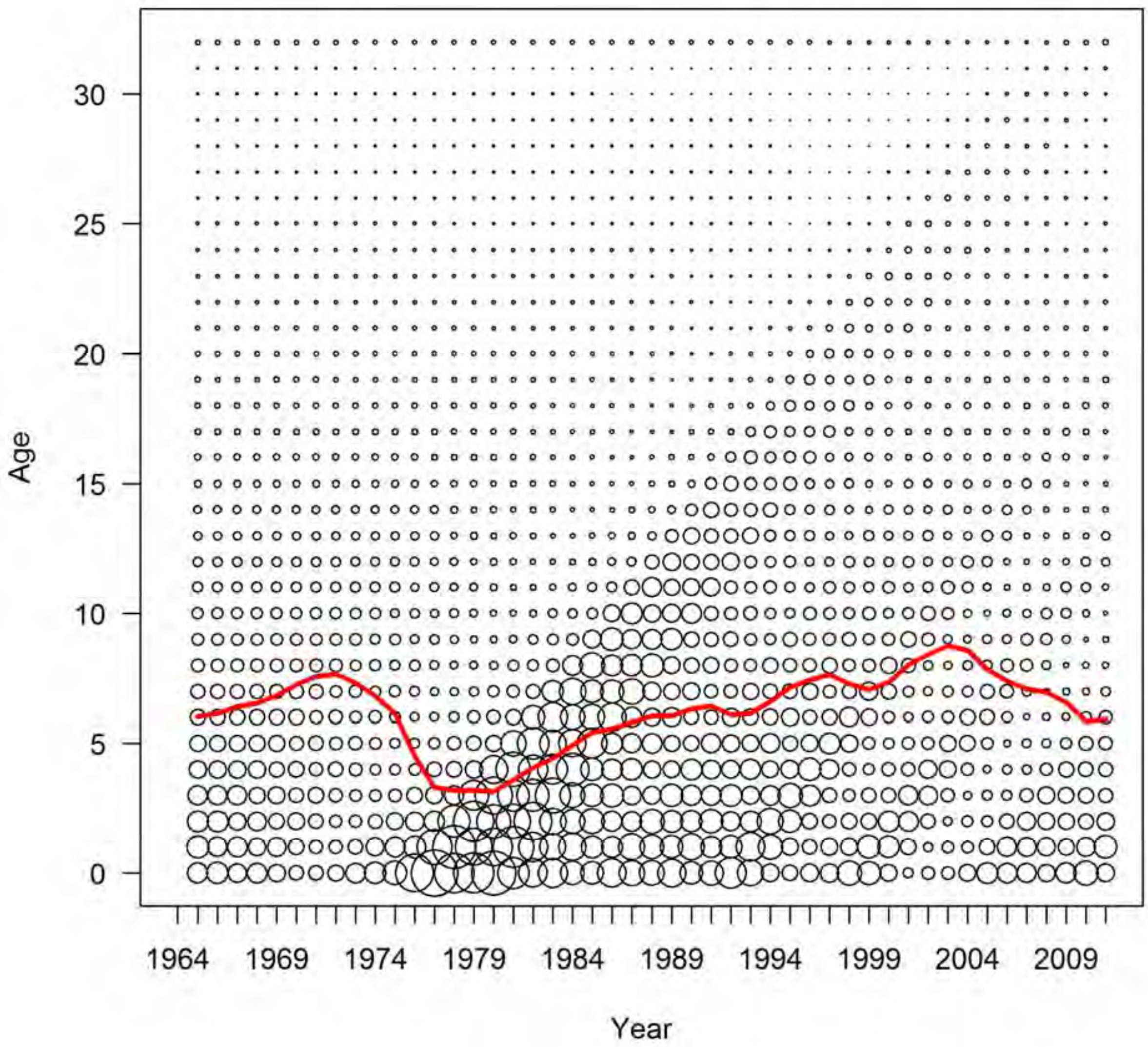
Log index NperTow+mm

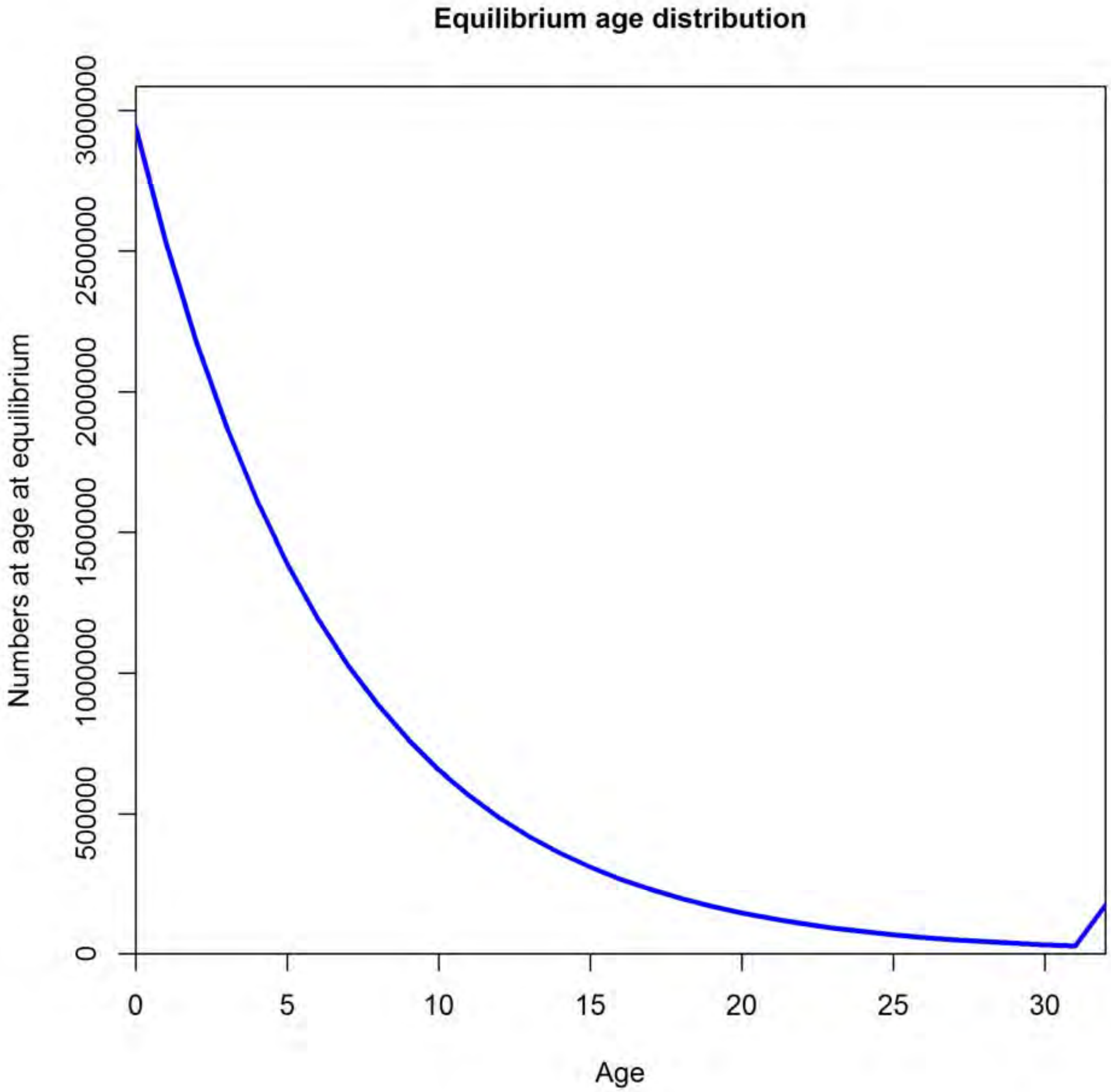


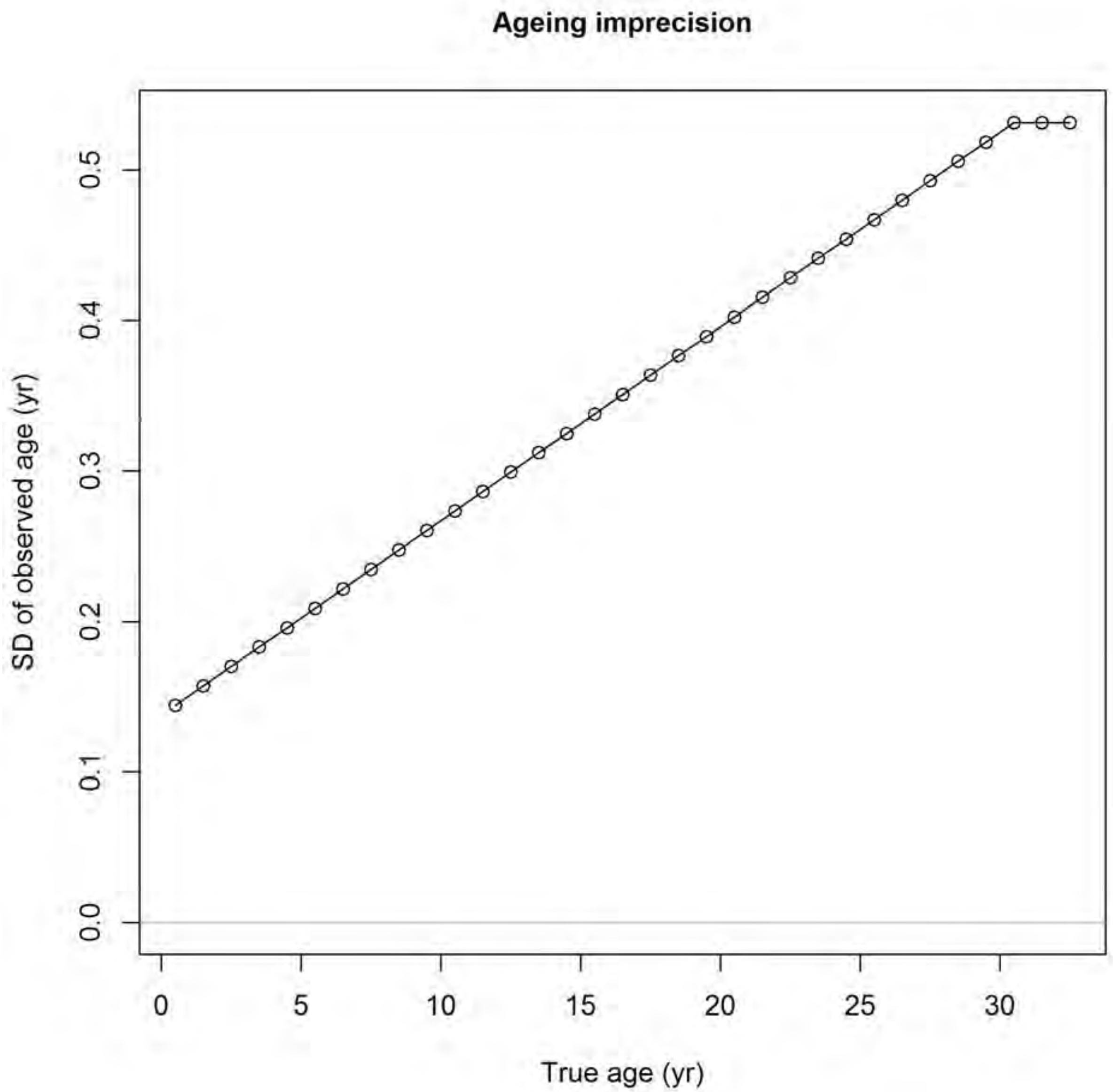

Log index SWAN
Log expected index
15.2
15.1
15.0
14.9
14.8
14.7
14.4
14.6
14.8
15.0
15.2
15.4
Log observed index

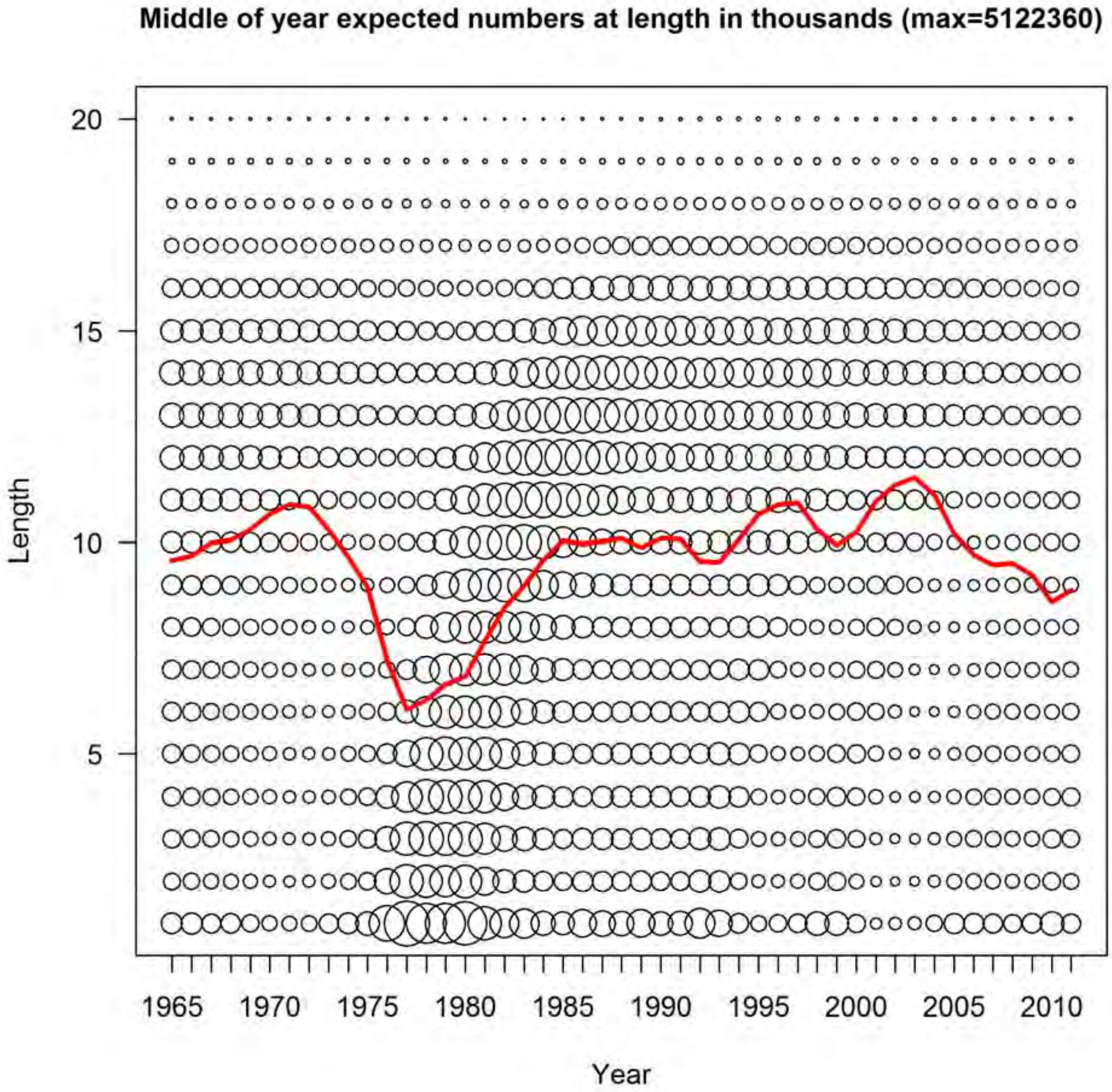

All cpue plot
Standardized index
0.5
1.0
1.5
2.0
Year
1980
1985
1990
1995
2000
2005
2010

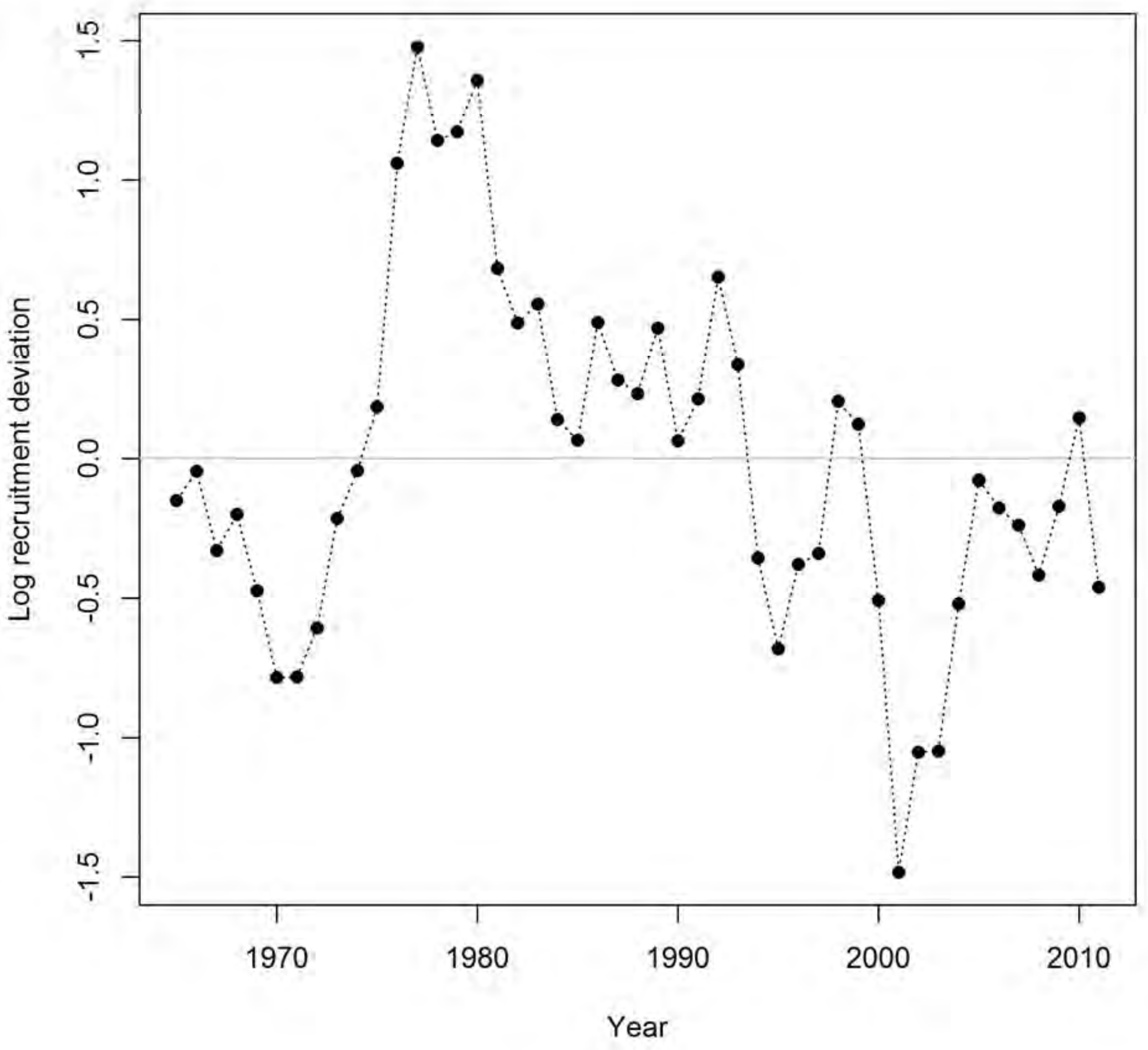
Middle of year expected numbers at age in thousands (max=9887690)

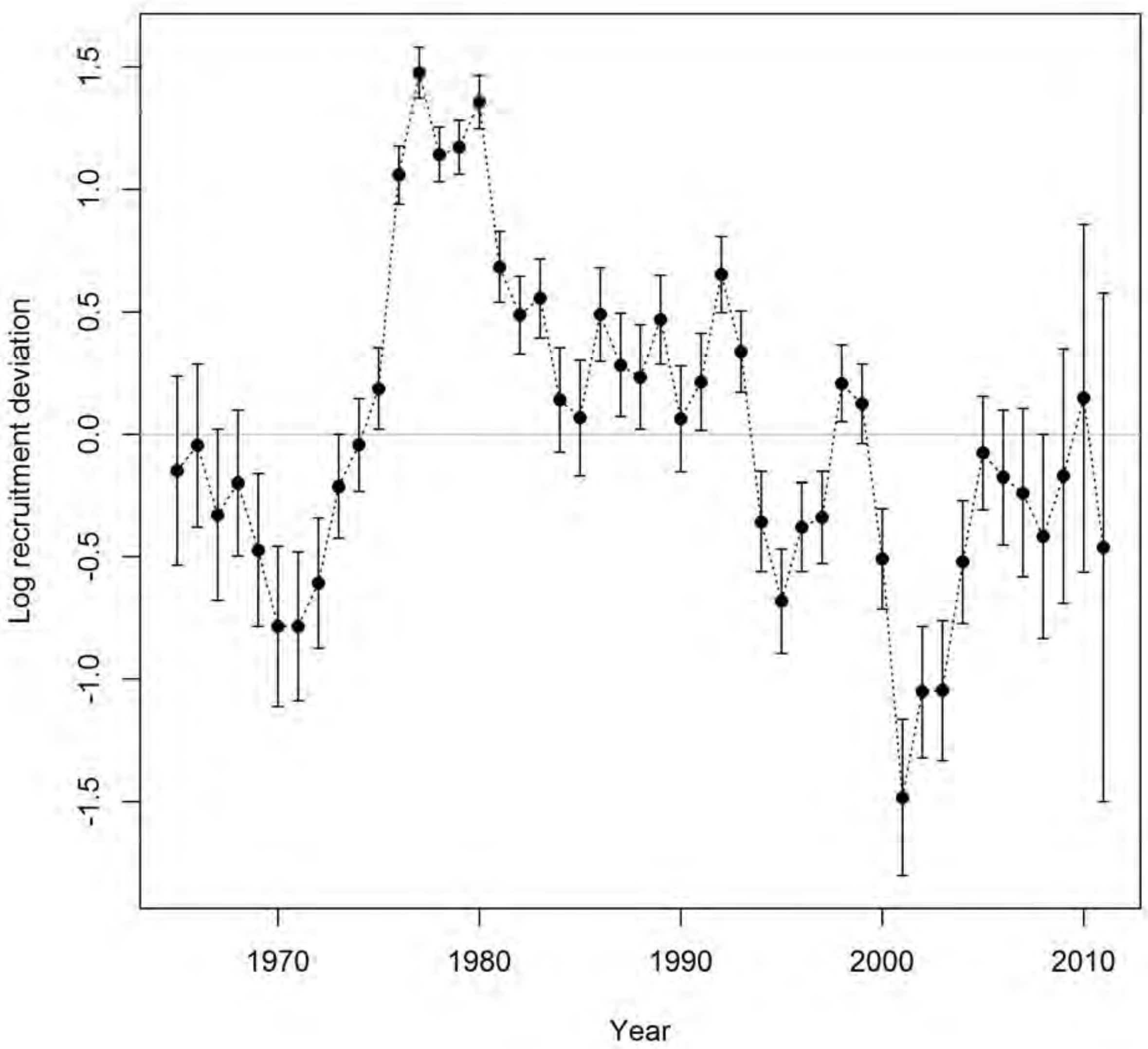


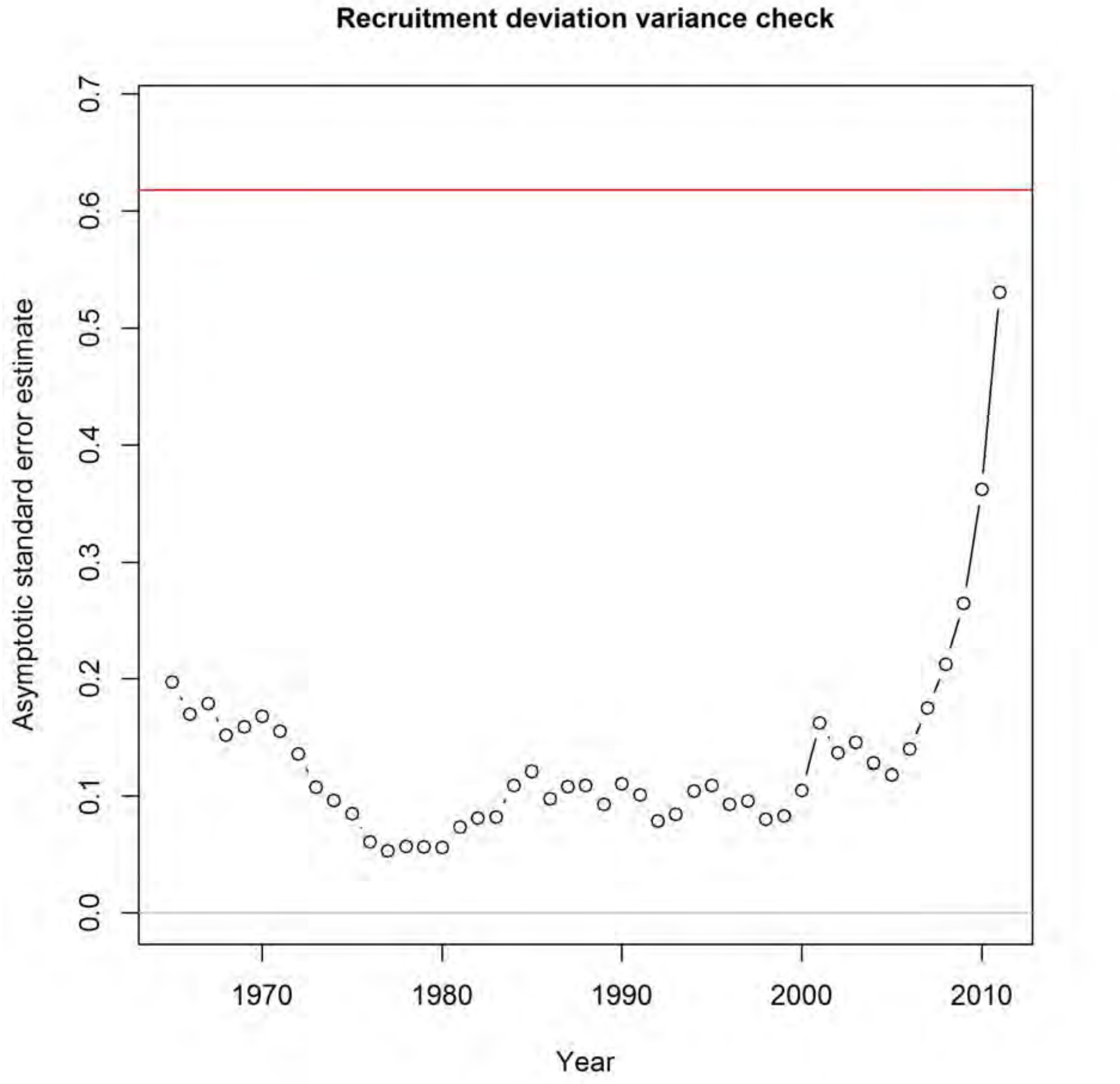

Equilibrium age distribution
Numbers at age at equilibrium
0
500000
1000000
1500000
2000000
2500000
3000000
Age
0
5
10
15
20
25
30

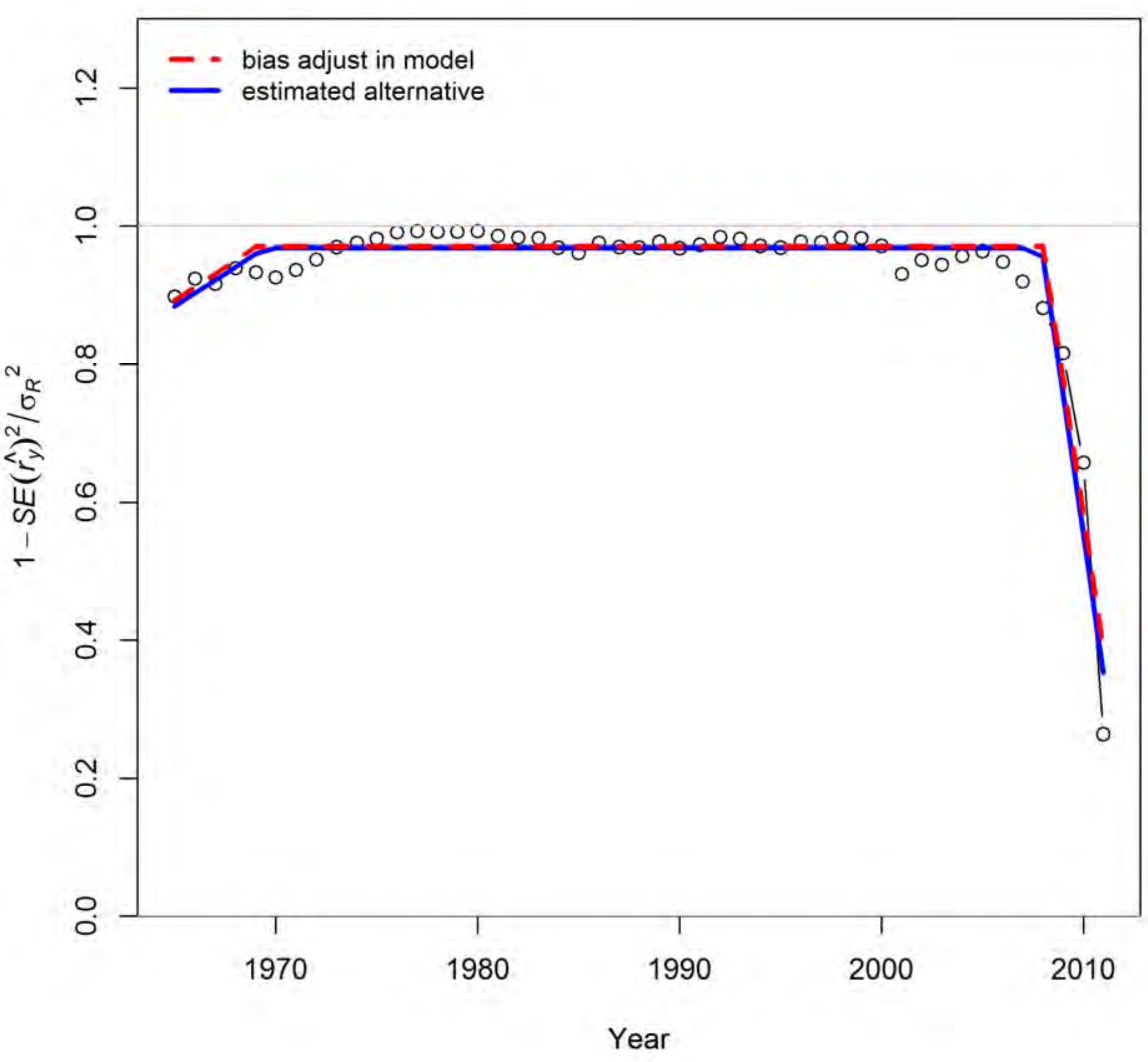

Ageing imprecision
SD of observed age (yr)
0.0
0.1
0.2
0.3
0.4
0.5
True age (yr)
0
5
10
15
20
25
30

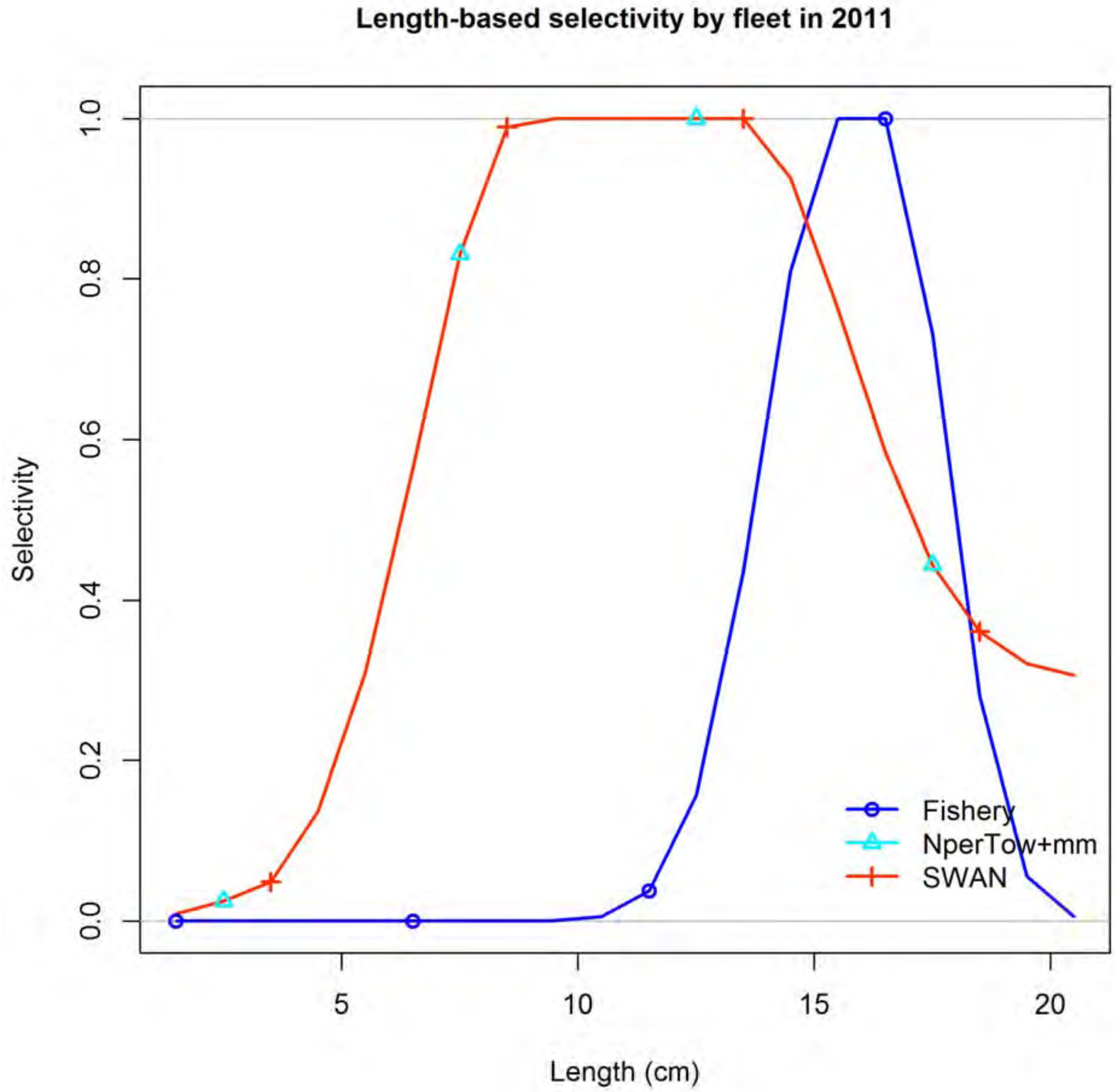

Middle of year expected numbers at length in thousands (max=5122360)
Length
20
15
10
5
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
Year

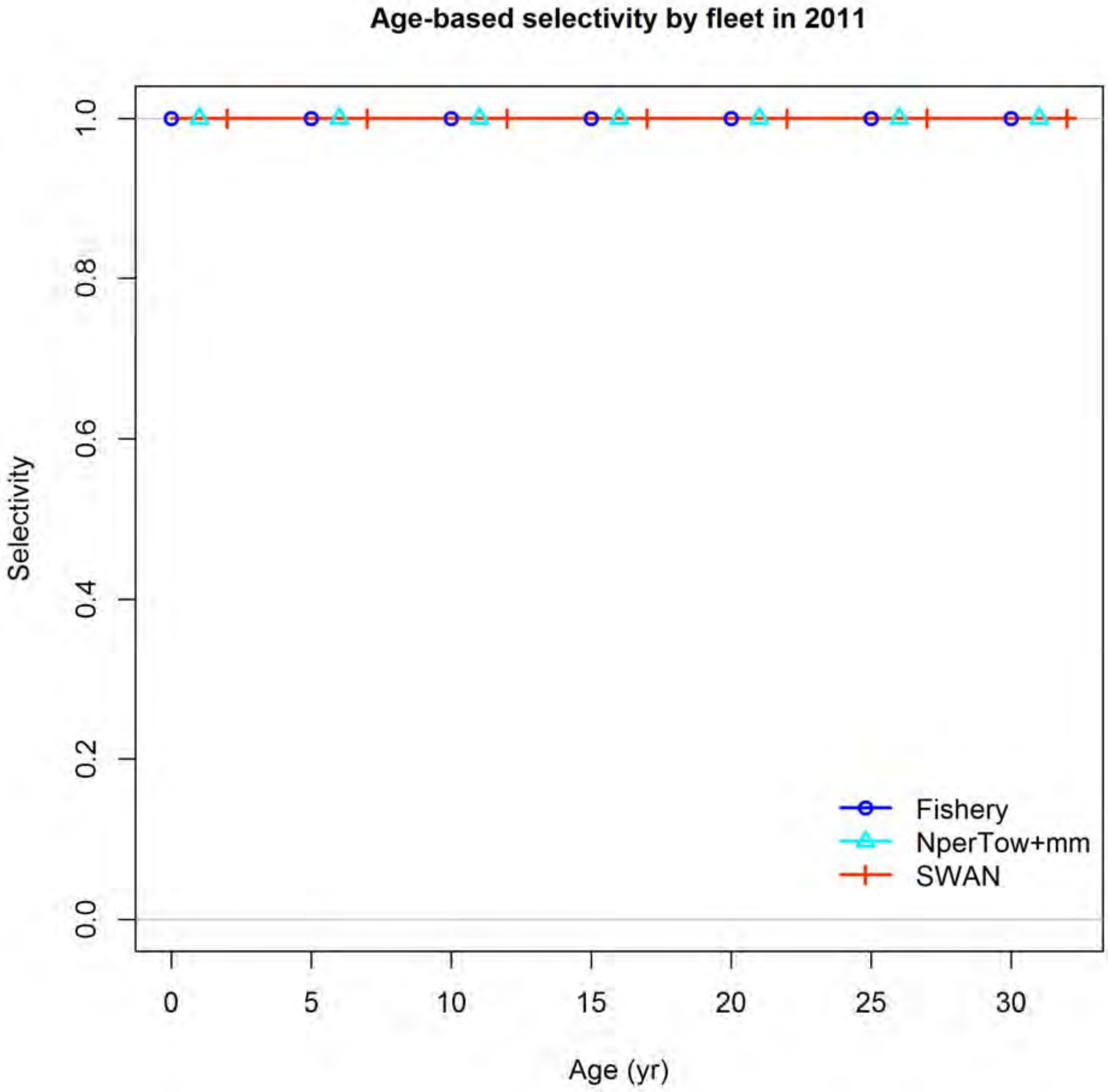

Log recruitment deviation
1.5
1.0
0.5
0.0
-0.5
-1.0
-1.5
1970
1980
1990
2000
2010
Year

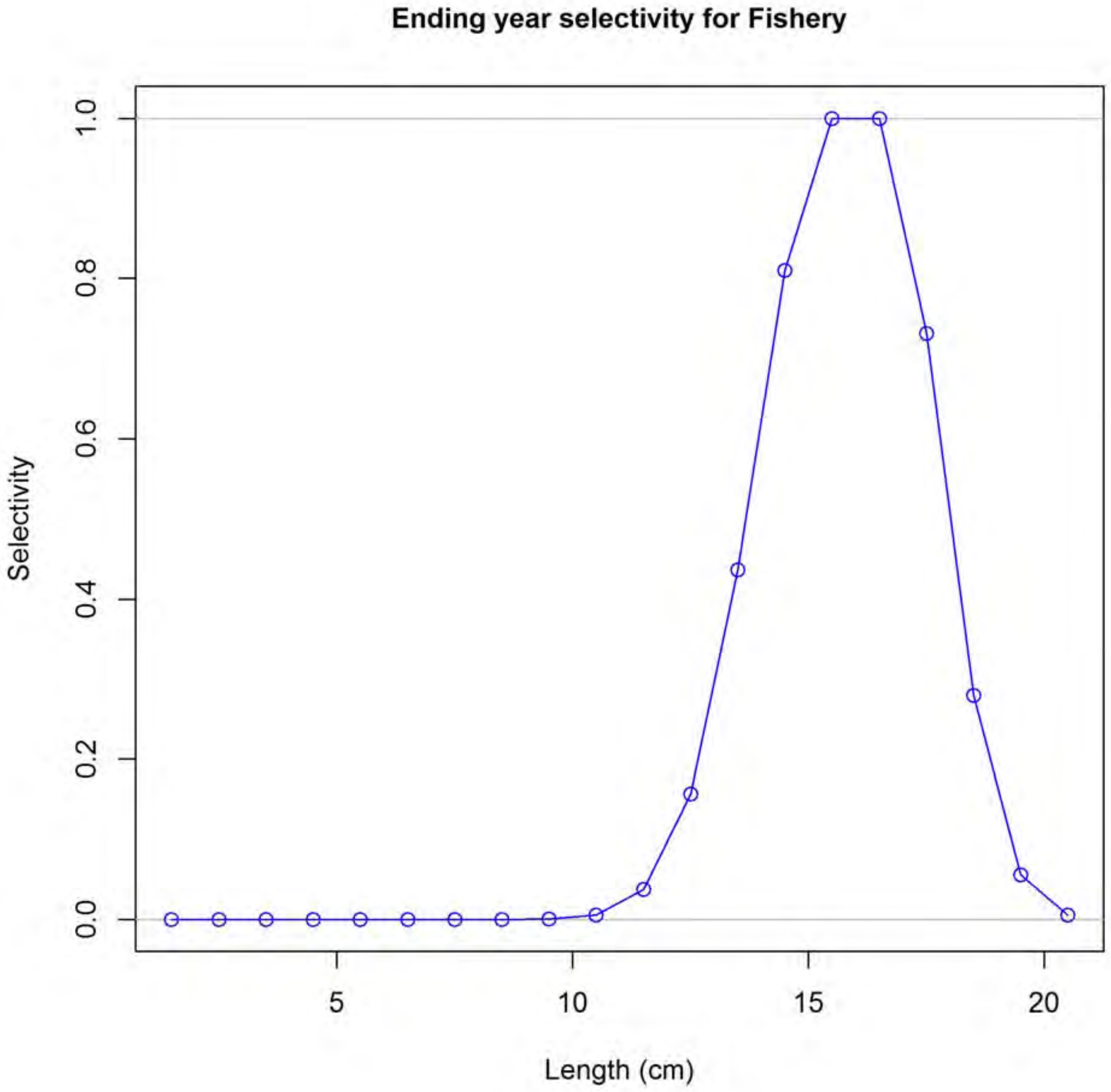

Log recruitment deviation
1.5
1.0
0.5
0.0
-0.5
-1.0
-1.5
1970
1980
1990
2000
2010
Year

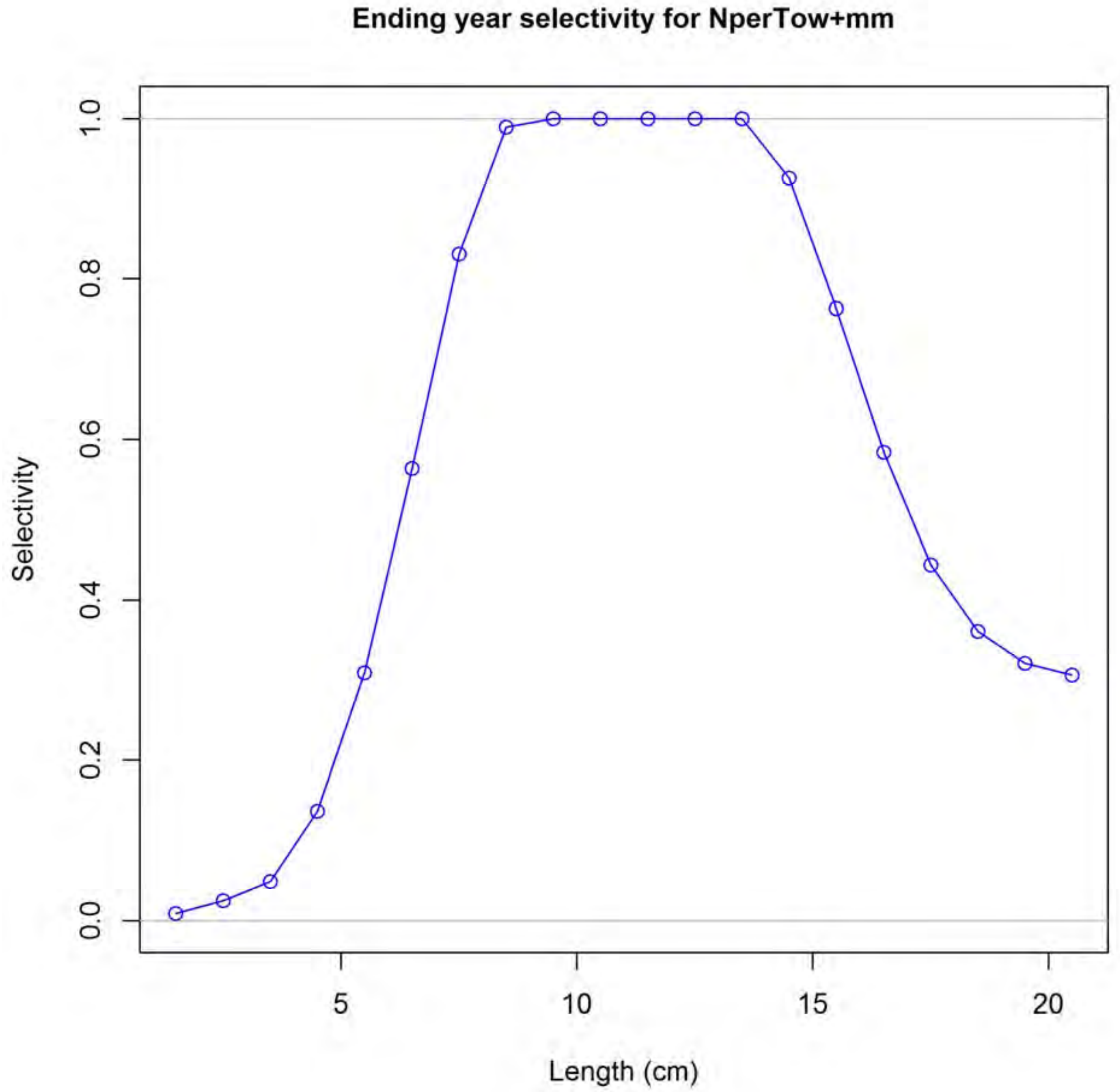

Recruitment deviation variance check
Asymptotic standard error estimate
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
1970
1980
1990
2000
2010
Year

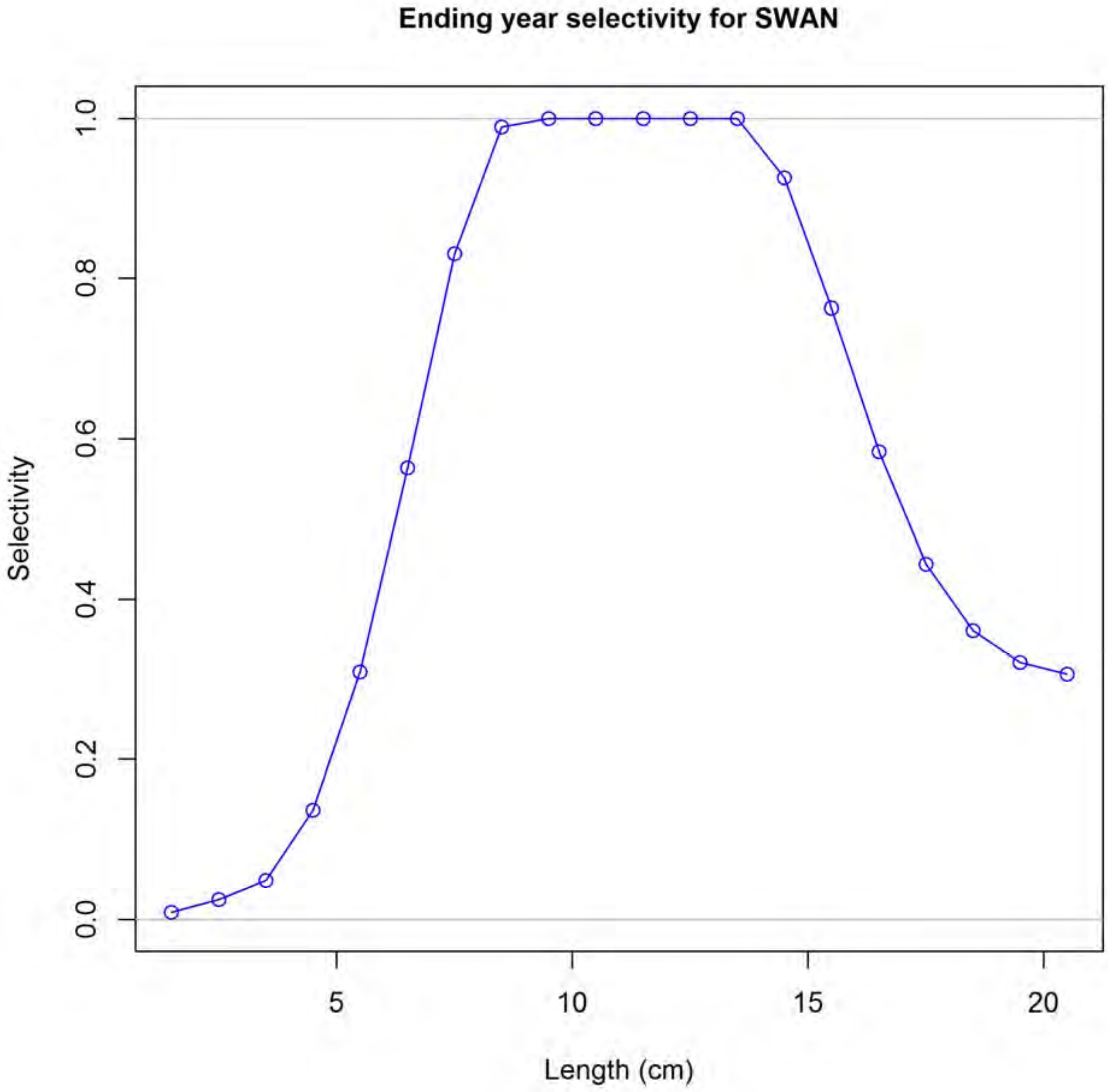

bias adjust in model
estimated alternative
$1 - SE(\hat{r}_y)^2/\sigma_R^2$
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1970
1980
1990
2000
2010
Year

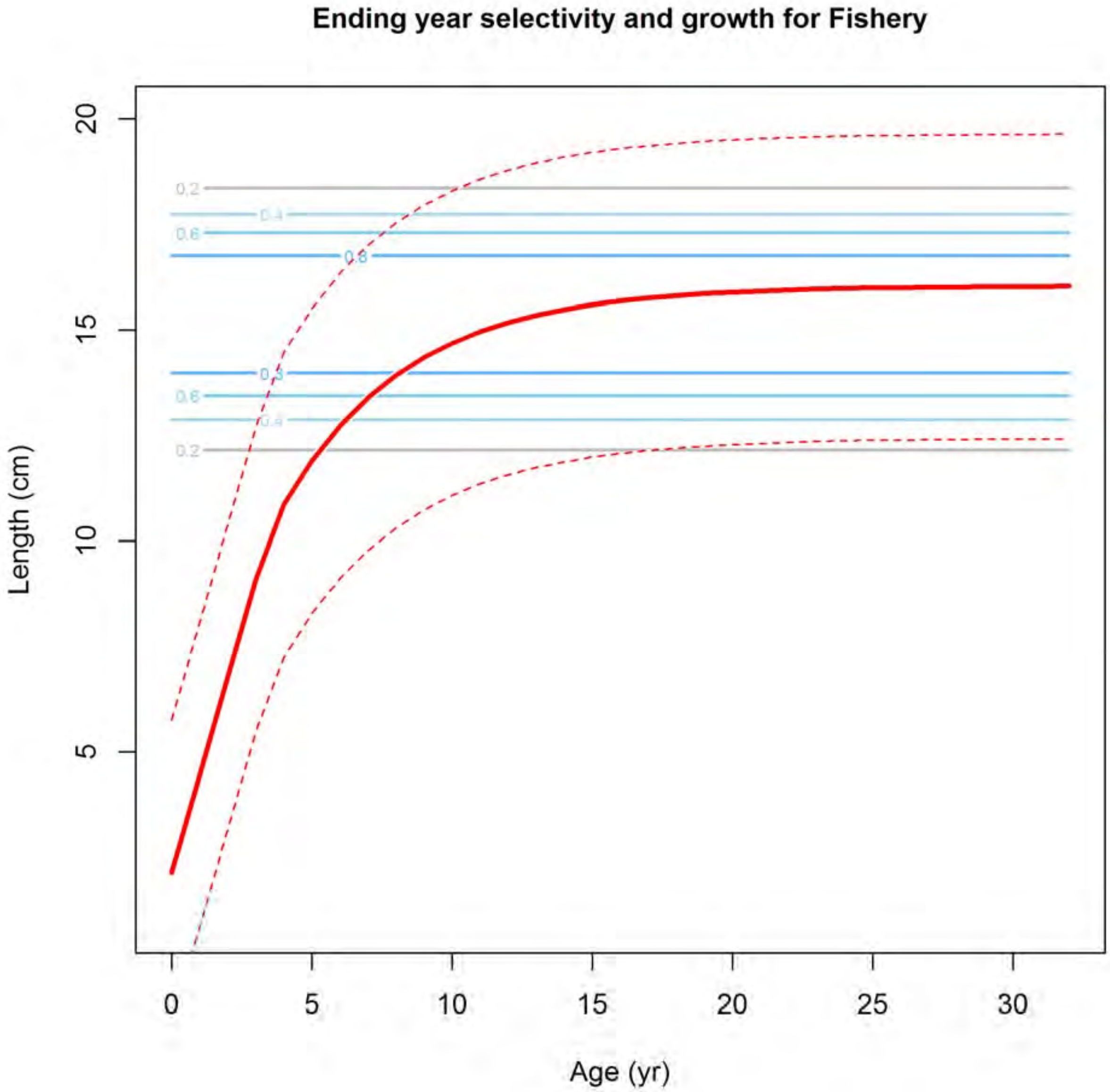

Length-based selectivity by fleet in 2011
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)
Fishery
NperTow+mm
SWAN

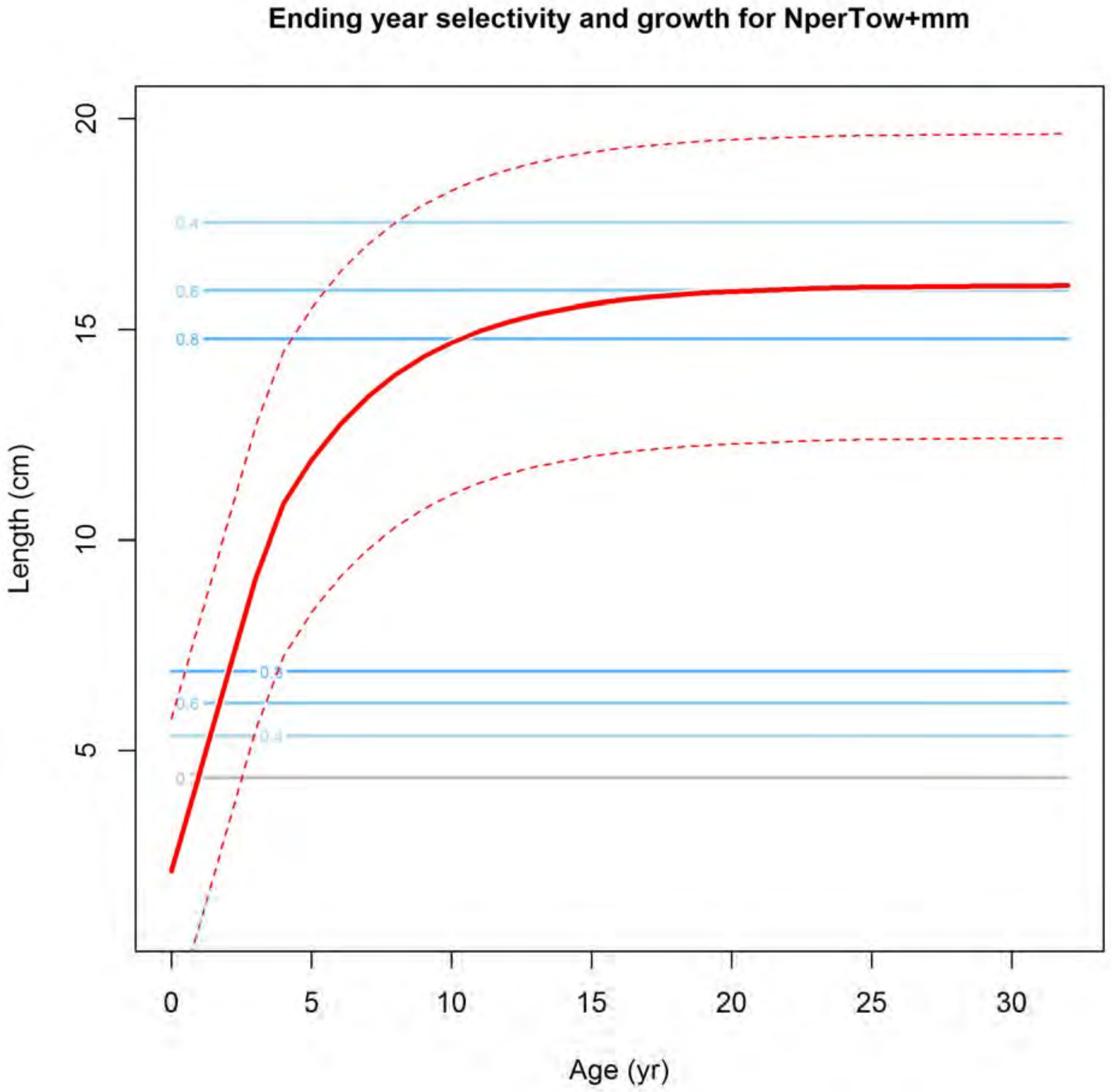

Age-based selectivity by fleet in 2011
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
Age (yr)
0
5
10
15
20
25
30
Fishery
NperTow+mm
SWAN

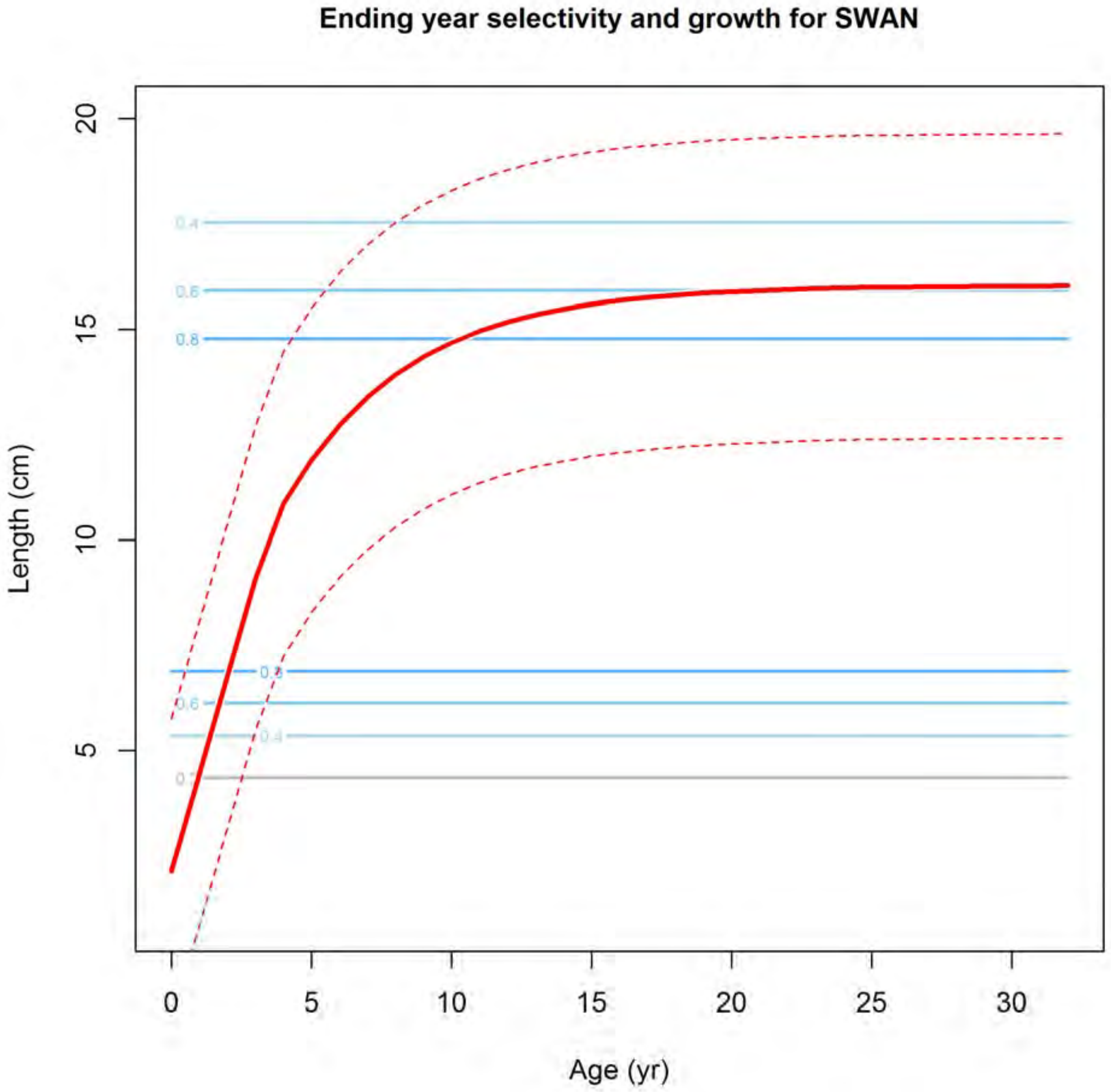

Ending year selectivity for Fishery
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)


Ending year selectivity for NperTow+mm
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)

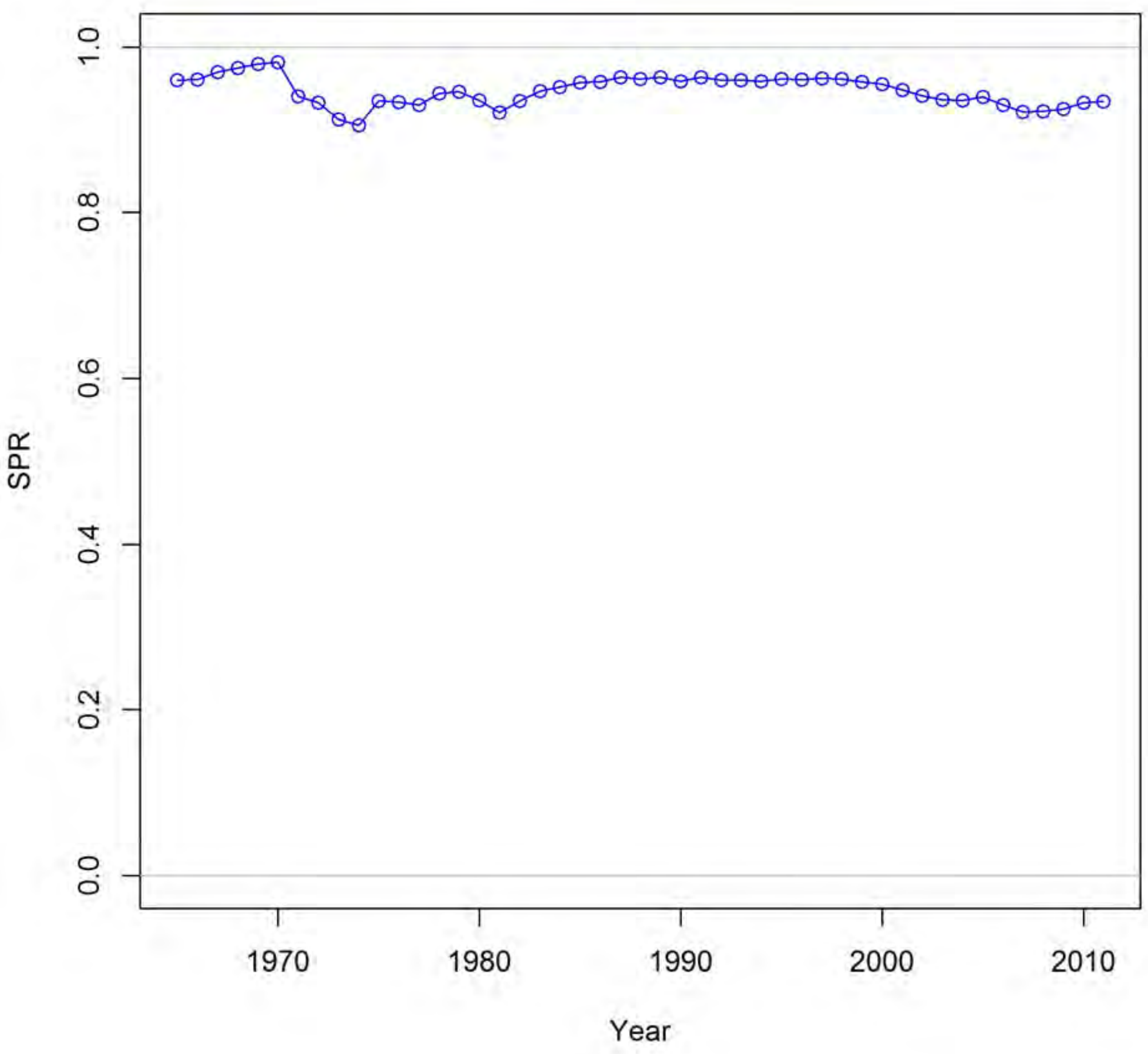

Ending year selectivity for SWAN
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)

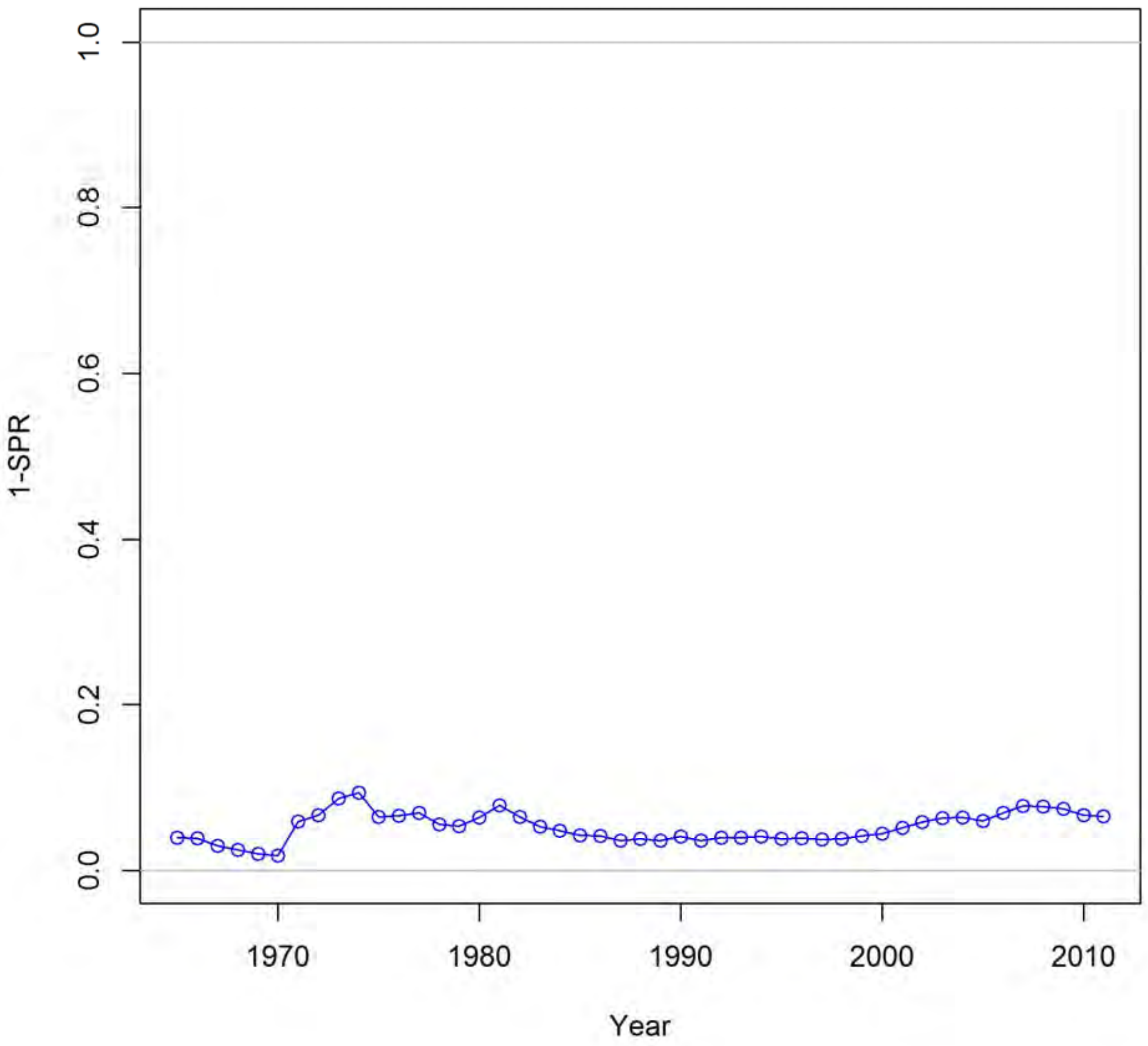

Ending year selectivity and growth for Fishery
Length (cm)
Age (yr)
5
10
15
20
0
5
10
15
20
25
30
0.2
0.4
0.6
0.8
0.8
0.6
0.4
0.2

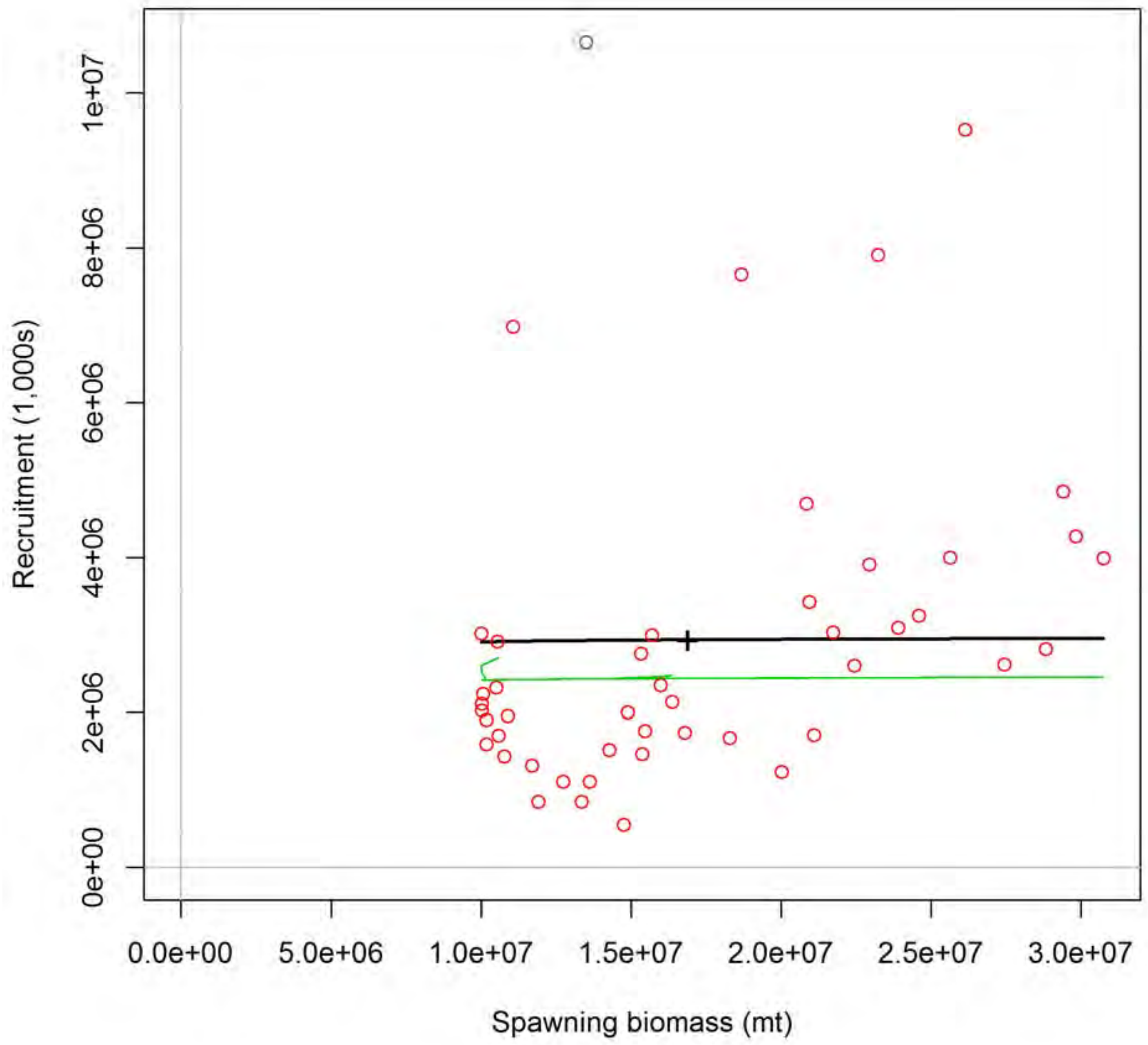

Ending year selectivity and growth for NperTow+mm
Length (cm)
20
15
10
5
0
5
10
15
20
25
30
Age (yr)
0.4
0.6
0.8
0.8
0.6
0.4
0.2

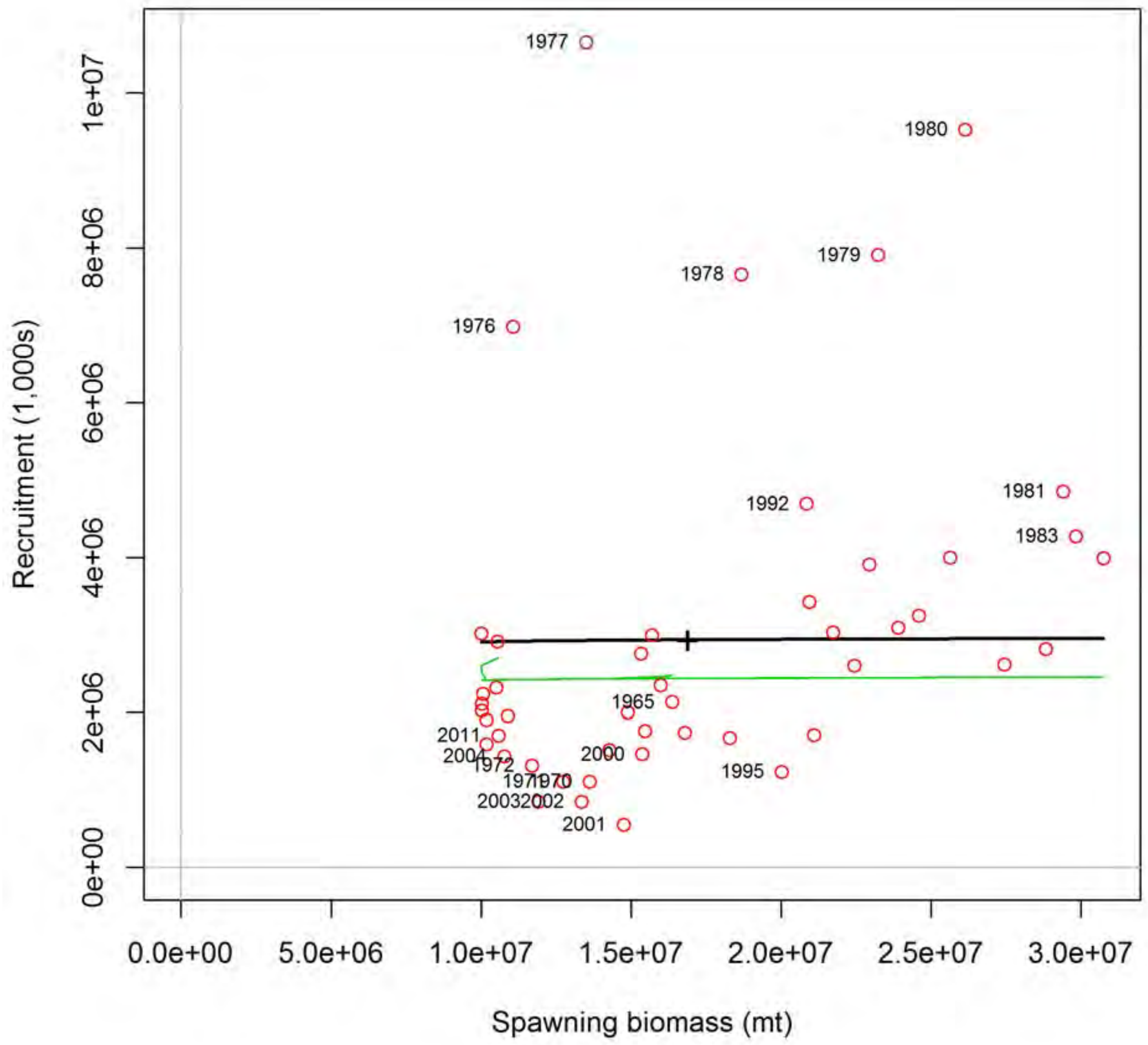


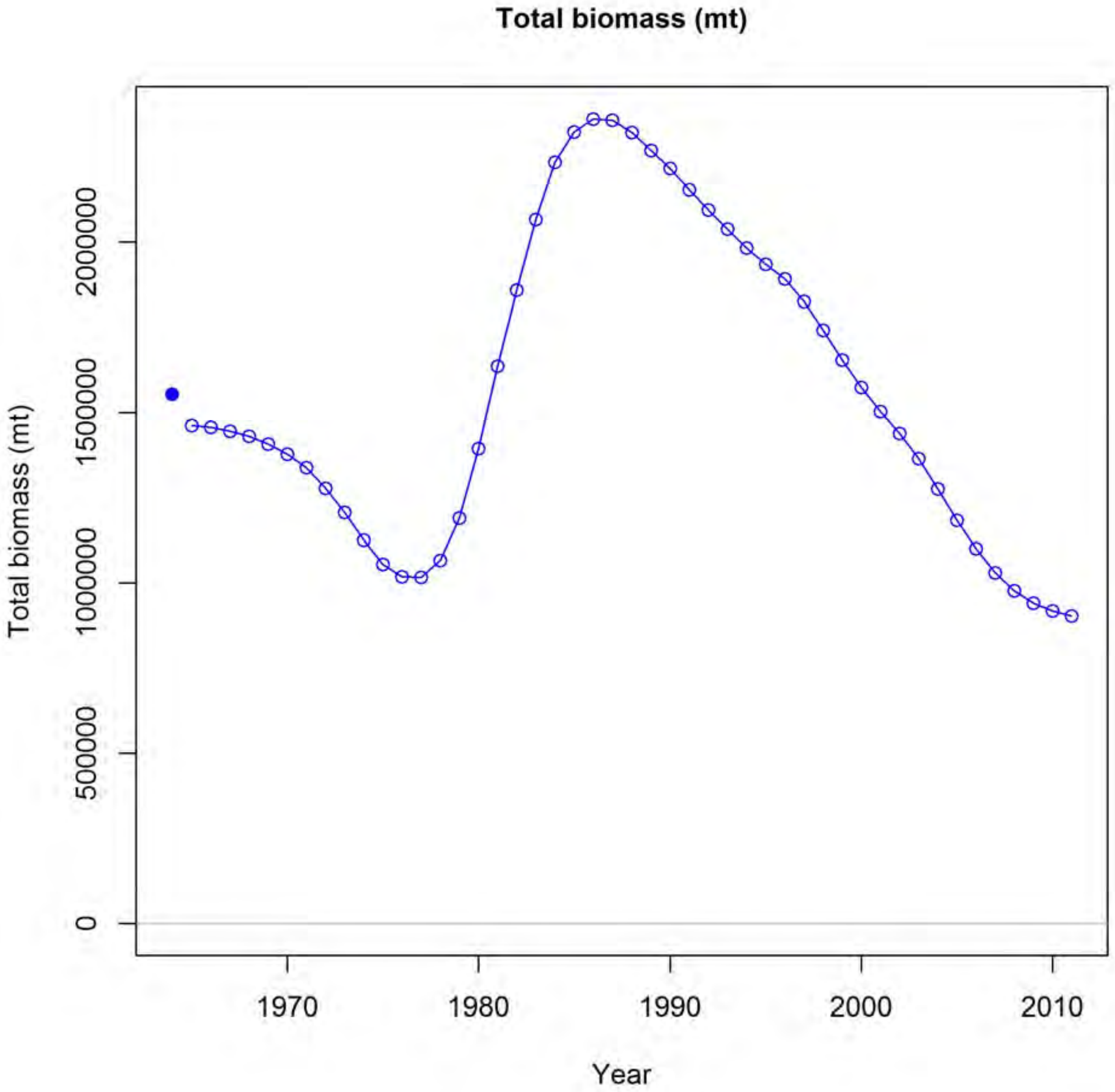
Ending year selectivity and growth for SWAN

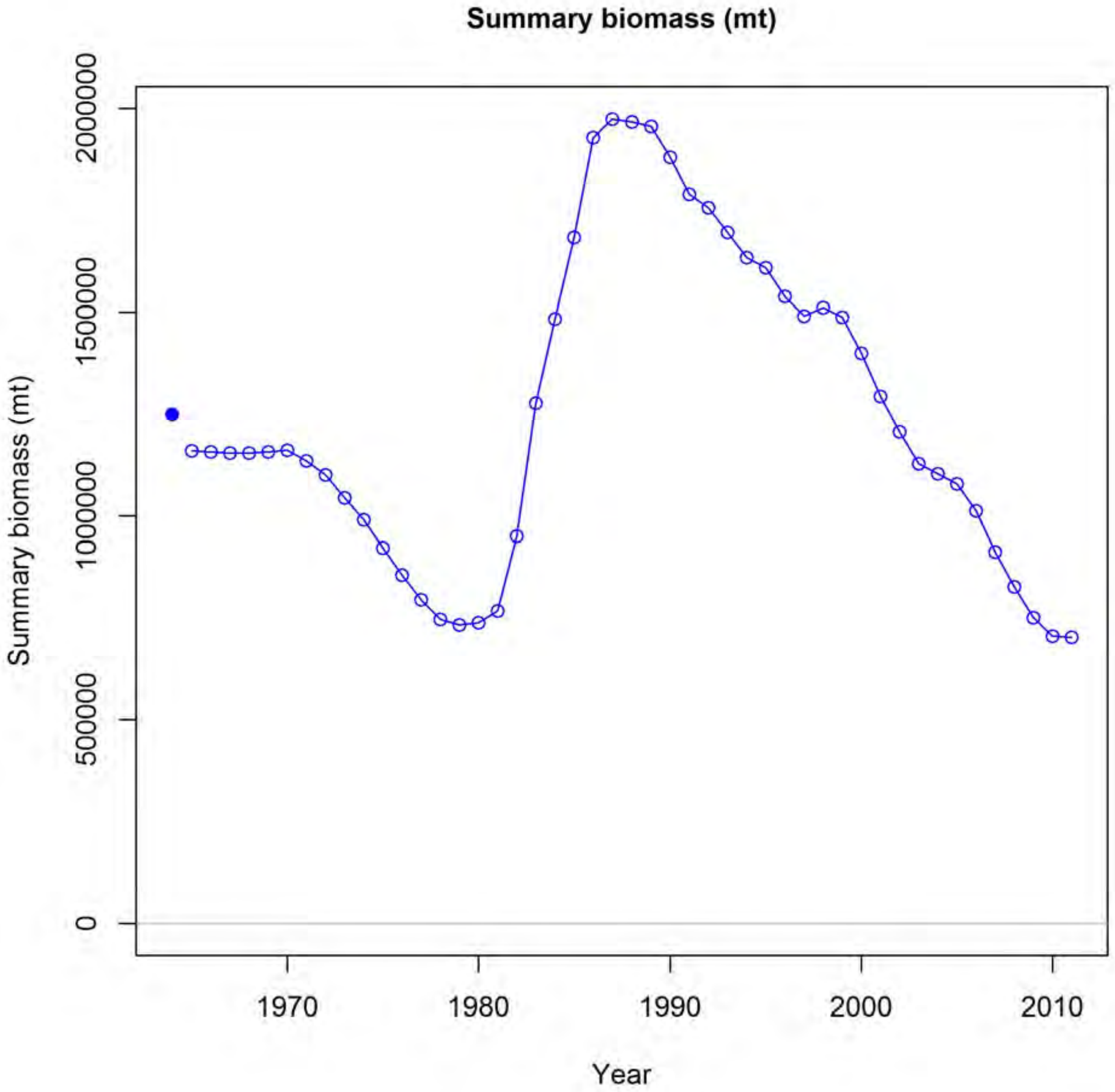

1.0
0.8
0.6
0.4
0.2
0.0
SPR
1970
1980
1990
2000
2010
Year

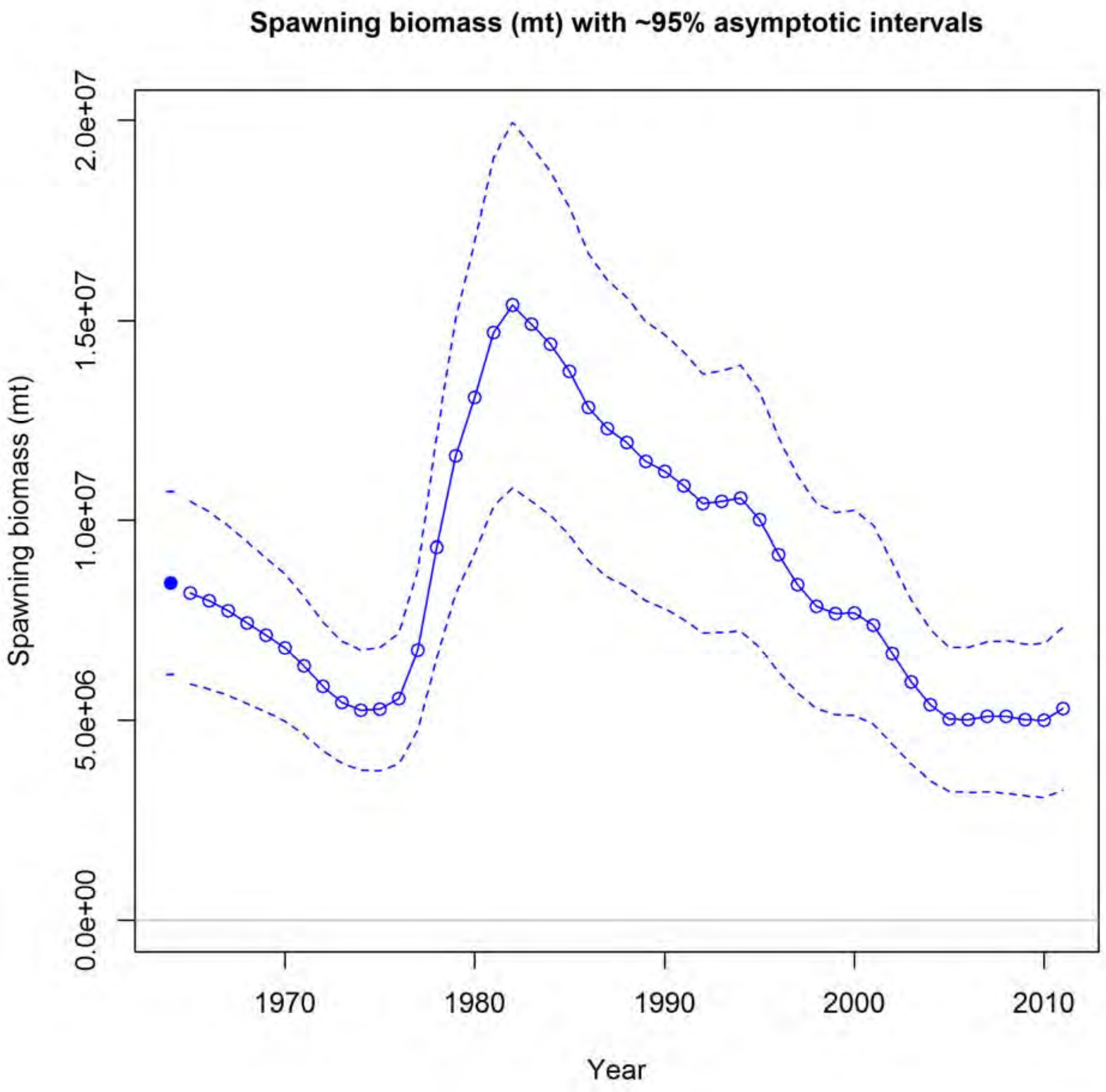

1-SPR
0.0
0.2
0.4
0.6
0.8
1.0
1970
1980
1990
2000
2010
Year

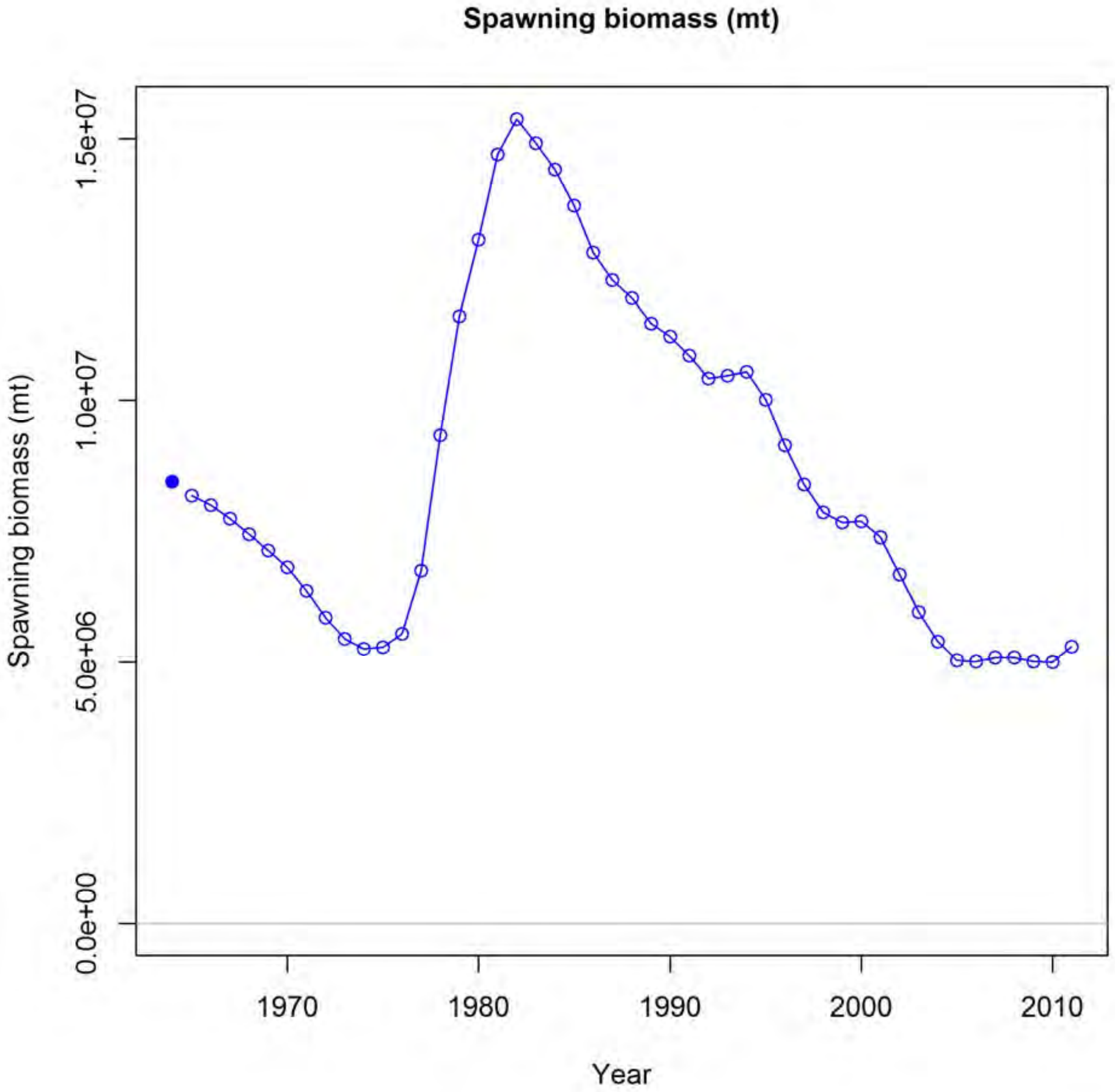

Recruitment (1,000s)
0e+00
2e+06
4e+06
6e+06
8e+06
1e+07
0.0e+00
5.0e+06
1.0e+07
1.5e+07
2.0e+07
2.5e+07
3.0e+07
Spawning biomass (mt)

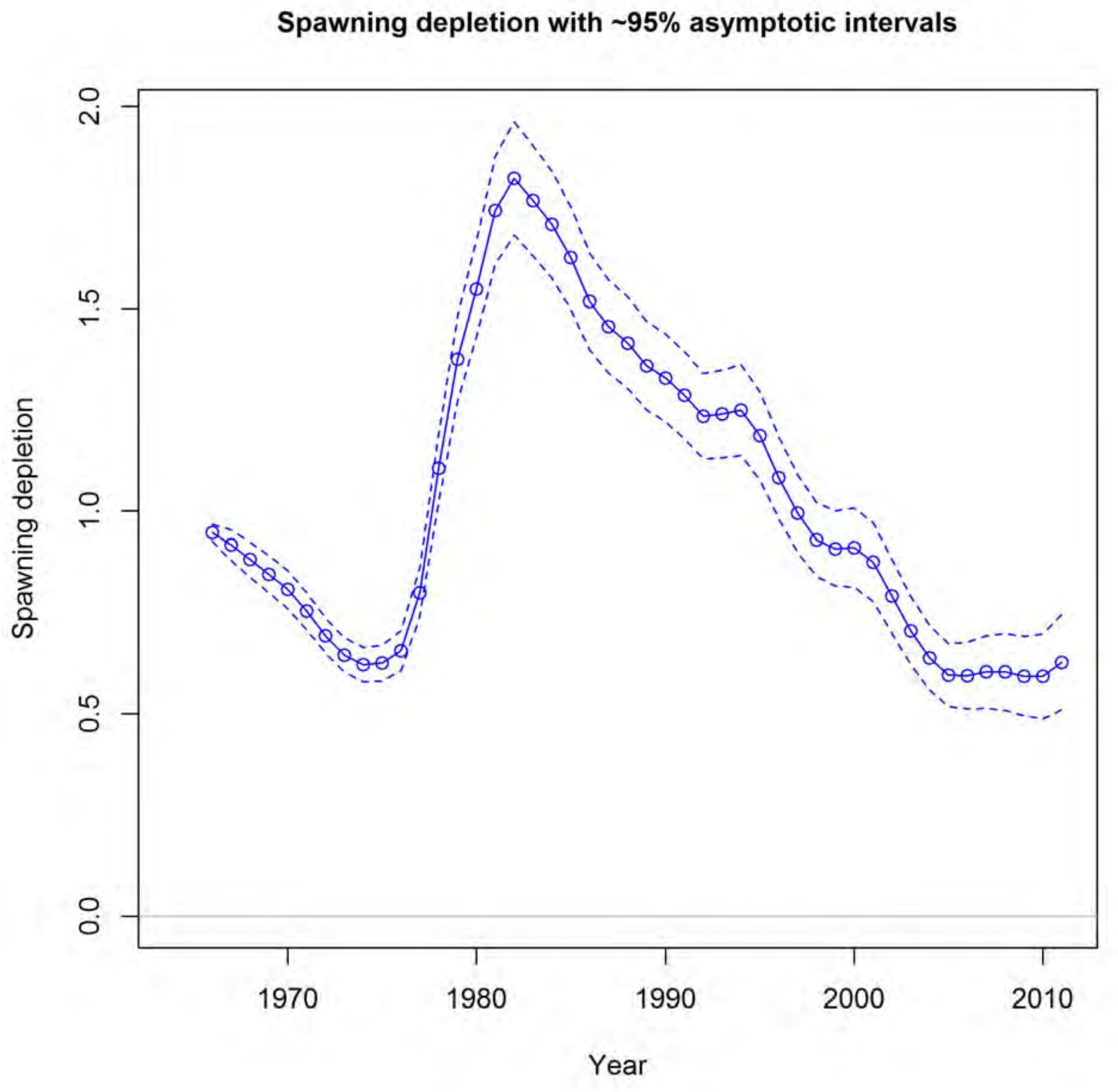

Recruitment (1,000s)
0e+00
2e+06
4e+06
6e+06
8e+06
1e+07
Spawning biomass (mt)
0.0e+00
5.0e+06
1.0e+07
1.5e+07
2.0e+07
2.5e+07
3.0e+07
1977
1980
1979
1978
1976
1992
1981
1983
1965
2011
2004
1972
1971
1970
2000
2003
2002
1995
2001

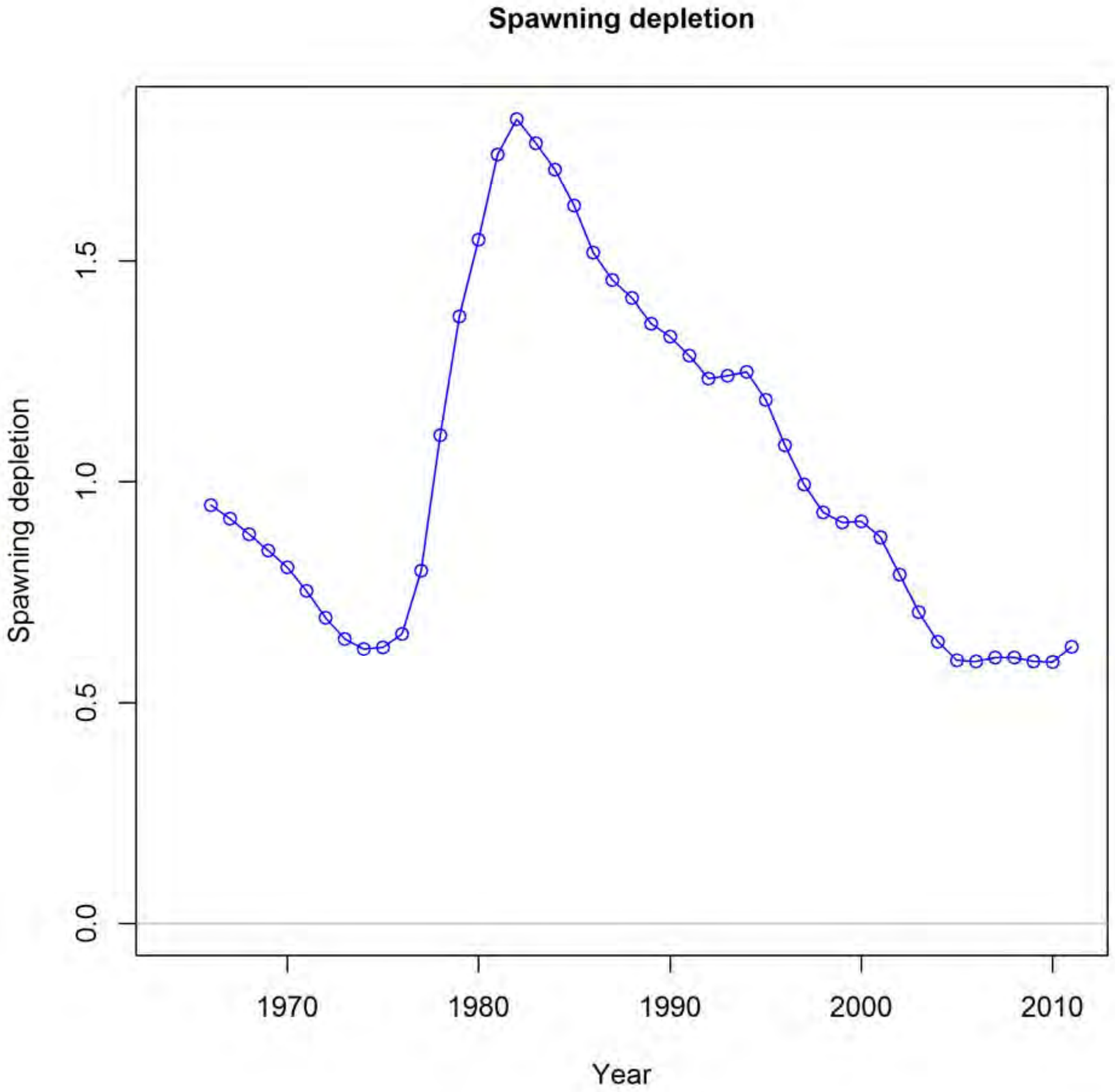

Total biomass (mt)
Total biomass (mt)
0
500000
1000000
1500000
2000000
1970
1980
1990
2000
2010
Year

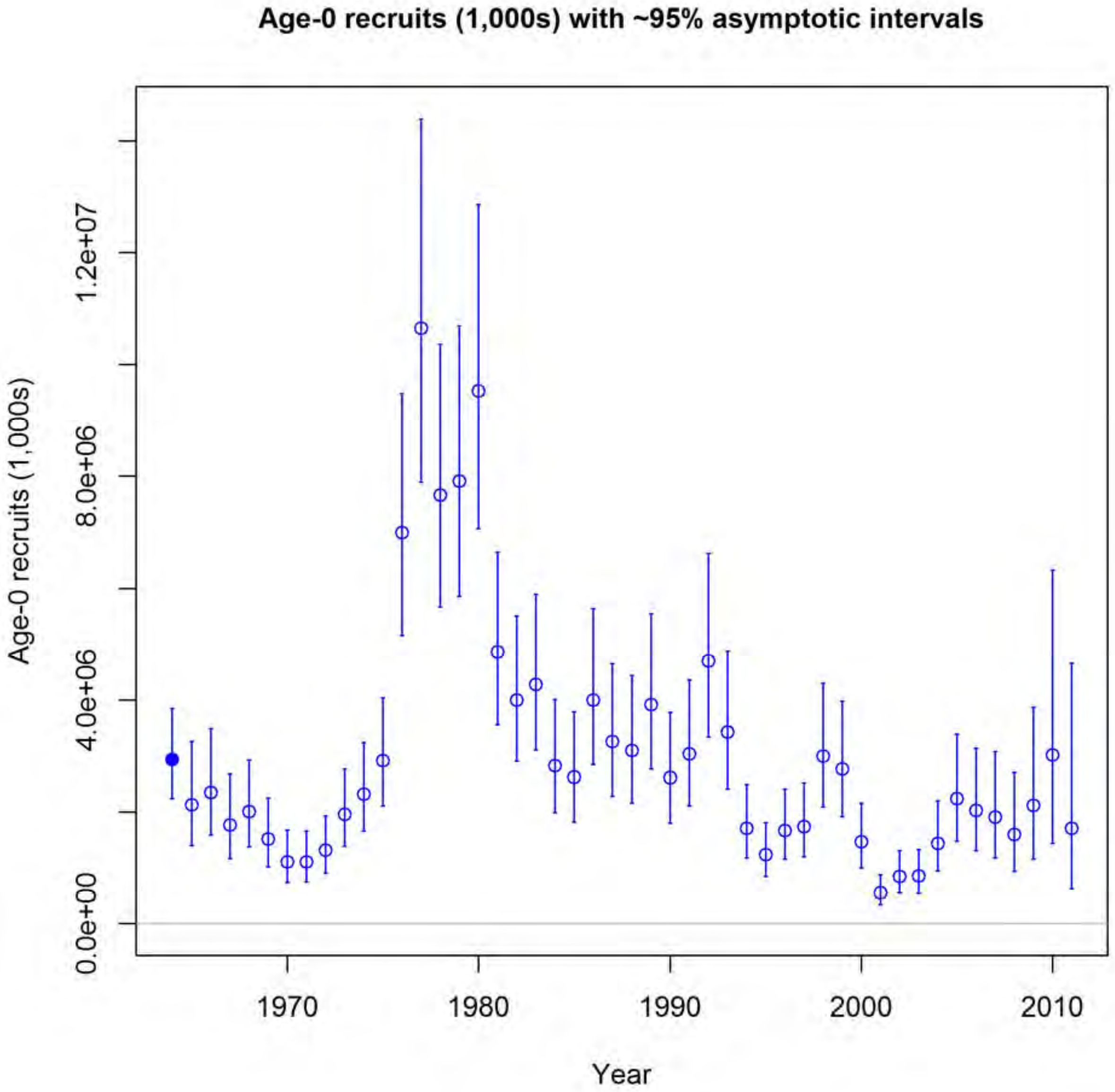

Summary biomass (mt)
Summary biomass (mt)
0
500000
1000000
1500000
2000000
1970
1980
1990
2000
2010
Year

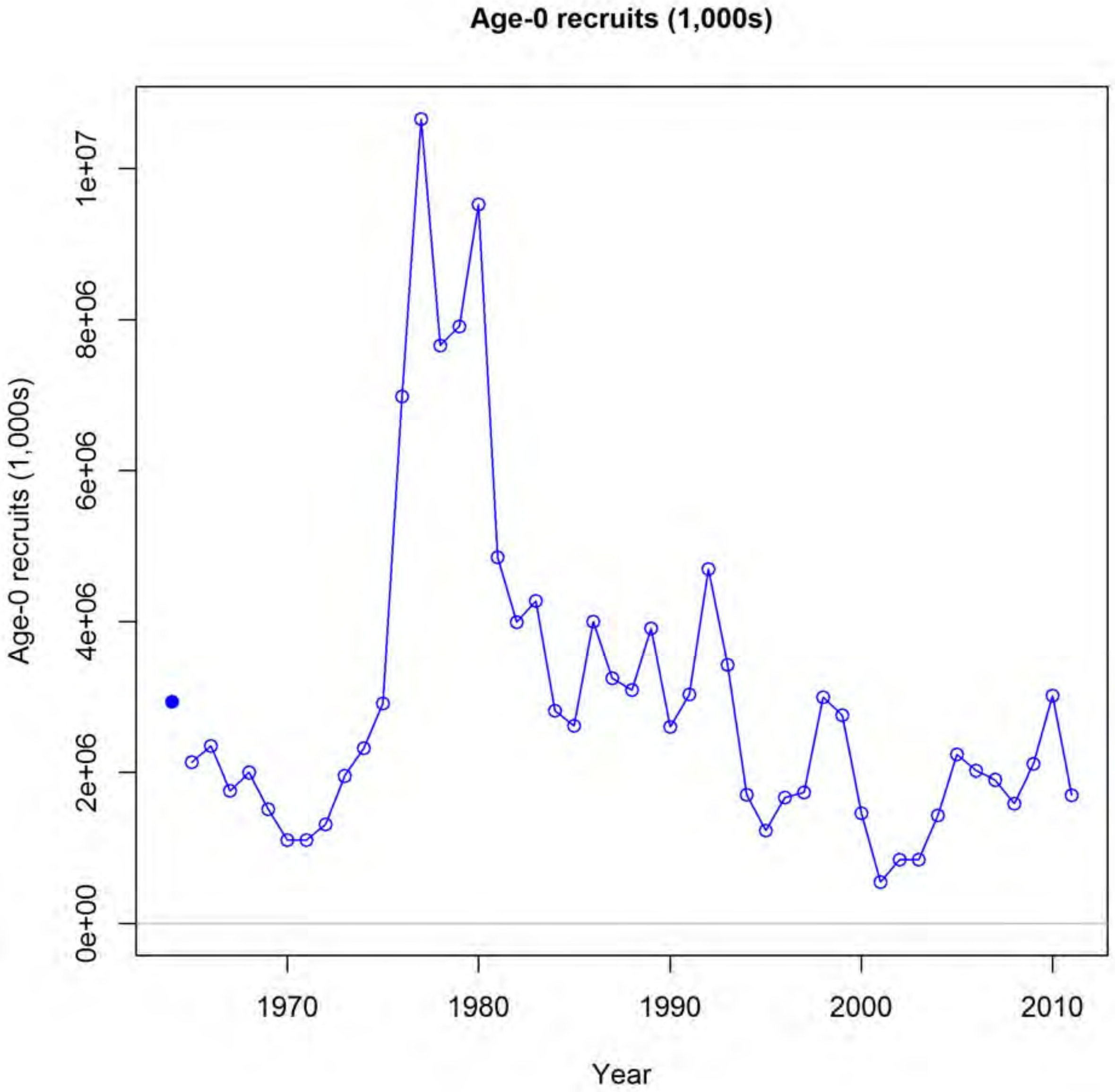

Spawning biomass (mt) with ~95% asymptotic intervals
Spawning biomass (mt)
0.0e+00
5.0e+06
1.0e+07
1.5e+07
2.0e+07
1970
1980
1990
2000
2010
Year


Spawning biomass (mt)
Spawning biomass (mt)
0.0e+00
5.0e+06
1.0e+07
1.5e+07
1970
1980
1990
2000
2010
Year

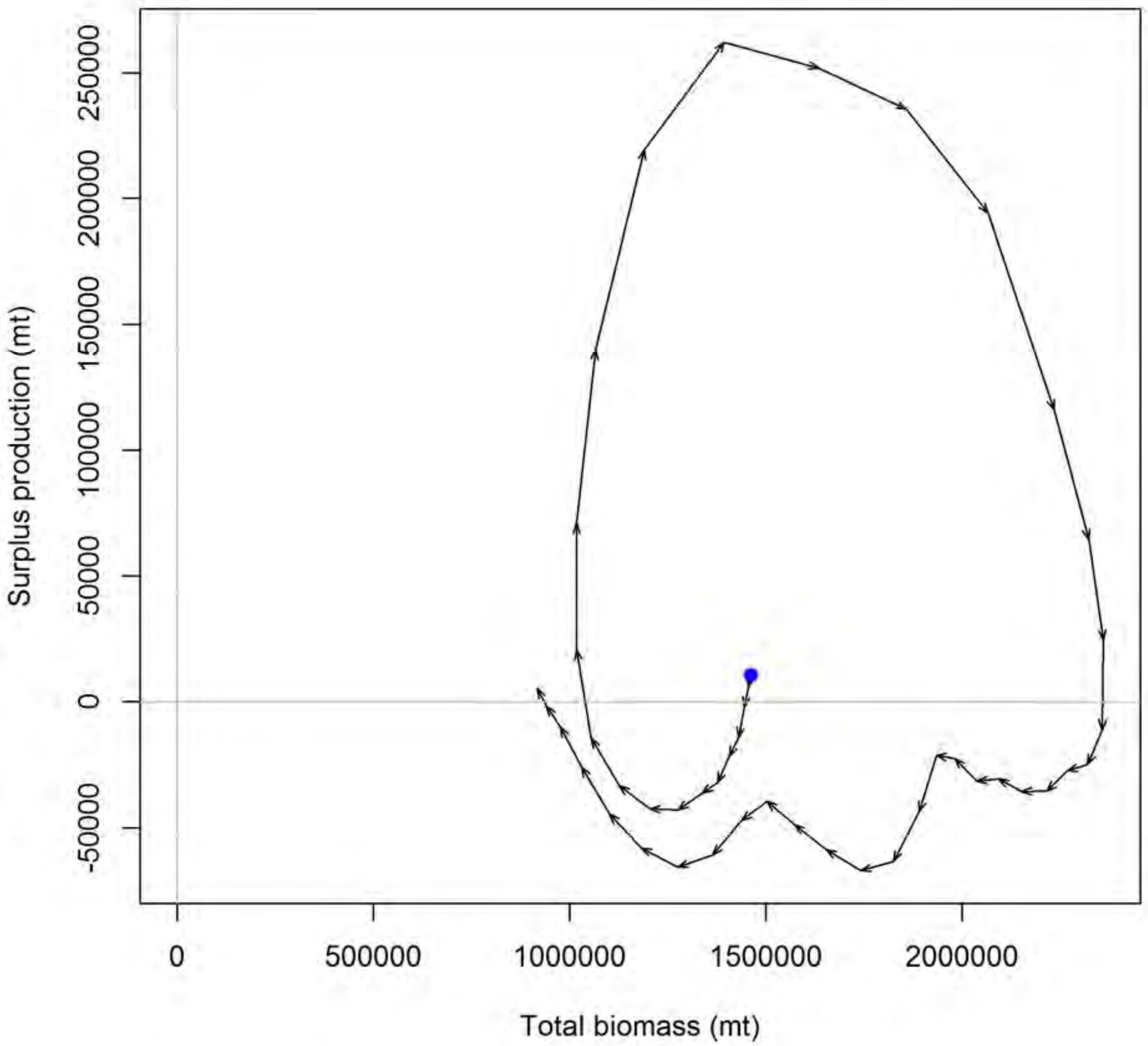

Spawning depletion with ~95% asymptotic intervals
Spawning depletion
0.0
0.5
1.0
1.5
2.0
1970
1980
1990
2000
2010
Year


Spawning depletion
Spawning depletion
0.0
0.5
1.0
1.5
1970
1980
1990
2000
2010
Year


Age-0 recruits (1,000s) with ~95% asymptotic intervals
Age-0 recruits (1,000s)
0.0e+00
4.0e+06
8.0e+06
1.2e+07
1970
1980
1990
2000
2010
Year



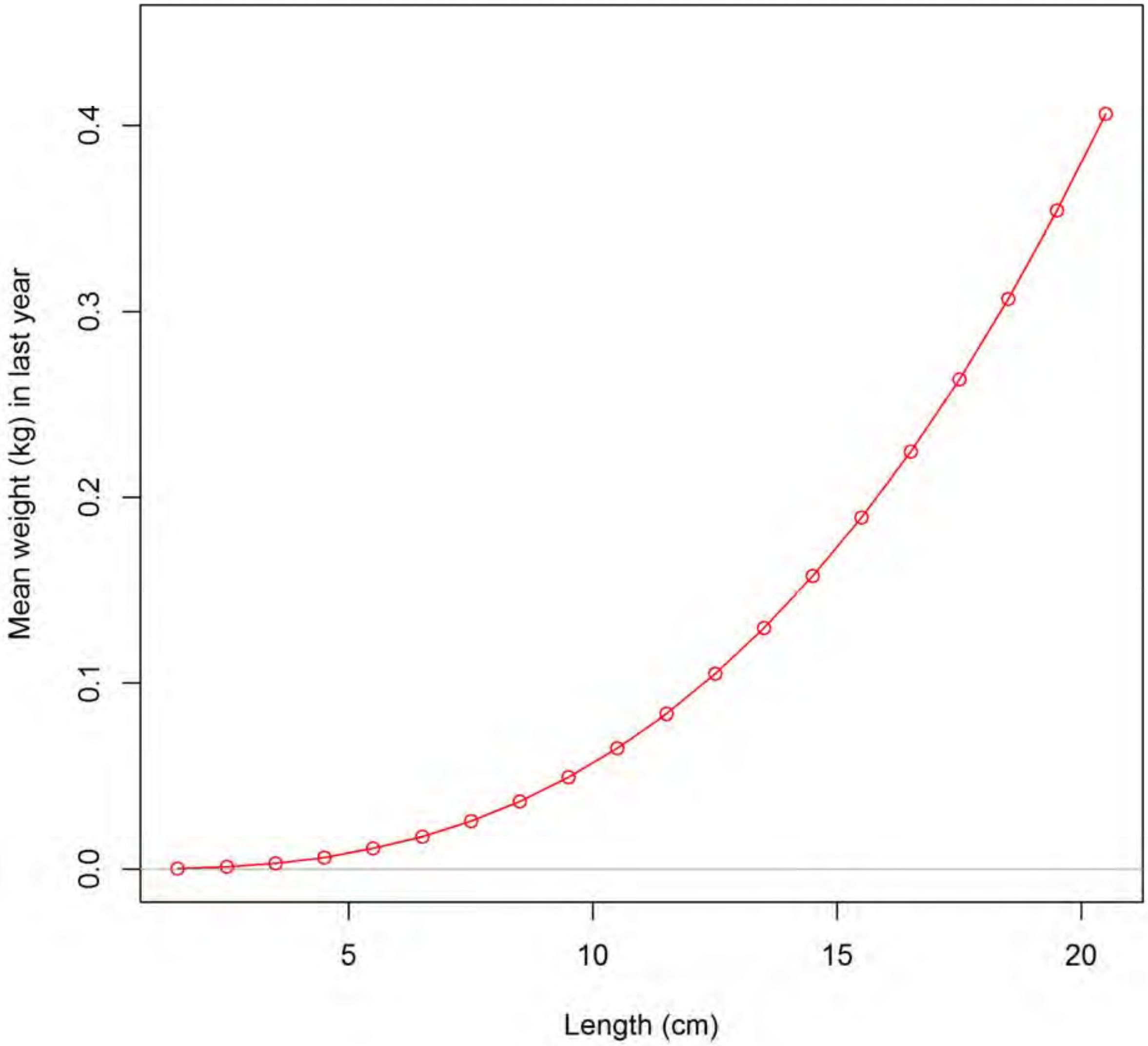
Age-0 recruits (1,000s)
Age-0 recruits (1,000s)
0e+00
2e+06
4e+06
6e+06
8e+06
1e+07
1970
1980
1990
2000
2010
Year

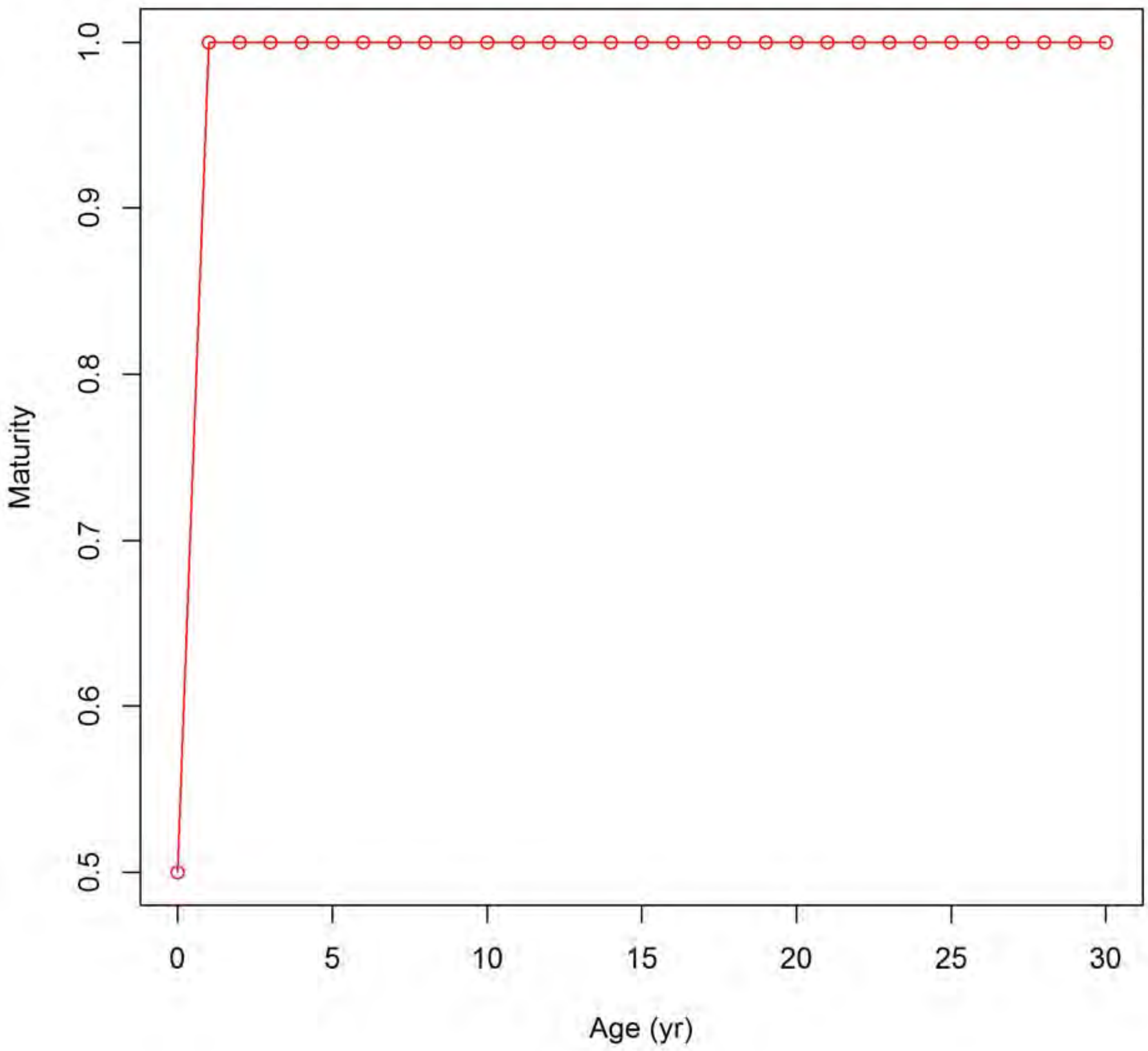


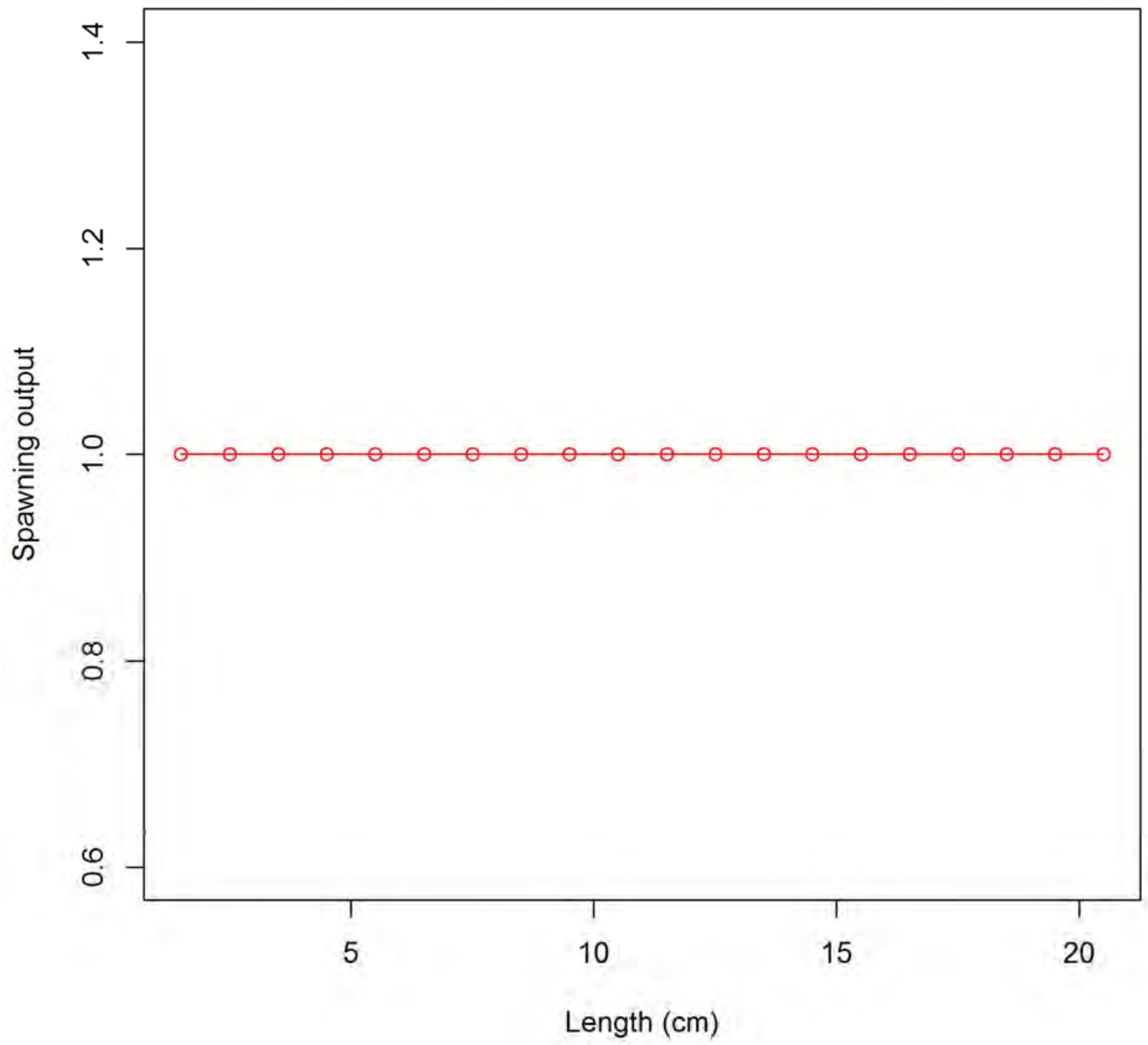
Surplus production (mt)
250000
200000
150000
100000
50000
0
-50000
0
500000
1000000
1500000
2000000
Total biomass (mt)

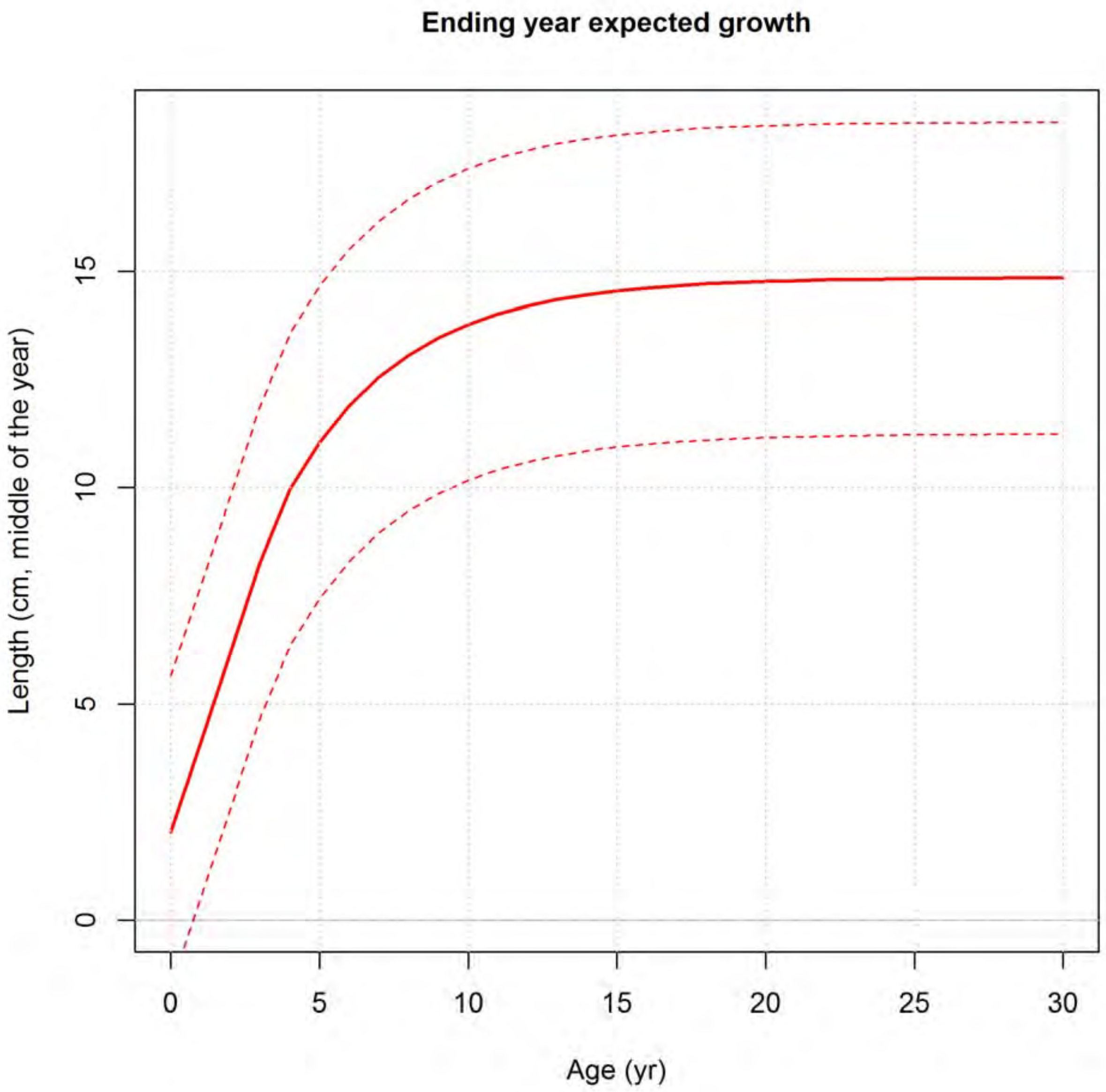
Appendix A7: SS3 Diagnostics for the GBK area

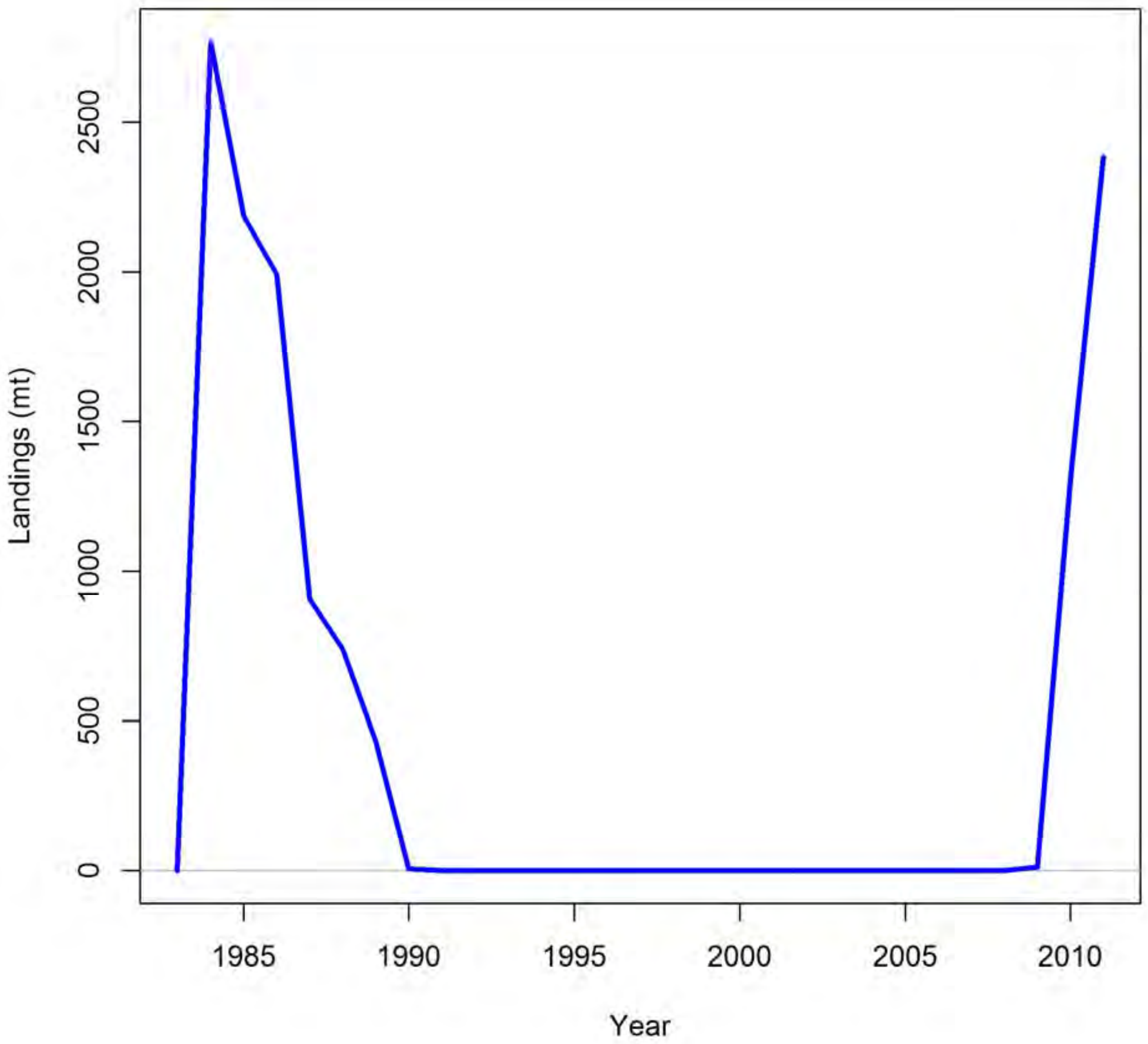
Plots created using the 'r4ss' package in R
Stock Synthesis version: SS−V3.24f
StartTime: Wed Jan 16 11:47:53 2013
Data_File: Surfclam_GBK−1.dat
Control_File: Surfclam_GBK−1.ctl

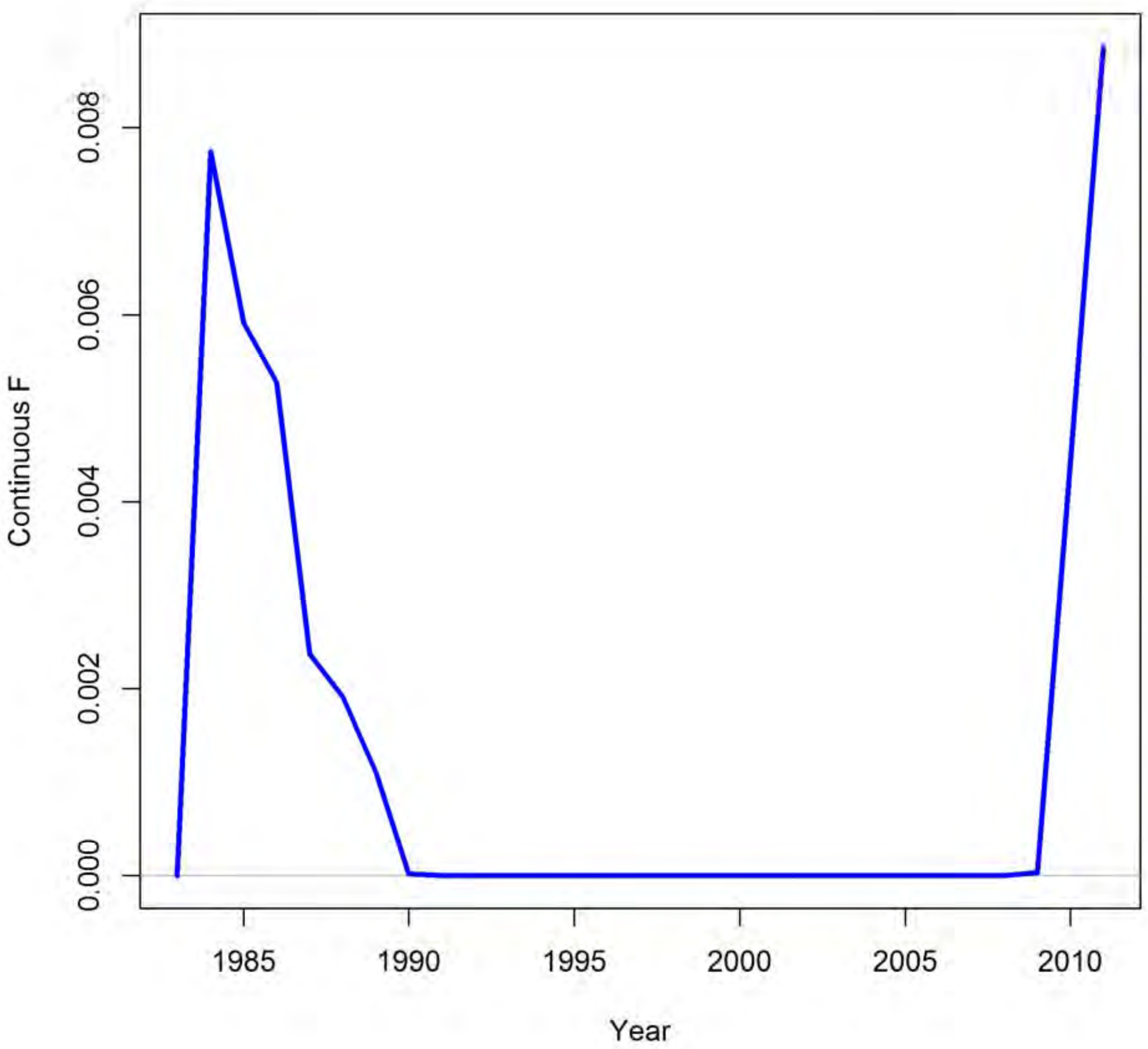

Mean weight (kg) in last year
0.0
0.1
0.2
0.3
0.4
5
10
15
20
Length (cm)

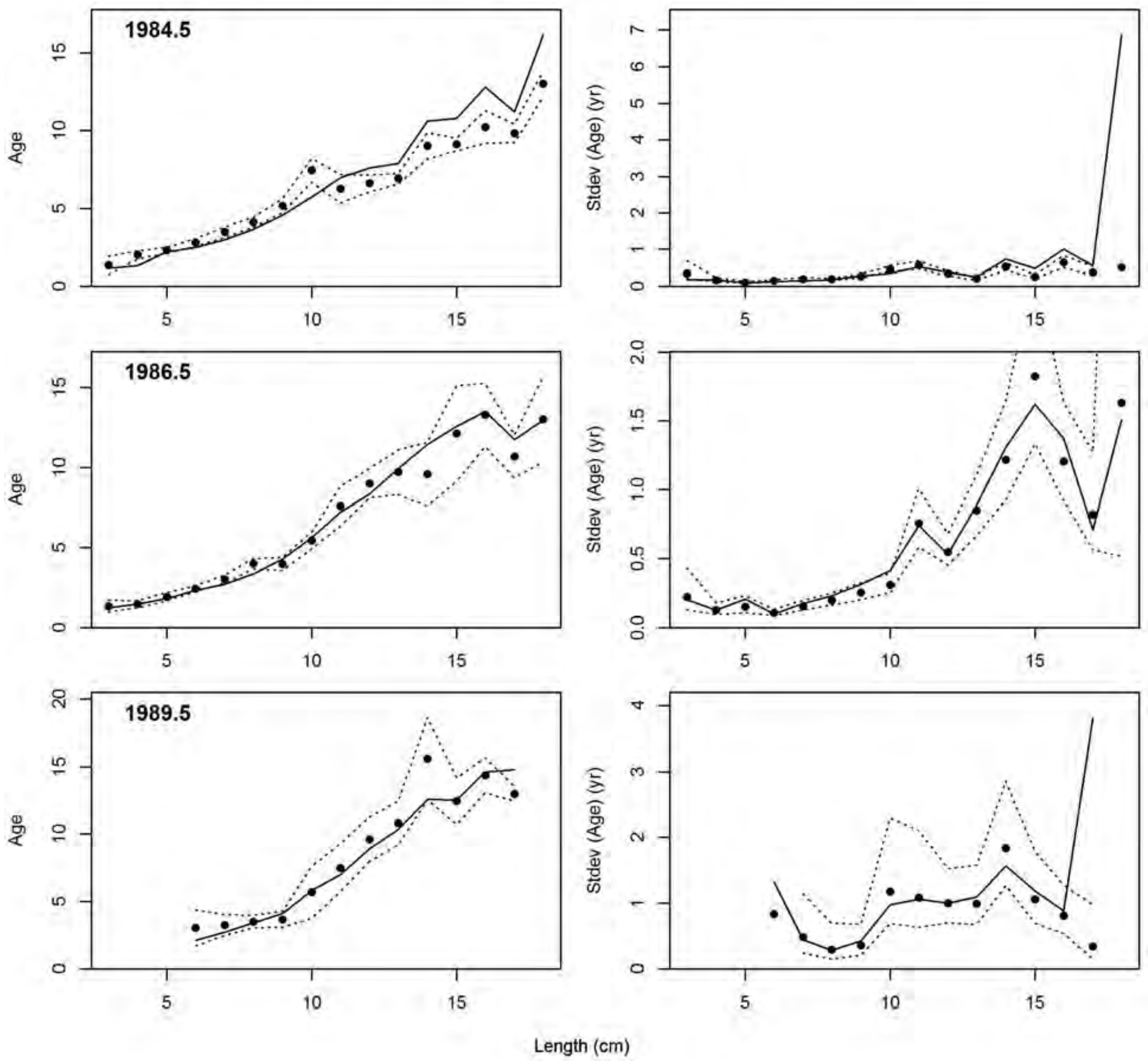

Maturity
1.0
0.9
0.8
0.7
0.6
0.5
0
5
10
15
20
25
30
Age (yr)


Spawning output
1.4
1.2
1.0
0.8
0.6
5
10
15
20
Length (cm)

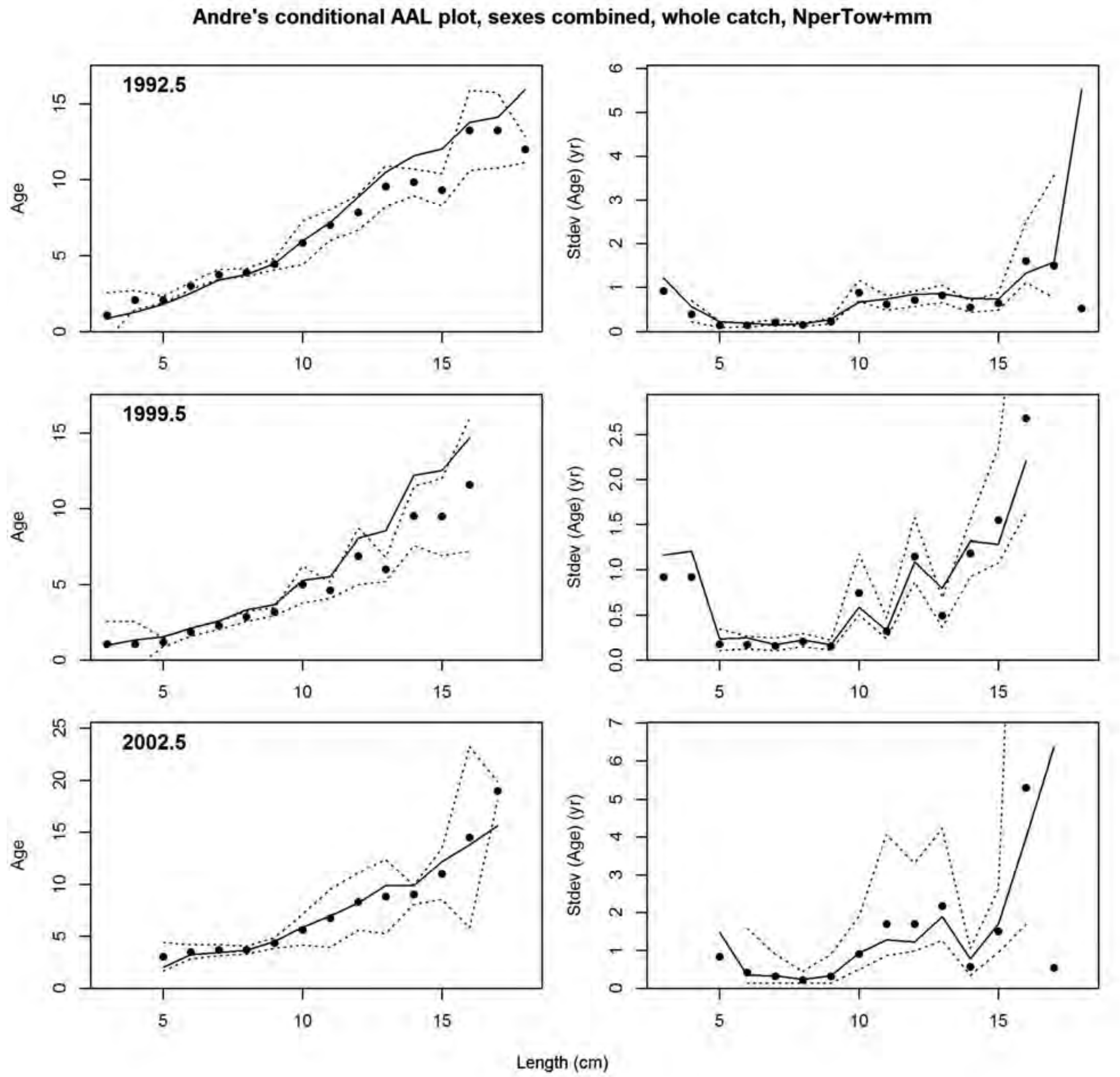

Ending year expected growth
Length (cm, middle of the year)
0
5
10
15
0
5
10
15
20
25
30
Age (yr)


Landings (mt)
0
500
1000
1500
2000
2500
1985
1990
1995
2000
2005
2010
Year

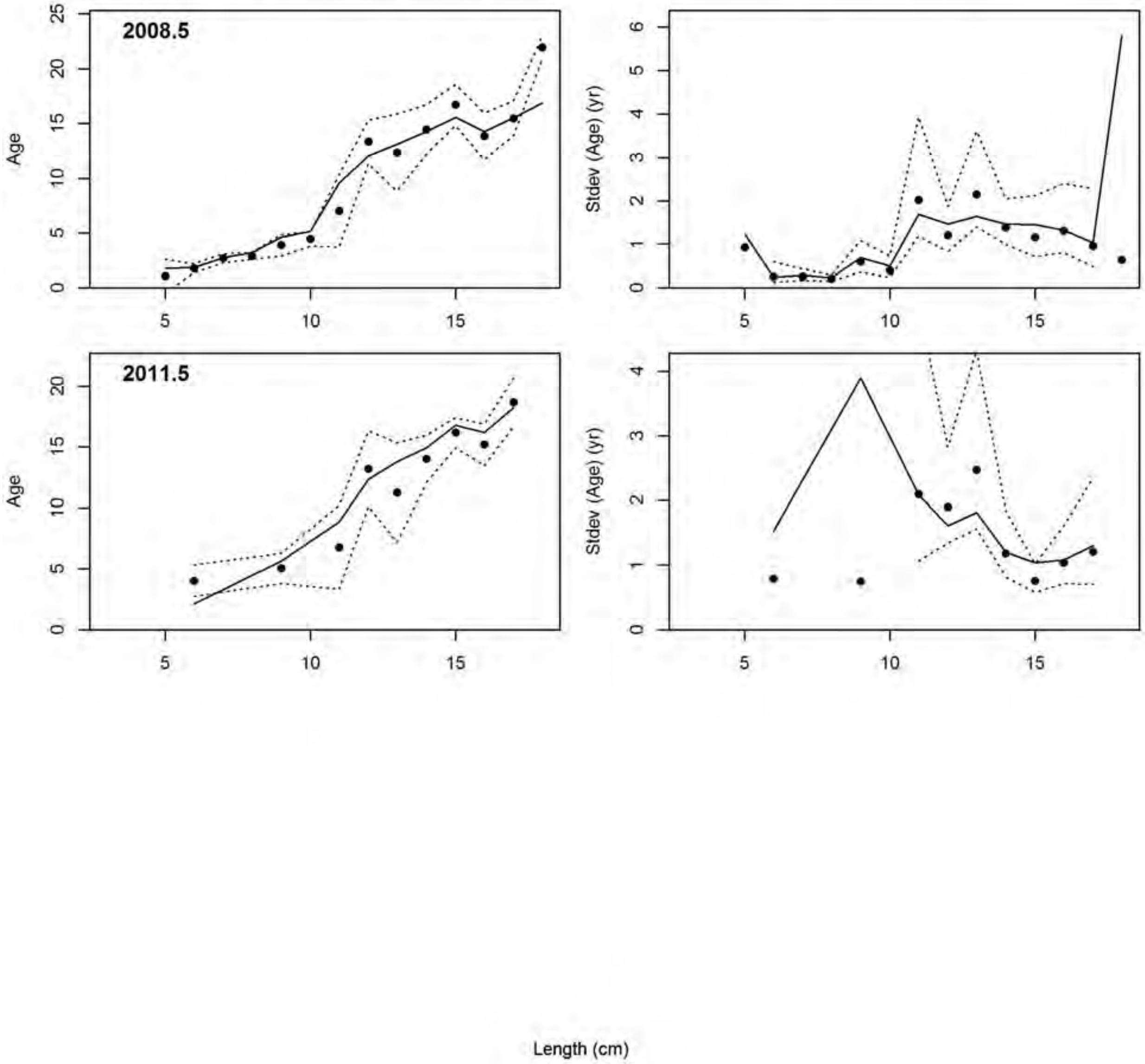

Continuous F
0.000
0.002
0.004
0.006
0.008
1985
1990
1995
2000
2005
2010
Year



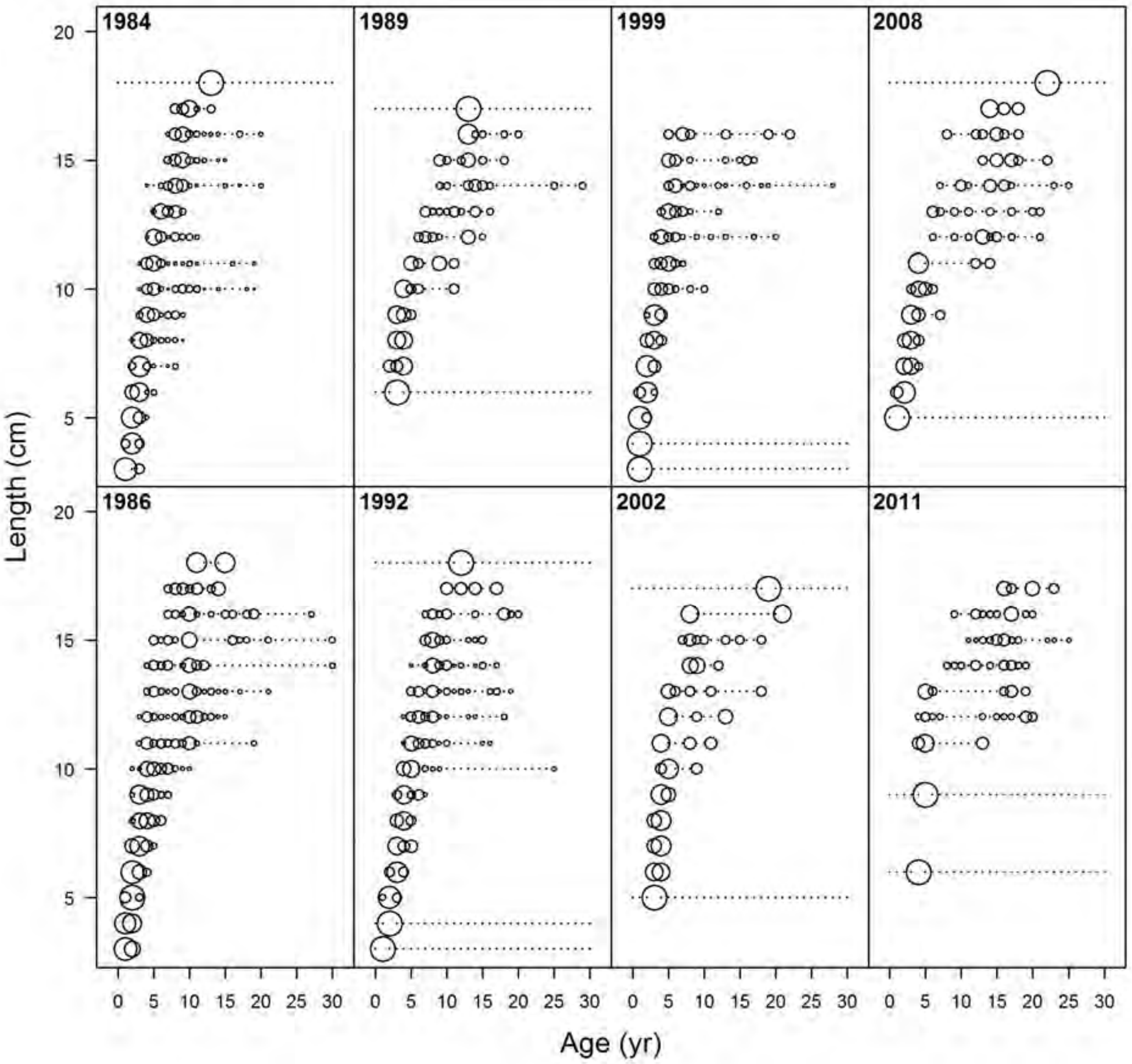
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm


Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm
1992.5
1999.5
2002.5
Age
Stdev (Age) (yr)
Length (cm)
0
5
10
15
20
25
0.0
0.5
1.0
1.5
2.0
2.5
1
2
3
4
5
6
7

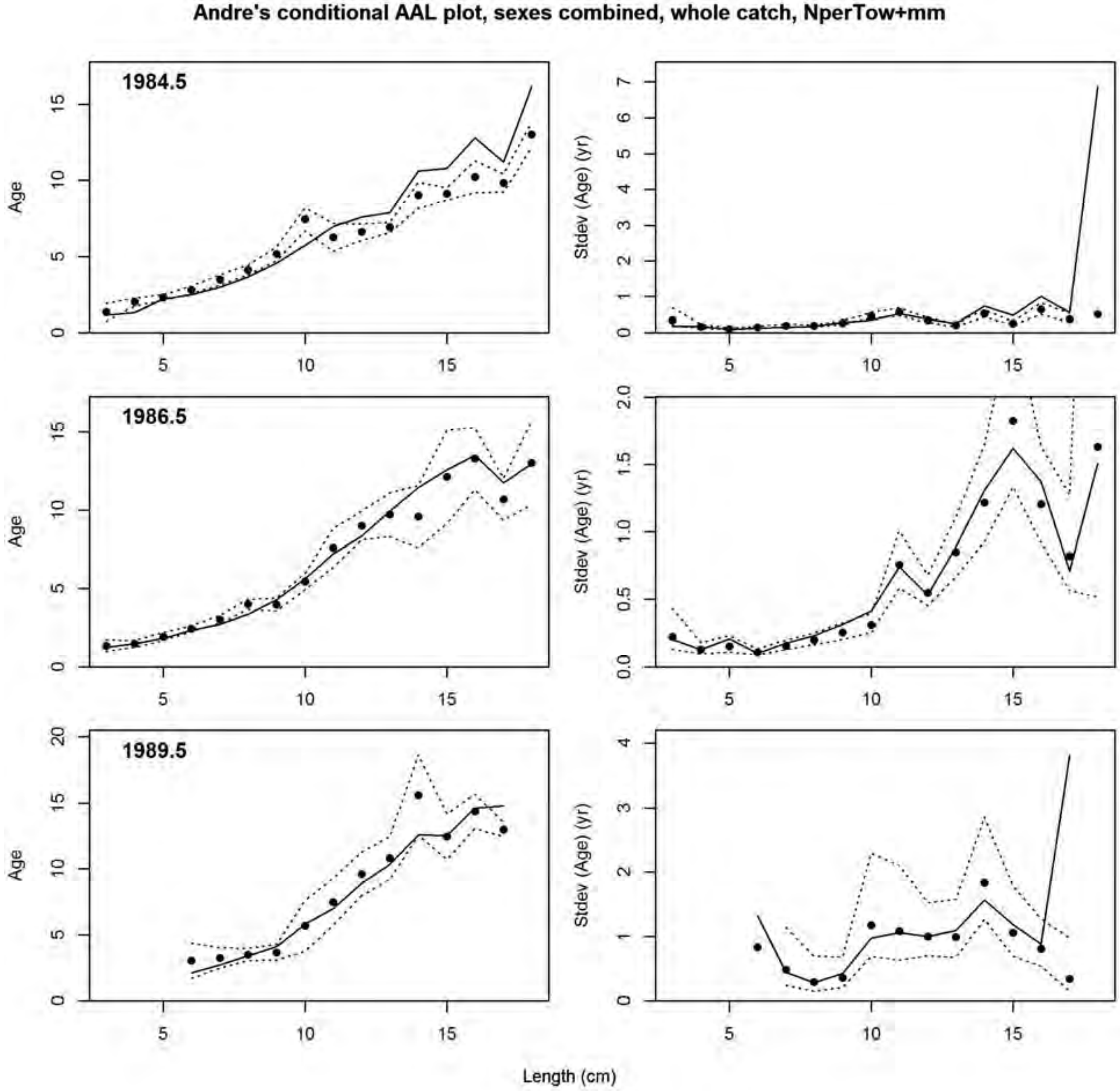
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm



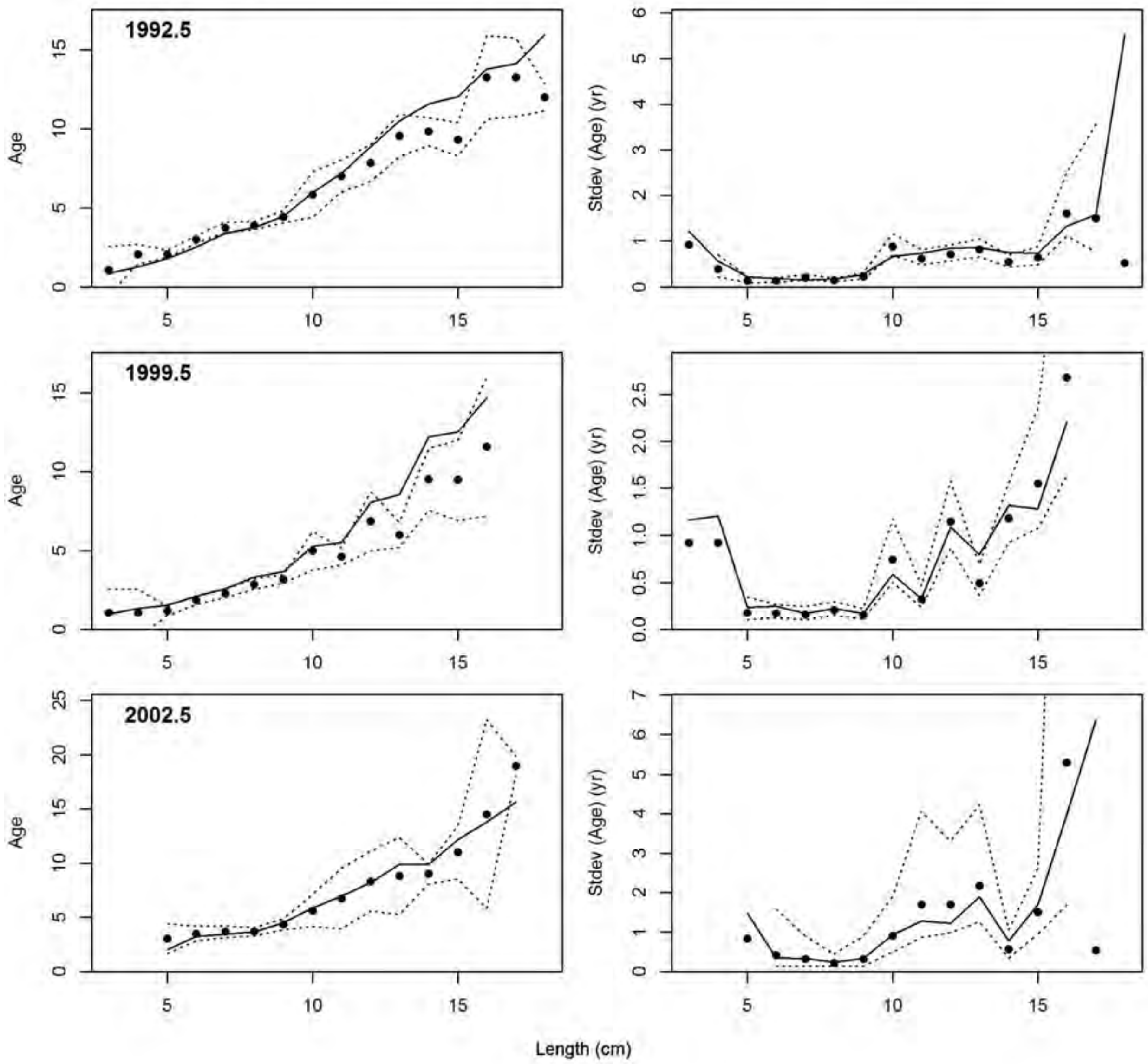
conditional age-at-length data, sexes combined, whole catch, NperTow+mm (max=1

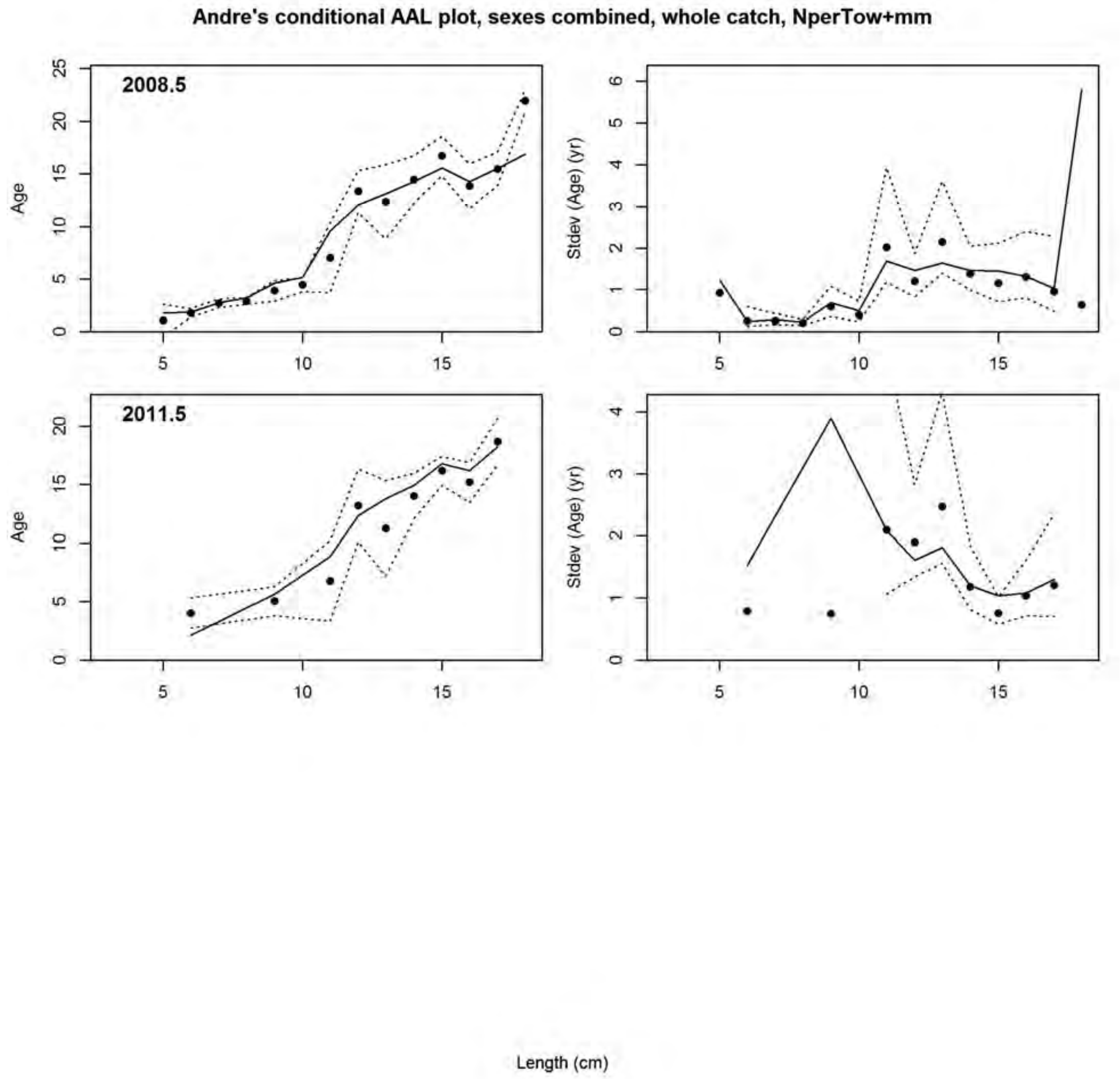


Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm

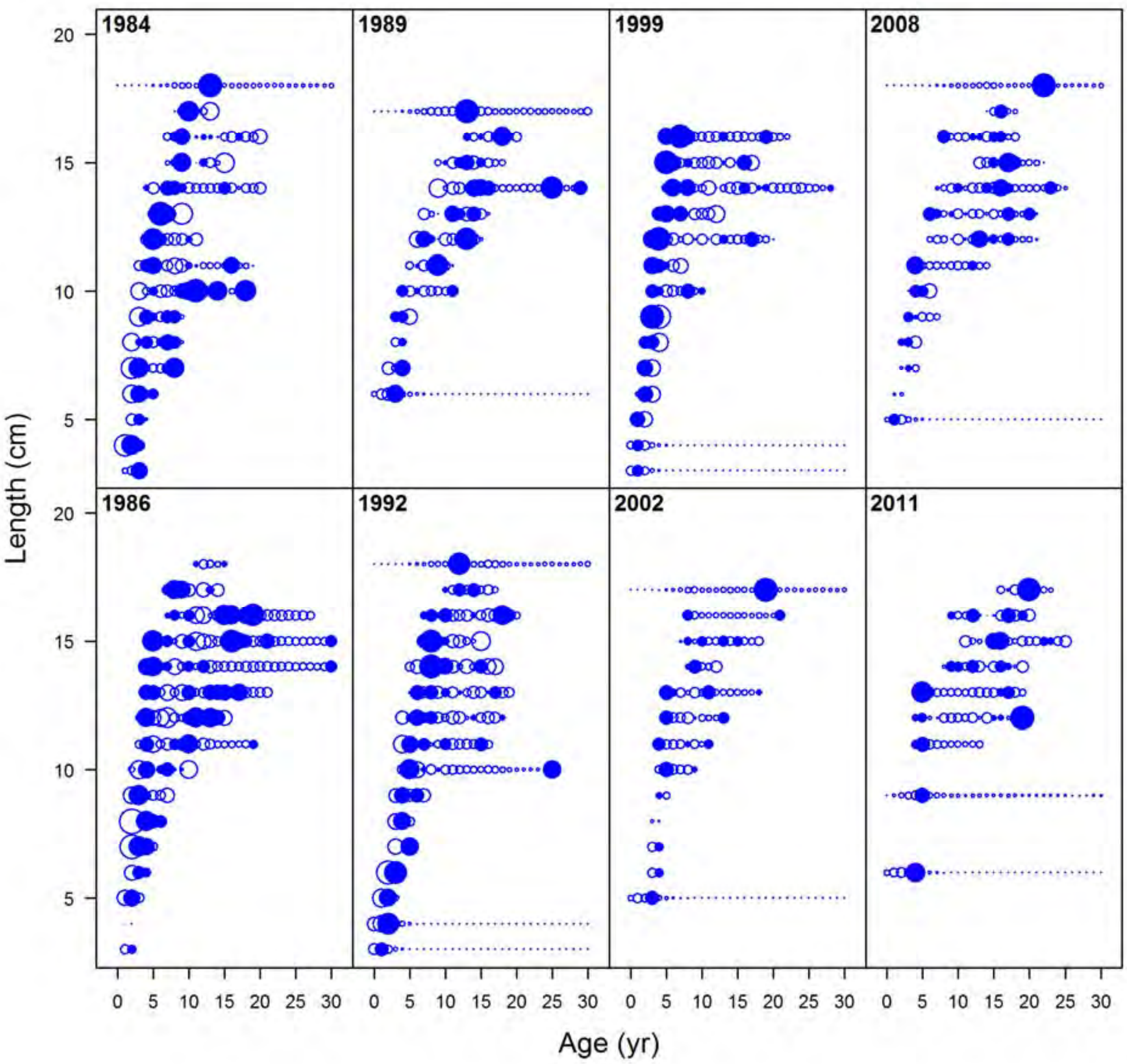


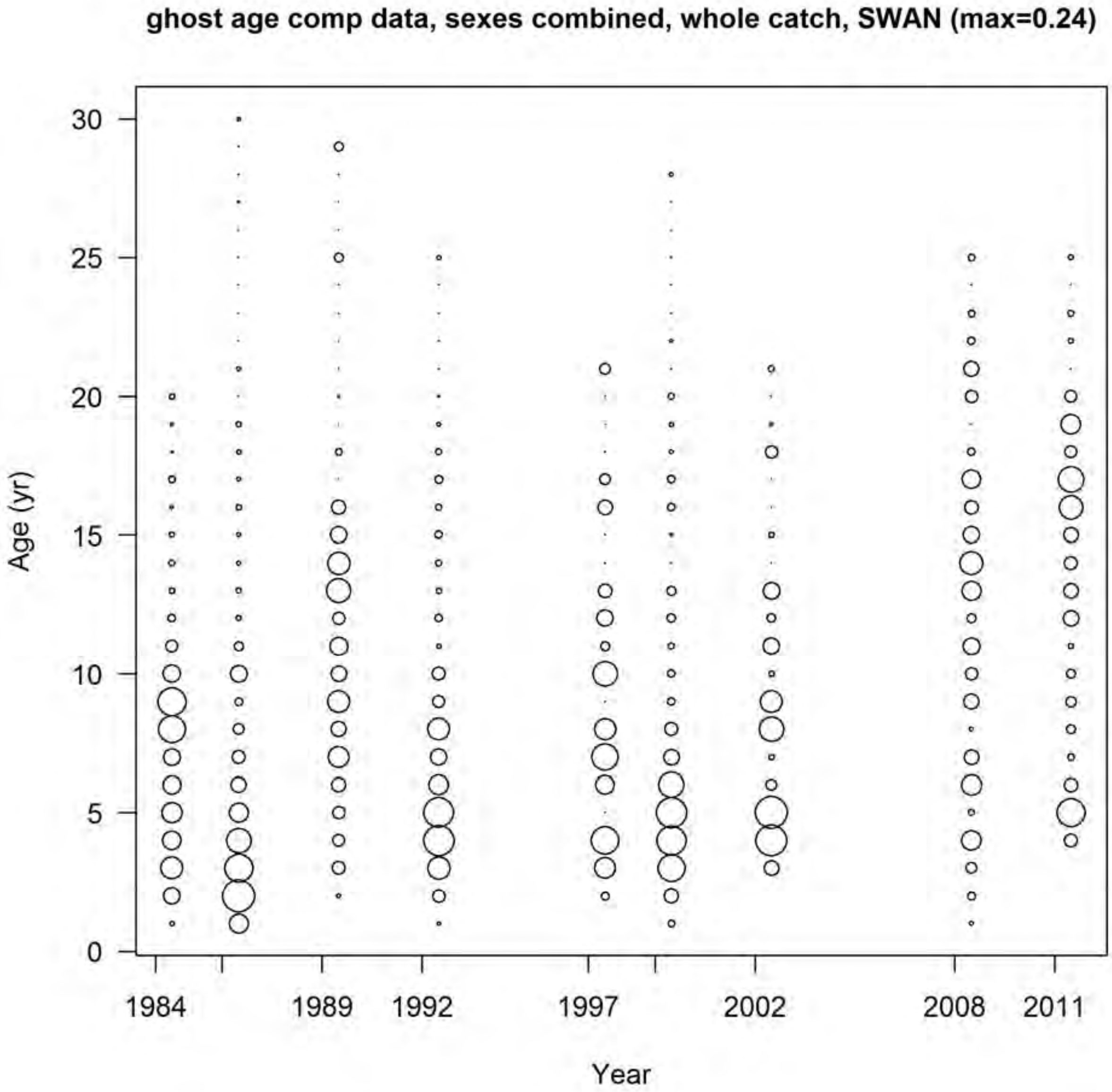
Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm

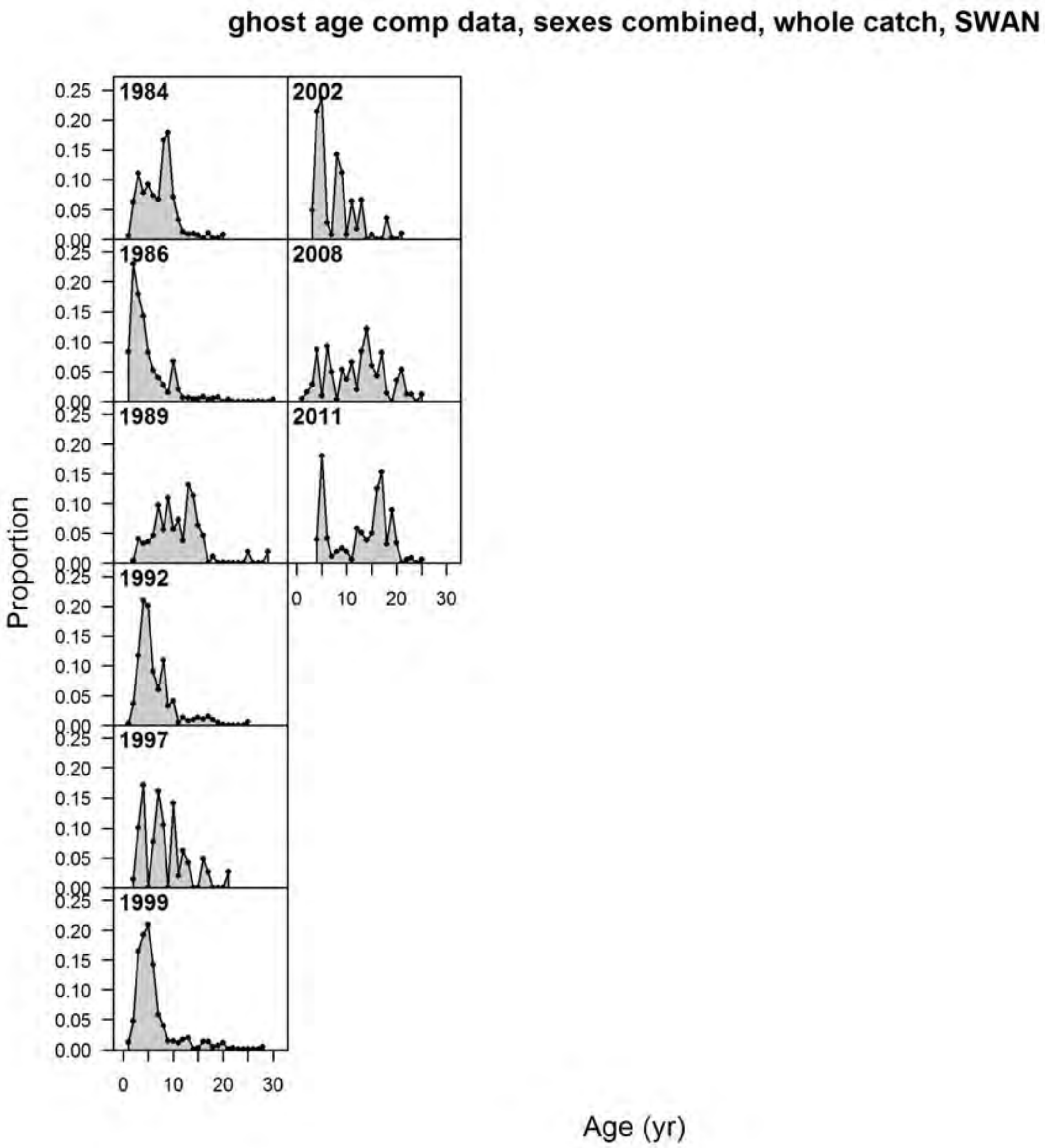



Andre's conditional AAL plot, sexes combined, whole catch, NperTow+mm
2008.5
2011.5
Age
Stdev (Age) (yr)
0
5
10
15
20
25
1
2
3
4
6
Length (cm)

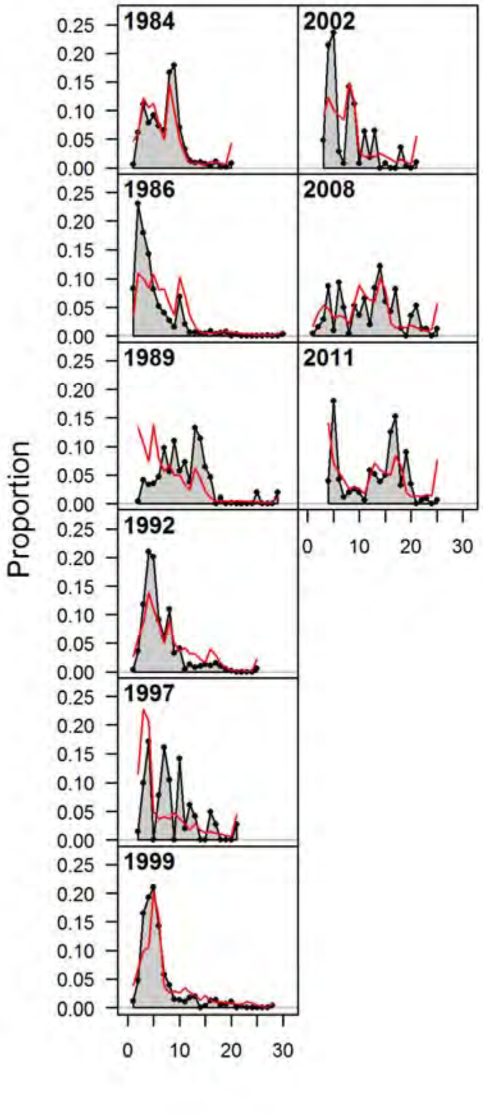
Pearson residuals, sexes combined, whole catch, NperTow+mm (max=6.03)

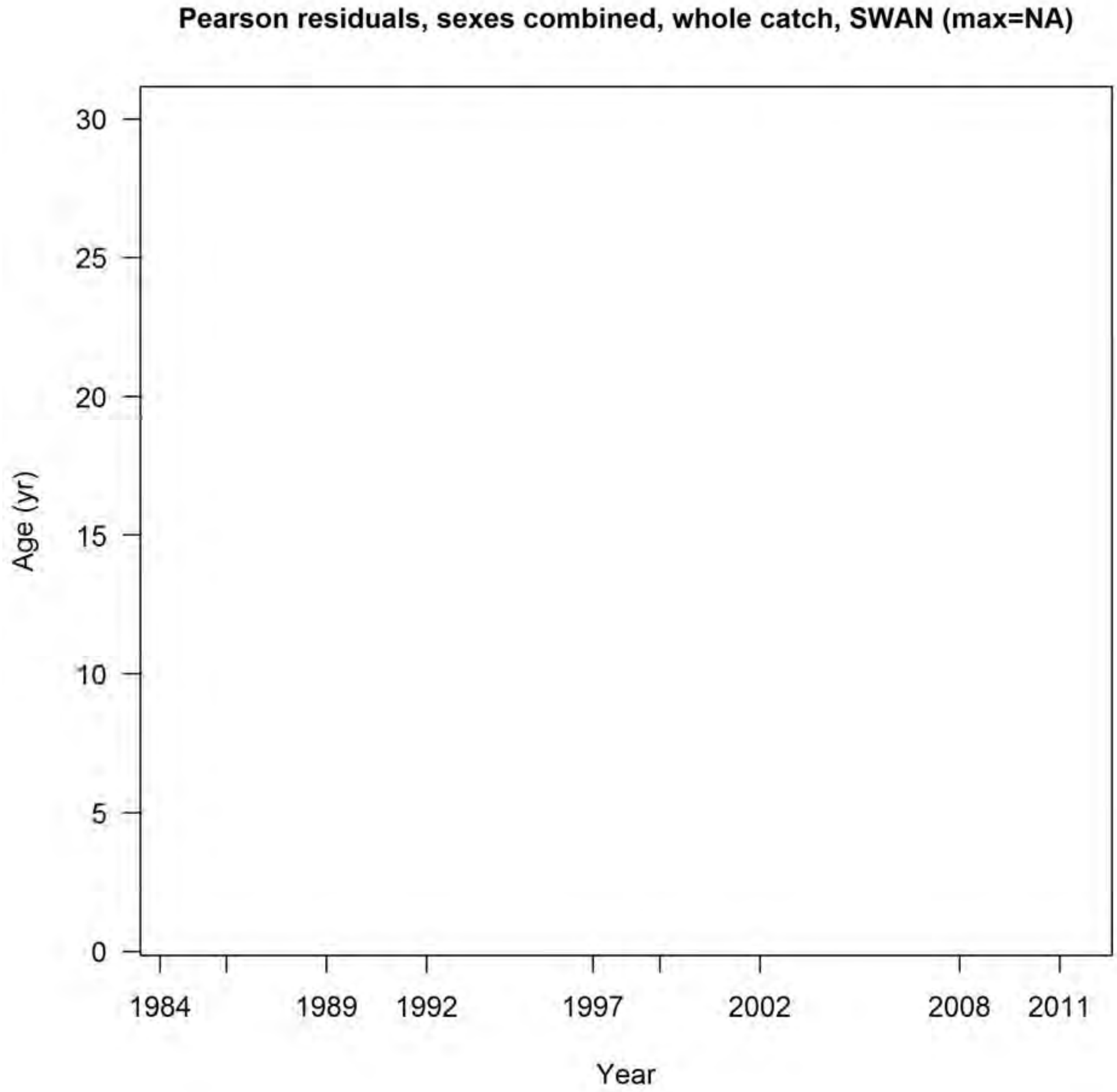


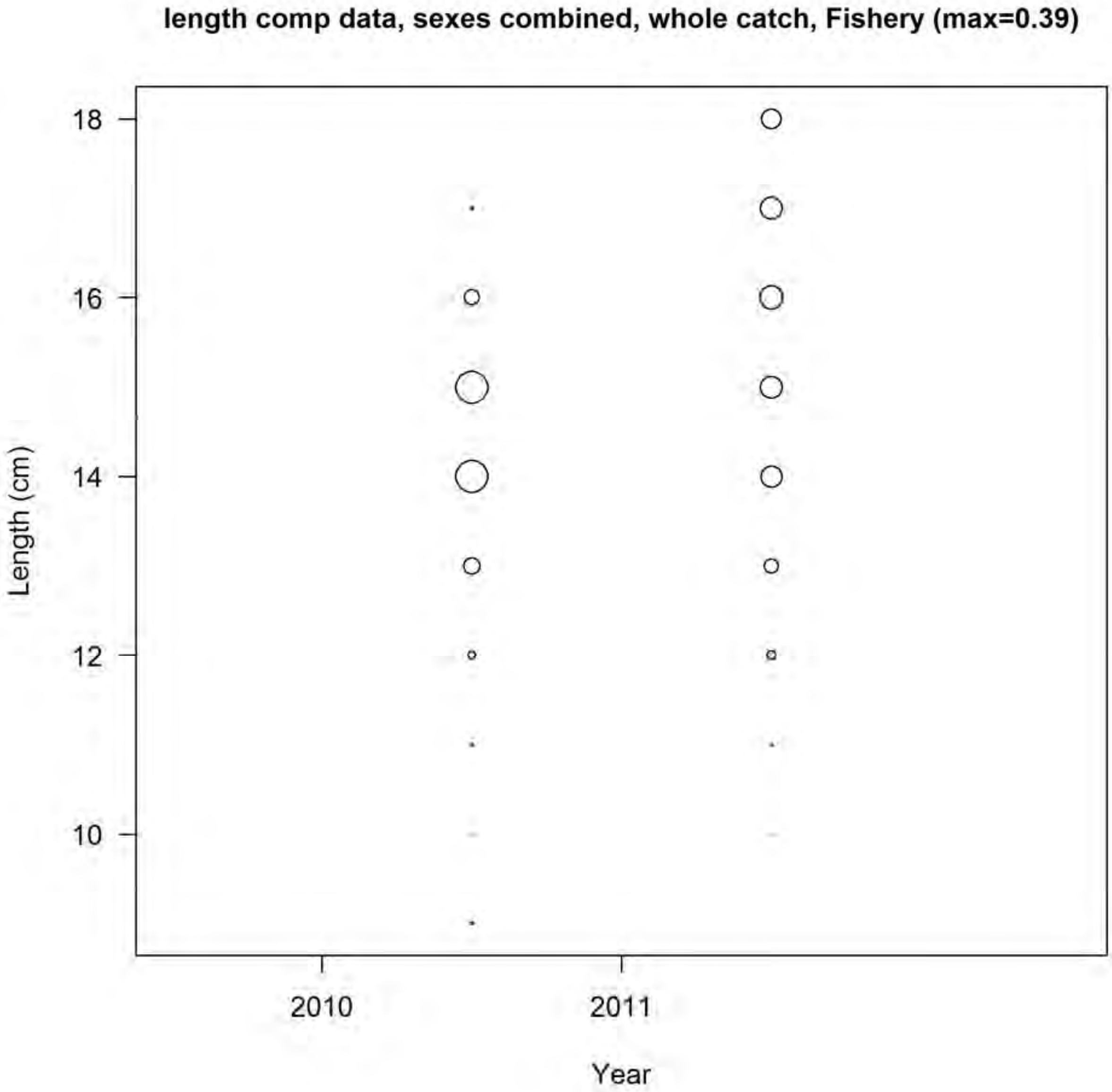

ghost age comp data, sexes combined, whole catch, SWAN (max=0.24)
Age (yr)
0
5
10
15
20
25
30
1984
1989
1992
1997
2002
2008
2011
Year

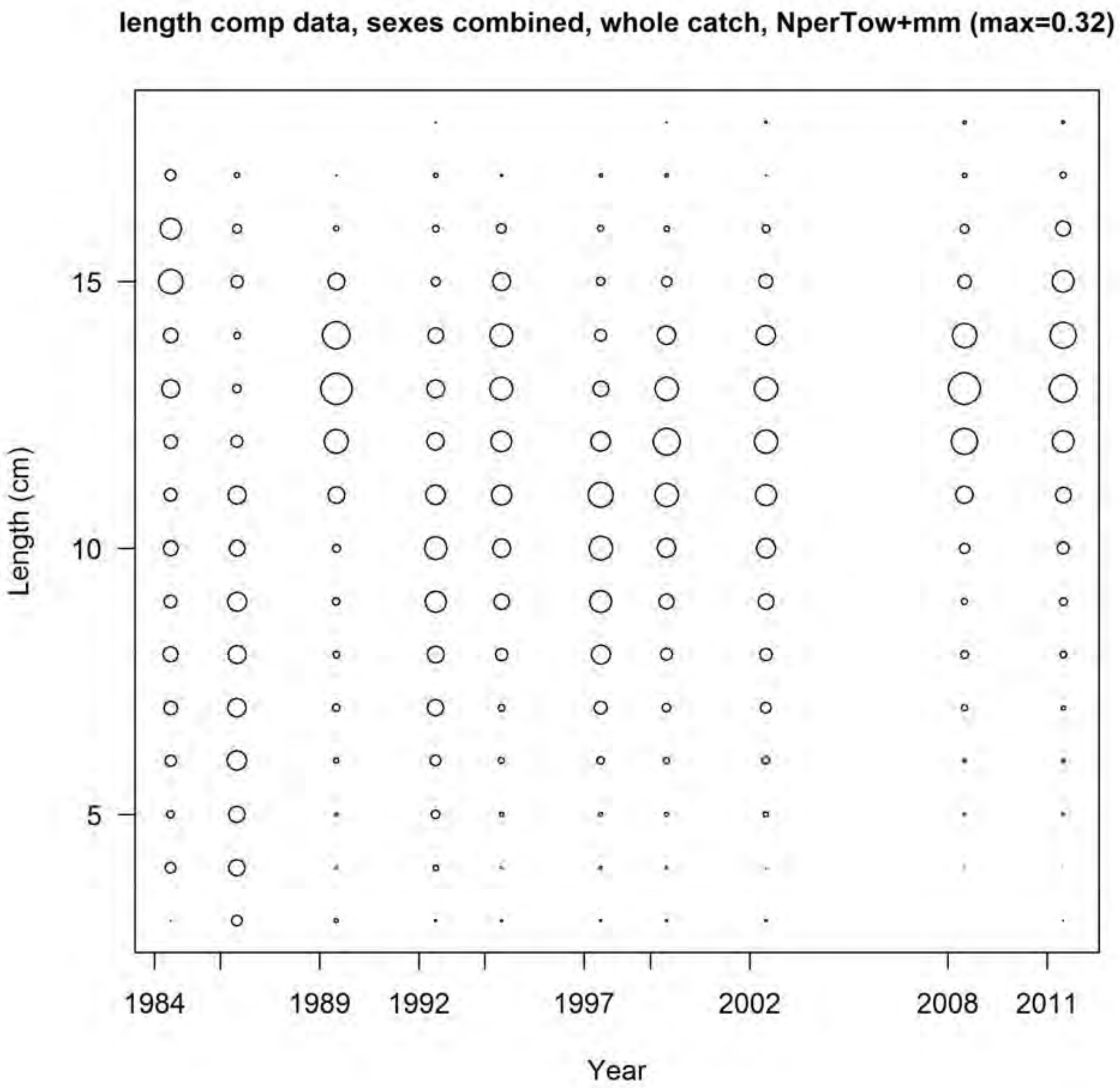

ghost age comp data, sexes combined, whole catch, SWAN
1984
2002
1986
2008
1989
2011
1992
1997
1999
Proportion
Age (yr)
0.00
0.05
0.10
0.15
0.20
0.25
0
10
20
30

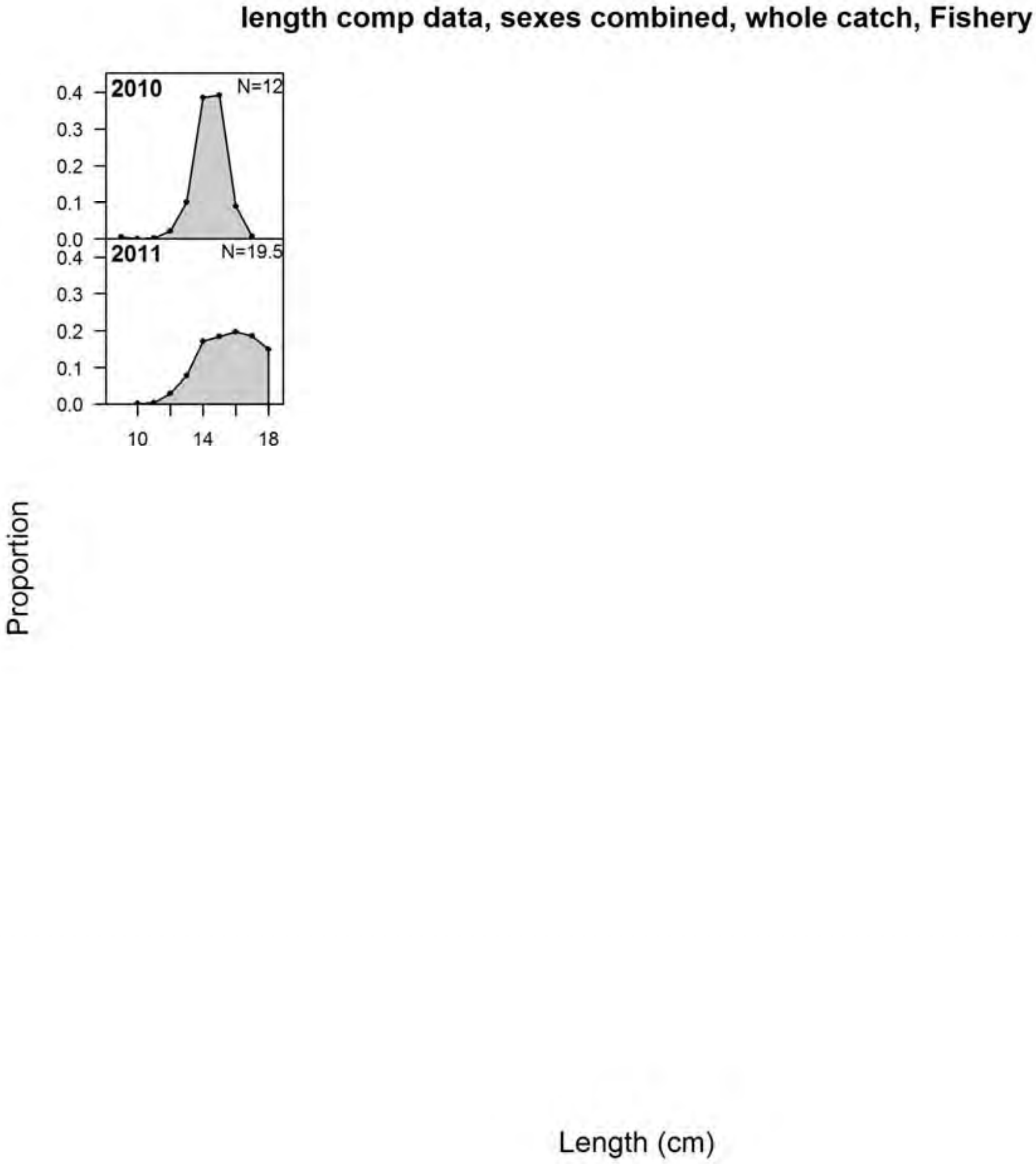
ghost age comps, sexes combined, whole catch, SWAN

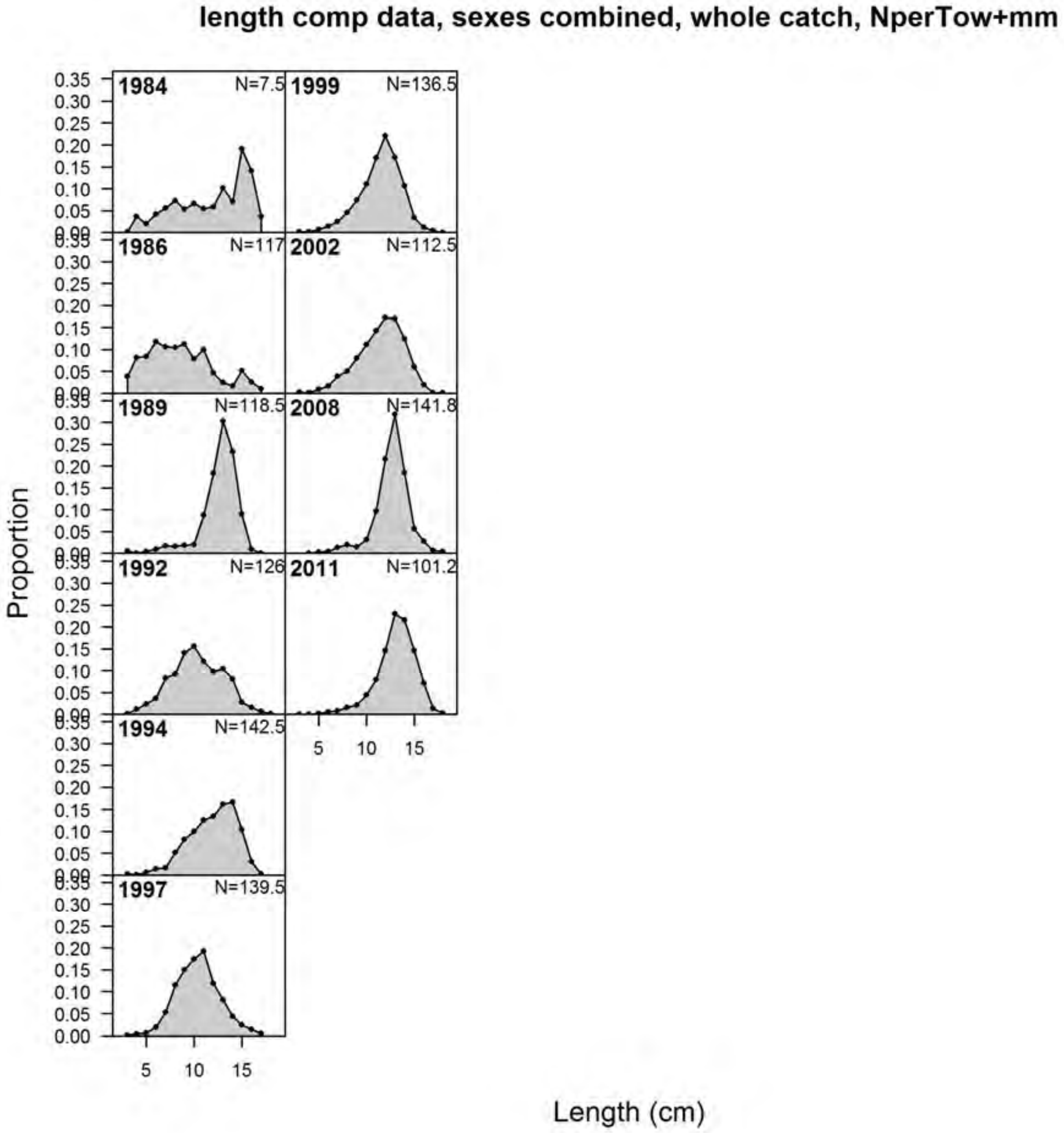


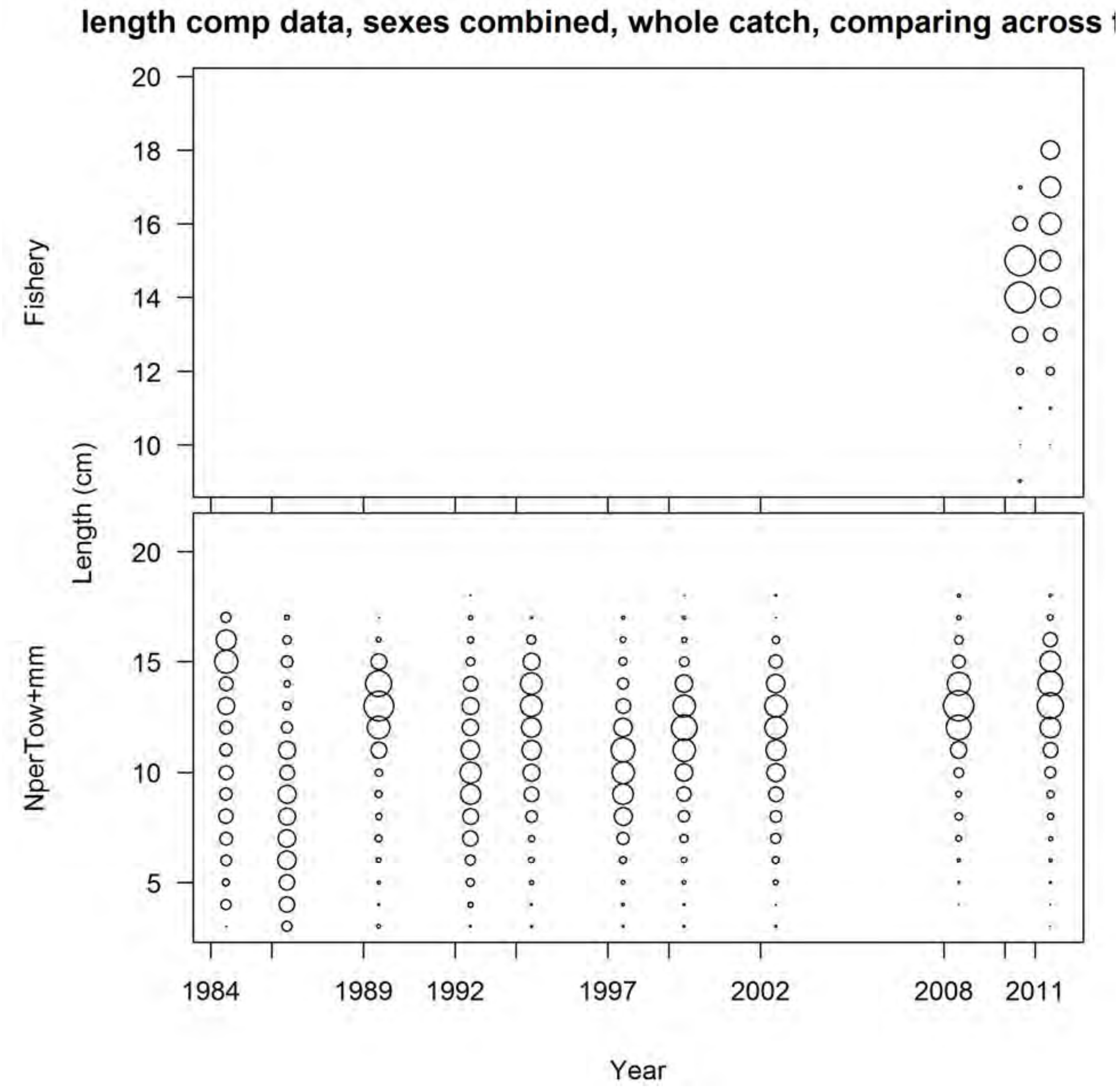

Pearson residuals, sexes combined, whole catch, SWAN (max=NA)
30
25
20
15
10
5
0
Age (yr)
1984
1989
1992
1997
2002
2008
2011
Year

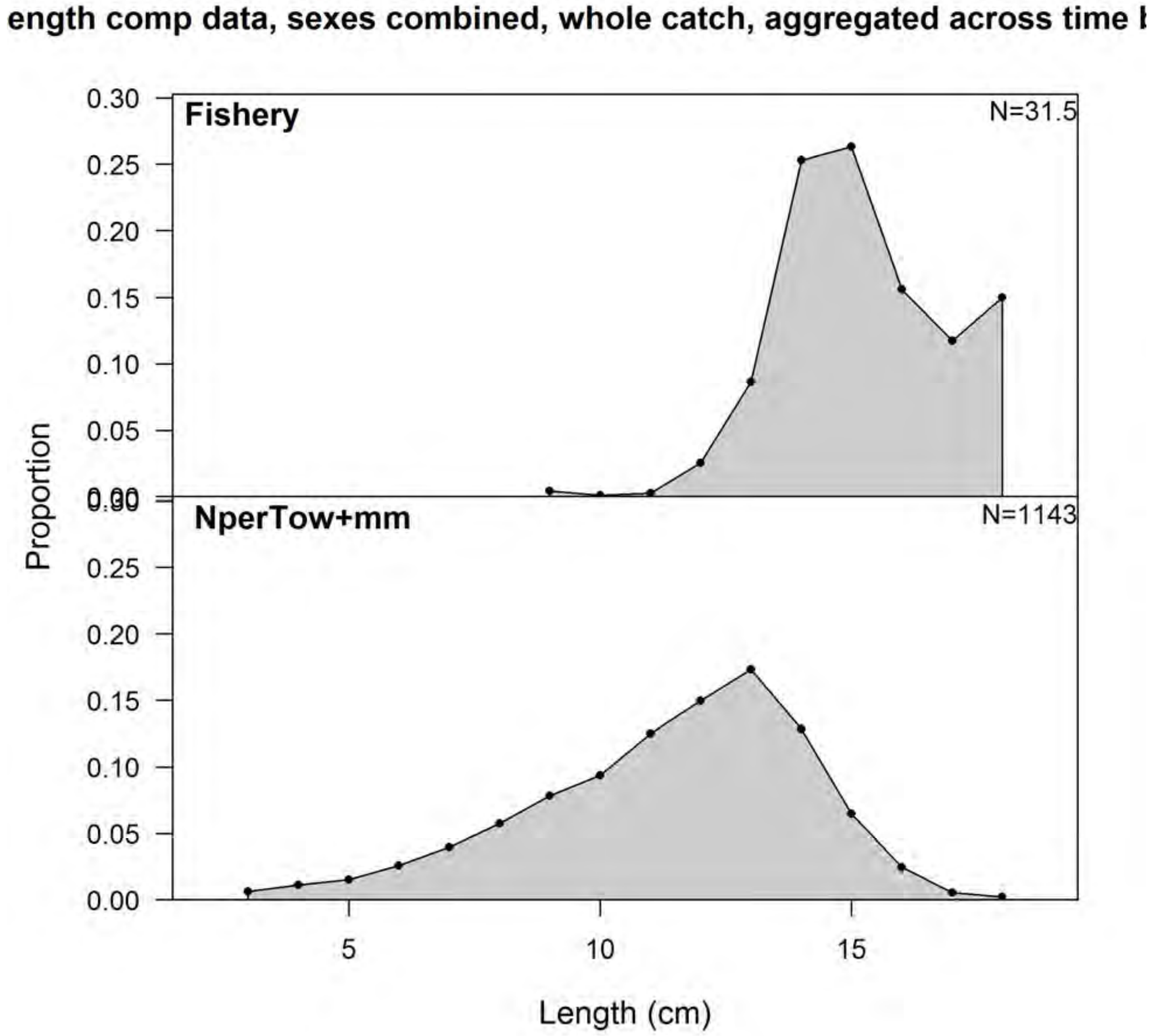

length comp data, sexes combined, whole catch, Fishery (max=0.39)
Length (cm)
18
16
14
12
10
2010
2011
Year

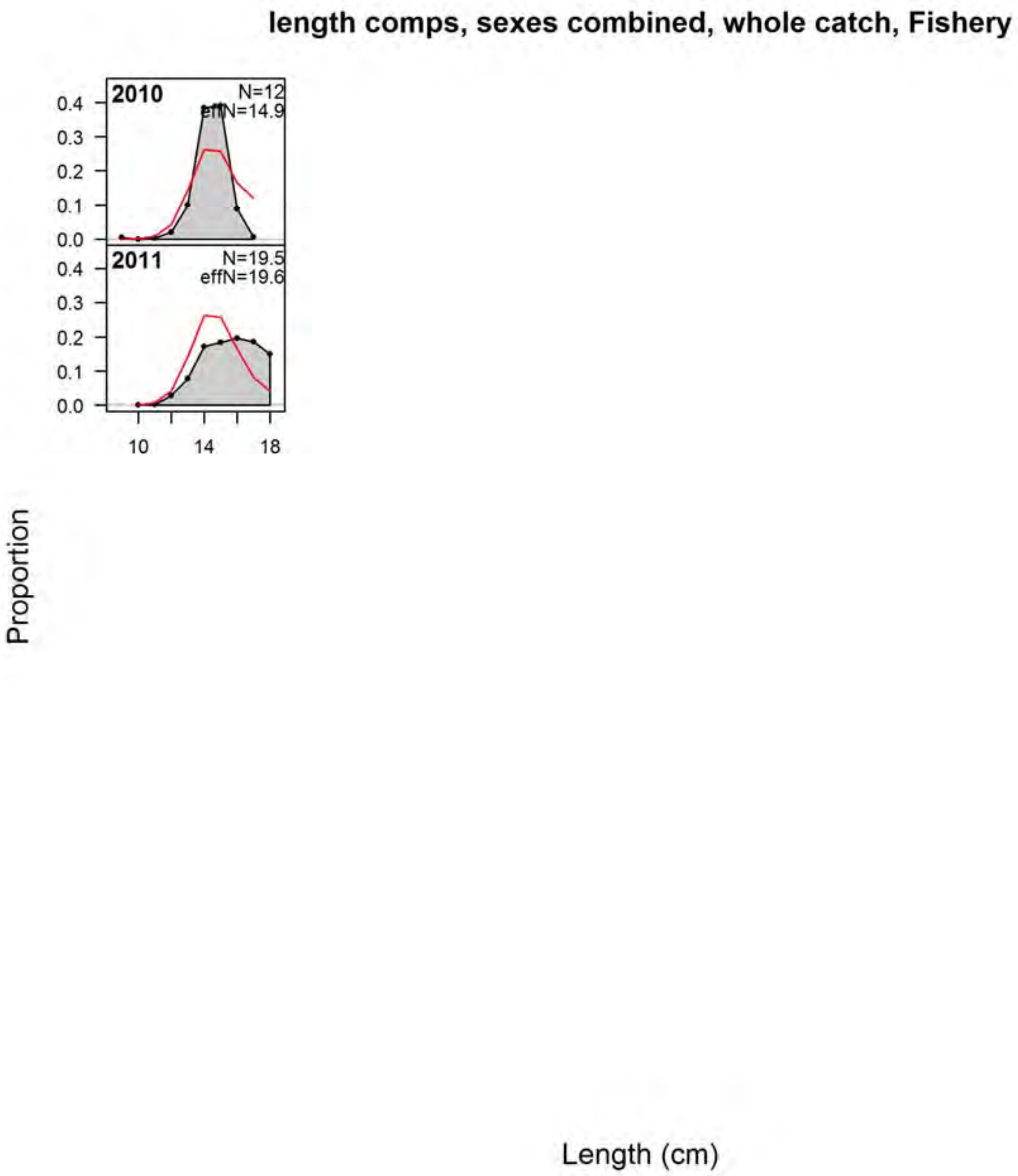

length comp data, sexes combined, whole catch, NperTow+mm (max=0.32)
Length (cm)
15
10
5
1984
1989
1992
1997
2002
2008
2011
Year

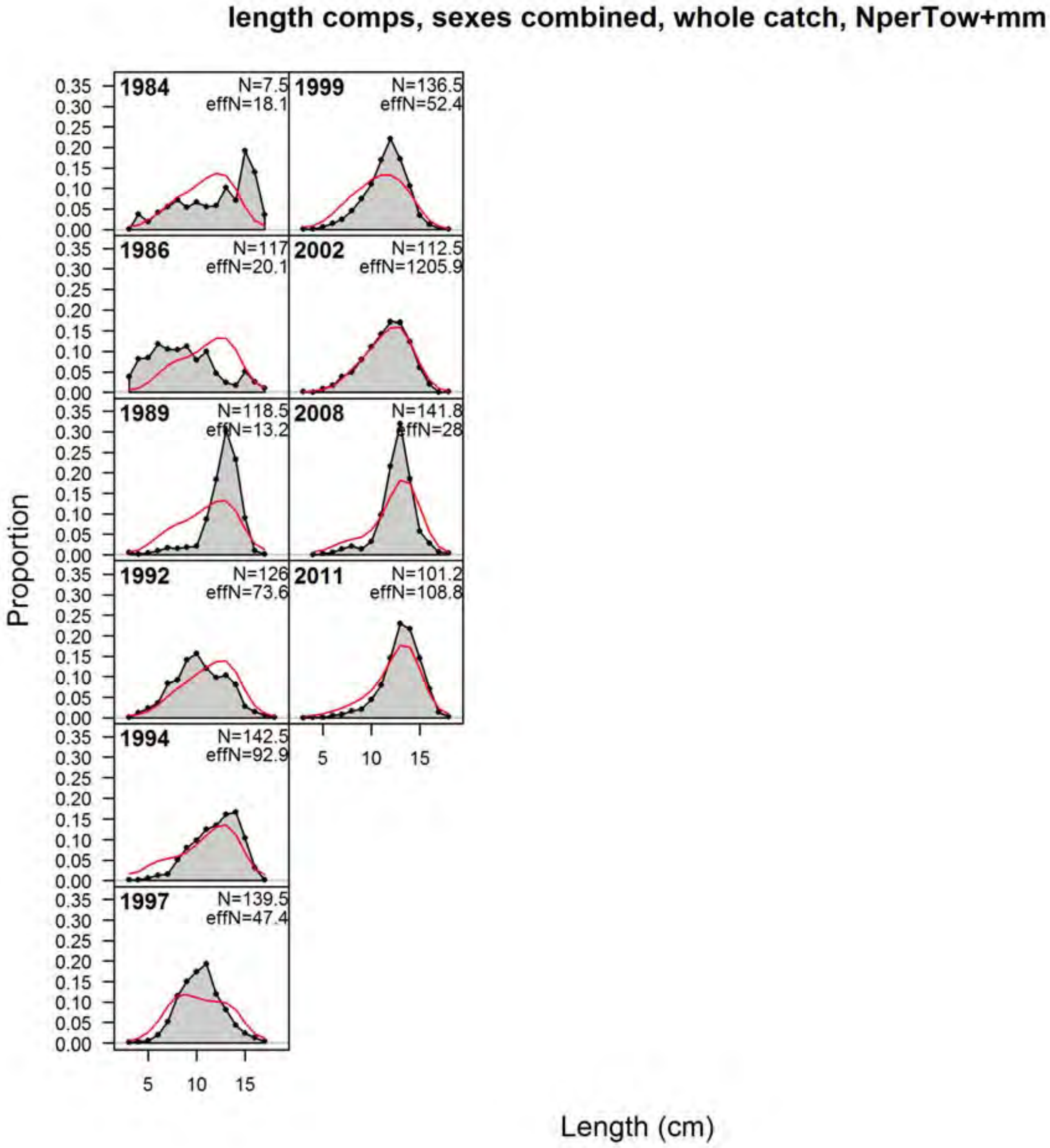

length comp data, sexes combined, whole catch, Fishery
2010
N=12
2011
N=19.5
0.0
0.1
0.2
0.3
0.4
10
14
18
Proportion
Length (cm)

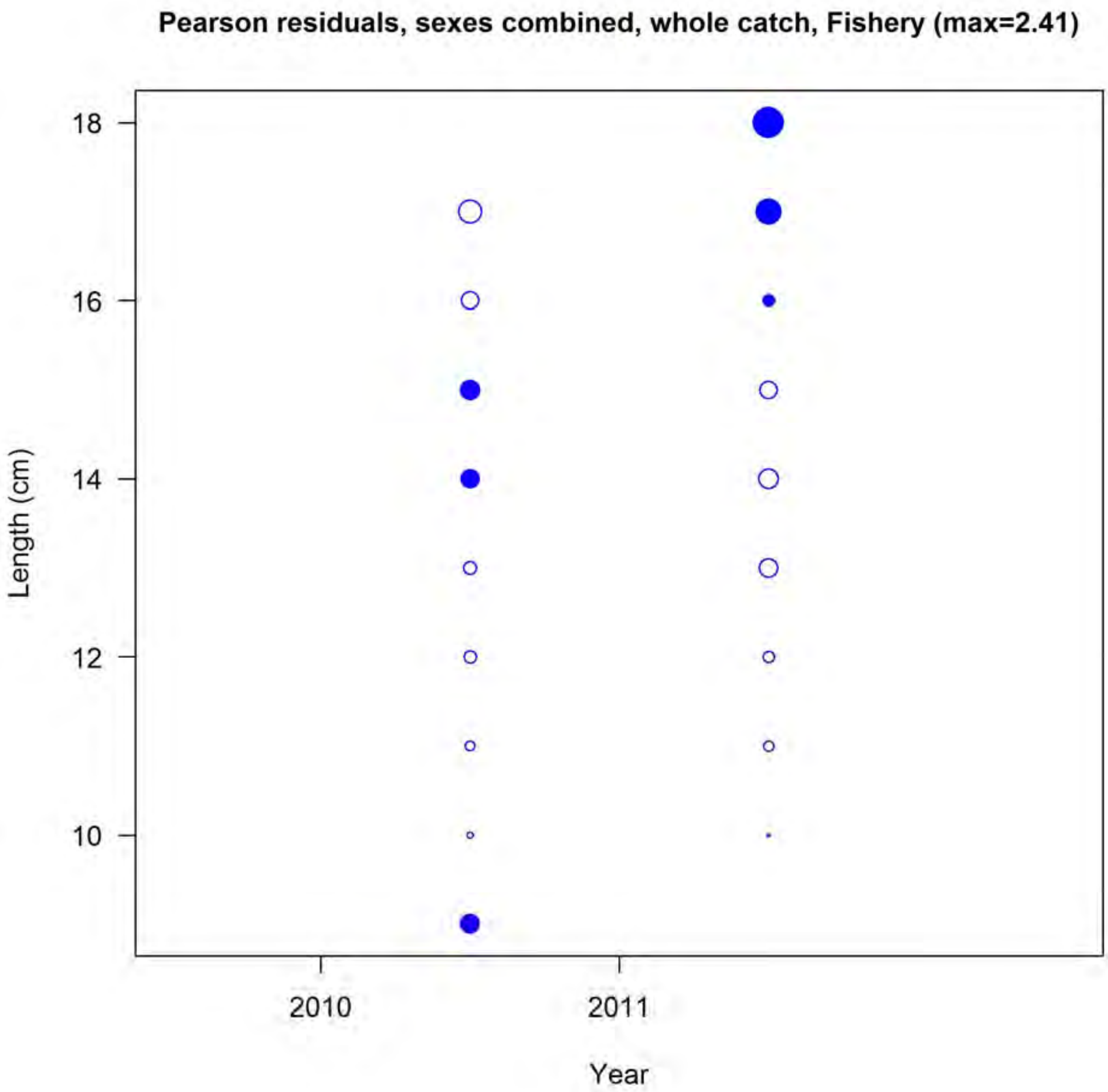

length comp data, sexes combined, whole catch, NperTow+mm
1984
N=7.5
1999
N=136.5
1986
N=117
2002
N=112.5
1989
N=118.5
2008
N=141.8
1992
N=126
2011
N=101.2
1994
N=142.5
1997
N=139.5
Proportion
0.00
0.05
0.10
0.15
0.20
0.25
0.30
0.35
5
10
15
Length (cm)


length comp data, sexes combined, whole catch, comparing across
Fishery
NperTow+mm
Length (cm)
20
18
16
14
12
10
20
15
10
5
1984
1989
1992
1997
2002
2008
2011
Year

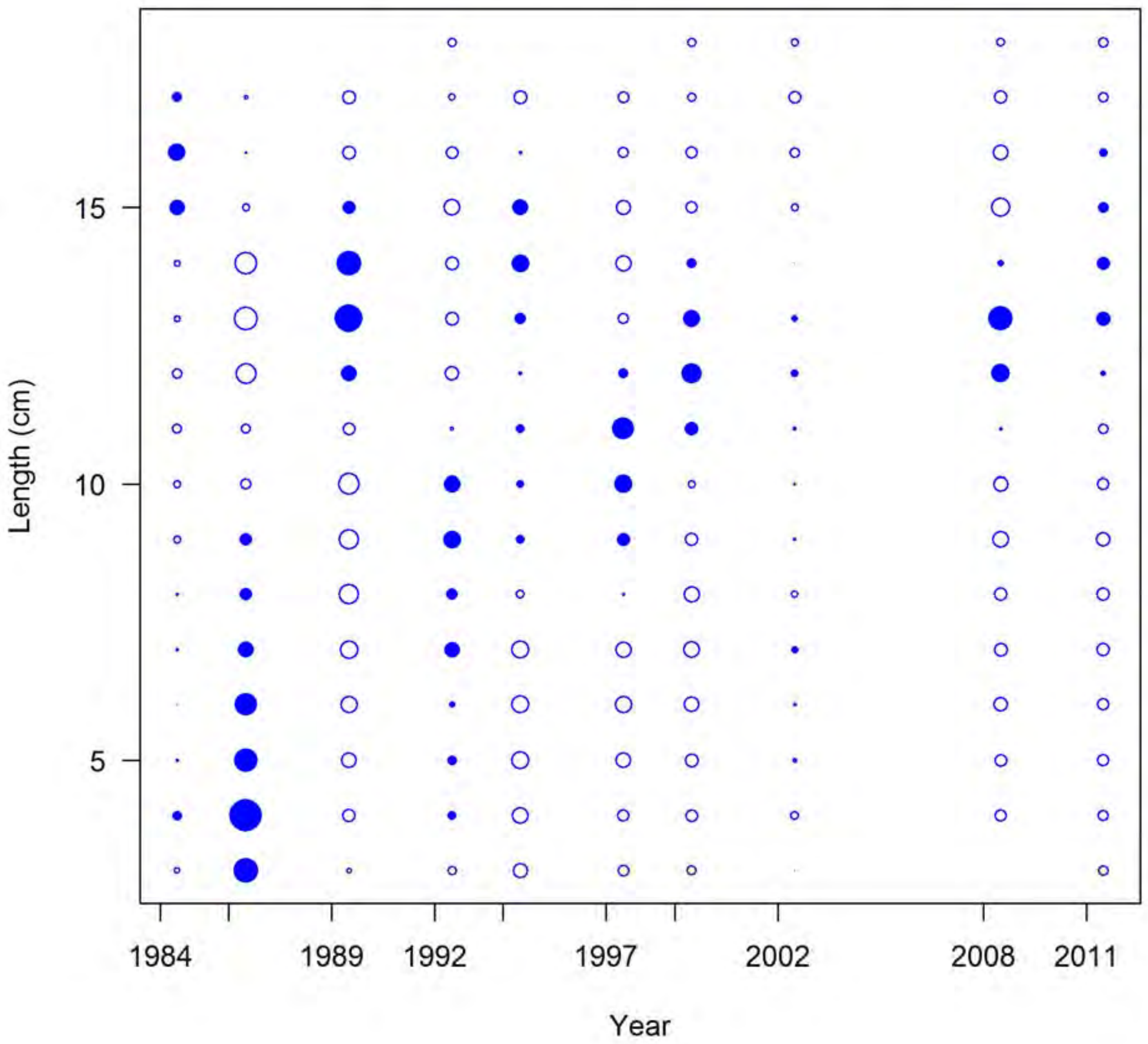

ength comp data, sexes combined, whole catch, aggregated across time b
Fishery
N=31.5
NperTow+mm
N=1143
Proportion
0.30
0.25
0.20
0.15
0.10
0.05
0.00
5
10
15
Length (cm)


length comps, sexes combined, whole catch, Fishery
2010
N=12
effN=14.9
2011
N=19.5
effN=19.6
0.0
0.1
0.2
0.3
0.4
10
14
18
Proportion
Length (cm)

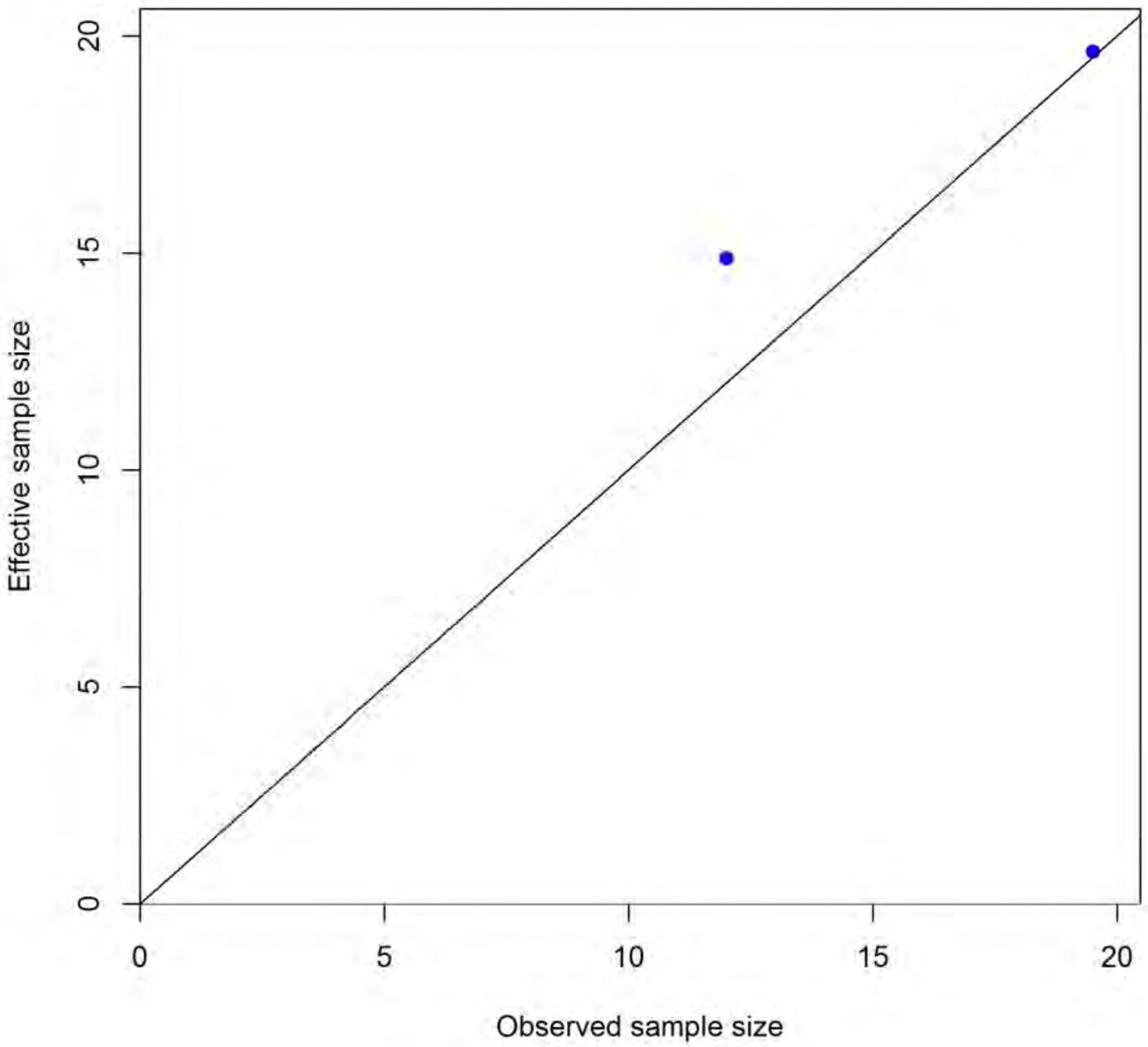

length comps, sexes combined, whole catch, NperTow+mm
1984 N=7.5 effN=18.1
1999 N=136.5 effN=52.4
1986 N=117 effN=20.1
2002 N=112.5 effN=1205.9
1989 N=118.5 effN=13.2
2008 N=141.8 effN=28
1992 N=126 effN=73.6
2011 N=101.2 effN=108.8
1994 N=142.5 effN=92.9
1997 N=139.5 effN=47.4
0.00 0.05 0.10 0.15 0.20 0.25 0.30 0.35
5 10 15
Proportion
Length (cm)


Pearson residuals, sexes combined, whole catch, Fishery (max=2.41)
Length (cm)
18
16
14
12
10
2010
2011
Year

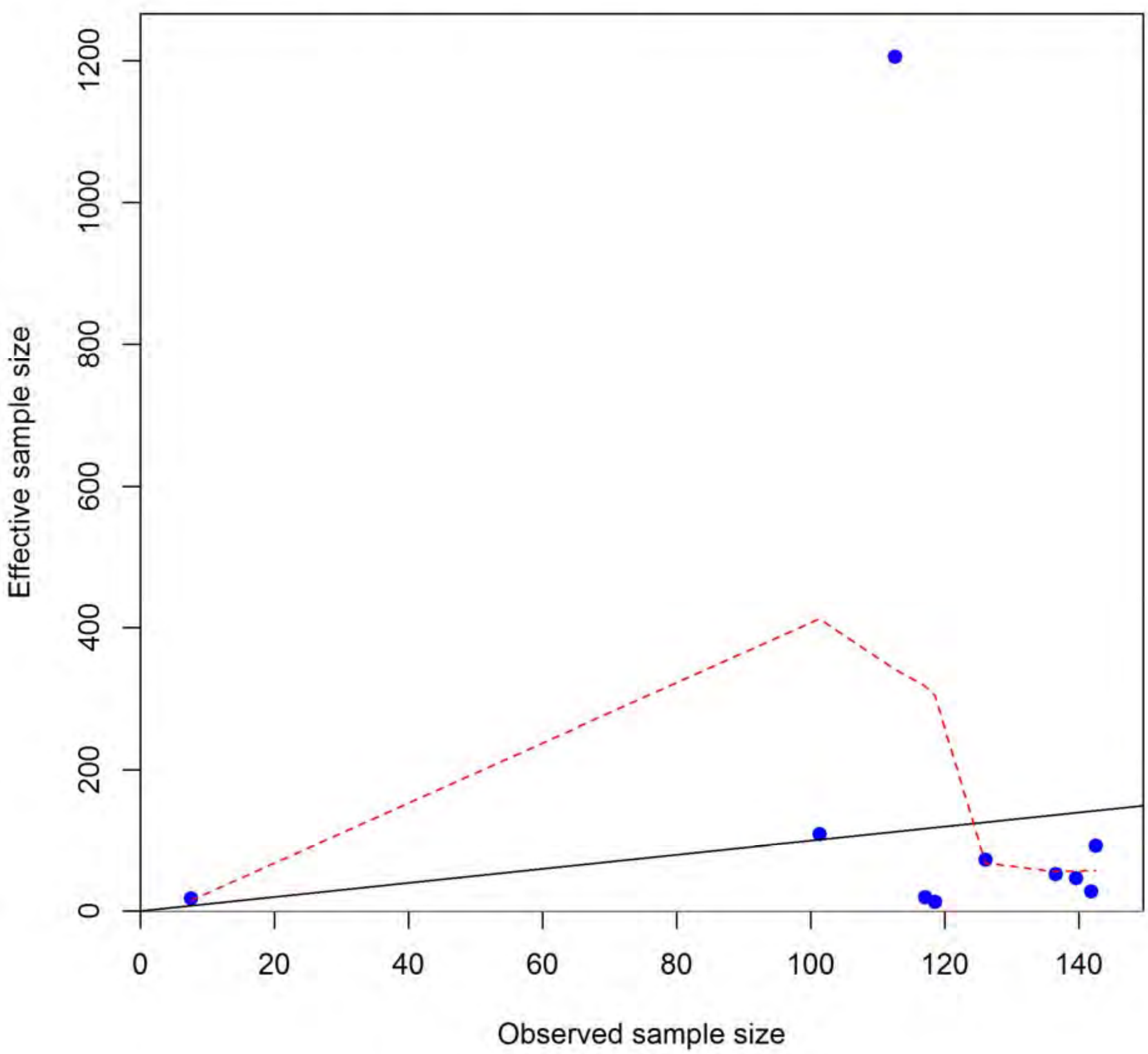
Pearson residuals, sexes combined, whole catch, NperTow+mm (max=7.5)

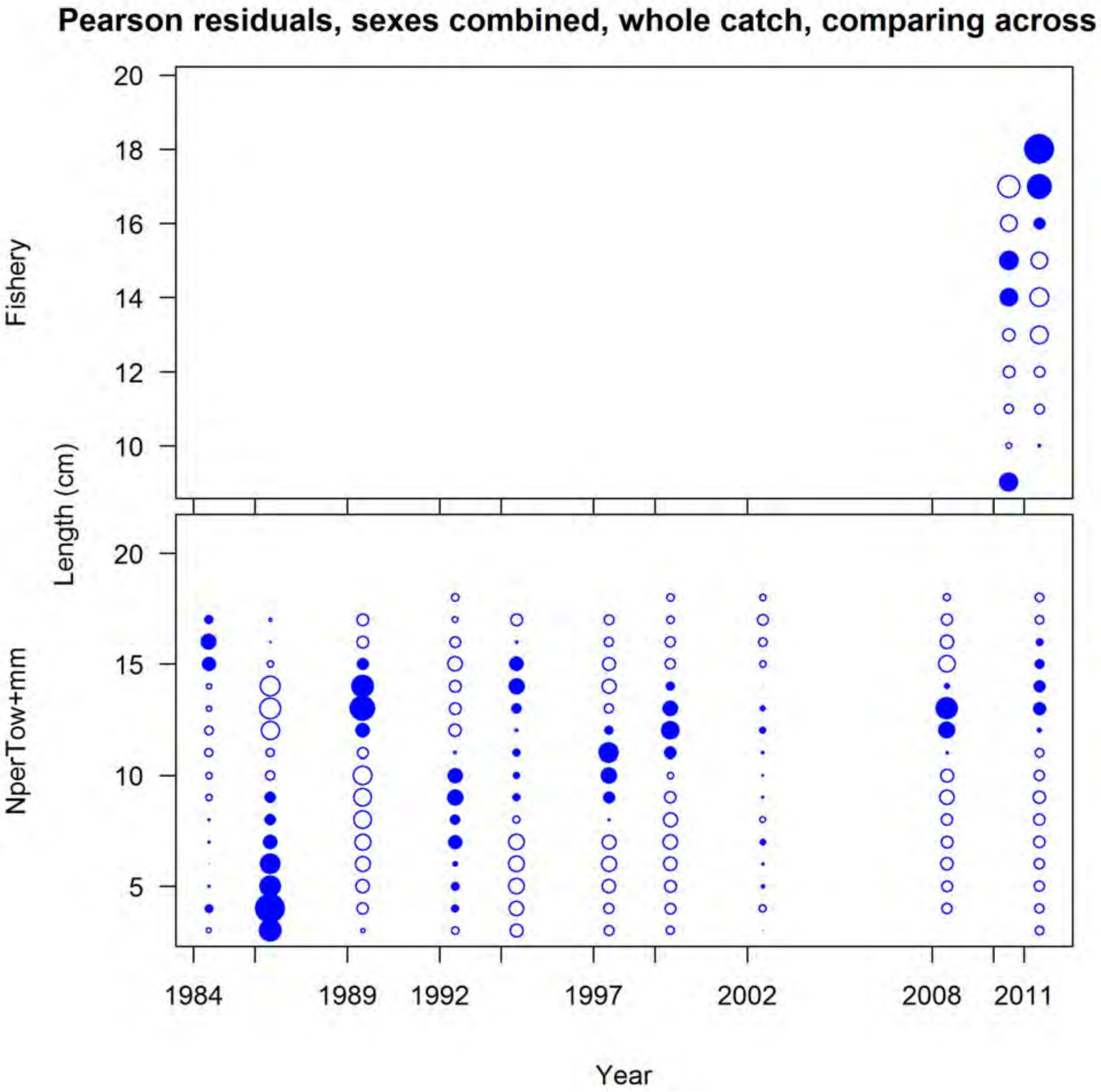


N-EffN comparison, length comps, sexes combined, whole catch, Fishery

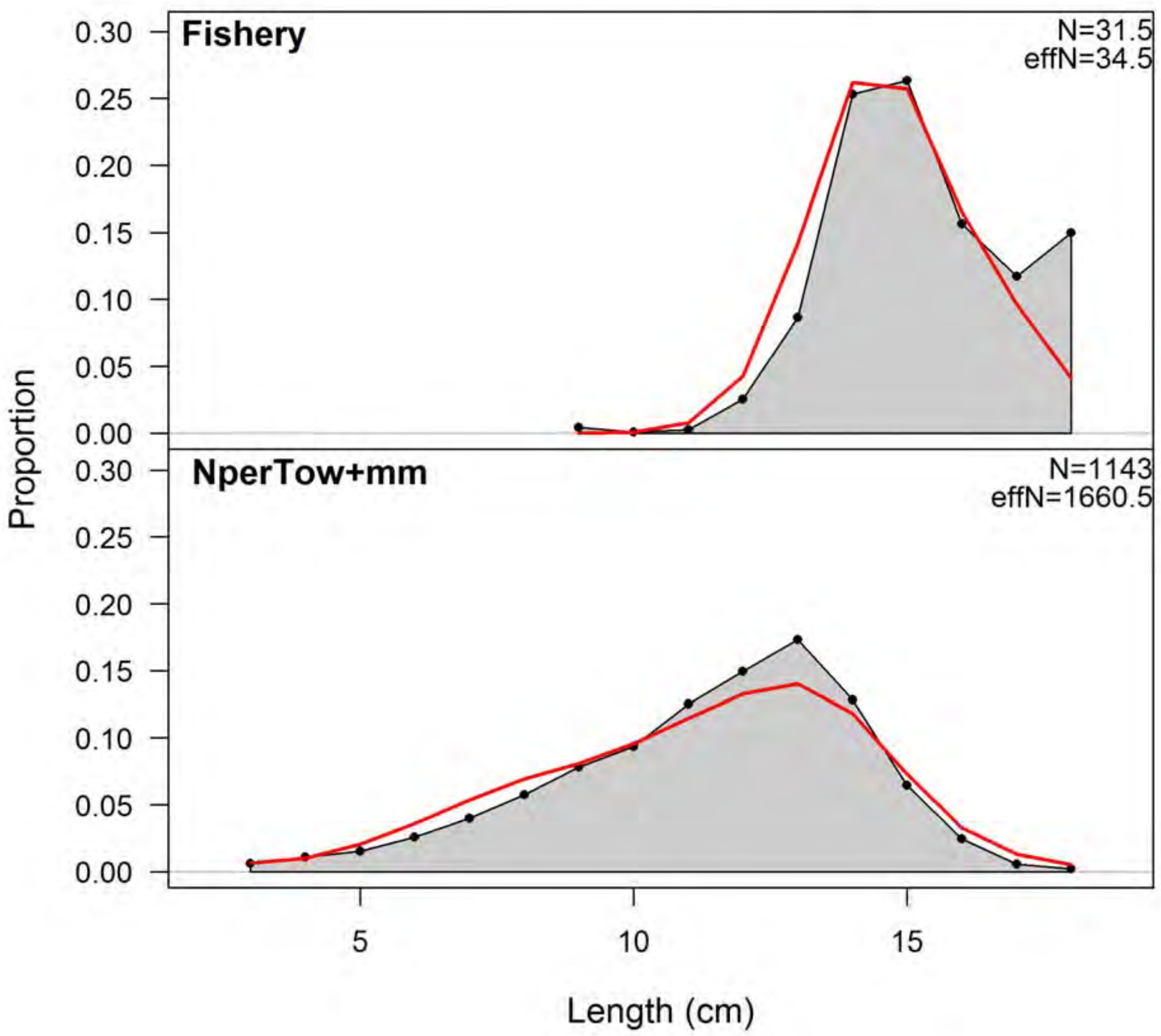


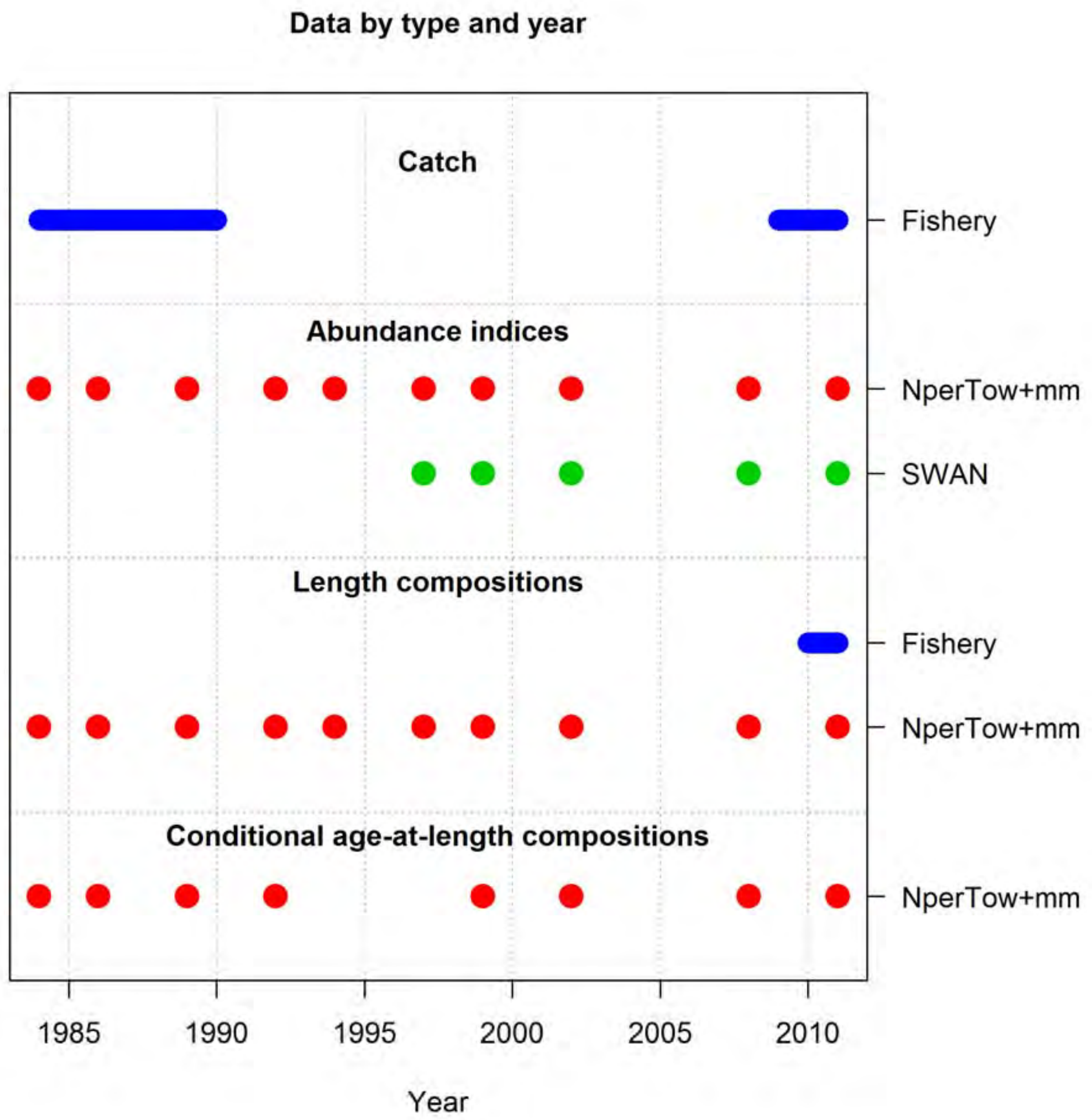
N-EffN comparison, length comps, sexes combined, whole catch, NperTow+mm

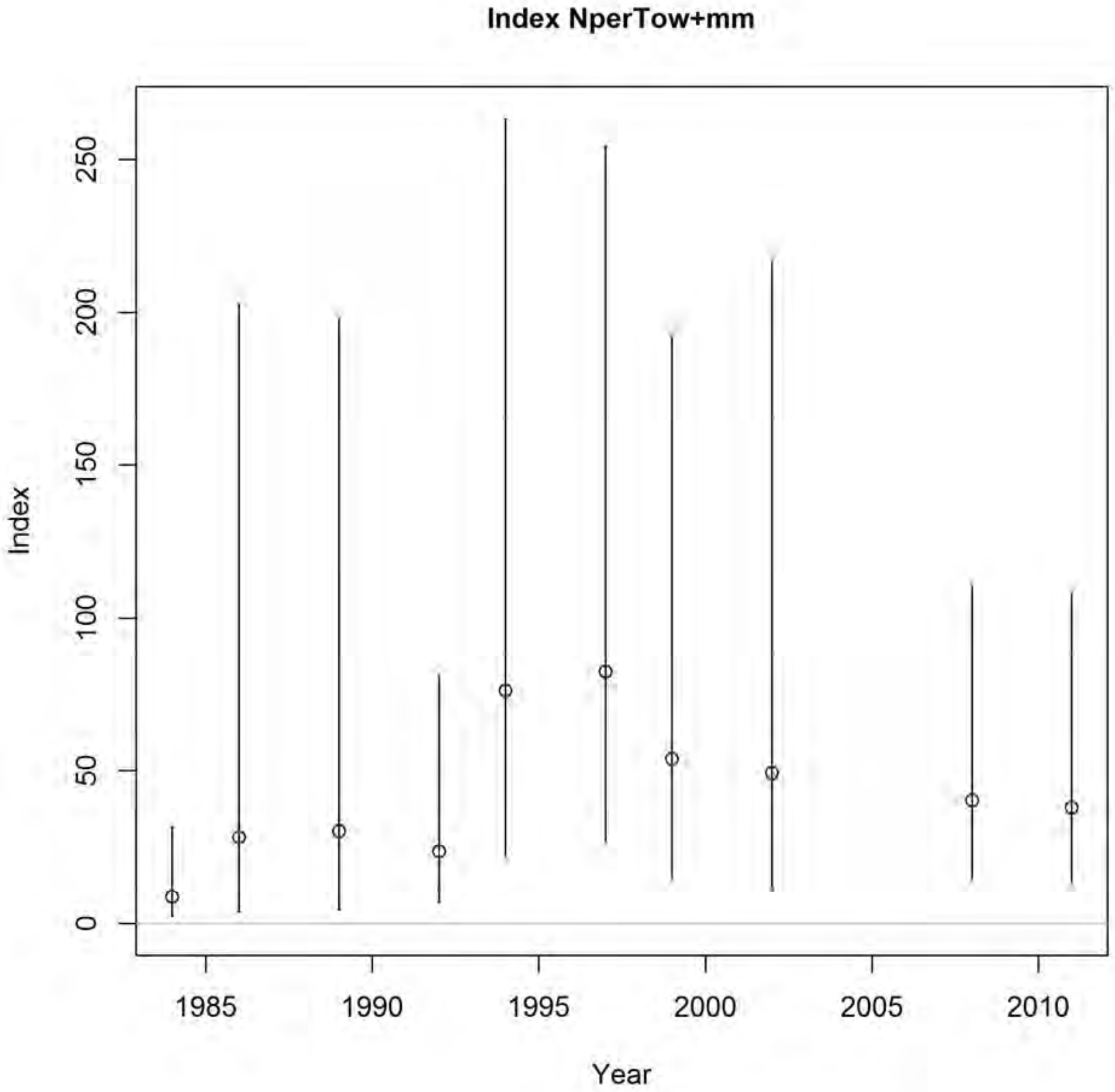


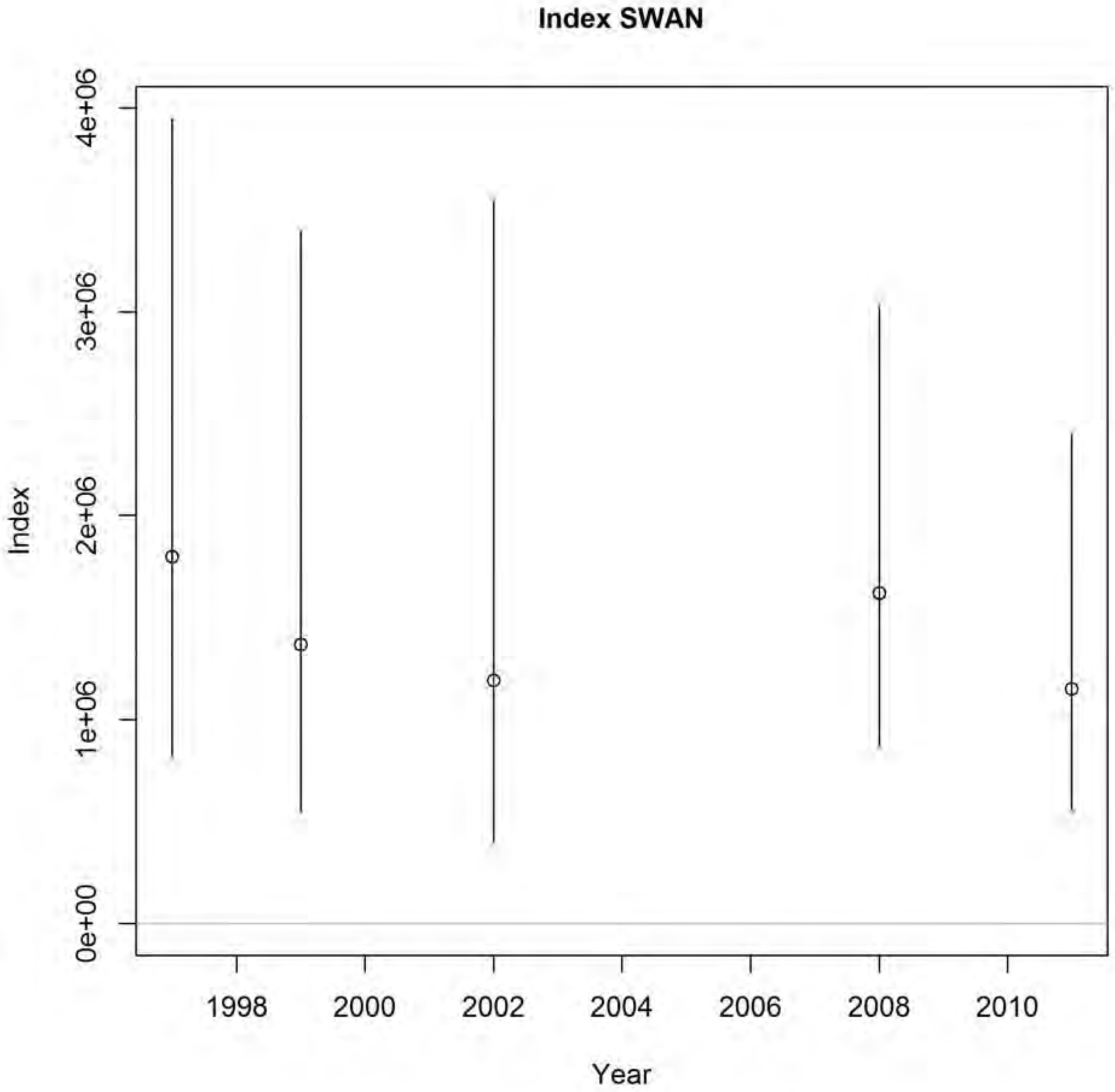

Pearson residuals, sexes combined, whole catch, comparing across
Fishery
NperTow+mm
Length (cm)
20
18
16
14
12
10
20
15
10
5
1984
1989
1992
1997
2002
2008
2011
Year

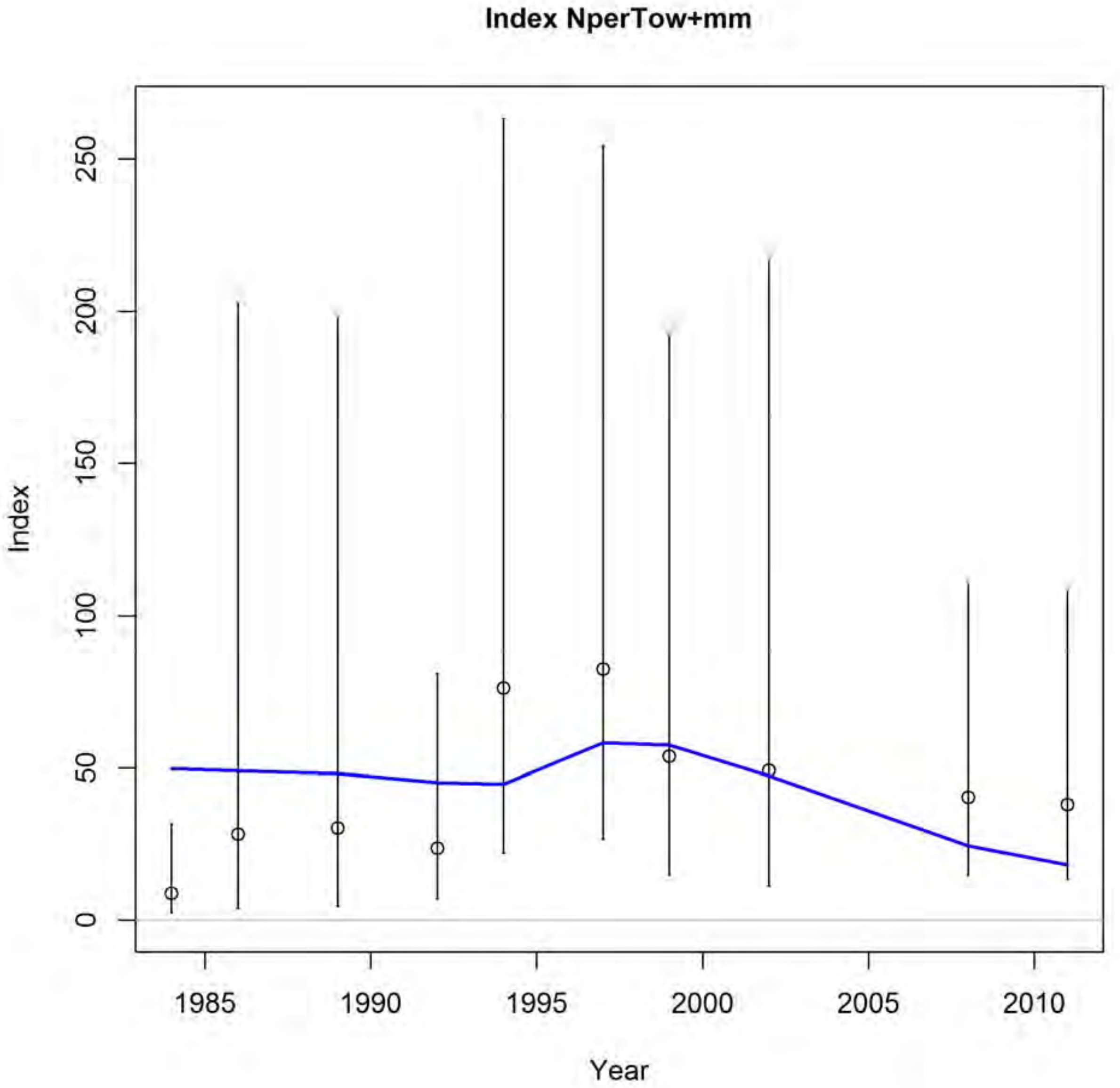
length comps, sexes combined, whole catch, aggregated across time by

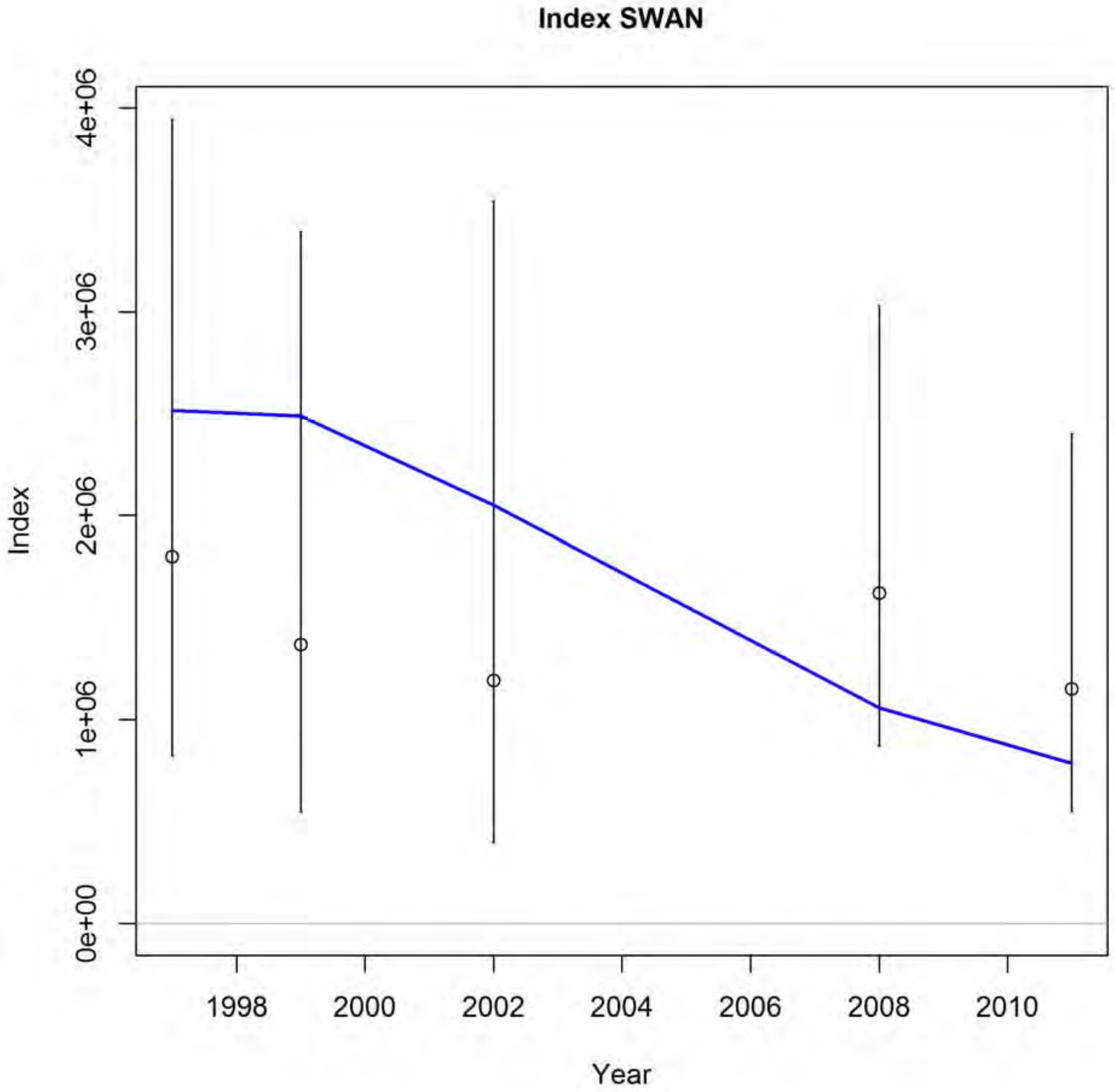


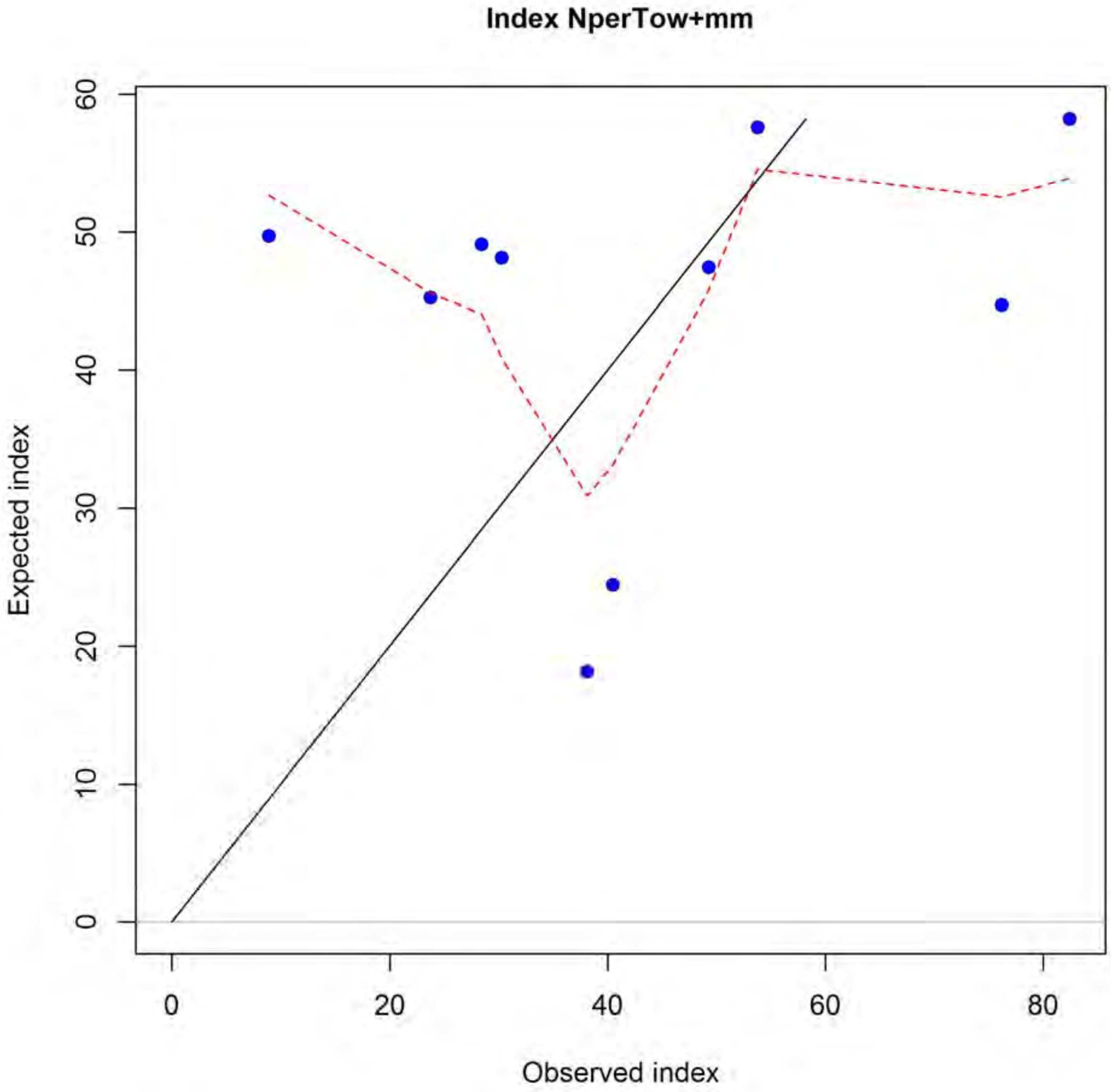

Data by type and year
Catch
Fishery
Abundance indices
NperTow+mm
SWAN
Length compositions
Fishery
NperTow+mm
Conditional age-at-length compositions
NperTow+mm
1985
1990
1995
2000
2005
2010
Year

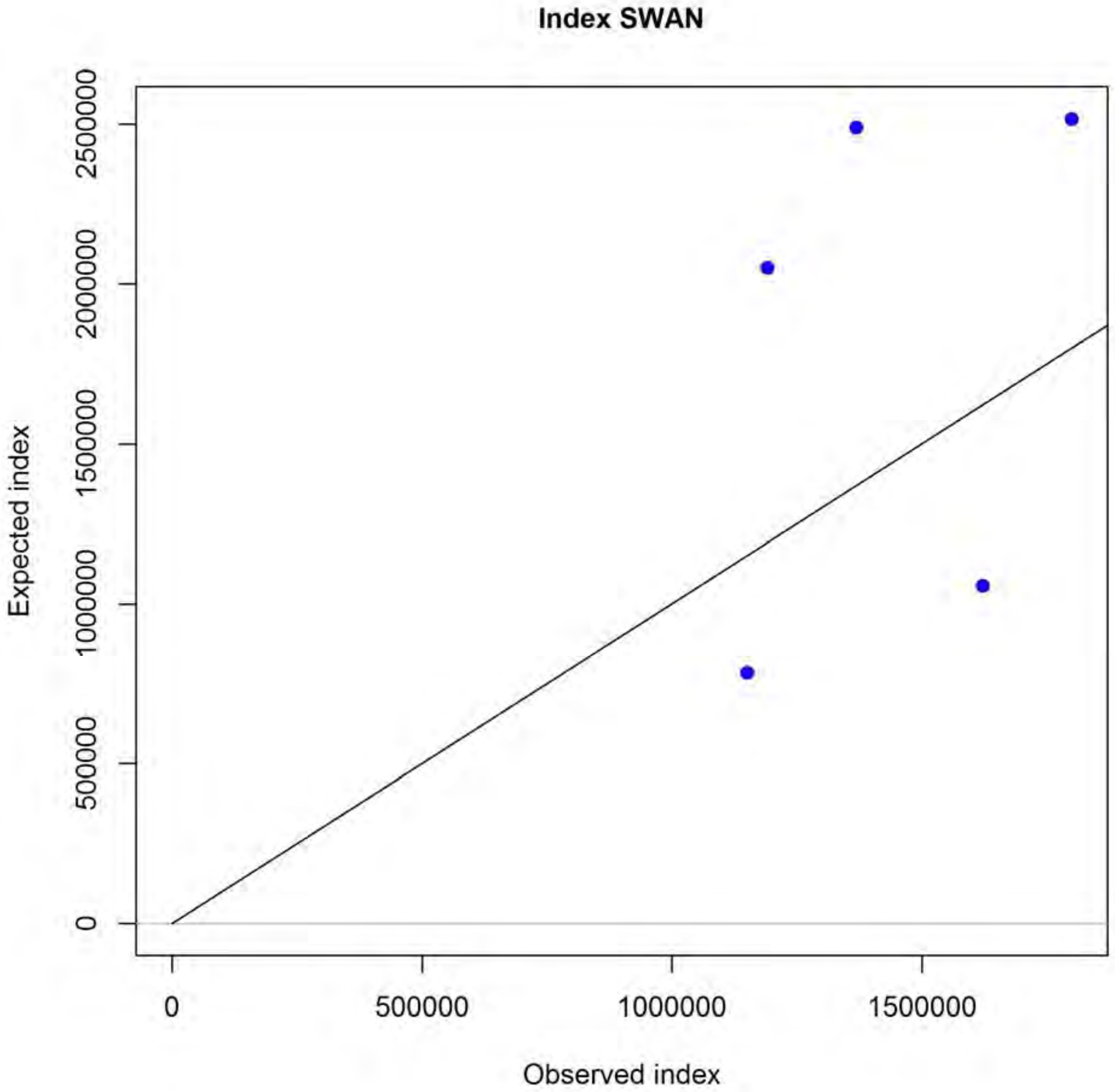

Index NperTow+mm
Index
0
50
100
150
200
250
Year
1985
1990
1995
2000
2005
2010

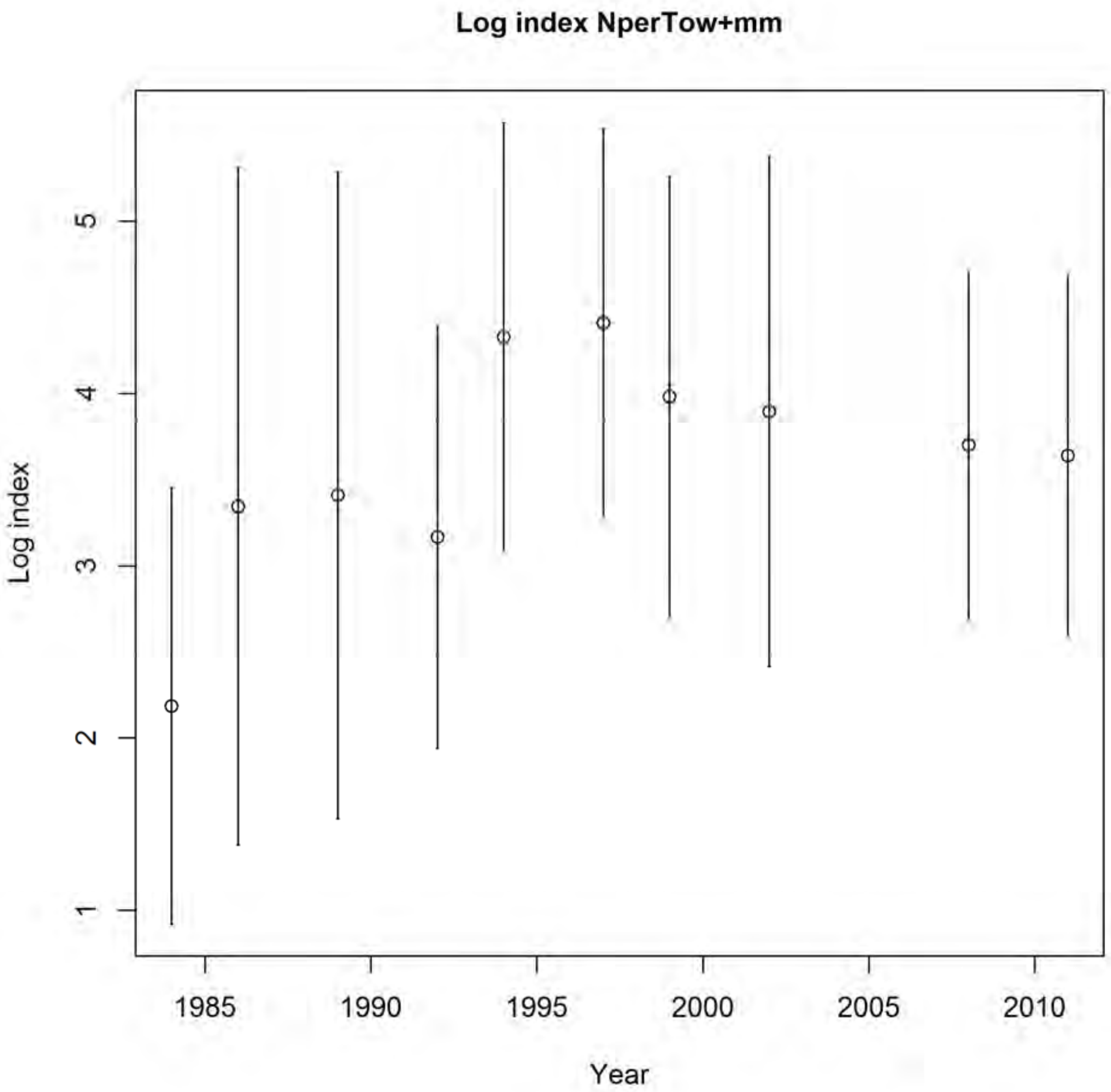

Index SWAN
Index
0e+00
1e+06
2e+06
3e+06
4e+06
1998
2000
2002
2004
2006
2008
2010
Year

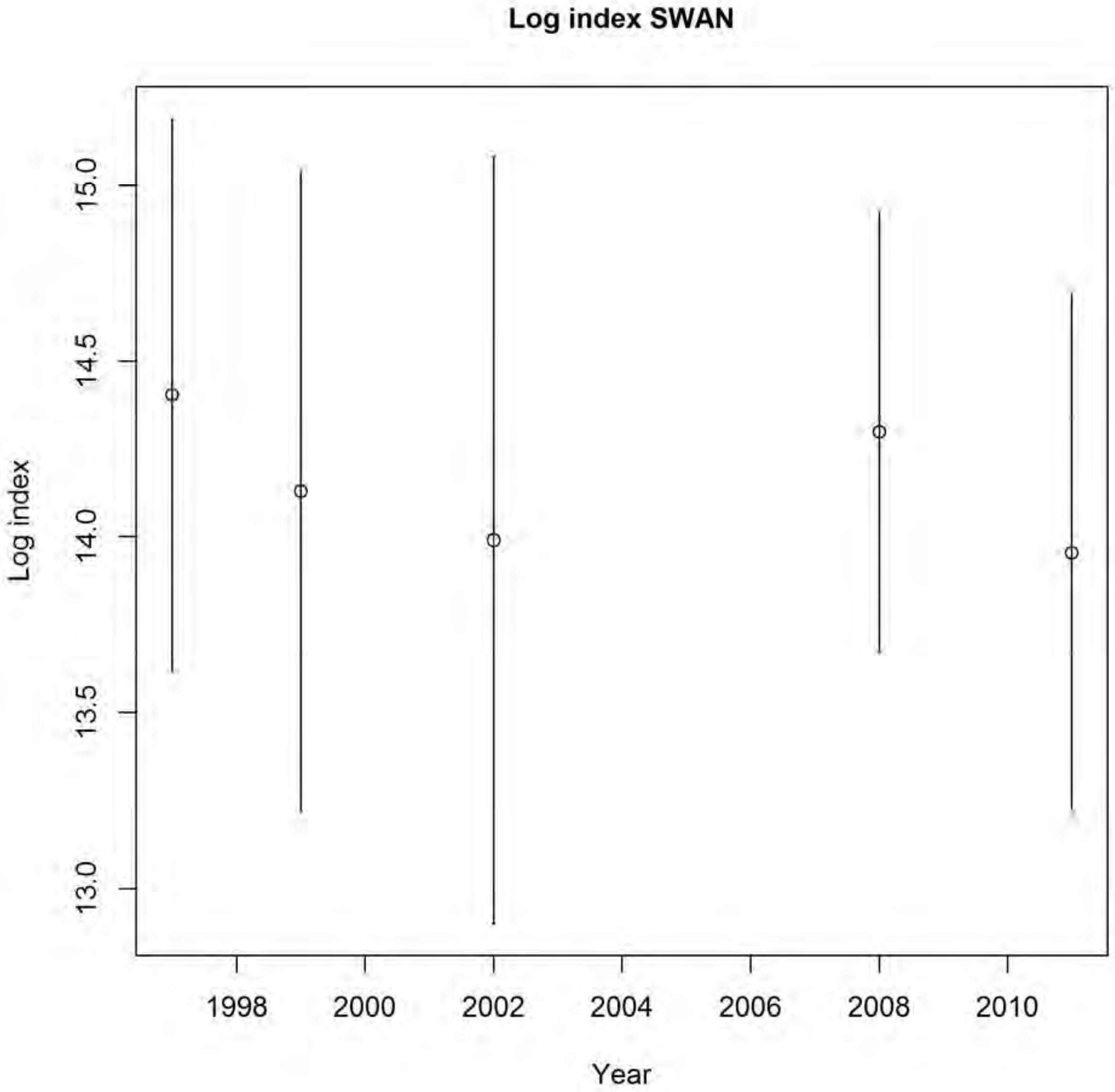

Index NperTow+mm
Index
0
50
100
150
200
250
Year
1985
1990
1995
2000
2005
2010

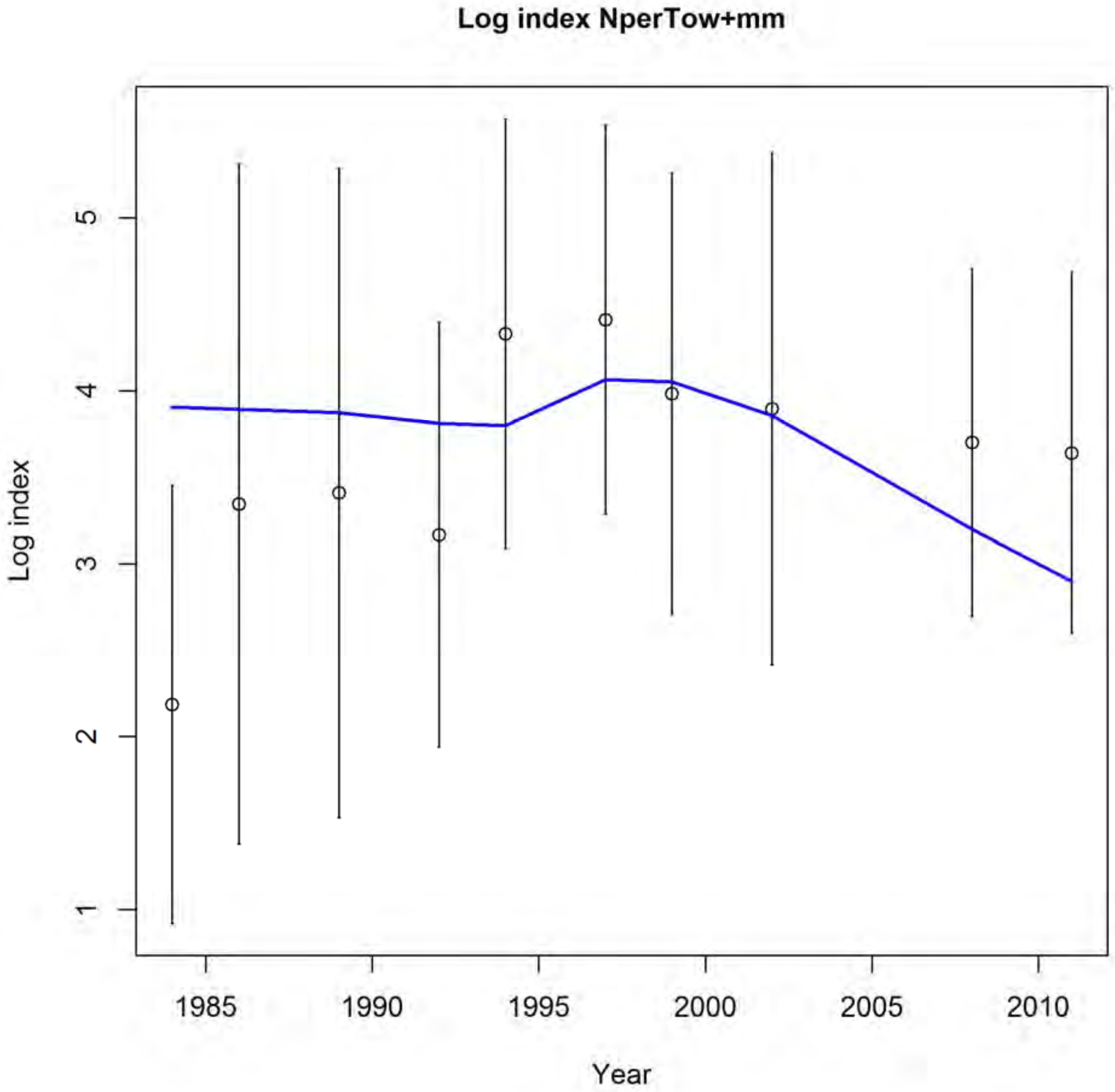

Index SWAN
Index
Year
0e+00
1e+06
2e+06
3e+06
4e+06
1998
2000
2002
2004
2006
2008
2010

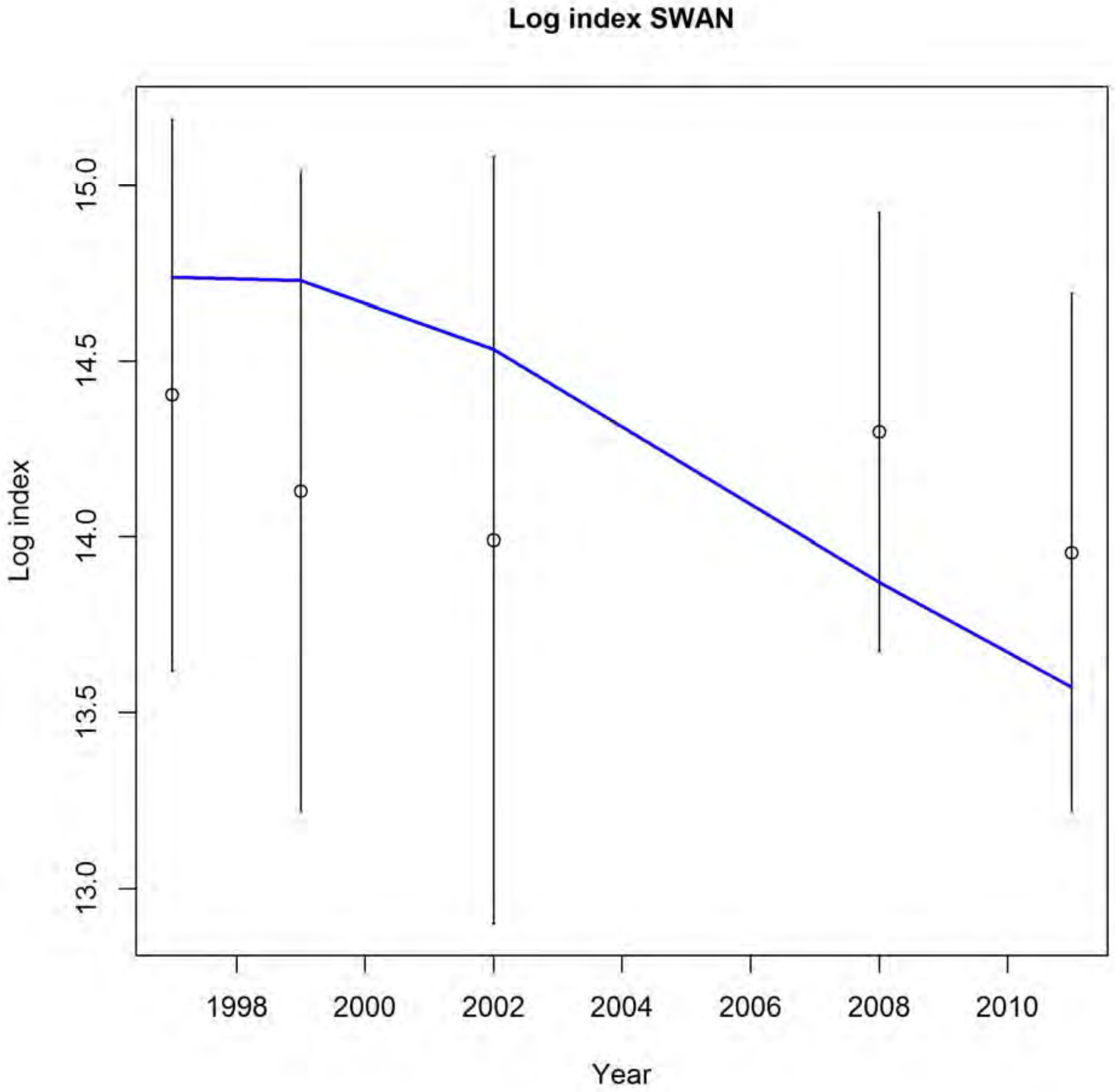

Index NperTow+mm
Expected index
60
50
40
30
20
10
0
0
20
40
60
80
Observed index

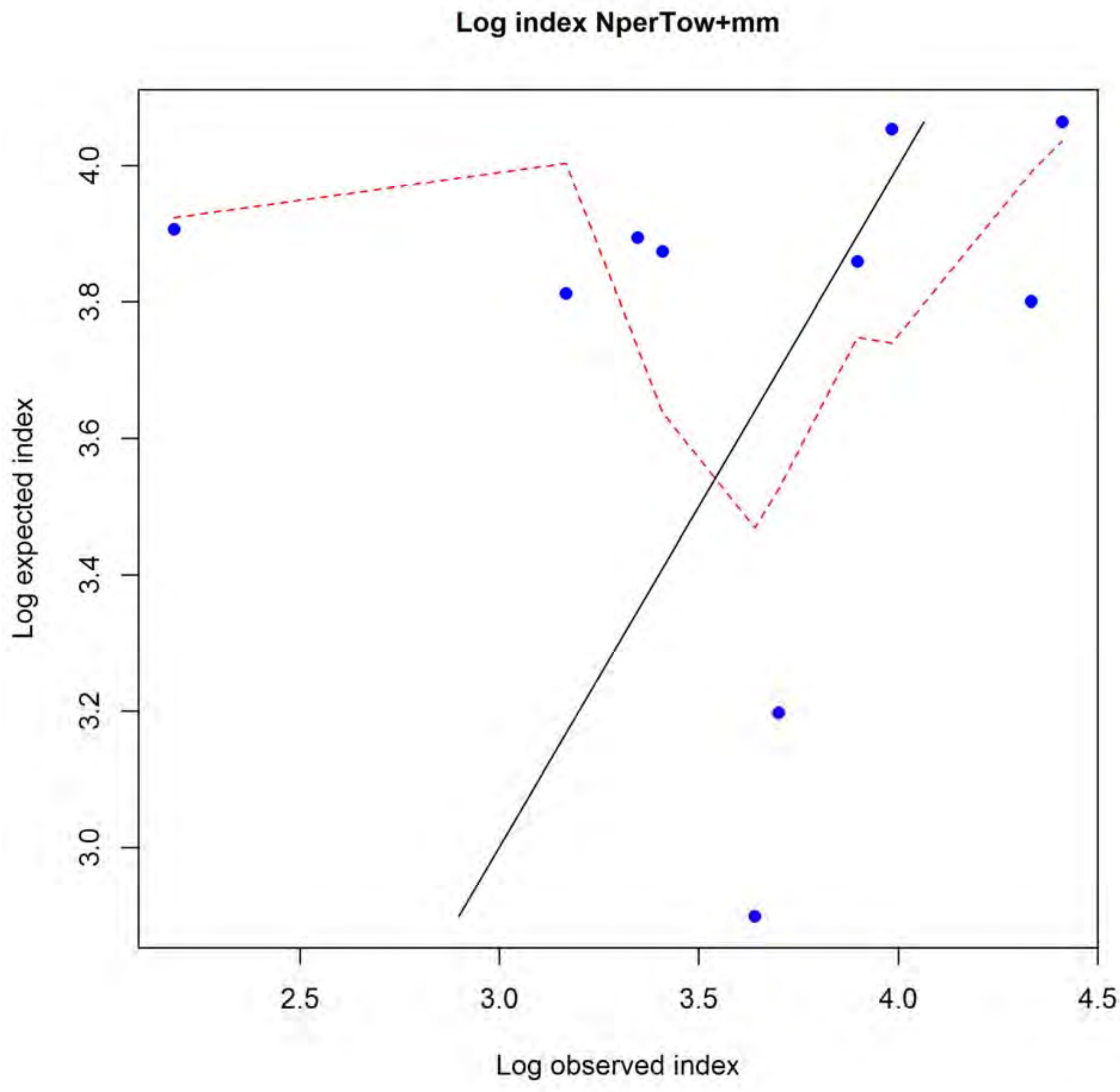

Index SWAN
Expected index
0
500000
1000000
1500000
2000000
2500000
Observed index
0
500000
1000000
1500000

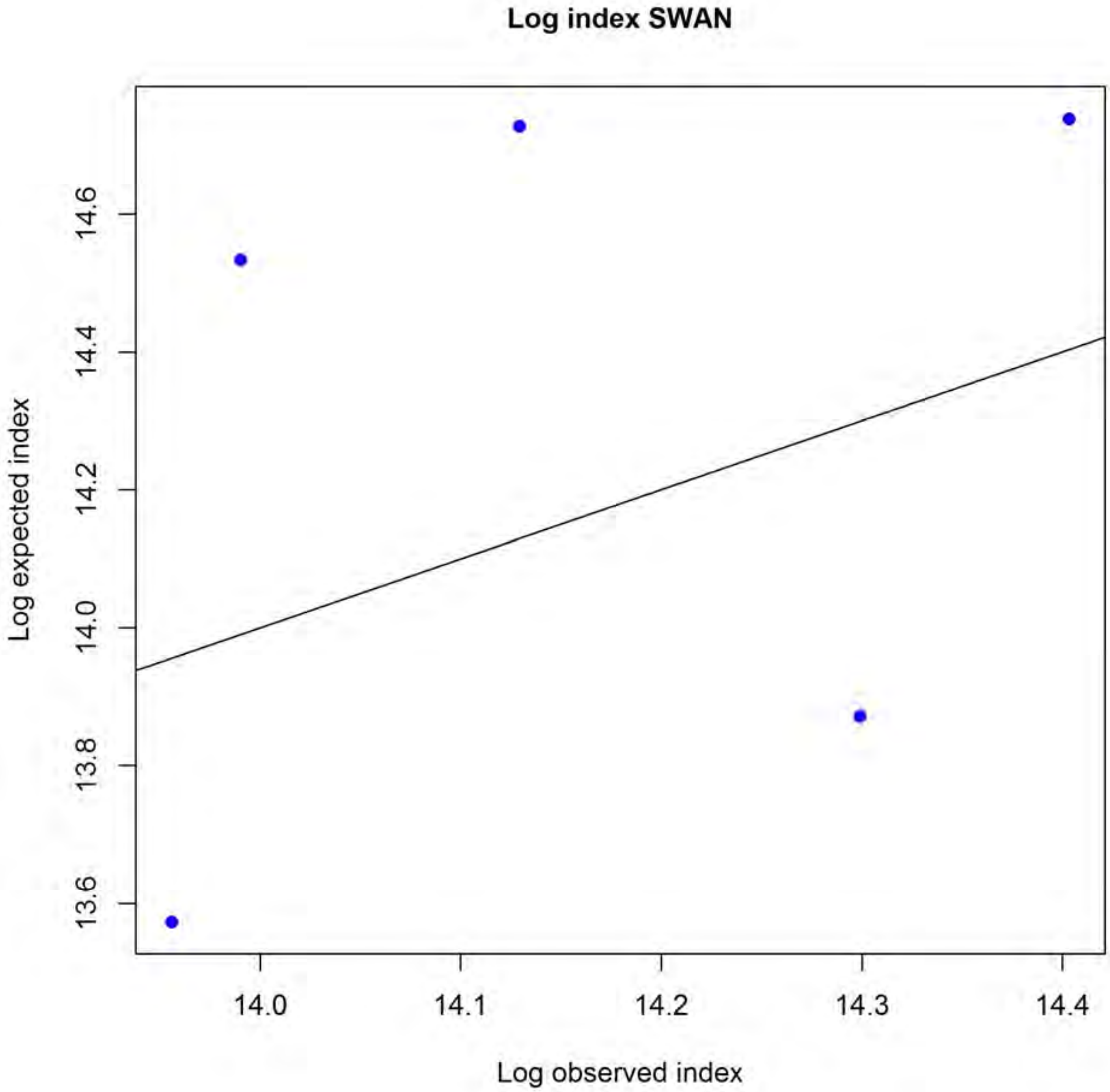

Log index NperTow+mm
Log index
1
2
3
4
5
1985
1990
1995
2000
2005
2010
Year

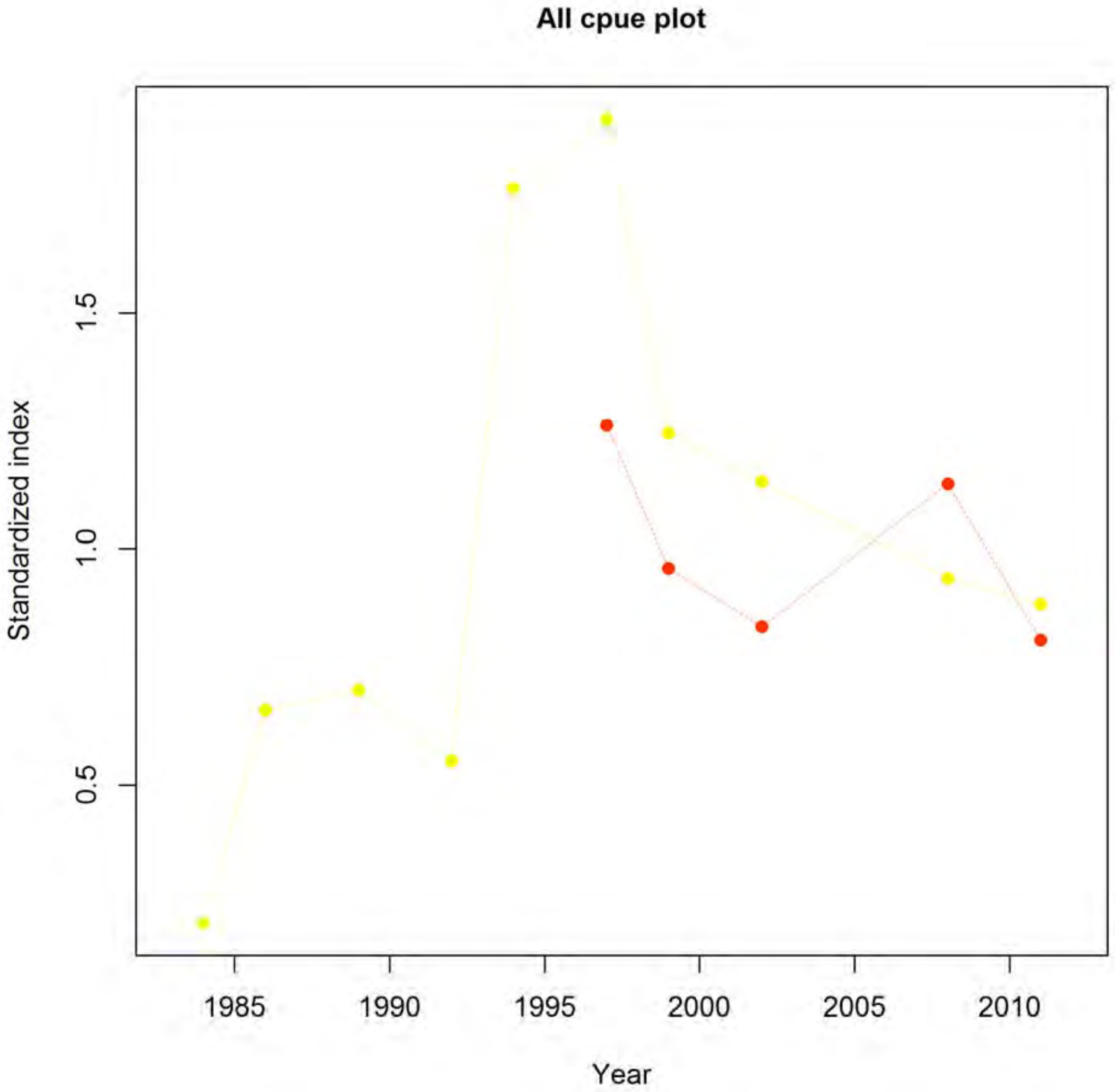

Log index SWAN
Log index
15.0
14.5
14.0
13.5
13.0
Year
1998
2000
2002
2004
2006
2008
2010


Log index NperTow+mm
Log index
1
2
3
4
5
1985
1990
1995
2000
2005
2010
Year

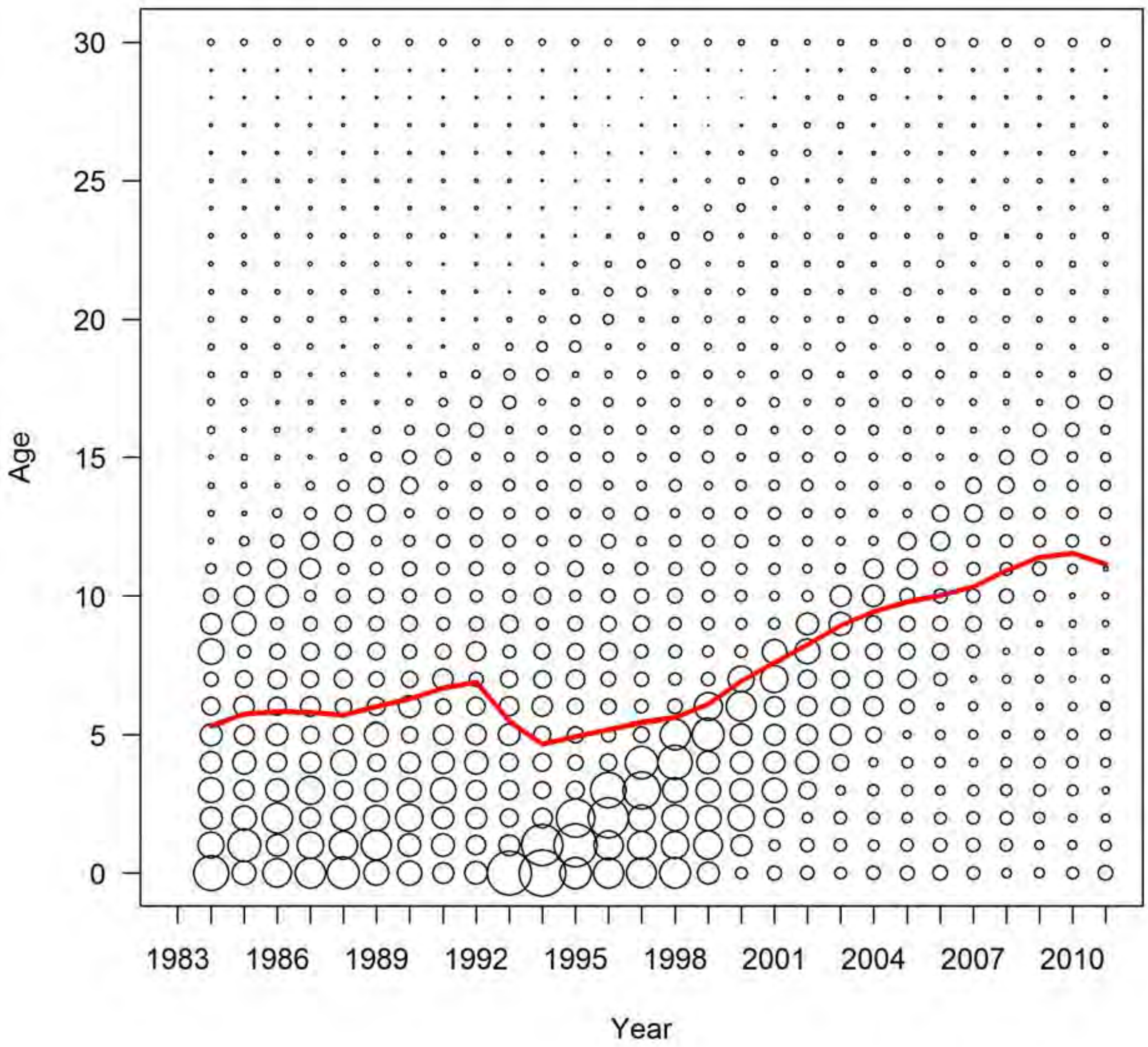

Log index SWAN
Log index
15.0
14.5
14.0
13.5
13.0
1998
2000
2002
2004
2006
2008
2010
Year

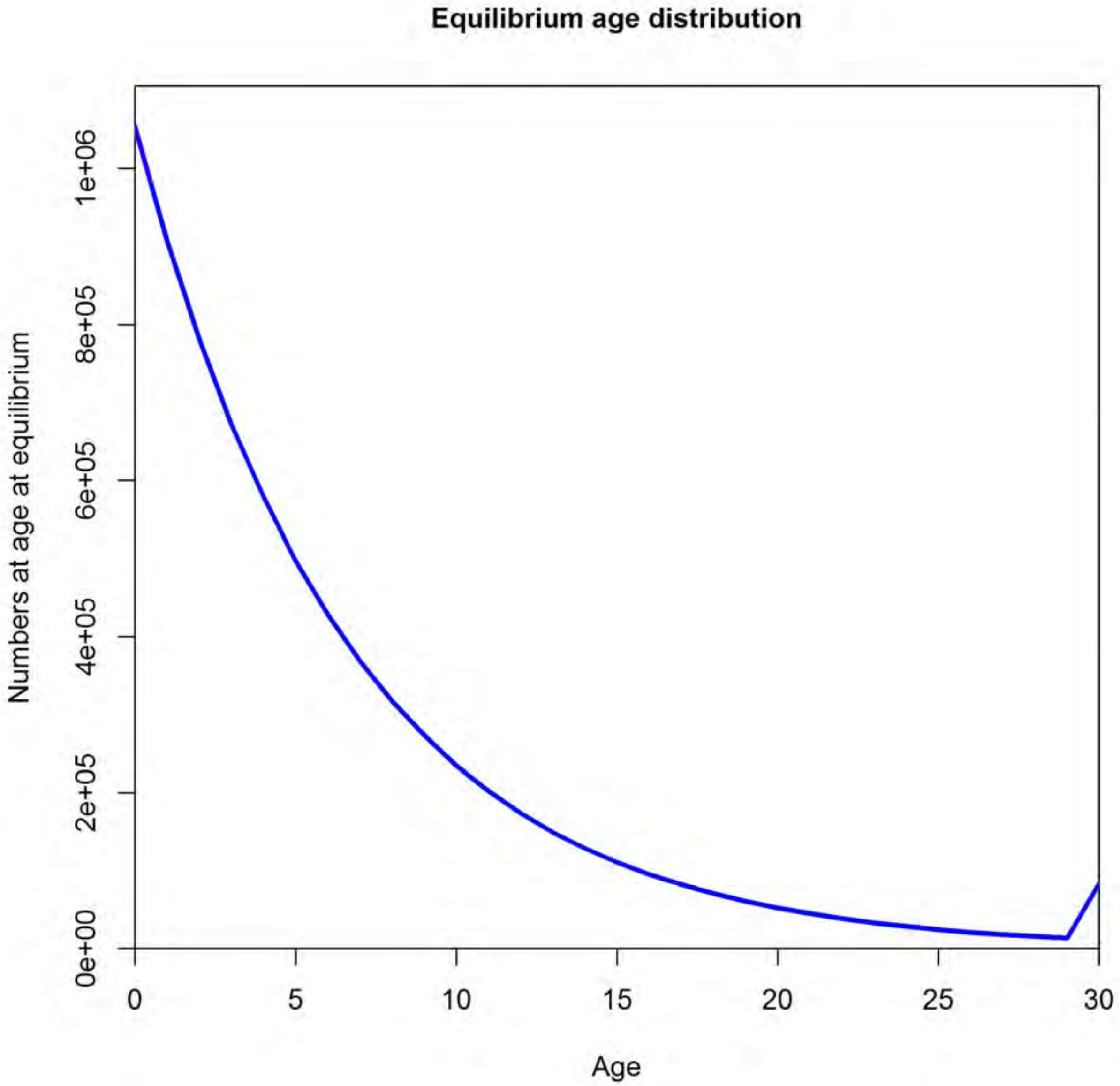

Log index NperTow+mm
Log expected index
Log observed index
4.0
3.8
3.6
3.4
3.2
3.0
2.5
3.0
3.5
4.0
4.5

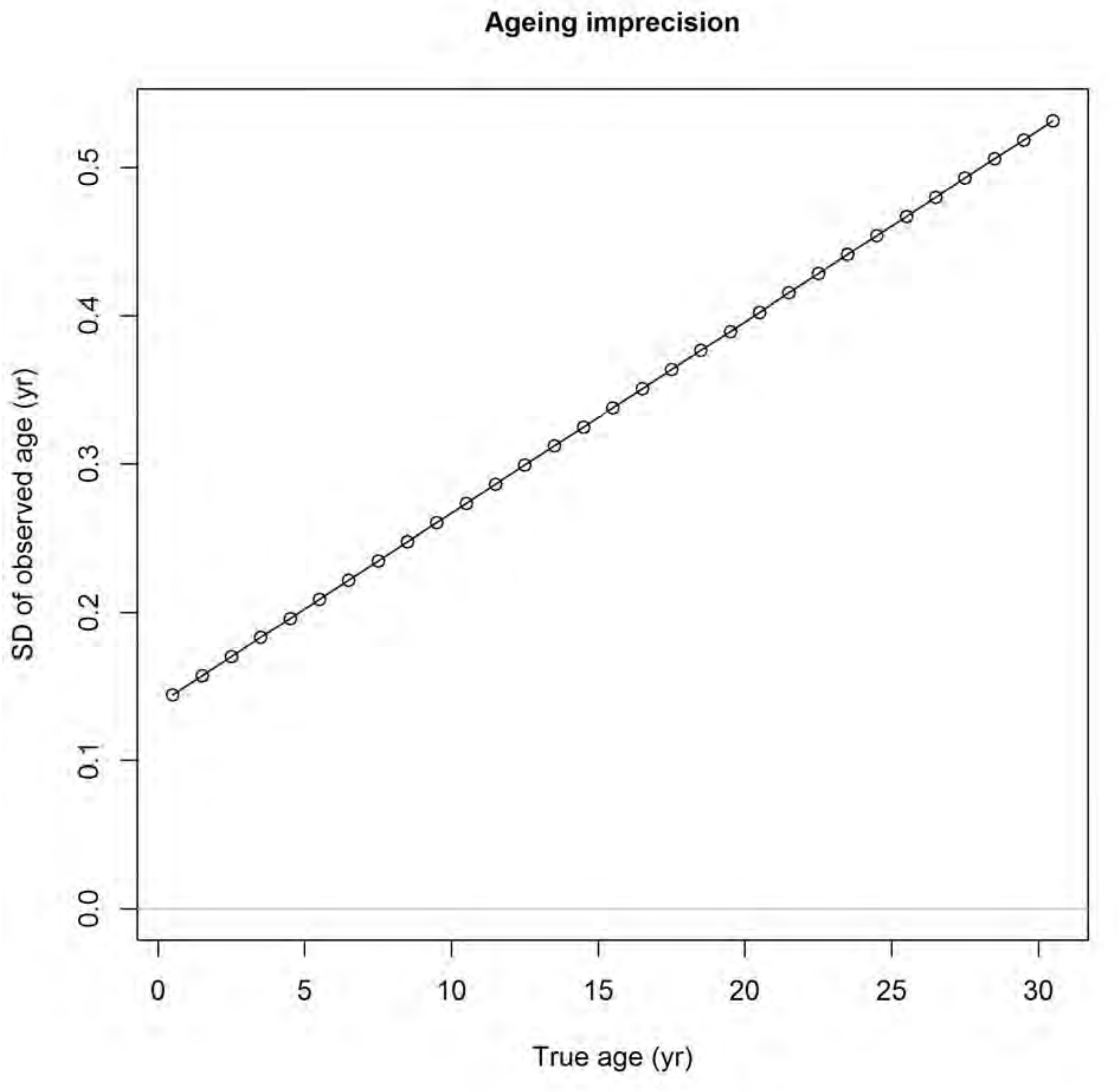

Log index SWAN
Log expected index
Log observed index
14.6
14.4
14.2
14.0
13.8
13.6
14.0
14.1
14.2
14.3
14.4


All cpue plot
Standardized index
0.5
1.0
1.5
1985
1990
1995
2000
2005
2010
Year

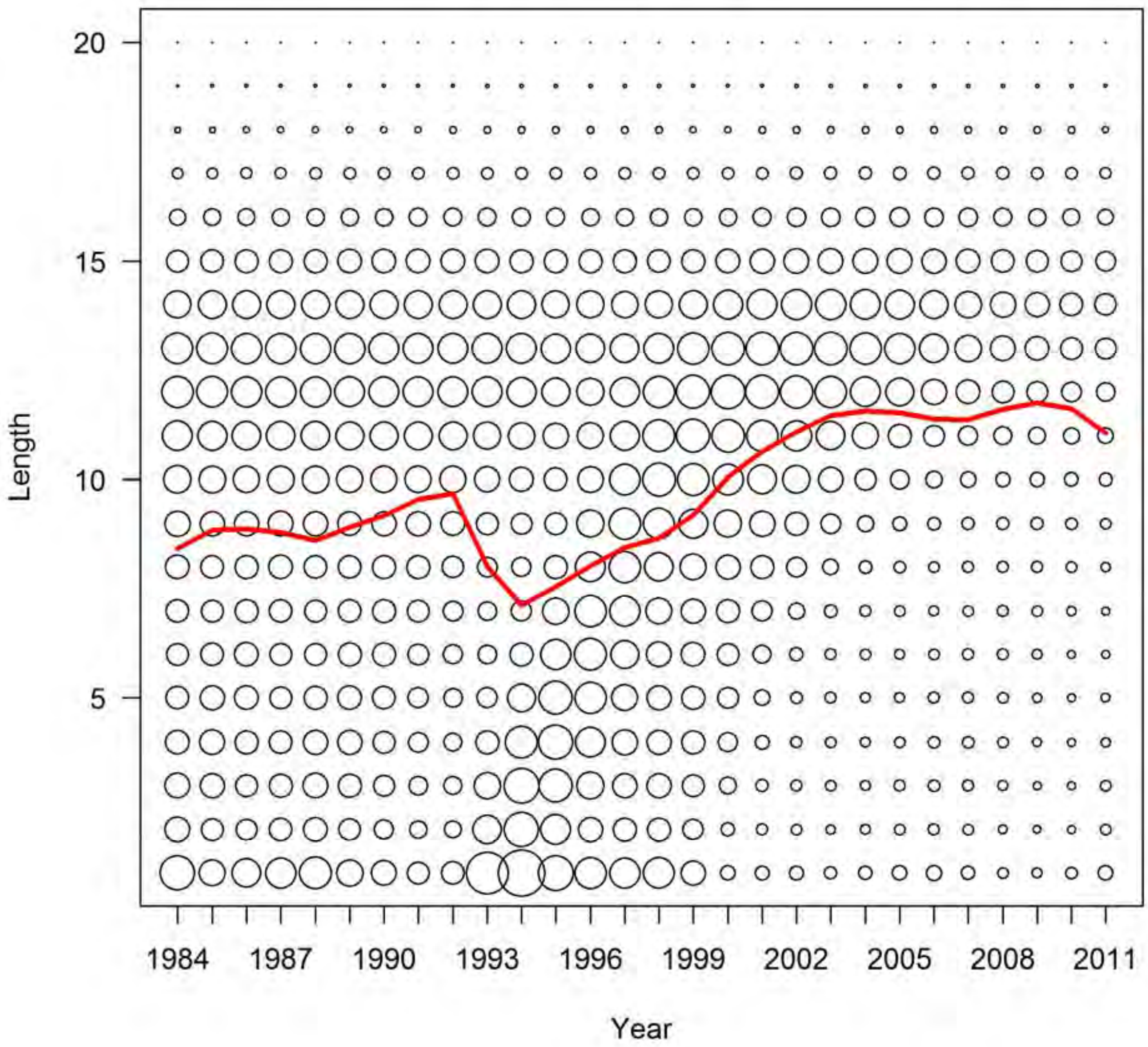
Middle of year expected numbers at age in thousands (max=3336850)

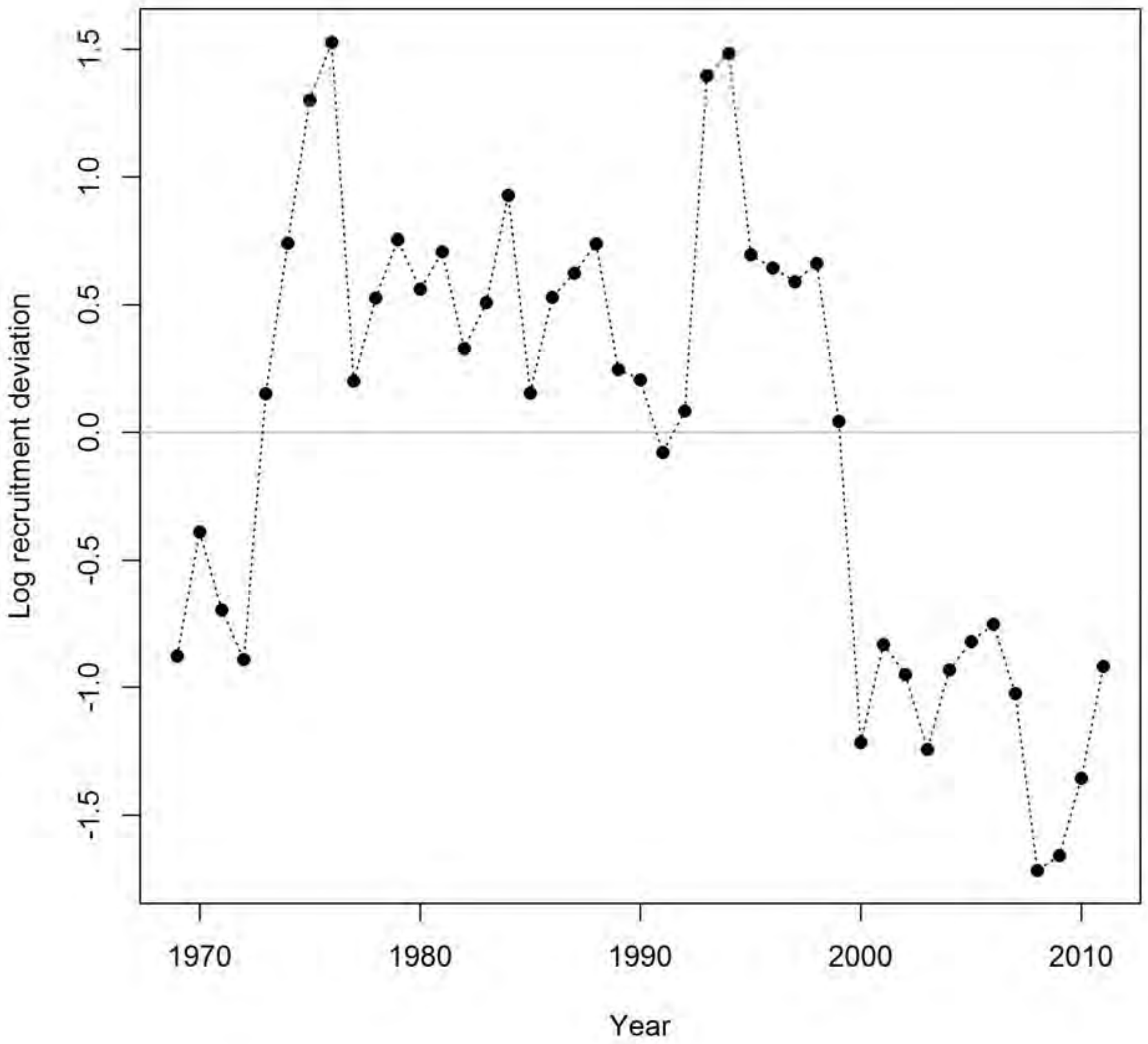


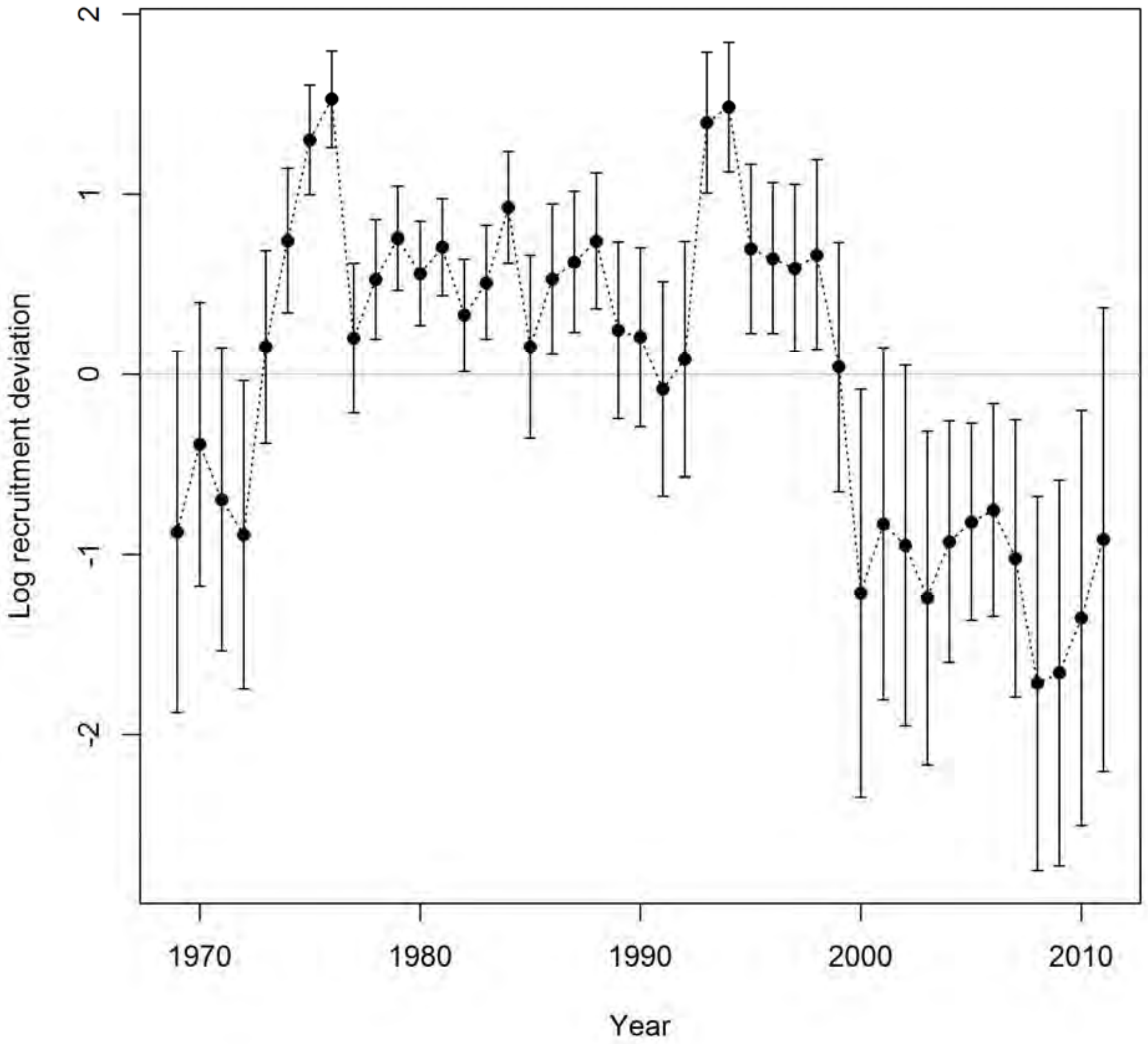

Equilibrium age distribution
Numbers at age at equilibrium
0e+00
2e+05
4e+05
6e+05
8e+05
1e+06
0
5
10
15
20
25
30
Age

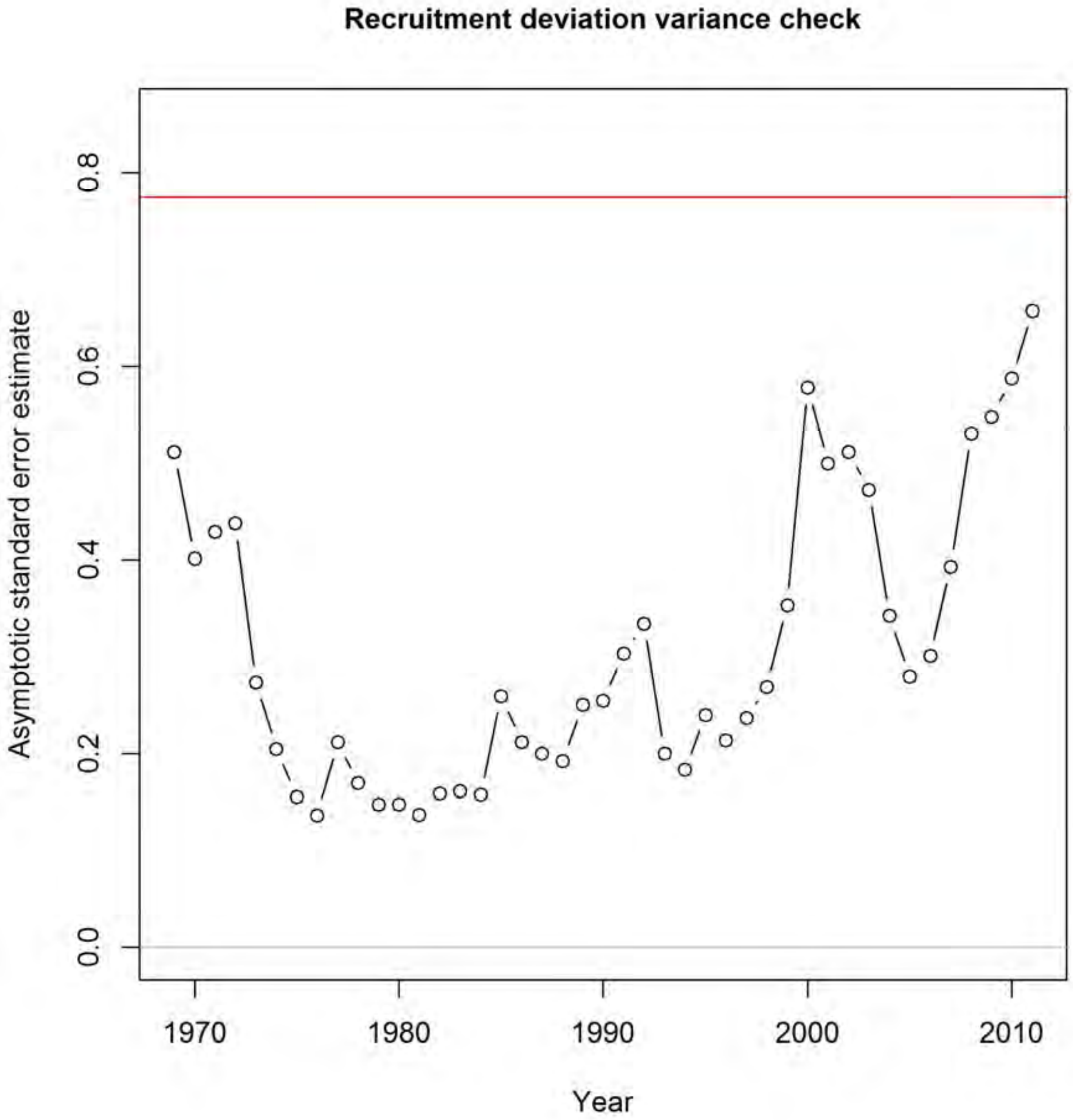

Ageing imprecision
SD of observed age (yr)
0.0
0.1
0.2
0.3
0.4
0.5
True age (yr)
0
5
10
15
20
25
30

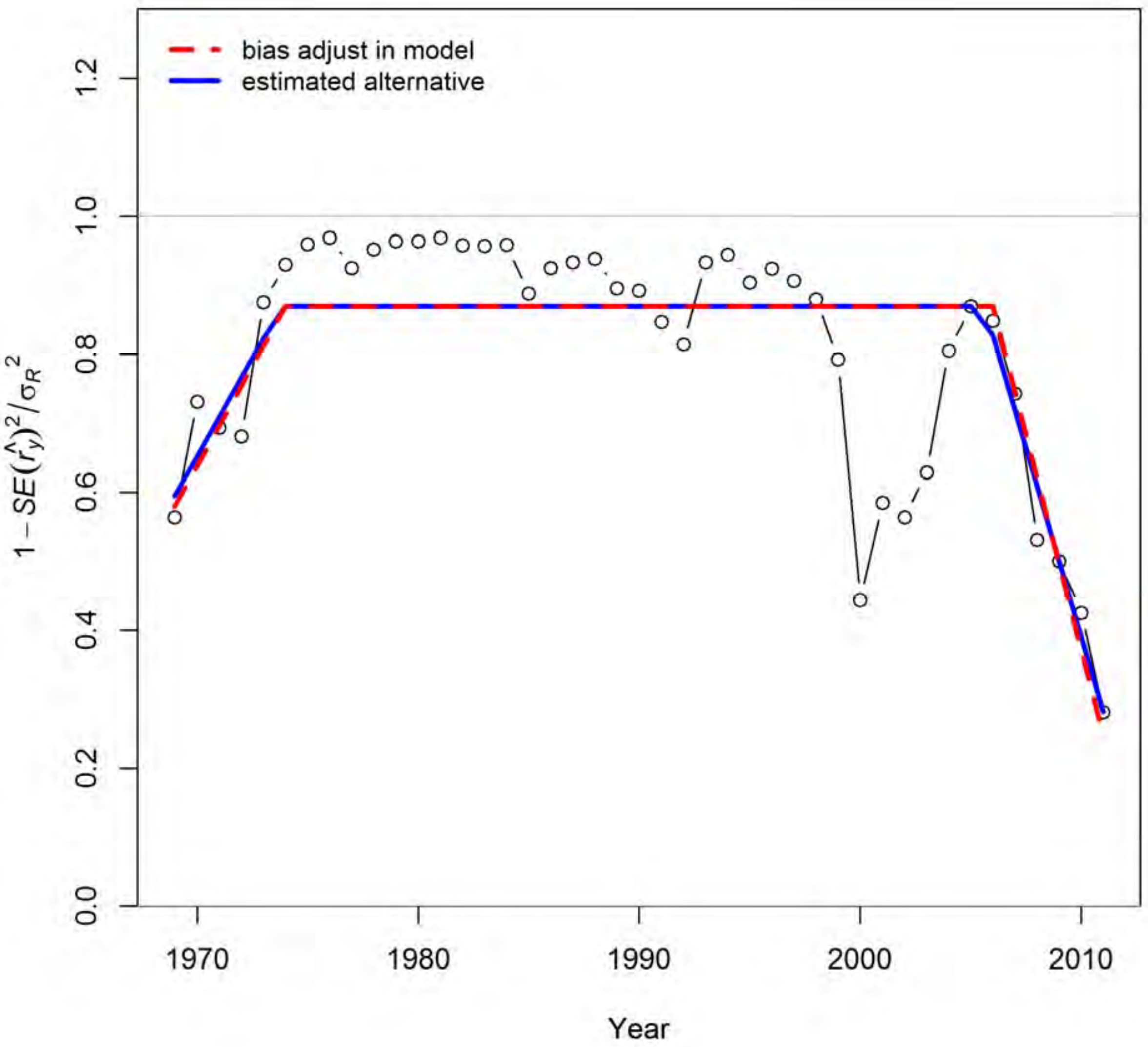
Middle of year expected numbers at length in thousands (max=1977530)

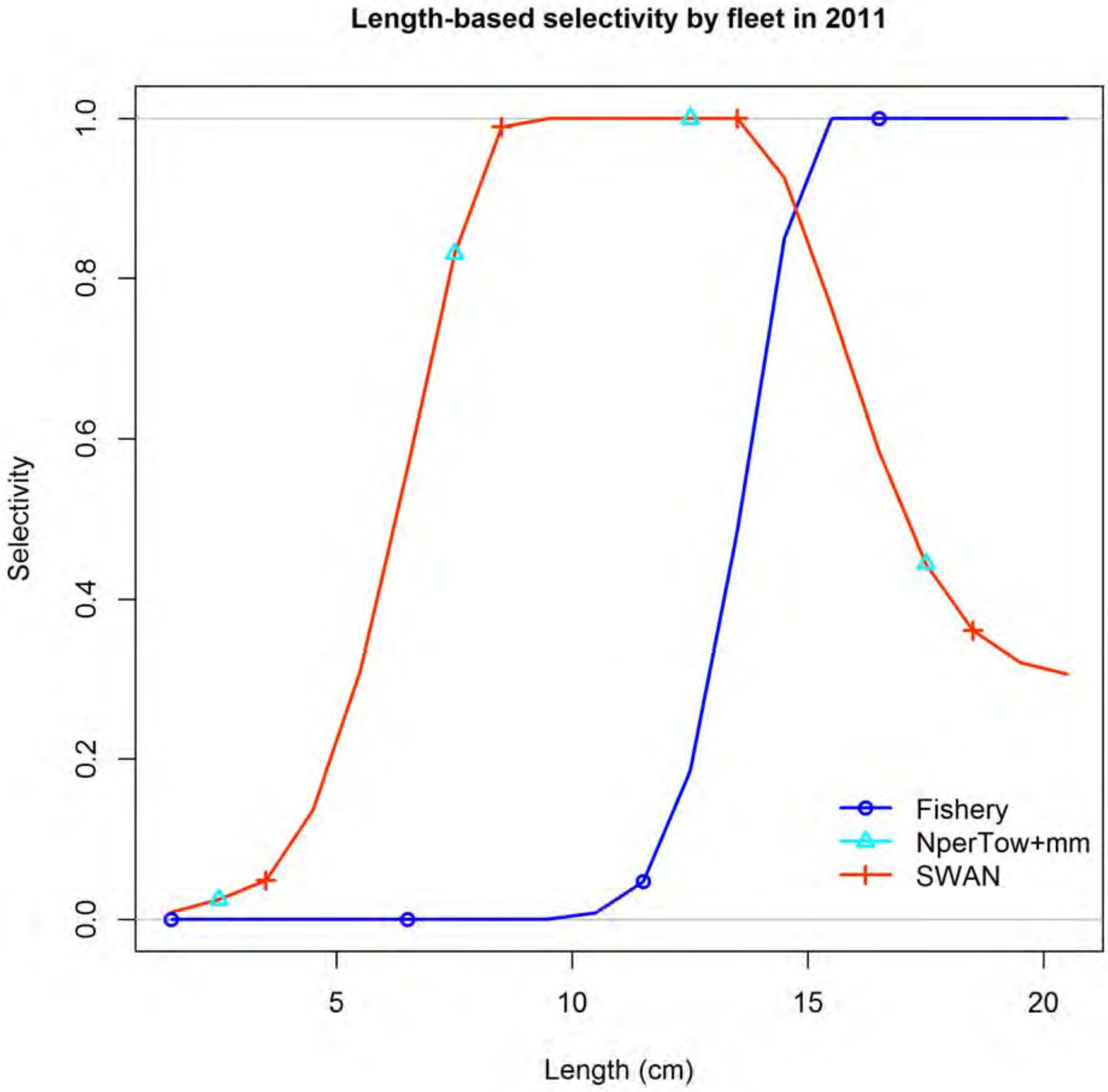



Log recruitment deviation
1.5
1.0
0.5
0.0
-0.5
-1.0
-1.5
1970
1980
1990
2000
2010
Year

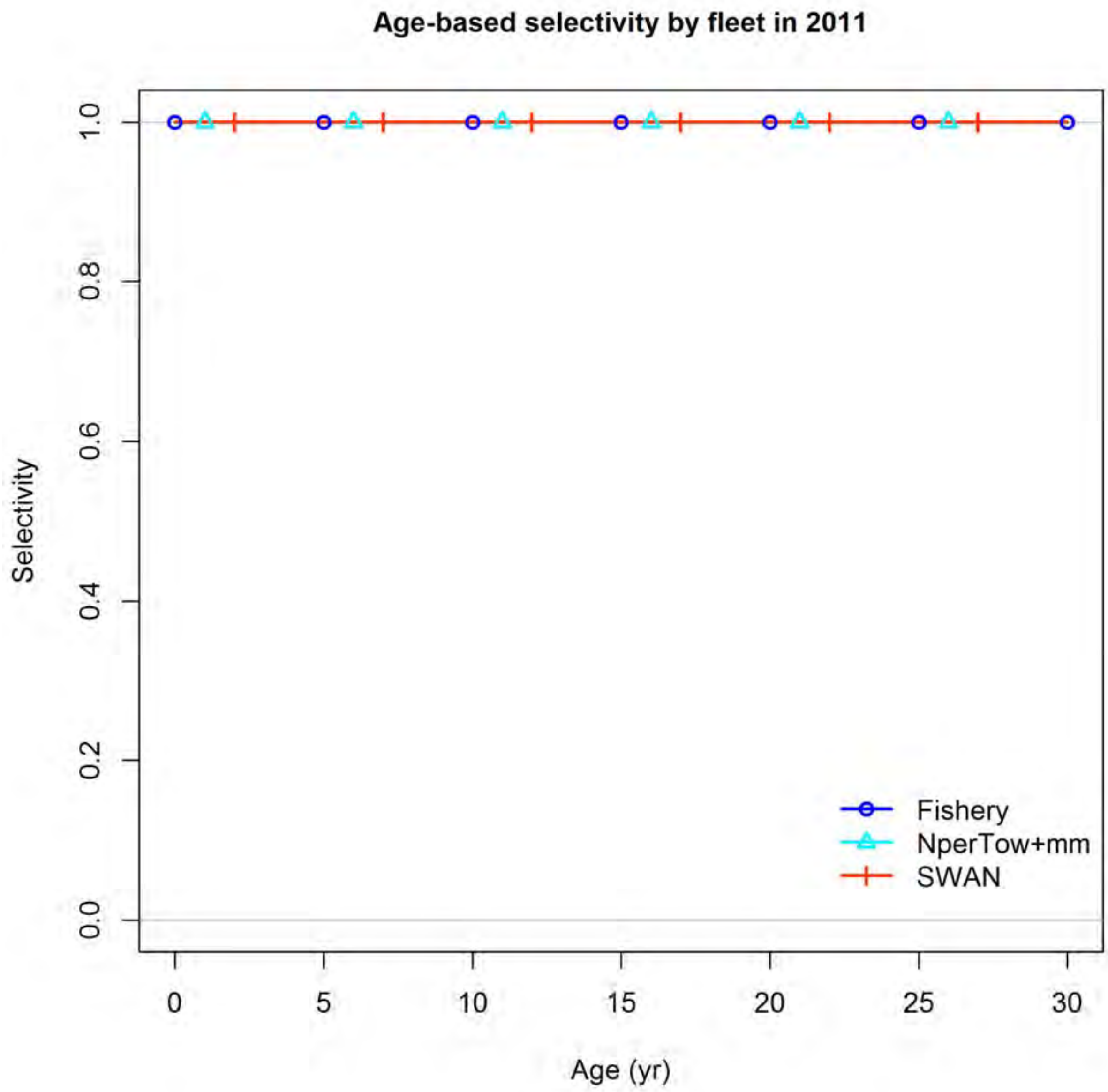

Log recruitment deviation
2
1
0
-1
-2
1970
1980
1990
2000
2010
Year


Recruitment deviation variance check
Asymptotic standard error estimate
0.0
0.2
0.4
0.6
0.8
1970
1980
1990
2000
2010
Year

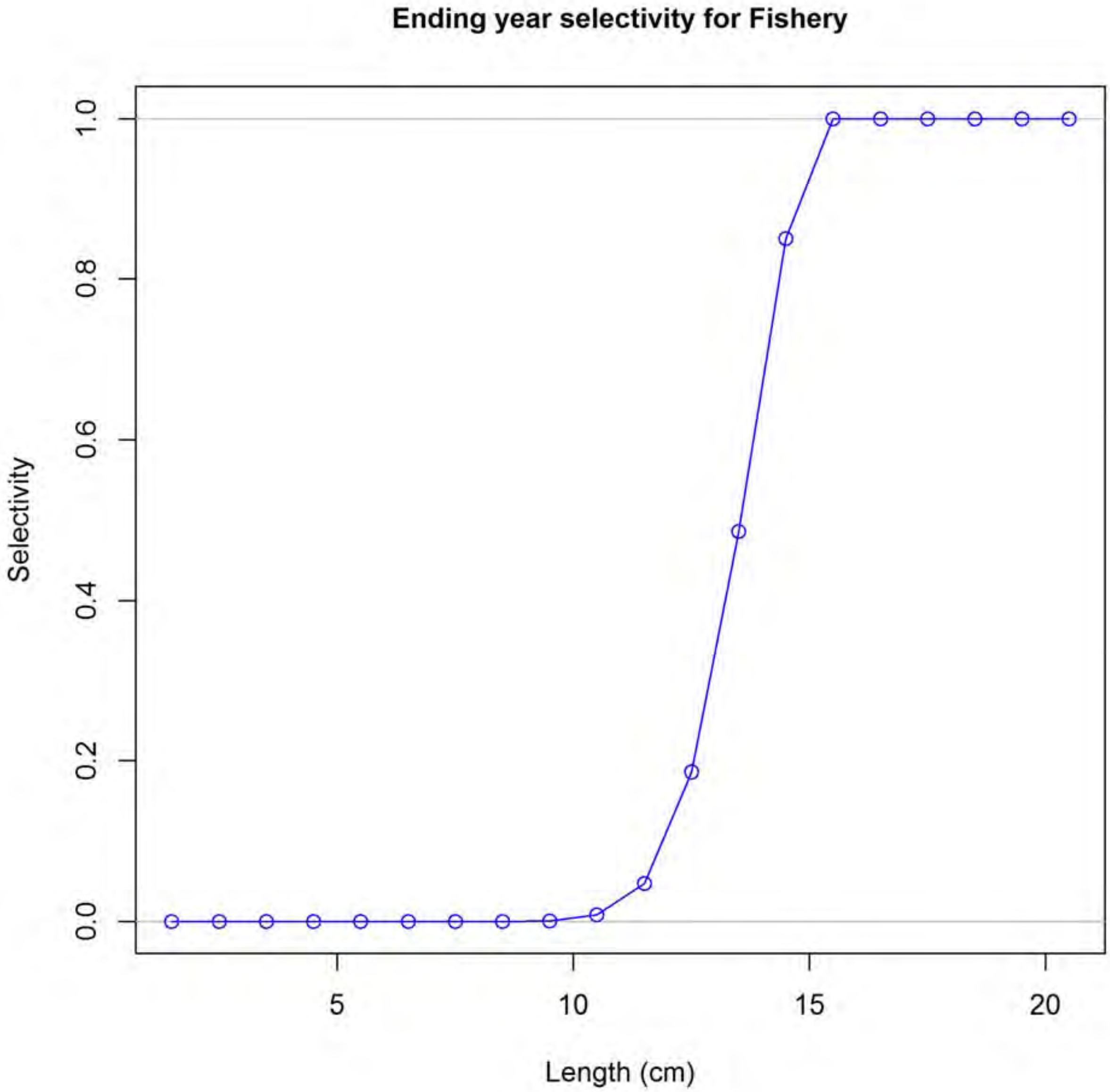

bias adjust in model
estimated alternative
$1 - SE(\hat{r}_y)^2/\sigma_R^2$
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1970
1980
1990
2000
2010
Year

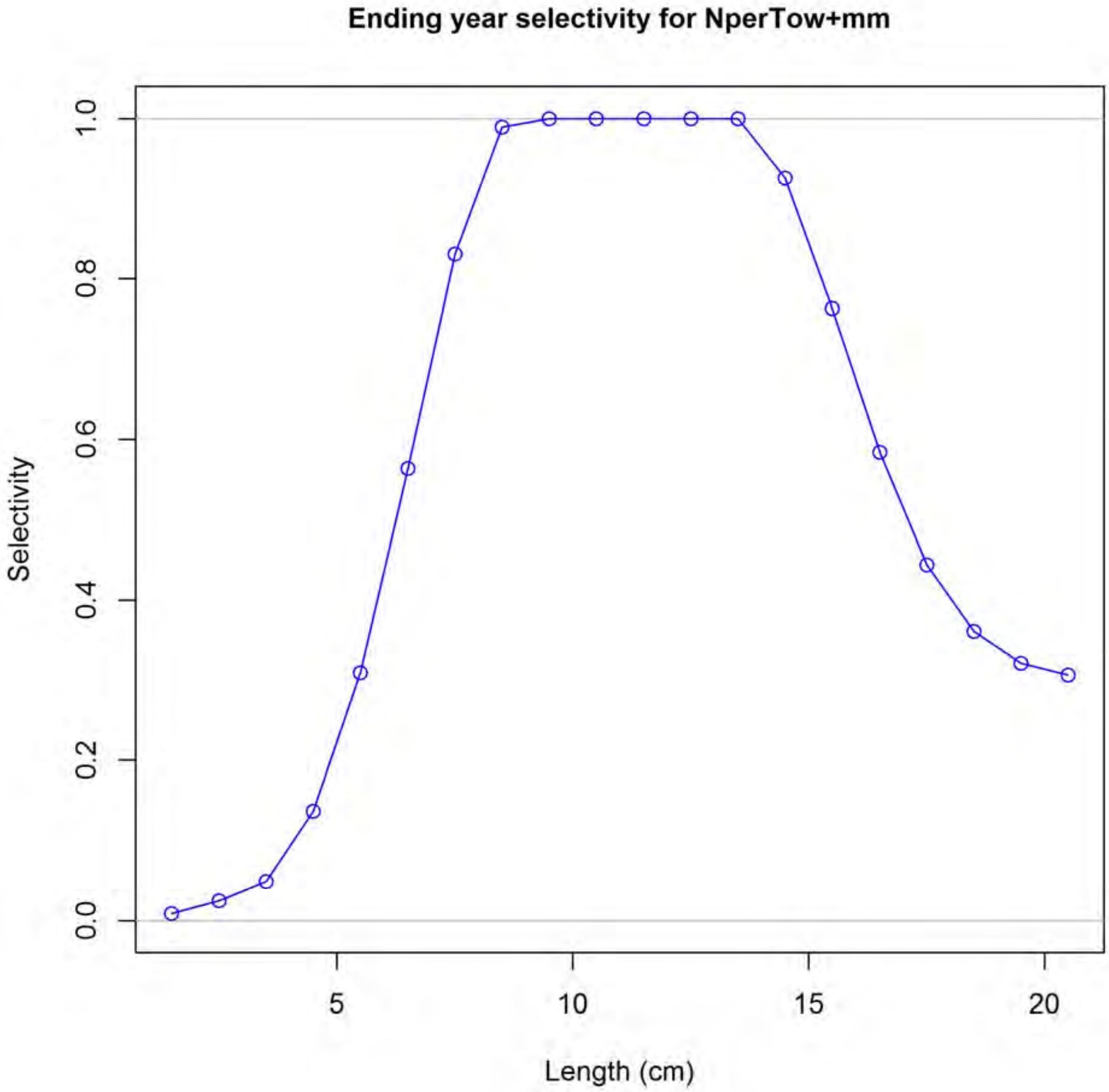


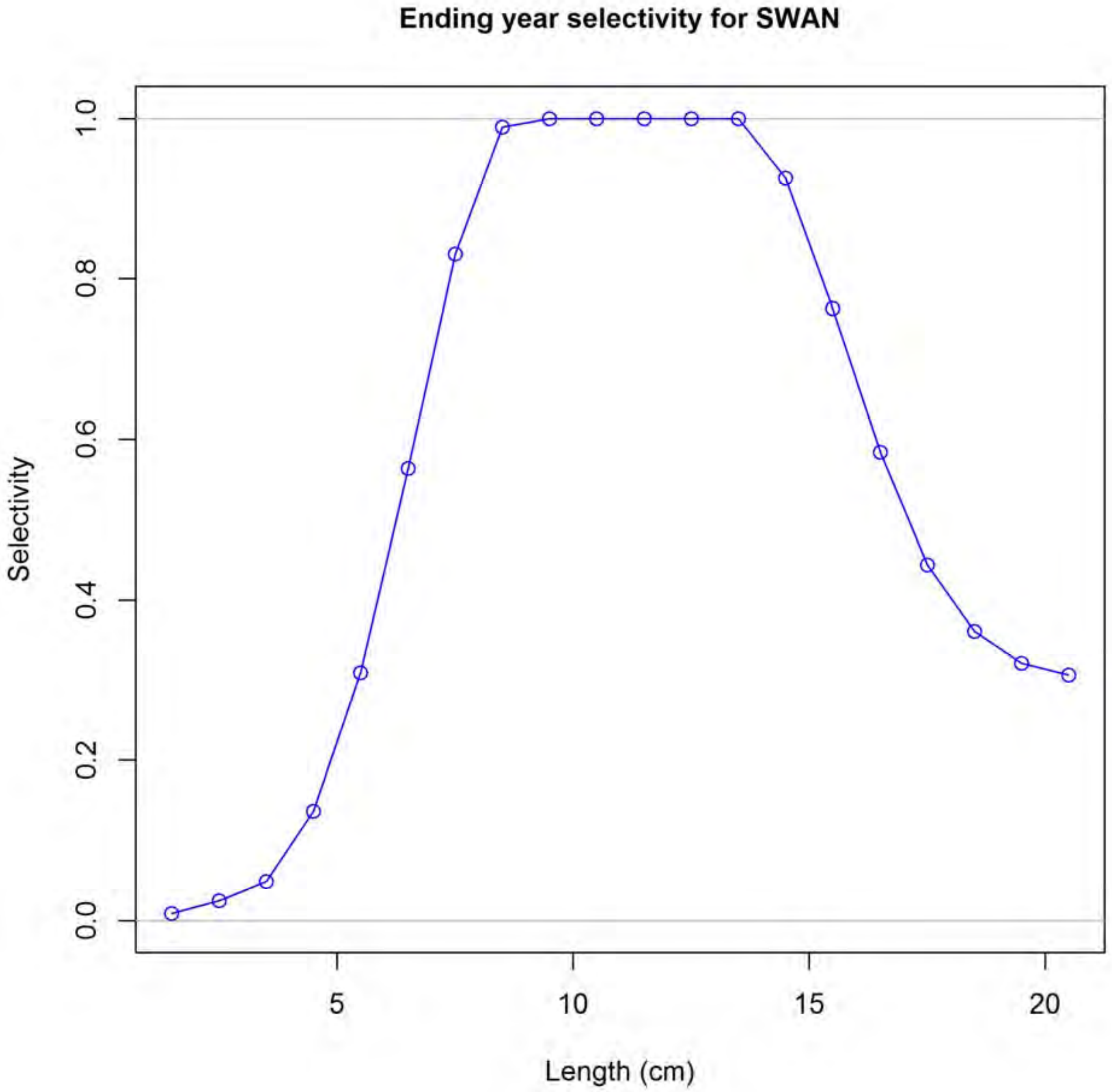
Length-based selectivity by fleet in 2011
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)
Fishery
NperTow+mm
SWAN


Age-based selectivity by fleet in 2011
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
Age (yr)
0
5
10
15
20
25
30
Fishery
NperTow+mm
SWAN

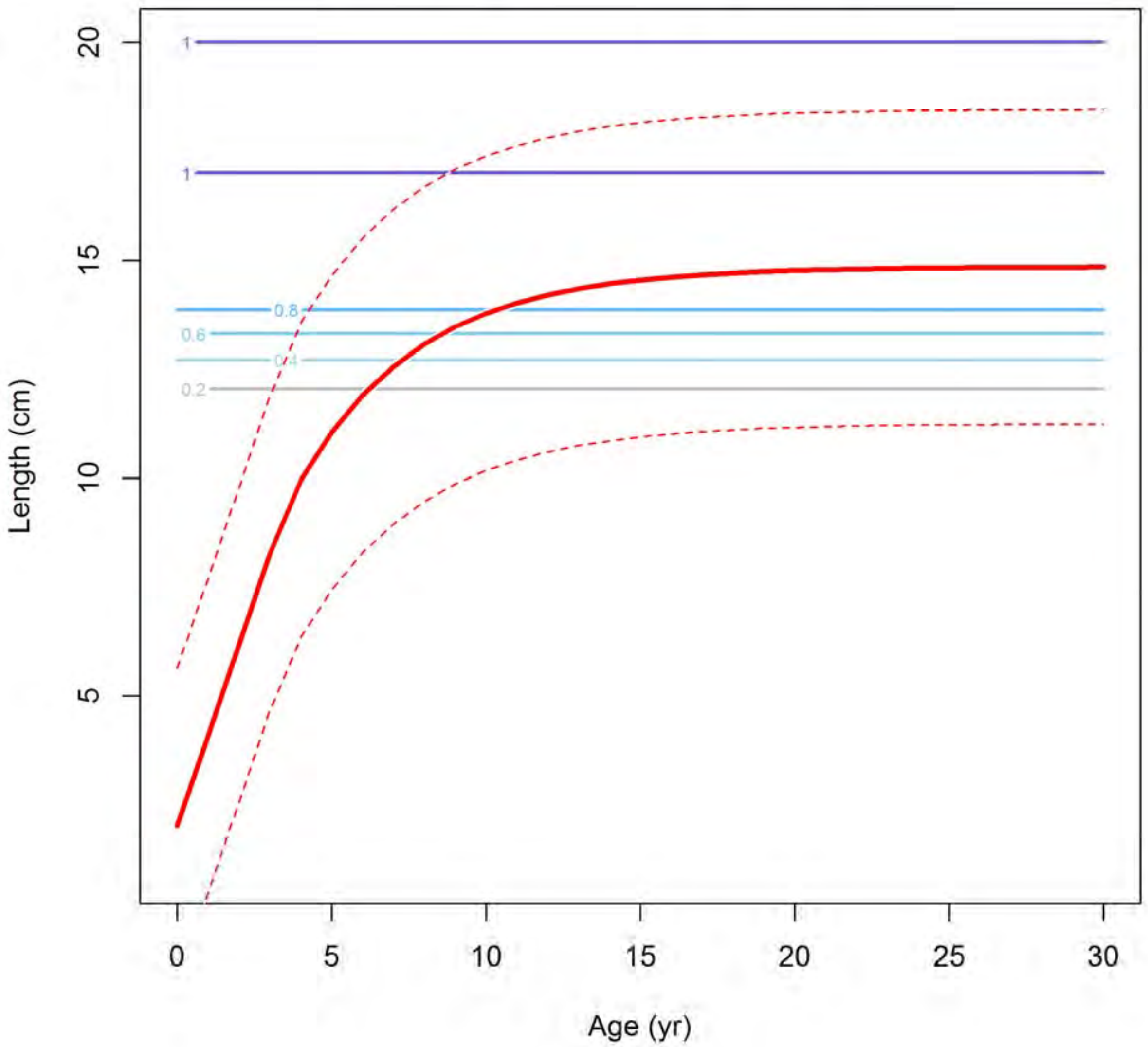
Ending year selectivity for Fishery

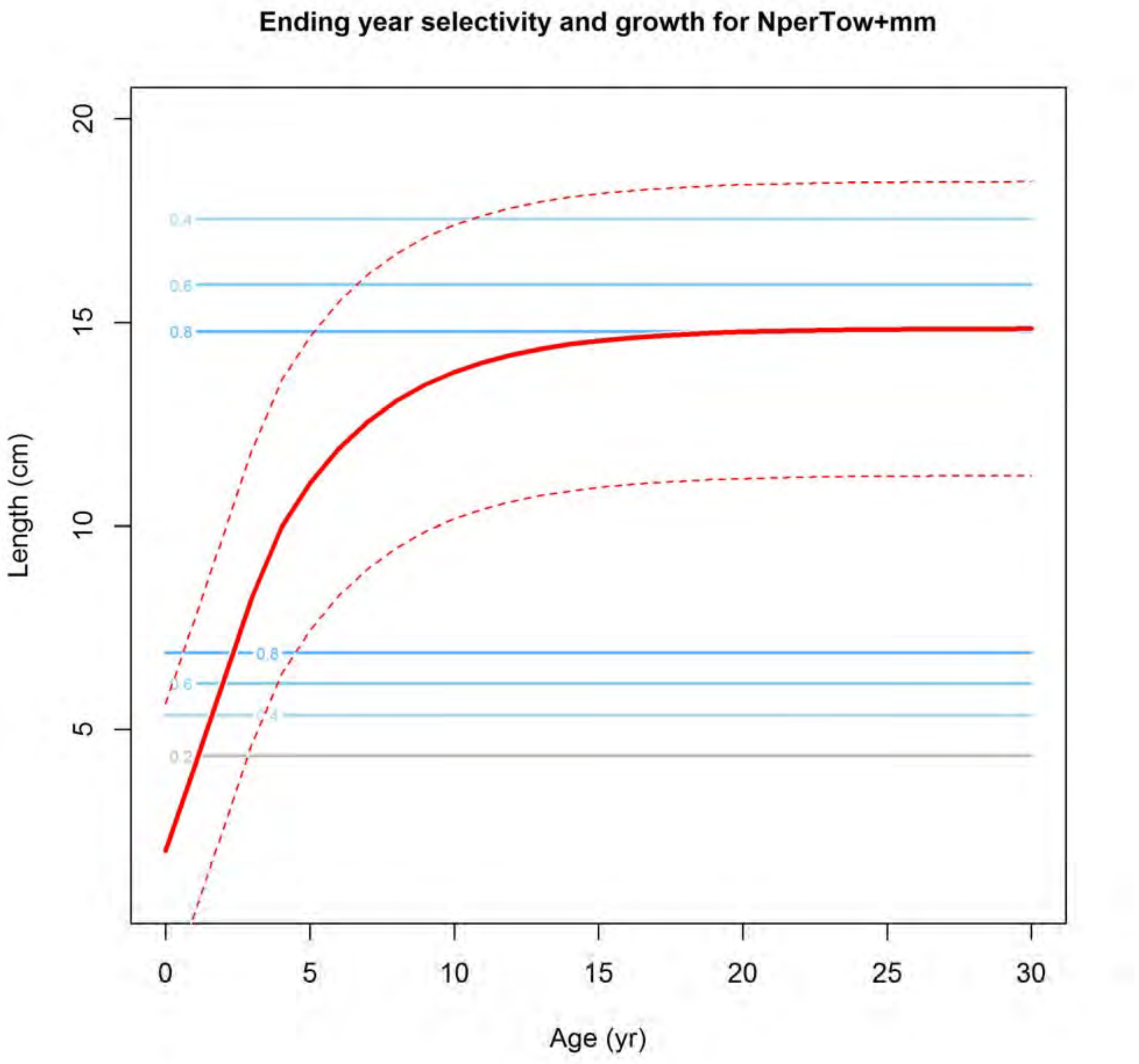


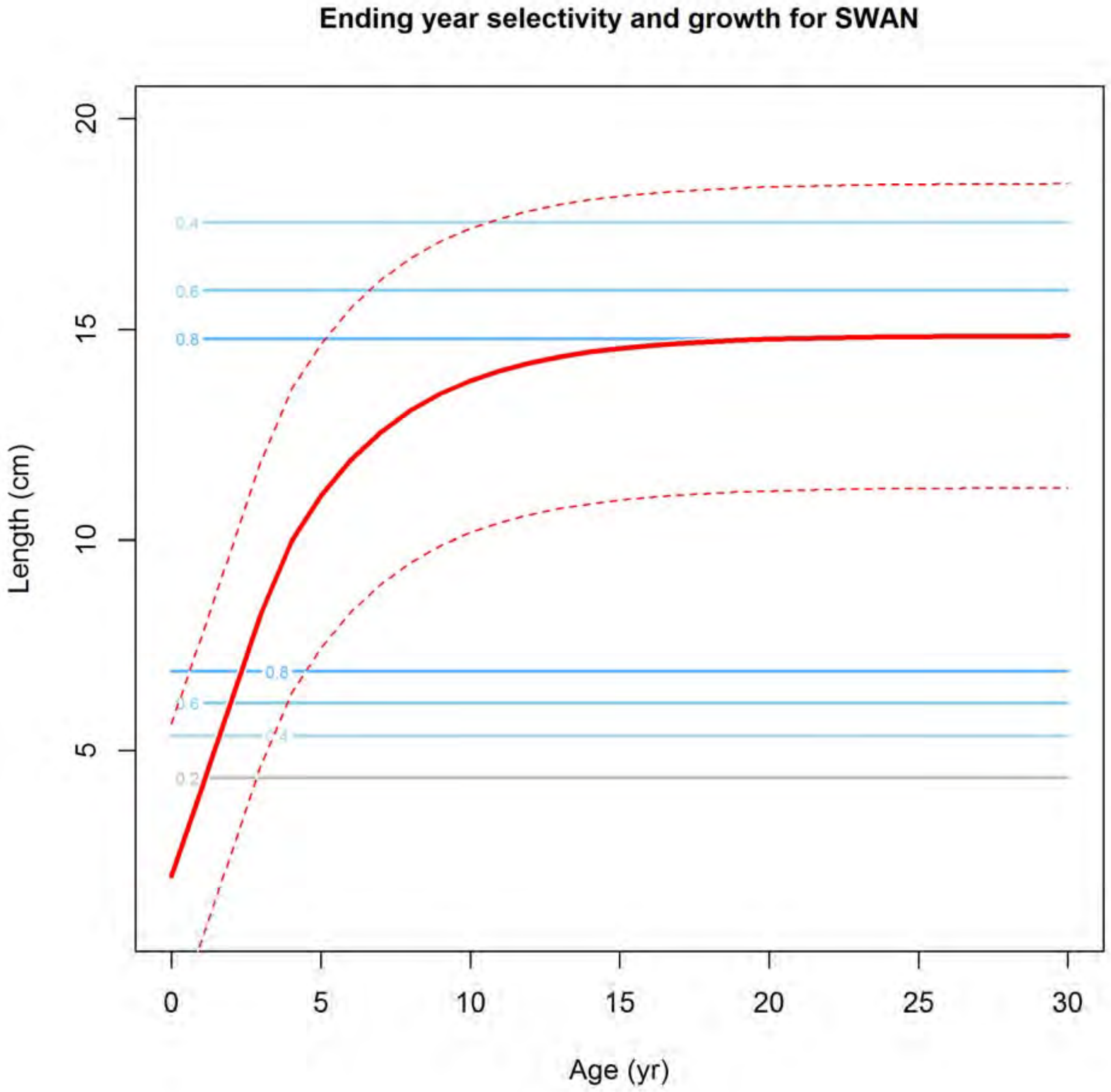

Ending year selectivity for NperTow+mm
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)

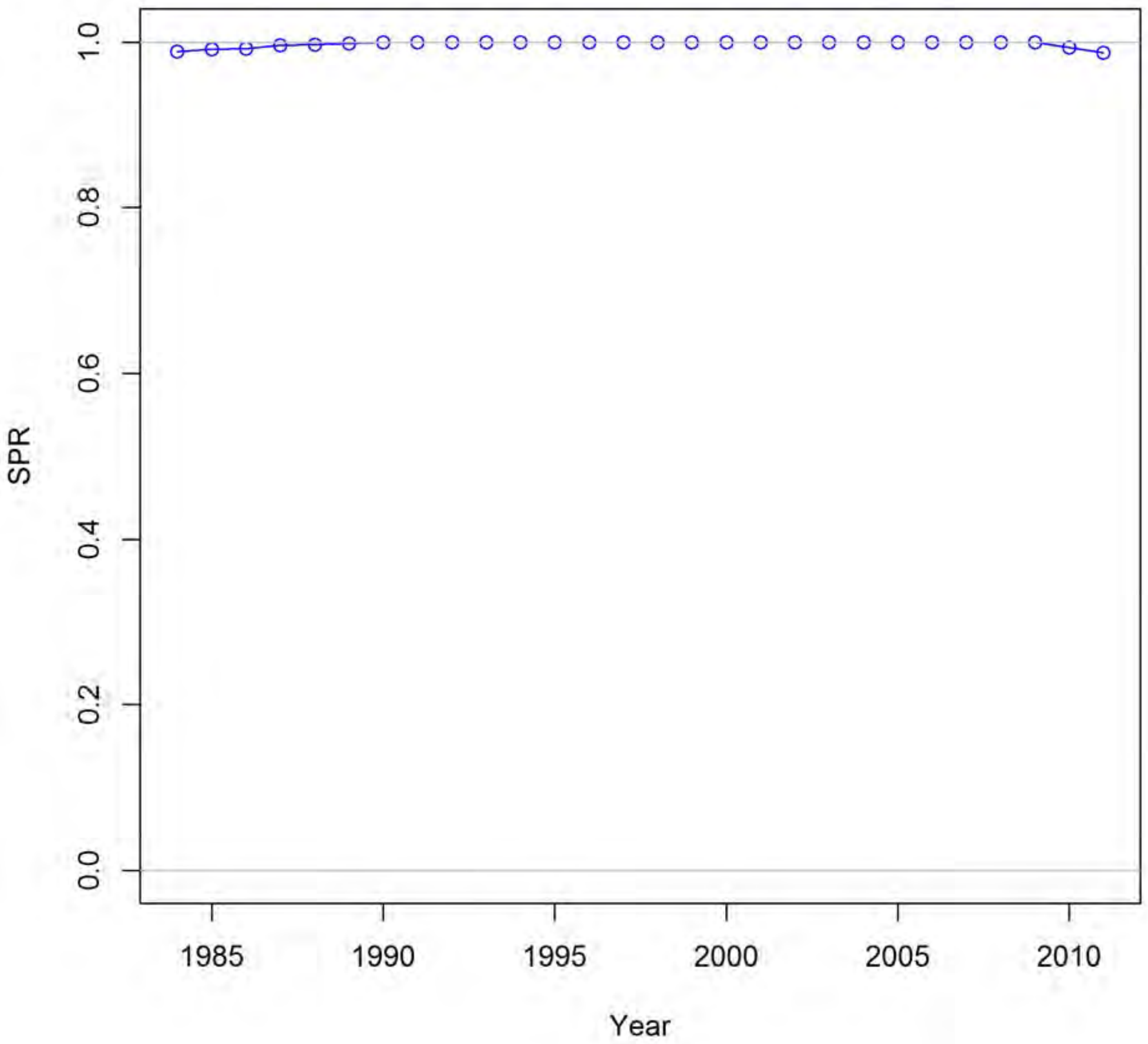

Ending year selectivity for SWAN
Selectivity
0.0
0.2
0.4
0.6
0.8
1.0
5
10
15
20
Length (cm)

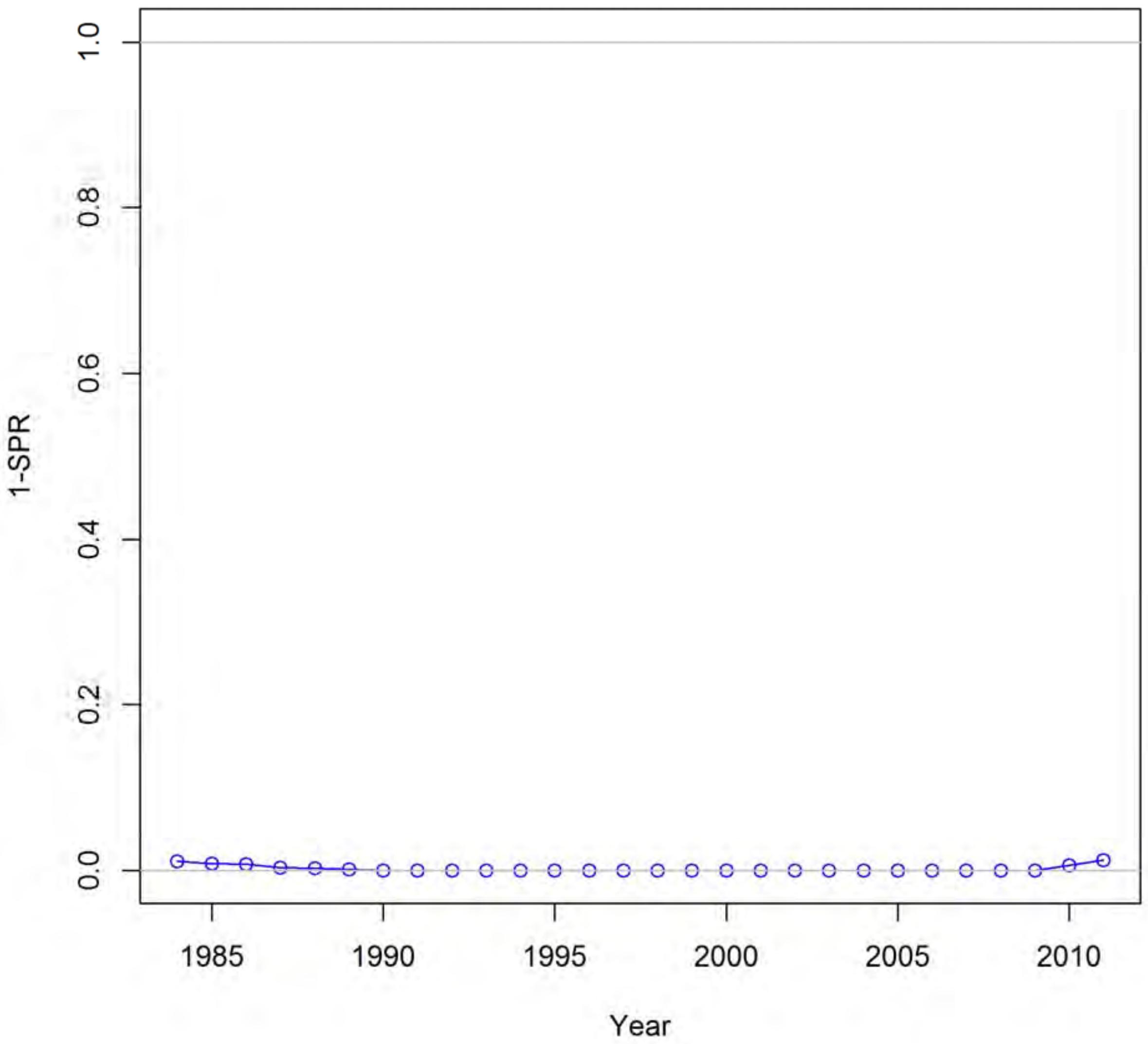
Ending year selectivity and growth for Fishery

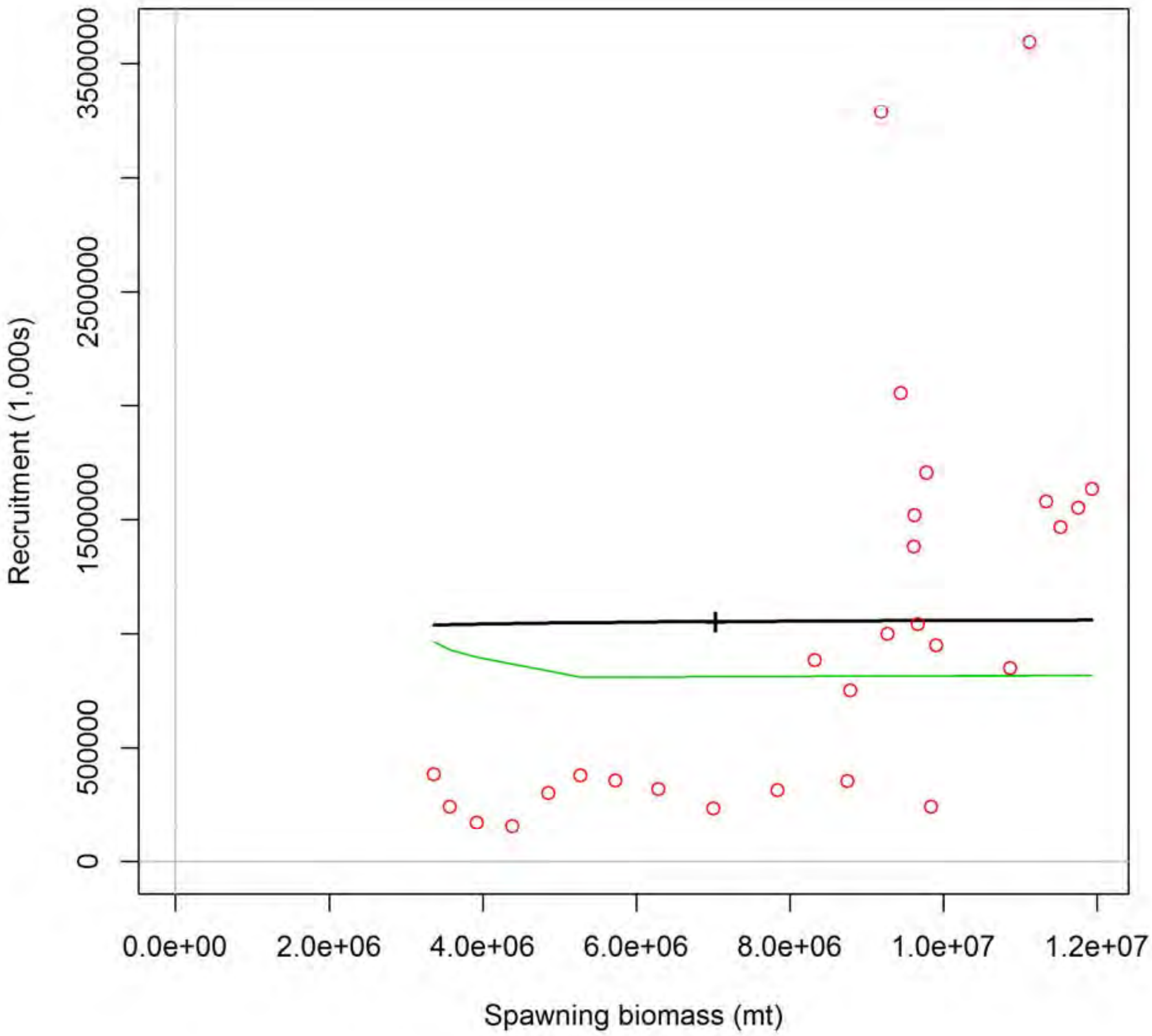


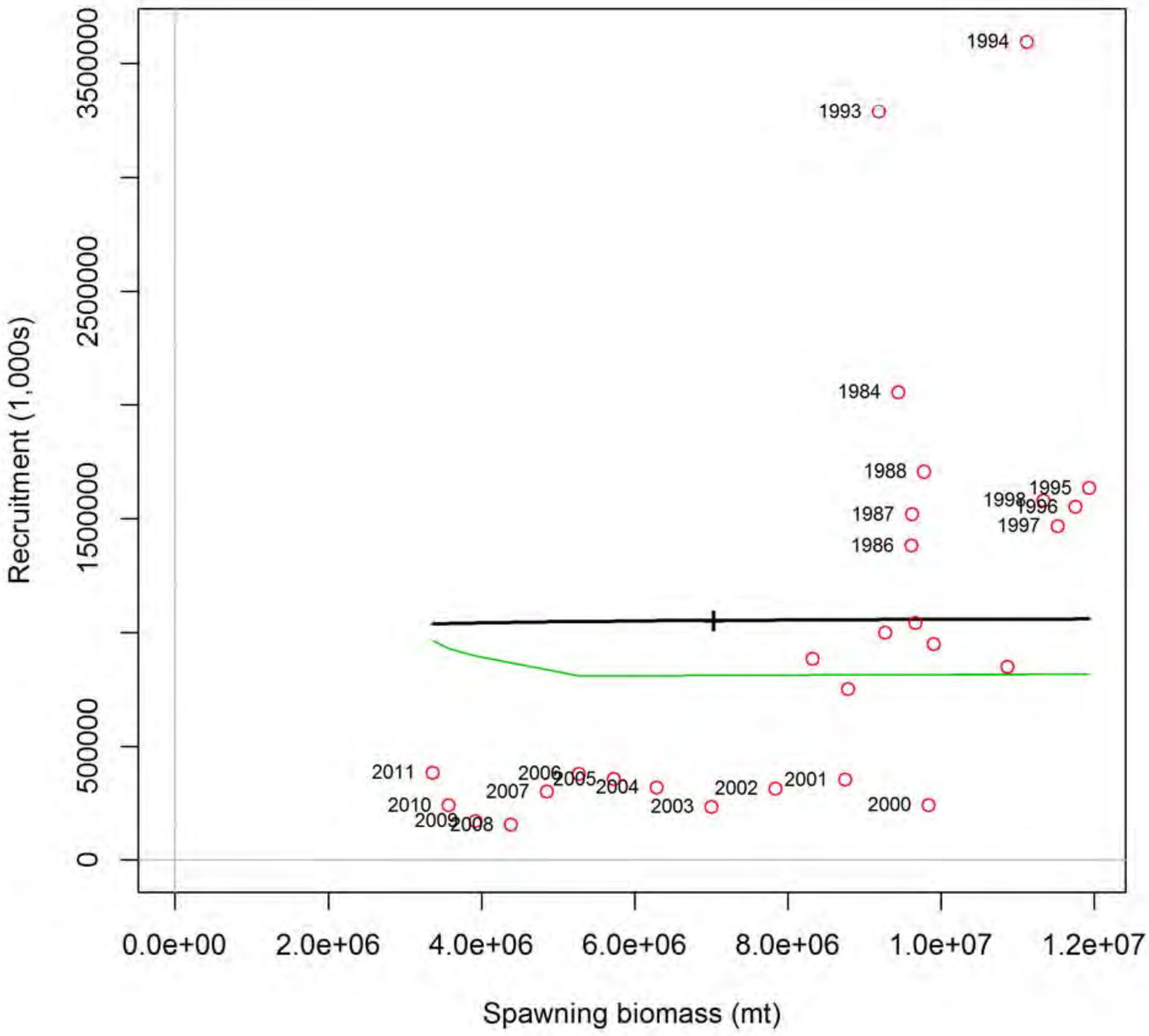

Ending year selectivity and growth for NperTow+mm
Length (cm)
Age (yr)
20
15
10
5
0
5
10
15
20
25
30
0.4
0.6
0.8
0.8
0.6
0.4
0.2


Ending year selectivity and growth for SWAN
Length (cm)
20
15
10
5
0.4
0.6
0.8
0.8
0.6
0.4
0.2
0
5
10
15
20
25
30
Age (yr)

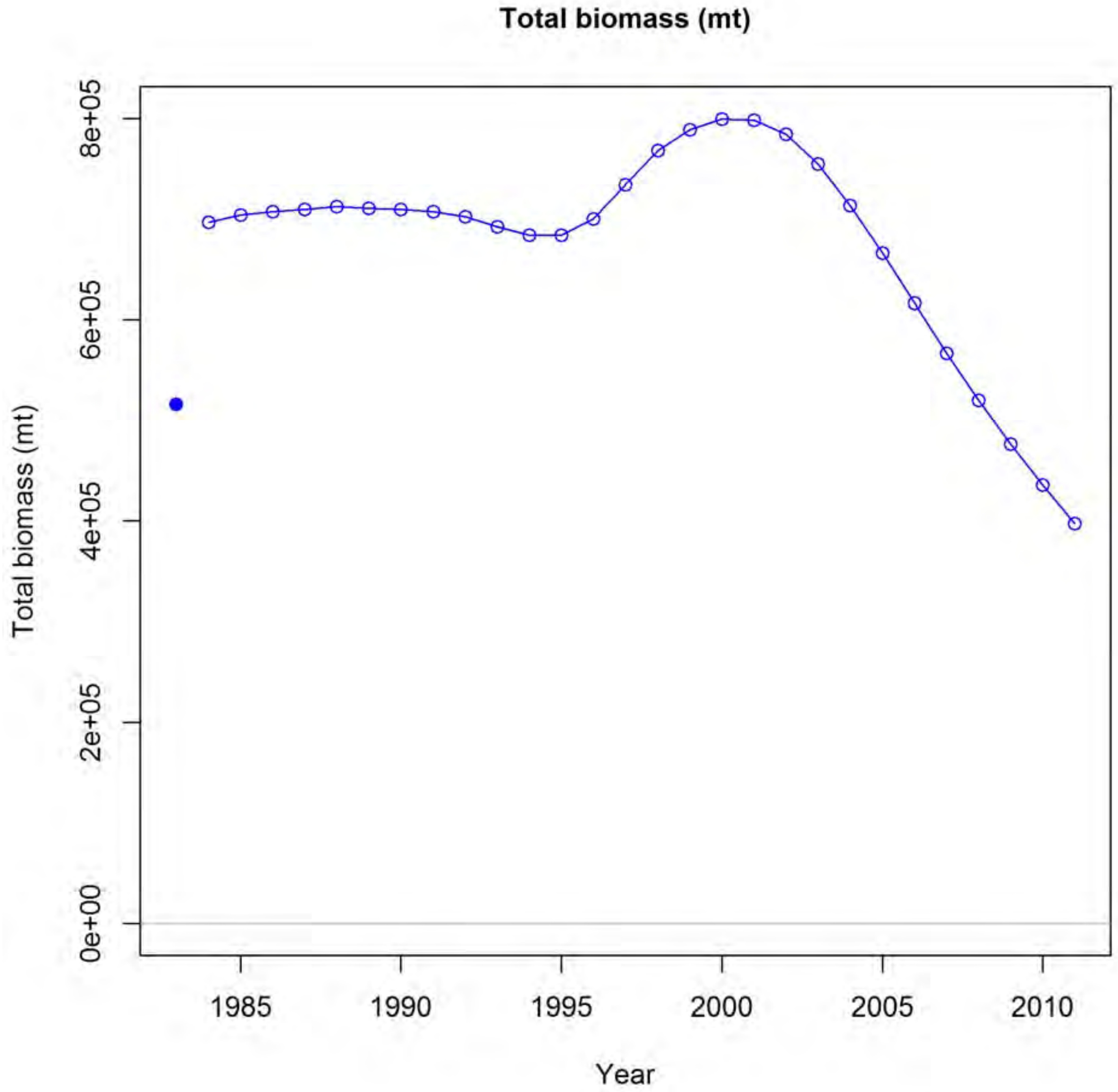

SPR
0.0
0.2
0.4
0.6
0.8
1.0
1985
1990
1995
2000
2005
2010
Year

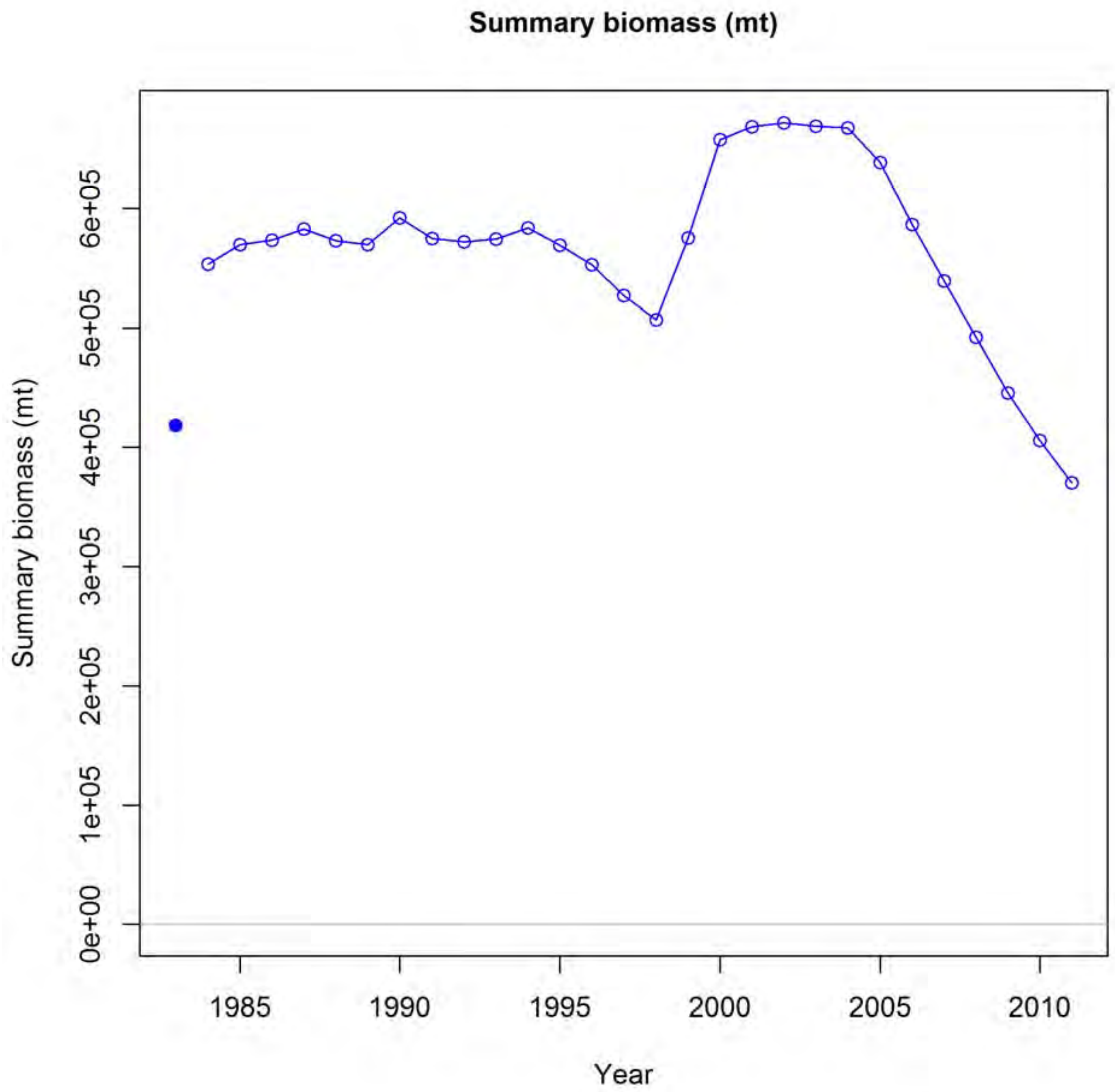

1-SPR
0.0
0.2
0.4
0.6
0.8
1.0
1985
1990
1995
2000
2005
2010
Year

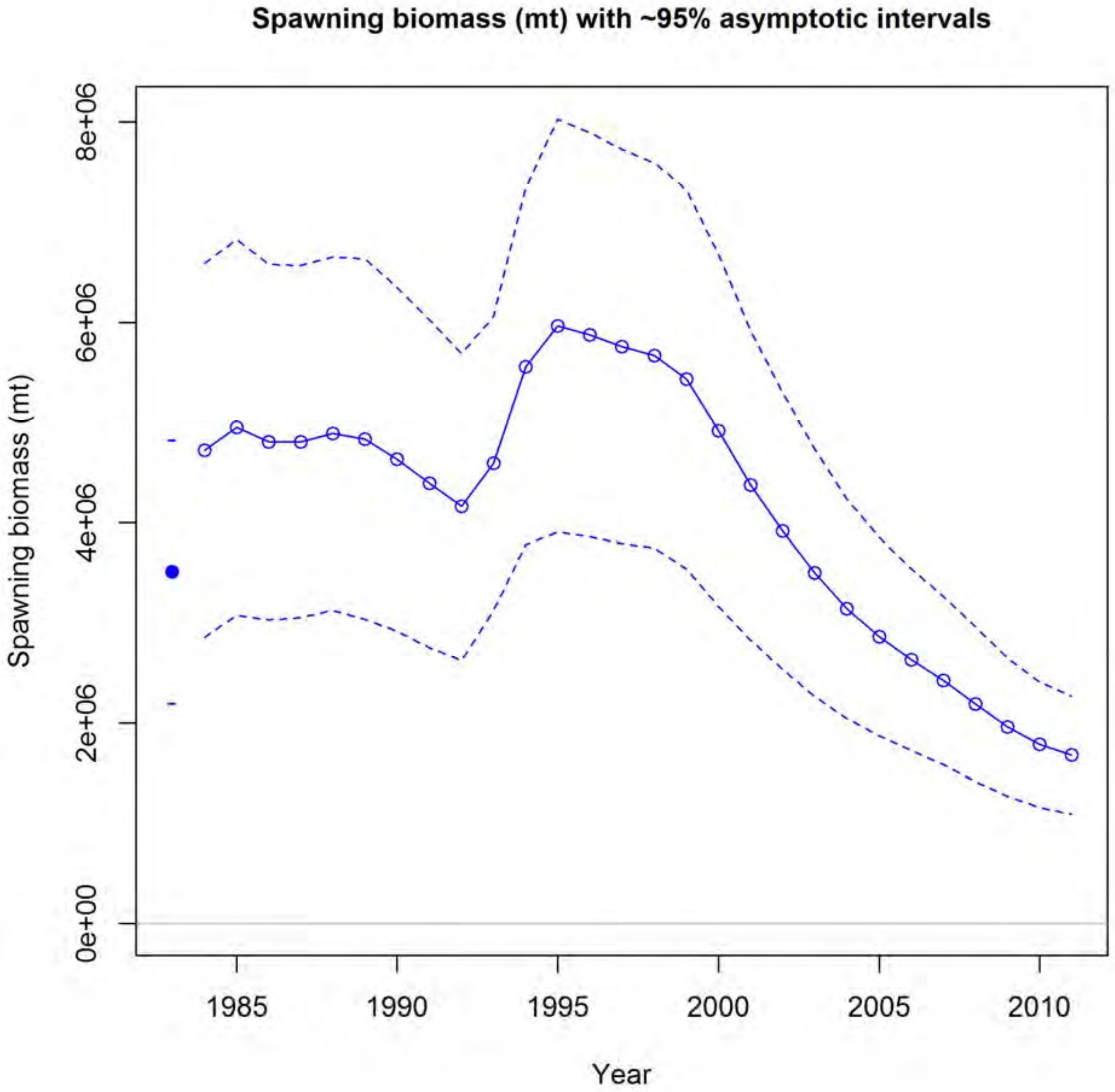

Recruitment (1,000s)
0
500000
1500000
2500000
3500000
0.0e+00
2.0e+06
4.0e+06
6.0e+06
8.0e+06
1.0e+07
1.2e+07
Spawning biomass (mt)



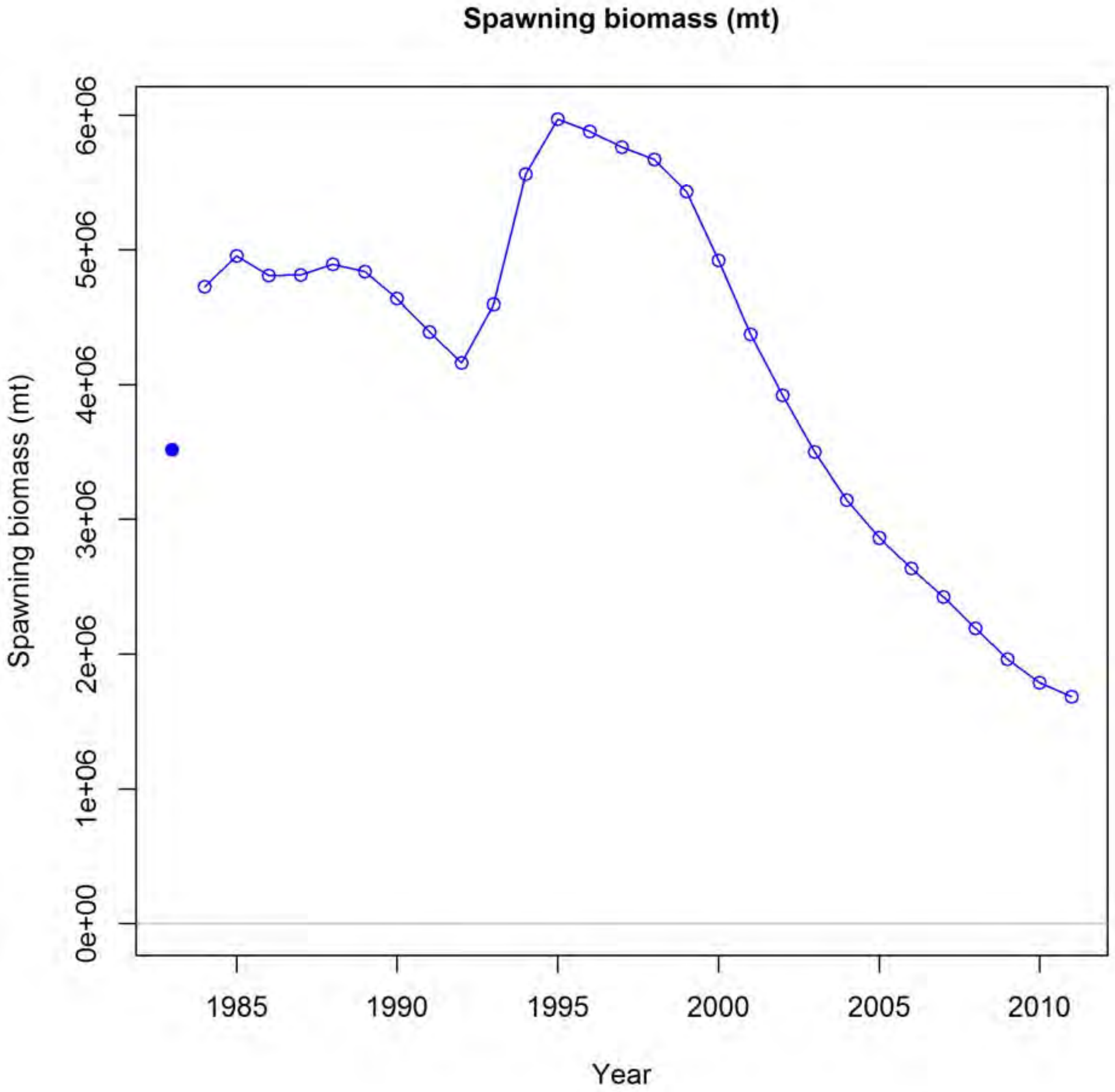
Recruitment (1,000s)
0
500000
1500000
2500000
3500000
Spawning biomass (mt)
0.0e+00
2.0e+06
4.0e+06
6.0e+06
8.0e+06
1.0e+07
1.2e+07
1994
1993
1984
1988
1987
1986
1995
1998
1996
1997
2011
2010
2009
2008
2007
2006
2005
2004
2003
2002
2001
2000

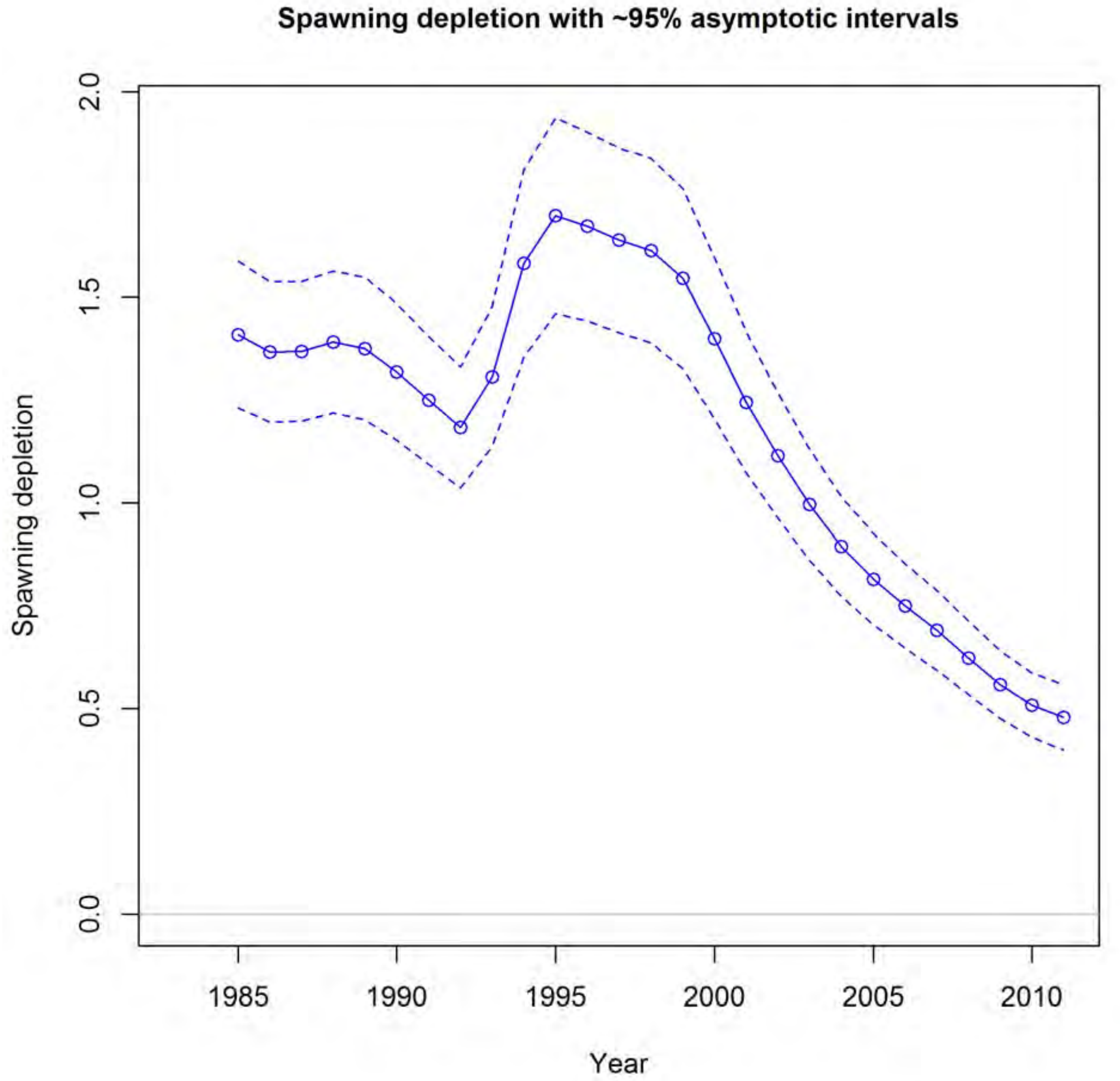

Total biomass (mt)
Total biomass (mt)
0e+00
2e+05
4e+05
6e+05
8e+05
1985
1990
1995
2000
2005
2010
Year

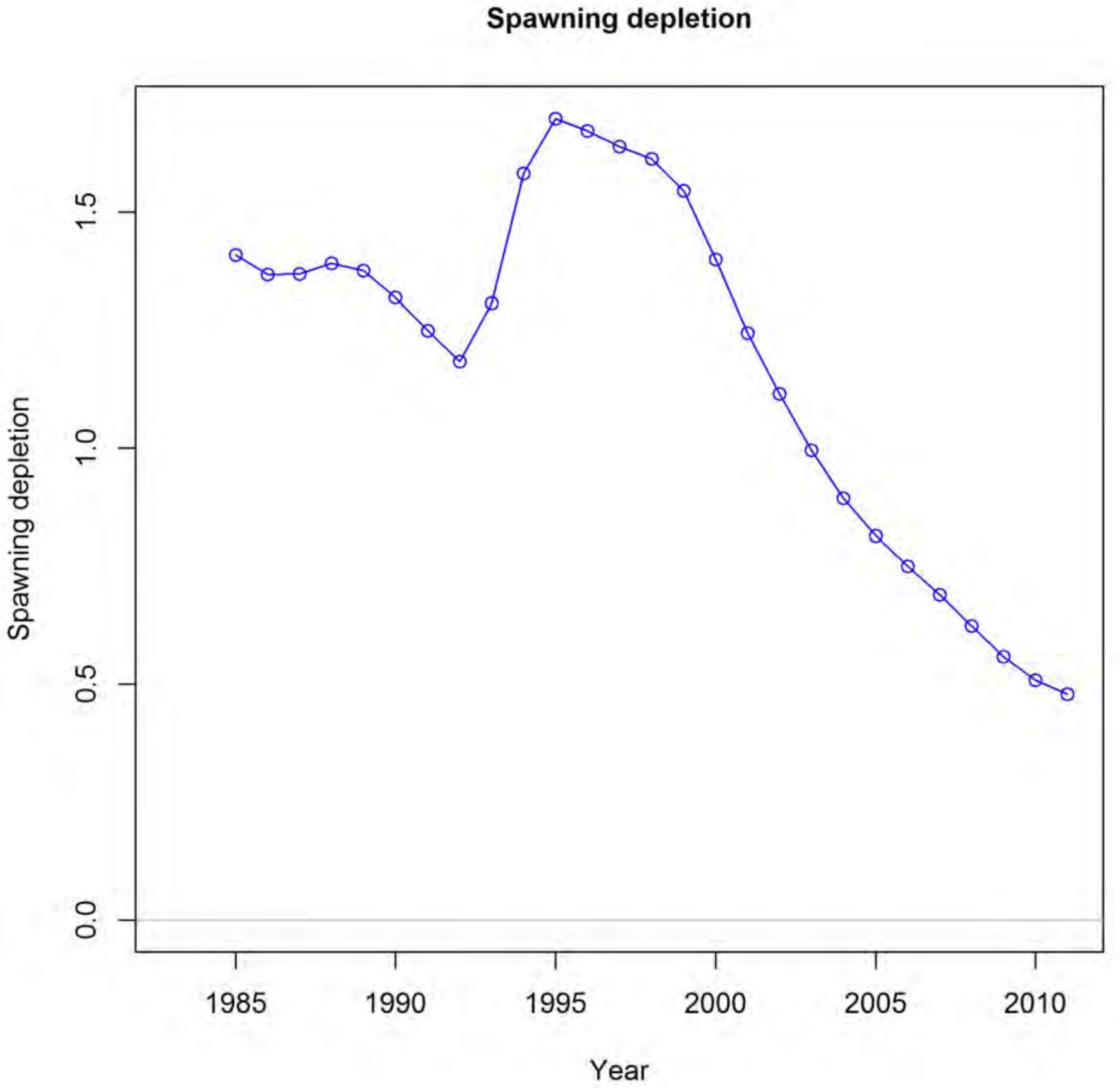

Summary biomass (mt)
Summary biomass (mt)
0e+00
1e+05
2e+05
3e+05
4e+05
5e+05
6e+05
1985
1990
1995
2000
2005
2010
Year



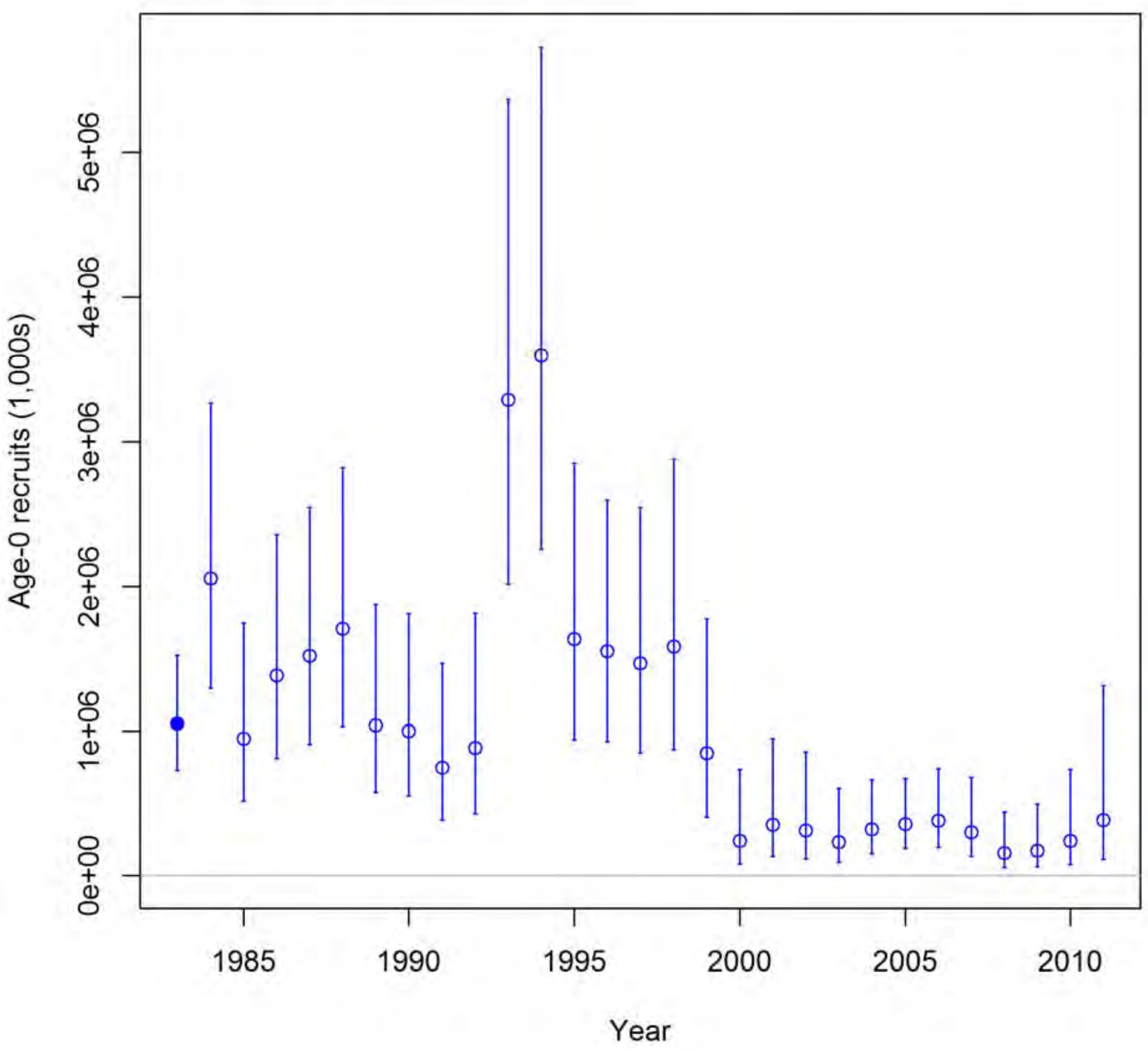
Spawning biomass (mt) with ~95% asymptotic intervals
Spawning biomass (mt)
0e+00
2e+06
4e+06
6e+06
8e+06
1985
1990
1995
2000
2005
2010
Year

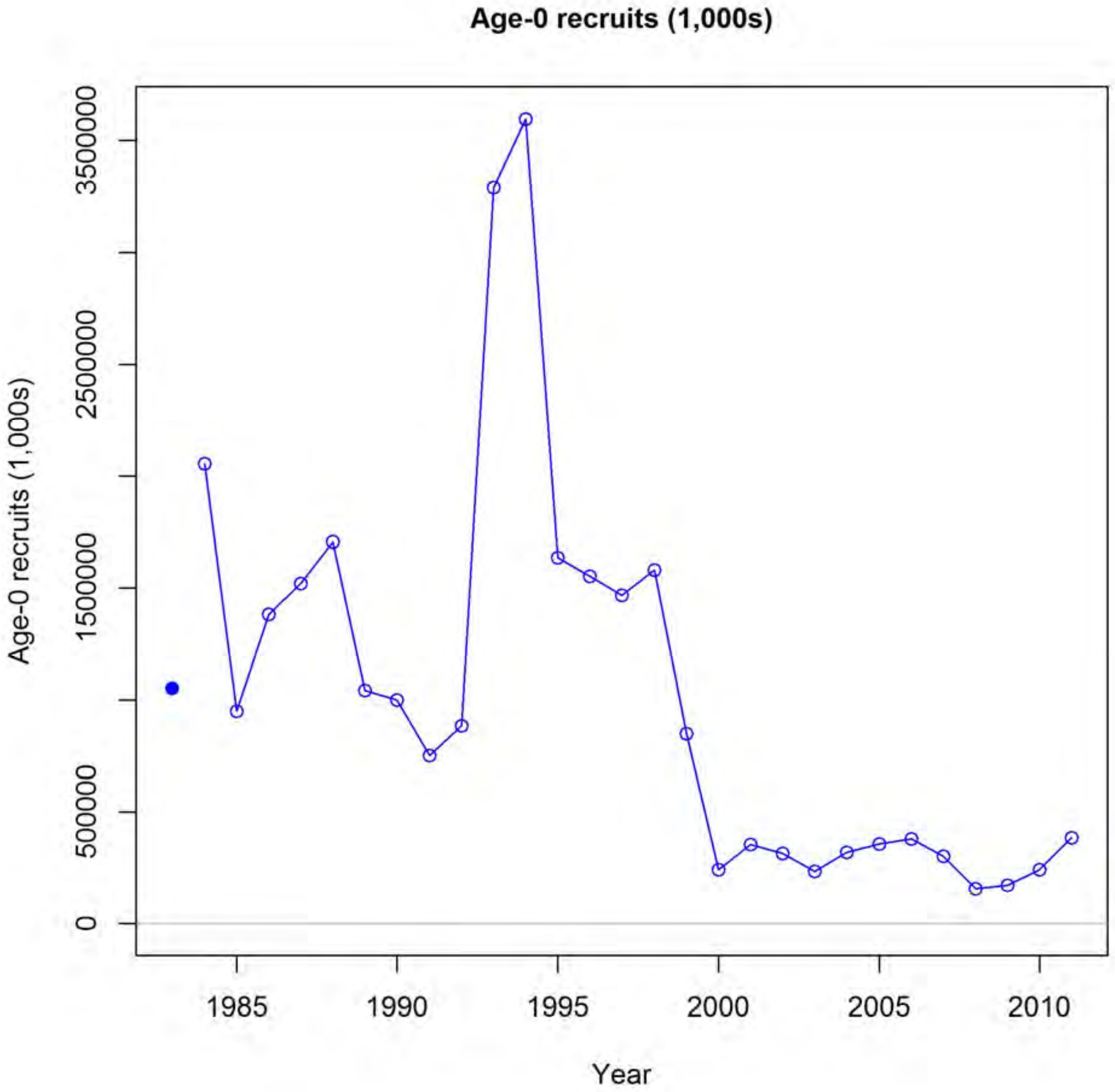

Spawning biomass (mt)
Spawning biomass (mt)
0e+00
1e+06
2e+06
3e+06
4e+06
5e+06
6e+06
1985
1990
1995
2000
2005
2010
Year

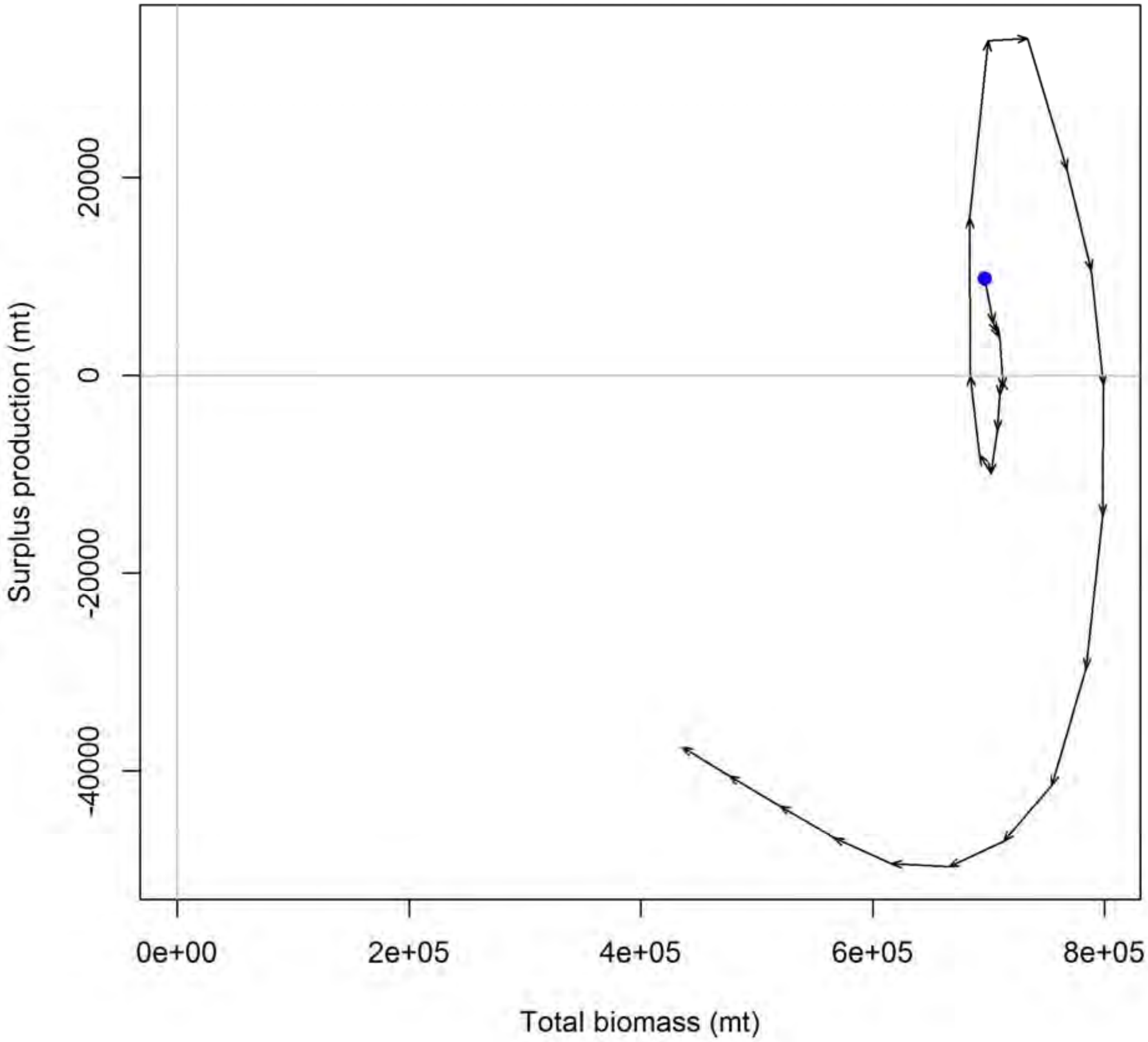

Spawning depletion with ~95% asymptotic intervals
Spawning depletion
0.0
0.5
1.0
1.5
2.0
1985
1990
1995
2000
2005
2010
Year


Spawning depletion
Spawning depletion
0.0
0.5
1.0
1.5
1985
1990
1995
2000
2005
2010
Year

Age-0 recruits (1,000s) with ~95% asymptotic intervals




Age-0 recruits (1,000s)
Age-0 recruits (1,000s)
0
500000
1500000
2500000
3500000
1985
1990
1995
2000
2005
2010
Year



Surplus production (mt)
20000
0
-20000
-40000
0e+00
2e+05
4e+05
6e+05
8e+05
Total biomass (mt)

Appendix A8: Swept area biomass analysis

Efficiency corrected swept-area biomass

Efficiency corrected swept area biomass and catch/biomass fishing mortality estimates have been used in past assessments to provide management advice. Although they no longer serve that purpose, they are still used to estimate scale in KLAMZ modeling.

Efficiency corrected swept area biomass and catch/biomass fishing mortality estimates were calculated with CVs for surfclams during 1997-2011 (years with dredge performance sensors deployed on surveys) on a regional basis, using the methods described in NEFSC (2010) (Table 1-2 and Figures 1-2).

Efficiency corrected swept-area biomass and fishing mortality estimates in this assessment for years prior to 2011 differ from estimates in previous assessments due to: 1) changes after the 2011 survey in the criteria used to judge a “bad” (with poor gear performance) survey tow; 2) the availability of data for 2011 that could be borrowed to help fill “holes” (unsampled strata) in the survey data for 2008; 3) new shell length meat weight relationships; 4) the updated estimate of survey dredge capture efficiency; and 5) use of a new survey dredge selectivity curve to calculate stock biomass.

A historical retrospective analysis was carried out to demonstrate the stability of efficiency corrected swept area biomass estimates. Swept-area biomass and fishing mortality calculations have changed from assessment to assessment as additional survey data accumulated and, mainly, as estimates of survey dredge efficiency were refined (Table 3, Figure 3).

Working group members were interested in seeing the ratio of swept area biomasses by region (Figure 4).

Appendix A8. Table 1. Efficiency corrected swept-area biomass estimates (1000 mt) and CVs for surfclams (120+ mm SL), by region.

	Estimate	CV
INPUT: Nominal tow distance (dn, nm)	0.15	
INPUT: Dredge width (nm)	0.00082	
Area swept per standard tow (a, nm2)	0.00012	10%

Area of assessment region (A, nm2) - no correction for stations with unsuitable clam habitat

S. Virginia and N. Carolina (SVA)	3,119	10%
Delmarva (DMV)	4,660	10%
New Jersey (NJ)	5,078	10%
Long Island (LI)	2,917	10%
Southern New England (SNE)	4,321	10%
Georges Bank (GBK)	5,772	10%
Total	25,867	

INPUT: Fraction suitable habitat (u)

S. Virginia and N. Carolina (SVA)	100%	10%
Delmarva (DMV)	100%	10%
New Jersey (NJ)	100%	10%
Long Island (LI)	100%	10%
Southern New England (SNE)	100%	10%
Georges Bank (GBK)	88%	10%

Habitat area in assessment region (A', nm2)			**INPUT: Biomass fraction in unsurveyd deep water**		
S. Virginia and N. Carolina (SVA)	3,119	14%	S. Virginia and N. Carolina (SVA)	0%	10%
Delmarva (DMV)	4,660	14%	Delmarva (DMV)	0%	10%
New Jersey (NJ)	5,078	14%	New Jersey (NJ)	0%	10%
Long Island (LI)	2,917	14%	Long Island (LI)	0%	10%
Southern New England (SNE)	4,321	14%	Southern New England (SNE)	0%	10%
Georges Bank (GBK)	5,079	14%	Georges Bank (GBK)	0%	10%

INPUT: Original survey mean catch from fishable stock (kg/tow, for tows adjusted to nominal tow distance using sensors)

	Estimates	CV	Estimates	CV	Estimates	CV	Estimates	CV	Estimates	CV	Estimates	CV
S. Virginia and N. Carolina (SVA) 120+ mm	0.0230	42%	0.0887	42%	0.4486	59%	0.0000	0%	0.0030	100%	0.0065	100%
Delmarva (DMV) 120+ mm	2.4641	19%	1.3336	18%	2.5392	20%	0.7967	16%	0.4146	34%	0.8732	43%
New Jersey (NJ) 120+ mm	6.3488	11%	4.5417	17%	3.8543	14%	2.3883	11%	3.9031	17%	1.8693	23%
Long Island (LI) 120+ mm	0.3672	66%	0.9268	51%	0.2407	64%	2.2825	36%	0.4535	24%	1.2362	35%
Southern New England (SNE) 120+ mm	1.4769	34%	0.8400	66%	0.6545	24%	0.6508	43%	1.2236	47%	0.2323	27%
Georges Bank (GBK) 120+ mm	2.0151	21%	2.4106	32%	2.2545	43%	3.9404	23%	4.3871	21%	3.8483	25%

Swept-area biomass without efficiency correction (B', 1000 mt):

S. Virginia and N. Carolina (SVA) 120+ mm	0.5817	47%	2.2433	47%	11.3402	63%	0.0000	20%	0.0753	102%	0.1641	102%
Delmarva (DMV) 120+ mm	93.0714	28%	50.3714	27%	95.9086	28%	30.0930	26%	15.6612	39%	32.9812	47%
New Jersey (NJ) 120+ mm	261.3123	23%	186.9338	26%	158.6390	24%	98.2987	23%	160.6465	26%	76.9379	31%
Long Island (LI) 120+ mm	8.6828	69%	21.9131	55%	5.6915	67%	53.9670	41%	10.7226	32%	29.2277	40%
Southern New England (SNE) 120+ mm	51.7246	39%	29.4211	69%	22.9215	31%	22.7916	47%	42.8541	51%	8.1361	34%
Georges Bank (GBK) 120+ mm	82.9608	29%	99.2444	38%	92.8198	47%	162.2261	31%	180.6177	29%	158.4357	32%
SVA to SNE	415	17%	291	19%	295	16%	205	17%	230	21%	147	21%
Total (including GBK)	498	15%	390	17%	387	17%	367	17%	411	17%	306	19%
INPUT: Survey dredge efficiency (e) from Patch mo	0.234	132%	0.234	132%	0.234	132%	0.234	132%	0.234	132%	0.234	132%

Efficiency adjusted swept area fishable biomass (B, 1000 mt)

S. Virginia and N. Carolina (SVA) 120+ mm	2.486	140%	9.587	140%	48.463	146%	0.000	134%	0.322	167%	0.701	167%
Delmarva (DMV) 120+ mm	398	135%	215	135%	410	135%	129	134%	67	138%	141	140%
New Jersey (NJ) 120+ mm	1,117	134%	799	135%	678	134%	420	134%	687	135%	329	136%
Long Island (LI) 120+ mm	37	149%	94	143%	24	148%	231	138%	46	136%	125	138%
Southern New England (SNE) 120+ mm	221	138%	126	149%	98	136%	97	140%	183	141%	35	136%
Georges Bank (GBK) 120+ mm	355	135%	424	137%	397	140%	693	136%	772	135%	677	136%
SVA to SNE	1,775	133%	1,243	133%	1,259	133%	877	133%	983	134%	630	134%
Total (including GBK)	2,130	133%	1,667	133%	1,655	133%	1,570	133%	1,755	133%	1,307	133%

Lower bound for 80% confidence intervals on fishable biomass (1000 mt, for lognormal distribution with no bias correction)

	Estimates	Estimates	Estimates	Estimates	Estimates	Estimates
S. Virginia and N. Carolina (SVA) 120+ mm	0.655	2.526	12.338		0.074	0.160
Delmarva (DMV) 120+ mm	108	59	111	35	18	37
New Jersey (NJ) 120+ mm	305	217	185	115	187	89
Long Island (LI) 120+ mm	9	24	6	61	12	33
Southern New England (SNE) 120+ mm	59	32	26	26	48	9
Georges Bank (GBK) 120+ mm	96	114	104	188	209	183
SVA to SNE	488	341	346	241	269	172
Total (including GBK)	586	458	455	431	482	358

Upperbound for 80% confidence intervals on fishable biomass (1000 mt, for lognormal distribution with no bias correction)

S. Virginia and N. Carolina (SVA) 120+ mm	9.433	36.381	190.363		1.409	3.070
Delmarva (DMV) 120+ mm	1,464	792	1,509	472	251	535
New Jersey (NJ) 120+ mm	4,089	2,936	2,485	1,538	2,522	1,215
Long Island (LI) 120+ mm	148	362	97	866	170	468
Southern New England (SNE) 120+ mm	827	502	362	370	700	129
Georges Bank (GBK) 120+ mm	1,308	1,584	1,507	2,562	2,847	2,505
SVA to SNE	6,461	4,535	4,580	3,192	3,590	2,302
Total (including GBK)	7,741	6,072	6,026	5,715	6,391	4,769

Appendix A8.Table 2. Fishing mortality estimates for surfclams based on catch and efficiency corrected swept area biomass estimates.

INPUT: Incidental mortality allowance	12%
INPUT: Assumed CV for catch	10%

INPUT: Landings (1000 mt, discard ~ 0)	**Estimates for 1997**	**Estimates for 1999**	**Estimates for 2002**	**Estimates for 2005**	**Estimates for 2008**	**Estimates for 2011**
S. Virginia and N. Carolina (SVA)	0.000	0.000	0.064	0.000	0.000	0.000
Delmarva (DMV)	1.540	0.648	4.489	1.668	3.223	1.427
New Jersey (NJ)	16.998	18.749	18.271	16.850	17.517	11.908
Long Island (LI)	0.073	0.157	1.130	0.759	1.317	0.437
Southern New England (SNE)	0.000	0.016	0.052	1.885	0.423	2.420
Georges Bank (GBK)	0.000	0.000	0.000	0.000	0.000	2.397
Total	18.611	19.570	24.006	21.163	22.481	18.589

Catch (1000 mt, landings + upper bound incidental mortality allowance)						
S. Virginia and N. Carolina (SVA)	0.000	0.000	0.072	0.000	0.000	0.000
Delmarva (DMV)	1.725	0.726	5.028	1.868	3.610	1.598
New Jersey (NJ)	19.038	20.999	20.463	18.872	19.619	13.337
Long Island (LI)	0.081	0.176	1.265	0.850	1.475	0.489
Southern New England (SNE)	0.000	0.018	0.058	2.112	0.474	2.710
Georges Bank (GBK)	0.000	0.000	0.000	0.000	0.000	2.685
Total	20.844	21.919	26.886	23.702	25.178	20.820

INPUT: Efficiency Corrected Swept Area Biomass for Fishable Stock (1000 mt)	**Estimates for 1997**	**CV**	**Estimates for 1999**	**CV**	**Estimates for 2002**	**CV**	**Estimates for 2005**	**CV**	**Estimates for 2008**	**CV**	**Estimates for 2011**	**CV**
S. Virginia and N. Carolina (SVA) 120+ mm	2	140%	10	140%	48	146%	0	134%	0	167%	1	167%
Delmarva (DMV) 120+ mm	398	135%	215	135%	410	135%	129	134%	67	138%	141	140%
New Jersey (NJ) 120+ mm	1,117	134%	799	135%	678	134%	420	134%	687	135%	329	136%
Long Island (LI) 120+ mm	37	149%	94	143%	24	148%	231	138%	46	136%	125	138%
Southern New England (SNE) 120+ mm	221	138%	126	149%	98	136%	97	140%	183	141%	35	136%
Georges Bank (GBK) 120+ mm	355	135%	424	137%	397	140%	693	136%	772	135%	677	136%
SVA to SNE	1,775	133%	1,243	133%	1,259	133%	877	133%	983	134%	630	134%
Total (including GBK)	2,130	133%	1,667	133%	1,655	133%	1,570	133%	1,755	133%	1,307	133%

Fishing mortality (y^{-1})												
S. Virginia and N. Carolina (SVA) 120+ mm	0.0000	NA	0.0000	NA	0.0015	146%	0.0000	NA	0.0000	NA	0.0000	NA
Delmarva (DMV) 120+ mm	0.0043	135%	0.0034	135%	0.0123	135%	0.0145	135%	0.0539	138%	0.0113	141%
New Jersey (NJ) 120+ mm	0.0170	134%	0.0263	135%	0.0302	135%	0.0449	134%	0.0286	135%	0.0406	136%
Long Island (LI) 120+ mm	0.0022	149%	0.0019	143%	0.0520	148%	0.0037	139%	0.0322	136%	0.0039	138%
Southern New England (SNE) 120+ mm	0.0000	138%	0.0001	149%	0.0006	136%	0.0217	141%	0.0026	142%	0.0780	137%
Georges Bank (GBK) 120+ mm	0.0000	NA	0.0000	NA	0.0000	NA	0.0000	NA	0.0000	NA	0.0040	136%
SVA to SNE	0.0117	133%	0.0176	134%	0.0214	133%	0.0270	133%	0.0256	134%	0.0400	134%
Total (including GBK)	0.0098	133%	0.0131	134%	0.0162	133%	0.0151	133%	0.0143	134%	0.0193	134%

Lower bound for 80% confidence intervals for fishing mortality (y^{-1}, for lognormal distribution with no bias correction)	**Estimates for 1997**	**Estimates for 1999**	**Estimates for 2002**	**Estimates for 2005**	**Estimates for 2008**	**Estimates for 2011**
S. Virginia and N. Carolina (SVA) 120+ mm	NA	NA	0.0004	NA	NA	NA
Delmarva (DMV) 120+ mm	0.0012	0.0009	0.0033	0.0039	0.0144	0.0030
New Jersey (NJ) 120+ mm	0.0046	0.0071	0.0082	0.0122	0.0078	0.0110
Long Island (LI) 120+ mm	0.0005	0.0005	0.0131	0.0010	0.0087	0.0010
Southern New England (SNE) 120+ mm	NA	0.0000	0.0002	0.0057	0.0007	0.0210
Georges Bank (GBK) 120+ mm	NA	NA	NA	NA	NA	0.0011
SVA to SNE	0.0032	0.0048	0.0059	0.0074	299.3489	0.0070
Total (including GBK)	0.0027	0.0036	0.0045	0.0041	628.5781	0.0039

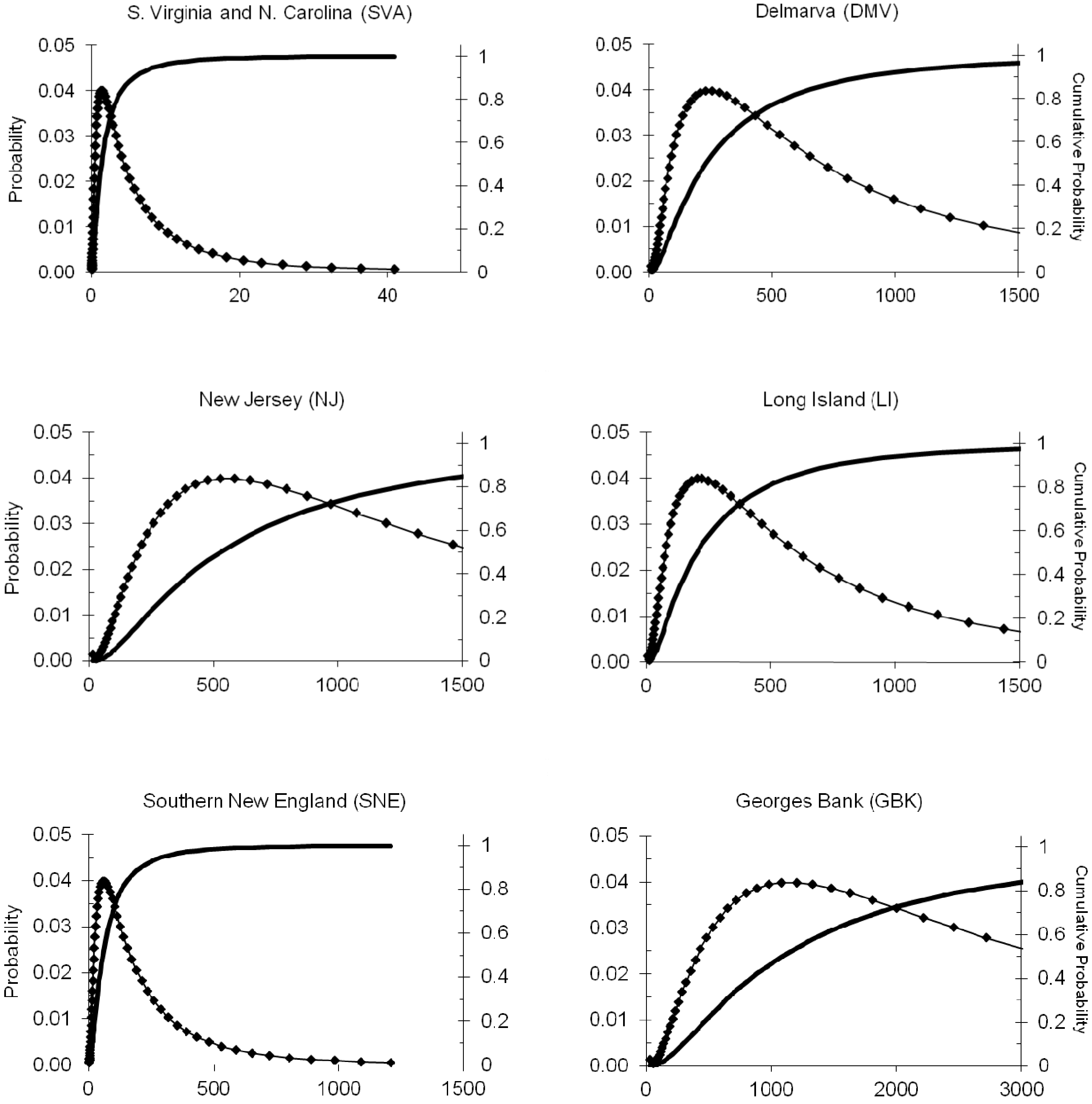
Upper bound for 80% confidence intervals for fishing mortality (y^{-1}, for lognormal distribution with no bias correction)						
S. Virginia and N. Carolina (SVA) 120+ mm	NA	NA	0.0059	NA	NA	NA
Delmarva (DMV) 120+ mm	0.0160	0.0124	0.0453	0.0535	0.2024	0.0091
New Jersey (NJ) 120+ mm	0.0626	0.0968	0.1109	0.1648	0.1052	0.0458
Long Island (LI) 120+ mm	0.0088	0.0073	0.2069	0.0139	0.1194	0.0023
Southern New England (SNE) 120+ mm	NA	0.0006	0.0022	0.0825	0.0099	0.1090
Georges Bank (GBK) 120+ mm	NA	NA	NA	NA	NA	NA
SVA to SNE	0.0428	0.0645	0.0779	0.0986	0.0938	0.0447
Total (including GBK)	0.0357	0.0480	0.0593	0.0551	0.0524	0.0175

Appendix A8. Table 3. Historical retrospective analysis of efficiency corrected swept area biomass estimates.

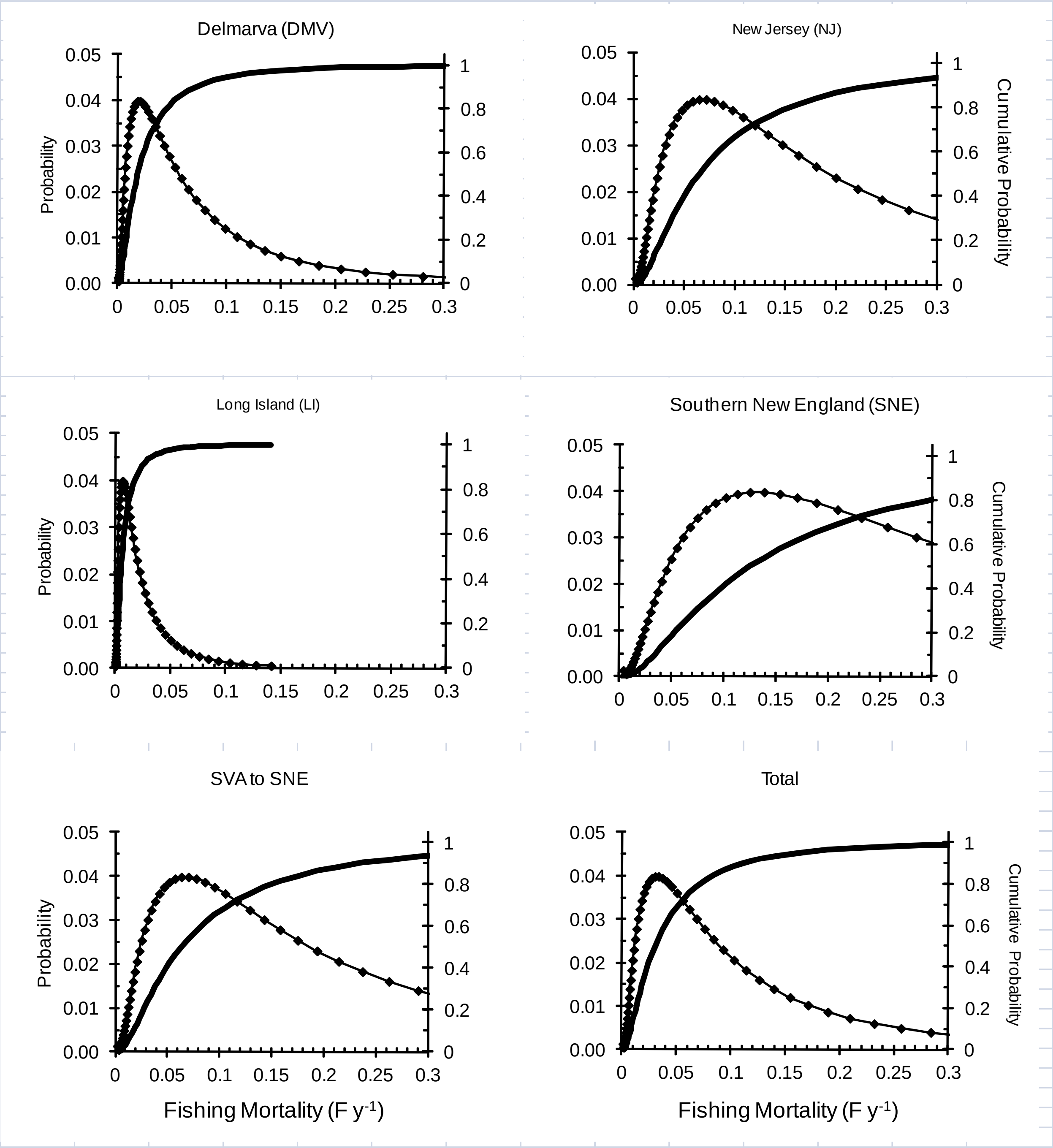
	SARC-26		SARC-30		SARC-37		SARC-44		SARC-49		New assessment	
Sizes	All		All		110+ and 120+		120+ mm		120+ mm		120+ mm	
Year	Biomass (1000 mt)	Survey efficiency (e)	Biomass (1000 mt)	Survey efficiency (e)	Biomass (1000 mt)	Survey efficiency (e)	Biomass (1000 mt)	Survey efficiency (e)	Biomass (1000 mt)	Survey efficiency (e)	Biomass (1000 mt)	Survey efficiency (e)
1997	1,130	0.897	1,106	0.588	1,146	0.460	1,913	0.226	1,276	0.372	2,130	0.234
1999			1,596	0.276	1,460	0.276	1,503	0.226	1,005	0.372	1,667	0.234
2002					803	0.389	1,479	0.226	1,082	0.372	1,655	0.234
2005							1,066	0.226	954	0.256	1,570	0.234
2008									1,038	0.372	1,755	0.256
2011											1,307	0.234

	SARC-26		SARC-30		SARC-37		SARC-44		SARC-49		New assessment	
Sizes	All		All		110+ and 120+		120+ mm		120+ mm		120+ mm	
Year	Fishing mortality	Survey efficiency (e)	Fishing mortality	Survey efficiency (e)	Fishing mortality	Survey efficiency (e)	Fishing mortality	Survey efficiency (e)	Fishing mortality	Survey efficiency (e)	Fishing mortality	Survey efficiency (e)
1997	0.0181	0.897	0.0188	0.588	0.0180	0.460	0.0109	0.226	0.0163	0.372	0.0098	0.234
1999			0.0137	0.276	0.0150	0.276	0.0146	0.226	0.0218	0.372	0.0131	0.234
2002					0.0330	0.389	0.0182	0.226	0.0248	0.372	0.0162	0.234
2005							0.0222	0.226	0.0248	0.372	0.0151	0.234
2008									0.0243	0.372	0.0143	0.234
2011											0.0193	0.234

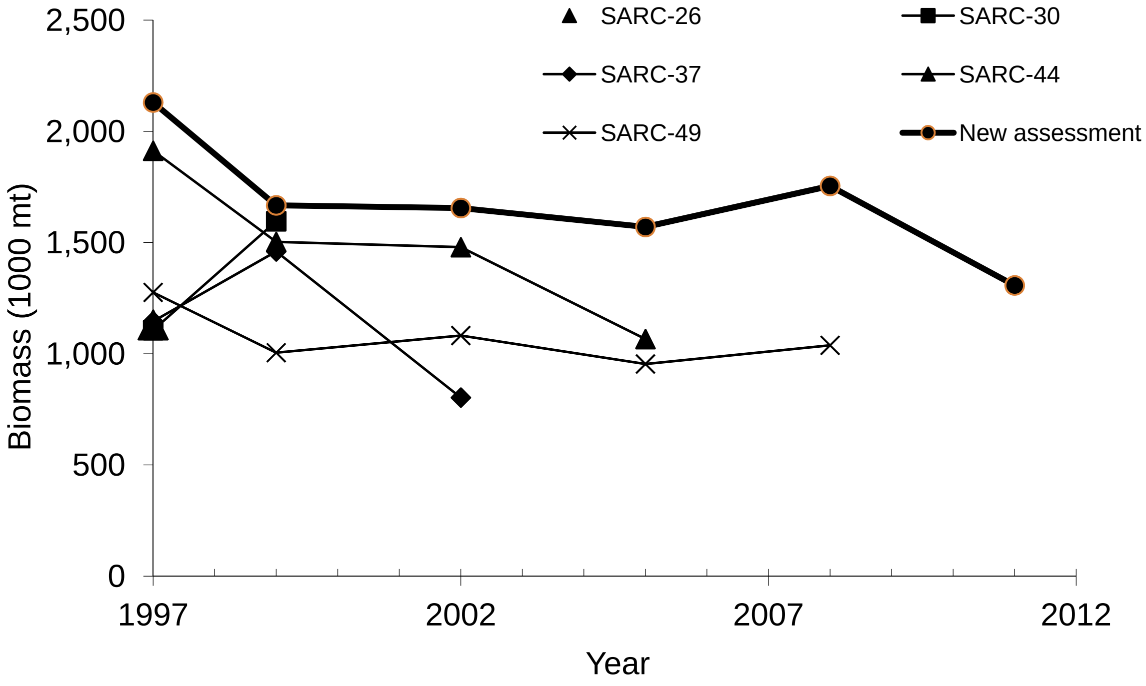
Appendix A8. Figure 1. Uncertainty in efficiency corrected swept area biomass estimates for surfclams in 2011. Note that the x-axis differs in the panel for SVA and GBK but is the same in other panels to facilitate comparisons.

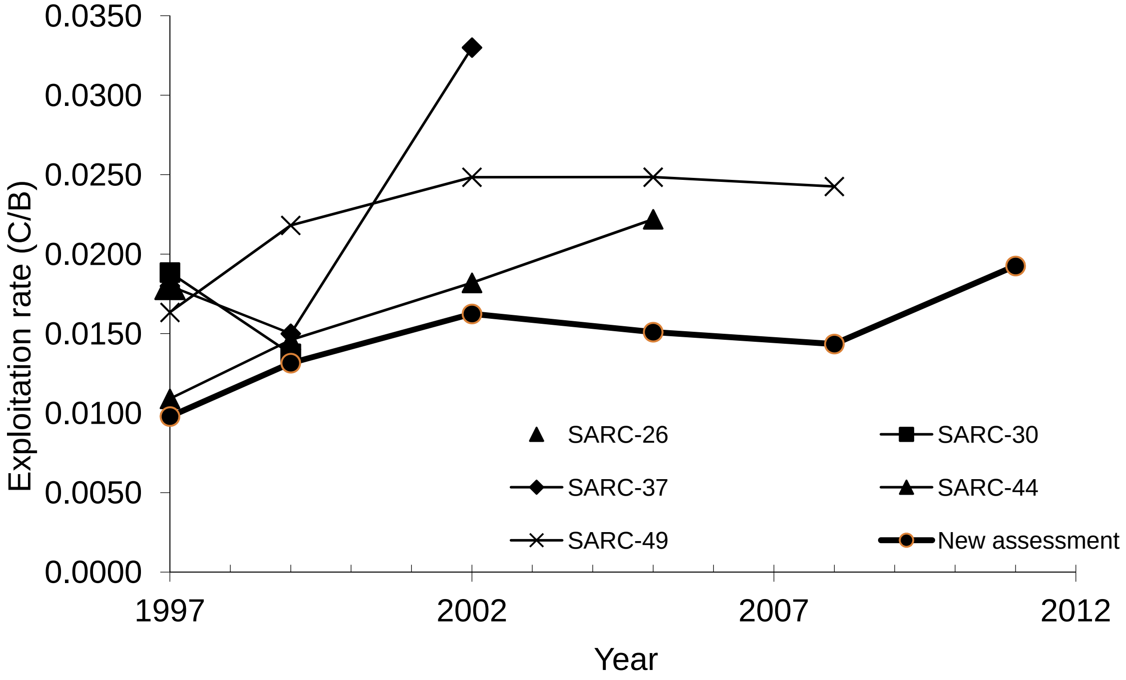


Appendix A8. Figure 2. Uncertainty in fishing mortality estimates for surfclams during 2011 based on catch data and efficiency corrected swept-area biomass. X-axes are scaled to the same maximum to facilitate comparisons.

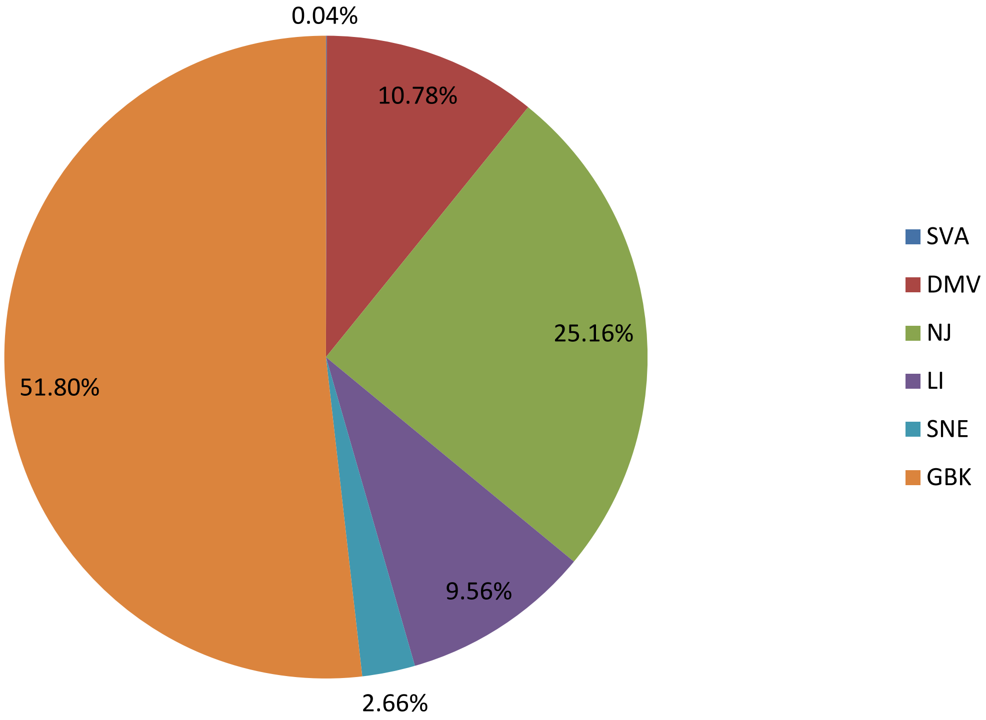


Appendix A8. Figure 3. Historical retrospective analysis of efficiency corrected swept area biomass and exploitation rate (catch / biomass).





Appendix A8. Figure 4. Percentage of total swept area biomass by region in 2011.



Appendix A9. Additional Sensitivity Testing and Decision Table Analyses

Uncertainty in estimating the scale of biomass has been a challenge in surfclam assessments for many years. We carried out additional sensitivity analyses to determine the likely effects of potential management actions (catch levels) if the biomass scale estimated in the basecase model is substantially too high or too low. The biomass reference points used in this assessment mitigate the scale problem to some degree because the calculation used to determine biomass status B2011/(B1999/4) is robust and does not change appreciably if the overall scale estimated by the assessment model changes, as long as trend can be estimated with relative accuracy and precision. In contrast, the calculation used to determine fishing mortality status F=M=0.15 is not robust to scale because it changes in proportion to the overall scale estimated by the assessment model.

In this appendix we estimate the probability of overfishing/overfished status for the entire stock and for the southern component by comparing projections against a wide range of possible biomass scales and catch levels (see TOR 4 and TOR 7 in the main document for the methods used in calculating overfished/overfishing status).

If the true catchability *q* for the NEFSC clam survey is higher than estimated in the basecase assessment, then the true biomass will be lower than estimated and *vice-versa*. The *q* estimated in the basecase model was 0.33, which was approximately equal to the 64th percentile of our prior distribution. It is possible that we misestimated *q*. With this in mind, one our sensitivity tests assumes that the true *q* is equal to the 75th percentile of our prior distribution so that true biomass levels are substantially lower than estimated in the basecase model. Other sensitivity analyses assume that the true *q* is equal to the 25th percentile of our prior distribution so that the true biomass level is much higher than estimated in the basecase model. These values of *q* produce a wide range of biomass estimates (Table A9.1). The two sensitivity runs are hereafter referred to as “high *q*” and “low *q*” and will be compared to the actual assessment runs called “basecase”.

In projection scenarios we used the estimated *q* (0.33 = basecase) to calculate reference points. The population variables (biomass and F) estimated in the high *q* and low *q* model runs were compared to the basecase reference point to determine the status of the population. This scenario demonstrates the possible outcomes of a situation in which the assessment was incorrect regarding scale, and the true scale of the biomass is considerably higher or lower than we believe. We tested several catch levels in projection scenarios, described in the main body of the report. In order of increasing catch they are: status quo, quota and OFL (see TOR 7 and Table A9.2). These catch levels were prorated between the southern area where most fishing occurs and GBK as described in the main body of the report (TOR 7). Separate simulations were run for the southern area and GBK and the results each pair of simulations were combined to evaluate effects on the entire stock.

Because a high *q* results in a lower biomass, high *q* is more likely to result in an overfished/overfishing status determination. The scenario in which an overfished/overfishing designation was most likely to occur was when the population was fished at the OFL level, particularly when true biomass was lower than estimated using our basecase model (Figure A9.3). Under the high *q*-low biomass state of nature, the cumulative probability of overfished status during any of the years from 2013 – 2017 was unlikely (probability < 10%) using the status quo or quota catch levels, but was relatively likely (45%) when using the OFL catch scenario (Table A9.3). Fishing at the OFL level is not currently allowed under the surfclam FMP.

The probability of overfishing at any point during the years 2013-2017 was essentially zero (Figure A9.4) at any level of *q*, unless the catch was set at the OFL, when overfishing was almost inevitable in simulations.

In the low *q* scenario, the population was unlikely to be overfished or have overfishing occur at any point over the next five years (Table A9.3; Figure A9.5 – A9.8).

For the southern area only and high *q* state, the true biomass in 2011 tended to stay above the threshold (Figure A9.9). In the high *q* state, the annual fishing mortality trajectory fell below the F threshold, except in F=OFL scenario (Figure A9.10).

Reference points are defined for the whole stock but the maximum annual probability of a hypothetical overfished condition for the southern area using the hypothetical reference point $B_{threshold}=B_{1999}/4$ for the south in any year between 2013 and 2017 was generally less than 5% except in the F=OFL scenario, where it rose to about 17% (Figure A9.11). The cumulative probability of overfished status over that time period varies from 14% to 42% (Table A9.4; Figure A9.12). Overfished status was unlikely under all fishing scenarios when testing the low *q* state (Figures A9.13 and A9.15; Table A9.4).

The maximum annual probability of hypothetical overfishing the southern area over the years from 2013 to 2017 was zero regardless of the *q* used, unless fishing was set to the OFL (Figures A9.14 and A9.16; Table A9.4).

Overfished status determinations for the northern (GBK) area are not possible at this time due to a lack of reference points. The likely trajectory of the population biomass given the various states of *q* and fishing scenarios is available in Table (A9.2) and Figures (A9.17 – A9.18).

Overfishing the northern area is unlikely (cumulative probability through 2017 < 1**%**), except where fishing is set to the OFL (Figures A9.19 – A9.22; Table A9.5).

Potential effects on biomass were summarized using an additional method. We also present results based on the probability that the stock would fall below the "true" (based on the *q* being tested) value of $B_{1999}/4$ (Table A9.6). In this case the each state of nature (or q level) would have a unique reference point. In contrast, the method used in all other analyses summarizes results based on the probability that the stock falls below the $B_{1999}/4$ biomass level estimated in the basecase assessment, so that each q level is tested against the same reference point.

These sensitivities demonstrate that conclusions about the probability of overfishing or overfished stock status during 2011-2018 using the basecase model would likely not change under a wide range of true biomass levels and catches at the status-quo or quota levels. However, overfishing and overfished conditions are likely at the OFL which is currently not permitted in the FMP.

Table A9.1. Biomass in 2011 given the basecase and 2 sensitivity scenarios used as states of nature in decision table analysis, one in which the biomass was underestimated in the base case (low *q*) and one in which the biomass was overestimated (high *q*).

Region	*q*=0.11	*q*=0.33 Basecase	*q*=0.39
South	2,399,830	704,366	600,320
North	1,118,680	370,217	312,684
Total	3,518,510	1,074,583	913,004

Table A9.2. Biomass in projections given different sensitivity scenarios involving a range of true states of nature (biomass level) and possible management actions (catch levels).

	State of nature: *q* low (B high)								
	Status-quo			Quota			F=0.15		
Year	South	North	Total	South	North	Total	South	North	Total
2011	2,399,830	1,118,680	3,518,510	2,399,830	1,118,680	3,518,510	2,399,830	1,118,680	3,518,510
2012	2,379,060	1,027,710	3,406,770	2,379,060	1,027,710	3,406,770	2,379,060	1,027,710	3,406,770
2013	2,350,010	939,531	3,289,541	2,350,010	939,531	3,289,541	2,350,010	939,531	3,289,541
2014	2,294,130	840,714	3,134,844	2,288,940	840,714	3,129,654	2,247,970	822,088	3,070,058
2015	2,298,590	753,353	3,051,943	2,288,690	753,353	3,042,043	2,213,700	722,861	2,936,561
2016	2,382,780	683,152	3,065,932	2,368,600	683,152	3,051,752	2,264,670	645,876	2,910,546
2017	2,322,830	637,951	2,960,781	2,305,000	637,951	2,942,951	2,177,370	597,389	2,774,759
2018	2,400,280	668,168	3,068,448	2,379,180	668,168	3,047,348	2,230,390	626,192	2,856,582
2019	2,488,280	710,556	3,198,836	2,464,300	710,556	3,174,856	2,296,280	667,943	2,964,223
2020	2,574,860	756,680	3,331,540	2,548,360	756,680	3,305,040	2,362,280	713,381	3,075,661
2021	2,657,440	803,286	3,460,726	2,628,730	803,286	3,432,016	2,425,390	758,827	3,184,217

	State of nature: *q* high (B low)								
	Status-quo			Quota			F=0.15		
Year	South	North	Total	South	North	Total	South	North	Total
2011	600,320	312,684	913,004	600,320	312,684	913,004	600,320	312,684	913,004
2012	595,561	285,915	881,476	595,561	285,915	881,476	595,561	285,915	881,476
2013	587,428	260,080	847,508	587,428	260,080	847,508	587,428	260,080	847,508
2014	576,571	227,784	804,355	571,561	227,784	799,345	532,181	209,198	741,379
2015	584,775	199,284	784,059	575,246	199,284	774,530	503,376	168,882	672,258
2016	626,825	176,141	802,966	613,143	176,141	789,284	513,398	139,021	652,419
2017	625,105	160,555	785,660	607,876	160,555	768,431	485,513	120,271	605,784
2018	659,520	166,515	826,035	639,107	166,515	805,622	496,442	124,930	621,372
2019	697,259	176,256	873,515	674,032	176,256	850,288	512,770	134,134	646,904
2020	733,435	187,321	920,756	707,722	187,321	895,043	528,862	144,568	673,430
2021	767,295	198,728	966,023	739,385	198,728	938,113	543,581	154,801	698,382

Table A9.3. Decision table for the whole surfclam stock, showing cumulative probability of overfished/overfishing status in any of the 5 years during 2013-2017, using 3 three different catch scenarios and assuming three states of nature (high, basecase and low biomass levels)

Whole stock overfished status probability

Catch	Low *q* (high B)	Basecase	High *q* (low B)
Status quo	0.001	0.019	0.082
Quota	0.001	0.022	0.098
OFL	0.002	0.122	0.448

Whole stock overfishing probability

Catch	Low *q* (high B)	Basecase	High *q* (low B)
Status quo	0	0	0
Quota	0	0	0.001
OFL	0	0.99	1

Table A9.4. Decision table for the southern area, showing cumulative probability of overfished/overfishing status in any of the 5 years from 2013-2017, using 3 three different catch scenarios and assuming three states of nature (high, basecase and low biomass levels).

Southern area overfished status probability

Catch	Low *q* (high B)	Basecase	High *q* (low B)
Status quo	0	0.053	0.136
Quota	0	0.061	0.156
OFL	0	0.163	0.42

Southern area overfishing probability

Catch	Low *q* (high B)	Basecase	High *q* (low B)
Status quo	0	0	0
Quota	0	0	0
OFL	0	0.99	1

Table A9.5. Decision table for the northern area, showing cumulative probability of overfished/overfishing status in any of the 5 years from 2013-2017, using 3 three different catch scenarios and assuming three states of nature (high, basecase and low biomass levels).

Northern area overfishing probability

Catch	Low *q* (high B)	Basecase	High *q* (low B)
Status quo	0	0	0.002
Quota	0	0	0.003
OFL	0	0.99	1

Table A9.6. Decision table for the whole stock and southern area, showing cumulative probability of overfished/overfishing status in any of the 5 years from 2013-2017, using 3 three different catch scenarios, and assuming three states of nature (high, basecase and low biomass levels). In this case the biomass reference point is derived from each assessment outcome (i.e. in the low q outcome, the reference point $B_{1999}/4$ is based on the low q biomass in 1999).

Whole stock overfished status probability

Catch	Low q (high B)	Basecase	High q (low B)
Status quo	0.001	0.019	0.004
Quota	0.001	0.022	0.006
OFL	0.002	0.122	0.118

Southern area overfished status probability

Catch	Low q (high B)	Basecase	High q (low B)
Status quo	0.003	0.053	0.027
Quota	0.004	0.061	0.032
OFL	0.006	0.163	0.139

Figure A9.1 Biomass results for projections with the high *q* (low biomass) scenario in which true whole stock biomass was substantially lower than estimated in the basecase model. The biomass reference point is from the basecase model.

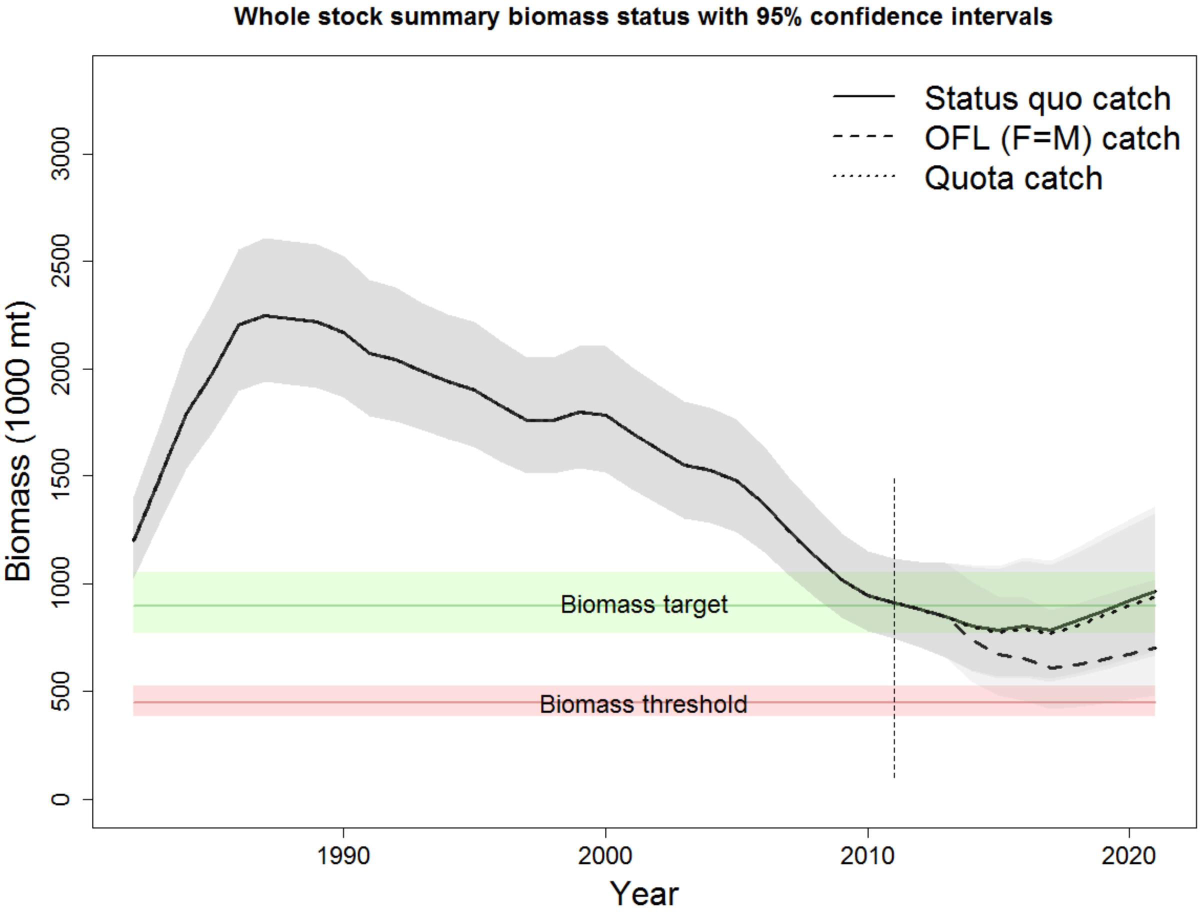


Figure A9.2. Fishing mortality results for projections with the high *q* (low biomass) scenario in which true whole stock biomass was substantially lower than estimated in the basecase model.

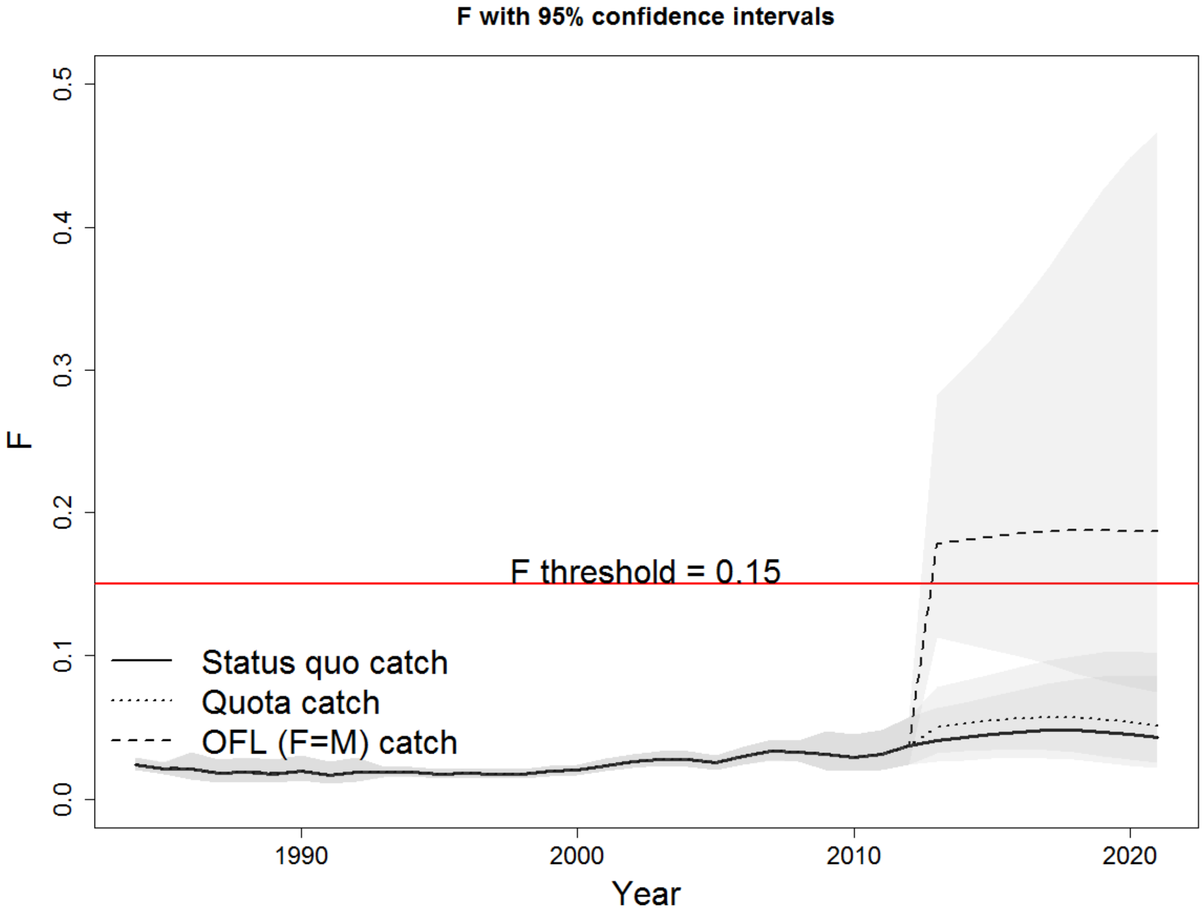


Figure A9.3. Biomass results for projections with the high *q* (low biomass) scenario in which whole stock biomass was substantially lower than estimated in the basecase model. Probabilities are for overfished stock status occurring given the minimum biomass projected between 2013-2017. The biomass reference point is from the basecase model.

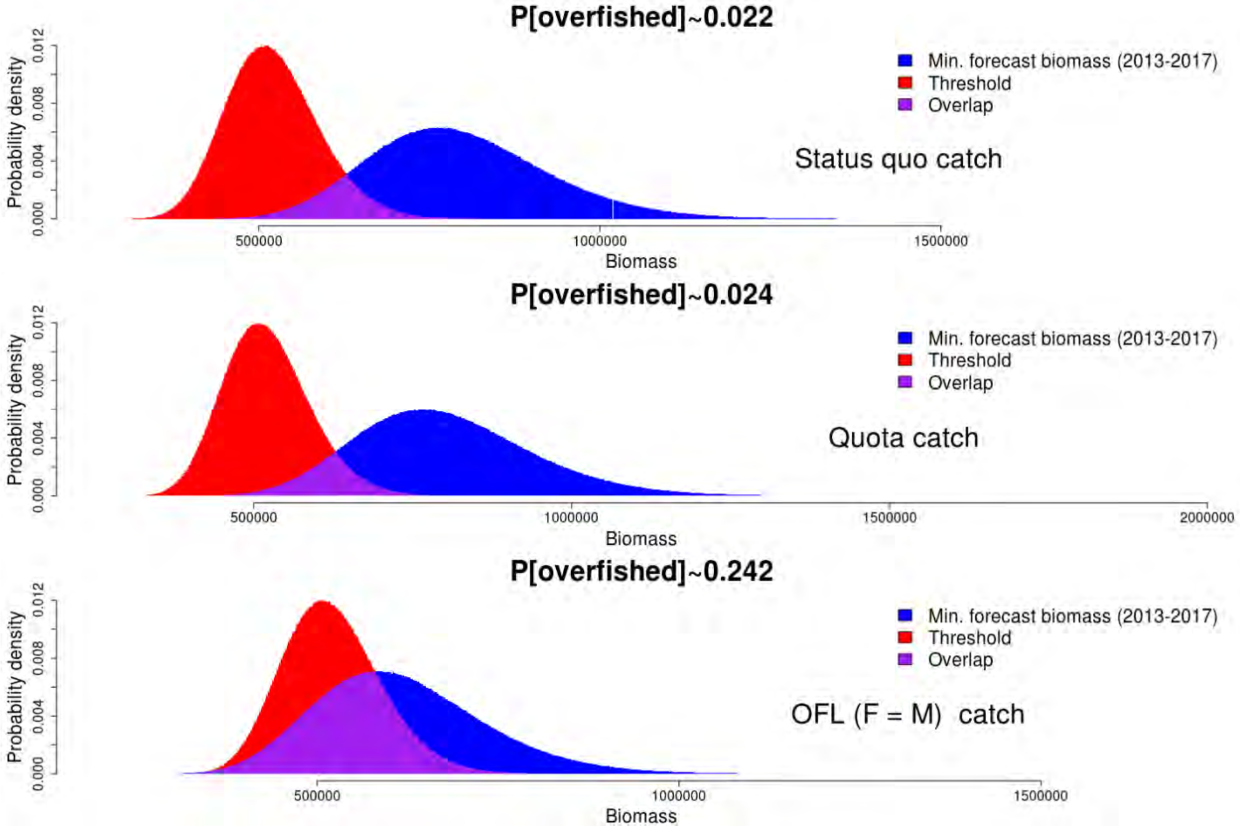


Figure A9.4. Fishing mortality results for projections with the high *q* (low biomass) scenario in which whole stock biomass was substantially lower than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.

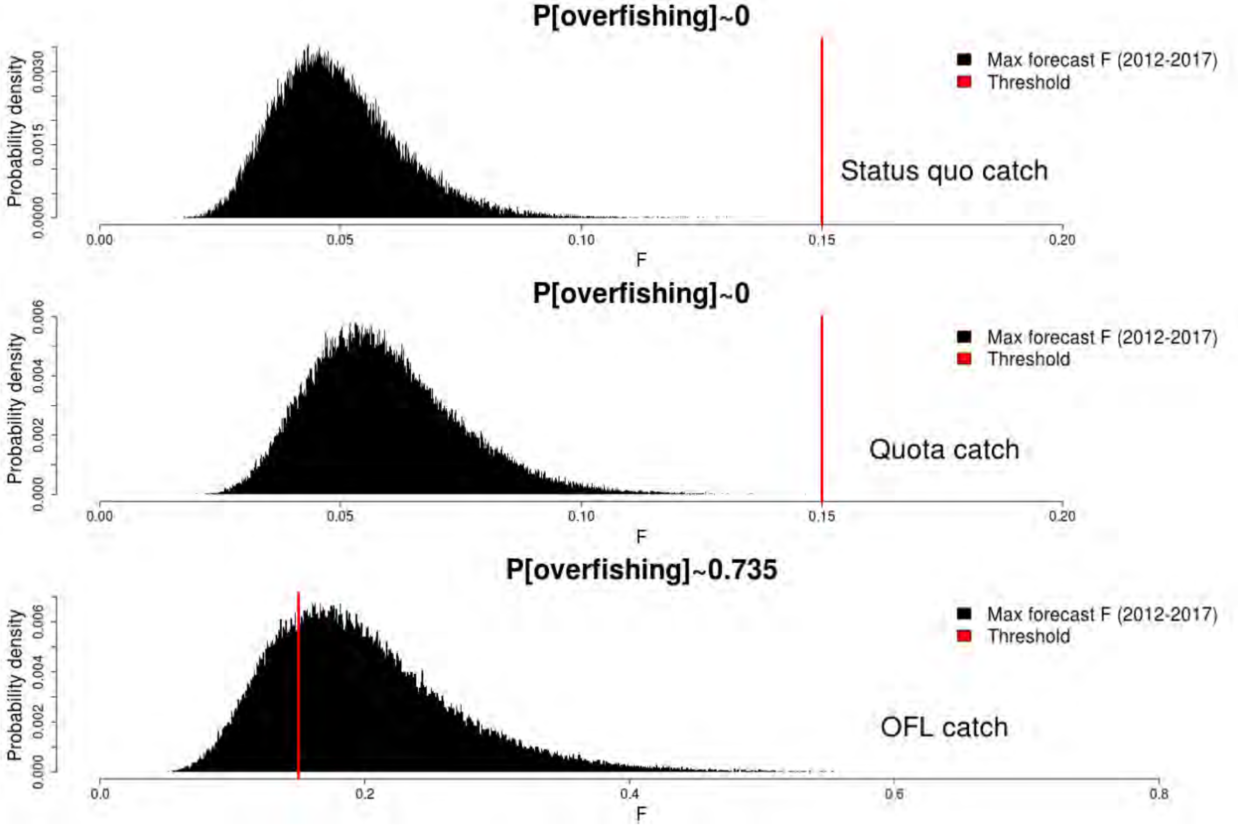


Figure A9.5. Biomass results for projections with the high *q* (low biomass) scenario in which true whole stock biomass was substantially larger than estimated in the basecase model. The biomass reference point is from the basecase model.

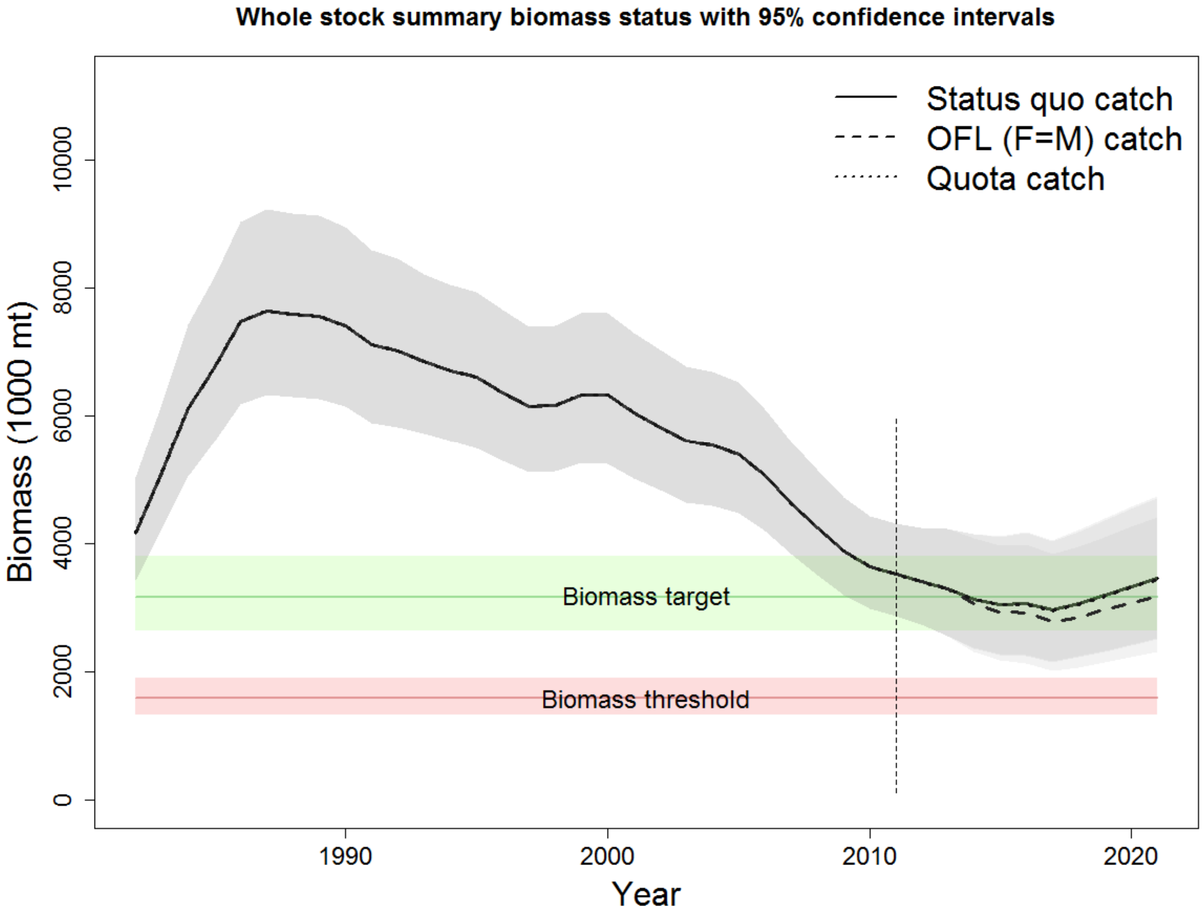


Figure A9.6. Fishing mortality results for projections with the low *q* (high biomass) scenario in which true whole stock biomass was substantially larger than estimated in the basecase model.

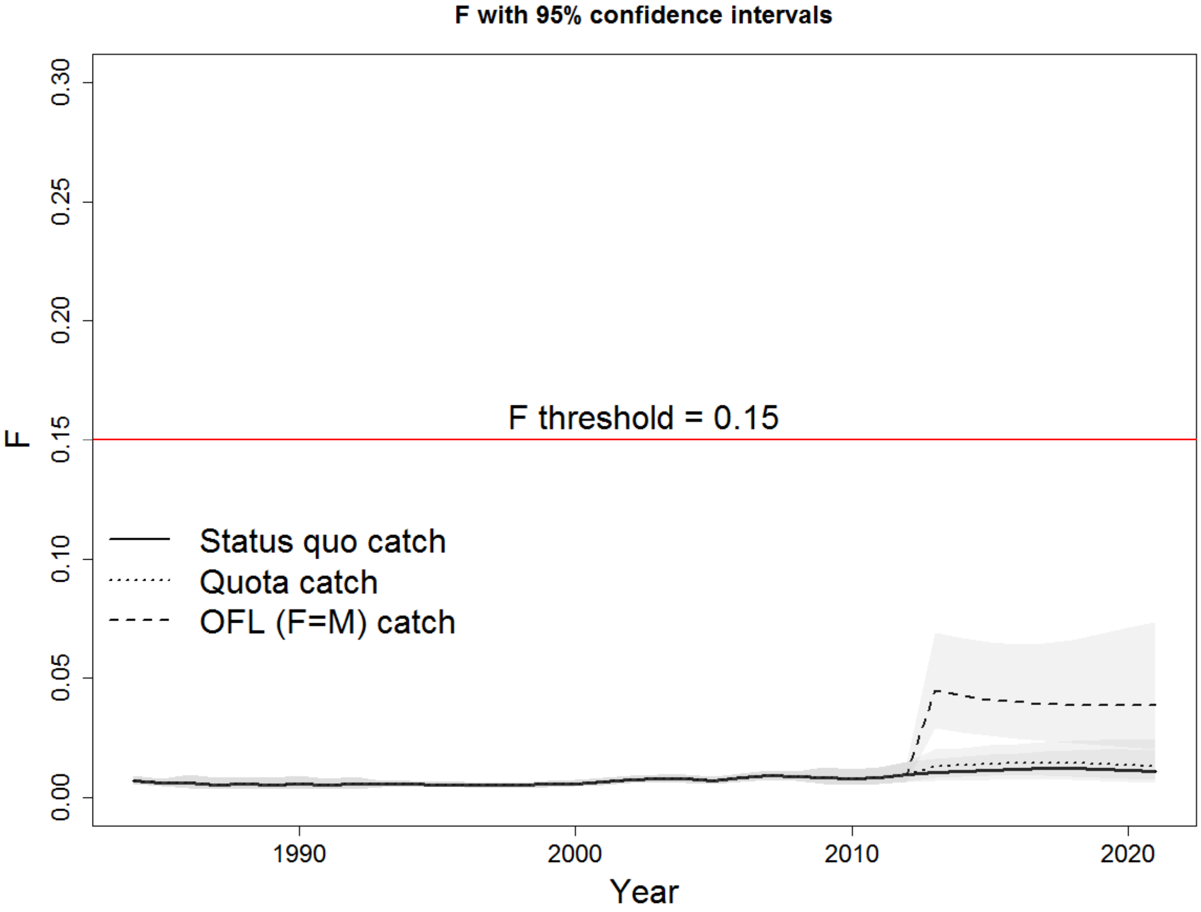


Figure A9.7. Biomass results for projections with the low *q* (high biomass) scenario in which whole stock biomass was substantially larger than estimated in the basecase model. Probabilities are for overfished stock status occurring given the minimum biomass projected between 2013-2017. The biomass reference point is from the basecase model.

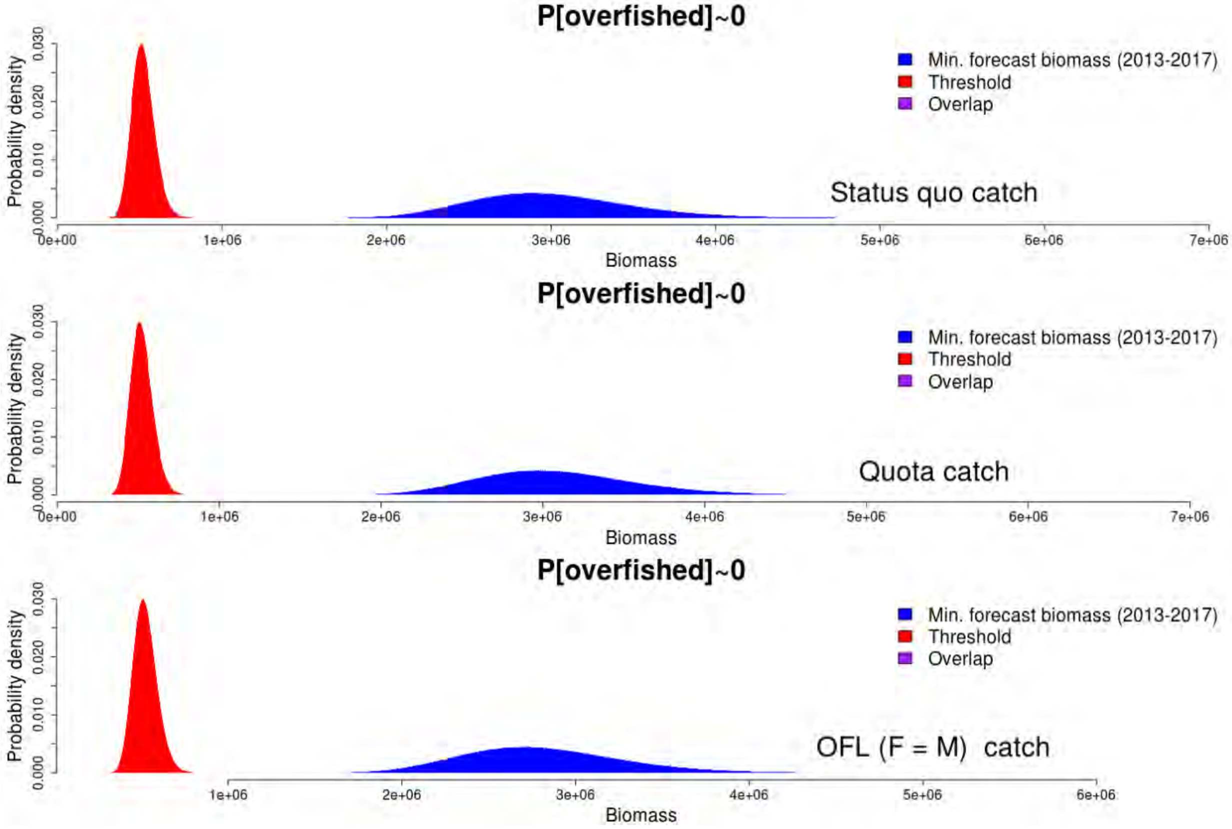


Figure A9.8. Fishing mortality results for projections with the low q (high biomass) scenario in which whole stock biomass was substantially larger than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.

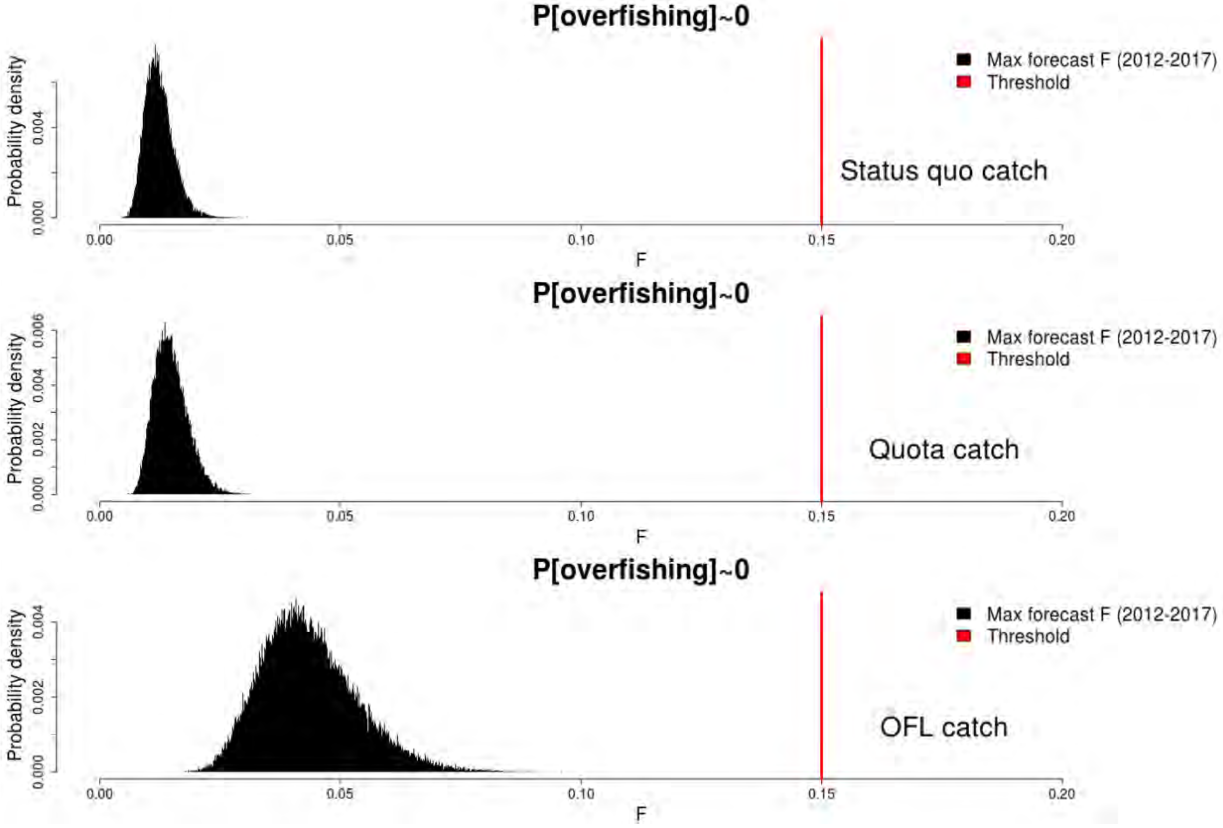


Figure A9.9. Biomass results for projections with the high *q* (low biomass) scenario in which true southern area biomass was substantially lower than estimated in the basecase model. The biomass reference point is from the basecase model.

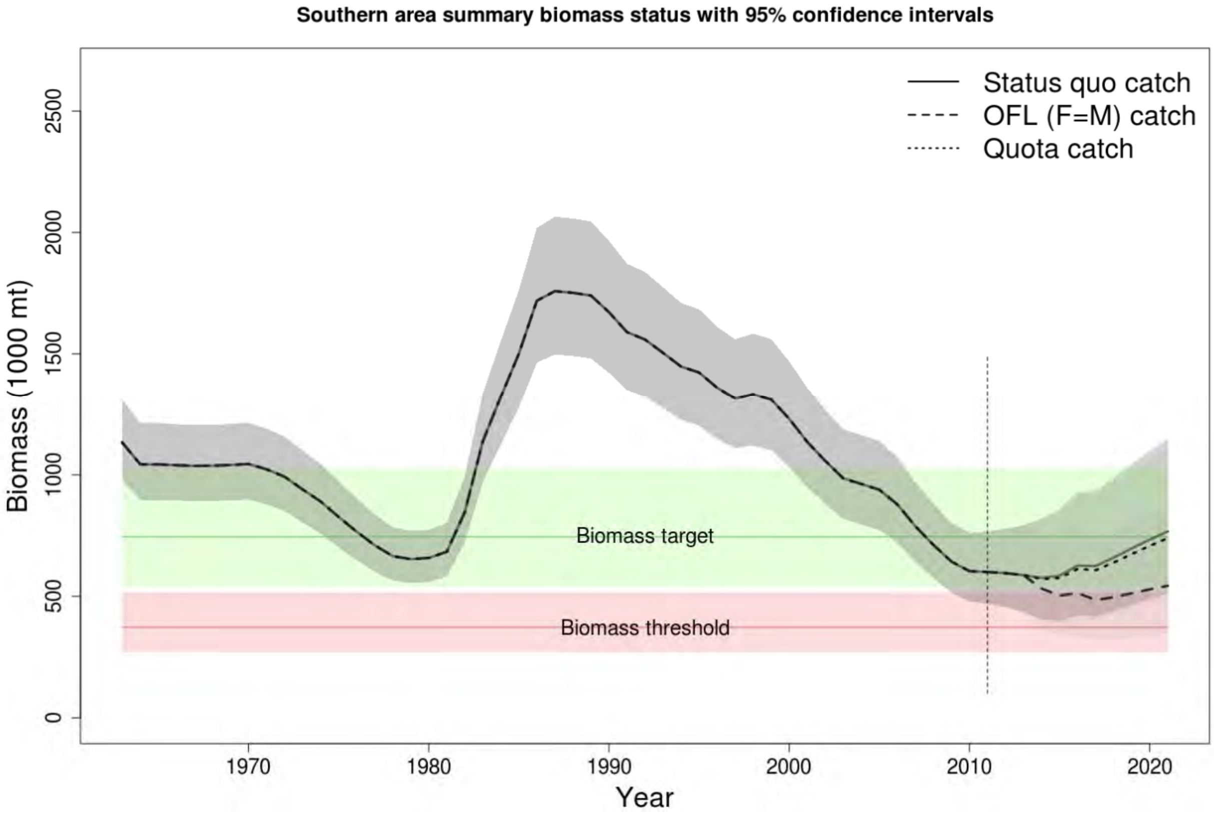


Figure A9.10. Fishing mortality results for projections with the high *q* (low biomass) scenario in which true southern area biomass was substantially lower than estimated in the basecase model.

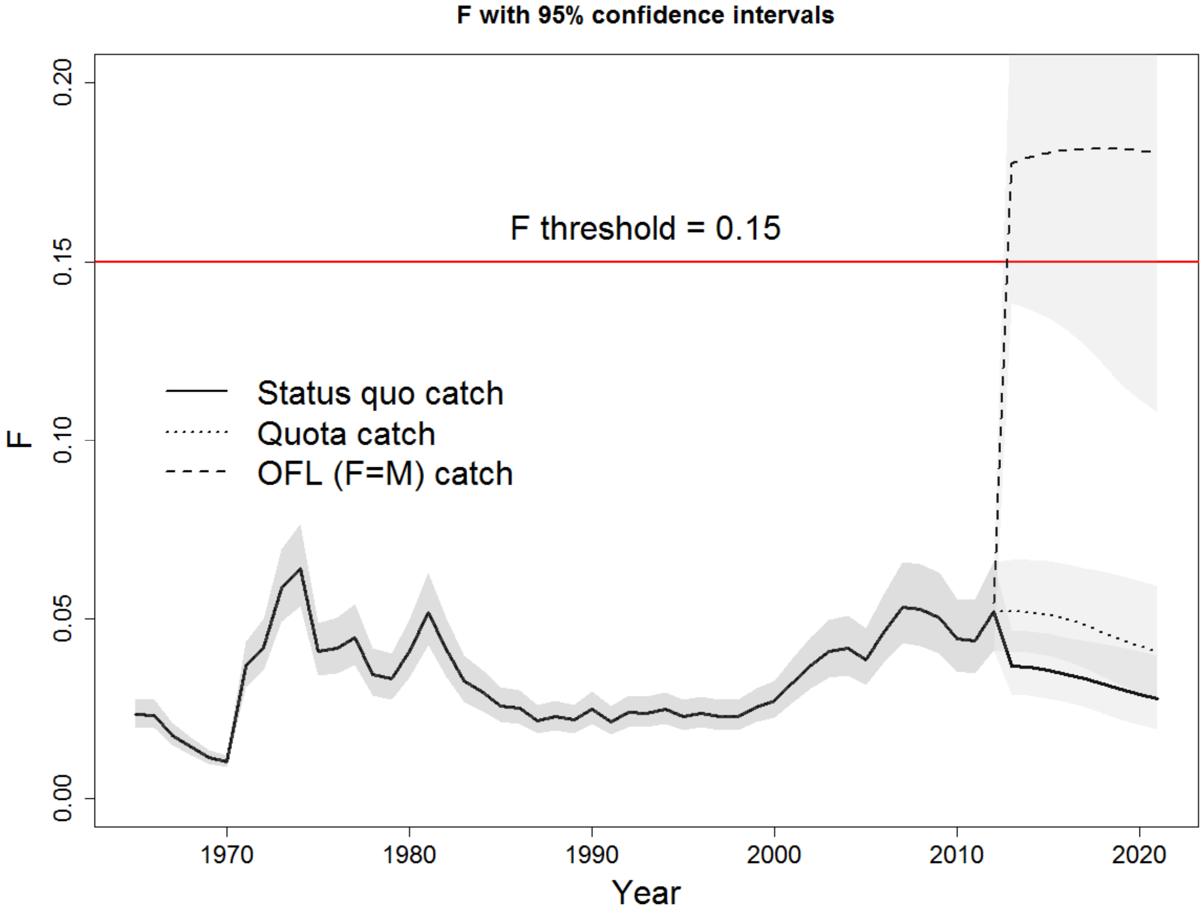


Figure A9.11. Biomass results for projections with the high *q* (low biomass) scenario in which southern area biomass was substantially lower than estimated in the basecase model. Probabilities are for overfished stock status occurring given the minimum biomass projected between 2013-2017. The biomass reference point is from the basecase model.

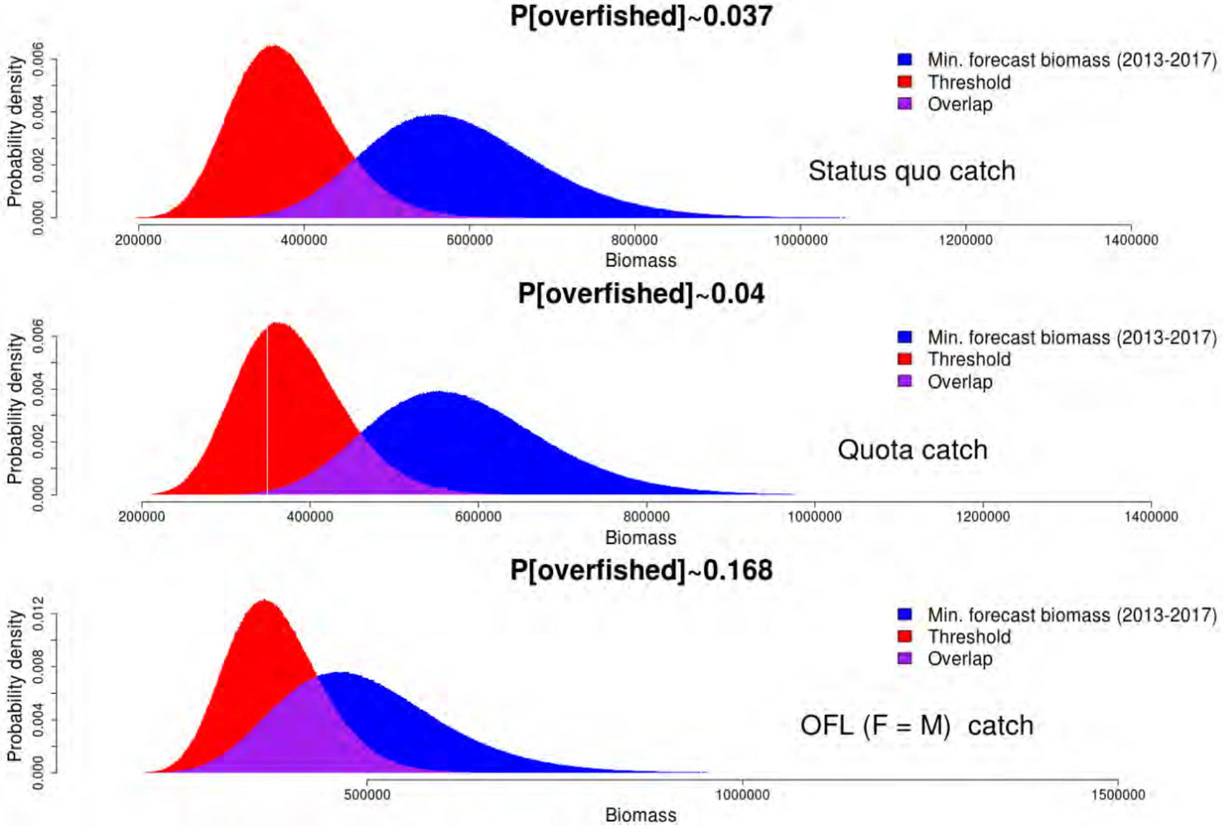
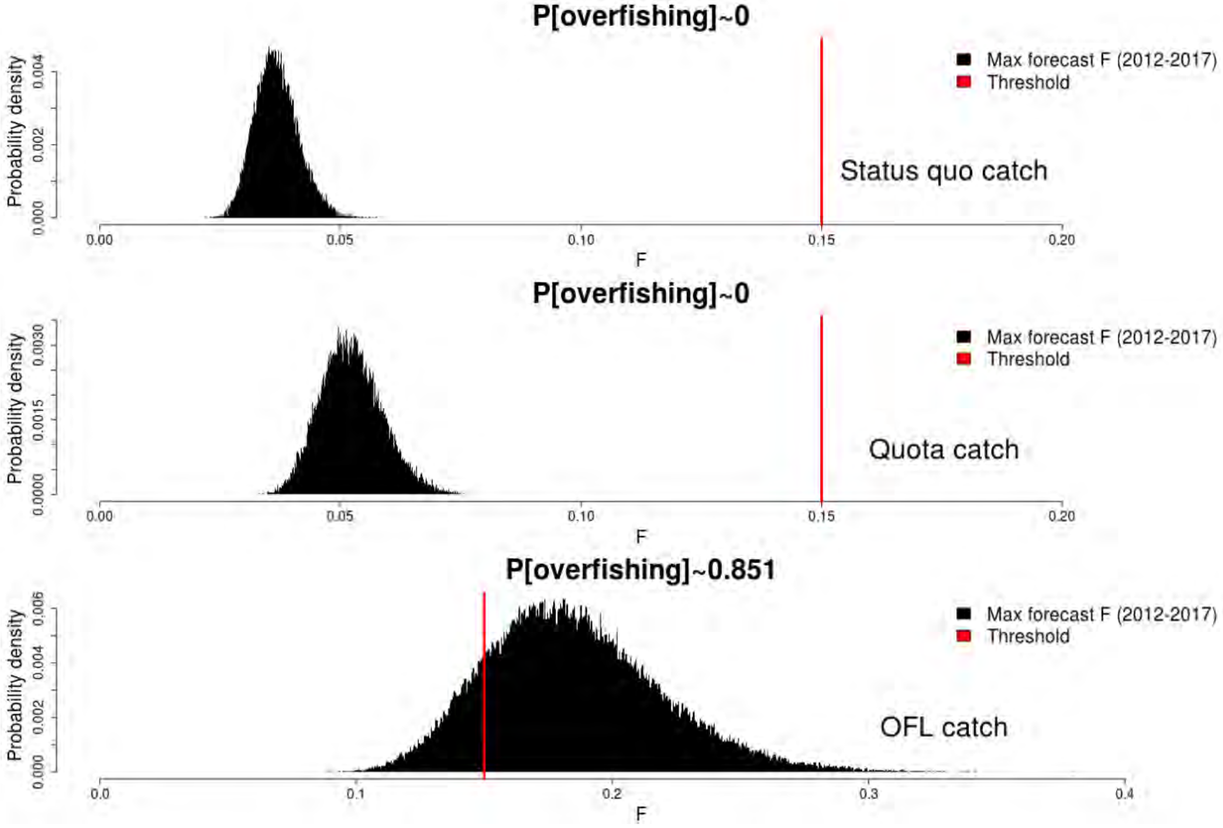


Figure A9.12. Fishing mortality results for projections with the high *q* (low biomass) scenario in which southern area biomass was substantially lower than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.



P[overfishing]~0

P[overfishing]~0

P[overfishing]~0.851

Figure A9.13. Biomass results for projections with the high *q* (low biomass) scenario in which true southern area biomass was substantially larger than estimated in the basecase model. The biomass reference point is from the basecase model.

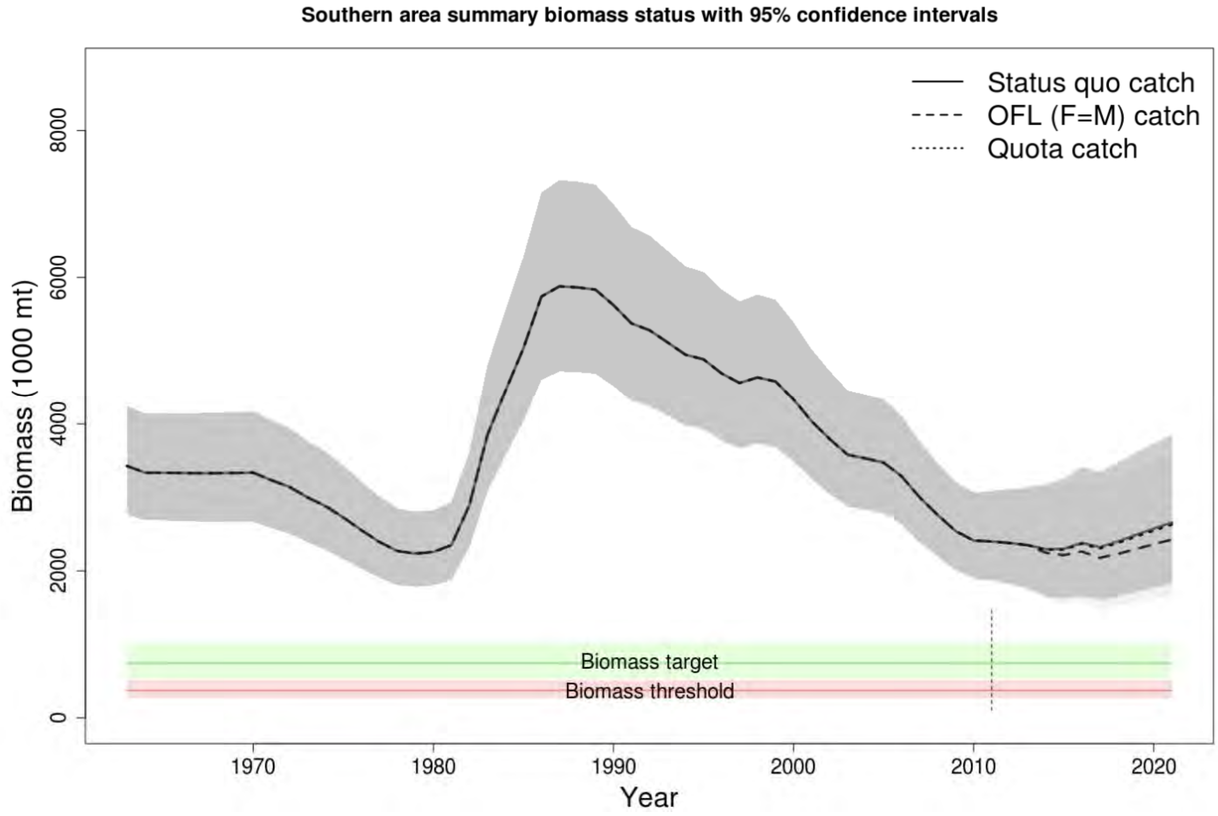


Figure A9.14. Fishing mortality results for projections with the low *q* (high biomass) scenario in which true southern area biomass was substantially larger than estimated in the basecase model.

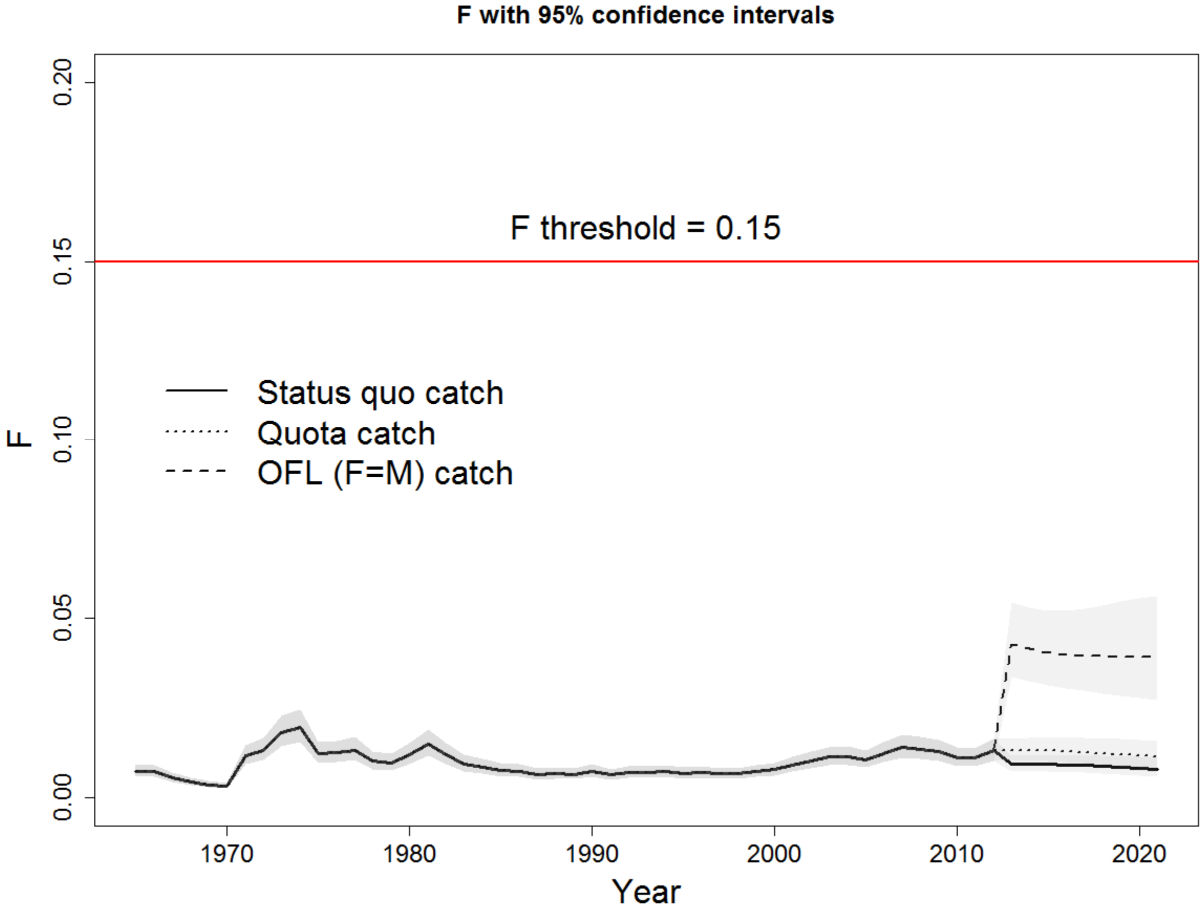


Figure A9.15. Biomass results for projections with the low *q* (high biomass) scenario in which southern area biomass was substantially larger than estimated in the basecase model. Probabilities are for overfished stock status occurring given the minimum biomass projected between 2013-2017. The biomass reference point is from the basecase model.

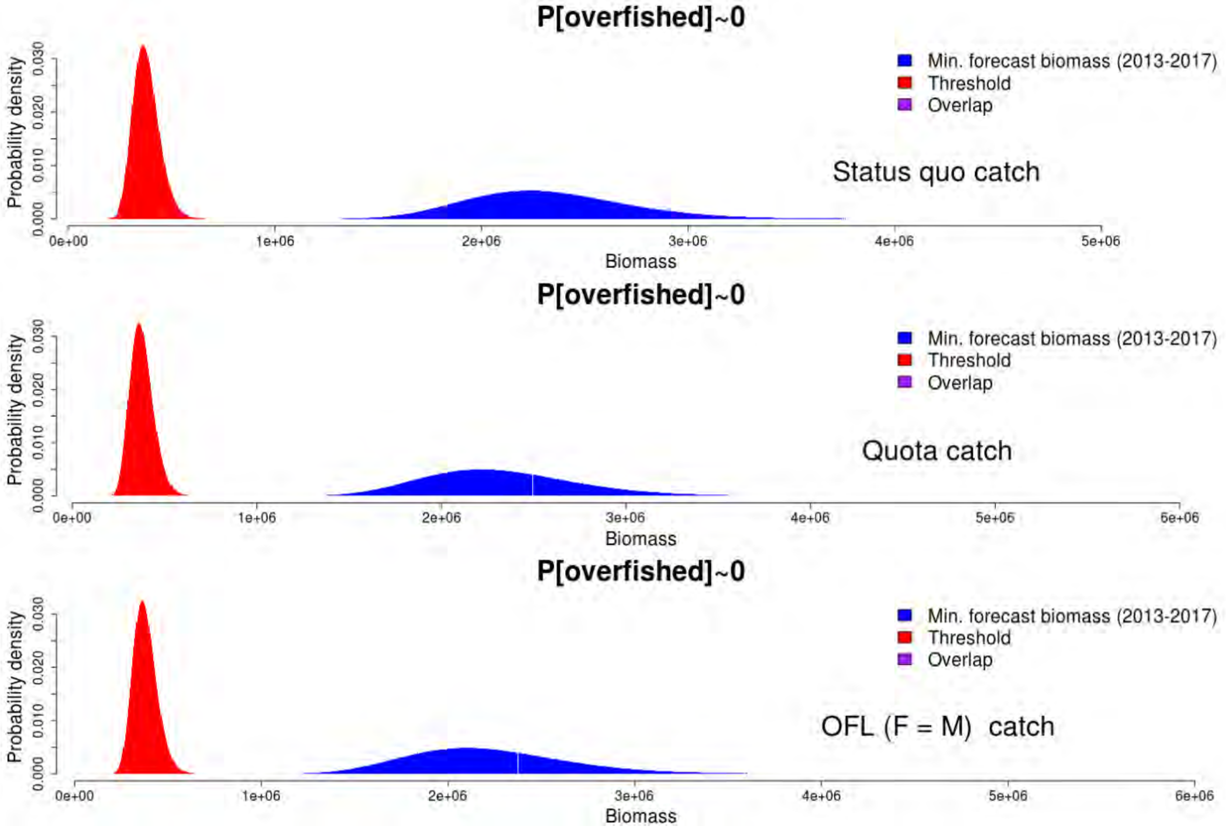


Figure A9.16. Fishing mortality results for projections with the low *q* (high biomass) scenario in which southern area biomass was substantially larger than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.

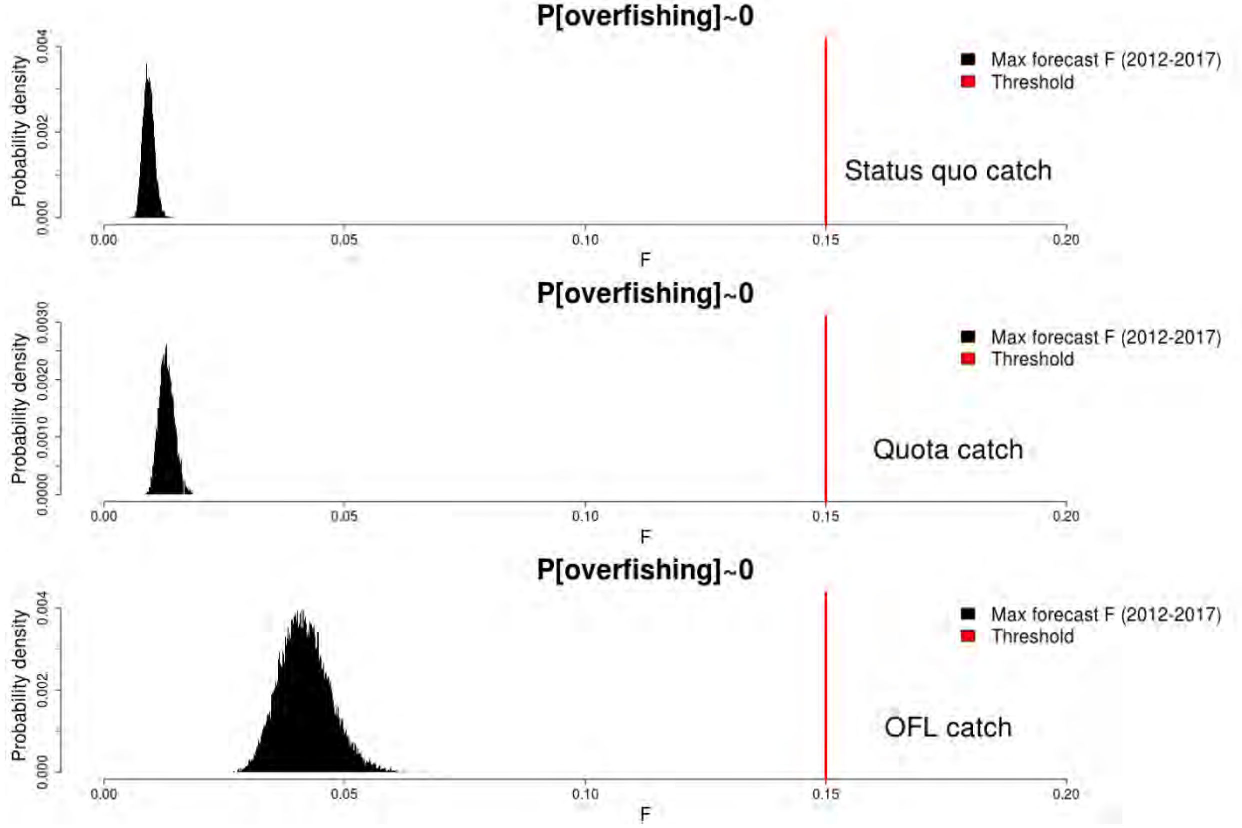


Figure A9.17. Biomass results for projections with the high *q* (low biomass) scenario in which true northern area biomass was substantially lower than estimated in the basecase model.

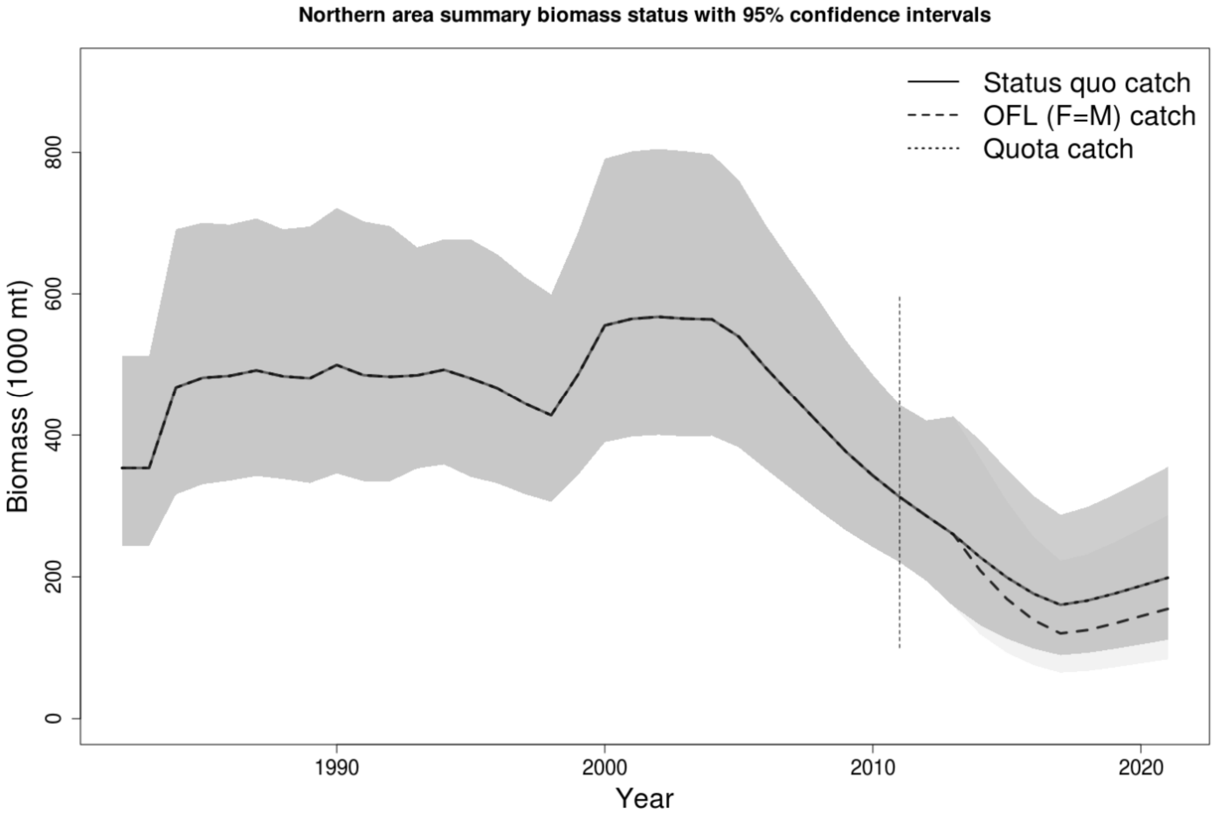


Figure A9.17 Biomass results for projections with the low *q* (high biomass) scenario in which true whole stock biomass was substantially lower than estimated in the basecase model.

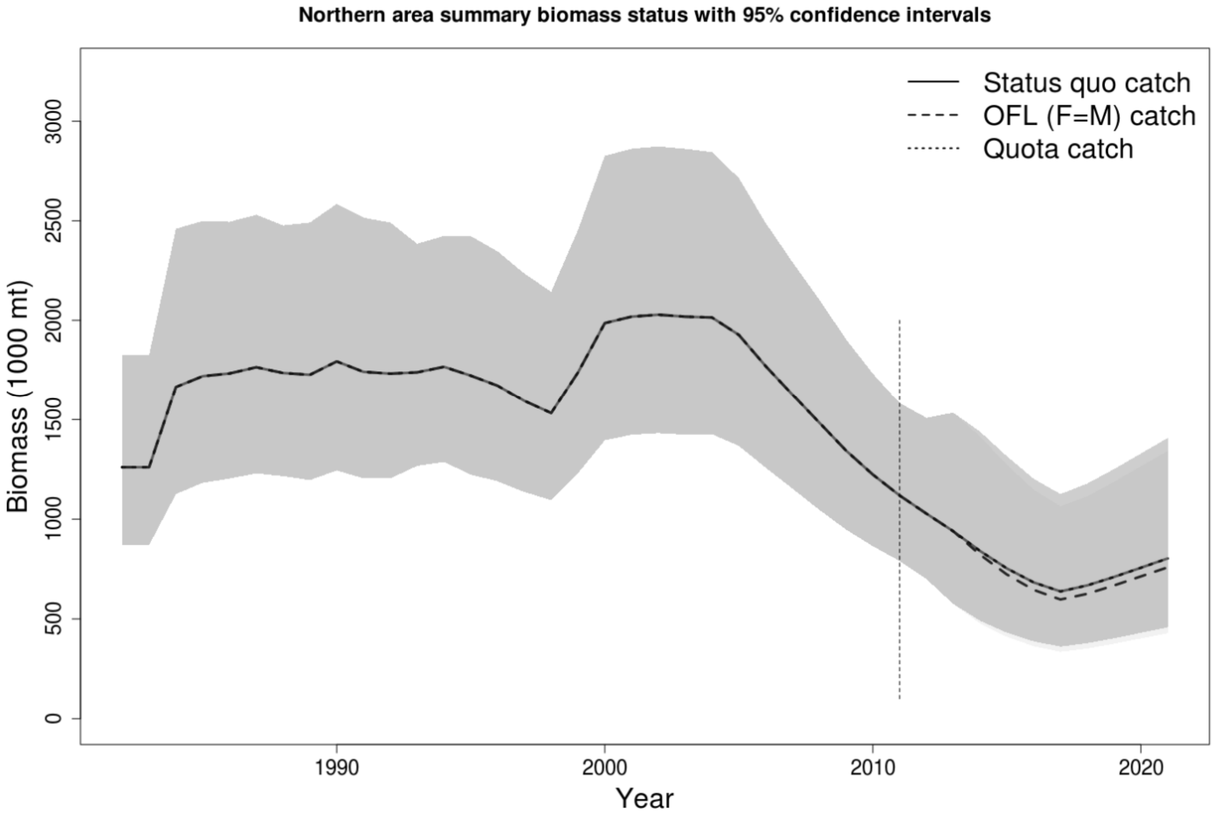


Figure A9.19. Fishing mortality results for projections with the high *q* (low biomass) scenario in which true northern area biomass was substantially lower than estimated in the basecase model.

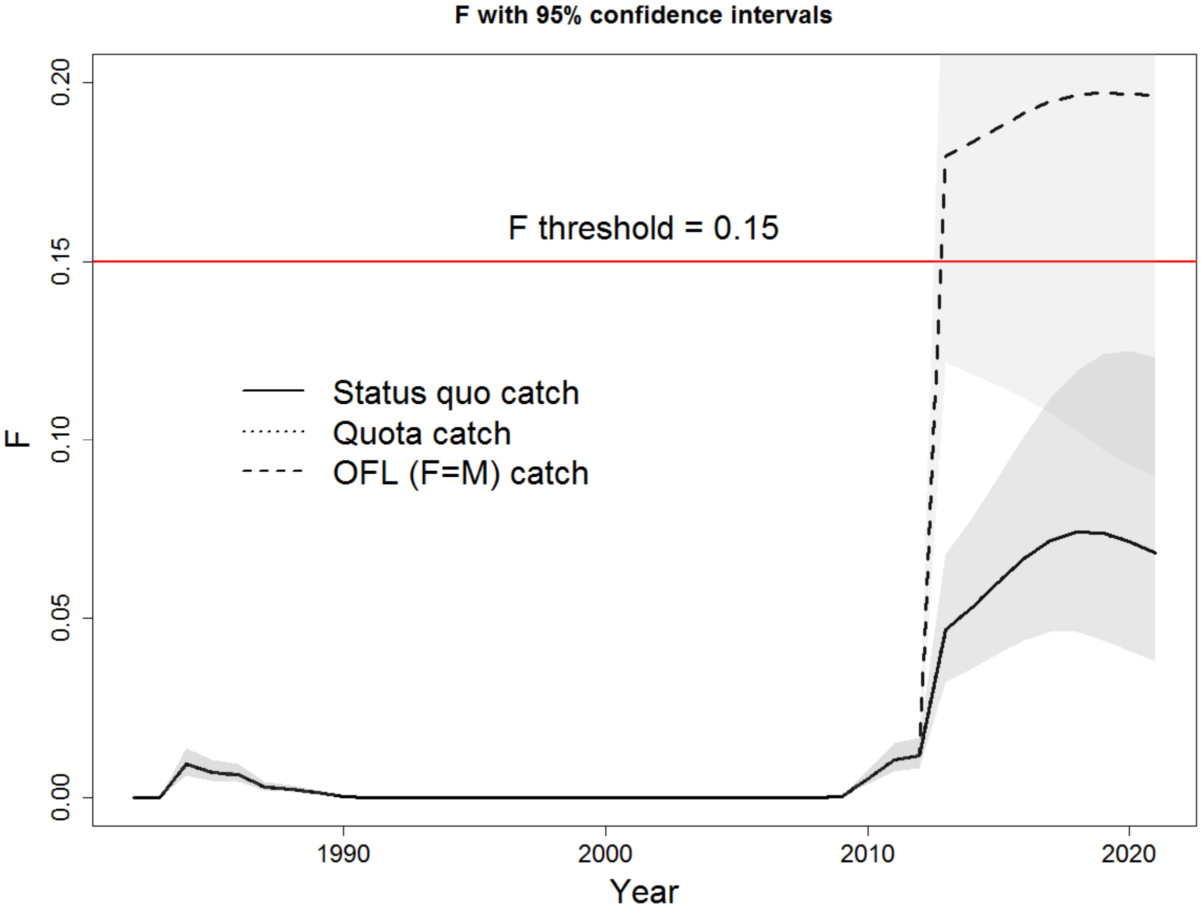


Figure A9.20. Fishing mortality results for projections with the high *q* (low biomass) scenario in which northern area biomass was substantially lower than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.

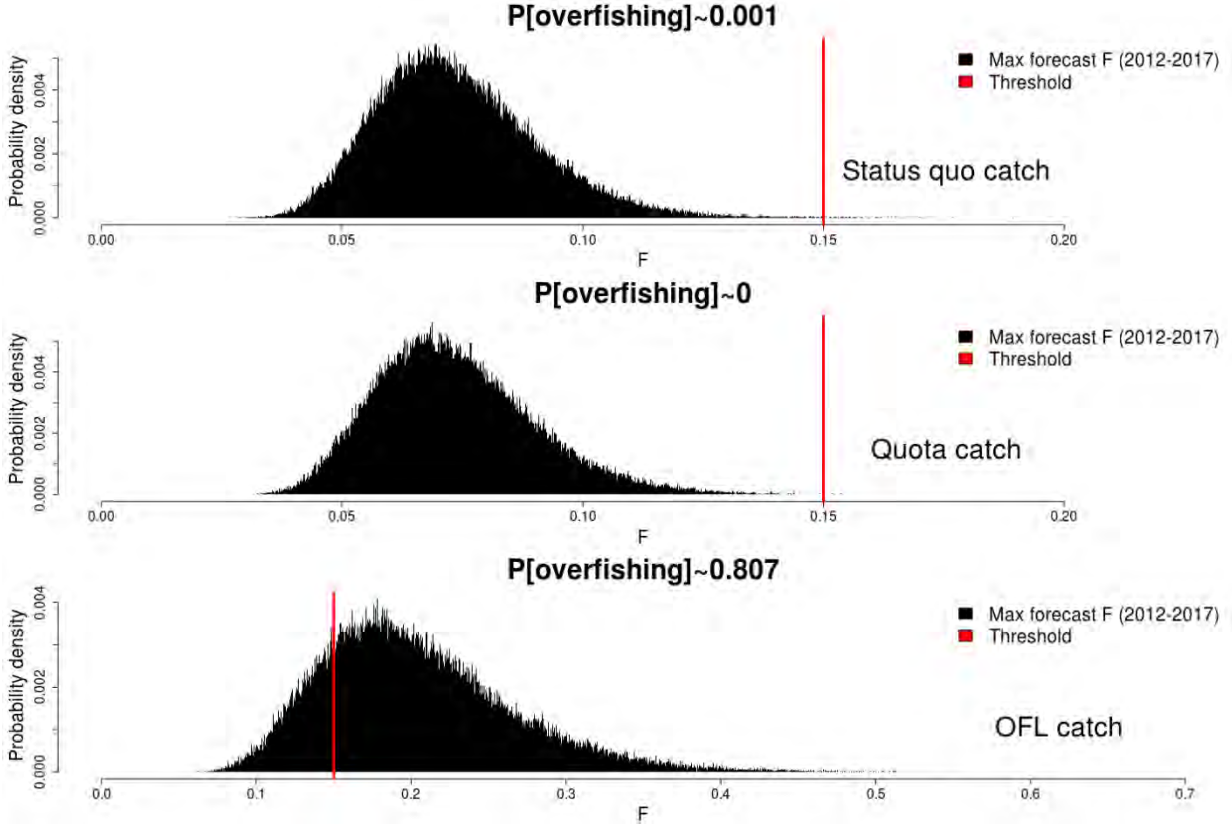


Figure A9.21. Fishing mortality results for projections with the low *q* (high biomass) scenario in which true northern area biomass was substantially larger than estimated in the basecase model.

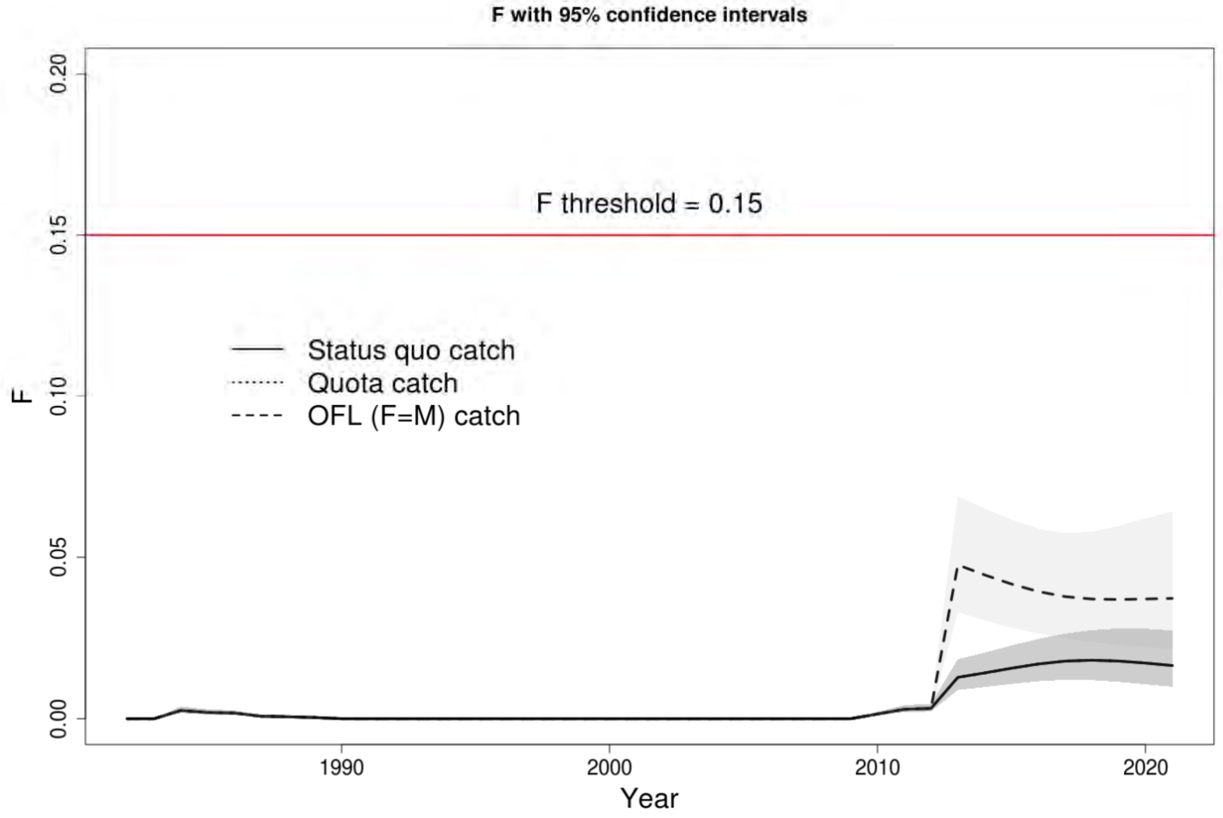
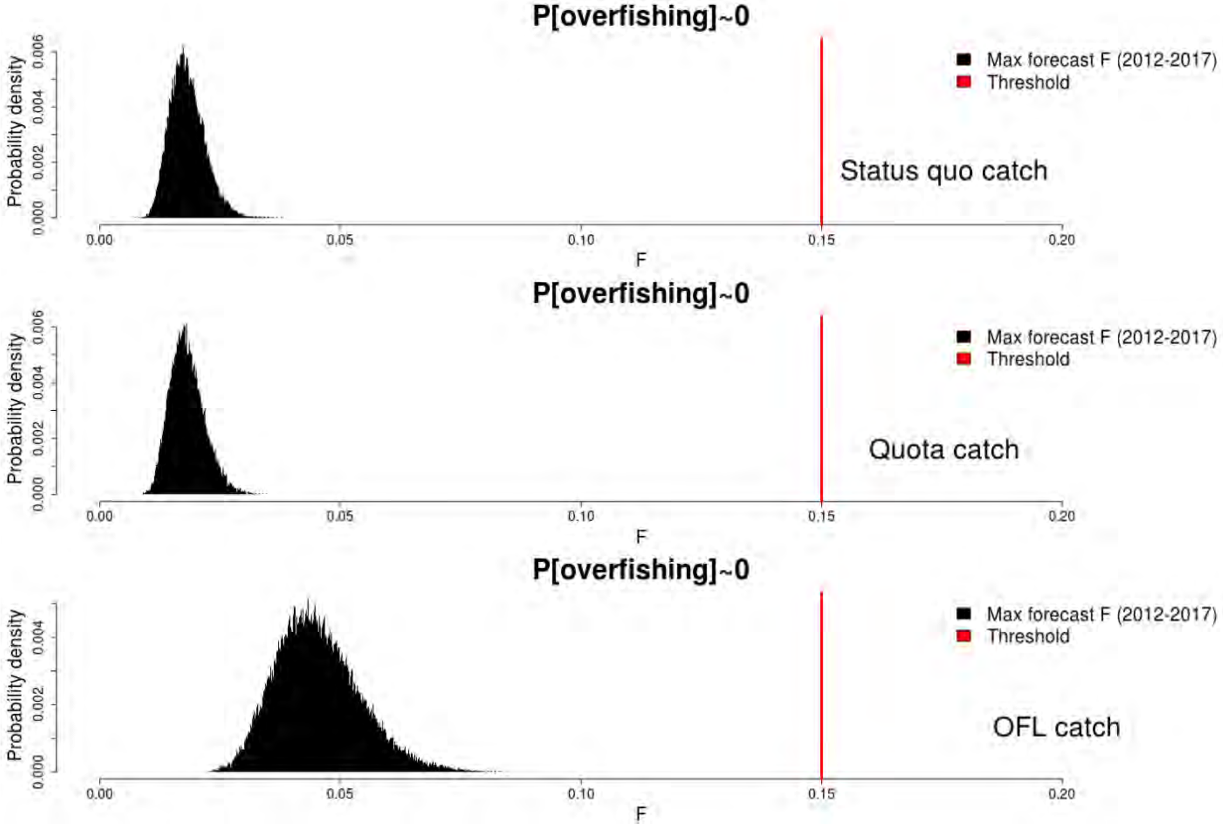


Figure A9.22. Fishing mortality results for projections with the low *q* (high biomass) scenario in which northern area biomass was substantially larger than estimated in the basecase model. Probabilities are for overfishing occurring given the minimum biomass projected between 2013-2017.



Appendix A10: Invertebrate Subcommittee

Persons who attended Invertebrate Subcommittee meetings and contributed to this report are:

Larry Jacobson (NEFSC, Chair)
Dan Hennen (NEFSC, assessment lead)
Toni Chute (NEFSC)
Chris Legault (NEFSC)
David Wallace (Wallace & Associates, Inc.)
Eric Powell (University of Southern Mississippi)
Daphne Munroe (Rutgers University)
Xinzhong Zhang (Rutgers University)
Fred Serchuk (NEFSC)
Jiashen Tang (NEFSC)
Jon Deroba (NEFSC)
Paul Rago (NEFSC)

Roger Mann (VIMS)
Tom Alspach (Sea Watch International, Inc.)
Tom Hoff (Wallace & Associates, Inc.)
Wendy Gabriel (NEFSC)
Jessica Coakly (MAFMC)
Jose Montanez (MAFMC)
Ed Houde (University of Maryland)
Doug Potts (NERO)
Guy Simmons (Sea Watch International, Inc.)
Bonnie McCray (Rutgers University)
Dvora Hart (NEFSC)
Carolyn Creed (Rutgers University)
Richard McBride (NEFSC)
Jeff Normant (NJ Division of Fish and Wildlife)
Jennifer O'Odwyer (NYSDEC)