1.2 Executive summary

This assessment is for Atlantic surfclam in the US EEZ (federal waters, 3-200 nm from shore) individual transferable quota (ITQ) fishery (Appendix 7). The assessment divides the US stock into a northern (Georges Bank or GBK) and a southern area (south of GBK to Cape Hatteras) for modeling purposes (Figures 6 and 7). However, the resource is managed as a single stock so estimates for the north and south are combined for status determination.

TOR 1. Estimate catch from all sources including landings and discards. Map the spatial and temporal distribution of landings, discards, fishing effort, and gross revenue, as appropriate. Characterize the uncertainty in these sources of data.

Commercial landings and fishing effort data are reported by processors based on cage tags, in logbooks by ten-minute square (TNMS) and considered reliable. Catch includes a 12% allowance for incidental mortality. Atlantic surfclam discards were near zero except during 1982-1993 when minimum size regulations were used (Table 3).

Landings, fishing effort and landings per unit effort (LPUE, bu per hour fished) shifted north after 2000 as fishery productivity in the south declined (Figures 13-18). During 2006-2015, total landings declined from about 27 to 18 (mean 21) thousand mt (Tables 4-5 and Figures 8-9). Fishing effort after 2006 varied without trend or declined in the south but is still relatively high. Effort increased dramatically in in the north (Table 6 and Figure 10). Processors prefer large Atlantic surfclam but the sizes of landed Atlantic surfclam have declined in the south (Figures 22-27).

TOR 2. Present the survey data being used in the assessment (e.g., indices of relative or absolute abundance, recruitment, state surveys, age-length data, etc.). Use logbook data to investigate regional changes in LPUE, catch and effort. Characterize the uncertainty and any bias in these sources of data. Evaluate the spatial coverage, precision, and accuracy of the new clam survey.

The NEFSC clam survey used the *RV Delaware II* and a small 5 ft dredge (RD) prior to 2012 and a commercial fishing vessel and modified commercial dredge (MCD) since. The entire resource was surveyed with the RD in 2011 (Tables 10-11). The MCD was used in 2012 and 2015 in the south but only on GBK in 2013. Data from the two periods are not comparable although capture efficiency and size selectivity estimates can be used to calculate relatively consistent swept-area stock size for 1997-2015. Based on two swept-area estimates, biomass declined in the south after 2011 (Figure 39). It is not possible to evaluate recent trends off GBK.

Landings per unit effort declined steadily for the stock as a whole and in the south to near record lows in 2015 but is high on GBK (Table 8 and Figure 12). Survey data and other information indicate that the biological condition of the Atlantic surfclam resource as a whole and in the south is better than fishery conditions would suggest. Landings, effort and LPUE do not reliably measure trends in overall Atlantic surfclam stock size because the fishery operates in relatively few TNMS such that most of the stock and habitat are not accessed by the fishery (Figures 19-21).

TOR 3. Determine the extent and relative quality of benthic habitat for Atlantic surfclam in the Georges Bank ecosystem to refine estimates of stock size based on swept area calculations.

The proportion of untrawlable ground that is potentially poor clam habitat was recalculated to be 14% which is slightly higher than the 12% figure used in this assessment. New information will be available soon for refining these imprecise estimates (Appendix 13).

TOR 4. Quantify changes in the depth distribution of Atlantic surfclam over time. Review changes over time in Atlantic surfclam biological parameters such as length, width, and growth.

The distribution of Atlantic surfclam in the south is shifting towards deeper water due to warming as suitable nearshore habitat areas have decreased and offshore habitats increased (Figures 72-77). Survey data indicate that overlap between Atlantic surfclam and ocean quahogs which inhabit relatively deep water habitat has increased (Figures 78-79). Maximum shell length had declined in the south while the von Bertalanffy growth parameter K increased (Figures 86-87).

TOR 5. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series (integrating results from TOR 3, as appropriate) and estimate their uncertainty. Include a historical retrospective analysis to allow a comparison with previous assessment results and previous projections.

The primary assessment was a statistical catch at age model implemented in SS3. Each of two areas were assessed separately and the results were combined to provide management advice for the stock (Part 1.7). The scale of absolute abundance was uncertain, which is a problem typical of low fishing mortality fisheries. The trend in biomass was relatively well determined. The southern area, where recent recruitment has been strong is near its unfished biomass (B_0). The northern area, where recent recruitment has been poor is at a lower level, but still above $\frac{1}{4}B_0$. Fishing mortality is low for both areas.

TOR 6. State the existing stock status definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, F_{MSY} and MSY) and provide estimates of their uncertainty. If analytic model-based estimates are unavailable, consider recommending alternative measurable proxies for BRPs. Comment on the scientific adequacy of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs, particularly as they relate to stock assumptions.

The current and recommended stock status definitions are listed in Table 4 (Part 1.8). The current stock status definitions were revised based on a management strategy evaluation (Part 8) and assessment model improvements, because the overfishing definition depended on the estimate of absolute abundance in the assessment, which is uncertain. The recommended stock status definitions are trend-based as trend is relatively well estimated in this assessment.

TOR 7. Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to any new model or models developed for this peer review.

1. *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.*
2. *Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR-5).*

The Atlantic surfclam population is not overfished and overfishing is not occurring under either the current or recommended reference point definitions and using either the previous or newly developed models (Part 1.9; Tables 27 - 29).

TOR 8. Develop approaches and apply them to conduct stock projections.

1. *Provide numerical annual projections (five years) and the statistical distribution (e.g., probability density function) of the OFL (overfishing level) (see Appendix 15). Consider cases using nominal as well as potential levels of uncertainty in the model. 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).*
2. *Comment on which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.*
3. *Describe this stock's vulnerability (see 15) to becoming overfished, and how this could affect the choice of ABC.*

Projections indicate that the population is unlikely to be overfished and that overfishing is unlikely to occur by 2025 using a wide range of possible biomass scales and assumed catches (Part 1.10; Tables 30 - 31).

TOR 9. Evaluate the validity of the current stock definition. Determine whether current stock definitions may mask fishery related reductions in sustainable catch on regional spatial scales. Make a recommendation about whether there is a need to modify the current stock definition.

The invertebrate subcommittee did not reach consensus on stock definitions. All members of the workgroup agree that stock definitions are unlikely to affect management, yield, or biological risk in the near term as long as fishing mortality rates remain low and overall abundance and biomass are relatively high in both the northern and southern areas. If fishing mortality increases substantially, or a portion of the stock declines substantially, then the current stock definition has the potential to mask conditions in the affected area and lead to reduced yield and biomass. The single stock assumption also complicates and adds uncertainty to stock status determinations based on current and recommended reference points (Part 1.11).

TOR 10. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in most recent SARC reviewed assessment and review panel reports. Identify new research recommendations.

Research recommendations were reviewed and evaluated and new ones were developed (Part 1.12).

Terms of Reference

A. Atlantic surfclams

1. Estimate catch from all sources including landings and discards. Map the spatial and temporal distribution of landings, discards, fishing effort, and gross revenue, as appropriate. Characterize the uncertainty in these sources of data.

2. Present the survey data being used in the assessment (e.g., indices of relative or absolute abundance, recruitment, state surveys, age-length data, etc.). Use logbook data to investigate regional changes in LPUE, catch and effort. Characterize the uncertainty and any bias in these sources of data. Evaluate the spatial coverage, precision, and accuracy of the new clam survey.

3. Determine the extent and relative quality of benthic habitat for surfclams in the Georges Bank ecosystem to refine estimates of stock size based on swept area calculations.

4. Quantify changes in the depth distribution of surfclams over time. Review changes over time in surfclam biological parameters such as length, width, and growth.

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

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

7. Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to any new model or models developed for this peer review.

   (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).

8. Develop approaches and apply them to conduct stock projections.

   (a) Provide numerical annual projections (five years) and the statistical distribution (e.g., probability density function) of the OFL (overfishing level) (see Appendix to the SAW TORs). Consider cases using nominal as well as potential levels of uncertainty in the model. 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 15) to becoming overfished, and how this could affect the choice of ABC.

9. Evaluate the validity of the current stock definition. Determine whether current stock definitions may mask fishery related reductions in sustainable catch on regional spatial scales. Make a recommendation about whether there is a need to modify the current stock definition.

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

1.3 TOR 1: Commercial

In this assessment for Atlantic surfclam the northern area was federal waters (3-200 nm from shore) on Georges Bank and the southern area was federal waters from south and west of Georges Bank to Cape Hatteras (Figures 6 and 7). Commercial landings were provided in meat weights for ease of comparison to survey data and in analyses, but were originally reported in units of industry cages. Landings per unit of fishing effort (LPUE) data were reported in this assessment as landings in bushels per hour fished, based on mandatory clam logbook reports. The spatial resolution of the clam logbook reports was 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 stock assessments (Northeast Fisheries Science Center 2013), "catch" was defined as the sum of landings, plus 12% of landings, plus discards. Based on prior calculations (Northeast Fisheries Science Center 2003), Atlantic surfclam catch in previous assessments was assumed to be 12% larger than landings to account for incidental mortality of clams in the path of the dredge. The 12% figure was considered an upper bound or overestimate because the area fished (e.g. 155 km^2 during 2004) is small relative to area covered by the stock (Walace and Hoff 2005). Furthermore, the ITQ (see below) clam fishery operates with little or no regulation induced inefficiency due to area closures, trip limits, size limits, etc. so that fishing effort and incidental mortality are reduced. The support for this estimate was reevaluated in this assessment based on data also used by Northeast Fisheries Science Center (2003), and more realistic algebraic relationships proposed by Dr. Deborah Hart (NEFSC, Woods Hole, MA) for sea scallops in Northeast Fisheries Science Center (2014).

The ratio of Atlantic surfclam in the path of a commercial dredge that are caught relative to those killed but not caught is $R = \frac{e}{c(1-e)}$ where e is capture efficiency and c is the fraction that die but are not caught. Indirect mortality due to contact with a clam dredge is in the range of 5-20% with an extreme upper bound of 50% (Table C10, (Northeast Fisheries Science Center 2003)). If F_L is fishing mortality for landed Atlantic surfclam and F_I is the incidental mortality rate then $F_I = \frac{F_L}{R} = \frac{F_L c(1-e)}{e}$ and $\frac{F_I}{F_L} = \frac{c(1-e)}{e}$. The ratio $\frac{F_I}{F_L}$ is the same as the ratio of numbers landed to numbers killed but not caught. If landed and incidental clams have the same size composition, then the ratio of landed weight to incidental weight is also $\frac{F_I}{F_L}$. The average efficiency of a commercial clam dredge for Atlantic surfclam is about 0.73 (Table A10 in NEFSC 2003). The range of estimates $c = 0.05$, 0.2 and 0.5 indicate that incidental losses are 2%, 7% and 18% of landings which together average about 13%. The Subcommittee concluded that the 12% incidental mortality estimate was reasonable for Atlantic surfclam.

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

Discard data

Discards were zero from 2008-2015 since the last assessment. Some discards occurred during 1979-1993; as the result of a minimum size (shell length) requirement for landing that was in place over that period (Table 3). No new information about discards was available for this assessment.

Age and size at recruitment to the fishery

Age at recruitment to the Atlantic surfclam fishery depends on growth rates, which vary both spatially and temporally (see 1.4). The age at recruitment depends on the area being modeled (north vs. south), and 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 section (1.7).

Landings, fishing effort and prices

Landings and fishing effort data for 1982-2015 were from mandatory logbook reports (similar but more detailed than standard Vessel Trip Reports used in most other fisheries) with information on the location, duration, and landings of each trip. Data for earlier years were from Northeast Fisheries Science Center (2003) and Mid-Atlantic Fishery Management Council (2006).

Landings data from Atlantic surfclam logbooks are considered accurate in comparison to other fisheries because of the Individual Transferable Quota (ITQ) and cage tag systems. However, effort data are not reliable for 1981–1990 due to regulations that restricted the duration of fishing to 6 hours. Effort data are considered reliable for years before 1985 and after 1990.

Atlantic surfclam landings were mostly from the US Exclusive Economic Zone (EEZ) during 1965 to 2011 (Table 4 and Figure 8). 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 2015. Landings have not reached the quota of 26,218 mt since it was set in 2004 because of limited markets. The quotas are set at levels much lower than might be permitted under the FMP. Approximate state landings are shown in Table 4, and more accurate state landings are available in Appendices (7). Both New Jersey and New York have seen a sharp decline in Atlantic surfclam biomass within their state territorial waters over the past 15 years, and an accompanying drop in landings (7).

The bulk of EEZ landings were from the DMV region (Figure 7) during 1979-1980. After 1980, the bulk of landings were from the NJ region (Table 5 and Figure 9). Landings from LI were modest but began increasing in 2001. Landings from SNE were modest but increased starting in 2004. The high proportion of landings on GBK reflects the high catch rates there (see below).

Total fishing effort increased after 1990 and has been relatively high, but stable since 2007, particularly in the DMV and NJ regions (Table 6 and Figure 10). The bulk of the fishing effort was in areas where the majority of landings come from.

Real ex-vessel prices for the inshore and EEZ fisheries have been stable, since the mid-1990s (Table 7 and Figure 11). Nominal revenues for Atlantic surfclam during 2013 were about $33 million.

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 8 and Figure 12). Standardized LPUE was not estimated for this assessment because the data are not used analytically and because Northeast Fisheries Science Center (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 in SVA, DMV and NJ, which have recently been at or near record lows. LPUE in GBK and SNE have generally been high.

LPUE is not an ideal measure of fishable biomass trends for sessile and patchy stocks like Atlantic surfclam because fishermen target high density beds and change their operations to maintain relatively high catch rates as stock biomass declines (Hilborn et al. 1992).

Spatial patterns in fishery data

Mean landings, fishing effort, and LPUE were calculated by ten-minute square (TNMS) from 1979-2015 in 5 year blocks (Figures 13 – 18). Only TNMS where more than ten bu of Atlantic surfclam were caught over the time period were included in maps. TNMS with reported landings less than 10 bu were probably in error, or from just a few exploratory tows. Inclusion of TNMS, with less than 10 bu distorted the graphical presentations because the area fished appeared unrealistically large.

Figures 13 – 18 show the spatial patterns of the Atlantic surfclam fishery over most of its history. In most blocks, the greatest concentration of fishing effort and landings occurred in the same thirty or so TNMS in the NJ region, with intermittent fishing activity in other regions and recent emphasis on SNE and GBK.

TNMS 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.

Important TNMS

TNMS important to the fishery were identified by choosing the 10 TNMS from with the highest mean landings during each 5 year time block. For example, a TNMS important during 1991-1995 could be selected regardless of its importance during earlier or later time periods. The list contains a subset of the total TNMS, because of overlap between the time periods and because the same TNMS tend to remain important. 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. Trends in landings, effort, and LPUE were plotted (Figures 19 – 21) for each TNMS to show changes in conditions over time within individual TNMS.

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 approximately 30 random landed Atlantic surfclam from selected fishing trips each year (Table 9).

Port sample length frequency data from the four regions show modest variation in size of landed Atlantic surfclam over time with declines in modal size in DMV and NJ since 2008 (Figures 22 – 28). Care should be taken in interpreting these due to small sample sizes in some cases (especially LI, SNE and GBK), but in general the data indicate that most landed Atlantic surfclam have been larger than 120mm SL. Commercial size distributions are discussed in detail in section (1.7).

Fishery management

The Atlantic surfclam is managed by the Mid-Atlantic Fishery Management Council (Council). The Council is one of eight regional fishery management councils created when the United States (U.S.) Congress passed Public Law 94-265, the Magnuson Fishery Conservation And Management Act of 1976 (also known as Magnuson-Stevens Act or MSA). The law created a system of regional fisheries management designed to allow for regional, participatory governance. The Council develops fishery management plans and recommend management measures to the Secretary of Commerce through the National Marine Fisheries Service (NMFS) for its fisheries in the Exclusive Economic Zone of the U.S. (EEZ; 3-200 miles off the east coast). There are also fisheries for Atlantic surfclam in New Jersey, New York, and Massachusetts within state waters (within 3 miles of shore); the state authorities are responsible for managing these fisheries, although fishing and survey data for state fisheries were presented in this document (see 7).

Atlantic surfclam is managed with another species (Ocean quahog, *Arctica islandica*) under a single fishery management plan, that was first developed by the Council in 1977. The Atlantic surfclam fishery was initially managed through limited-entry restrictions, quarterly quotas, and fishing time restrictions. By the mid-1980s, effort limitation combined with overcapacity in the fishery meant that capacity utilization was very low, with vessels operating only 6 hours every other week in 1990. An individual transferrable quota (ITQ) system was established in 1990 which initially allocated shares to vessel owners based on a formula including historical catch and vessel size. Economic efficiency improved and management monitoring decreased as a result of initial ITQ implementation, but it also led to consolidation and displacement of labor (particularly non-vessel owning captains and crew). ITQ shares can be traded or leased to any non-foreign person or entity, with no pre-conditions of vessel ownership. Market consolidation and existing vertical integration have increased over time. From 1990 to 2005, the Atlantic surfclam fleet size decreased by about 70%.

Under the current management system, managers set an annual catch limit for Atlantic surfclam and allocate landings to the ITQ shares. The Council's annual catch limit recommendations for the upcoming fishing year(s) cannot exceed the acceptable biological catch (ABC) recommendation of its Scientific and Statistical Committee (SSC). The SSC serves as the Councils primary scientific/technical advisory body, and provides ongoing scientific advice for fishery management decisions, including recommendations for ABC, preventing overfishing, maximum sustainable yield, and achieving rebuilding targets.

In order to participate in the Atlantic surfclam fishery, fishermen must have a permit to commercially harvest and sell Atlantic surfclam (using valid ITQ shares), and there are mandatory reporting and vessel-monitoring requirements, as well as clam cage-tagging requirements. There is a minimum size for Atlantic surfclam, which can be suspended by managers if it is demonstrated the harvest of small Atlantic surfclam is below a certain threshold. Fishing areas can be closed due to environmental degradation or due to the toxins that cause paralytic shellfish poisoning (PSP). PSP is a public health concern for Atlantic surfclam. It is caused by saxitoxins, produced by the alga *Alexandrium fundyense* (red tide), that accumulate in shellfish, and has resulted in fishery closures in the Georges Bank Area of the EEZ. NMFS recently (2013) reopened portions of the closed areas to harvest of Atlantic surfclam for those vessels using a protocol for onboard screening and dockside testing to verify that clams harvested from these areas are safe. Areas can also be closed to Atlantic surfclam fishing if the abundance of small clams in an area meets certain threshold criteria. This small Atlantic surfclam closure provision was applied during the 1980's with three area closures (off Atlantic City, NJ, Ocean City, MD, and Chincoteague, VA), with the last of the three areas reopening in 1991.

1.4 TOR 2: Survey

NEFSC clam surveys

Survey data used in this assessment were from 2 different sampling platforms. The first was the NEFSC clam surveys conducted during 1982–2011 by the *RV 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 (Atlantic surfclam 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 Atlantic surfclam 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 Northeast Fisheries Science Center (2003). The second survey platform was the *ESS Pursuit*, a commercial vessel that was contracted to conduct the NEFSC clam survey since 2012, when the *RV Delaware II* was retired. The *ESS Pursuit* used a modified commercial dredge described in detail in Hennen et al. (2016). Surveys conducted from the *ESS Pursuit* have taken place in August each year since 2012.

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 (Northeast Fisheries Science Center 2003)).

NEFSC clam surveys were organized around NEFSC shellfish strata and stock assessment regions (Figure 7). Most Atlantic surfclam landings originate from areas covered by the survey. The survey did not cover GBK during 2005 and provided marginal coverage there 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 10). Survey data were never borrowed from surveys before the previous, or beyond the next survey. A model–based imputation was investigated for the last assessment (Northeast Fisheries Science Center 2013), but the imputation tended to over–emphasize unsampled years and areas. Alternative approaches to imputing missing strata were not further pursued in this assessment.

Surveys followed a stratified random sampling design, allocating a pre-determined number of tows to each stratum. A standard tow was 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 was typically longer than 0.125 nm (Weinberg et al. 2002). These problems were eliminated in 2012 when the survey was switched to the *ESS Pursuit*. For trend analysis, when using data from before 2012, 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 start and stop times recorded on the bridge. Stations used to measure trends in Atlantic 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. Other non-random stations were occupied for a variety of purposes (e.g. selectivity experiments) but not used to estimate trends in abundance. Locations and catches of all stations in the survey have been mapped (Figures 29–34).

Occasionally, randomly selected stations were found to be too rocky or rough to tow, particularly on GBK. The proportion of random stations that could not be fished was an estimate of the proportion of habitat in an area that was not suitable habitat for Atlantic surfclam (1.5). These estimates were used in the calculation of Atlantic surfclam swept-area biomass (see below).

Following most survey tows, all Atlantic surfclam in the survey dredge are counted and shell length is measured to the nearest mm. 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 1997–2015 surveys (see below).

Survey tow distance and gear performance based on sensor data

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), 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 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.

Determination of time fishing

The determination of time fishing, the "fishing seconds" for each tow (after 1997), was based on a measurement of the pitch of the dredge during each second of the tow. Pitch data were smoothed using a 7 second moving average and then compared to a "critical angle" to determine when the dredge was fishing effectively. When the dredge was above the critical angle it was assumed to be pitched too steeply for the blade to penetrate the sediment. When the dredge was pitched below the critical angle, it was assumed to be near enough to horizontal that the blade should penetrate and thus be actively fishing.

It is important to find a critical angle for tow distance that is neither too small, nor too large. 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. Further discussion of the determination of critical angle as well as summaries of dredge performance by year are in appendices (16–18).

NEFSC clam survey trends and composition data

NEFSC clam survey data for Atlantic surfclam, including the number and weight caught per tow were tabulated by year, region and for the entire stock (Table 11). Mean numbers per tow were used in the plots of trends 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 Atlantic surfclam (Figure 35) provide some evidence for recruitment trends over time. Recruitment appears to be increasing in DMV, NJ, LI, and SNE since the last assessment. Survey trends for fishable (120+mm) Atlantic surfclam (Figures 36) show evidence of decreasing abundance in the SVA, and possibly LI regions, but there are increasing trends in abundance in DMV, NJ and SNE. We cannot make inference on trends in abundance or recruitment on GBK because there is only one data point available from the new survey. Based on survey data for the entire southern area, recruitment and fishable abundance have been increasing since the last assessment in 2011 (Figures 37 – 39).

Survey age–length keys and stratified mean length composition data were used to estimate the age composition of Atlantic surfclam 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 2015 when surveys occurred. Ages ranged from 1-37 (Figures 40 – 46). Specific year classes and trends in length and age composition are discussed in the context of the assessment model (see 1.6).

Shell length composition data (Figures 47 – 52) can be helpful in visually identifying shifts in population demography. For example, there is evidence of recent recruitment in the southern area regions.

Dredge efficiency

Changes to the NEFSC survey involved changes to the survey gear. In particular, shifting the survey dredge from the research dredge (RD) used on the *RV Delaware II* to the modified commercial dredge (MCD) used on the *ESS Pursuit* was an important modification in that it necessitated a re-evaluation of capture efficiency. Fortunately the MCD was the same dredge that was used in previous depletion experiments (Northeast Fisheries Science Center 2013) so estimates of capture efficiency already exist. These are discussed in detail in Appendix 4 and Northeast Fisheries Science Center (2013).

Estimates of survey dredge efficiency were used to generate prior distributions for capture efficiency for each survey in the assessment model (see 1.7). A comparison of the prior distribution for the RD to the prior distribution for the MCD shows that the MCD has higher and more precisely estimated efficiency (Figure 53).

Size selectivity

Selectivity data were collected on the *ESS Pursuit* during selectivity experiments in 2008 – 2015. Data from the experiments were used to estimate size-selectivity for the MCD. The MCD was configured for survey operations, rather than commercial fishing operations. Thus, the size selectivity estimates for the commercial dredge used by the *ESS Pursuit* during cooperative survey work are not directly applicable to commercial catch data. Selectivity experiments are described in Hennen et al. (2016).

The data available for each selectivity study site included shell length data from: one MCD tow, and one F/V selectivity tow using either a commercial dredge lined with wire mesh or a specially designed selectivity dredge (SD). Gear testing work done in 2014 showed that the SD and the lined commercial dredge should be interchangeable in selectivity studies (Hennen et al. (2016)).

Shell length data from selectivity experiments conducted since the last assessment were tabulated using 1 mm shell length size groups (Tables 12 – 13). Survey size selectivity was estimated using data from 47 total sites.

Selectivity was modelled as a generalized additive mixed model (GAMM), where the shell length bin was a factor, predicting the binomial proportion of the survey catch over the total catch (SD + MCD). The fully saturated model was

$$P_L = e^{(\alpha + s(L) + s[YrSta, L] + offset)} \tag{1}$$

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 variables, in this case shell length (L) and a random effect (indicated with braces) due to station and year. The final term is an offset (Pinheiro and Bates (2006)) based on the tow distance at each station. Tow distance 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, while the SD was towed for 2 min.

Using the GAMM 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)[2] 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 modeller.

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 the differences between them (Figure 54). 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

$$CI_L = elogit^{(\rho_L \pm 1.96\sigma_L)} \tag{2}$$

Where CI_L is the approximate confidence interval for selectivity at length L, ρ_L is the corresponding logit scale model estimate, σ_L is the standard error and elogit is the inverse of the logit function.

Selectivity estimates (Tables 14 – 15; Figure 55) were used to generate swept area and survey index plots (Figures 35 – 39) and are useful for comparison to assessment model results.

[2] See R package mgcv documentation

Shell length, meat weight relationships

The shell length-meat weight (SLMW) relationships are important because they are used to convert numbers of Atlantic surfclam 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 Atlantic 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 and Dichmont (2004)) were used to predict clam meat weight, using equations of the form:

$$MW = e^{(\alpha+\beta_0 ln(L)+\beta_1 c_1+\beta_2 c_2+\cdots+\beta_n c_n)} \tag{3}$$

where MW was meat weight, L was shell length, $c_1, \cdots, c_n$ were covariate predictors (*e.g.*, region or depth), and α and β_i were the estimated parameters. 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 reasonable for the distributions of meat weights (McCullagh and Nelder 1989). The GLMM in all analyses used the quasi-Poisson family with a log link. Quasi-Poisson is a Poisson distribution with a variance inflation parameter that relaxes the Poisson requirement that the mean must equal the variance. Because shell length to meat weight relationships for Atlantic surfclam 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 16). The best model by AIC and BIC was a model with fixed effects for shell length, depth, and region and random effects for shell length slope and the intercept, using both the year and the station as the grouping variables.

Regional differences in meat weight are meaningful, particularly for the largest animals (Figure 56), though some of the differences between regions can be explained by the different depths found there (Figure 57).

Age and growth

Atlantic surfclam were measured at sea and the shells were retained for ageing in the laboratory. Shells for ageing 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 (Northeast Fisheries Science Center 2010). Age and length samples are available for most regions, but not from every survey (Table 17).

Plots of age vs. shell length by year and region (Figures 58 – 64) indicate that growth patterns have been relatively constant in most regions over time with DMV and NJ, where growth has slowed and maximum size has decreased over the last two decades.

Von Bertalanffy parameters for growth in shell length were estimated for each region and each survey year for which sufficient data existed (Table 18). The Von Bertalanffy growth curve used in the calculations was:

$$L_a = L_\infty(1 - e^{(-k(a-t_0))}) \quad (4)$$

Where L_a was length (mm) at age a, and L_∞, k and t_0 are Von Bertalanffy parameters.

Atlantic surfclam are thought to mature very early. Data are limited but Atlantic surfclam off New Jersey may reach maturity as early as 3 months after settlement and at lengths of less than 5 mm (Chintala and Grassle 1995; Chintala 1997).

Survey trends and LPUE for important ten-minute squares

We analyzed commercial LPUE and survey data for 1982 - 2011 for important ten-minute squares (TNMS see section 1.3) in the southern New Jersey and Delmarva regions where fishing is traditionally concentrated to better understand potential fishing effects on key southern fishing grounds. Modes in size composition data from the commercial catch declined steadily in these areas over the last decade (Figures 22 – 28) but the declines are not clear in survey size composition data through 2011 when survey gear changed (Figures 47 – 52), probably due to size selective removals of large clams on fishing grounds. Survey and LPUE data suggest that abundance trends in areas where fishing occurs were similar to trends for the New Jersey and Delmarva regions as a whole. Thus, fishing seems to have had modest effects on abundance in TNMS where fishing was highest.

TNMS were much smaller than survey strata and not all squares were sampled during each year. We therefore analyzed the data "as is" (ignoring the unsampled squares) and after filling the holes with imputed survey "data" from a GAM model. The GAM model (mgcv library in the R programming language) was $gam(N_{tow} \sim s(Y, tnms) + tnms)$. In this model, N_{tow} is the number of Atlantic surfclam caught in the tow, Y is the survey year (continuous) and $tnms$ is the ten-minute square (a categorical factor). About 5% of survey tows had zero catch so we fit the model using the default log link function assuming errors from a Tweedie distribution, which is a combination of a logistic distribution (for zero observations) and a Gamma distribution (for positive catches). Given these specifications, the model handles zero and non-zero catches directly while estimating a different intercept (average catch rate) and different interannual trends for each TNMS. In effect, there was a separate model for each TNMS.

The imputed data from the fitted GAM model (R^2 = 0.48, deviance explained=75%, N=299) amount to interpolations between years with observed data and extrapolations for missing years at the beginning and end of the time series (Figure 65). Extrapolation is possible in the mgcv GAM software as long as the years involved are within the range of years in the dataset as a whole, even though the models for different TNMS are nearly independent. The surveys were usually triennial so that interpolations and extrapolations were over relatively long periods of time (1-11 years). Extrapolation is not valid from a statistical point of view and should (along with interpolations over many years) be viewed with caution but the analysis was exploratory and results did not depend strongly on using imputed data (see below).

Interannual time series for the New Jersey and Delmarva regions were calculated by averaging all values (observed and/or predicted values) for each region and year. TNMS were the same size so the annual averages amounted to stratified random mean numbers per survey tow. Results with and without imputed data were similar (Figure 66). All results indicate that abundance declined rapidly during 1995-2005 (on fishing grounds) to current relatively low levels (Figure 67).

LPUE and survey data for important TNMS show that LPUE remained high as abundance declined off New Jersey (correlation coefficient $\rho = 0.2$, Figure 67). Survey trends in important TNMS and for New Jersey as a whole were strongly correlated ($\rho = 0.79$). In contrast to New Jersey, trends in survey and LPUE in important TNMS off Delmarva had a linear relationship and were strongly correlated ($\rho = 0.59$). Survey trends in important TNMS and for Delmarva as a whole were also strongly correlated ($\rho = 0.52$).

Evaluation of new survey

Spatial coverage

The assessment working group reviewed information showing fishing activity and survey catches in an area south of Nantucket that is not routinely surveyed, they also evaluated several approaches for identifying Atlantic surfclam habitat based on data from multiple surveys, multi-beam acoustic data, published studies, environmental measurements and habitat suitability models (Appendix 13). Such data would be useful for expanding the survey to cover new grounds, restratification and in improving the NEFSC clam survey design. The approaches presented appeared potentially useful and should be further developed for consideration by a future working group tasked specifically with evaluating survey design. NEFSC Survey Branch personnel and program managers would need to be heavily involved in the discussions.

Changes in the spatial distribution of biomass

We calculated relative swept-area survey biomass of Atlantic surfclam (all sizes) by region and area during 2012-2015. No adjustments were made for capture efficiency, size selectivity or changes when the new survey began in 2012 to keep the analysis simple and because these parameters may be the same for all regions in the same year and should tend to "cancel out". The proportion of biomass in year y and region r was calculated $p_{y,r} = \frac{p_{y,r}}{\sum_s p_{y,s} A_s}$ where A_s is the area (nm^2) of one of the regions. The northern and southern areas were sampled in different years after 2011, so data from the survey in the northern area during 2013 was used in these calculations for both 2012 and 2015.

Results show the increase in the proportion of total biomass in GBK and declines off DMV during 1982-2011 measured using the old survey dredge (Figure 68). These patterns were attributed to rising water temperatures in the last assessment. Unexpectedly, proportions of total biomass on GBK dropped during 2012-2015 while fractions in NJ and DMV increased based on survey data from the new dredge. Biomass indices increased after 2011 in all regions because the new dredge is more efficient and sweeps more ground. However, increases during 2012-2015 in the south were larger than increases in the north. It is possible that these patterns reflect changes in spatial distribution but they may also be due to reduced capture efficiency in the new survey using the MCD in the relatively rough and rocky GBK region. The latter possibility could be investigated by conducting depletion studies on GBK to estimate capture efficiency directly.

Precision

The MCD survey was expected to be more precise than the original survey because the new dredge is more efficient (see 4), tow distance is more consistent (see below) and because the area swept by a tow in the new survey is larger (RD mean about 580 m^2 with CV=25% and MCD mean=1764 m^2 with CV=11%). However, there was no clear reduction in CVs for survey abundance indices (stratified mean catch per tow) with the MCD (Table 11 and Figures 35–38). Lower numbers of tows beginning in 2012 reduced the precision of abundance indices for the southern area. There is no evidence that the variance among individual tows in the same stratum was reduced after 2012 (Table 10 and Figure 35–38). However, swept-area stock size estimates were probably more precise (Figure 38) when using the MCD despite little or no improvement in abundance indices because capture efficiency estimates for the MCD are more precise than estimates for the RD (Figure 53; Table 10) and 4).

Borrowing should be less common in the future because NEFSC expects to survey the northern and southern areas completely during sequential years rather than in parts (see 2 for a discussion of the borrowing required for this assessment). This plan and the goal of reducing the frequency of unsampled strata are important because of the difficulties in borrowing lengths and ages from other years now that length and age data are used in the assessment model. Borrowing from adjacent surveys is a type of imputation, but further work on imputation techniques is warranted. NEFSC (2007) used negative binomial GAM models to impute catches for strata with no data that could not be filled by borrowing but with modest effect on results. Model based approaches might have larger effect if all strata with missing data were imputed.

The total number of stations in the NEFSC clam survey is limited by the time devoted to the survey with deductions for transit time, bad weather, etc. The proportion of the total number of random survey stations in each stratum for each region (northern or southern) in the new survey was based on stratum area and on the mean catch and variance in catch for Atlantic surfclam plus ocean quahogs in previous surveys (Cochran 1977). This standard approach minimizes the variance of the total stratified mean catch per tow but does not minimize the variance for either individual species.

It might be possible to improve overall precision of the clam survey by changing the relative amounts of time used to sample the northern and the southern areas in subsequent years without changing the total time or cost for the survey as a whole.

The precision of stratified random mean estimates like clam survey abundance estimates depends on the number of tows and variance in catches within each stratum (Table 10). The reduction in the number of tows in the southern area after 2011 increased the standard deviation and CVs of the stratified means.

Tow distance with the RD varied strongly with depth and among years when tow procedures changed unintentionally (Figure 69). Tow distances since 2012 have been less variable in general and relatively constant across depth and years. These changes should improve precision of survey data for recent years.

Analysis of precision was complicated by limited number or zero samples in some strata and years, a high proportion of tows with no catch (about 60%), different temporal trends among strata, and the tendency for variance to increase with the mean catch per tow. We dealt with these problems by considering variance in the proportion of positive tows and variance in log catch for positive tows separately, and by calculating the variance of randomized quantile residuals from models with likelihoods that were calculated using the compound Tweedie distribution which accommodates both zeroes and positive values. These analyses used data from random tows with sensor data collected since 1997, from strata that were sampled consistently (in all surveys) in each region (Table 19). In particular, we used survey data from northern strata 55, 57, 59, 61, 70-71 and 73-74 sampled during 1997, 1999, 2002, 2008, 2011 (RD), and 2014 (MCD) and data for southern strata 9, 10, 13, 17, 18, 21-22, 25-26, 29-30, 33-34 and 84-93 sampled during 1997, 1999, 2002, 2005, 2008, 2011 (RD), 2012 and 2015 (MCD) in the south.

To begin, we calculated the mean and standard deviations for a dummy variable that identified positive tows (=1 if Atlantic surfclam were caught and 0 otherwise) and for log of Atlantic surfclam catch in positive tows (Figure 70). There were no obvious changes in the proportion of positive tows in the new survey or in the variance of the dummy variable or log positive catch. Higher proportions and lower variance in the dummy variable for positive tows might be expected using a dredge that affords higher precision although positive tows are likely at even low Atlantic surfclam densities using either survey dredge (10).

Next, we fit a series of GLM and GAM models to catches (tows with and without catch combined) and used AIC to determine the "best" (by AIC) model (Table 20). The best model (gamB) explained 45% of the total deviance, 23% of the total variance and the residuals were close to normally distributed. The distributions and standard deviation of residuals from the best model do not indicate increases in precision of individual tows beginning in 2012 (Figure 71).

1.5 TOR 3: Habitat

This TOR was driven by concern that relatively high densities of clams measured by survey tows in easy to sample areas on Georges Bank might be applied to rocky low density habitats that are difficult to sample so that model and swept-area biomass estimates are biased high. In stock assessment calculations, stock biomass $B = \frac{bA}{ae} = bQ$ where b is mean catch per tow, A is the area surveyed (the parameter of concern), a is area swept and e is capture efficiency. In recent assessments, the area surveyed on Georges Bank (A) was reduced by 12% assuming that the proportion of untowable stations represents grounds that were poor habitat with no Atlantic surfclam. For this assessment, the working group reviewed survey procedures and recalculated the proportion of untowable ground.

A list of random survey stations is prepared prior to the first leg of each clam survey and the captain determines towability when the ship reaches each random station. In the past, during the 1999, 2002, and 2005 surveys (Georges Bank was not surveyed in 2005), untowable stations were noted in station logs using a special "SHG=151" code. In the more recent survey during 2013-2014, text in comment fields and other SHG codes can be used to determine if a station was untowable, if the dredge was filled with rocks and no Atlantic surfclam, or the dredge was damaged by rocks. Based on "151" codes, 12/83=14% of random stations on GBK were not trawlable. In later years, 13/74=18% of random stations were not trawlable. The combined average (14%) is somewhat higher than the 12% figure used in this assessment. New habitat databases and models under development will soon be available for refining estimates of poor Atlantic surfclam habitat for GBK (Appendix 13). In addition, procedures for dealing with untrawlable stations in the survey may need to be modified so that this information is collected routinely and is clear in the survey database.

1.6 TOR 4: Depth and changes in biological parameters

As ocean temperatures increase, the distribution and biology of Atlantic surfclam are potentially changing with potential effects on fishery productivity. For example, increasing water temperature may result in changes to the biological parameters that describe growth (Munroe et al. 2016). Increasing water temperature may also be driving a shift in Atlantic surfclam distribution, to deeper water in the southern area (Weinberg et al. 2002). It is reasonable to assume that any responses to temperature would be strongest in the southern-most regions (SVA, DMV and NJ), where ocean temperatures are warmest and probably nearest the warm water tolerance for Atlantic surfclam.

Depth and temperature

Survey stations are distributed randomly relative to depth within a stratum and the same strata tend to be sampled over time within a region (Table 10). Therefore, if the depth distribution of Atlantic surfclam were trending over time, the depth at which most of the animals were caught within a region might be expected to increase. Plots of the depth at which the median cumulative catch within each region occurs over time show this relationship in two regions, DMV and NJ (Figures 72 – 77).

Warming coastal waters might change the spatial overlap between Atlantic surfclam in relatively shallow water and ocean quahogs that are found in adjacent deeper water. Overlap is important because the fishery operates most efficiently where only one species is caught. The depth at which 95% of the cumulative catch of Atlantic surfclam was taken during 1982-2011 clam surveys was used as the offshore habitat boundary for Atlantic surfclam and the depth at which 5% of the cumulative quahog catch was used as the inshore boundary for ocean quahogs (Figure 78). In the 1980s and with the exception of the LI region, the two habitat boundaries were similar indicating that the habitat was partitioned across depth as expected. There was no evidence that the inshore boundary of ocean quahog habitat changed in later years but there was clear evidence that the offshore boundary of Atlantic surfclam habitat shifted to deeper water in the southern NJ and DMV regions and, surprisingly, in the northern most GBK region. By the mid- to late 1990s, the overlap in Atlantic surfclam and ocean quahog habitat was pronounced in the south. The shift on GBK may have been due to increases in Atlantic surfclam abundance (Figure 36). In contrast, abundance generally decreased after 1982 in the south and the change in habitat boundaries was more likely. Results for LI were anomalous given that the offshore boundary for Atlantic surfclam was consistently deeper than the inshore boundary for ocean quahogs, probably due to high density beds of ocean quahogs in cold shallow water (Figure 78) and the increased presence of clay as substrate, which tends to contain more ocean quahog than Atlantic surfclam.

The sampling properties of presence-absence data from NEFSC survey tows were characterized analytically (Appendix 11). Results show that survey tows are almost certain to detect clams at relatively low densities (roughly 0.013 per m^2, corresponding to about 15 encounters per tow in the RD). Thus, presence absence data are useful for detecting clams at relatively low densities but not for tracking trends in abundance when density is higher. Based on these results, presence-absence data were used in this assessment to quantify extent but neither quality of habitat nor density of clams.

Presence-absence GAM models showed that the probability of co-occurrence (both species in the same tow) decreased almost linearly during 1982-2011 in the SNE region while increasing almost linearly in the LI and NJ regions (Figure 79 and Appendix 10). Trends were not statistically significant in the DMV and GBK regions where strong changes in abundance may complicate interpretation.

The amount of habitat for Atlantic surfclam was quantified by dividing the area surveyed consistently in each region into relatively small areas based on latitude and longitude as well as two other coordinate systems (Appendix 12). Presence-absence GAM models with time and position as predictor variables were selected from a set of candidates based on AIC. Habitat was quantified by summing the predicted probability of a positive tow from the best model over all of the small areas in each region and year. Results suggest that habitat area declined in the south in the DMV area due to losses in shallow water, increased along the central Mid-Atlantic Bight (NJ and LI areas) due to increases in deep water and varied without trend in the north (SNE and GBK areas). Temperature data were not available but these changes were likely due to water temperatures increasing above the preferred range for Atlantic surfclam in nearshore coastal areas off DMV (Weinberg 2005) and above the lower bound of the preferred range in deep waters off NJ and LI.

Temperature was recorded as part of the survey station data (beginning in 2002), and may be a useful indicator of habitat preference for Atlantic surfclam. Plots of the temperature and depth recorded at each survey station over time, against the total number of Atlantic surfclam caught are provided here (Figures 80 – 85). The results indicate that temperature and depth preferences vary by region, but appear to be relatively consistent over (recent) time. This may be indicative of local adaptation, or there may be other local factors, potentially correlated with temperature and depth, that influence habitat preference in each region.

Changes in biological parameters

If increasing ocean temperature negatively affects the fitness of Atlantic surfclam, one might expect to see decreases in the biological parameters that describe growth, particularly in the southernmost regions where water temperatures are highest. Analysis indicates that DMV and NJ have experienced declines in average maximum length (L_∞) through time (Figure 86). NJ and SNE have shown decreases in the rate at which an animal approaches its theoretical maximum size (K; Figure 87).

1.7 TOR 5: Model

The Atlantic surfclam assessment model was implemented in SS3[3] (Methot and Wetzel 2013). Separate SS3 models were developed for Atlantic surfclam in the southern and northern areas. 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 in the north, and differences in occurrence of strong year classes) made it too difficult to estimate "average" population dynamics for the areas combined. Also, data would be lost if 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 quantities for the combined regions were estimated by combining elements for the southern and northern areas.

Configuration

Fishery and survey selectivity were functions of size rather than age in SS3 models. Conditional age at length data, rather than traditional age composition data, were used in fitting models. The conditional age vector with indices 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 year-class strength is preserved. Size composition data are not used twice (once as size composition data and once in calculation of traditional catch at age). Finally, the sampling distribution of conditional age data is probably easier and more accurately characterized as a multinomial, conditional on the number of ages (at t and L) actually sampled.

The same types of data (Figures 88 and 89) were available for both areas, although more precise and numerous data were available for the southern area. The additional data for the south made it possible to estimate additional catchability, recruitment and selectivity parameters, as well as biomass and mortality over a longer time period (Tables 21 – 22). It was necessary to borrow some of these parameter estimates from the south in modelling Atlantic surfclam in the north because data were so limited and catches were nearly zero over much of the time series.

Dome shaped survey selectivity curves with parameters fixed at field study estimates were used in SS3 models for the MCD survey in the south and north and the RD survey in the south (Figure 90). 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 55; Northeast Fisheries Science Center (2013)). Allowing the model for the north to estimate the ascending limb of the RD survey selectivity curve was helpful in reducing diagnostic problems.

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 Atlantic surfclam 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

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

from the CV for mean numbers per tow in each year (and assumed 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 based on the recommendations of (Francis 2011). The initial standard deviations for survey trend data were tuned, if necessary, based on preliminary model fits by adding a constant to the standard deviation for each observation in the time series (Francis 2011).

Priors for survey dredge capture efficiency

The prior distributions for survey dredge capture efficiencies were important because the models are not otherwise strongly informed regarding scale. The last Atlantic surfclam assessment (Northeast Fisheries Science Center 2013) details the work that was done to estimate a prior for the distribution of capture efficiency for the research dredge (RD) last used in 2011. Appendix 4 details the work done to estimate a prior for the distribution of the modified commercial dredge (MCD) used since 2011.

Issues

South

The Atlantic surfclam assessment for the south is unable to estimate scale (absolute stock size) although trends in biomass were estimated more reliably. This is typical of a low F fishery. In general, there are several different scenarios involving combinations of selectivity, biological parameters and biomass scale that might explain the observed population dynamics when fishing mortality cannot account for it. Therefore the model is easily shifted from one scale to another based on small changes in the data or model.

Some of the issues with the assessment model for the south stem from the fact that there are only two years of data in the MCD survey. Because of this limitation, the prior distribution on the MCD survey catchability was very influential (see section 1.7).

The base model has some poorly determined parameters (Table 23). Most of these are recruitment deviations, which are generally difficult to estimate when the survey and commercial gear do not sample the youngest animals well. Both the survey and commercial gear have selectivity curves such that they are unlikely to capture very many animals less than about 3 years old. Therefore, poorly determined recruitment deviations are not unexpected. The model has particular trouble estimating the Q parameter associated with the catchability of the MCD survey. This is probably because the survey contributes only two data points to the model. This parameter is therefore strongly influenced by its prior distribution. Sensitivities in which the prior distribution was turned off were run and are discussed in section 1.7. The other poorly determined parameter is the one that describes the width of the plateau of the selectivity curve for the fishery (see Figure 90). This parameter was difficult to determine because the commercial length comps show conflicting tendencies over time. In the early years of the fishery the length composition was heavily weighted towards longer clams, and in later years the composition was broader and shows a higher proportion of smaller clams. This pattern was difficult to fit with one selectivity curve. Sensitivity runs designed to estimate this parameter better using time varying selectivity are described in section 1.7.

Other potential issues are the use of assumed parameter values for M, steepness and growth. The growth values estimated in the model were near the experimentally derived values presented in part 1.4 and the M value used was based on observed longevity. There was no experimental basis for the assumed steepness ($h = 0.95$) used in the assessment model as there were no observations of recruitment at low stock size available. The h used was high and resulted in no apparent relationship between spawning biomass and recruitment. Sensitivities testing each of these assumptions are described in section 1.7.

North

The Atlantic surfclam assessment model for the north is also uncertain relative to scale. As in the south, the model does not have enough information to estimate scale with precision because the population is lightly fished and there is little contrast in the survey indices. The model from the north also suffers from a shorter time series for catch, survey, age composition, and length composition data.

The estimated biomass trend in the early part of the time series does not fit the survey index well. The early part of the time series is uncertain relative to trend because the survey index increased rapidly in the absence of any prior fishery removals that would have accounted for the population being in a depleted state (where the increase would represent recovery). There is no support for a low biomass in the early part of the time series in the composition data either. With no mechanism to explain the increase from 1984 to 1995 (or more precisely the low biomass in 1984), the model does not believe the survey. Sensitivities to explore the affect of forcing the model to fit the survey index better are discussed in section 1.7.

The base model has some poorly determined parameters (Table 24). Most of these are recruitment deviations, which are generally difficult to estimate when the survey and commercial gear do not sample the youngest animals well. Model precision can be improved by increasing the weighting on the MCD survey. This approach was not taken because the MCD survey consists of only one data point and because increased model precision is not desirable when the information provided to the assessment is uncertain. In the case of the northern area, the model has little meaningful information and should not reflect an unrealistically precise estimate of biomass. Sensitivity runs in which the MCD index was heavily weighted and also removed from the calculation of the likelihood surface are described in section 1.7.

Fit and estimates from basecase models

South

The biological parameters used in the assessment model were based on experimentally derived values (Figures 91 – 93). Fishery selectivity was estimated and retained the domed shape seen in the last assessment (Figure 90). The fit to the surveys was acceptable and the residuals did not show trends or high variance (Figures 94 – 96). The fit to the composition data was generally tight, with the possible exception of the MCD survey which showed conflicting length composition over only two years and was difficult to fit well with one selectivity pattern (Figures 97 – 109). Data weighting decisions are shown in Figure 110. Model time series results are shown and in Figure 111 and parameter estimates are shown in Table 22.

North

The biological parameters used in the assessment model were based on experimentally derived values (Figures 112 – 114). Fishery selectivity was partially estimated and shared the domed shape seen in the model for the south (Figure 115). The fit to the surveys was reasonable given the constraints of the data and the residuals did not have high variance (Figures 116 – 117). The fit to the composition data was generally tight, with the possible exception of the MCD survey which had only a single year of data (Figures 118 – 128). Selectivity for the MCD survey was assumed because allowing the model to fit a single year of data would have resulted in overfitting. Data weighting decisions are shown in Figure 129. Model time series results are shown and in Figure 130 and parameter estimates are shown in Table 22.

Likelihood profile analysis

South

Likelihood profile analysis of the model for the southern area consisted of fixing the unfished recruitment parameter (R_0) at successive values that bracketed the R_0 solution (from the base case model) and estimating all of the other parameters in the model.

Likelihood profile results for the south indicate that goodness of fit for the priors on survey catchability were best near the basecase model run (Table 25 and Figure 131). Survey age data support higher R_0 (higher biomass) and length composition data lower R_0 (lower biomass). However, the differences in total likelihood were small (Table 25). The one area of data conflict that appears to make a substantial difference in total likelihood is between the parameter prior distributions (on survey catchability), which prefers the solution, and the age composition data, which prefer a lower values of R_0.

North

There is model tension between the RD survey index and its composition data (Table 26 and Figure 132) in the model for the north. The composition data support a higher R_0 (higher biomass), while the survey data support a smaller R_0 (lower biomass). The biomass scale at the solution is set by prior distributions on survey catchability, which affect the MCD survey and RDscale index (RDscale did not contribute to the likelihood in the north because the Q were not estimated and RDscale was not fit for trend. See Table 21).

Sensitivities

Experimental model runs testing the effects of model manipulations (for example with either extra parameters or fewer sources of data) were informative.

South

Natural mortality was fixed at $M = 0.15$, based on the observed longevity of Atlantic surfclam in the base model, and an experimental run was conducted to estimate it. M was estimable and decreased with age (Figure 133). Estimating M produced a slightly better fit to the commercial length composition data, but a slightly worse fit to the survey length composition data compared to the base run. The fits to other data were unchanged. There was virtually no change in either the trend or scale of biomass and the base model was preferred due to parsimony.

The growth parameter K was fixed at values derived in the last assessment (Northeast Fisheries Science Center 2013) in the base model run. An experimental run attempted to estimate it. The Von Bertalanffy K parameter and the coefficients of variation around the growth curve were estimable. The estimated K was slightly less than the K assumed in the base model, while the estimated parameters describing uncertainty around the growth curve were nearly identical to the values used in the base model. Estimating growth had virtually no effect on the model fit and the base model was preferred due to parsimony.

There is experimental evidence that growth has changed over time in at least part of the southern stock area (Figure 86). In one sensitivity run growth was allowed to vary over time. The closest SS3 equivalent to the Von Bertalanffy L_∞ parameter was estimated for two time blocks (<2000 and >1999). This run had a negligible effect on the biomass estimates in the model (Figure 134). An additional run in which the Von Bertalanffy K parameters were fit in each of the time blocks produced only slight changes as well. This run improved the fit to the length composition very slightly at a minor cost to the fit of the conditional age at length composition data. There was virtually no change in either the trend or scale of biomass and the base model was preferred due to parsimony.

Although commercial selectivity was estimated, it may have changed over time. The evidence for this is in the apparent lack of fit to commercial length composition data that occurs in the early years of the time series (Figure 97) and the fact that the the model has trouble estimating the parameter describing the width of the plateau of the commercial selectivity curve (Table 23). The gear used by the commercial fishery has not changed substantially over time so any changes in fishery selectivity were probably due to changes in behavior. That is, the fishery probably targeted the beds with the largest and oldest clams first, and then later moved to beds of smaller clams when those were fished down. Sensitivity runs where commercial length composition was allowed to vary in time blocks (<1986 and >1985) produced better fits to the commercial length composition data. The overall fit to the commercial length composition data was already fairly good, however, and there was virtually no change in either the trend or scale of biomass and the base model was preferred due to parsimony.

The base model somewhat underfits the RD survey (Figure 96). This base case solution is fairly stable. In order to force the model into a fit tight enough to reduce the standard deviation of the standardized residuals of the fit the RD survey, the lambda (likelihood weighting component) of the RD survey had to be increased by a factor of 10. Forcing the model to better fit the RD survey trend changed the overall biomass trend somewhat (Figure 135). It also caused a degradation in the fit to the composition data, and the conditional age at length composition data in particular. Biomass scale was unchanged and given the large weight being put on the survey data, the base model was preferred.

There is tension between the survey data and the composition data (Figure 131). The weights associated with each of these data sources determines the shape and scale of the model to some extent. A sensitivity run in which the variance associated with the composition data (both length, and age at length composition data) was increased relative to the base model, so that the harmonic mean of the effective sample size matched the mean of the input sample size (implicitly decreasing the information content of the RD survey). This was compared to the base model and the previous sensitivity run (Figure 135) in which the weight of the RD survey was increased (implicitly decreasing the information content of the composition data). The trade off between the composition data and the RD survey indices are clear in the comparison in that weighting the composition data more heavily tends to smooth out the biomass trajectory, while weighting the RD survey tends to introduce additional topography to the trend. All three runs show similar scales, while the base model is a compromise between the two in trend.

Profile analysis showed that the prior distribution associated with the MCD survey was influential in the base model solution. A sensitivity run in which the prior was not used confirmed this. The scale of estimated biomass shifted considerably, though the trend was very similar to the base run (Figure 136). The fits to the composition data were not affected by the removal of the prior distribution on catchability. When the prior distribution for the RD survey was removed, the effect on the model was almost undetectable, further indicating that the prior on the MCD survey is influential. When both prior distributions were removed, the model estimated a lower biomass (R_0 near the lower end of the range covered in the likelihood profile analysis), but the trend and fit to composition data were similar to the base model.

The MCD survey has only two data points in the base model, which is a small sample size to use for estimating trend. When the MCD survey index contribution to the likelihood was removed (multiplied by $\lambda = 0$), the scale of the estimated biomass shifted and trend was not strongly affected (Figure 137). This implies that the MCD survey and its prior are important for setting scale in the assessment model, but that they do not have a strong influence on the trend, even over the period that the MCD survey covers (>2011).

The steepness of the stock recruit relationship is assumed ($h = 0.95$) in the base model. There are no observations of the stock at low biomass in the time series (typical of a low F fishery) and so there is little information available with which to estimate steepness. A sensitivity run estimated steepness at 0.33 ($cv = 0.54$; Figure 138), but it is difficult to credit this estimate given the lack of information available to the model. The lower steepness value had little affect on the scale, trend or fit of the model, but would have an affect on biological reference points, if they were derived from the stock-recruit relationship. This aspect will be discussed further in 1.8.

The split survey in 2012 and 2013 caused some difficulty in compiling the data for the assessment model. In particular, the inclusion of 2013 data with the 2012 ages (Table 17) introduced additional observation error in the conditional age at length composition data. The error in the length composition data was expected to matter less because Atlantic surfclam grow relatively slowly, are fished lightly and the length composition from one year to the next should not change very much. A sensitivity run in which the conditional age at length information from 2013 was left out of the model was indistinguishable from the base run (Figure 139).

A comparison of the biomass time series estimated in several sensitivity runs demonstrated that the model varied in scale (Figure 140), but was relatively stable in trend (Figure 141). Allowing flexibility in the model by estimating more parameters, including time varying growth and natural mortality, produced runs that started and ended at similar biomass levels and had confidence intervals with a high degree of overlap. The run that used no prior information for estimating the catchability of the surveys had a different scale than the other runs, but showed a similar trend. In general, the model for the southern area was stable over many different configurations.

North

The assessment model for the north does not fit the survey well in the early part of the time series. One possible explanation for this is that the population was in a period of low recruitment and is currently in a period of high recruitment. A sensitivity run in which the parameter R_0 was estimated for each of two time periods (before 1995 and after) did estimate a lower R_0 for the early part of the time series, but did not substantively improve the fit to the survey index (Figure 142).

It was not possible to estimate recruitment variation (around the stock recruitment curve) in the model for the north in any of the runs tested. It is possible that the assumed value for recruitment variation was too low to provide the model enough flexibility to fit the early part of the RD survey well. A sensitivity run in which the recruitment variation was increased by 100% did not improve the fit to the survey (Figure 143).

Forcing the model to fit the RD survey better, by increasing its likelihood weight by a factor of ($\lambda = 10$), caused a degradation in the fit to the composition data (Figures 144 – 145). It was necessary to increase the weight on the survey by an order of magnitude before the model was able to fit the early RD index well (Figure 146).

The model was sensitive to the inclusion and relative weighting of the MCD survey. The MCD survey contributed only one year to the model. The MCD composition data were not particularly well fit by the model using an assumed selectivity, but it was not reasonable to allow the model to estimate selectivity given the single data point. When the variance associated with the MCD survey index was reduced (increasing its relative information content), the model produced a far more certain biomass estimate for the whole time series (Figure 147). While this result improved model diagnostics, it relied heavily on the information provided by a single data point and is therefore unstable. This is easily demonstrated by removing the MCD index from the likelihood calculation (making its contribution 0), which resulted in a model with a different biomass scale. The change in scale indicates that the entire model depends heavily on the catchability parameter for the MCD survey. The dependence on the prior for the MCD index in setting scale was clear from sensitivity runs in which the MCD index was included but the prior on catchability for it was not (Figure 148).

The inclusion of the 2014 conditional age at length composition with the 2013 data introduced additional observation error to the model. Removing the age data from 2014 would be expected to cause a bigger change in the model for the north, than the corresponding removal of 2013 data from the model for the south, because a higher proportion of the total data for the north came from the staggered year (Table 17). A sensitivity run in which the conditional age at length information from 2014 was left out of the model was similar to the base run (Figure 149), and the base run was preferred in order to make use of more of the available data and because the differences were minor relative to the uncertainty in the model.

A comparison of the biomass time series estimated in several sensitivity runs demonstrated that the model was relatively stable in trend (Figure 150), except when the trend was forced to fit the early part of the survey time series in the run called "WeightRD". Allowing flexibility in the model by including time varying recruitment, or increasing the variance around recruitment produced runs that started and ended at similar biomass levels and had confidence intervals with a high degree of overlap. Removing either the trend or the prior on catchability for the MCD survey tended to reduce the scale of the estimated biomass, though trends were still similar (Figure 151).

Internal retrospective

South

There is a shift in scale when the MCD survey drops out of the assessment (retrospective peels that do not include years after 2011; Figure 152). The Atlantic surfclam model for the southern area however, does not have a retrospective pattern in trend, which can be seen in a plot of the relative biomass from each retrospective run (Figure 153). Relative biomass was determined by dividing the biomass in each year and run by 25% of the virgin biomass estimated in that run.

North

The shift in scale in the model for the northern area is larger than in the southern area, and the trend is less stable over 10 peels of retrospective analysis (Figure 154 - 155).

Whole stock results

A simulation testing the relative merits of different approaches to combining F from multiple areas when absolute abundance was poorly determined, demonstrated that the abundance weighted average F was negatively biased when the correlation between abundance and F was close to -1 (See 9). The simulation also showed that the geometric mean of the F from each of two areas was close to the true combined F at all correlation levels. However, the geometric mean was strongly negatively biased when F is very low and in fact undefined when $F = 0$, which is true for a substantial proportion of the Northern area time series. The abundance weighted mean was therefore the preferred method of calculating combined F for the stock and determining the stock status relative to 2015.

Whole stock fishing mortality was $F_W = \frac{(C_S+C_N)}{(\widehat{N_S}+\widehat{N_N})}$ where C_S and C_N were the catch in numbers from each area and $\widehat{N_S}$ and $\widehat{N_N}$ were average fully selected abundances

$$\widehat{N_a} = \sum_L s_L \frac{N_L(1-e^{Z_L})}{Z_L}$$

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 part 1.9.

The F in projections was far enough from zero to allow the use of the geometric mean as a method for combining F from different areas. Therefore, the whole stock fishing mortality in projections was $F_W = e^{log(F_S)+log(F_N)}$. Whole stock projection results are discussed in 1.10. Fortunately, this choice had little effect on the whole stock results because F was so low. If F increases in the future it may be prudent to revisit the method for combining F from different areas in order to minimize the potential bias caused by correlation between F and abundance (9).

Whole stock spawning biomass estimates for clams was $SSB_W = e^{log(\frac{SSB_S}{SSB_{Threshold,S}})+log(\frac{SSB_N}{SSB_{Threshold,N}})}$, where $SSB_{Threshold,A} = \frac{SSB_{0,A}}{4}$ and A was area (either N or S). The variance around $\frac{SSB_A}{SSB_{Threshold,A}}$ was

$$\sigma^2_{SSB_A} = \left(\frac{\widehat{SSB}_A}{\widehat{SSB_{Threshold}}}\right)^2 \left(\frac{\sigma^2_{SSB_A}}{\widehat{SSB}_A^{\,2}} + \frac{\sigma^2_{Threshold,A}}{\widehat{SSB_{Threshold,A}}^{\,2}} - \left(\frac{2 * cov[SSB_A, SSB_{Threshold}]}{(\widehat{SSB}_A * \widehat{SSB_{Threshold}})}\right)\right)$$

Historical retrospective

The estimated whole stock biomass in this assessment is higher in scale than previous assessments (Figure 156). The scale shift over time reflects the difficulty in determining scale in the Atlantic surfclam assessment, progress as priors for catchability were developed, and is typical of a low F fishery.

1.8 TOR 6: Reference points

Current reference points

According to the harvest control rule in the FMP for Atlantic surfclam, overfishing occurred whenever the annual fishing mortality rate on the whole stock was larger than the overfishing limit (OFL), which was defined as a proxy for F_{MSY} ($F_{Threshold} = M = 0.15\,y^{-1}$). B_{Target} was defined as a proxy for B_{MSY} ($B_{Target} = \frac{1}{2} B_{1999}$ where B_{1999} was near the highest estimated biomass in previous assessments). The stock was overfished if total biomass fell below $B_{Threshold}$, which was $\frac{1}{2} B_{MSY}$ ($B_{Threshold} = \frac{1}{2} B_{MSY} = \frac{1}{4} B_{1999}$).

Current and recommended biological reference points (BRP) for Atlantic surfclam are proxies because spawner-recruit relationships required to determine F_{MSY} and B_{MSY} directly have not been estimated (low stock size has never been observed). Both current and recommended biomass reference points are based on trends/status ratios such as $\frac{B_{2015}}{B_{Threshold}}$ rather than absolute biomass estimates because the overall level of Atlantic surfclam biomass is uncertain. The current fishing mortality reference point is a fishing mortality rate but the recommended reference point is based on relative catch, again because of the uncertainty in biomass.

Reference points may be selected based on fishery performance and/or policy (risk aversion). Recommendations in this assessment are based on fishery performance criteria leaving MAFMC to consider policy to consider risk involved in setting catch targets, with the advice of its Scientific and Statistical Committee.

The $B_{MSY} = \frac{1}{2} B_{1999\,proxy}$ currently used for Atlantic surfclam has no theoretical justification beyond the notion that the biomass in 1999 was high at that time and might approximate carrying capacity. The major advantage was that both B_{1999} and biomass in the terminal year (e.g. B_{2015}) were estimated in the same model so that uncertainty in the overall scale of population size cancelled out in ratios used to determine stock status such as $\frac{B_{2015}}{\frac{1}{2} B_{1999}}$. In effect, the current approach is based on estimated trends in biomass but not on the absolute size of the estimates themselves. This property is important because sensitivity and historical retrospective analyses in this assessment show that estimated stock size trends are more robust for Atlantic surfclam than estimates of scale (Figures 147 - 156).

F_{MSY} and proxies depend on spawner-recruit, and yield/spawning biomass per-recruit relationships. Proxies for F_{MSY} are often set at some fraction of M ($F_{MSY} = cM$, $c < 1$ such that M is an upper bound for F_{MSY}) or at the fishing mortality rate corresponding to some fraction of maximum average reproductive output per recruit ($F_{SPR\%}$, Zhou et al. 2012). Existing $F_{SPR\%}$ proxies are not applicable to Atlantic surfclam because the analyses on which they are based generally assume that individuals mature and recruit to the fishery at about the same time. In addition, F_{MSY} cannot be computed directly because we have never observed a low stock size and thus have no way to characterize the stock recruit relationship. The current $F_{MSY\,proxy}$, $F = M = F_{Threshold}$ relies on biomass scale, and status determination relative to fishing mortality was therefore subject to the uncertainty associated with scale in the assessment.

Simulation analyses can be used to identify robust reference points that work well across a range of potential spawner-recruit curves and life-history patterns. This assessment includes management strategy evaluation (MSE) simulations which were tailored to Atlantic surfclam and the uncertainties about their life history and dynamics (8). The MSE analysis included two scenarios of particular interest. The primary scenario reflects current practice in managing two spatial areas (Northern and Southern) with different biological properties and independent recruitment patterns as a single unit. The secondary scenario uses separate harvest control rules for each unit and provides a means for assessing the potential costs and benefits of managing the two regions as a unit or separately.

MSE

MSE simulations were used to evaluate how MAFMC control rule parameters (a simplified version) affect average biomass relative to virgin biomass $\frac{SSB}{B_0}$[4], average relative yield measured as $\frac{Y}{B_0}$, interannual variation in yield cv(Y) and the proportion of years with no fishing ($t_{F=0}$). Simulations included a relatively wide and realistic range of random inputs for recruitment parameters, natural mortality, Beverton-Holt and Ricker spawner-recruit patterns, and other important, but uncertain parameters (8).

MSE results for combined region management and assuming both Beverton-Holt and Ricker recruitment patterns showed that $F_{Threshold}$ (F_{MSY} proxy) in the simulations, B_{Target} (B_{MSY} proxy) and $B_{Threshold}$ were all important for Atlantic surfclam in the MAFMC control rule (Figures 234 - 235). However, a wide range of different combinations of these parameters performed well based on MSE results. To simplify analysis we base recommendations on results for $F_{Threshold} < M = 0.15$ (an upper bound for F_{MSY}) and MAFMC control rule values of B_{MSY}=B_{Target}=$\frac{1}{2}B_0$, $B_{Threshold}$=$\frac{1}{4}B_0$.

For simulations at B_{Target}=$\frac{1}{2}B_0$, and considering combined area management, and with two spawner-recruit patterns, $F_{Threshold}$ values near 0.12 maximized yield while maintaining relatively high average spawning biomass with low interannual variation in yield and infrequent years with no fishing (Tables 41 - 44 and Figures 236 - 237).

Recommendations

$F_{MSY\,proxy} = 0.12$ is preferred over $F_{MSY proxy} = 0.15$ because higher levels of biomass, lower levels of variation in catch and less frequent years with no fishing would be expected according to the MSE (Appendix 8). $F_{MSY\,proxy} = 0.12$ is lower than the upper bound estimate $M = 0.15$, as should be expected. It is slightly larger than the range of $F_{MSY} = cM$ proxies for finfish with $0.63 < c < 0.74$ and $0.09 < F_{MSY} < 0.11$ (Zhou et al. (2012)). $F_{Threshold} = 0.12$, $B_{Threshold} =\frac{1}{4}B_0$ and $B_{Target} =\frac{1}{2}B_0$ provided high levels of catch and stock biomass at relatively low levels of variation in catch and years with no fishing. There is no reason to change the biomass reference points $B_{Target} =\frac{1}{2}B_0$ or $B_{Threshold} =\frac{1}{2}B_{Target}$ because they performed well in MSE simulations. These results were robust to assumptions about the underlying spawner recruit curve (Figures 236 - 237).

Based on the MSE analysis, mean stock biomass would increase by about one-third at $F_{Threshold} =$ 0.12, with no change in mean yield if Atlantic surfclam in both areas were managed separately

[4]Because Atlantic surfclam mature before age 1, there is no practical difference between B_0 and SSB_0 and the terms may be used interchangeably

although variance in catch and the number of years with no catch would increase (Figure 235). The simulations assume that all available yield is taken. Changes in average biomass, average yield, variance in catch, and years with no fishing would be smaller in the current fishery where catches are low relative to the levels calculated using the MAFMC control rule.

The recommendation $F_{Threshold} = 0.12$ is superior to $F_{Threshold} = 0.15$ on theoretical grounds but it shares an important implementation problem given that estimated fishing mortality rates are uncertain due to uncertainty in the scale of the biomass estimates. Thus, it would be very difficult to reliably compare an estimated fishing mortality rate to $F_{Threshold}$ and determine if overfishing is occurring. The assessment working group concluded it would be better to employ an $F_{Threshold}$ reference point based on trends using the average fishing mortality rate between 1982 and 2015 (the period for which we have survey data) in the southern area.

$$E_{y=1982}^{2015}[F_y] = F^*$$

The catch during that time period did not appear to result in overfishing. There is no evidence of overfishing in the current age/size compositions and current biomass estimates are near B_0 (see 1.7 and 14). The highest average fishing mortality between 1982 and 2015 for the southern area in sensitivity analyses was $F^*_{Max} = 0.03$. There is a high probability that $\frac{F_{MSY}}{F^*} > 4$ because

$$\frac{F_{MSY}}{F^*_{Max}} = \frac{0.12}{0.03} = 4$$

and F^*_{Max} was taken from the sensitivity run with the lowest biomass and thus highest F of any model run for the southern area. In addition, catch curve total mortality $(F + M)$ estimates for the southern area during this time period averaged 0.14, compared to the assumed M of 0.15. Empirical exploitation rates < 0.05, provding further evidence that F was low (14). Thus any F^* calculated from another model run would likely be lower than F^*_{Max}.

The recommended fishing mortality reference point is

$$F_{OFL} = F_{Threshold} = F^* \frac{F_{MSY}}{F^*_{Max}}$$

rather than a specific rate such as 0.12. It is important that F^* be calculated using the period between 1982 and 2015 in this, and in future assessments, as that was a period during which overfishing was very unlikely. Allowing the years that compose the reference point to shift over time would allow the reference point to normalize to current behavior. That is, the reference point would decrease during a regime of less fishing pressure and increase during a regime of more fishing pressure, which is not a desirable characteristic for a reference point.

There are three primary advantages to this recommendation. First, the status ratio used to identify overfishing

$$\frac{F_y}{F_{Threshold}} = \frac{F_y}{F^* \frac{F_{MSY}}{F^*_{Max}}}$$

provides information about relative exploitation rates that is not available in the ratio $\frac{F_y}{0.12}$ given the high degree of certainty in estimated trends and high degree of uncertainty in the scale of biomass estimates. Second, the recommended reference point is robust because it will adjust to changes in the scale of Atlantic surfclam biomass estimates, which can be expected in future assessments, at least over the short term. Finally, the scaling factor $\frac{F_{MSY}}{F^*_{Max}}$ can be re-examined and/or replaced as biomass estimates improve.

Table 4: Biological reference points used in the last assessment and the revised values used in the current assessment.

Reference point	Previous assessment	Revised
$F_{MSY} = F_{Theshold}$	$M = 0.15$	$F^* \frac{F_{MSY}}{F^*_{Max}}$
K	B_{1999}	B_0
$B_{MSY} = B_{Target}$	$\frac{B_{1999}}{2}$	$\frac{B_0}{2}$
$\frac{B_{MSY}}{2} = B_{Threshold}$	$\frac{B_{1999}}{4}$	$\frac{B_0}{4}$

1.9 TOR 7: Stock status

The assessment model was configured some what differently from the base model in the last assessment (Northeast Fisheries Science Center 2013), with the most important change being the addition of the new survey MCD survey. No new data from the RD survey has been collected since the previous assessment. It was not possible to add the new survey data to the previous assessment model because it was not configured to accept data from a different survey. Therefore, the previous assessment model cannot be directly compared to the model used in the current assessment, though a reasonable effort has been made to do so in (6). It is, however, possible to compare the current assessment estimates of biomass and fishing mortality to the current and recommended biological reference points.

Current reference points

Comparing the terminal biomass (B_{2015}) and fishing mortality estimates (F_{2015}) to the current reference points (Table 4) shows a low probability of either overfishing or overfished status for the Atlantic surfclam stock in the US EEZ (Table 27; Figure 157). The current $F_{threshold}$ was a point estimate with no associated uncertainty. Therefore the probability of overfishing was equal to the probability of overlap between the distribution of F_{2015} and the point estimate of $F_{threshold}$.

Recommended reference points

There is a near zero probability that the Atlantic surfclam stock in the US EEZ is experiencing overfishing ($F_{2015} < F_{Threshold}$; Table 28; Figure 158–159), and there is a low probability that the Atlantic surfclam stock in the US EEZ is overfished ($B_{2015} < B_{Threshold}$; Table 29; Figure 158 and 160). According to the recommended reference point definitions, the Atlantic surfclam stock is not overfished and overfishing is not occurring.

1.10 TOR 8: Projections

Basecase models were used to project biomass of Atlantic surfclam, catch (mt), and fully recruited fishing mortality in both areas, and in the combined stock during 2016-2025 (Tables 30 - 31 and Figure 161). Three harvest policies were assumed: 1) $F = F_{Threshold} = F_{OFL}$ (F at the OFL), 2) status quo catch (20333 mt) and 3) the maximum allowed catch under the current FMP or "quota level" catch (29364 mt) in the combined areas. Results indicate that biomass will remain higher than the biomass threshold and projected fishing mortality levels will be lower than the fishing mortality threshold for the entire resource.

Projection calculations were carried out in SS3 for the two areas using basecase models. Results for the whole stock were derived by combining projections for the northern and southern areas. Thus, the distribution of catches, relative growth rates, etc., were the same as in the terminal years of the base case models. Catches were landings multiplied by 1.12 to account for assumed 12% incidental mortality. Catches during 2016 were assumed the same as during 2015. For lack of better information, catches in the northern area during 2016-2025 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.

Projections for each year assumed time series average recruitment with uncertainty in starting stock size equal to the uncertainty in the final (non-forecast) model year (Figure 162). Projected total catch for the combined area was obtained by adding catches estimates for the southern and northern areas. Fishing mortality for the combined area (whole stock) was computed as the geometric mean (see Appendix 9) of the F from each area (calculated separately for each catch scenario). Overfishing status determination in each year (y) for the combined area was computed as $\frac{F_y}{F_{Threshold}} = \frac{F_y}{F^* \frac{F_{MSY}}{F^*_{Max}}}$ (see 1.8), where F^* was the mean F for the whole stock between 1982 and 2015 (Table 31). Whole stock spawning stock biomass was the sum of the spawning stock biomass from each area. These were considered unreliable due to scale uncertainty and are only included to document the calculation of projected catch at the OFL. Whole stock status ratios were the geometric mean of the status ratios from each area. Overfished status ratios were computed as $\frac{SSB_y}{SSB_{Threshold}} = \frac{SSB_y}{0.25 SSB_0}$.

It is unlikely 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. The distributions of SSB_y and $SSB_{Threshold}$ were assumed log normal with means equal to their respective point estimates and variances equal to their delta method variances. One million draws from 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 SSB_y and $SSB_{Threshold}$ estimated in the model. 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 (Shertzer et al. 2008). The probability of the whole stock being overfished was low for all projection scenarios considered (Figure 163).

The most likely fishing scenario is probably status quo catch, because the fishery is market limited and has been catching less than the quota since 2004 (Table 4). The quota scenario with higher catches was therefore a reasonable upper bound on likely fishing pressure over the next ten years. Using the quota scenario, the maximum probability of being overfished in any one year in next five

(P^*) was low (Figure 163) and the cumulative probability of being overfished at any time during the next ten years ($1 - \prod_y \{1 - p_y^*\}$) (Table 32), where p_y^* is the P^* value for each year was also low (see Shertzer et al. (2008)).

Projected fishing mortality levels are lower than the fishing mortality threshold for the entire resource under all scenarios except $F = F_{OFL}$ for each of the stock areas (Figure 164; Table 31). The cumulative probability of experiencing overfishing using the status quo catch or quota scenarios in any of the projection years was also low (Table 32).

In order to test the sensitivity of the projections to uncertainty in biomass scale, as well as model specification, quota scenario projections were conducted using the sensitivity runs with the lowest and highest biomass scale from 1.7 ("NoQPriors" and "EstimateM" for the southern area; see Figure 140). For the northern area the sensitivity runs with the lowest scale were the runs that excluded the MCD survey and the scale was too low to be creditable. Projection sensitivities for the northern area were run with the two models with the highest and lowest creditable scales ("HighRecrVariance" and "WeightRD"; see Figure 150). Projecting forward using the status quo catch scenario with these sensitivity runs showed that probabilities of overfishing and overfished status for the southern, northern and whole stock areas were similar in projection over a wide range of initial biomass scales (Table 33). The projection sensitivity results indicate that the status of the stock over the projected time horizon is robust to uncertainty in biomass scale, when recruitment remains near time series average values.

Probability distributions of the catch at the OFL were generated by repeated draws from a lognormal distribution of catch in each year, with a mean equal to the point estimate of the catch and a cv equal to the model estimated cv for each catch value (Figures 165 - 167; Table 34).

1.11 TOR 9: Stock definitions

Atlantic surfclam are assumed in the fishery management plan to be one unit stock throughout their range in US waters. The stock assessment workgroup discussed stock definitions of Atlantic surfclam at length during the last assessment (SAW 56) without reaching consensus. After reviewing all of the information presented, the SARC 56 review panel, "could not and did not choose to draw any conclusions as to whether a one- or two-stock definition was appropriate (SARC 56, 2013)." Ideas and arguments about Atlantic surfclam stock structure were summarized in two tables for SAW 56 which are also presented in this report (Figures 168 and 169).

The validity of the current stock definition was discussed by the working group briefly again in this assessment without reaching consensus. Opinions on this issue are strongly divided between industry-supported academic scientists and other members. As a result, the working group was unable to develop consensus recommendations as to whether there is a need to modify the current stock definition. Most of the stock definition discussion to date has focused on whether Georges Bank should be treated as a separate stock, both because it tends to be reproductively isolated due to persistent oceanographic conditions, and because it is unique based on the biological and fishery factors listed in Figure 168.

Below, the workgroup chair has summarized opinions of the assessment workgroup for purposes of addressing this TOR. Working group members agree that Atlantic surfclam consists of two or more meta-populations with different population dynamics, degrees of connectivity, fishery, exploitation, recruitment, post-settlement survival, growth rates, and shell height-meat weight patterns. However, some working group members view these differences as clinal and suggest that stock distinctions could be drawn in other places or not at all. They suggest that flexibility and lack of potential constraints on fishing activity are the most important benefits from the one stock approach. The multi-stock approach could lead to management constraints on the fishery that might not be necessary.

Other workgroup members noted that reference points like F_{MSY} and B_{MSY} are not well defined for heterogeneous stocks with independent population dynamics. Proxy reference points might not protect either population unit or maximize yield when used by the Council's SSC to set catch and landings limits intended to prevent Atlantic surfclam from being overfished, or overfishing from occurring. Stock conditions may suffer overall because problems in one area will be masked by conditions in the other. As shown in MSE analyses for Atlantic surfclam in this assessment (see 8) and in other studies, yield is reduced at F_{MSY} because productive areas in good condition may be fished too lightly while unproductive areas in poor condition may be fished too hard. These disadvantages are pronounced and likely to be important if fishing mortality rates approach or exceed F_{MSY}.

All members of the workgroup agree that stock definitions are unlikely to affect management, yield, or biological risk in the near term as long as fishing mortality rates remain low and overall abundance and biomass are relatively high.

The single stock assumption complicates and adds uncertainty to stock status determinations based on current and recommended reference points because biomass trend estimates for the whole stock are sensitive to independent errors in estimating scale for each area. Stock status conclusions in this assessment were robust to this problem because stock size was relatively high in both areas such that overfished status and overfishing were unlikely in either.

1.12 TOR 10: Research recommendations

The following are research recommendations from the previous assessment, in no particular order:

1. Determine the best spatial and temporal distribution to use for Atlantic surfclam assessment models.

   There have been no changes in stock definition, but the consensus of the assessment working group is that two areas modeled independently (northern area and southern area) with the results combined is the best configuration for stock assessment.

2. Biomass reference points need to be reconsidered.

   The SS3 model used for the assessment estimates B_0 for both southern and northern areas upon which biomass reference points can be based. See discussion of reference points in 1.8.

3. Has Atlantic surfclam biomass shifted offshore into deeper water over time?

   Sections 11 and 12 address this question analytically.

4. Look into a better way to implement regime change into the SS3 model. Look into patterns which may match other species and climate indices.

   Model sensitivity runs for the southern area were done with two possible growth stanzas. The model did estimate decreased growth in the second stanza, but the differences in outcome were negligible. See 1.7 for details.

5. Look at habitat on Georges Bank

   Section 13 lists methods explored in order to better determine the Atlantic surfclam habitat in the northern area that can be sampled effectively with a hydraulic clam dredge. These approaches will become available when stratification of the survey is reconsidered in the coming year. The working group agreed that the current approach was adequate for now.

New research recommendations, in no priority order:

1. Include Nantucket Shoals in the surveyed area for Atlantic surfclam.
2. Re-stratify northern area to make the survey more efficient and effective.
3. Examine coefficients used to convert commercial catches in bushels to meat weights.

1.13 Literature cited

References

Baranov, F. I. (1918). On the question of the biological basis of fisheries. *Nauchnyi Issledovatelskii Iktiologicheskii Institut, Izvestiia 1*(1), 71–128. 8

Beverton, R. J. and S. J. Holt (1957). *On the dynamics of exploited fish populations.* Chapman and Hall, London. 8

Chintala, M. M. (1997). Population biology of surfclams (*Spisula solidissima*) in inshore new jersey waters. Master's thesis, Rutgers University, New Brunswick, NJ. 109 p. 1.4

Chintala, M. M. and J. P. Grassle (1995). Early gametogenesis and spawning in juvenile atlantic surfclams, *Spisula solidissima* (dillwyn, 1819). *Journal of Shellfish Research 14*(2), 301–306. 1.4

Cochran, W. G. (1977). *Sampling techniques.* John Wiley & Sons. 1.4, 2

Deroba, J. and J. Bence (2008). A review of harvest policies: Understanding relative performance of control rules. *Fisheries Research 94*, 210–223. 8

Deroba, J. and J. Bence (2012). Evaluating harvest control rules for Lake Whitefish in the Great Lakes: accounting for variable life-history traits. *Fisheries Research 121-122*, 88–103. 8

Francis, R. C. (2011). Data weighting in statistical fisheries stock assessment models. *Canadian Journal of Fisheries and Aquatic Sciences 68*(6), 1124–1138. 1.7

He, X., M. Mangel, and A. MacCall (2006). A prior on steepness in stock-recruitment relationships, based on an evolutionary persistence principle. *Fisheries Bulletin 104*, 428–433. 8

Hennen, D. R., L. D. Jacobson, and J. Tang (2012). Accuracy of the patch model used to estimate density and capture efficiency in depletion experiments for sessile invertebrates and fish. *ICES Journal of Marine Science: Journal du Conseil 69*(2), 240–249. 3

Hennen, D. R., R. Mann, N. Charriere, and V. A. Nordahl (2016). Testing the performance of a hydraulic clam dredge modified to capture small animals. Technical Report NOAA Technical Memorandum NMFS-NE-237, National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026. 1.4, 1.4

Hilborn, R., C. J. Walters, et al. (1992). Quantitative fisheries stock assessment: choice, dynamics and uncertainty. *Reviews in Fish Biology and Fisheries 2*(2), 177–178. 1.3

McCullagh, P. and J. Nelder (1989). *Generalized Linear Models, 2nd Ed.* Boca Raton, FL: Chapman and Hall. 1.4, 8, 8

Methot, R. D. and C. R. Wetzel (2013). Stock synthesis: a biological and statistical framework for fish stock assessment and fishery management. *Fisheries Research 142*, 86–99. 1.7

Mid-Atlantic Fishery Management Council (2006). Overview of the surfclam and ocean quahog fisheries and quota considerations for 2007. Technical report, Mid-Atlantic Fishery Management Council, Dover, DE. 1.3

Munroe, D., D. Narvez, D. Hennen, L. Jacobsen, R. Mann, E. Hofmann, E. Powell, and J. Klinck (2016). Fishing and bottom water temperature as drivers of change in maximum shell length in atlantic surfclams (*Spisula solidissima*). *Estuarine, Coastal and Shelf Science 173*, 65–78. 1.6

Northeast Fisheries Science Center (2003). Report of the 37th northeast regional stock assessment workshop (37th saw). a. Atlantic Surfclams. Technical Report NEFSC Ref. Doc. 3-16, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026. 1.3, 1.3, 1.4

Northeast Fisheries Science Center (2007). Report of the 44th northeast regional stock assessment workshop (44th saw). a. Atlantic Surfclams. Technical Report NEFSC Ref. Doc. 7-10, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026. 1.3

Northeast Fisheries Science Center (2010). Report of the 49th northeast regional stock assessment workshop (49th saw). a. Atlantic Surfclams. Technical Report NEFSC Ref. Doc. 10-13, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026. 1.4, 8

Northeast Fisheries Science Center (2013). Report of the 56th northeast regional stock assessment workshop (56th saw). a. Atlantic Surfclams. Technical Report NEFSC Ref. Doc. 13-10, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026. 1.3, 1.4, 1.4, 1.7, 1.7, 1.7, 1.9, 168, 169, 2, 3, 3, 4, 6, 6, 8, 8, 236, 237, 14, 14

Northeast Fisheries Science Center (2014). Report of the 59th northeast regional stock assessment workshop (59th saw). Technical Report NEFSC Ref. Doc. 14-09, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026. 1.3

Ortiz, M. and F. Arocha (2004). Alternative error distribution models for standardization of catch rates of non-target species from pelagic longline fishery: billfish species in the venezuelan tuna longline fishery. *Fisheries Research 70*, 275–297. 8

Pinheiro, J. and D. Bates (2006). *Mixed-effects models in S and S-PLUS*. Springer Science & Business Media. 1.4

R Core Team (2013). *R: a language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. 8

Rago, P. J., J. R. Weinberg, and C. Weidman (2006). A spatial model to estimate gear efficiency and animal density from depletion experiments. *Canadian Journal of Fisheries and Aquatic Sciences 63*(10), 2377–2388. 3

Restrepo, V. and J. Powers (1999). Precautionary control rules in us fisheries management: specification and performance. *ICES Journal of Marine Science 56*, 846–852. 8

Ricker, W. E. (1954). Stock and recruitment. *Journal of the Fisheries Board of Canada 11*(5), 559–623. 8

Shertzer, K. W., M. H. Prager, and E. H. Williams (2008). A probability-based approach to setting annual catch levels. *Fishery Bulletin 106*(3), 225–232. 1.10

Venables, W. N. and C. M. Dichmont (2004). Glms, gams and glmms: an overview of theory for applications in fisheries research. *Fisheries research 70*(2), 319–337. 1.4

von Bertalanffy, L. (1938). A quantitative theory of organic growth (inquiries on growth laws. ii). *Human biology 10*(2), 181–213. 8

Walace, D. and T. Hoff (2005). Hydraulic clam dredge effects on benthic habitat off the northeastern united states. *American Fisheries Society Symposium 41*, 691–693. 1.3

Weinberg, J. R., T. G. Dahlgren, and K. M. Halanych (2002). *Influence of rising sea temperature on commercial bivalve species of the US Atlantic coast.* 1.4, 1.6

Weinberg, J. R., P. J. Rago, W. W. Wakefield, and C. Keith (2002). Estimation of tow distance and spatial heterogeneity using data from inclinometer sensors: an example using a clam survey dredge. *Fisheries Research 55*(1), 49–61. 10

Zhou, S., S. Yin, J. Thorson, A. Smith, and M. Fuller (2012). Linking fishing mortality reference points to life history traits: an empirical study. *Canadian Journal of Fisheries and Aquatic Sciences 69*, 1292–1301. 1.8