

B. ASSESSMENT FOR ATLANTIC SEA SCALLOPS *(Placopecten magellanicus)*

B1.0 CONTRIBUTORS

Invertebrate Subcommittee¹

B2.0 TERMS OF REFERENCE

1. Characterize the commercial catch, effort and CPUE, including descriptions of landings and discards of that species.
2. Estimate fishing mortality, spawning stock biomass, and total stock biomass for the current year and characterize the uncertainty of those estimates. If possible, also include estimates for earlier years.
3. Either update or redefine biological reference points (BRPs; proxies for B_{MSY} and F_{MSY}), as appropriate. Comment on the scientific adequacy of existing and redefined BRPs.
4. Evaluate current stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 3).
5. Recommend modeling approaches and data to use for conducting single and multi-year stock projections, and for computing TACs or TALs.
6. If possible,
 - a. provide numerical examples of short term projections (2-3 years) of biomass and fishing mortality rate, and characterize their uncertainty, under various TAC/F strategies and
 - b. compare projected stock status to existing rebuilding or recovery schedules, as appropriate.
7. Review, evaluate and report on the status of the SARC/Working Group Research Recommendations offered in recent SARC reviewed assessments.

¹ Meetings and members of the Invertebrate Subcommittee who helped prepare this assessment are listed in Appendix B1.

B3.0 EXECUTIVE SUMMARY

B3.1 TOR 1. Characterize the commercial catch, effort and CPUE, including descriptions of landings and discards of that species. (Completed – Section 4)

U.S. sea scallop landings averaged about 26,000 mt meats during 2002-2006, about twice their long-term average. Fishing effort reached its maximum in 1991 (at about 52,000 days absent), and then declined during the 1990s so that effort in 1999 was less than half that in 1991. Effort has been increasing in recent years, primarily due to increased landings and effort in the open access general category (day boat) sector. Landings per unit effort (LPUE) showed general declines from the mid-1960s through the mid-1990s, with brief occasional increases due to strong recruitment. LPUE more than quadrupled between 1998 and 2001, and remained high during 2001-2006. Discards of sea scallops was unusually high during 2001-2004, averaging about 10% of landings (by weight), but declined during 2005-2006, probably due to changes in gear regulations that reduced catches of small individuals. Sea scallops are occasionally caught and discarded in other fisheries such as the *Loligo* squid and summer flounder fisheries but the overall discards in other fisheries is small relative to total sea scallop landings.

B3.2 TOR 2. Estimate fishing mortality, spawning stock biomass, and total stock biomass for the current year and characterize the uncertainty of those estimates. If possible, also include estimates for earlier years. Completed (Section 5).

A dynamic size-based stock assessment model (CASA) was used as the primary model for sea scallops. This model was introduced in the previous benchmark sea scallop assessment but not used for estimation purposes due to its preliminary nature at that time. CASA was used in this assessment to estimate fishing mortality, (spawning) stock biomass and egg production.

Data used in CASA included commercial catch, LPUE, and commercial shell height compositions, the NMFS sea scallop and winter trawl surveys, the SMAST (School for Marine Sciences and Technology, University of Massachusetts, Dartmouth) small camera video survey, growth increment data from scallop shells, and shell height/meat weight data adjusted to take into account commercial practices and seasonality. Fishing mortality was also estimated using the rescaled F method employed in the last several assessments. The CASA and rescaled F methods gave similar results, especially for the most recent years.

The sea scallop stock was assessed in two components (Mid-Atlantic Bight and Georges Bank) separately and then combined. Estimates of fishing mortality were made from 1975-2006 in the Mid-Atlantic, and from 1982-2006 in Georges Bank and in the whole stock. Whole stock fishing mortality gradually increased during the 1980s, and peaked in 1992 at $F = 1.3$. Fishing mortality has generally declined afterwards, and the estimated fishing mortality $F = 0.23$ in 2006 was the lowest in the 1982-2006 time series.

Spawning stock biomass gradually increased from around 20,000 mt meats during 1982-1983 to a peak of 37,000 mt in 1990, and then declined to less than 17,000 mt meats by 1993. Biomass has been increasing since then, and the estimated 2006 biomass of 166,200 mt meats is the highest in the 1982-2006 time series.

Possible mild retrospective patterns were observed in the model in both regions, but not in the stock as a whole because the regional retrospectives were in different directions. CASA model estimates were reasonably precise: 95% confidence intervals for 2006 fishing mortality and spawning stock biomass were (0.17,0.32) and (152,182) thousands mt meats, respectively.

B3.3 TOR 3. Either update or redefine biological reference points (BRPs; proxies for B_{MSY} and F_{MSY}), as appropriate. Comment on the scientific adequacy of existing and redefined BRPs. Completed (Section 6).

The per recruit reference points F_{MAX} and biomass at F_{MAX} that are used as proxies for F_{MSY} and B_{MSY} were updated in this assessment based on new growth information and changes in fishery selectivity, using the CASA model. The new recommended fishing mortality threshold is 0.29, compared to the current reference point of 0.24. The new recommended biomass target is 108.6 thousand mt meats, and the recommended biomass threshold is half the biomass target, or 54.3 thousand mt meats. The current biomass reference points are a target of 5.6 kg/tow in the NEFSC sea scallop survey, adjusted for the assumed selectivity of the liner as in previous assessments, and a threshold of 2.8 kg/tow (adjusted).

The changes in fishery selectivity and new estimates of growth make updated yield per recruit curves flatter than previous curves so that F_{MAX} is more difficult estimate precisely and sensitive to assumption. In addition, the spatial variability in fishing mortality in the sea scallop fishery tends to cause per recruit reference points to overestimate the true (numbers-weighted) fishing mortality that maximizes yield per recruit. While this assessment recommends adoption of the new reference points, it also recommends that different types of biological reference points be considered for the next assessment.

B3.4 TOR 4. Evaluate current stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 3). Completed (Section 7).

The U.S. sea scallop stock is not overfished and overfishing is not occurring, under both the existing and proposed new BRPs, and using the new and previous method of estimating fishing mortality. Fishing mortality in 2006 was $F=0.23$ using the CASA model, and 0.20 using the rescaled F approach. Both of these figures are below the current overfishing threshold of 0.24, and the new proposed overfishing threshold of 0.29. Stock biomass was estimated in 2006 as 166.2 thousand mt, which is above the proposed biomass target of 108.6 thousand mt meats and the new biomass threshold of 54.3 thousand mt meats. Adjusted NEFSC survey biomass in 2006 was 7.3 kg/tow, above the current biomass target of 5.6 kg/tow, and the current biomass threshold of 2.8 kg/tow.

B3.5 TOR 5,6. Recommend modeling approaches and data to use for conducting single and multi-year stock projections, and for computing TACs or TALs.

If possible, provide numerical examples of short term projections (2-3 years) of biomass and fishing mortality rate, and characterize their uncertainty, under various TAC/F strategies and compare projected stock status to existing rebuilding or recovery schedules, as appropriate. Completed (Section 8)

The recommended projection model is spatially explicit and accommodates differences among regions in recruitment, growth, initial size structure, shell height/meat weight relationships, management approach (open vs. closed areas and catch quota vs. limits on fishing effort), intensity of fishing effort, and other factors. Two example short-term projections were conducted, both of which forecast modest increases in stock biomass and landings during 2007-2009. Sea scallop stock biomass is above its biomass target and not subject to a rebuilding or recovery plan.

B3.6 TOR 7. Review, evaluate and report on the status of the SARC/Working Group Research Recommendations offered in recent SARC reviewed assessments. Completed (section 9)

Collaborators made substantial progress on a number of important research recommendations since the last assessment. In particular, new growth and shell height/meat weight data and models were incorporated into the assessment, estimates of rock chain adjustment factors for survey data as well as dredge selectivity estimates were improved, the CASA stock assessment model was tested, improved and used to estimate fishing mortality and biomass for status-determination purposes, and results from collaborative research programs (i.e. video surveys and selectivity studies) were integrated into assessment calculations.

B4.0 INTRODUCTION AND LIFE HISTORY

The Atlantic sea scallop, *Placopecten magellanicus*, is a bivalve mollusk that occurs on the eastern North American continental shelf. Major aggregations in US waters occur in the Mid-Atlantic from Virginia to Long Island, on Georges Bank, in the Great South Channel, and in the Gulf of Maine (Hart and Rago 2006). In Georges Bank and the Mid-Atlantic, sea scallops are harvested primarily at depths of 30 to 100 m, while the bulk of landings from the Gulf of Maine are from near-shore relatively shallow waters (< 40 m). This assessment focuses on the two main portions of the sea scallop stock and fishery, Georges Bank in the north and the Mid-Atlantic in the south (Figure B3-1). Results for Georges Bank and the Mid-Atlantic are combined to evaluate the stock as a whole.

US landings during 2003-2006 exceeded 25,000 mt (meats) each year, roughly twice the long-term mean.² During 2005, US ex-vessel sea scallop revenues were over \$430 million, which was higher than for any other US fishery. Unusually strong recruitment in the Mid-Atlantic Bight area and increased yield-per-recruit due to effort reduction measures are the key reasons for high recent landings. The mean meat weight of a landed scallop in 2006 was over 25 g, compared to less than 14 g during the early to mid 1990s.

Area closures and reopenings have a strong influence on sea scallop population dynamics (Figure B3-1). Roughly one-half of the productive scallop grounds on Georges Bank and Nantucket Shoals were closed to both groundfish and scallop gear during most of the time since December 1994. Limited openings to allow scallop fishing in closed areas contributed more than half of Georges Bank landings during 1999-2000 and 2004-2006.

In the Mid-Atlantic, there have been four rotational scallop closures. Two areas (Hudson Canyon South and Virginia Beach) were closed in 1998 and then reopened in 2001. Although the small Virginia Beach closure was unsuccessful, scallop biomass built up in Hudson Canyon Closed Area while it was closed, and substantial landings were obtained from Hudson Canyon during 2001-2005. A third rotational closure, the Elephant Trunk area east of Delaware Bay, was closed in 2004, after extremely high densities of small scallops were observed by surveys during 2002 and 2003. The Elephant Trunk area reopened during March 2007 and preliminary reports indicate very high catch levels consistent with expectations and recent survey data. A fourth closed area (Delmarva), directly south of the Elephant Trunk area, was closed in 2007 and is scheduled to reopen in 2010.

² In this assessment, landings and biomass figures are metric tons (mt) of scallop meats, unless otherwise indicated.

B4.1 Assessment history

Early attempts to model sea scallop population dynamics (NEFSC 1992, 1995, 1997, 1999) were not useful because biomass estimates were less than the minimum swept area biomass obtained from the NEFSC scallop survey (NEFSC 1999). In lieu of model based estimates, fishing mortality in the most recent three assessments (SARC-29,32 and 39; NEFSC 1999, 2001, 2004) was estimated using a simple rescaled F method which relies heavily on survey and landings data (the rescaled F and other models were tested by simulation as part of this assessment, see Appendix B12). In the last assessment, a length-structured forward projecting model (CASA based on Sullivan et al. 1990 and Methot 2000) was introduced for preliminary evaluation. The CASA model was refined and tested and was used as the primary model for estimating fishing mortality, biomass and biological reference points for this assessment.

B4.2 Life History and Distribution

Sea scallops are found in the Northwest Atlantic Ocean from North Carolina to Newfoundland along the continental shelf, typically on sand and gravel bottoms (Hart and Chute 2004). Sea scallops feed by filtering phytoplankton, microzooplankton, and detritus particles. Sexes are separate and fertilization is external. Larvae are planktonic for 4-7 weeks before settling to the bottom. Scallops recruit to the NEFSC survey at 40 mm SH, and to the current commercial fishery at around 90-105 mm SH, although sea scallops between 70-90 mm were common in landings prior to the mid-1990s.³

According to Amendment 10 of the Atlantic Sea Scallop Fishery Management Plan, all sea scallops in the US EEZ belong to a single stock. However, the US sea scallop stock can be divided into Georges Bank, Mid-Atlantic, Southern New England, and Gulf of Maine regional components based on survey data, fishery patterns, and other information (NEFSC 2004, Figure B3-1).

B4.3 Age and growth

Sea scallop growth is traditionally modeled using the von Bertalanffy growth equation. Previous sea scallop assessments used the growth curves estimated by Serchuk et al. (1979), but reviewers expressed concern about lack of recent information on growth. As a result, a scallop growth study was carried out using shells collected during the 2001-2006 NEFSC scallop surveys (see Appendices B2 and 3). Growth curves based on new data have lower L_{∞} and higher K values than previous estimates (see table below and Figure B3-2). The growth parameter t_0 was not estimated and its value is not relevant to this assessment.

Growth parameters for sea scallops

Source	Region	L_{∞}	SE	K	SE
New	Mid-Atlantic Bight	131.6	0.4	0.495	0.004
	Georges Bank	146.5	0.3	0.375	0.002
Serchuk et al. (1979)	Mid-Atlantic Bight	151.84		0.2997	
	Georges Bank	152.46		0.3374	

³ Scallop body size is measured as shell height (SH, the maximum distance between the umbo and shell margin).

B4.4 Maturity and fecundity

Sexual maturity commences at age 2; sea scallops > 40 mm that are reliably detected in the surveys used in this assessment are all considered mature individuals. Although sea scallops reach sexual maturity at a relatively young age, individuals younger than 4 years may contribute little to total egg production (MacDonald and Thompson 1985; NEFSC 1993).

According to MacDonald and Thompson (1985) and McGarvey et al. (1992), annual fecundity (reproductive output, including maturity, spawning frequency, oocyte production, etc.) increases quickly with shell height in sea scallops ($Eggs=0.0000003396 SH^{4.07}$). Spawning generally occurs in late summer or early autumn. DuPaul et al. (1989) found evidence of spring, as well as autumn, spawning in the Mid-Atlantic Bight area. Almeida et al. (1994) and Dibacco et al. (1995) found evidence of limited winter-early spring spawning on Georges Bank.

B4.5 Shell height/meat weight relationships

Shell height-meat weight relationships allow conversion from numbers of scallops at a given size to equivalent meat weights. They are expressed in the form $W=\exp(\alpha+\beta \ln(L))$, where W is meat weight in grams and L is shell height in mm. NEFSC (2001) combined the shell height/meat weight relationships from Serchuk and Rak (1983) with relationships from NEFSC (1999; later published as Lai and Helser 2004) to obtain “blended” estimates that were used in the last two assessments (see table below).

Shell height/meat weight parameters

	α	β	γ
Mid-Atlantic Bight			
Haynes (1966)	-11.0851	3.0431	
Serchuk & Rak (1983)	-12.1628	3.2539	
NEFSC (2001)	-12.2484	3.2641	
Lai and Helser (2004)	-12.3405	3.2754	
New	-12.01	3.22	
New with depth effect	-9.18	3.18	-0.65
Georges Bank			
Haynes (1966)	-10.8421	2.9490	
Serchuk & Rak (1983)	-11.7656	3.1693	
NEFSC (2001)	-11.6038	3.1221	
Lai and Helser (2004)	-11.4403	3.0734	
New	-10.70	2.94	
New with depth effect	-8.62	2.95	-0.51

New shell height/meat weight data was collected during annual NEFSC sea scallop surveys during July of 2001-2006. Unlike previous studies, where meats were either frozen or brought in live and then weighed on land, meats were weighed at sea just after they were shucked (Appendix B4). Shell height/meat weight relationships based on new data give slightly higher predicted meat weights at a given shell height than NEFSC (2001), and nearly identical values at large shell heights (Figure B3-3).

Meat weights also depend on depth, with meat weights decreasing with depth, probably because of reduced food (phytoplankton) supply. Analysis of the new data indicated that depth had a significant effect on the intercept but not the slope of the shell height/meat weight

relationship. Estimated coefficients for the relationship $W = \exp(\alpha + \beta \ln(L) + \gamma \ln(D))$, where D is depth in meters, are given above (see Figure B3-4). In this assessment, depth-adjusted shell height/meat weight relationships were used to calculate survey biomass information, and traditional relationships were used in the models (CASA and SAMS), where depth is not explicit.

Meat weights for landed scallops may differ from those predicted based on research survey data for a number of reasons. First, the shell height/meat weight relationship varies seasonally, in part due to the reproductive cycle, so that meat weights collected during the NEFSC survey in July and August may differ from those in the rest of year. Additionally, commercial fishers concentrate on speed, and often leave some meat on the shell during shucking (Naidu 1987, Kirkley and DuPaul 1989). On the other hand, meats may gain weight due to water uptake during storage on ice (DuPaul et al. 1990). Finally, fishers may target areas with relatively large meat weight at shell height, and thus may increase commercial meat weights compared to that collected on the research vessel.

Observer and landings data were used to adjust survey shell height/meat relationships for use with the commercial catch. On select tows, observers measured the shell heights of about 100 scallops, and used a graduated cylinder to determine the total volume of the meats sampled after they were shucked in the normal manner by a crew member. Data collected at sea included the number of meats, sample weight, individual shell height measurements and the depth of the tow.

Volumetric measurements by observers were converted into meat weights assuming a conversion factor of 1.05 g/cc (Caddy and Radley-Walters 1972; Smolowitz et al. 1989). The observed average meat weight (b) for each observer sample was calculated as the sample weight divided by the number of meats in the sample. In the next step, the predicted average meat weight of the sample (p) was computed based on shell height/meat weight/depth relationships from survey data and observer shell height measurements and depth data. Anomalies (a) were computed as $a = (b - p)/p$ and averaged monthly for the Mid-Atlantic Bight and Georges Bank regions to estimate a monthly time series of meat weight anomalies (Figure B3-5). Gains in meat weight during storage on ice are highly variable and uncertain but for this assessment, meats were assumed to have gained by 3% to account for absorption of water during storage and transport when computing numbers of scallops landed (DuPaul et al. 1990).

Negative meat weight anomalies mean that fishery meat weights were less than predicted based on summer sea scallop survey relationships, and vice-versa. The mean anomaly during July in the Mid-Atlantic, and August on Georges Bank were slightly negative, probably due to loss of meat during commercial shucking. Both regions show a marked drop in meat weights between August and October, coinciding with the September-October spawning period, similar to the declines noted in Haynes (1966) and Serchuk and Smolowitz (1989).

Anomalies in the Mid-Atlantic were negative in all months, with the highest meat weight in July when the research vessel samples are taken. The monthly anomalies in Georges Bank were positive only in June and July. The estimated anomalies on Georges Bank for February through May are uncertain because they were based on a limited number of observed trips and samples.

Average monthly height/meat weight anomalies were averaged using the fraction of scallops landed during each month and year to calculate average annual shell height/meat weight anomalies for the commercial fishery, i.e. the dot-product between two vectors,

$$A_y = (L_{y1}, L_{y2}, \dots, L_{y12}) \cdot (a_1, a_2, \dots, a_{12})$$

where A_y represents the annual shell height/meat weight anomaly, L_{yk} is the fraction of the total (regional) landings in year y landed in month k , and a_k is the average shell height/meat weight anomaly in month k (Figure B3-6).

In computing numbers of sea scallops landed in the Georges Bank and Mid-Atlantic each year for this assessment, reported landings (mt meats) were divided by the average weight of individuals in the catch. The average weight of individual sea scallops in the catch was calculated based on size composition, shell-height meat relationship, annual anomaly, and adjustment for water absorption.

B4.6 Natural mortality estimates from survey “clapper” data

Following previous assessments, (e.g., NEFSC 2001, 2004), the natural mortality rate for sea scallops in this assessment was assumed to be $M = 0.1 \text{ y}^{-1}$ for scallops with shell heights > 40 mm. This estimate is based on Merrill and Posgay (1964), who estimated M based on ratios of clappers to live scallops in survey data. Clappers are shells from dead scallops that are still intact (i.e., both halves still connected by the hinge ligament). The basis of the estimate (Dickie 1955) is an assumed balance between the rate at which new clappers are produced ($M \cdot L$, where L is the number of live scallops) and the rate at which clappers separate ($S \cdot C$, where S is the rate at which shell ligaments degrade, and C is the number of clappers). At equilibrium, the rates of production and loss must be equal, so that $M \cdot L = S \cdot C$ and:

$$M = C / (L \cdot S).$$

Merrill and Posgay estimated $S = 1.58 \text{ y}^{-1}$ from the amount of fouling on the interior of clappers. The observed ratio C/L was about 0.066 and M was estimated to be about 0.1 y^{-1} . MacDonald and Thompson (1986) found a similar overall natural mortality rate, though they suggested that natural mortality increases at larger shell heights.

Clapper ratios were calculated for sea scallops in the Mid-Atlantic and on Georges Bank (Figure B3-7). Clapper ratios for both areas tend to be lower than observed by Merrill and Posgay (1964). It is unclear whether lower clapper ratios for recent years are because of lower natural mortality, differences in the clapper separation rate or changes in clapper catchability due to the change from an unlined to a lined dredge.

There have been recent increases in clapper ratios on Georges Bank. These may represent episodic mortality events, but also could be related to the increases in size/age in the Georges Bank stock. Larger size classes tend to have higher clapper ratios, but it is unclear whether this is due to increased separation time of larger clappers or to increased natural mortality as scallops age, or a combination of both (NEFSC 2004).

B5.0 COMMERCIAL AND RECREATIONAL CATCH (TOR 1)

The US sea scallop fishery is conducted mainly by about 350 vessels with limited access permits. However, landings have increased recently from vessels with open access general category permits, which tend to be smaller vessels that fish relatively near-shore beds. General category permits allow landings up to 400 lbs of scallop meats per trip or day (whichever is greater) without requiring a limited-access permit.

Principal ports in the sea scallop fishery are New Bedford, MA, Cape May, NJ, and Hampton Roads, VA. New Bedford style scallop dredges are the main gear type in all regions,

although some scallop vessels use otter trawls in the Mid-Atlantic (Table B4-1). Recreational catch is negligible; a small amount of catch in the Gulf of Maine may be due to recreational divers.

B5.1 Management history

The sea scallop fishery in the US EEZ is managed under the Atlantic Sea Scallop Fishery Management Plan (FMP), implemented on May 15, 1982. From 1982 to 1994, the primary management control was a minimum average meat weight requirement for landings. Figure B4-1 gives a timeline of all management measures implemented since 1984.

FMP Amendment 4 (NEFMC 1993), implemented in 1994, changed the management strategy from meat count regulation to effort control for the entire US EEZ. Effort controls were included that incrementally restricted days-at-sea (DAS), minimum ring size, and crew limits (Figure B4-1). To comply with legal requirements, Amendment 7 was implemented during 1998 with more stringent days-at-sea limitations and a mortality schedule intended to rebuild the stocks within ten years. Subsequent analyses considering effects of closed areas indicated that the stocks would rebuild with less severe effort reductions than called for in Amendment 7, and the Amendment 7 days-at-sea schedule was modified by Frameworks 12-15. Frameworks 11-13 permitted temporary access to the Georges Bank closed areas in 1999-2001, and Frameworks 14-16 provided for the controlled reopening of the Mid-Atlantic rotational closures.

A new set of regulations was implemented as Amendment 10 during 2004. This amendment formalized an area based management system, with provisions and criteria for new rotational closures, and separate allocations (in days-at-sea or TACs) for reopened closed areas and general open areas. Amendment 10 closed an area offshore of Delaware Bay (the Elephant Trunk area) where high numbers of small scallops were observed in the 2002 and 2003 surveys. This area reopened in 2007, when an area directly to the south was closed (Delmarva closure). Amendment 10 also increased the minimum ring size to 4" and, together with subsequent frameworks, allowed limited reopening of portions of the groundfish closed areas. Limited-access scallop vessels are restricted to a 7-man crew, which tends to limit the processing power of scallop vessels because regulations require most scallops to be shucked at sea.

B5.2 Landings

Landings from the Georges Bank and the Mid-Atlantic regions dominate the fishery. Proration of total commercial sea scallop landings into Georges Bank, Mid-Atlantic, Southern New England, and Gulf of Maine regions generally followed procedures in Wigley et al. (1998).

Sea scallop landings in the US increased substantially after the mid-1940s (Figure B4-2), with peaks occurring around 1960, 1978, 1990, and 2004. Maximum US landings were 29,109 mt meats during 2004. US Georges Bank landings had peaks during the early 1960's, around 1980 and 1990, but declined precipitously during 1993 and remained low through 1998 (Figure B4-3). Landings in Georges Bank during 1999-2004 were fairly steady, averaging almost 5000 mt annually, and then increased in 2005-2006, primarily due to reopening of portions of the groundfish closed areas to scallop fishing.

Until recently, the Mid-Atlantic landings were lower than those on Georges Bank. Mid-Atlantic landings during 1962-1982 averaged less than 1,800 mt per year. An upward trend in both recruitment and landings has been evident in the Mid-Atlantic since the mid-eighties. Landings peaked in 2004 at 24,494 mt before declining during 2005-2006.

Landings from other areas (Gulf of Maine and Southern New England) are minor in comparison (Table B4-1). Most of the Gulf of Maine stock is assessed and managed by the State of Maine because it is primarily in state waters. Gulf of Maine landings are generally a small percentage of the total. Gulf of Maine landings in 2006 were less than 1% of the total US sea scallop landings. Gulf of Maine landings average 475 mt meats during 1982-2006. Maximum landings in the Gulf of Maine were 1,614 mt during 1980. Southern New England landings averaged 116 mt meats during 1982-2006, with a maximum of 403 mt in 2005.

B5.3 Fishing effort and LPUE

Regulatory and reporting changes cause uncertainty in comparing trends in fishing effort and catch rates before and after 1994. Prior to 1994, landings and effort data were collected during port interviews by port agents and based on dealer data. Since 1994, commercial data are available as dealer reports (DR) and in vessel trip report (VTR) logbooks. DR data are total landings, and, since 1998, landings by market category. VTR data contain information about area fished, fishing effort, and retained catches of sea scallops. Ability to link DR and VTR reports in data processing is reduced by incomplete data reports and other problems, although there have been significant improvements since 1994 (Wigley et al. 1998). These problems make it difficult to precisely estimate catches and fishing effort, and to prorate catches and fishing effort among areas and gear types.

Landings per unit effort (LPUE) (Figure B4-4) shows a general downward trend from the beginning of the time series to around 1998, with occasional spikes upward probably due to strong recruitment events. LPUE increased considerably from 1999-2003 as the stock recovered; further increases in LPUE on Georges Bank were seen in 2005-2006, due primarily to the reopening of portions of the groundfish closed areas. Note the close correspondence in most years between the LPUE in the Mid-Atlantic and Georges Bank, probably reflecting the mobility of the fleet; if one area has higher catch rates, it is fished harder until the rates are equalized. Although comparisons of LPUE before and after the change in data collection procedures during 1994 need to be made cautiously, there is no clear break in the LPUE trend in 1994.

Nominal fishing effort (days absent) in the US sea scallop fishery generally increased from the mid-1960s to about 1990 (Figure B4-5a). Effort decreased during the 1990s, first because of low catch rates, and later as a result of effort reduction measures. Effort increased during 2000-2006, initially due to reactivation of latent effort among limited access vessels, and more recently due to large increases in the general category fishery.

However, LPUE in the limited access fishery has averaged about 1600 lbs/day in recent years, compared to the 400 lbs per day absent (by regulation) by a general category vessel. Thus, a day absent fishing by a general category vessel does not result in the same amount of mortality as a day absent by a limited access vessel. Adjusted days absent on trips with landings less than 500 lbs was therefore calculated as pounds landed from the trip divided by the mean LPUE of trips landing more than 500 lbs that year (Figure B4-5b). After this adjustment, the increase in effort is much more modest than what would appear based on the unadjusted data.

Another factor affecting the relationship between effort and mortality is the shucking capacity of a seven-man crew. During recent years, vessels have been able to catch scallops faster than they can be shucked. Thus, these vessels often stop actively fishing to allow the crew to shuck and process the catch before putting the gear back into the water. Data from observed (open area) trips indicates that the number of hours actually fished during a day absent dropped

from around 18 in the mid-1990s to 14 or less during the most recent years (Figure B4-6a). The number of hours fished during trips to formerly closed areas is considerably less (Figure B4-6b).

Spatial distribution of effort during 1998-2006 can be assessed using data from vessel monitoring systems (VMS) that are required on most sea scallop vessels (Figure B4-7). Average speed can be inferred as distance traveled between polling events (when location data are transmitted via satellite) divided by time elapsed. Vessels traveling between 1 and 5 knots were assumed to be actively fishing. Higher speeds likely indicate steaming, whereas speeds between 0-1 knots suggest that the vessel is probably processing the catch without fishing, as discussed above. Spatial distribution of fishing effort reflects limited openings of portions of the groundfish closed areas during 1999-2001 and 2004-2006, the rotational closure of the Hudson Canyon South and Virginia Beach areas from 1998-2001, and the Elephant Trunk closure between 2004-2007.

B5.4 Discards and discard mortality

Sea scallops are sometimes discarded on directed scallop trips because they are too small to be economically profitable to shuck, or because of high-grading during access area trips to previously closed areas. Ratios of discard to total catch (by weight) were recorded by sea samplers aboard commercial vessels since 1992, though sampling intensity on non-access area trips was low until 2003 (Figure B4-8, Table B4-2).

Discard to kept ratios during scallop fishing were variable. Higher discard ratios tend to be related to strong recruitment, which induce higher numbers of undersized scallops in commercial catches. Discard ratios were low during 2005-2006, probably due to new gear regulations (e.g., 4" rings) that went into effect at the end of 2004. Sea scallop discards in the sea scallop fishery were calculated as the discard to landings ratio for observed sea scallop trips times total sea scallop landings.

Sea scallops are also caught and either landed or discarded in fisheries that target finfish and other invertebrates. To estimate of the scallop bycatch in trawl fisheries for other target species, observer sea sample data from trawl trips targeting other species were used to calculate the ratio of pounds of scallops caught for every pound of the target species landed (observers ask the captain to declare the target species for each tow).

To estimate total sea scallop discard in other directed fisheries, discard to landed ratios were multiplied times total landings of target species from VTR records. The target species on a VTR record was the species with the most landings. This procedure may understate discards to some extent because VTR records may not include all landings.

The trawl fisheries with the largest bycatch of scallops for the years analyzed (1994-2006) were longfin squid, summer flounder, yellowtail, haddock, cod and monkfish. No data were available for the clam fisheries due to lack of observer coverage but hydraulic clam dredges used in the clam fishery have minimal bycatch of fish, sea scallops, and other invertebrates. Discards of scallops in other fisheries is negligible compared to landings. In total, an estimated mean of 94 mt meats of scallops were landed and 68 mt meats were discarded per year in 1994-2006 by the six fisheries targeting other species that were most likely to catch them (Table B4-3).

Discarded sea scallops may suffer mortality on deck due to crushing, high temperatures, or desiccation. There may also be mortality after they are thrown back into the water from physiological stress and shock, or from increased predation due to shock and inability to swim or shell damage (Veale et al. 2000, Jenkins and Brand 2001). Murawski and Serchuk (1989) estimated that about 90% of tagged scallops were still living several days after being tagged and

placed back in the water. Total discard mortality (including mortality on deck) is uncertain but has been estimated as 20% in previous assessments (NEFSC 2001, 2004). Though there is considerable uncertainty due to the limited data, an estimate of about 10% (on deck) + 10% (after release) = 20% total mortality of discarded sea scallops seems reasonable.

B5.5 Incidental Mortality

Scallop dredges likely kill and injure some scallops that are contacted but not caught, primarily due to damage (e.g., crushing) caused to the shells by the dredge. Caddy (1973) estimated that 15-20% of the scallops remaining in the track of a dredge were killed. Murawski and Serchuk (1989) estimated that less than 5% of the scallops remaining in the track of a dredge suffered non-landed mortality. Caddy's study was done in a relatively hard bottom area in Canada, while the Murawski and Serchuk study was in sandy bottom off the coast of New Jersey. It is possible that the difference in indirect mortality estimated in these two studies was due to different bottom types (Murawski and Serchuk 1989).

In order to use the above estimates to relate landed and non-landed fishing mortality in stock assessment calculations, it is necessary to know the efficiency e of the dredge (the probability that a fully recruited scallop in the path of a dredge is captured). Denote by c the fraction of scallops that suffer mortality among sea scallops in the path of the dredge but not caught. The best available information indicates that $c = 0.15-0.2$ (Caddy 1973), and $c < 0.05$ (Murawski and Serchuk 1989). The ratio R of scallops in the path of the dredge that were caught, to those killed but not caught is:

$$R = e/[c(1-e)]$$

If scallops suffer direct (i.e., landed) fishing mortality at rate F_L , then the rate of indirect (non-landed) fishing mortality will be (Hart 2003):

$$F_I = F_L / R = F_L c (1-e)/e.$$

If, for example, the commercial dredge efficiency e is 50%, then $F_I = F_L c$, where F_L is the fully recruited fishing mortality rate for sea scallops. Assuming $c = 0.15$ to 0.2 (Caddy 1973) gives $F_I = 0.15 F_L$ to $0.2 F_L$. With $c < 0.05$ (Murawski and Serchuk 1989) $F_I < 0.05 F_L$. For this assessment, incidental mortality was assumed to be $0.15 F_L$ in Georges Bank and $0.04 F_L$ in the Mid-Atlantic.

B5.6 Commercial shell height data

Since most sea scallops are shucked at sea, it has often been difficult to obtain reliable commercial size compositions. Port samples of shells brought in by fishers have been collected, but there are questions about whether the samples were representative of the landings and catch. Port samples taken during the meat count era often appear to be selected for their size rather than being randomly sampled, and the size composition of port samples from 1992-1994 differed considerably from those collected by sea samplers during this same period. For this reason, size compositions from port samples after 1984 when meat count regulations were in force are not used in this assessment.

Sea samplers have collected shell heights of kept scallops from commercial vessels since 1992, and discarded scallops since 1994. Although these data are likely more reliable than that from port sampling, sea sample data must be interpreted cautiously for years prior to 2003 (except for the access area fisheries) due to limited observer coverage. Shell heights from port and sea sampling data indicate that sea scallops between 70-90 mm often made up a considerable portion of the landings during 1975-1998, but sizes selected by the fishery have increased since then, so that scallops less than 90 mm were rarely taken during 2002-2006 (Figure B4-9).

Dealer data (landings) have been reported by market categories (under 10 meats per pound, 10-20 meats per pound, 20-30 meats per pound etc) since 1998 (Figure B4-10). These data also indicate a trend towards larger sea scallops in landings. While nearly half the landings in 1998 were in the smaller market categories (more than 30 meats per pound), nearly 80% of the 2006 landings were in the two largest market categories (10-20 count and under 10 count).

B5.7 Commercial gear selectivity

New gear regulations, requiring at least 4" rings on dredges with 10" twine tops, were implemented with Amendment 10 in 2004. They were required in the Hudson Canyon South Access Area in July 2004, in the groundfish closure access programs when these opened in November 2004, and in all areas since December 2004. A study was conducted to determine the selectivity of the new gear by towing a commercial dredge side by side with an NEFSC lined sea scallop survey dredge (Yochum 2006; Appendix B5). The new gear has a more gradual selectivity curve that is shifted to the right compared to the gear with 3.5" rings in use during 1996-2004 (Figure B4-11).

B5.8 Economic trends in the sea scallop fishery

Economic benefits from the sea scallop fishery have increased in recent years providing a larger supply of scallops for the consumers and higher revenue for the fishermen at lower costs. Landings from the northeast sea scallop fishery increased dramatically after 2001, surpassing all levels observed historically (Figure B4-12).⁴ Scallop ex-vessel revenue fell to its lowest recorded level of \$92 million during 1998 (Figure B4-13). Since 1998, revenue from scallops has increased steadily each year, exceeding \$440 million in 2005 and \$380 million in 2006.

Historical trends in the sea scallop fishery for three time periods are compared in the table below. The first period, from 1989 to 1992, summarizes the scallop fishery during a period when annual landings averaged above 16,000 mt and revenues averaged \$215 million. During the period from 1993 to 1998, overfishing in the previous years combined with the effort reduction measures and closure of the Georges Bank groundfish areas resulted in a dramatic decline in scallop landings and revenues. The period from 1999 to 2006 corresponds to the rebuilding of the sea scallop biomass and the consequent increase in scallop landings, revenues and exports to historical high levels. The average revenue per year for this period, over \$270 million, was more than double the average revenue of \$116 million per year during 1993-1998.

⁴ Although part of the increase in 2004 was due to some overfishing in the Mid-Atlantic, which is expected to decline in 2005, there is no question that increased scallop landings since 1999 were due primarily to increased scallop biomass.

Summary of economic trends in the scallop fishery (dollar values adjusted for inflation and expressed as 2006 prices)

Data - Annual averages	Period		
	1989-1992	1993-1998	1999-2006
Ex-vessel Price of scallops (\$ per lb.)	4.2	5.8	5.2
Scallop Revenue (\$ million)	215.0	115.9	270.6
Average meat count	37.7	36.5	21.7

There were some significant changes affecting scallop ex-vessel prices and revenues after 1999:

- In the past scallop prices increased when landings declined, and vice-versa. As Figure B4-12 shows, however, both landings and the ex-vessel price of scallops increased after 2001.
- The shifts in landings towards larger scallops that command a higher price was important factor increasing revenues after 1999 (Figure B4-10).
- Scallop revenues in 2005 and 2006 were more than three times higher than in 1994-98.

B6.0 FISHING MORTALITY AND STOCK BIOMASS (TOR #2)

NEFSC sea scallop survey data used in this assessment to estimate fishing mortality and biomass are from 1982-2006 for Georges Bank and 1975-2006 for the Mid-Atlantic. Sea scallop surveys were conducted by NEFSC in 1975 and annually after 1977 to measure abundance and size composition of sea scallops in the Georges Bank and Mid-Atlantic regions (Figure B5-1). The 1975-1978 surveys used a 3.08 m (10') unlined dredge. A 2.44 m (8') survey dredge with a 4.4 cm (1.75") plastic liner has been used consistently since 1979. The northern edge of Georges Bank was not surveyed until 1982, so survey data for this area are incomplete for this area during 1975-1981.

The *R/V Albatross IV* was used for all NEFSC scallop surveys except during 1990-1993, when the *R/V Oregon* was used instead. Surveys by the *R/V Albatross IV* during 1989 and 1999 were incomplete on Georges Bank. In 1989, the *R/V Oregon* and *R/V Chapman* were used to sample the South Channel and a section of the Southeast Part. Serchuk and Wigley (1989) found no significant differences in catch rates for the *R/V Albatross IV*, *R/V Oregon* and *R/V Chapman* based on a complete randomized block gear experiment (3 vessels x 13 stations=39 tows) in stratum 34.

The *F/V Tradition* was used to complete the 1999 survey on Georges Bank. The *F/V Tradition* towed the standard NMFS scallop survey dredge as well as a New Bedford commercial scallop dredge side by side. For the purposes of computing survey trends, only data from the NEFSC survey dredge was used. NEFSC (2001) found no statistically significant differences in catch rates between the two vessels from 21 comparison stations after adjustments were made for tow path. Therefore, as in previous assessments (e.g., NEFSC 2004), survey indices for the period 1990-93 based on data from the *R/V Oregon* were used without adjustment, and survey dredge tows from the *F/V Tradition* in 1999 were used after adjusting for tow distance.

Calculation of mean numbers of scallops per tow, mean meat weight per tow and variances in this assessment were standard calculations for stratified random surveys (Serchuk and Wigley 1989; Wigley and Serchuk 1996; Smith 1997) with some extensions described below.

B6.1 Imputed survey data

No valid tows were performed during some years for a few strata. In these cases, survey values for the missing strata were imputed using a generalized linear model (see Appendix B6). Results were very similar to the “borrowing” procedure used in previous assessments. Imputed data were supplied after all post-stratification was completed so that survey data (real or imputed) were available for every stratum assumed in calculations.

B6.2 Rock chains

Rock chains have been used on the NEFSC sea scallop survey dredge since 2004 in certain hard bottom strata to enhance safety at sea and increase reliability (NEFSC 2004). Preliminary analysis in the last assessment (NEFSC 2004) was augmented by additional gear experiments and statistical analysis (Appendix B9) to estimate rock chain effects on survey data. Results were difficult to interpret because rock chain effects appear to have varied from year to year. However, the best overall estimate was that rock chains increased survey catches on hard grounds by 1.31 times (CV 0.196).

To accommodate rock chain effects in hard bottom areas, survey data collected prior to 2004 from strata 49-52 were multiplied by 1.31 prior to calculating stratified random means for larger areas. Variance due to the rock chain adjustment was accommodated by calculating the variance of the adjusted strata means $\sigma^2 = 1.32^2 \sigma_n^2 + 0.257^2 n^2$ where n is the mean catch per tow for the stratum, σ_n^2 was the variance for mean catch per tow and $0.257=1.31*0.196$ was the standard error of the adjustment factor.

B6.3 Stratum areas and post-stratification

NEFSC shellfish survey stratum areas calculated using GIS by NEFSC (2001) were used in this assessment (Figure B5-1). Relatively high abundance of sea scallops in closed areas makes it necessary to post-stratify survey data by splitting NEFSC shellfish strata that cross open/closed area boundaries. After post-stratification, adjacent strata were grouped into regions corresponding to the various open and closed areas. Finally, in cases where the closed or open portion of an NEFSC survey stratum was very small, it was necessary to combine the small portion with an adjacent stratum to form a new slightly larger stratum (NEFSC 1999).

Rules for splitting strata along open/closed boundaries, assigning small portions to adjacent strata, and grouping strata into regions were the same as in NEFSC (1999) and Table B5-4 in NEFSC 2001), with a few refinements. The Closed Area II region was broken into two new regions by assigning the closed portions of survey strata 6621, 6610 and 6590 in Closed Area II to the new “Closed Area II (South)” region. All other portions of Closed Area II were assigned to the new “Closed Area II (North)” region. This allows the assessment to accommodate disparate population dynamics of the northern and southern areas of Closed Area II. The southern part of Closed Area II was heavily fished in 1999-2000. A very large (1998) year class was observed there during the 2000 and subsequent NMFS scallop surveys. By contrast, the northern portion of Closed Area II has not been fished since December, 1994.

NEFSC (2004) post-stratified the Nantucket Lightship Closed Area by defining a new stratum in the northeast corner of this area. Surveys show considerably higher recruitment and biomass in this area than elsewhere in the Nantucket Lightship area. Extra tows that have been

added to the northeast corner of the Nantucket Lightship Area in surveys during recent years are random with respect to the new stratification scheme and were used to increase the accuracy of abundance estimates.

B6.4 Survey dredge selectivity

Beginning in 1979, NEFSC sea scallop surveys used a 2.44-m (8-ft) wide dredge equipped with 5.1-cm (2-in) rings and a 3.8-cm (1.5 in) plastic mesh liner. Serchuk and Smolowitz (1980) compared catches from lined and unlined survey dredges, and found that the unlined dredge caught more large (>75 mm) while the lined dredge retained more small scallops. Other experiments comparing unlined commercial gear with a lined survey dredge found similar apparent reductions in catches of large scallops (NEFSC 2001, 2004; Yochum 2006). Based on these data, NEFSC (1995; 1997) assumed that the efficiency of the lined dredge was greater at small shell heights than at larger ones, and estimated a declining logistic selectivity curve with relatively low selectivity on scallops 60+ mm SH (Figure B5-2). In retrospect, the declining logistic shape of the estimated selectivity curve used in previous assessments was due to using shell height composition data from the unlined dredge in Serchuk and Smolowitz (1980) as a standard in estimating the selectivity of the lined dredge.

Shell height data from SMAST video surveys during 2003-2006 (Appendix B8) were used in this assessment as the standard in re-estimating survey dredge selectivity. The video survey data was particularly useful in this context because video cameras sample sea scallops 40+ mm SH (small camera) and 70+ mm SH (large camera, Appendix B7) with nearly full efficiency. Results (Appendix B8) indicate that the survey dredge has constant selectivity and efficiency for sea scallops 40+ mm SH, corresponding to the 38 mm mesh liner used in the survey dredge. For this reason, no adjustment was made to dredge survey shell height composition or abundance indices in this assessment to accommodate survey dredge selectivity.

The net effect of new assumptions about survey dredge selectivity is to reduce the absolute magnitude of survey abundance indices because the relative abundance of large sea scallops is not artificially increased. More importantly, the relative abundance of small scallops is higher in unadjusted dredge survey composition data. A number of analyses in this assessment are carried out using survey data with and without the selectivity adjustment to link results from new and previous methods. However, survey time series without selectivity adjustments are preferable on technical grounds.

B6.5 Non- and fully-recruited survey indices

Following NEFSC (2004), and for comparative purposes, unadjusted dredge survey data were partitioned into non-recruited (not vulnerable to commercial dredges) and fully recruited (completely vulnerable to commercial dredges) groups by applying a commercial 3.5" dredge selectivity function developed by consensus (NEFSC 1995):

$$s_h = \begin{cases} 0 & \text{if } h \leq h_{\min} \\ \frac{h - h_{\min}}{h_{\text{full}} - h_{\min}} & \text{if } h_{\min} < h < h_{\text{full}} \\ 1 & \text{if } h \geq h_{\text{full}} \end{cases}$$

where $h_{min} = 65$ mm and $h_{full} = 88$ mm. Note that fishery selectivity has changed over time, and the above curve approximates fishery selectivity during the mid- to late 1990s. Current fishery selectivity has shifted considerably towards larger scallops. However, non- and fully recruited abundance and biomass indices are useful in describing historical trends based on a familiar measure.

B6.6 Survey abundance and biomass trends

Biomass and abundance trends for the Mid-Atlantic Bight and Georges Bank are presented in Table B5-1 and Figure B5-3. Only random tows were used except in the post-stratified portion of the Nantucket Lightship Area (see above). Variances for strata with zero means were assumed to be zero. Confidence intervals were obtained by bootstrapping (Smith 1997, Figure B5-4).

In the Mid-Atlantic Bight, abundance and biomass were at low levels during 1975-1997, and then increased rapidly during 1998-2003, due to area closures, reduced fishing mortality, changes in fishery selectivity, and strong recruitment. Biomass was relatively stable during 2003 to 2006. In Georges Bank, biomass and abundance increased during 1995-2000 after implementation of closures and effort reduction measures. Abundance and biomass have been modestly declining during recent years, due to poor recruitment and to reopening of portions of the groundfish closed areas. Survey shell height frequencies show a trend to larger shell heights in both regions in recent years, coinciding with the period of increased biomass and abundance and recent recruitment levels (Figure B5-5).

Sea scallop biomass during 1994 (just before the Georges Bank closed areas and effort reduction measures were implemented), and during the most recent 2006 survey (Figure B5-6), shows considerable increases since 1994 in most areas. Increases are especially pronounced in the Georges Bank closed areas and the Elephant Trunk area that was closed during 2004-2007 after exceptional recruitment was observed there.

B6.7 SMAST video survey

Video survey data collected by the School for Marine Sciences and Technology (SMAST), University of Massachusetts, Dartmouth during May-September of 2003-2006 was used in this assessment. SMAST survey data are counts and shell height measurements from images that were recorded by two types of video camera. The “large” camera was mounted 1.575 m above the bottom in the center of the sampling frame with an effective sampling area of 3.235 m² of sea bottom. The “small” camera was mounted 0.7 m above the bottom with an effective sample area of 0.788 m². The effective sampling area includes the area within the sample frame plus an extra 75 mm around the edge of the frame to account for scallops on the edge of the frame. Data from the small camera were used to estimate the size selectivity of the NEFSC scallop dredge (Appendix B8), the large camera (Appendix B7) and as an input to the CASA model. All calculations assume that the small camera has 100% sampling efficiency and flat selectivity for sea scallops 40+ mm SH. Selectivity of the large camera is >90% for scallops 70+ mm SH (Appendix B7).

The SMAST survey is based on a systematic sampling pattern with stations centered on a 5.6 x 5.6 km grid pattern (Stokesbury et al. 2004). Four “quadrats” are sampled at each station and one image taken with each camera is analyzed from each quadrat. The sampling frame and cameras are placed on the bottom at the center of the grid where video footage from the first quadrat is collected. The sampling frame is then raised until the sea floor is no longer visible and

the ship is allowed to drift approximately 50 m in the current before the sampling frame is lowered and video footage from the second quadrat image is collected. The third and fourth images are collected in the same manner. All scallops with any portion of their shell lying within the sample area are counted. Measurements are taken from images projected on a digitizing tablet from all specimens where the umbo and shell margins are clearly visible.

The precision of measurements must be considered in interpreting video shell height data. Based on work in progress (K. Stokesbury, SMAST, pers. comm.) and NEFSC (2004), video shell height measurements from the large camera have a standard deviation of 6.1 mm across a wide range of sea scallop shell heights (see NEFSC 2004, Appendix 1). The standard deviation of measurements from small camera images is assumed in this assessment to be 6.1 mm also for lack of better information.

Video survey data (Tables B5-2 and B5-3) in this assessment are expressed as densities (number m⁻²). Variances for estimated densities are approximated from the variance among station means in each year. Areas sampled in the video survey differ somewhat from the areas sampled in the dredge survey (Figure B5-7). There was some variability in the areas covered during each year (Figure B5-7 and Tables B5-2 and B5-3).

B6.8 Simple biomass and fishing mortality estimates

The NEFSC survey can be used to obtain an estimate of absolute biomass provided dredge efficiency can be estimated. Commercial dredge efficiency has been estimated at 0.4 – 0.55 in Georges Bank and 0.57 in the Mid-Atlantic (NEFSC 1999, 2001; Gedamke et al. 2004, 2005). Based on the data discussed above, a liner reduces the efficiency of the survey dredge by a factor of about 0.715. Thus, these commercial dredge efficiencies translate into survey dredge efficiencies of about 0.29-0.36 in Georges Bank and 0.41 in the Mid-Atlantic. Comparison of abundances between the NEFSC dredge and SMAST video survey suggests that survey dredge efficiency is about 0.38 on Georges Bank and 0.43 in the Mid-Atlantic (Appendix B8). Based on these figures, the survey swept area biomasses and abundances were calculated using an estimated survey dredge efficiency of 0.36 on Georges Bank and 0.42 in the Mid-Atlantic, and using an estimated mean tow path of 4516 m² (NEFSC 2004), using the formula

$$B_y^* = \frac{b_y A}{a e}$$

where b_y is mean meat weight per tow from the survey in year y , B_y^* is survey stock biomass, a is the area (nm²) swept by a standard tow, e is efficiency, and A is the size (nm²) of the stock area or region.

Fishing mortality rates cF_y (biomass-weighted) can then be estimated as:

$${}^cF_y = \frac{C_y}{B_y^*}$$

where C_y is the meat weight of scallops killed by fishing during the calendar year (Ricker 1975). The survey is conducted during July-August, approximating the average annual biomass. However, C_y represents reported landings only, and the mortality estimate will be biased low if there were non-reported landings, or if there was non-yield fishing mortality. Additionally, these

estimates are biomass-based mortality rates, which tend to be biased low compared to numbers-based mortality rates, particularly when there is spatial heterogeneity in fishing mortality (Hart 2001). Because of these issues and uncertainty in the estimates of dredge efficiency, this simple fishing mortality estimator is used only as an indicator of fishing mortality trends (NEFSC 1999, 2001, 2004).

B6.9 Survey-based (two-bin) method

The survey-based approach divides the survey data for each year into two shell height size bins. The first bin approximates the size range of new recruits to the fishery. The second bin includes sea scallops of all larger sizes.

Based on updated growth information, the first bin for Georges Bank consisted of scallops of 80-100.8 mm shell height and the second bin consisted of all scallops larger than 100.8 mm. An 80 mm sea scallop was almost fully recruited to the fishery (except during the most recent period) and will grow to 100.8 mm in one year, according to growth increments from collected shells. For the Mid-Atlantic region, the first bin consisted of 80-98 mm scallops and the second bin consisted of scallops larger than 98 mm. Using these data, survey-based fishing mortalities were calculated:

$${}^s F_t = -\ln\left(\frac{P_{t+1}}{R_t + P_t}\right) - M,$$

where R_t was the mean population number of scallops per standard survey tow in the first bin (new recruits) during survey year t , and P_t was the mean number of scallops per standard survey tow in the second bin. Survey years are the annual period between NEFSC sea scallop surveys (summer to summer).

B6.10 Rescaled catch-biomass method

Rescaled catch-biomass estimates were used during the last three assessments as the primary estimator of fishing mortality rates (NEFSC 1999, 2001, 2004; Hart 2006). Rescaled survey-based estimates were computed as:

$${}^r F_y = {}^c F_y \left(\frac{{}^s \bar{F}}{{}^c \bar{F}} \right)$$

where average catch-biomass ${}^c \bar{F}$ and survey-based ${}^s \bar{F}$ fishing mortality rates were for a time period of many years that contains year y . This estimator is based on the idea that the catch-biomass estimate tracks the trend in fishing mortality accurately, while the appropriate overall scale is given by mean survey fishing mortality rates. The rescaled F gives a smoother trend than the survey fishing mortalities, and, unlike the simple catch/biomass method, is numbers based and does not require assumptions about dredge efficiency and incidental mortality. For this assessment, survey and landings data from 1979-2006 for sea scallops in the Mid-Atlantic and 1982-2006 for sea scallops on Georges Bank were used to estimate the ratio of ${}^c \bar{F}$ and ${}^s \bar{F}$.

As in NEFSC (2004), coefficients of variation (CVs) for rescaled fishing mortality estimates were approximated considering variability in the survey data (measured by CVs for random stratified means), and landings data (assumed CV of 10%).

B6.11 Whole-stock rescaled F estimates

Because of differences in e.g., growth rates, between Georges Bank and the Mid-Atlantic, fishing mortalities were calculated separately for the two areas. Whole-stock estimates of fishing mortality are required, however, for comparison to biological reference points used to identify overfishing and overfished stock conditions.

Whole stock estimates were calculated by averaging estimates for Georges Bank and the Mid-Atlantic using the area surveyed in the NEFSC dredge survey in each region as weights. A variety of evidence indicates that dredge efficiency on Georges Bank is lower than in the Mid-Atlantic, so swept-area abundances in the Mid-Atlantic were multiplied by 0.875 before averaging (0.875 is approximately the ratio of survey dredge efficiencies between the two areas, see Appendix B8). Results for the whole stock were only very slightly sensitive to the assumed value of this factor.

Survey-based and rescaled F estimates both show generally increasing fishing mortality until the early 1990s, with reductions during 1994-2006 (Table B5-4, Figure B5-8).

B6.12 Model-based fishing mortality and biomass estimates

CASA model estimates are the best scientific information about sea scallop population dynamics available in this assessment (a complete technical description of the CASA model is in Appendix B10). A CASA model for sea scallops was presented for preliminary review in the last stock assessment (NEFSC 2004) and received positive comments. Simulation testing described in this assessment indicates generally good model performance. Base case model estimates for Georges Bank and the Mid-Atlantic Bight use all of the available data and appear reasonable in comparison to estimates from the rescaled F model used previously (see below). Sensitivity analyses (see below) suggest that base case estimates for sea scallops are reasonably robust. CASA models in this assessment are used to estimate fishing mortality, biomass and biological reference points based on the same assumptions and using the same computer code, ensuring that the fishing mortality and biomass measures are comparable to biological reference points. CASA model estimates appear relatively precise.

B6.13 Whole stock biomass, abundance and mortality

Biomass, egg production, abundance, recruitment and fishable mean abundance were estimated for the whole stock by adding estimates for the Mid-Atlantic Bight and Georges Bank. Whole stock fishing mortality rates for each year were calculated $F = (C_M + C_G) / (\bar{N}_M + \bar{N}_G)$ where C_M and C_G are catch numbers for the Mid-Atlantic Bight and Georges Bank. Terms in the denominator are average fishable abundances during each year calculated in the original CASA model $\bar{N} = \sum_L \frac{N_L(1 - e^{-Z_L})}{Z_L}$ with the mortality rate for each size group (L) adjusted for fishery selectivity. The simple ratio formula used to calculate whole stock F is an “exact” solution because the catch equation $C = F\bar{N}$.

Asymptotic delta method variances calculated in CASA with AD-Model Builder software were used to compute variances and coefficients of variation (CV) for whole stock estimates assuming that estimation errors for Georges Bank and the Mid-Atlantic Bight were independent. In particular, variances for biomass, abundance and catch estimates were the sum of the variances for Georges Bank and the Mid-Atlantic Bight. CVs for the ratios estimating whole stock F were approximated $CV_F = \sqrt{CV_C^2 + CV_{\bar{N}}^2}$, which is exact if catch number C_N and average abundance \bar{N} are independent (Deming 1960). The CV for measurement errors in catch for each region was 0.05, the same as assumed in fitting the CASA model.

Whole stock estimates indicate that annual abundance, annual egg production and biomass (Table B5-5 and Figures B5-9 to B5-11) were relatively high during 2006. In contrast, recruitment was relatively low during 2006 (Table B5-5 and Figure B5-12). Fishing mortality during 2006 (Table B5-5) was similar to rescaled F fishing mortality estimates used in the last assessment (Figure B5-13). CV values indicate that abundance, biomass and fishing mortality estimates were relatively precise for individual regions and for the stock as a whole (Table B5-6 and Figure B5-14). The relatively small CVs but likelihood profiles and MCMC probability intervals (not shown) confirmed the asymptotic variances for recent biomass and fishing mortality.

The apparent precision of the estimates for sea scallops may be surprising and the CVs calculated in this assessment certainly do not capture all of the underlying uncertainties. However, estimates were relatively precise because of the long time series of relatively precise dredge survey data (CVs averaging 23% for Georges Bank during 1982-2006 and 12% for the Mid-Atlantic Bight during 1979-2006) and recent video survey data (overall CVs averaging 14% during 2003-2006). The assumption of flat selectivity curves for the two surveys substantially enhances precision, as does the prior information about sampling efficiency in the video survey.

B6.14 Retrospective patterns

CASA model runs for Georges Bank and the Mid-Atlantic show possible retrospective patterns that cancel out when estimates for the two areas were combined (Figure B5-15). The possible retrospective tendencies may be due to anomalously high dredge survey abundance for Georges Bank in 2000 and anomalously high dredge and video survey abundances for Mid-Atlantic Bight during 2003. Bootstrapped survey estimates show unusually high variances for survey data during these years (Figure B5-3). When areas are combined, effects of unusual survey data and possible uncertainties in allocating landings between the two areas are diminished. The closure of the Elephant Trunk area during 2004-2006, and closures and reopenings on Georges Bank may be partially responsible for the retrospective patterns. Preliminary model runs that included spring and fall bottom trawl survey data for the Mid-Atlantic Bight (not shown) showed no evidence of retrospective patterns.

B6.15 CASA models for the Mid-Atlantic Bight and Georges Bank

CASA models for the Mid-Atlantic Bight and Georges Bank were configured as described in Table B5-7. Estimated parameters and asymptotic standard deviations are given in Tables B5-8 and B5-9. Diagnostics indicate that base case models for both areas fit reasonably well in most cases (Figures B5-16 to B5-19).

There was a noticeable lack of fit to commercial shell height composition data for 1975-1980 in the Mid-Atlantic Bight because shell height composition data from the 10 ft unlined

dredge survey for 1975, and 1977-1978 showed a different pattern with higher frequencies of large scallops (Figure B5-19). In retrospect, the commercial fishery during the late 1970s would have been better modeled with a separate dome-shaped fishery selectivity pattern with low selectivity on the largest scallops which were probably outside of traditional fishing grounds. However, sensitivity analysis showed that estimates were almost unchanged when data 1975-1978 were omitted (see below). Commercial shell height composition data during the late 1970s probably had little effect because the data were down-weighted using low effective sample sizes in goodness of fit calculations. Sea scallop population dynamics during years prior to 1979 and the advent of the modern sea scallop dredge survey is an important topic for future research.

B6.15.1 Likelihood profile analysis

Likelihood profile analysis indicates that base case CASA models for sea scallops on Georges Bank and in the Mid-Atlantic Bight struck a reasonable balance between different sources of information and key data sources generally supported similar estimates of recent fishing mortality and biomass. Likelihood profiles are useful because they identify the statistical support among various data sources for a range of recent biomass and fishing mortality estimates (Tables B5-10 and B5-11). Profiles were constructed by holding the survey scaling parameter (catchability coefficient) for the SMAST video small camera survey fixed at a series of values while estimating all other parameters in the model. The scaling parameter for the SMAST video survey was ideal for this purpose because it would be expected to have values near 0.5 and because this parameter has a direct impact on recent biomass and fishing mortality estimates. At each point in the likelihood profile, estimated 2006 biomass and fishing mortality and “naked” (unweighted) likelihood were recorded for each type of data and constraint.

In interpreting likelihood profiles, it is useful to know that a difference of 1.92 likelihood points is often used to identify differences that are statistically significant at the $p=0.05$ level. The 1.92 rule of thumb is approximate and based on asymptotic arguments.

The total likelihood for the base case Georges Bank model had a well defined minimum around the base case solution (Table B5-10). The trend in the dredge survey, which is the most important source of trend information, and short trend in video survey data fit best near the base case solution. Commercial landings and LPUE data and the constraint on recruitment support higher 2006 biomass levels, although the likelihoods for commercial catch and LPUE were relatively flat. The likelihood for the prior on efficiency of the SMAST video survey was lowest at 0.5 (as expected) supporting a higher 2006 biomass estimate. All three types of shell height composition data support lower 2006 biomass estimates but the likelihoods for shell height composition data were relatively flat.

The total likelihood for the base case Mid-Atlantic Bight model had a well defined minimum around the base case solution (Table B5-11). The trend in the dredge survey, which is the most important source of trend information in the model, and short trend in video survey data fit best near the base case solution. In contrast, the winter bottom trawl survey fit best at lower 2006 biomass levels and the short trend in unlined 10 ft scallop dredge survey data fit best at higher 2006 biomass levels, although the likelihood surface for both was relatively flat. Fall and spring bottom trawl survey data (which did not affect model estimates) support lower 2006 biomass estimates. Commercial landings and LPUE data and the constraint on recruitment deviations fit best at lower 2006 biomass levels although the likelihood surface for catch and LPUE was relatively flat. The likelihood for the prior on efficiency of the SMAST video survey was lowest at 0.5 (as expected) supporting a higher 2006 biomass estimate. Commercial and

survey shell height composition data, with the exception of the unlined 10 ft scallop dredge survey, support higher biomass 2006 estimates although likelihood surfaces were relatively flat for the dredge and winter bottom trawl shell height composition data.

B6.15.2 Sensitivity analysis

Several alternative model runs were carried out with CASA models for the Mid-Atlantic Bight to identify uncertainties and affects of modeling decisions. Mid-Atlantic Bight models were used for sensitivity analysis because of the similarity in structure between models for the two areas and because more types of information were available for the Mid-Atlantic Bight.

Results indicate that biomass estimates for sea scallops in the Mid-Atlantic Bight region were robust to uncertainties and modeling decisions (Table B5-12 and Figure B5-20). The only sensitivity analysis run with substantially different recent biomass and fishing mortality estimates was one that included fall and spring bottom trawl trend and shell height composition in fitting the model. As described under profile analysis, the fall and spring trend data support lower biomass estimates than the base case model.

B7.0 BIOLOGICAL REFERENCE POINTS (TOR 3)

Biological reference points, fishing mortality rates and biomass estimates used in status determination here are for the entire US sea scallop stock. Because of the lack of well-defined stock-recruitment relationships for sea scallops, per recruit reference points F_{MAX} and B_{MAX} are used by managers as proxies for F_{MSY} and B_{MSY} . F_{MAX} is the fishing mortality rate for fully recruited scallops that generates maximum yield-per-recruit. B_{MAX} for sea scallops is the product of BPR_{MAX} (biomass per recruit at $F = F_{MAX}$, from yield-per-recruit analysis) and median numbers of recruits.

The current biological reference points are $F_{MAX} = 0.24$ and $B_{MAX} = 5.6$ kg/tow (in survey units, adjusted for the survey dredge liner as in previous assessments NEFSC 2001, 2004). The current F_{MAX} reference point was originally calculated by Applegate et al. (1998) using an age-based (Thompson-Bell) yield per recruit calculation. NEFSC (2004) found a similar value for F_{MAX} using a size-based yield per recruit calculation, and left this reference point unchanged. The current value of B_{MAX} was calculated in NEFMC (2003) as a product of BPR_{MAX} (from the per recruit calculations in NEFSC 2001) with median survey recruitment from 1979-2002 (Mid-Atlantic) and 1982-2002 (Georges Bank).

The CASA model was used to recalculate per recruit curves for Georges Bank and the Mid-Atlantic Bight assuming the selectivity patterns during 2006, growth increment data, etc. Yield and biomass per recruit curves for the two regions were fairly similar (Figure B6-1), although growth patterns are different and fishery selectivity curves for the two areas during 2006 were offset by about 10 mm (Figures B3-2 and B6-1).

Per recruit curves for the two areas were combined to approximate a per recruit model for the whole stock. The goal was to estimate curves that would have been calculated if the two regions had been modeled together. Whole stock yield- and biomass per recruit curves (Figure B6-1) were calculated by averaging yield per recruit curves for the two regions using median recruitment during 1983-2006 (the longest period with recruit estimates for both areas) as weights. F_{MAX} (F_{MSY} proxy) and B_{MAX} (B_{MSY} proxy, 40+ mm SH on January 1) are from the whole stock per recruit curves (Table B6-1). As in previous sea scallop assessments (NEFSC 2004), the B_{MSY} target reference point for the whole stock was estimated as the product of

biomass per recruit at F_{MAX} and median recruitment for the whole stock during 1983-2006 (Table B6-1).

The per recruit reference points F_{MAX} and B_{MAX} are reasonable proxies for F_{MSY} and B_{MSY} provided that recruitment is independent of stock size or has reached its asymptotic value at B_{MAX} , and if fishing mortality as well as other parameters do not vary over space. There was no compelling evidence of a spawner-recruit relationship for either area that would tend to undermine F_{MAX} as an F_{MSY} proxy. As in previous assessments, the biomass threshold was $B_{MSY}/2$.

However, there are special considerations for sedentary organisms such as sea scallops where fishing mortality is not uniform and particularly when closed areas are present. In such a case, mean yield-per-recruit, averaged over all recruits, may be different than yield-per-recruit obtained by a conventional per-recruit calculation performed on a recruit that suffers the mean fishing mortality risk (Hart 2001). This condition is exaggerated, as in the case of the scallop fishery, with use of rotational or long-term closures. Recent research indicates that the (numbers- or biomass- weighted) fishing mortality that achieves maximum or optimal yield may be less than that indicated by a conventional yield-per-recruit analysis when there is spatial variability in fishing mortality (Hart 2001, 2003).

B7.1 Examination of possible stock-recruit relationships

This section was added at the request of the SARC panel. Sea scallop recruitment and egg production for the Mid-Atlantic and Georges Bank were estimated using the CASA model (Figure B6-2, Table B5-5). Mid-Atlantic sea scallop egg production remained low from 1975-1997, but increased about 10-fold from 1997-2006. Sea scallop recruitment was poor from 1975-1981 and was moderately strong but variable from 1982-1995. The 1996-2001 year classes were all very strong; all but the 2000 year class was larger than any year class during 1975-1995. Recruitment was below average in 2002 and 2004, but was strong in 2003. The plot of recruitment vs. egg production (Figure B6-3a) suggests the possibility that the increased egg production was at least partially responsible for the strong recent recruitment. However, the period of strong recruitment started before any increase in egg production, so that autocorrelated environmental factors may also explain the increase in recruitment. A fit of a Beverton-Holt curve to the data, assuming log-normal errors, suggests the possibility that recruitment overfishing was occurring prior to 1999, when egg production was less than 20 quintillion. This fit ignores any import of larvae from Nantucket Shoals and Georges Bank, which might be significant but is not quantifiable at this time. Assuming the proposed target biomass (108.6 thousand mt meats) was equally split between Mid-Atlantic and Georges Bank, this reference point corresponds to about 60 quintillion eggs. Egg production at the target biomass appears to be sufficient to saturate the stock-recruit relationship, so that there is little concern of recruitment overfishing if biomass remains at or over the proposed target. It also appears that the biomass threshold, corresponding to about 30 quintillion eggs, is a reasonable point to take action to prevent possible recruitment overfishing.

Georges Bank egg production was relatively low from 1982-1995, but has increased substantially since then (Figure B6-2b). Recruitment appears fairly trendless, with strong recruitment during the late 1980s, and a very strong 1998 year class. Except for the 2001 year class, recruitment during 1999-2004 has been below average. A plot of recruitment vs. egg production (Figure B6-3b) gives no indication that the recent increase in egg production has led to an increase in recruitment. A fit of these data to a Beverton-Holt stock-recruit curve, assuming log-normal errors, suggests that the stock-recruit curve is already saturated at 20 quintillion eggs,

about the number of eggs released in 1996-1997, so that the subsequent increases in egg production had little effect. However, this analysis neglects the egg production in the Canadian side of Georges Bank which in many years may have contributed 30-50% of the total egg production. Additionally, there are no observations below the estimated half-saturation point of the stock-recruit curve, so that the half-saturation point cannot be well estimated. However, again it can be concluded that there is little concern regarding recruitment overfishing if biomass is at or over its target (corresponding to about 60 quintillion eggs on Georges Bank) or even at the biomass threshold (corresponding to about 30 quintillion eggs).

B8.0 STATUS DETERMINATION (TOR 4)

According to the Amendment 10 overfishing definition (NEFMC 2003), sea scallops are overfished when the survey biomass index for the whole stock falls below $1/2 B_{MAX}$. Overfishing occurs if fishing mortality exceeds F_{MAX} . As described above, managers use F_{MAX} from yield-per-recruit analysis and B_{MAX} as proxies for F_{MSY} and B_{MSY} . F_{MAX} is the fishing mortality rate for fully recruited scallops that generates maximum yield-per-recruit (see above). The current target biomass level B_{TARGET} was calculated as the median recruitment in the survey time series times BPR_{MAX} , the biomass per recruit obtained when fishing at F_{MAX} . The current management reference points are $F_{MAX} = 0.24 \text{ y}^{-1}$ and $B_{TARGET} = 5.6 \text{ kg/tow}$ (adjusted for the liner as in previous assessments).

Overfishing was not occurring in the sea scallop stock and overfishing was not occurring during 2006, based on the reference points currently in use and the fishing mortality estimator used in previous assessments (NEFSC 2001, 2004). Based on the 2006 NEFSC scallop survey, sea scallop biomass (adjusted for assumed dredge selectivity) was about 7.3 kg/tow, well above $B_{MAX} = 5.6 \text{ kg/tow}$. The overall rescaled F fishing mortality estimate for the whole stock 2006 was 0.20 (rescaled F), which is below the overfishing threshold of $F_{MAX} = 0.24$.

Based on the new recommended reference points and CASA model estimates, the US sea scallop stock was not overfished and overfishing was not occurring in 2006. This assessment proposes the new reference points of $F_{MAX} = 0.29$, a target biomass reference point of 108.6 thousand mt meats, and a biomass threshold of 54.3 thousand mt meats. The best estimate for fully recruited fishing mortality during 2006 is $F = 0.23$ (95% confidence interval 0.17-0.32, Figure B7-1), which is well below the proposed threshold $F_{MAX} = 0.29$. Based on the variance in estimated fishing mortality, there is only a 7% chance that fishing mortality was above the recommended fishing mortality threshold during 2006. Estimated stock biomass for sea scallops during 2006 was 166 thousand mt (95% confidence interval: 152-182 thousand mt, Figure B7-2). Based on the variance in estimated biomass, there is less than a 0.1% probability that the sea scallop biomass was below the biomass threshold of 54.3 mt meats.

B9.0 STOCK PROJECTIONS (TOR 5-6)

Example stock projections were made for two assumed scenarios. Under the first scenario, $F=0.20$ (the current target) during 2007-2009. The second set of projects assumes $F=0.24$ (the current F_{MSY} proxy and fishing mortality threshold, and a potential new target) during 2007-2009.

Because of the sedentary nature of sea scallops, fishing mortality of sea scallops can vary considerably in space even in the absence of area specific management (Hart 2001). Area management such as rotational and long-term closures can make variation even more extreme

(Figure B4-7). Projections that ignore such variation might be unrealistic and misleading. For example, suppose 80% of the stock biomass is in areas closed to fishing (as occurred in some years in Georges Bank). A stock projection that ignored the closure and assumed a whole-stock F of 0.2 would forecast landings nearly equal to the entire stock biomass of the areas remaining open to fishing. Thus, using a non-spatial forecasting model can lead to setting a level of landings that appears sustainable if all areas were fished uniformly, but is in fact unsustainable for a given area management policy.

For this reason, a spatial forecasting model (the Scallop Area Management Simulator, SAMS) was developed for use in sea scallop management. Various versions of SAMS have been used since 1999 (NEFSC 2004). Growth is modeled in SAMS and CASA in a similar manner, except that each area of Georges Bank and the Mid-Atlantic in SAMS has its own stochastic growth transition matrix derived from the shell increments collected in that area. Mortality and recruitment are also area-specific. Fishing mortality can either be explicitly specified in each area, calculated using a simple fleet dynamics model which assumes fishing effort is proportional to fishable biomass, or a combination of the two. Shell height/meat weight relationships were from the 2001-2006 R/V data, adjusted using the mean annual fishery shell height/meat weight anomaly during 1997-2006 (see Appendix B4 and Figure B3-6).

Projected recruitment is modeled stochastically with the log-transformed mean and covariance for recruitment in each area matching that observed in NEFSC dredge survey time series. Initial conditions were based on the 2006 NEFSC sea scallop survey with uncertainty measured by bootstrapping as described by Smith (1997). Survey dredge efficiencies were set in SAMS so that the mean 2006 biomass matched estimates from the CASA model. Further details regarding the SAMS model are given in Appendix B11.

For these simulations, the stock area was split into 15 subareas, six in the Mid-Atlantic (Virginia Beach, Delmarva, Elephant Trunk, Hudson Canyon South, New York Bight, and Long Island) and nine on Georges Bank (Closed Area I, II and Nantucket Lightship EFH closures, Closed Area I, II and Nantucket Lightship access areas, Great South Channel, Northern Edge and Peak, and Southeast Part). The Delmarva area was closed on a rotational basis in 2007, and is assumed to be fished at 0.2 for the first year (since the simulation starts in July 2006), and then closed during the remainder of the simulation.

The Elephant Trunk area was reopened in 2007 after a three year closure, and scheduled to remain a special access area with its own TAC and target fishing mortality for the three years of the simulation. It is subject to an increasing pattern of fishing mortality during the three year simulations (0.16, 0.24, 0.32 in the first set of simulations; 0.16, 0.29, 0.38 in the second set of simulations). The Hudson Canyon South area was closed in 1998-2001 and 2007 is the last year of its special access program with estimated fishing mortality of 0.4. It is scheduled to be a part of the fully open areas in 2008-9.

The EFH closure portions of the three groundfish closed areas (Closed Area I, II and Nantucket Lightship Closed Area) are closed long-term to all bottom-tending mobile gear, and are assumed closed during the entire simulation period. Two out of three of the access portions of the groundfish closed areas are opened each year: Closed Area I and Nantucket Lightship in 2007, Closed Area II and Nantucket Lightship in 2008, and Closed Areas I and II in 2009.

Target total allowable catch (TAC) levels have already been set for the 2007 groundfish access area program (NEFMC 2005, about 2500 mt in each area). Fishing mortality in these areas in 2008-9 was assumed to be 0.2, as specified in sea scallop Amendment 10 (NEFMC 2003). All other areas (Virginia Beach, New York Bight, Long Island, South Channel, Northern

Edge and Peak, Southeast Part, and after 2007, Hudson Canyon South) are part of the open area pool.

In projections, fishing effort was allocated to areas so that the overall fishing mortality rate was 0.24 in the first year (based on current regulations described in NEFMC 2005) and 0.2 during 2008-2009 (first set of simulations) or 0.24 (second set of simulations). Fishing effort was distributed among the open areas according to a simple fleet dynamics model, where fishing mortality in each area was assumed to be proportional to fishable biomass.

Under both scenarios, biomass and landings are expected to increase modestly in the next three years (Figure B8-1,2). Under the first scenario ($F = 0.20$), landings are expected to rise from a little more than 26,000 mt meats in 2006-2007, to over 32,000 mt in 2008-2009, compared to a range of 26,000 mt in 2006-2007 to over 34,000 mt in 2008-2009 in the $F = 0.24$ scenario. On the other hand, biomass is projected to increase more during 2006-2009 in the $F = 0.20$ scenario (22%) than in the $F = 0.24$ simulation (15%). Roughly 40% of the landings are projected to come from the special access areas (Elephant Trunk and the groundfish closed areas). None of the 400 model runs resulted in a biomass below the new biomass target (108.6 thousand mt) indicating that overfished stock conditions are unlikely in the near future.

Simulated landings are more variable than biomass, because the landings stream is more dependent on the abundances of a few key areas (such as the Elephant Trunk) while total biomass includes sea scallops in closed areas and areas lightly fished. Much of the variation among the simulation runs for each scenario was due to bootstrapping of survey data to set initial conditions (rather than variable recruitment) because simulated recruits did not have time during the short simulations to grow and completely recruit to the fishery.

B10.0 RESEARCH RECOMMENDATIONS (TOR 7)

Agencies, academic institutions, and contractors made considerable progress in key areas of scallop research since the last assessment. In this section, progress on recommendations in the previous assessment (NEFSC 2004) is reviewed and new research recommendations are presented.

B10.1 Research recommendations from NEFSC (2004)

1. More comparison tows between standard survey dredges and those equipped with rock chains are necessary to more precisely estimate the correction factor(s) needed to convert between survey tows with and without rock chains. *Additional field work and statistical analysis has been completed although more research would be required to precisely estimate rock chain effects, which may vary from year to year (see Appendix B9).*
2. Explore potential for surveying hard bottom areas not currently covered using survey dredges equipped with rock chains. Some experimental paired tows have been carried out on the (hard-bottom) northern edge of Georges Bank, where rocks are occasionally seen. This topic is under discussion and progress is expected when the current NEFSC sea scallop survey is replaced by a proposed optical-dredge survey after 2008.
3. Explore the use of VMS and landings data to characterize condition of the resource on grounds not covered by the survey. *Some work is underway to interpret catch rates on unsurveyed grounds using VMS and other data. Grounds covered by NEFSC surveys may be expanded after 2008.*

4. Further work is required to better characterize the selectivity of the commercial dredges with 4 inch rings relative to the standard NEFSC survey dredge. *A comprehensive paired-tow field study to estimate contact selectivity of commercial dredges with 4 inch rings was completed (see Appendix B5 and Yochum 2006). In addition, CASA model estimates for 2006 provide useful estimates of fishery selectivity that integrate the effects of contact selectivity, discard and targeting.*
5. Because assumptions about growth are important in almost any stock assessment model, better estimation of scallop growth, including variability in growth, is important in improving the precision of sea scallop stock assessments. *Appendices B2-B3 describe new growth data and growth parameter estimates. Variation among regions is accommodated and variability over time is noted.*
6. Work presented during the assessment indicates substantial variability in shell height-meat weight relationships due to depth, season, year and possibly area. Additional work on this subject may be useful, especially with respect to area-based management. *See Appendix B4 and Section 3 of this report for new data, depth based shell height/meat weight relationships, and approaches to calculating shell height-meat weight in the commercial fishery.*
7. Based on recent work on scallops in the US and Canada, there is a potential for tracking year-to-year variability in natural mortality based on clapper data. Use of clapper data in stock assessment models to estimate natural mortality should be investigated. *Work on this topic is underway but has not been completed.*
8. The statistical properties of the new “CASA” model should be fully evaluated prior to the next meeting. The properties of concern include performance in the face of process errors (e.g. variability in natural mortality and growth), measurement errors in data, and characterization of uncertainty. In addition, use of smaller time steps and shell height groups might be helpful. It may prove possible to apply the model or similar models to smaller geographic areas. *Appendices B10-B12 describe progress along these lines and software used to test the sea scallop stock assessment model.*
9. There appears to be considerable scope for reducing variability in scallop survey data by changing the allocation of tows to survey strata. *A more adaptive allocation scheme has been adopted, which has resulted in lower variance in the most recent surveys (Table B5-1).*
10. Comparison of SMAST video survey with the NEFSC survey has proved valuable in estimating efficiency of survey and commercial dredges and in improving abundance estimates. The benefits of future video surveys could be enhanced by increasing coordination in carrying out the video and NEFSC surveys on the same grounds, so that the NEFSC scallop strata are fully covered by the video survey. More intense video surveys in small areas, such as was done in 1999-2002, can help reduce the variances of the efficiency estimates. *SMAST video survey data were fully incorporated in this assessment. Cooperative analyses were carried using video and dredge survey data to characterize selectivity in both surveys and to refine estimates of dredge efficiency for sea scallops in the Mid-Atlantic and on Georges Bank (see Appendices B7-B8). A paired photographic/dredge comparison study is planned for this summer.*
11. This assessment demonstrates the potential for fully incorporating results of cooperative surveys in stock assessment models for scallops. Areas where additional information could be obtained by cooperative research include abundance in areas not normally surveyed by NEFSC, gear properties, and temporal and spatial variation in shell height/meat weight relationships, mortality, recruitment and growth. *Results of a 4 inch ring selectivity study*

conducted by the Virginia Institute of Marine Science (VIMS, Appendix B5, Yochum 2006) and SMAST video surveys (Section 5, Appendices B7-B8) were incorporated in this assessment.

B10.2 New Research Recommendations

1. Refine estimates of natural mortality focusing on variation among regions, size groups and over time. Abundance trends in closed areas where no fishing occurs may provide important information about the overall level of natural mortality and time trends. Survey clapper catches may provide information about spatial, temporal and size related patterns in natural mortality.
2. Evaluate the within and between reader error rates in identification and measurement of growth increments on scallop shells.
3. Improve estimates of incidental and discard mortality rates.
4. Consider using autocorrelated recruitment in SAMS projection model runs. CASA model estimates indicate that sea scallop recruitment may be autocorrelated.
5. Consider modeling the spatial dynamics of the fishing fleet in the SAMS projection model based on catch rates, rather than exploitable abundance, of scallops in each area.
6. Evaluate assumptions about the spatial dynamics of the fishing fleet in the SAMS model by comparing predicted distributions to VMS data.
7. Investigate the feasibility and benefits of using information about the size composition of sea scallops in predicting the spatial distribution of the fishing fleet in the SAMS projection model.
8. Evaluate the accuracy of the SAMS projection model retrospectively by comparison to historical survey abundance trends.
9. Consider implementing discard mortality calculations in the CASA model that are more detailed and involve discarded shell height composition data from at sea observers.
10. Consider implementing a two or more “morph” formulation in the CASA model to accommodate scallops that grow at different rates.
11. Consider approaches to implementing seasonal growth patterns in the CASA model to improve fit to shell height composition data. Scallops grow quickly at small sizes and growth rates vary by season.

B11.0 REFERENCES (including references cited in scallop appendices)

- Almeida F, Sheehan T, Smolowitz R. 1994. Atlantic sea scallop, *Placopecten magellanicus*, maturation on Georges Bank during 1993. NEFSC Ref. Doc. 94-13.
- Applegate A, Cadrin S, Hoenig J, Moore C, Murawski S, Pikitch E. 1998. Evaluation of existing overfishing definitions and recommendations for new overfishing definitions to comply with the Sustainable Fisheries Act. Final Report, June 17, 1998. New England Fisheries Management Council, Saugus, MA.
- Beverton RJH, Holt SJ. 1956. A review of methods for estimating mortality rates in fish populations, with special reference to sources of bias in catch sampling. Rapp P-v Reun Cons Int Explor Mer 140:67-83.
- Butler JL, Jacobson LD, Barnes JT, Moser HG. 2003. Biology and population dynamics of cowcod (*Sebastes levis*) in the southern California Bight. Fish Bull. 101: 260-280.

- Caddy JF. 1973. Underwater observations on tracks of dredges and trawls and some effects of dredging on a scallop ground. *J Fish Res Bd Can.* 30:173-180.
- Caddy JF, Radley-Walters C. 1972. Estimating count per pound of scallop meats by volumetric measurement. *Fish Res Brd Can Man Rep Ser.* 1202.
- Deming WE. 1960. *Sample design in business research.* New York: Wiley & Sons Inc; 517 p.
- Dibacco C, Robert G, Grant J. 1995. Reproductive cycle of the sea scallop, *Placopecten magellanicus* (Gmelin, 1791), on northeastern Georges Bank. *J Shellfish Res.* 14:59-69.
- Dickie LM. 1955. Fluctuations in abundance of the giant scallop, *Placopecten magellanicus* (Gmelin), in the Digby Area of the Bay of Fundy. *J Fish Res Bd Can.* 12:797-857.
- DuPaul WD, Fisher RA, Kirkley JE. 1990. An evaluation of at-sea handling practices: Effects on sea scallop meat quality, volume and integrity. *Contr rep to Gulf and S Atlant Fisheries.*
- DuPaul WD, Kirkley JE, Schmitzer AC. 1989. Evidence of a semiannual reproductive cycle for the sea scallop, *Placopecten magellanicus* (Gmelin, 1791), in the Mid-Atlantic region. *J Shellfish Res.* 8:173-178.
- Fournier D, Archibald CP. 1982. General theory for analyzing catch at age data. *Can J Fish Aquat Sci.* 39:1195-1207.
- Gedamke T, DuPaul WD, Hoenig JM. 2004. A spatially explicit open-ocean DeLury analysis to estimate commercial scallop dredge efficiency. *N Amer J Fish Manage.* 24:335-351.
- Gedamke T, DuPaul WD, Hoenig JM. 2005. Index-removal estimates of dredge efficiency for sea scallops on Georges Bank. *N Amer J Fish Manage.* 25:1122-1129.
- Harris BP, Stokesbury KDE. 2006. Shell growth of sea scallops (*Placopecten magellanicus*) in the southern and northern Great South Channel, USA. *ICES J Mar Sci.* 63:811-821.
- Hart DR. 2001. Individual-based yield-per-recruit analysis, with an application to the Atlantic sea scallop, *Placopecten magellanicus*. *Can J Fish Aquat Sci.* 58:2351-2358.
- Hart DR. 2003. Yield- and biomass-per-recruit analysis for rotational fisheries, with an application to the Atlantic sea scallop (*Placopecten magellanicus*). *Fish Bull.* 101:44-57.
- Hart DR. 2006. Sea scallop stock assessment update for 2005. NEFSC Ref Doc. 06-20; 14 p.
- Hart DR, Chute AS. 2004. Essential fish habitat source document: Sea scallop, *Placopecten magellanicus*, life history and habitat characteristics, 2nd ed. NOAA Tech Memo NMFS NE 189; 21 p.
- Hart DR, Rago PJ. 2006. Long-term dynamics of US Atlantic sea scallop *Placopecten magellanicus* populations. *N Amer J Fish Manage* 26:490-501.
- Haynes EB. 1966. Length-weight relationship of the sea scallop *Placopecten magellanicus* (Gmelin). *Res Bull Int Comm Northw Atl Fish.* 3:32-48.
- Holling CS. 1959. Some characteristics of simple types of predation and parasitism. *Can Entomol.* 91:385-398.
- Jenkins SR, Brand AR. 2001. The effect of dredge capture on the escape response of the great scallop, *Pecten maximus* (L.): implications for the survival of undersized discards. *J Exp Mar Biol Ecol.* 266:33-50.
- Kirkley JE, DuPaul WD. 1989. Commercial practices and fishery regulations: The United States northwest Atlantic sea scallop, *Placopecten magellanicus* (Gmelin, 1791), fishery. *J Shellfish Res.* 8:139-149.
- Lai HL, Helser T. 2004. Linear mixed-effects models for weight-length relationships. *Fish Res.* 70:377-387.
- MacDonald BA, Thompson RJ. 1985. Influence of temperature and food availability on the ecological energetics of the giant scallop *Placopecten magellanicus*. II. Reproductive output and total production. *Mar Ecol Prog Ser.* 25:295-303.
- MacDonald BA, Thompson RJ. 1986. Production, dynamics and energy partitioning in two populations of the giant scallop *Placopecten magellanicus* (Gmelin). *J Exp Mar Biol Ecol.* 101:285-299.
- McCullagh P, Nelder JA. 1989. *Generalized linear models*, 2nd ed. Chapman & Hall.

- McGarvey R, Serchuk FM, McLaren IA. 1992. Statistics of reproduction and early life history survival of the Georges Bank sea scallop (*Placopecten magellanicus*) population. J Northw Atl Fish Sci. 13:83-89.
- Merrill AS, Posgay JA. 1964. Estimating the natural mortality rate of sea scallop. Res Bull Int Comm Northw Atlant Fish. 1:88-106.
- Merrill AS, Posgay JA, Nichy F. 1966. Annual marks on shell and ligament of sea scallop (*Placopecten magellanicus*). Fish Bull. 65:299-311.
- Methot RD. 2000. Technical description of the stock synthesis assessment program. NOAA Tech Memo NMFS NWFSC 43;1-46.
- Millar RB. 1992. Estimating the size-selectivity of fishing gear by conditioning on the total catch. J Am Stat Assoc. 87: 962-968.
- Millar RB, Fryer RJ. 1999. Estimating the size-selection curves of towed gears, traps, nets and hooks. Rev Fish Biol Fish. 9(1):86-116.
- Millar RB, Broadhurst MK, Macbeth WG. 2004. Modeling between-haul variability in the size selectivity of trawls. Fish Res. 67:171-181.
- Murawski SA, Serchuk FM. 1989. Environmental effects of offshore dredge fisheries for bivalves. ICES CM 1989/K:27.
- Naidu KS. 1987. Efficiency of meat recovery from Iceland scallops (*Chlamys islandica*) and sea scallops (*Placopecten magellanicus*) in the Canadian offshore fishery. J Northw Atl Fish Sci. 7:131-136.
- NEFMC. 1993. Amendment #4 and supplemental environmental impact statement to the sea scallop fishery management plan. Saugus MA: NEFMC.
- NEFMC. 2003. Final Amendment 10 to the Atlantic sea scallop fishery management plan with a supplemental environmental impact statement, regulatory impact review, and regulatory flexibility analysis. Newburyport MA: NEFMC.
- NEFMC. 2005. Framework adjustment 18 to the Atlantic sea scallop FMP, including an environmental assessment, regulatory impact review, regulatory flexibility analysis and stock assessment and fishery evaluation (SAFE) report. Newburyport MA: NEFMC.
- Northeast Fishery Science Center (NEFSC). 1992. Report of the 14th Northeast Regional Stock Assessment Workshop (14th SAW). [By Northeast Regional Stock Assessment Workshop No. 14.] NEFSC Ref. Doc. 92-07, 140 p.
- NEFSC. 1995. 20th Northeast Regional Stock Assessment Workshop (20th SAW), Stock Assessment Review Committee (SARC) Consensus Summary of Assessment. NEFSC Ref Doc. 95-18; 211 p.
- NEFSC. 1997. 23rd Northeast Regional Stock Assessment Workshop (23rd SAW), Stock Assessment Review Committee (SARC) Consensus Summary of Assessment. NEFSC Ref. Doc. 97-05; 191 p.
- NEFSC 1999. 29th Northeast Regional Stock Assessment Workshop (29th SAW). Stock Assessment Review Committee (SARC) consensus summary of assessments. NEFSC Ref Doc. 99-14.
- NEFSC. 2001. 32nd Northeast Regional Stock Assessment Workshop (32nd SAW). Stock Assessment Review Committee (SARC) consensus summary of assessments. NEFSC Ref Doc. 01-05; 289 p.
- NEFSC. 2004. 39th Northeast Regional Stock Assessment Workshop (39th SAW) Assessment Summary Report & Assessment Report. NEFSC Ref. Doc. 04-10a (16 p) and b (211 p).
- Overholtz WJ, Jacobson L, Melvin GD, Cieri M, Power M, Clark K. 2004. Stock assessment of the Gulf of Maine-Georges Bank herring complex, 1959-2002. NEFSC Ref Doc. 04-06.
- Pennington M, Burmeister L-M, Hjellvik V. 2002. Assessing the precision of frequency distributions estimated from trawl-survey samples. Fish Bull. 100:74-80.
- Pinheiro JC, Bates DM. 2000. Mixed-effects models in S and S-PLUS. Springer.
- Posgay JA. 1979. Population assessment of the Georges Bank sea scallop stocks. Rapp R-v Reun Cons Int Explor Mer. 175:109-113.
- Press WH, Flannery BP, Teukolsky SA, Vetterling WT. 1990. Numerical recipes. New York: Cambridge Univ Press.
- Rice JA. 1987. Mathematical statistics and data analysis. Wadsworth.
- Ricker WE. 1975. Computation and interpretation of biological statistics of fish populations. Bull Fish Res Brd Can Bull. 191; 382 pp.

- Sainsbury KJ .1980.. Effect of individual variability on the von Bertalanffy growth equation. Can J Fish Aquat Sci. 37:241-247.
- Serchuk FM, Rak RS. 1983. Biological characteristics of offshore Gulf of Maine scallop populations: size distributions, shell height-meat weight relationships and relative fecundity patterns. NMFS Woods Hole Lab Ref Doc. 83-07; 42 p.
- Serchuk FM, Smolowitz RJ. 1980. Size selection of sea scallops by an offshore scallop survey dredge. ICES CM 1980/K:24.
- Serchuk FM, Smolowitz RJ. 1989. Seasonality in sea scallop somatic growth and reproductive cycles. J Shellfish Res. 8:435.
- Serchuk FM, Wigley SE. 1989. Current resource conditions in USA Georges Bank and Mid-Atlantic sea scallop populations: Results of the 1989 NMFS sea scallop research vessel survey. NEFSC SAW-9, Working Paper No. 9, 52 p.
- Serchuk FM, Wood PW, Posgay JA, Brown BE. 1979. Assessment and status of sea scallop (*Placopecten magellanicus*) populations of the northeast coast of the United States. Proc Natl Shellfish Assoc. 69:161-191.
- Smith SJ. 1997. Bootstrap confidence limits for groundfish trawl survey estimates of mean abundance. Can J Fish Aquat Sci. 54:616-630.
- Smolowitz RJ, Serchuk FM, Reidman RJ. 1989. The use of a volumetric measure for determining sea scallop meat count. NOAA Tech. Memo F/NER-1.
- Stevenson JA, Dickie LM. 1954. Annual growth rings and rate of growth of the giant scallop *Placopecten magellanicus* (Gmelin) in the Digby Area of the Bay of Fundy. J Fish Res Board Can. 11:660-671.
- Stokesbury KDE, Harris BP, Marino MC II, Nogueira JI. 2004. Estimation of sea scallop abundance using a video survey in off-shore U.S. waters. J Shellfish Res. 23: 33-40.
- Sullivan PJ, Lai H-L, Gallucci VF. 1990. A catch-at-length analysis that incorporates a stochastic model of growth. Can J Fish Aquat Sci 47:184-198.
- Tan FC, Cai D, Roddick DL. 1988. Oxygen isotope studies on sea scallops, *Placopecten magellanicus*, from Browns Bank, Nova Scotia. Can J Fish Aquat Sci. 45:1378-1386.
- Thouzeau G, Robert G, Smith SJ. 1991. Spatial variability in distribution and growth of juvenile and adult sea scallops *Placopecten magellanicus* (Gmelin) on eastern Georges Bank (Northwest Atlantic). Mar Ecol Prog Ser. 74:205-218.
- Veale LO, Hill AS, Brand AR. 2000. An in situ study of predator aggregations on scallop (*Pecten maximus* (L.)) dredge discards using a static time-lapse camera system. J Exp Mar Biol Ecol. 255:111-129.
- Wigley SE, Serchuk FM. 1996. Current resource conditions in Georges Bank and Mid-Atlantic sea scallop populations: results of the 1994 NEFSC sea scallop research vessel survey. NEFSC Ref Doc. 96-03.
- Wigley SE, Terceiro M, DeLong A, Sosebee K. 1998. Proration of 1994-96 USA commercial landings of Atlantic cod, haddock, and yellowtail flounder to unit stock areas. NEFSC Ref Doc. 98-2.
- Yochum N. 2006. Size-selectivity of the commercial sea scallop (*Placopecten magellanicus*) dredge. Masters thesis, Virginia Institute of Marine Science, College of William and Mary.

ATLANTIC SEA SCALLOP TABLES

Table B4-1. US sea scallop landings (mt meats) 1964-2006.

Year	Gulf of Maine			Georges Bank			S. New England			Mid-Atlantic Bight			Total		
	dredge	trawl	sum	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum
1964		0	208		0	6,241	6,241		52	3	55		0	137	137
1965		0	117		3	1,478	1,481		2	24	26		0	3,974	3,974
1966		0	102		0	883	884		0	8	8		0	4,061	4,061
1967		0	80		4	1,217	1,221		0	8	8		0	1,873	1,873
1968		0	113		0	993	994		0	56	56		0	2,437	2,437
1969		1	122		8	1,316	1,324		0	18	19		5	846	851
1970		0	132		5	1,410	1,415		0	6	6		14	459	473
1971		4	358		18	1,311	1,329		0	7	7		0	274	274
1972		1	524		5	816	821		0	2	2		5	653	658
1973		0	460		15	1,065	1,080		0	3	3		4	245	249
1974		0	223		15	911	926		0	4	5		0	937	938
1975		6	741		13	844	857		8	42	50		52	1,506	1,558
1976		3	364		38	1,723	1,761		4	3	7		819	2,972	3,791
1977		4	254		27	4,709	4,736		1	10	11		255	2,564	2,819
1978		242	1	0	243	0	5,569	5,532	25	2	27		207	0	4,642
1979		401	5	1	407	7	6,285	6,253	61	5	66		2857	29	2,888
1980		1,489	122	3	1,614	2	5,419	5,382	34	0	133		2,202	85	2,366
1981		1,225	73	7	1,305	0	7,843	7,787	56	0	69		772	14	2,788
1982		631	28	5	664	0	6,322	6,204	119	0	126		1,602	6	2,161
1983		815	72	7	895	4	4,284	4,247	32	0	243		3,092	19	3,121
1984		651	18	10	678	3	3,043	3,011	29	3	164		3,695	53	3,750
1985		408	3	10	421	0	2,894	2,860	34	0	82		3,230	49	3,281
1986		308	2	6	316	0	4,438	4,428	10	0	78		3,407	386	3,799
1987		373	0	9	382	0	4,851	4,821	30	0	68		7,639	1,168	8,808
1988		506	7	13	526	0	6,054	6,036	18	0	68		6,071	938	8,7017
1989		600	0	44	644	0	5,661	5,637	25	0	138		7,894	534	8,433
1990		545	0	28	574	0	9,982	9,972	10	0	116		6,364	541	10,6915
1991		527	3	75	605	0	9,311	9,235	77	0	71		6,408	878	14,7300
1992		676	2	45	722	0	8,238	8,230	7	0	124		4,562	570	5,137
1993		763	2	32	797	0	3,655	3,637	18	0	66		2,412	393	2,808
1994		375	17	0	392	1	1,205	1,195	13	1	35		5,180	792	0
1995		446	3	6	455	1	1,199	1,175	17	7	48		5,750	317	1
1996		634	9	3	646	0	2,061	1,992	69	0	82		4,156	759	3
1997		601	9	12	622	0	2,053	2,030	23	0	87		2,391	337	0
1998		474	7	2	483	0	2,039	2,021	18	0	100		2,356	531	5
1999		239	2	2	243	0	5,085	5,079	6	0	80		3,516	890	8
2000		139	5	1	144	0	5,039	5,016	20	3	74		7,652	1,200	0
2001		251	8	1	260	0	4,597	4,563	35	0	29		13,989	1,621	0
2002		492	6	1	499	0	5,541	5,478	63	0	20		15,435	1,616	5
2003		394	8	1	403	0	4,823	4,799	23	0	103		18,796	1,292	1
2004		125	9	0	134	0	4,357	4,343	14	0	120		23,082	1,402	13
2005		134	9	0	143	3	9,502	9,381	118	1	403		14,588	1,040	6
2006		227	1	1	229	0	17,286	17,239	47	0	370		8,232	584	3
									4	0			26,063	636	5
															26,704

Table B4-2. Estimated discards on directed scallop trips 1992-2006 (mt meats).

	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Mid-Atlantic	143	13	853	324	24	8	60	11	871	854	1637	2417	2644	579	213
Georges Bank	448	282	3	38	135	29	5	162	1129	865	128	313	91	286	628
Total	591	295	856	363	159	37	65	173	2000	1719	1765	2729	2735	864	842

Table B4-3. Estimated scallop landings and discards in non-scallop otter trawl fisheries, 1994-2006 (mt meats, “*” indicates < 1 mt meats)

	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
longfin squid	32	26	42	51	42	47	46	1	9	22	42	86	115
summer flounder	7	13	20	12	10	11	12	*	3	10	18	31	34
scup	2	3	5	3	3	2	1	*	1	4	4	10	13
yellowtail flounder	2	1	2	3	3	5	7	*	1	4	7	7	4
haddock	*	*	*	1	1	2	2	*	1	2	4	7	3
cod	1	1	2	2	1	1	1	*	*	1	*	1	1
monkfish	*	*	1	1	*	*	*	*	*	*	*	*	*

Scallops discarded by otter trawl fisheries (mt):

	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
longfin squid	52	58	33	38	68	65	60	112	108	57	68	42	13
summer flounder	12	28	15	9	17	15	16	26	33	26	30	15	4
scup	4	6	4	2	4	2	2	4	8	10	7	5	1
yellowtail flounder	4	2	2	2	5	6	9	17	11	9	12	4	*
haddock	*	*	*	*	2	2	3	7	9	6	6	3	*
cod	2	2	1	1	1	1	1	3	2	2	1	*	*
monkfish	1	1	1	*	*	1	1	1	1	*	*	*	*
prop landed	0.38	0.31	0.56	0.57	0.38	0.42	0.43	0.01	0.08	0.28	0.38	0.67	0.90
total landings (mt)	46	43	72	72	60	67	69	2	15	42	76	142	172
total discards (mt)	74	97	57	54	98	93	92	170	171	109	124	70	19

Table B5-1. NEFSC survey stratified mean indices for (a) Mid-Atlantic sea scallop survey, (b) Georges Bank sea scallop dredge survey, (c) combined Mid-Atlantic and Georges Bank sea scallop dredge survey, and (d) winter bottom trawl survey (Mid-Atlantic only). Unlike previous assessments, no adjustments were made to sea scallop dredge survey data for the liner. Holes were filled with imputed data (Appendix B6).

(a) Mid-Atlantic Bight												
Year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	N positive tows	Proportion Positive Tows	N Strata Surveyed	Area Surveyed (nm ²)	Mean Julian Date	Mean Meat Weight (g / scallop)	
1979	30.7	0.09	0.571	0.11	166	152	0.92	30	8,350	141	18.6	
1980	41.1	0.11	0.498	0.08	167	157	0.94	30	8,350	149	12.1	
1981	30.8	0.16	0.390	0.12	167	152	0.91	29	8,316	169	12.7	
1982	30.7	0.11	0.408	0.08	185	168	0.91	30	8,350	174	13.3	
1983	27.7	0.09	0.358	0.08	193	171	0.89	30	8,350	216	12.9	
1984	29.4	0.11	0.341	0.09	204	186	0.91	29	8,172	213	11.6	
1985	67.0	0.12	0.541	0.08	201	188	0.94	30	8,350	210	8.1	
1986	117.9	0.10	0.954	0.09	226	210	0.93	30	8,350	218	8.1	
1987	122.3	0.09	0.843	0.07	226	211	0.93	30	8,350	195	6.9	
1988	131.2	0.10	1.281	0.07	227	206	0.91	29	8,229	200	9.8	
1989	166.6	0.09	1.166	0.07	244	226	0.93	30	8,350	170	7.0	
1990	205.6	0.22	1.254	0.17	216	192	0.89	30	8,350	214	6.1	
1991	75.7	0.10	0.729	0.12	229	211	0.92	30	8,350	216	9.6	
1992	40.6	0.11	0.414	0.07	229	200	0.87	30	8,350	218	10.2	
1993	127.5	0.10	0.573	0.07	214	205	0.96	30	8,350	217	4.5	
1994	125.8	0.11	0.772	0.08	227	214	0.94	30	8,350	181	6.1	
1995	168.7	0.13	1.160	0.10	227	217	0.96	30	8,350	179	6.9	
1996	55.8	0.08	0.564	0.07	211	188	0.89	30	8,350	217	10.1	
1997	41.3	0.13	0.428	0.06	225	209	0.93	30	8,350	208	10.4	
1998	157.6	0.18	0.808	0.14	226	208	0.92	30	8,350	208	5.1	
1999	234.0	0.22	1.708	0.18	226	209	0.92	30	8,350	204	7.3	
2000	283.6	0.15	2.979	0.13	229	201	0.88	30	8,350	203	10.5	
2001	306.3	0.13	3.322	0.13	227	205	0.90	30	8,350	188	10.8	
2002	301.0	0.11	3.743	0.12	206	184	0.89	30	8,350	206	12.4	
2003	641.3	0.16	5.678	0.10	201	181	0.90	30	8,350	217	8.9	
2004	468.8	0.11	5.232	0.07	248	220	0.89	30	8,350	194	11.2	
2005	360.1	0.09	6.045	0.09	241	223	0.93	30	8,350	203	16.8	
2006	378.1	0.10	5.883	0.07	230	215	0.93	30	8,350	201	15.6	

(b) Georges Bank

Year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	N positive tows	Proportion Positive Tows	N Strata Surveyed	Area Surveyed (nm2)	Mean Julian Date	Mean Meat Weight (g / scallop)
1982	133.4	0.56	0.847	0.32	129	103	0.80	28	7,250	204	6.4
1983	50.7	0.29	0.600	0.24	138	117	0.85	29	7,216	232	11.8
1984	30.1	0.12	0.411	0.10	138	114	0.83	29	7,172	227	13.7
1985	53.8	0.19	0.560	0.18	170	144	0.85	30	7,281	235	10.4
1986	90.4	0.18	0.689	0.12	194	164	0.85	29	7,029	234	7.6
1987	106.6	0.21	0.903	0.17	190	156	0.82	30	7,281	214	8.5
1988	81.8	0.18	0.715	0.15	192	150	0.78	30	7,281	217	8.7
1989	39.6	0.26	0.364	0.17	254	209	0.82	30	7,281	217	9.2
1990	175.9	0.29	1.093	0.23	194	155	0.80	30	7,281	226	6.2
1991	241.2	0.48	1.160	0.28	194	170	0.88	30	7,281	228	4.8
1992	247.1	0.55	1.683	0.47	191	165	0.86	30	7,281	230	6.8
1993	62.5	0.30	0.427	0.19	182	150	0.82	30	7,281	230	6.8
1994	38.0	0.18	0.357	0.13	194	156	0.80	30	7,281	195	9.4
1995	109.7	0.25	0.672	0.17	193	164	0.85	30	7,281	213	6.1
1996	113.5	0.18	1.124	0.16	189	164	0.87	30	7,281	234	9.9
1997	80.6	0.14	1.273	0.15	206	175	0.85	30	7,281	224	15.8
1998	271.2	0.28	3.703	0.35	230	197	0.86	30	7,281	222	13.7
1999	159.8	0.15	2.559	0.16	198	186	0.94	30	7,281	215	16.0
2000	715.5	0.29	6.336	0.21	188	168	0.89	30	7,281	222	8.9
2001	357.8	0.15	5.128	0.15	225	211	0.94	30	7,281	217	14.3
2002	297.9	0.14	5.976	0.13	229	207	0.90	30	7,281	221	20.1
2003	225.8	0.12	5.419	0.14	225	206	0.92	30	7,281	233	24.0
2004	269.9	0.10	7.132	0.11	230	212	0.92	30	7,281	210	26.4
2005	210.5	0.12	5.698	0.11	227	211	0.93	30	7,281	213	27.1
2006	151.0	0.10	4.502	0.11	231	212	0.92	30	7,281	216	29.8

(c) Whole stock

Year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	N positive tows	Proportion Positive Tows	N Strata Surveyed	Area Surveyed (nm2)	Mean Julian Date	Mean Meat Weight (g / scallop)
1982	78.6	0.44	0.613	0.21	314	271	0.86	58	15,600	188	7.8
1983	38.4	0.18	0.471	0.15	331	288	0.87	59	15,566	224	12.3
1984	29.7	0.08	0.374	0.06	342	300	0.88	58	15,344	220	12.6
1985	60.8	0.11	0.550	0.10	371	332	0.89	60	15,631	222	9.0
1986	105.1	0.09	0.831	0.07	420	374	0.89	59	15,378	226	7.9
1987	115.0	0.10	0.871	0.09	416	367	0.88	60	15,631	204	7.6
1988	108.2	0.09	1.017	0.07	419	356	0.85	59	15,510	208	9.4
1989	107.4	0.09	0.792	0.07	498	435	0.87	60	15,631	192	7.4
1990	191.8	0.18	1.179	0.14	410	347	0.85	60	15,631	219	6.1
1991	152.8	0.36	0.930	0.17	423	381	0.90	60	15,631	222	6.1
1992	136.8	0.46	1.006	0.36	420	365	0.87	60	15,631	224	7.4
1993	97.3	0.12	0.505	0.09	396	355	0.90	60	15,631	223	5.2
1994	84.9	0.10	0.579	0.07	421	370	0.88	60	15,631	187	6.8
1995	141.2	0.12	0.932	0.09	420	381	0.91	60	15,631	195	6.6
1996	82.7	0.12	0.825	0.10	400	352	0.88	60	15,631	225	10.0
1997	59.6	0.10	0.822	0.11	431	384	0.89	60	15,631	215	13.8
1998	210.5	0.18	2.157	0.28	456	405	0.89	60	15,631	214	10.2
1999	199.4	0.15	2.104	0.12	424	395	0.93	60	15,631	209	10.6
2000	484.8	0.21	4.542	0.14	417	369	0.88	60	15,631	212	9.4
2001	330.3	0.10	4.163	0.10	452	416	0.92	60	15,631	201	12.6
2002	299.6	0.09	4.783	0.09	435	391	0.90	60	15,631	213	16.0
2003	447.7	0.12	5.557	0.08	426	387	0.91	60	15,631	224	12.4
2004	376.1	0.08	6.117	0.07	478	432	0.90	60	15,631	201	16.3
2005	290.4	0.07	5.883	0.07	468	434	0.93	60	15,631	208	20.3
2006	272.3	0.08	5.240	0.06	461	427	0.93	60	15,631	208	19.2

(d) Winter trawl survey (Mid-Atlantic only)

Year	Abundance index (mean N/tow)	CV	N tows	N positive tows	Proportion Positive Tows	N Bottom Trawl Strata Surveyed	Area Surveyed (nm ²)	Mean Julian Date
1992	50.4	0.49	79	40	0.506	18	22,118	50
1993	25.8	0.25	84	46	0.548	18	22,092	43
1994	84.5	0.56	54	41	0.759	18	22,118	37
1995	43.9	0.30	81	42	0.519	20	22,536	46
1996	49.9	0.31	91	44	0.484	18	22,118	43
1997	4.7	0.21	83	40	0.482	22	22,701	41
1998	57.3	0.22	90	38	0.422	23	22,625	45
1999	71.3	0.49	91	46	0.505	24	22,813	38
2000	96.7	0.32	103	39	0.379	23	22,583	49
2001	22.6	0.24	118	50	0.424	24	22,813	37
2002	101.9	0.43	118	48	0.407	23	22,753	47
2003	158.0	0.82	86	24	0.279	24	22,813	47
2004	53.3	0.35	106	36	0.340	24	22,813	45
2005	81.0	0.44	82	41	0.500	23	22,753	43
2006	65.1	0.29	106	57	0.538	24	22,813	45

Table B5-2. SMAST large camera video survey data for sea scallops 40+ mm SL in the Mid-Atlantic Bight and Georges Bank regions, 2003-2006. Four camera images were taken at each station.

Year	Density (N/m ²)	SE	CV	N stations	Area surveyed (km ²)
Mid-Atlantic Bight					
2003	0.4420	0.0756	16.5	851	26,270
2004	0.2236	0.0219	9.8	840	25,930
2005	0.2038	0.0246	11.9	868	26,794
2006	0.1897	0.0188	9.9	903	27,875
Georges Bank					
2003	0.1474	0.0124	8.4	924	28,523
2004	0.1202	0.0143	11.8	941	29,048
2005	0.1115	0.0121	10.8	943	29,110
2006	0.1060	0.0112	10.6	943	29,110
Georges Bank - Open Areas					
2003	0.0900	0.0085	9.3	513	15,836
2004	0.0711	0.0091	12.6	531	16,392
2005	0.0621	0.0073	11.6	536	16,546
2006	0.0597	0.0084	14.1	531	16,546
Georges Bank – Closed Areas					
2003	0.2191	0.0253	11.5	411	12,687
2004	0.1838	0.0304	16.5	410	12,656
2005	0.1767	0.0259	14.6	407	12,564
2006	0.1658	0.0230	13.8	412	12,564

Table B5-3. SMAST small camera video survey data for sea scallops 40+ mm SL in the Mid-Atlantic Bight and Georges Bank regions, 2003-2006. Four camera images were taken at each station.

Year	Density (N/m ²)	SE	CV	N stations	Area surveyed (km ²)
Mid-Atlantic Bight					
2003	0.4899	0.1179	20.2	845	26,084
2004	0.2119	0.0242	11.3	837	25,837
2005	0.1957	0.0248	12.2	866	26,733
2006	0.1896	0.0200	10.5	895	27,628
Georges Bank					
2003	0.1538	0.0163	10.3	905	27,937
2004	0.1155	0.0136	11.2	929	28,677
2005	0.0975	0.0125	12.7	917	28,307
2006	0.1348	0.0138	10.0	939	28,986
Georges Bank - Open Areas					
2003	0.0798	0.0103	12.1	503	15,527
2004	0.0743	0.0129	16.0	521	16,083
2005	0.0549	0.0084	14.8	526	16,237
2006	0.0936	0.0125	13.1	531	16,392
Georges Bank - Closed Areas					
2003	0.2468	0.0338	13.5	402	12,409
2004	0.1679	0.0260	15.1	408	12,595
2005	0.1557	0.0268	17.2	391	12,070
2006	0.1885	0.0270	14.0	408	12,595

Table B5-4. Survey and rescaled fishing mortality estimates for (a) Mid-Atlantic, (b) Georges Bank, (c) overall.

(a) Mid-Atlantic

	80-98	98+	SurveyF	SE	Landings	MinEBms	EBms	CV	CBI	CV	RescaledF	CV
1979	7.2	14.2			2,888	3,003	7,150	0.10	0.40	0.14	0.58	0.15
1980	4.8	11.7	0.50	0.12	1,975	2,471	5,884	0.07	0.34	0.12	0.48	0.13
1981	6.6	7.5	0.69	0.12	731	1,853	4,411	0.10	0.17	0.14	0.24	0.15
1982	7.9	7.9	0.48	0.15	1,610	1,953	4,649	0.08	0.35	0.13	0.50	0.13
1983	4.4	7.9	0.58	0.14	3,109	1,713	4,077	0.08	0.76	0.13	1.09	0.13
1984	10.0	6.2	0.60	0.11	3,675	1,674	3,985	0.09	0.92	0.13	1.32	0.14
1985	11.6	9.0	0.49	0.15	3,276	2,210	5,263	0.07	0.62	0.12	0.89	0.13
1986	30.8	12.5	0.40	0.13	3,359	3,846	9,158	0.07	0.37	0.12	0.53	0.13
1987	28.8	11.1	1.26	0.12	7,803	3,355	7,989	0.06	0.98	0.12	1.40	0.12
1988	46.6	18.8	0.65	0.10	6,178	5,829	13,878	0.07	0.45	0.12	0.64	0.13
1989	36.3	13.3	1.49	0.12	7,973	4,482	10,670	0.07	0.75	0.12	1.07	0.13
1990	35.6	9.7	1.54	0.11	6,435	4,034	9,606	0.10	0.67	0.14	0.96	0.15
1991	22.9	11.7	1.25	0.18	7,011	3,237	7,707	0.11	0.91	0.15	1.30	0.15
1992	10.8	8.5	1.30	0.13	4,955	1,939	4,616	0.07	1.07	0.12	1.54	0.13
1993	9.5	6.0	1.08	0.11	2,778	1,596	3,800	0.07	0.73	0.12	1.05	0.13
1994	31.4	5.8	0.88	0.12	5,912	3,010	7,166	0.08	0.82	0.13	1.18	0.13
1995	38.1	10.8	1.13	0.11	5,976	4,140	9,858	0.09	0.61	0.13	0.87	0.14
1996	27.8	8.5	1.65	0.11	4,828	2,668	6,353	0.06	0.76	0.12	1.09	0.13
1997	8.9	10.3	1.16	0.10	2,728	2,110	5,024	0.06	0.54	0.11	0.78	0.12
1998	21.5	8.0	0.78	0.09	2,891	2,453	5,840	0.14	0.50	0.17	0.71	0.17
1999	61.4	22.0	0.19	0.23	4,414	6,826	16,251	0.15	0.27	0.18	0.39	0.18
2000	121.5	57.1	0.28	0.21	8,852	14,163	33,720	0.13	0.26	0.16	0.38	0.17
2001	93.5	94.3	0.54	0.22	15,611	16,137	38,422	0.14	0.41	0.17	0.58	0.18
2002	102.5	110.7	0.43	0.22	17,056	19,003	45,246	0.12	0.38	0.16	0.54	0.16
2003	107.7	146.5	0.28	0.17	20,089	25,929	61,736	0.09	0.33	0.14	0.47	0.14
2004	133.0	90.8	0.93	0.13	24,497	23,419	55,760	0.06	0.44	0.12	0.63	0.12
2005	97.4	182.4	0.10	0.11	15,634	32,115	76,464	0.09	0.20	0.13	0.29	0.14
2006	91.5	163.6	0.21	0.13	8,819	29,905	71,203	0.08	0.12	0.13	0.18	0.13
Mean7906	43.2	38.1	0.77	0.03	7,181	8,038	19,139	0.03	0.54	0.03	0.77	0.03
Mean7994	19.1	10.1	0.88	0.03	4,354	2,888	6,876	0.02	0.64	0.02	0.92	0.04
Mean9506	75.4	75.4	0.6	0.05	10,950	14,906	35,490	0.03	0.40	0.04	0.57	0.05

Table B5-4 continued
(b) Georges Bank

	80-100.8	100.8+	SurveyF	SE	Landings	MinEBms	Ebms	CV	CBI	CV	RescaledF	CV
1982	7.2	7.7			6,322	2,429	6,747	0.12	0.94	0.15	0.87	0.17
1983	5.7	7.1	0.64	0.24	4,284	2,509	6,970	0.10	0.61	0.14	0.57	0.16
1984	6.9	8.7	0.29	0.31	3,043	1,958	5,439	0.10	0.56	0.14	0.52	0.16
1985	6.4	6.4	0.79	0.40	2,894	2,272	6,310	0.14	0.46	0.17	0.43	0.19
1986	8.2	7.6	0.42	0.29	4,438	2,384	6,621	0.09	0.67	0.13	0.62	0.16
1987	18.2	7.3	0.67	0.18	4,851	3,461	9,614	0.13	0.50	0.16	0.47	0.18
1988	18.9	6.4	1.28	0.21	6,054	2,812	7,812	0.10	0.77	0.14	0.72	0.16
1989	5.1	6.9	1.20	0.22	5,661	1,401	3,891	0.32	1.46	0.34	1.35	0.35
1990	38.1	7.0	0.45	0.18	9,982	3,511	9,753	0.22	1.02	0.24	0.95	0.25
1991	18.8	21.6	0.63	0.28	9,311	2,997	8,326	0.09	1.12	0.14	1.04	0.16
1992	18.2	15.4	0.87	0.16	8,238	3,764	10,457	0.17	0.79	0.19	0.73	0.21
1993	6.4	15.0	0.70	0.21	3,655	1,297	3,604	0.10	1.01	0.14	0.94	0.16
1994	8.9	17.6	0.10	0.15	1,205	1,522	4,229	0.12	0.28	0.16	0.27	0.17
1995	6.3	28.0	-0.15	0.24	1,199	1,955	5,430	0.12	0.22	0.15	0.21	0.17
1996	18.7	21.5	0.36	0.23	2,061	4,568	12,688	0.14	0.16	0.17	0.15	0.19
1997	41.5	30.6	0.17	0.24	2,053	6,084	16,900	0.14	0.12	0.17	0.11	0.19
1998	45.9	70.6	-0.08	0.41	2,039	17,818	49,493	0.39	0.04	0.40	0.04	0.41
1999	46.1	55.3	0.65	0.40	5,085	12,928	35,911	0.31	0.14	0.32	0.13	0.33
2000	83.3	227.6	-0.91	0.30	5,039	23,790	66,084	0.20	0.08	0.23	0.07	0.24
2001	84.9	82.7	1.22	0.26	4,597	24,426	67,849	0.15	0.07	0.18	0.06	0.20
2002	83.5	128.7	0.16	0.21	5,541	31,380	87,166	0.13	0.06	0.17	0.06	0.18
2003	20.2	136.2	0.34	0.20	4,823	28,510	79,195	0.13	0.06	0.17	0.06	0.18
2004	50.0	162.4	-0.14	0.18	4,357	38,419	106,721	0.11	0.04	0.15	0.04	0.17
2005	31.0	141.7	0.00	0.16	9,502	30,882	85,783	0.11	0.11	0.15	0.10	0.17
2006	22.4	99.5	0.66	0.16	17,286	24,271	67,418	0.11	0.26	0.15	0.24	0.17
Mean8206	28.0	52.8	0.43	0.05	5,341	11,094	30,816	0.05	0.46	0.06	0.43	0.06
Mean8294	12.8	10.4	0.64	0.07	5,380	2,486	6,905	0.04	0.78	0.06	0.71	0.07
Mean9506	44.5	98.8	0.19	0.08	5,299	20,419	56,720	0.05	0.11	0.06	0.11	0.07

Table B5-4 continued
(c) Whole stock

	SurveyF	Landings	Ebms	CBI	RescaledF	CV
1982	0.63	7933	11396	0.70	0.74	0.08
1983	0.44	7392	11047	0.67	0.72	0.07
1984	0.65	6718	9424	0.71	0.91	0.07
1985	0.41	6170	11573	0.53	0.64	0.08
1986	0.95	7797	15779	0.49	0.56	0.07
1987	0.87	12654	17603	0.72	0.91	0.08
1988	1.43	12232	21690	0.56	0.67	0.07
1989	1.02	13634	14561	0.94	1.13	0.13
1990	0.92	16417	19358	0.85	0.96	0.10
1991	0.96	16323	16034	1.02	1.16	0.08
1992	0.89	13192	15073	0.88	0.91	0.09
1993	0.64	6433	7404	0.87	1.00	0.07
1994	0.79	7116	11395	0.62	0.90	0.08
1995	0.87	7175	15288	0.47	0.69	0.08
1996	0.42	6889	19041	0.36	0.52	0.08
1997	0.09	4780	21924	0.22	0.28	0.08
1998	0.43	4930	55334	0.09	0.17	0.15
1999	-0.43	9499	52162	0.18	0.25	0.13
2000	0.92	13891	99805	0.14	0.19	0.10
2001	0.28	20208	106272	0.19	0.29	0.09
2002	0.30	22597	132412	0.17	0.27	0.09
2003	0.43	24911	140932	0.18	0.31	0.08
2004	0.43	28855	162481	0.18	0.35	0.07
2005	0.06	25136	162481	0.15	0.22	0.08
2006	0.60	26105	162247	0.16	0.20	0.07
Mean8206	0.60	13160	52509	0.48	0.75	
Mean8294	0.82	10506	14245	0.74	0.87	
Mean9506	0.38	16248	94198	0.21	0.33	

Table B5-5. CASA model estimates for sea scallop recruitment, stock biomass, stock abundance (*top panel*), catch numbers, fishable biomass and fully recruited fishing mortality (*lower panel*) on Georges Bank (1982-2006), in the Mid-Atlantic Bight (1975-2006), and for the whole stock (1982-2006). CVs for all estimates are given in a separate table.

Year	Recruitment (January 1, millions)			Stock biomass (January 1, 40+ mm SL, mt)			Stock abundance (January 1, 40+ mm SL, millions)		
	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock
1975	--	--	--	--	11,318	--	--	571	--
1976	--	413	--	--	11,865	--	--	753	--
1977	--	250	--	--	11,319	--	--	824	--
1978	--	109	--	--	11,897	--	--	750	--
1979	--	71	--	--	9,546	--	--	528	--
1980	--	164	--	--	8,061	--	--	469	--
1981	--	136	--	--	7,664	--	--	473	--
1982	--	135	--	10,966	8,907	19,873	1,828	524	2,351
1983	181	208	388	11,457	9,221	20,678	1,141	582	1,723
1984	269	305	574	9,465	8,130	17,595	752	654	1,406
1985	369	780	1,149	10,898	7,739	18,637	832	1,044	1,875
1986	826	732	1,558	13,078	10,508	23,585	1,237	1,446	2,684
1987	640	984	1,624	14,782	15,503	30,284	1,445	1,921	3,365
1988	478	647	1,125	16,623	15,698	32,321	1,413	1,823	3,236
1989	1,111	837	1,948	17,453	17,491	34,944	1,720	1,894	3,614
1990	859	423	1,281	20,955	16,211	37,166	2,005	1,600	3,605
1991	1,004	231	1,235	18,876	14,837	33,713	1,941	1,206	3,147
1992	243	217	460	14,476	10,366	24,842	1,317	804	2,121
1993	315	1,145	1,460	7,894	8,780	16,674	736	1,293	2,029
1994	265	682	947	5,923	13,632	19,554	587	1,747	2,334
1995	658	303	960	9,249	14,359	23,608	967	1,435	2,402
1996	352	103	455	14,989	12,177	27,167	1,220	925	2,146
1997	418	500	918	19,500	10,027	29,526	1,313	881	2,194
1998	752	2,048	2,800	24,385	14,202	38,587	1,641	2,257	3,898
1999	752	1,695	2,447	31,783	27,069	58,852	2,054	3,599	5,653
2000	1,850	1,451	3,302	39,549	44,664	84,212	3,093	4,418	7,511
2001	470	1,444	1,915	52,681	59,007	111,688	3,366	4,825	8,191
2002	367	1,121	1,488	64,628	64,744	129,372	3,174	4,657	7,831
2003	744	3,211	3,956	72,724	70,580	143,305	3,186	6,014	9,200
2004	262	312	575	78,623	78,448	157,071	2,987	5,563	8,550
2005	453	1,776	2,229	84,106	78,387	162,493	2,935	5,360	8,295
2006	225	370	594	81,047	85,161	166,208	2,637	4,833	7,469

Table B5-5 continued

Year	Catch numbers (all sizes, millions)			Fishable mean abundance (all sizes, millions)			Fully recruited fishing mortality (F, annual)		
	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock
1975	--	67	--	--	393	--	--	0.171	--
1976	--	138	--	--	349	--	--	0.395	--
1977	--	134	--	--	424	--	--	0.316	--
1978	--	230	--	--	437	--	--	0.526	--
1979	--	145	--	--	341	--	--	0.427	--
1980	--	90	--	--	266	--	--	0.339	--
1981	--	36	--	--	290	--	--	0.125	--
1982	298	75	373	127	300	427	2.346	0.249	0.873
1983	431	142	573	455	271	726	0.947	0.525	0.789
1984	149	180	330	383	220	603	0.390	0.819	0.547
1985	153	196	349	354	216	569	0.432	0.909	0.613
1986	239	262	501	342	401	742	0.701	0.653	0.675
1987	300	585	885	475	490	964	0.631	1.196	0.918
1988	371	499	870	557	591	1,148	0.667	0.843	0.758
1989	322	599	921	520	589	1,109	0.619	1.018	0.831
1990	592	500	1,092	560	590	1,150	1.057	0.847	0.949
1991	619	496	1,115	465	503	968	1.331	0.987	1.152
1992	586	312	898	363	328	691	1.614	0.952	1.300
1993	268	174	442	223	251	474	1.201	0.696	0.934
1994	74	499	573	234	439	673	0.317	1.136	0.851
1995	57	512	569	346	555	900	0.166	0.922	0.632
1996	125	343	468	344	469	813	0.363	0.731	0.576
1997	138	136	275	452	270	722	0.306	0.504	0.380
1998	111	140	250	470	274	744	0.236	0.510	0.337
1999	185	259	444	603	544	1,147	0.308	0.475	0.387
2000	167	536	703	747	1,201	1,948	0.223	0.447	0.361
2001	185	838	1,023	1,048	1,629	2,677	0.177	0.515	0.382
2002	224	745	969	999	1,251	2,250	0.224	0.596	0.431
2003	206	812	1,019	1,068	1,331	2,399	0.193	0.610	0.425
2004	129	955	1,084	1,576	1,305	2,881	0.082	0.731	0.376
2005	250	685	935	1,580	1,678	3,258	0.158	0.408	0.287
2006	431	368	799	1,390	2,105	3,495	0.310	0.175	0.229

Table B5-6. Coefficients of variation for CASA model estimates of sea scallop recruitment, stock biomass, stock abundance (top panel), catch numbers, fishable biomass and fully recruited fishing mortality (bottom panel) on Georges Bank (1982-2006), in the Mid-Atlantic Bight (1975-2006), and for the whole stock (1982-2006).

Year	Recruitment (January 1, millions)			Stock biomass (January 1, 40+ mm SL, mt)			Stock abundance (January 1, 40+ mm SL, millions)		
	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock
1975	--	--	--	--	0.042	--	--	0.042	--
1976	--	0.116	--	--	0.043	--	--	0.037	--
1977	--	0.137	--	--	0.037	--	--	0.034	--
1978	--	0.169	--	--	0.032	--	--	0.031	--
1979	--	0.165	--	--	0.033	--	--	0.031	--
1980	--	0.097	--	--	0.037	--	--	0.033	--
1981	--	0.125	--	--	0.040	--	--	0.036	--
1982	--	0.148	--	0.030	0.037	0.024	0.030	0.037	0.025
1983	0.173	0.126	0.105	0.042	0.039	0.029	0.038	0.039	0.029
1984	0.145	0.143	0.102	0.048	0.045	0.033	0.043	0.051	0.033
1985	0.163	0.111	0.092	0.051	0.050	0.036	0.051	0.054	0.038
1986	0.119	0.151	0.095	0.047	0.046	0.033	0.048	0.044	0.032
1987	0.151	0.113	0.091	0.044	0.039	0.029	0.044	0.038	0.029
1988	0.189	0.152	0.119	0.044	0.039	0.030	0.046	0.038	0.030
1989	0.125	0.112	0.086	0.042	0.038	0.028	0.047	0.035	0.029
1990	0.159	0.137	0.116	0.035	0.036	0.025	0.046	0.036	0.030
1991	0.092	0.147	0.080	0.035	0.037	0.025	0.039	0.035	0.027
1992	0.185	0.137	0.117	0.037	0.038	0.027	0.038	0.038	0.028
1993	0.109	0.059	0.052	0.040	0.040	0.028	0.039	0.036	0.027
1994	0.123	0.090	0.073	0.044	0.034	0.027	0.042	0.032	0.026
1995	0.071	0.128	0.063	0.041	0.036	0.027	0.034	0.032	0.023
1996	0.113	0.200	0.098	0.037	0.036	0.026	0.033	0.035	0.024
1997	0.101	0.108	0.074	0.037	0.043	0.029	0.036	0.048	0.029
1998	0.080	0.059	0.048	0.039	0.039	0.028	0.038	0.036	0.026
1999	0.098	0.081	0.064	0.040	0.030	0.026	0.041	0.029	0.024
2000	0.069	0.101	0.059	0.044	0.028	0.025	0.042	0.027	0.023
2001	0.210	0.102	0.093	0.047	0.027	0.026	0.046	0.026	0.024
2002	0.181	0.122	0.102	0.051	0.028	0.029	0.050	0.028	0.026
2003	0.102	0.061	0.053	0.056	0.030	0.032	0.055	0.034	0.029
2004	0.183	0.240	0.155	0.061	0.036	0.035	0.061	0.039	0.033
2005	0.134	0.088	0.075	0.065	0.047	0.041	0.065	0.047	0.038
2006	0.226	0.256	0.181	0.075	0.057	0.047	0.074	0.056	0.045

Table B5-6 continued

Year	Catch numbers (all sizes, millions)			Fishable mean abundance (all sizes, millions)			Fully recruited fishing mortality (F, annual)		
	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock	Georges Bank	Mid- Atlantic Bight	Whole stock
1975	--	0.050	--	--	0.046	--	--	0.065	--
1976	--	0.050	--	--	0.055	--	--	0.069	--
1977	--	0.050	--	--	0.058	--	--	0.065	--
1978	--	0.050	--	--	0.045	--	--	0.055	--
1979	--	0.050	--	--	0.039	--	--	0.056	--
1980	--	0.050	--	--	0.043	--	--	0.062	--
1981	--	0.050	--	--	0.046	--	--	0.067	--
1982	0.050	0.050	0.041	0.072	0.044	0.038	0.070	0.065	0.190
1983	0.050	0.050	0.040	0.056	0.050	0.040	0.077	0.066	0.103
1984	0.050	0.050	0.036	0.060	0.068	0.046	0.082	0.084	0.138
1985	0.050	0.050	0.036	0.068	0.094	0.055	0.087	0.112	0.178
1986	0.050	0.050	0.035	0.083	0.082	0.058	0.099	0.098	0.139
1987	0.050	0.050	0.037	0.070	0.077	0.052	0.089	0.095	0.138
1988	0.050	0.050	0.036	0.065	0.065	0.046	0.082	0.087	0.121
1989	0.050	0.050	0.037	0.074	0.060	0.047	0.090	0.080	0.118
1990	0.050	0.050	0.035	0.065	0.048	0.040	0.075	0.068	0.103
1991	0.050	0.050	0.036	0.057	0.044	0.035	0.060	0.056	0.084
1992	0.050	0.050	0.037	0.051	0.051	0.036	0.050	0.063	0.078
1993	0.050	0.050	0.036	0.054	0.070	0.045	0.060	0.089	0.102
1994	0.050	0.050	0.044	0.061	0.063	0.046	0.076	0.078	0.108
1995	0.050	0.050	0.045	0.054	0.047	0.035	0.072	0.064	0.095
1996	0.050	0.050	0.039	0.066	0.043	0.037	0.079	0.062	0.093
1997	0.050	0.050	0.035	0.063	0.071	0.047	0.086	0.084	0.131
1998	0.050	0.050	0.036	0.074	0.078	0.055	0.098	0.094	0.158
1999	0.050	0.050	0.036	0.082	0.079	0.057	0.084	0.091	0.130
2000	0.050	0.050	0.040	0.089	0.065	0.053	0.091	0.077	0.111
2001	0.050	0.050	0.042	0.092	0.052	0.048	0.093	0.065	0.097
2002	0.050	0.050	0.040	0.180	0.071	0.089	0.189	0.078	0.146
2003	0.050	0.050	0.041	0.188	0.067	0.092	0.203	0.074	0.141
2004	0.050	0.050	0.044	0.082	0.070	0.055	0.091	0.075	0.147
2005	0.050	0.050	0.039	0.083	0.082	0.058	0.093	0.091	0.139
2006	0.050	0.050	0.035	0.098	0.076	0.060	0.109	0.092	0.164

Table B5-7. Configuration of basecase CASA models for sea scallops in the Mid-Atlantic Bight and Georges Bank.

Category	Mid-Atlantic Bight (MAB)	Georges Bank (GBK)	Explanation
Model configuration			
Modeled years	1975-2006	1982-2006	Models start in first year with dredge survey data; Unlined dredge survey for MAB starts in 1975; Lined dredge survey for GBK with complete coverage starts 1982
L-infinity (mm SH)	131.6	147	New growth studies based on shell increments (used only to define plus group in model)
Population shell height bins (mm SH)	5	5	Same as resolution of shell height composition data for lined dredge survey
First size in model (mm SH)	20	20	Biomass and fishing mortality estimates are for 40+ mm SH; Starting model at 20 mm SH allows for 20 mm of "burn-in" before recruits enter size groups (40+ mm SH) used to calculate biomass and fishing mortality
Size range new recruits (mm)	20-69.9 (shell height size groups 1-10)	20-69.9 (shell height size groups 1-10)	Based on inspection of trawl survey shell height composition data for years with strong recruitment; model estimates a unimodal beta distribution with two parameters to represent the shell height distribution for new recruits across this range of sizes
Population dynamics			
Stock shell height/meat weight parameters	a = -10.7; b = 2.942	a = -12.2484; b = 3.2641	Same as previous assessment; based on dredge survey data collected during June
Incidental mortality multiplier	0.04	0.15	Total mortality of all size groups is increased by this proportion of full recruited F; based on published studies although available information is limited and uncertain; assumed same for all size groups
Population shell height composition in 1st year	1975 unlined dredge survey, adjusted for selectivity	1982 lined dredge survey, no adjustments	Not estimable in model
Natural mortality rate (y-1)	0.1	0.1	Same for all years and shell height groups
Growth matrix method	Observed increments (n=2,244)	Observed increments (n=2,692)	Observed increment from each are used to form growth transition matrix; Different increment data sets and transition matrices for GBK and MAB
Size range used to summarize stock biomass and mortality rates	40-130+	40-145+	Survey data reliable for 40+ mm SH
Fecundity at size			Annual fecundity at size parameters from McGarvey et al. (1992)

Table B5-7 continued.

Commercial			
Fishery shell height/meat weight parameters	1992-2006	1975-2006 1982-1995 (logistic); 1996-1998 (domed); 1999-2001 (logistic); 2002-2003 (domed); 2004-2006 (logistic)	Population shell height/meat weight are modified by annual anomalies based on observer data and seasonal landings patterns to account for seasonal patterns in shell height/meat weight relationships, meat loss during shucking and absorption of water during storage Predicted values are a nonlinear function of abundance and biomass
Commercial LPUE			
Fishery selectivity periods	1975-1981; 1982-1995; 1997-2001; 2002-2006 (all logistic)		Logistic patterns used for MAB in all years and GBK when fishery had access to entire scallop size range; Dome (double logistic) pattern used for GBK when substantial numbers of large scallops were in closed areas not open to fishing
Shell height-meat weight			The average weight of scallops in the commercial catch was calculated in the model based on predicted shell heights and survey shell-height meat weight relationships, with adjustments based on annual mean meat weight anomalies.
Survey trends			
NEFSC lined dredge	1979-2006	1982-2006	Likelihood calculations use CV for stratified random means; 40+ mm SH; Flat selectivity pattern (not estimated)
SMAST video (small camera)	2003-2006	2003-2006	Likelihood calculations use CV for stratified random means; 40+ mm SH; Flat selectivity pattern; Assumed 100% efficiency; Densities (N/m ²) converted to swept area biomass based on area covered by dredge survey, then multiplied by 0.5 for use with prior (see below)
Winter bottom trawl	1992-2006	1992-2006	Likelihood calculated based on internally estimated variance; 40+ mm SH; Logistic selectivity pattern
Fall and spring bottom trawl	1964-2006 (fall); 1968-2006 (spring)	1964-2006 (fall); 1968-2006 (spring)	Less reliable than dredge or video surveys (particularly for GBK); Used only for comparison, did not affect model estimates; Likelihood calculated based on internally estimated variance; 40+ mm SH; Domed (double logistic) selectivity pattern
10 ft unlined dredge	1975-1976, 1978	NA	Likelihood calculations use CV for stratified random means; 40+ mm SH; Logistic selectivity pattern fixed per reanalysis of paired tow experiments in Serchuk and Smolowicz (1980) using Millars (1980) SELECT model

Table B5-7 continued.

Shell height composition data	All years with trend data	All years with trend data	Five mm shell height groups; Assumed effective sample size used in likelihood calculations was adjusted based on preliminary model fits; Assumed standard deviation for shell height measurement errors = 1.6 mm for dredge survey and 6 mm for video survey
Dredge and video survey shell height composition	All years with trend data	All years with trend data	Ten mm shell height groups; Assumed effective sample size used in likelihood calculations was adjusted based on preliminary model fits; Assumed standard deviation for shell height measurement errors = 1.6 mm
Bottom trawl shell height composition	1969-1984 (port samples); 1992-2006 (observers)	1982-1984 (port samples); 1992-2006 (observers)	Five mm shell height groups; Assumed effective sample size used in likelihood calculations was adjusted based on preliminary model fits; Assumed standard deviation for shell height measurement errors = 6 mm
Commercial shell height composition	Beta distribution with mean 0.5 and CV=20%	Beta distribution with mean 0.5 and CV=20%	Video survey trend data were scaled up to swept area biomass and then multiplied by 0.5 before use in model assuming 100% sampling efficiency, a flat survey selectivity pattern and areas of grounds covered in dredge survey (see above); Under these conditions, the expected survey scaling parameter is 0.5; 20% CV is ad-hoc but based on standard deviation of eight video-dredge survey comparisons and meant to imply a moderate level of uncertainty
Prior information			
Prior on SMAST survey scaling parameter (catchability coefficient)			

Table B5-8. Parameters, standard errors, and CVs estimated in the basecase model for sea scallops on Georges Bank during 1982-2006.

ID	Parameter	Estimate	Standard error	CV
1	Log N first year	21.3870	0.0299	0.00
2	Log beta distribution parameter for shell height distribution of new recruits	-0.6388	0.4253	0.67
3	Log beta distribution parameter for shell height distribution of new recruits	-1.9992	0.6445	0.32
4	Log mean recruitment	20.0020	0.0306	0.00
5	Log survey scaling parameter for lined dredge survey	-2.4385	0.0517	0.02
6	Log survey scaling parameter for video small camera survey	-0.4984	0.0795	0.16
7	Log mean F	-0.8534	0.0365	0.04
8	Log LPUE scaling parameter	-2.9814	0.0624	0.02
9	Log LPUE shape parameter	-2.8181	1.3191	0.47
10	Log fishery selectivity parameter 1982-1995 (logistic)	3.5776	0.1877	0.05
11	Log fishery selectivity parameter 1982-1995 (logistic)	-0.7623	0.1916	0.25
12	Log fishery selectivity parameter 1996-1998 (domed, double logistic)	3.1260	0.0973	0.03
13	Log fishery selectivity parameter 1996-1998 (domed, double logistic)	-1.3862	0.1090	0.08
14	Log fishery selectivity parameter 1996-1998 (domed, double logistic)	-4.9991	2.8192	0.56
15	Log fishery selectivity parameter 1996-1998 (domed, double logistic)	-3.1237	0.1567	0.05
16	Log fishery selectivity parameter 1999-2001 (logistic)	2.8067	0.1696	0.06
17	Log fishery selectivity parameter 1999-2001 (logistic)	-1.7953	0.1876	0.10
18	Log fishery selectivity parameter 2002-2003 (domed, double logistic)	4.0713	0.6400	0.16
19	Log fishery selectivity parameter 2002-2003 (domed, double logistic)	-0.5435	0.6571	1.21
20	Log fishery selectivity parameter 2002-2003 (domed, double logistic)	2.6844	0.5917	0.22
21	Log fishery selectivity parameter 2002-2003 (domed, double logistic)	-2.1137	0.4999	0.24
22	Log fishery selectivity parameter 2004-2006 (logistic)	3.2284	0.2959	0.09
23	Log fishery selectivity parameter 2004-2006 (logistic)	-1.4596	0.3071	0.21
24	Log F deviation 1982	1.7063	0.0736	0.04
25	Log F deviation 1983	0.7985	0.0754	0.09
26	Log F deviation 1984	-0.0893	0.0781	0.87
27	Log F deviation 1985	0.0146	0.0807	5.53
28	Log F deviation 1986	0.4978	0.0903	0.18
29	Log F deviation 1987	0.3929	0.0815	0.21
30	Log F deviation 1988	0.4486	0.0771	0.17
31	Log F deviation 1989	0.3733	0.0858	0.23
32	Log F deviation 1990	0.9088	0.0749	0.08
33	Log F deviation 1991	1.1395	0.0655	0.06
34	Log F deviation 1992	1.3323	0.0580	0.04
35	Log F deviation 1993	1.0368	0.0629	0.06
36	Log F deviation 1994	-0.2955	0.0744	0.25
37	Log F deviation 1995	-0.9403	0.0707	0.08
38	Log F deviation 1996	-0.1595	0.0755	0.47

Table B5-8 continued.

39	Log F deviation 1997	-0.3299	0.0809	0.25
40	Log F deviation 1998	-0.5921	0.0927	0.16
41	Log F deviation 1999	-0.3259	0.0753	0.23
42	Log F deviation 2000	-0.6465	0.0804	0.12
43	Log F deviation 2001	-0.8802	0.0815	0.09
44	Log F deviation 2002	-0.6432	0.1731	0.27
45	Log F deviation 2003	-0.7901	0.1868	0.24
46	Log F deviation 2004	-1.6484	0.0768	0.05
47	Log F deviation 2005	-0.9907	0.0780	0.08
48	Log F deviation 2006	-0.3177	0.0921	0.29
49	Log recruitment deviation 1983	-0.9893	0.1688	0.17
50	Log recruitment deviation 1984	-0.5931	0.1437	0.24
51	Log recruitment deviation 1985	-0.2772	0.1613	0.58
52	Log recruitment deviation 1986	0.5302	0.1225	0.23
53	Log recruitment deviation 1987	0.2751	0.1528	0.56
54	Log recruitment deviation 1988	-0.0176	0.1863	10.57
55	Log recruitment deviation 1989	0.8262	0.1300	0.16
56	Log recruitment deviation 1990	0.5687	0.1566	0.28
57	Log recruitment deviation 1991	0.7252	0.0985	0.14
58	Log recruitment deviation 1992	-0.6924	0.1815	0.26
59	Log recruitment deviation 1993	-0.4355	0.1112	0.26
60	Log recruitment deviation 1994	-0.6060	0.1202	0.20
61	Log recruitment deviation 1995	0.3021	0.0732	0.24
62	Log recruitment deviation 1996	-0.3227	0.1121	0.35
63	Log recruitment deviation 1997	-0.1527	0.0973	0.64
64	Log recruitment deviation 1998	0.4361	0.0763	0.17
65	Log recruitment deviation 1999	0.4364	0.0920	0.21
66	Log recruitment deviation 2000	1.3361	0.0640	0.05
67	Log recruitment deviation 2001	-0.0337	0.2072	6.14
68	Log recruitment deviation 2002	-0.2821	0.1729	0.61
69	Log recruitment deviation 2003	0.4255	0.0946	0.22
70	Log recruitment deviation 2004	-0.6168	0.1725	0.28
71	Log recruitment deviation 2005	-0.0713	0.1244	1.75
72	Log recruitment deviation 2006	-0.7711	0.2133	0.28

Table B5-9. Parameters, standard errors, and CVs estimated in the basecase model for sea scallop in the Mid-Atlantic Bight during 1975-2006.

ID	Parameter	Estimate	Standard error	CV
1	Log N first year	20.1720	0.0420	0.00
2	Log beta distribution parameter for shell height distribution of new recruits	0.4947	0.1284	0.26
3	Log beta distribution parameter for shell height distribution of new recruits	0.1924	0.1167	0.61
4	Log mean recruitment	19.9660	0.0197	0.00
5	Log survey scaling parameter for lined dredge survey	-2.4483	0.0261	0.01
6	Log survey scaling parameter for video small camera survey	-0.4457	0.0686	0.15
7	Log survey scaling parameter for winter bottom trawl survey	-3.8376	0.0878	0.02
8	Log survey scaling parameter for unlined dredge survey	-1.6114	0.1289	0.08
9	Log selectivity parameter for winter bottom trawl survey (logistic)	2.1731	0.2215	0.10
10	Log selectivity parameter for winter bottom trawl survey (logistic)	-1.8471	0.2392	0.13
11	Log mean F	-0.6134	0.0266	0.04
12	Log LPUE scaling parameter	-2.4049	0.1715	0.07
13	Log LPUE shape parameter	0.0779	0.2555	3.28
14	Log fishery selectivity parameter 1975-1981 (logistic)	2.9808	0.2115	0.07
15	Log fishery selectivity parameter 1975-1981 (logistic)	-1.2975	0.2258	0.17
16	Log fishery selectivity parameter 1982-1995 (logistic)	3.1507	0.1119	0.04
17	Log fishery selectivity parameter 1982-1995 (logistic)	-1.2083	0.1189	0.10
18	Log fishery selectivity parameter 1997-2001 (logistic)	3.0164	0.0871	0.03
19	Log fishery selectivity parameter 1997-2001 (logistic)	-1.5086	0.0943	0.06
20	Log fishery selectivity parameter 2002-2006 (logistic)	3.0759	0.1205	0.04
21	Log fishery selectivity parameter 2002-2006 (logistic)	-1.5409	0.1304	0.08
22	Log F deviation 1975	-1.1534	0.0661	0.06
23	Log F deviation 1976	-0.3152	0.0693	0.22
24	Log F deviation 1977	-0.5374	0.0645	0.12
25	Log F deviation 1978	-0.0289	0.0552	1.91
26	Log F deviation 1979	-0.2376	0.0554	0.23
27	Log F deviation 1980	-0.4680	0.0593	0.13
28	Log F deviation 1981	-1.4691	0.0629	0.04
29	Log F deviation 1982	-0.7775	0.0606	0.08
30	Log F deviation 1983	-0.0312	0.0614	1.97
31	Log F deviation 1984	0.4141	0.0758	0.18
32	Log F deviation 1985	0.5184	0.1025	0.20
33	Log F deviation 1986	0.1876	0.0892	0.48
34	Log F deviation 1987	0.7921	0.0891	0.11
35	Log F deviation 1988	0.4431	0.0816	0.18
36	Log F deviation 1989	0.6315	0.0766	0.12
37	Log F deviation 1990	0.4471	0.0672	0.15
38	Log F deviation 1991	0.5999	0.0563	0.09
39	Log F deviation 1992	0.5641	0.0610	0.11
40	Log F deviation 1993	0.2510	0.0852	0.34
41	Log F deviation 1994	0.7407	0.0745	0.10
42	Log F deviation 1995	0.5324	0.0618	0.12
43	Log F deviation 1996	0.3007	0.0590	0.20
44	Log F deviation 1997	-0.0719	0.0769	1.07
45	Log F deviation 1998	-0.0603	0.0864	1.43

Table B5-9 continued

46	Log F deviation 1999	-0.1305	0.0835	0.64
47	Log F deviation 2000	-0.1924	0.0721	0.37
48	Log F deviation 2001	-0.0511	0.0617	1.21
49	Log F deviation 2002	0.0952	0.0735	0.77
50	Log F deviation 2003	0.1194	0.0690	0.58
51	Log F deviation 2004	0.3004	0.0692	0.23
52	Log F deviation 2005	-0.2832	0.0837	0.30
53	Log F deviation 2006	-1.1299	0.0860	0.08
54	Log recruitment deviation 1976	-0.1280	0.1159	0.91
55	Log recruitment deviation 1977	-0.6291	0.1383	0.22
56	Log recruitment deviation 1978	-1.4576	0.1678	0.12
57	Log recruitment deviation 1979	-1.8824	0.1627	0.09
58	Log recruitment deviation 1980	-1.0493	0.0984	0.09
59	Log recruitment deviation 1981	-1.2355	0.1246	0.10
60	Log recruitment deviation 1982	-1.2455	0.1465	0.12
61	Log recruitment deviation 1983	-0.8155	0.1268	0.16
62	Log recruitment deviation 1984	-0.4310	0.1423	0.33
63	Log recruitment deviation 1985	0.5085	0.1145	0.23
64	Log recruitment deviation 1986	0.4442	0.1509	0.34
65	Log recruitment deviation 1987	0.7404	0.1151	0.16
66	Log recruitment deviation 1988	0.3209	0.1502	0.47
67	Log recruitment deviation 1989	0.5787	0.1134	0.20
68	Log recruitment deviation 1990	-0.1046	0.1367	1.31
69	Log recruitment deviation 1991	-0.7094	0.1461	0.21
70	Log recruitment deviation 1992	-0.7706	0.1345	0.17
71	Log recruitment deviation 1993	0.8925	0.0623	0.07
72	Log recruitment deviation 1994	0.3743	0.0900	0.24
73	Log recruitment deviation 1995	-0.4386	0.1283	0.29
74	Log recruitment deviation 1996	-1.5190	0.1952	0.13
75	Log recruitment deviation 1997	0.0643	0.1067	1.66
76	Log recruitment deviation 1998	1.4737	0.0627	0.04
77	Log recruitment deviation 1999	1.2844	0.0826	0.06
78	Log recruitment deviation 2000	1.1293	0.1015	0.09
79	Log recruitment deviation 2001	1.1245	0.1030	0.09
80	Log recruitment deviation 2002	0.8712	0.1189	0.14
81	Log recruitment deviation 2003	1.9235	0.0609	0.03
82	Log recruitment deviation 2004	-0.4066	0.2325	0.57
83	Log recruitment deviation 2005	1.3312	0.0861	0.06
84	Log recruitment deviation 2006	-0.2387	0.2479	1.04

Table B5-10. Likelihood profile analysis for the basecase Georges Bank CASA model. For ease of interpretation, the likelihood for each type of data and constraint in the basecase model was subtracted from the likelihoods for the same type of data or constraint at each point.

Data type or constraint	Survey scaling parameter for small camera video survey								
	0.31	0.41	0.51	0.60	0.61	0.70	0.80	0.90	0.98
Fishing mortality 2006	0.18	0.22	0.27	0.31	0.31	0.35	0.39	0.43	0.46
Stock biomass 2006 (mt)	132,919	108,337	92,633	81,798	81,088	73,374	66,930	61,843	58,486
Neg. log likelihood for major components	0.07	-0.07	-0.07	-0.01	0.00	0.09	0.21	0.34	0.46
<i>Commercial catch weight</i>	-0.05	-0.03	-0.02	0.00	0.00	0.01	0.03	0.04	0.05
<i>Commercial LPUE</i>	0.24	-0.30	-0.29	-0.03	0.00	0.40	0.89	1.43	1.86
<i>Recruitment deviations</i>	1.18	-0.15	-0.52	-0.07	0.00	1.39	4.36	10.62	26.79
<i>Efficiency prior - camera survey</i>	20.13	6.98	1.39	-0.01	0.00	1.21	4.24	8.52	12.59
<i>Total for survey trends</i>	8.07	4.56	2.02	0.13	0.00	-1.40	-2.58	-3.49	-4.08
<i>Total for shell height composition</i>	29.63	10.98	2.51	0.01	0.00	1.70	7.16	17.46	37.68
Total major components (unweighted)									
Survey trends									
<i>Dredge survey</i>	12.37	4.21	0.84	0.00	0.00	0.63	2.19	4.32	6.28
<i>Camera survey</i>	7.76	2.77	0.55	-0.01	0.00	0.58	2.05	4.20	6.31
Shell height composition data									
<i>Commercial fishery</i>	1.09	0.65	0.30	0.02	0.00	-0.22	-0.43	-0.61	-0.73
<i>Dredge survey</i>	6.54	3.66	1.61	0.10	0.00	-1.11	-2.03	-2.74	-3.19
<i>Camera survey</i>	0.43	0.24	0.11	0.01	0.00	-0.07	-0.12	-0.15	-0.16

Table B5-11. Likelihood profile analysis for the basecase Mid-Atlantic Bight CASA model. For ease of interpretation, the likelihood for each type of data and constraint in the basecase model was subtracted from the likelihoods for the same type of data or constraint at each point. The table includes likelihoods for NEFSC fall and winter bottom trawl surveys which were included for information only and did not affect model estimates.

Data type or constraint	Survey scaling parameter for small camera video survey									
	0.30	0.40	0.50	0.60	Basecase 0.64	0.70	0.80	0.90	1.00	
Fishing mortality 2006	0.34	0.38	0.41	0.43	0.44	0.45	0.47	0.49	0.50	
Stock biomass 2006 (mt)	107,515	96,247	88,483	82,709	80,771	78,193	74,534	71,490	68,905	
Neg. log likelihood for major components										
Commercial catch weight	1.171	0.482	0.155	0.020	0.000	-0.003	0.046	0.141	0.268	
Commercial LPUE	0.37	0.24	0.13	0.03	0.00	-0.05	-0.12	-0.19	-0.25	
Recruitment deviations	5.69	3.30	1.64	0.41	0.00	-0.54	-1.31	-1.93	-2.45	
Efficiency prior - camera survey	0.98	-0.47	-0.90	-0.45	0.00	1.01	3.98	10.23	55.01	
Total for survey trends	50.89	22.20	7.82	1.22	0.00	-0.71	0.36	3.42	7.86	
Total for shell height composition	-7.26	-5.06	-2.88	-0.80	0.00	1.16	3.01	4.75	6.40	
Total major components (unweighted)	51.84	20.70	5.96	0.43	0.00	0.87	5.96	16.41	66.84	
Survey trends used to fit model										
Dredge survey	13.73	5.48	1.65	0.17	0.00	0.07	0.81	2.11	3.78	
Camera survey	32.27	13.93	4.80	0.71	0.00	-0.33	0.59	2.84	6.00	
Winter bottom trawl survey	4.90	2.79	1.37	0.34	0.00	-0.44	-1.05	-1.53	-1.92	
Unlined 10 ft scallop dredge survey	-0.02	-0.01	-0.01	0.00	0.00	0.00	0.01	0.01	0.01	
Survey trends that did not affect model estimates										
Fall bottom trawl survey	15.10	9.01	4.59	1.17	0.00	-1.58	-3.87	-5.81	-7.49	
Spring bottom trawl survey	15.43	9.26	4.74	1.22	0.00	-1.65	-4.04	-6.09	-7.87	
Shell height composition data										
Commercial fishery	-3.36	-2.16	-1.16	-0.31	0.00	0.44	1.10	1.71	2.26	
Dredge survey	0.49	-0.06	-0.18	-0.08	0.00	0.14	0.44	0.77	1.13	
Camera survey	-3.35	-2.17	-1.17	-0.31	0.00	0.44	1.12	1.74	2.31	
Winter bottom trawl survey	-1.16	-0.75	-0.40	-0.11	0.00	0.15	0.38	0.58	0.77	
Unlined 10 ft scallop dredge survey	0.11	0.07	0.03	0.01	0.00	-0.01	-0.03	-0.05	-0.06	

Table B5-12. Sensitivity analysis runs using the CASA model for sea scallops in the Mid-Atlantic Bight.

Scenario	Estimates for 2006	
	Biomass (mt)	Fishing mortality (y^{-1})
Basecase	85,161	0.18
Eliminate prior on video survey efficiency	83,061	0.18
Drop anomalous 2003 dredge and video trend observations	83,520	0.18
No constraint on recruitment variability	88,815	0.17
Use spring and fall bottom trawl surveys	69,440	0.22
Start 1979	85,870	0.17

Table B6-1. Biological reference points and stock status measures for the whole stock of sea scallops from CASA model runs.

Estimate	Value
F_{MSY} proxy (F_{MAX} , y-1)	0.29
Current F (y^{-1})	0.23
BPR at F_{MAX} (g)	86.3
Median 83-06 recruitment (millions)	1,258
B_{MSY} proxy	108,628
Biomass threshold (mt)	54,314
Current biomass (mt)	166,208

ATLANTIC SEA SCALLOP FIGURES

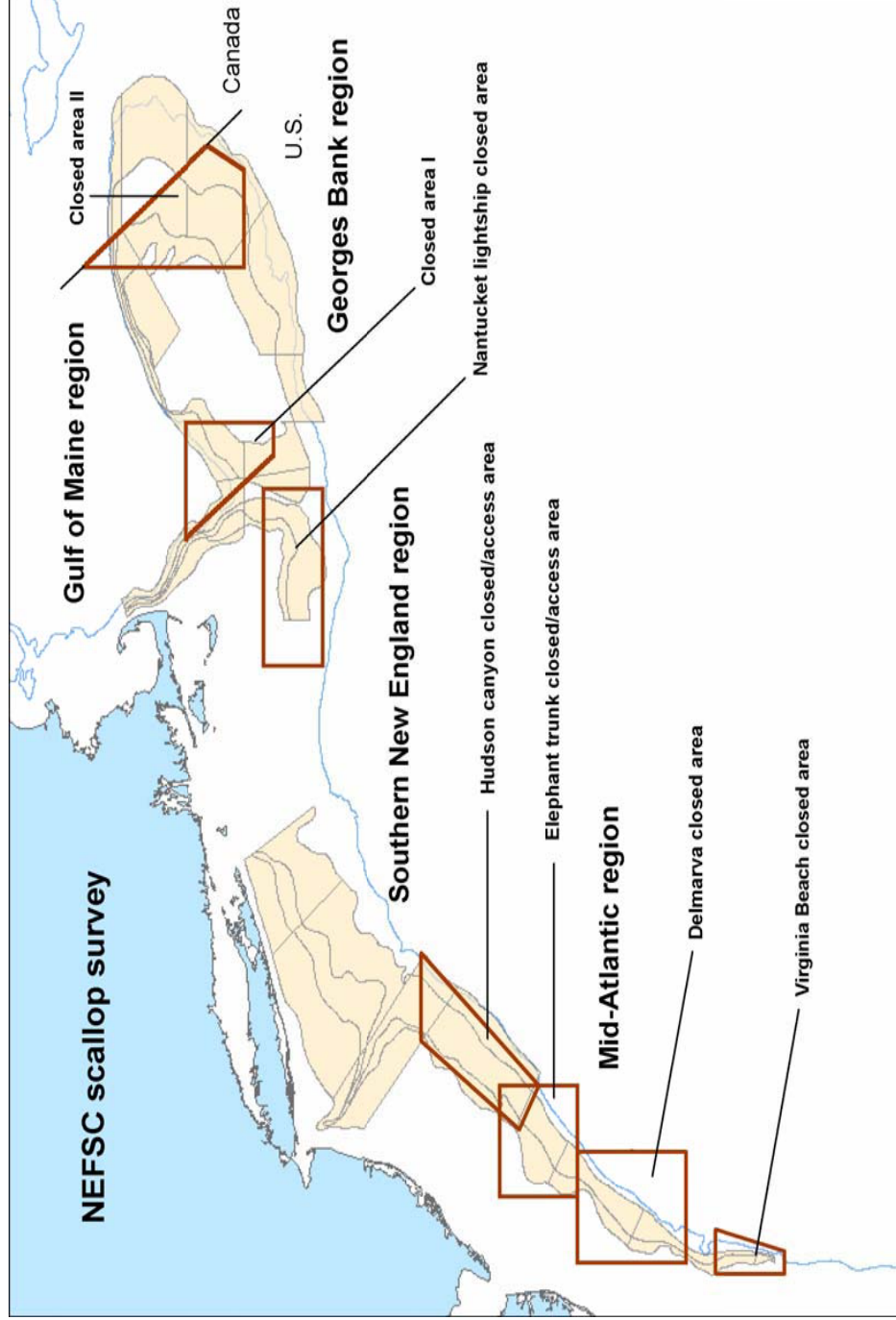


Figure B3-1. Map of sea scallop grounds and regions. The shaded regions are the shellfish strata regularly sampled in the NEFSC sea scallop survey

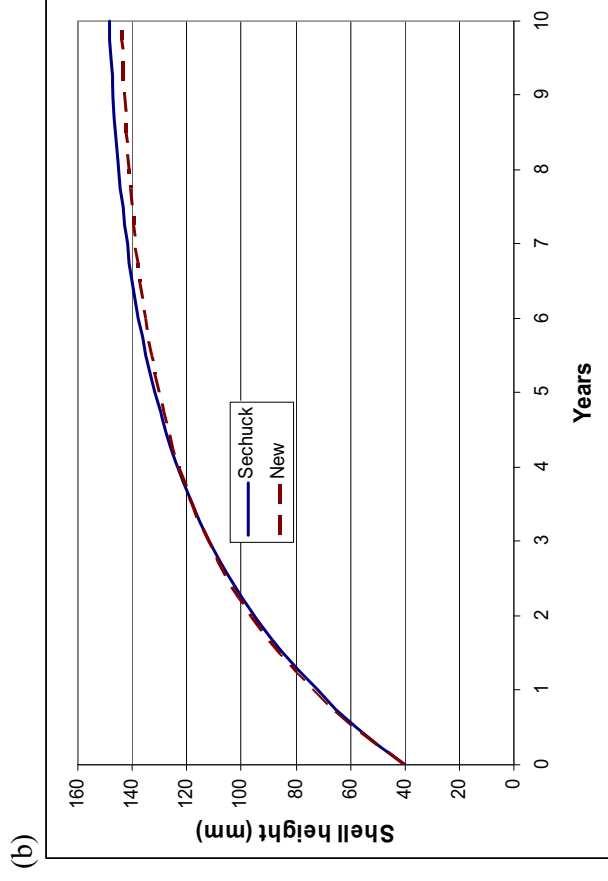
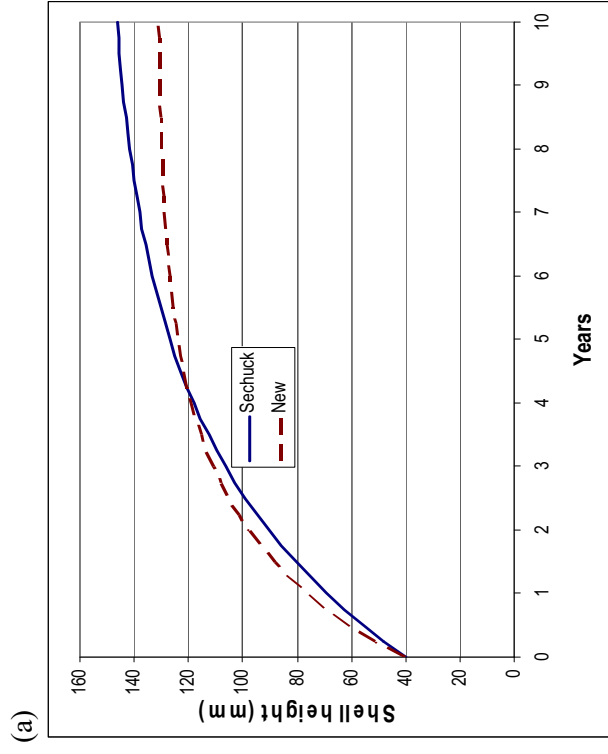


Figure B3-2. Comparison of growth rates for a hypotheticalal sea scallop starting at 40 mm SH based on growth curves from Serchuk et al. (1979) and new growth curves in (a) Mid-Atlantic Bight and (b) Georges Bank.

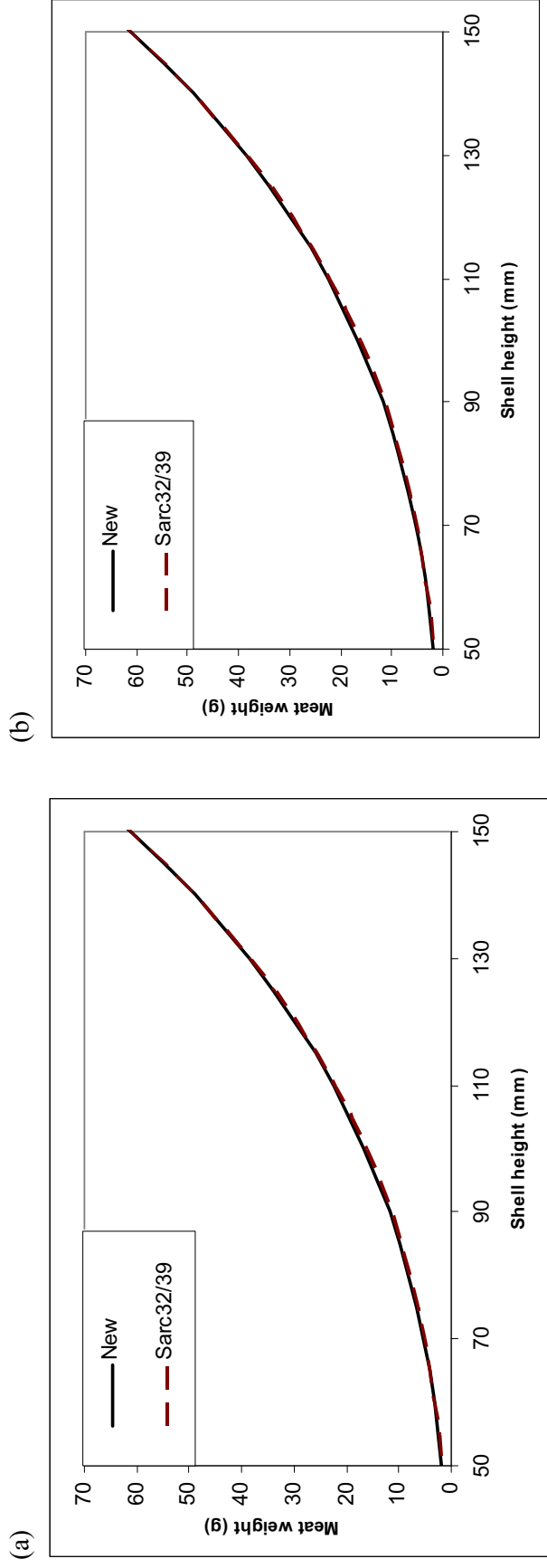


Figure B3-3. Comparison of new shell height/meat weight relationships (calculated ignoring depth effects) for (a) Georges Bank and (b) Mid-Atlantic with relationship used in the previous two assessments (NEFSC 2001, 2004).

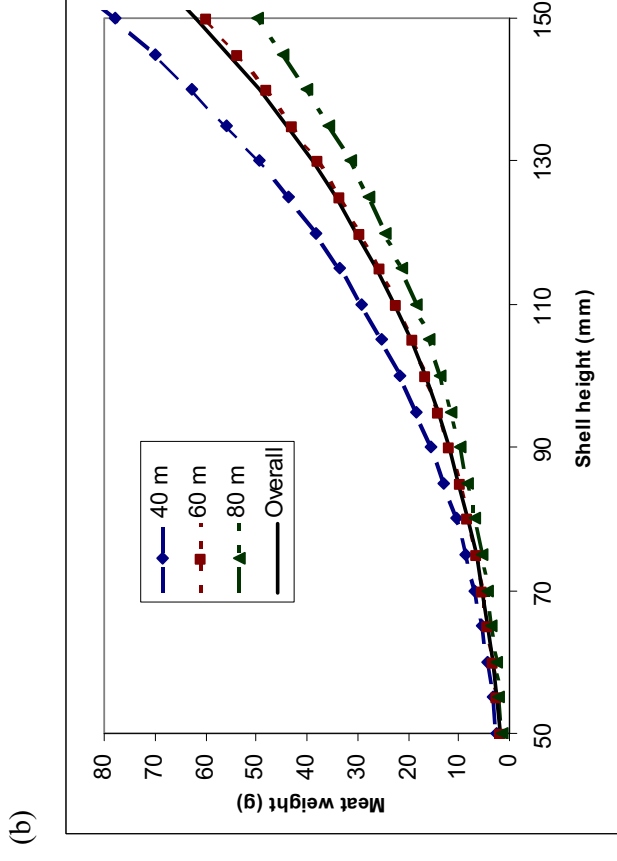
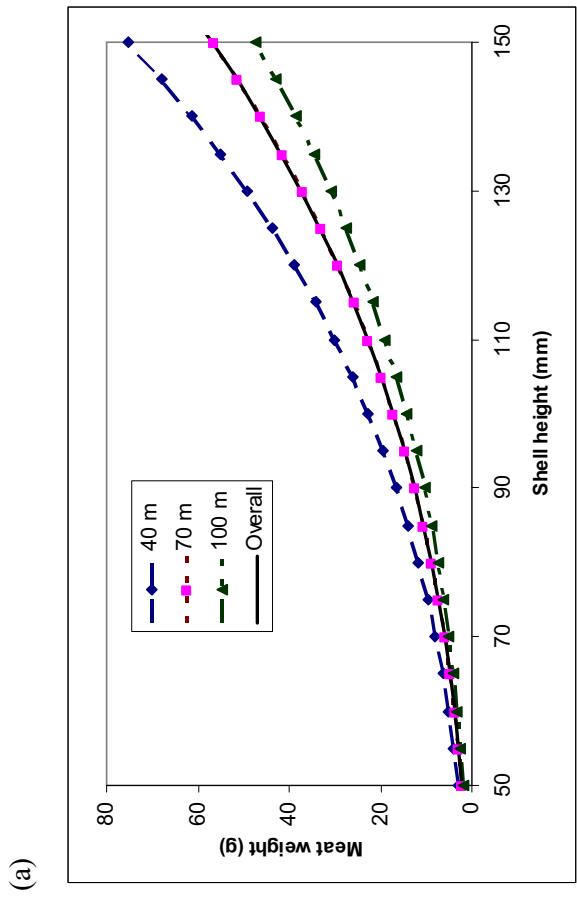


Figure B3-4. Depth effects on shell height/meat weight relationships for sea scallops (a) on Georges Bank and (b) in the Mid-Atlantic.

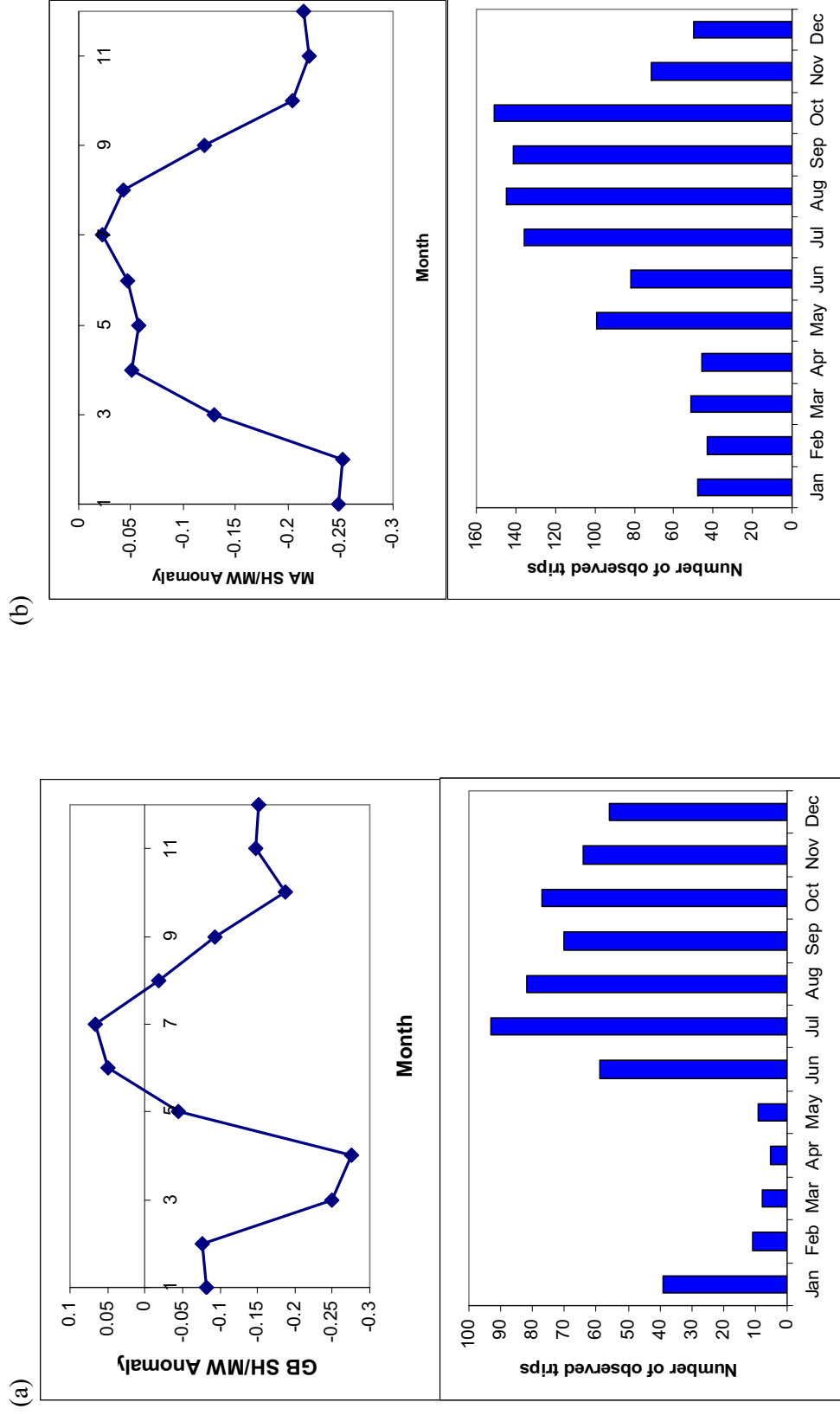


Figure B3-5. Monthly shell height/meat weight anomalies on (a) Georges Bank and (b) in the Mid-Atlantic, together with the number of observed trips by month.

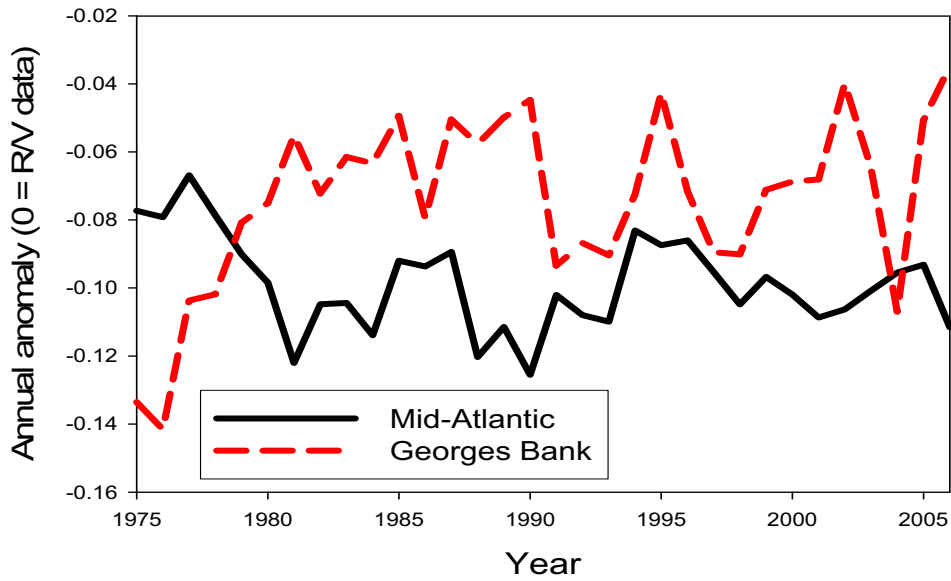


Figure B3-6. Estimated annual shell height/meat weight anomalies for sea scallops on Georges Bank and in the Mid-Atlantic, with no adjustment for water uptake.

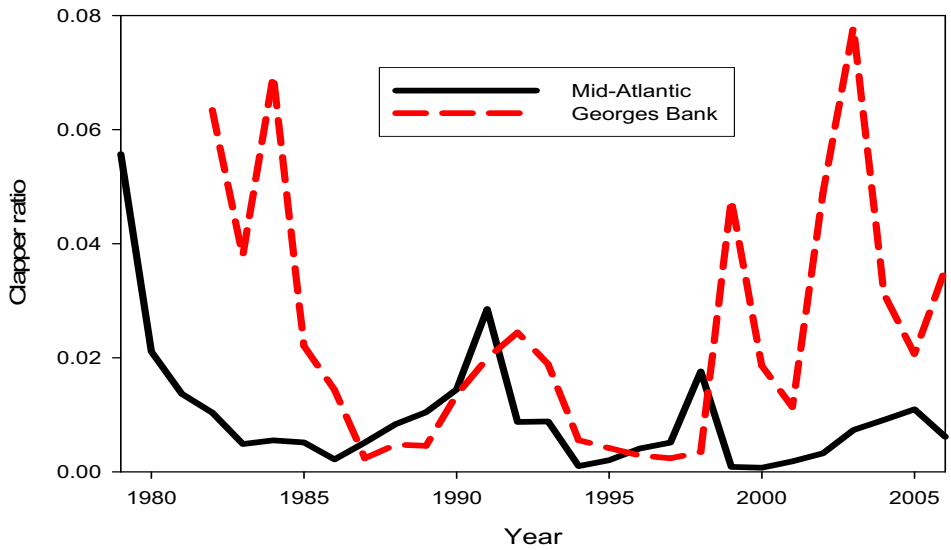


Figure B3-7. Ratio of clapper to live sea scallops in the NEFSC sea scallop survey

	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	
Meat Count																								
Minimum Shell Height																								
DAS Full Time																								
Part Time																								
Occasional																								
Minimum Ring Size																								
Gear																								
Maximum Crew																								
Area Closure																								
Area Reopening																								

Figure B4-1. Summary of management measures from 1984-2006. "mpp" means meats per pound.

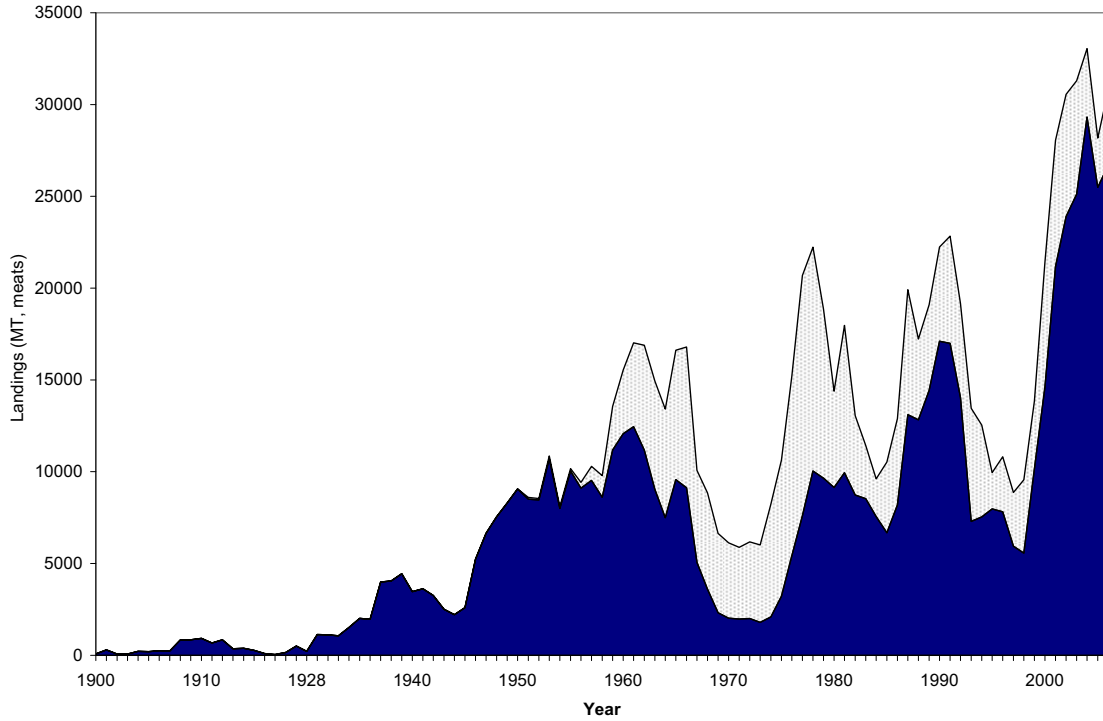


Figure B4-2. Sea scallop landings in NAFO areas 5 and 6 (US plus the Canadian portion of Georges Bank). US landings are shown by dark fill. Canadian landings are shown by light fill.

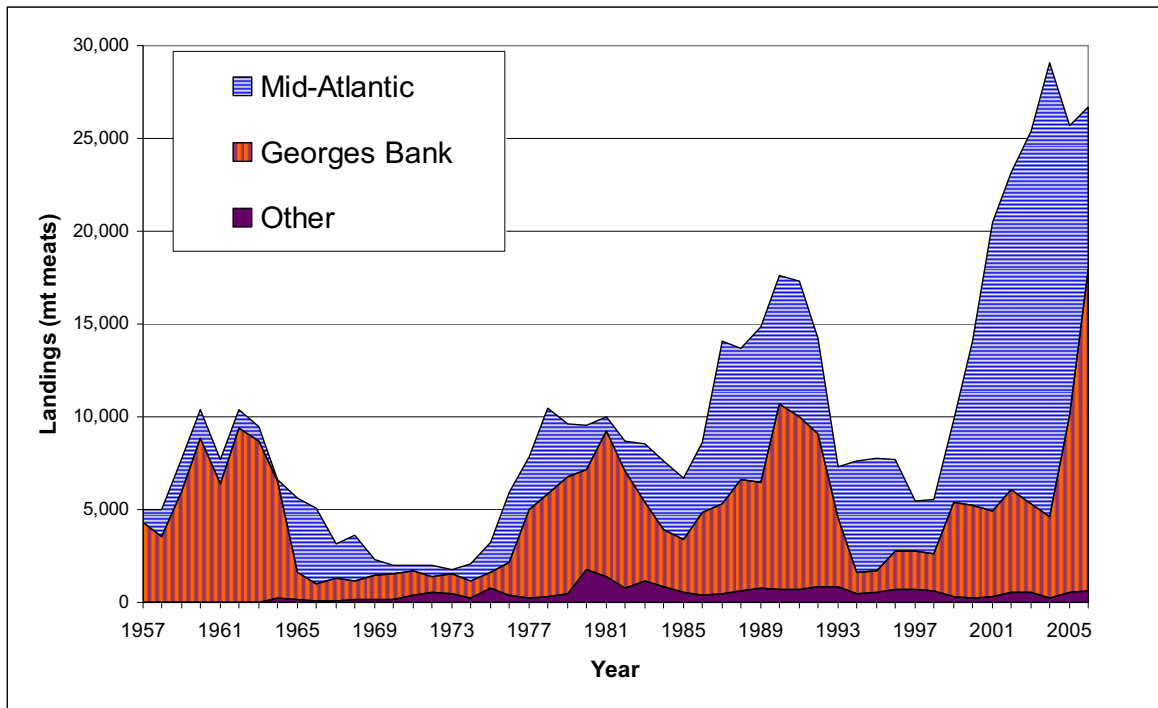


Figure B4-3. US sea scallop landings by area, 1957-2006. “Other” landings (i.e., southern New England and the Gulf of Maine) are not available prior to 1964.

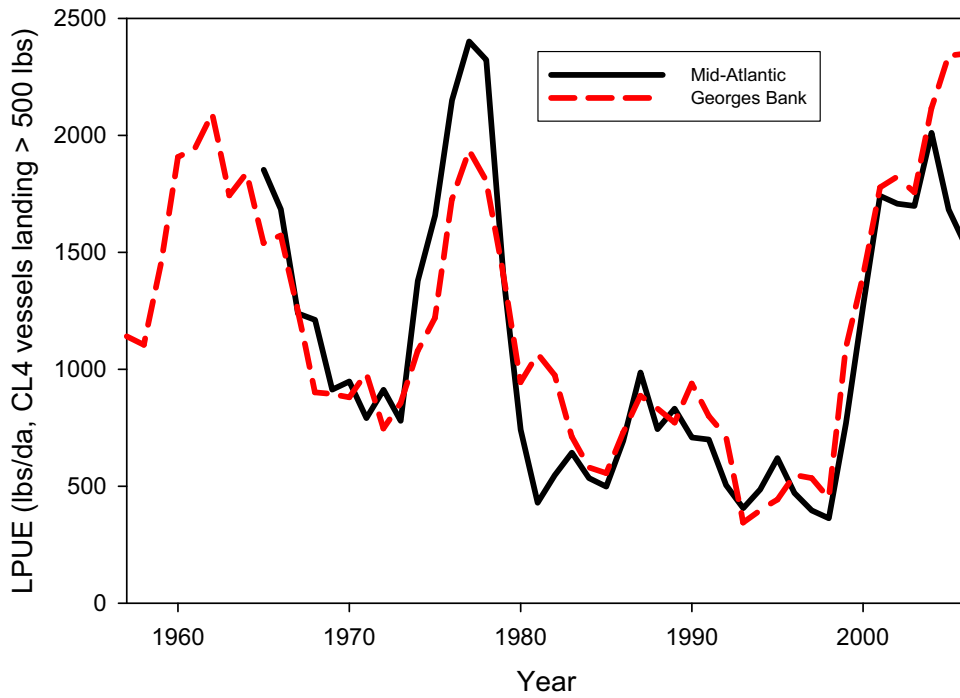


Figure B4-4. Landings per unit effort (lbs meats per days absent for vessels >150 GRT and trips >500 lbs meats) in the sea scallop fishery.

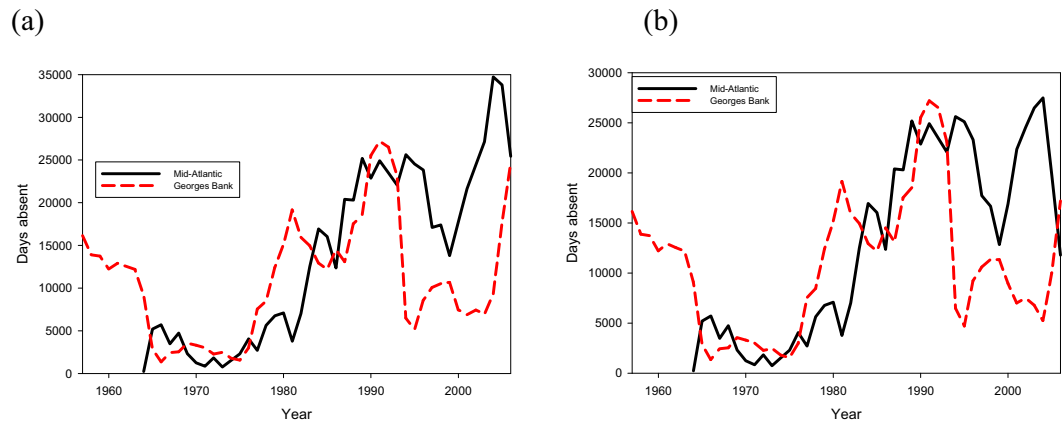


Figure B4-5. Fishing efforts (days absent) for the sea scallops fishery. (a) unadjusted, (b) adjusted for trips landing less than 500 lbs meats.

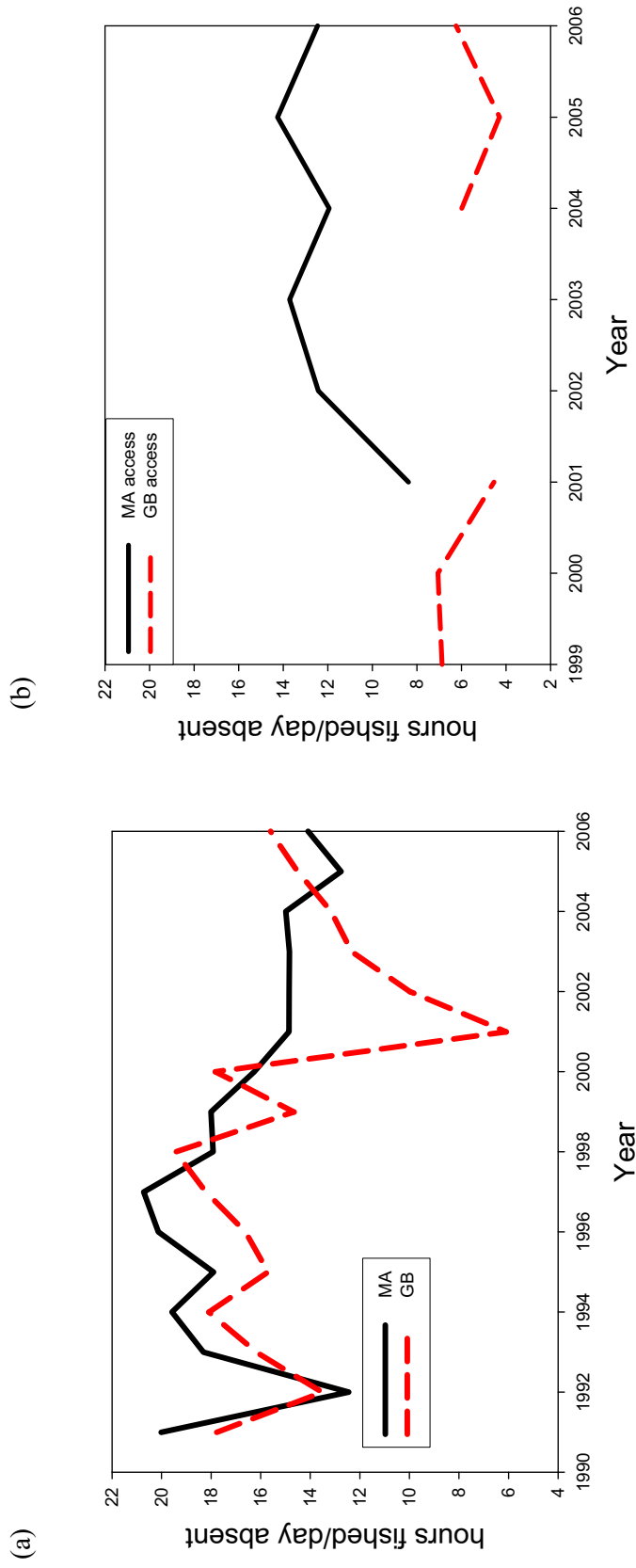


Figure B4-6. Hours fished per day absent in (a) regular trips in open area, and (b) special access area trips in previously closed areas, derived from observer sea sampling data

Figure B4-7. The spatial distribution of fishing effort based on VMS data for the limited access scallop fleet during 1998-2006 (through Sept. 15), and general category vessels during 2006 (through Sept. 15). Effort categories are classified based on 20th percentiles of effort during FY 2000.

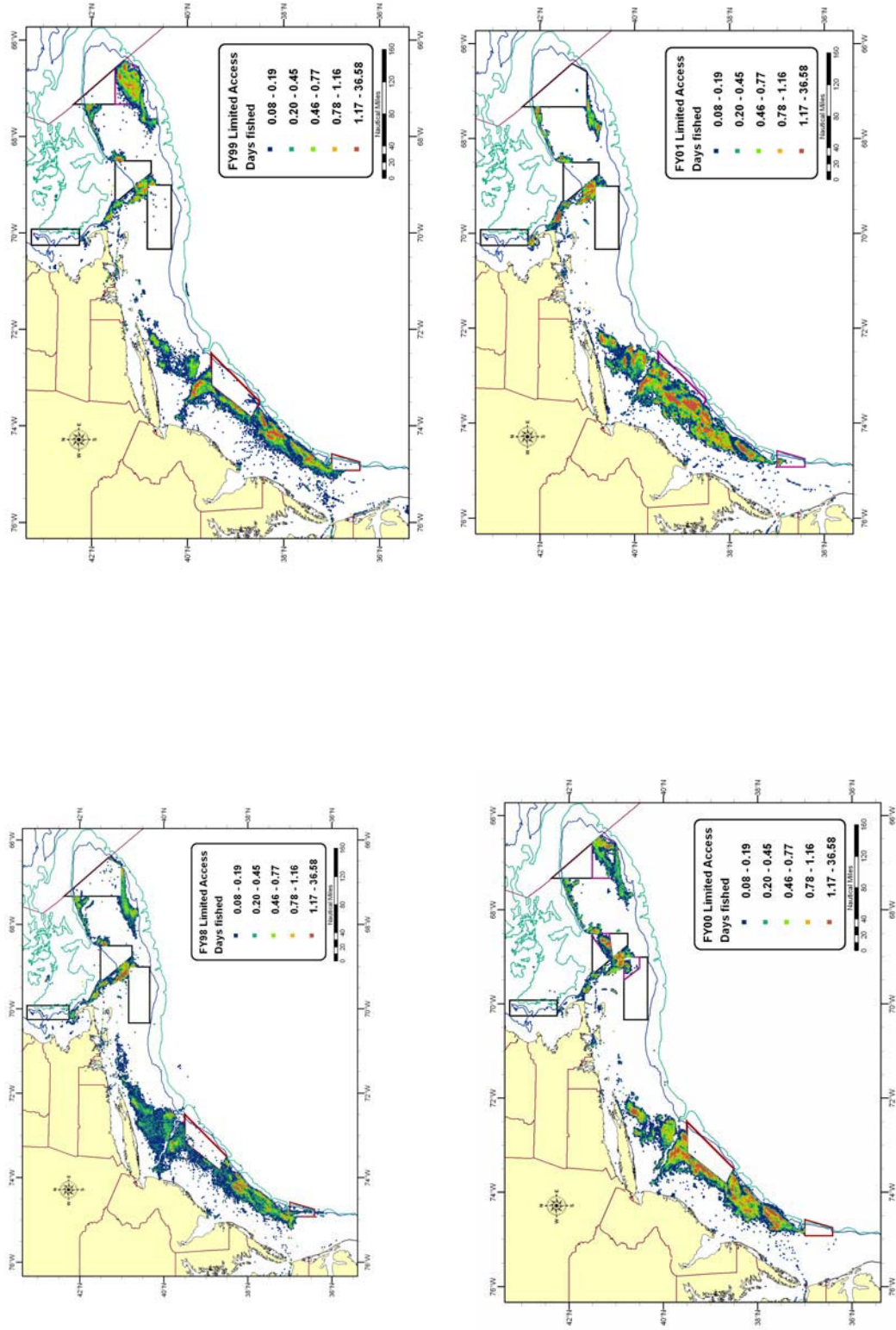


Figure B4-7 continued

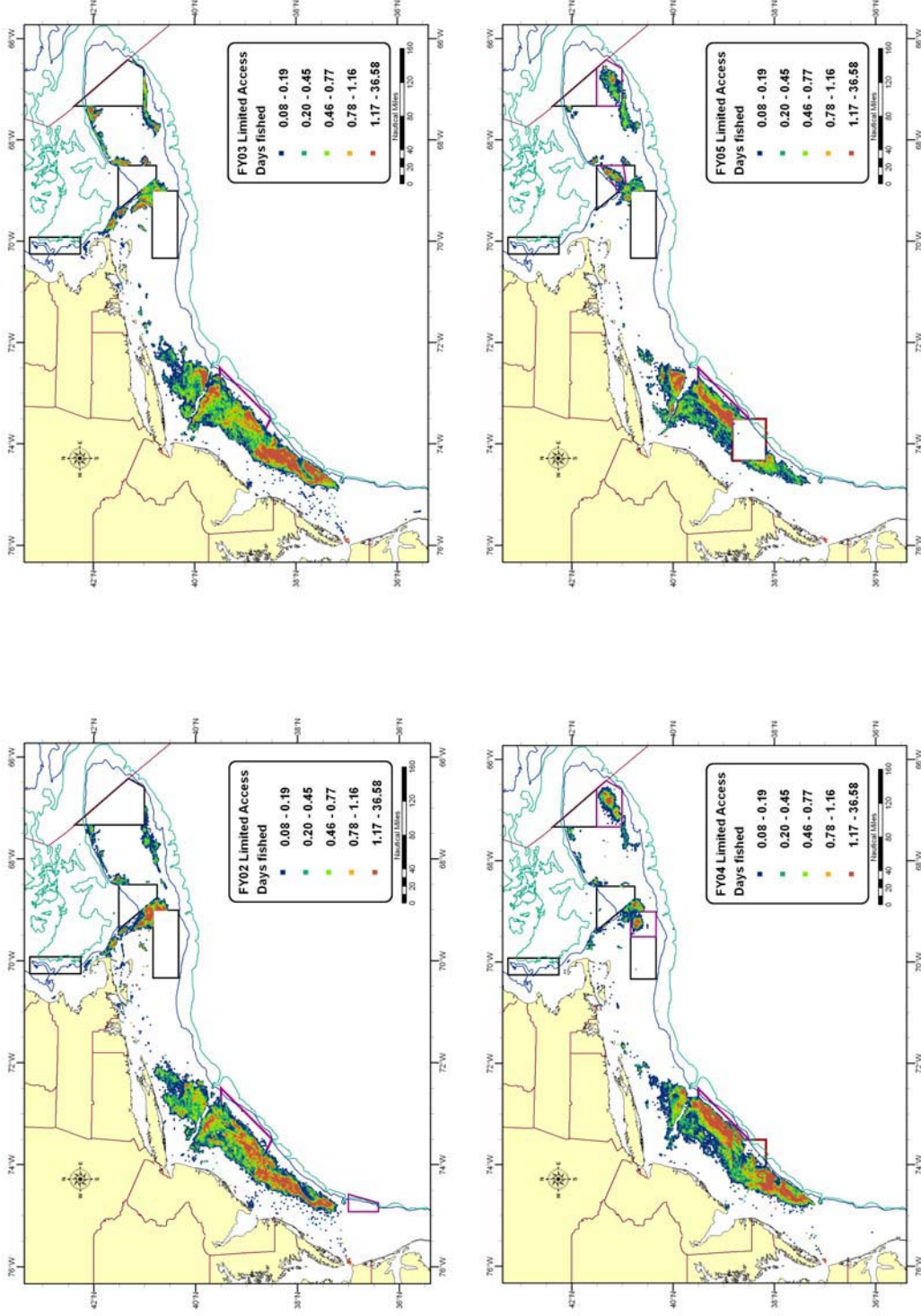


Figure B4-7 continued

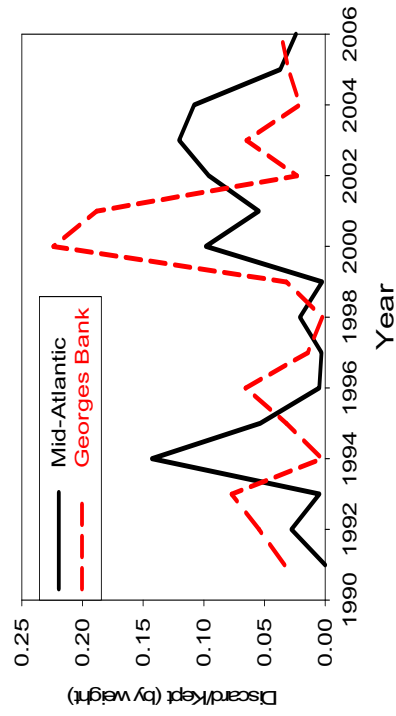
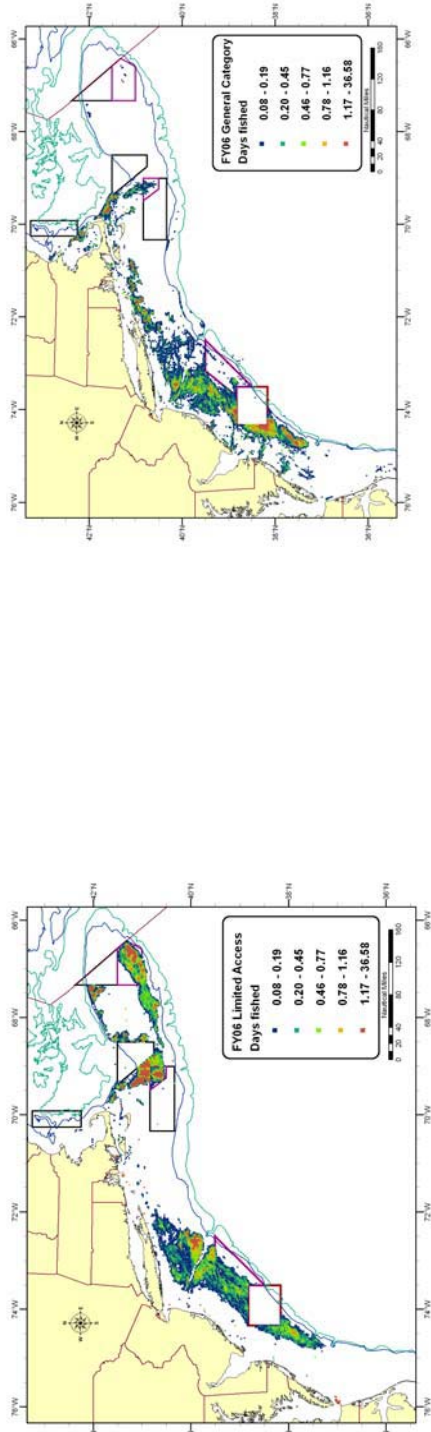


Figure B4-8. Discard to kept ratio for sea scallops in directed sea scallop trips.

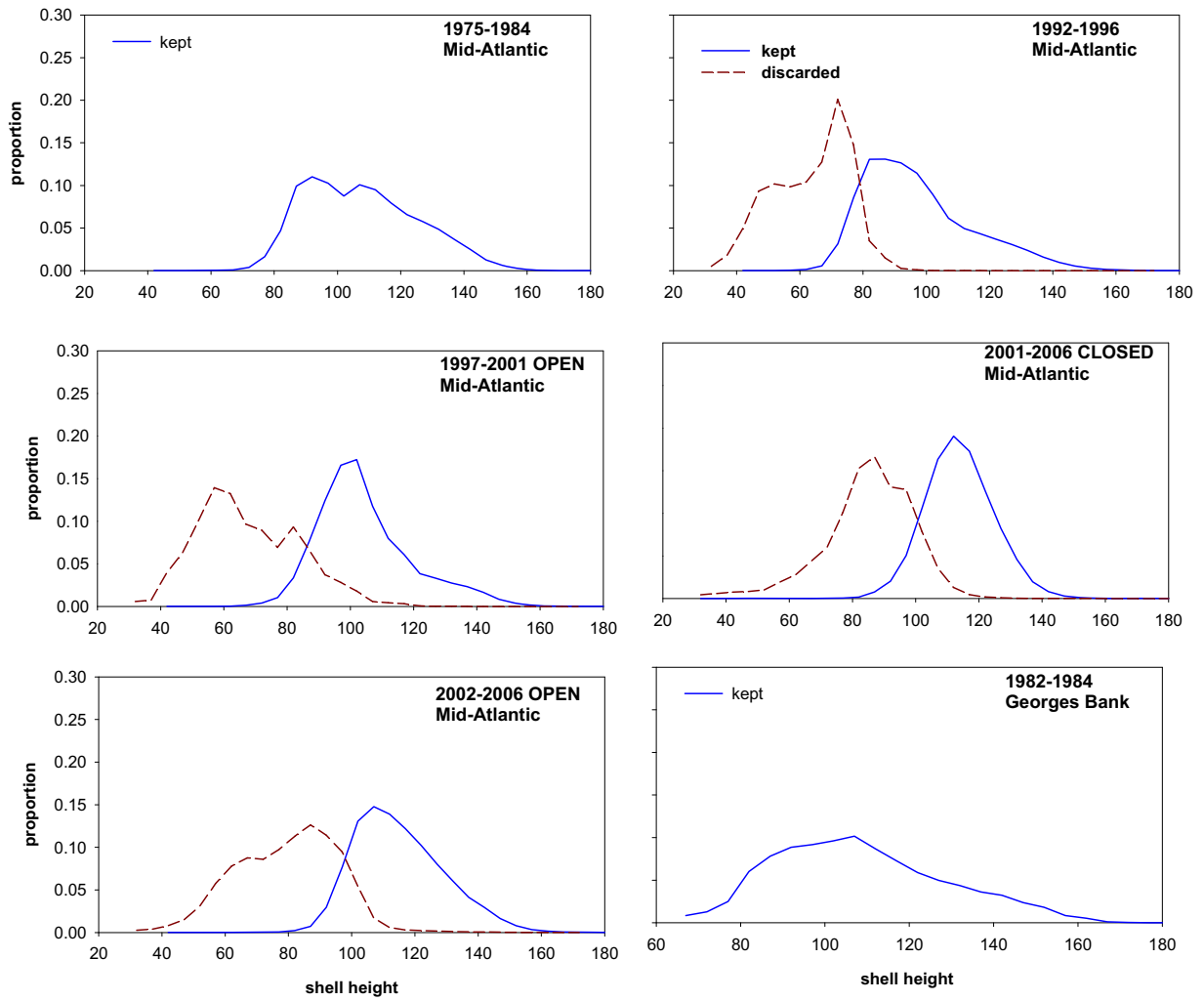
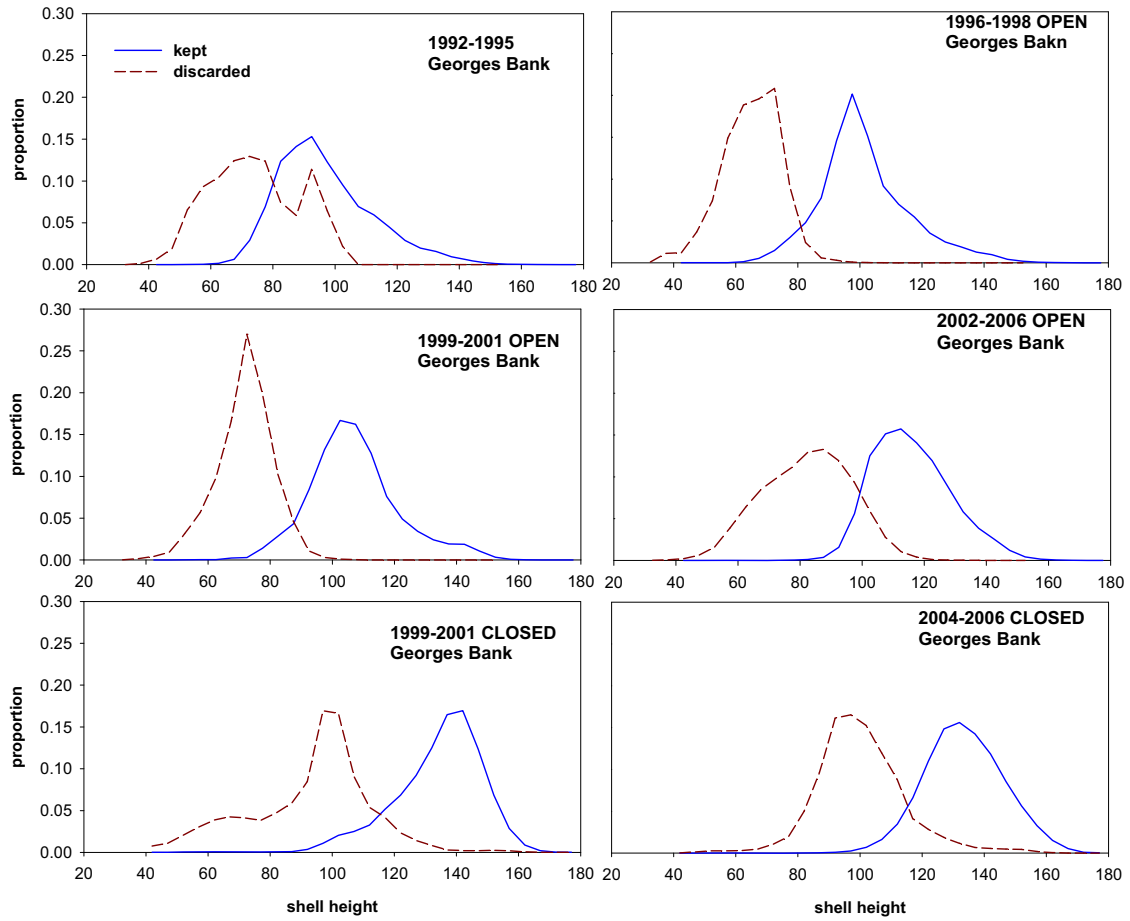


Figure B4-9. Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops, from port sampling (1975-1984) and sea sampling (1992-2006).

Figure B4-9 continued



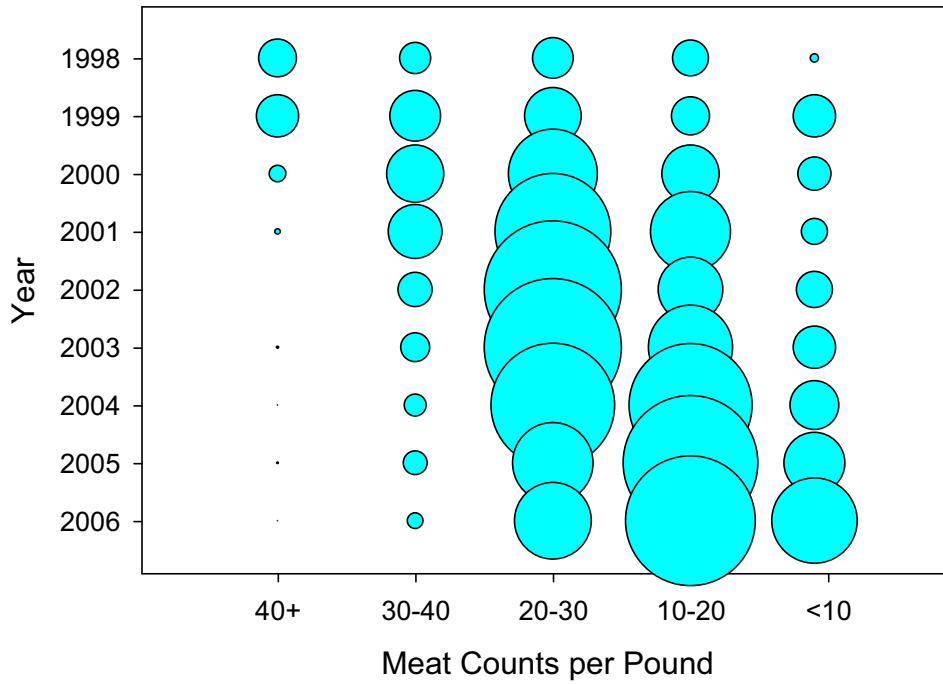


Figure B4-10. Commercial landings by meat count category, 1998-2006.

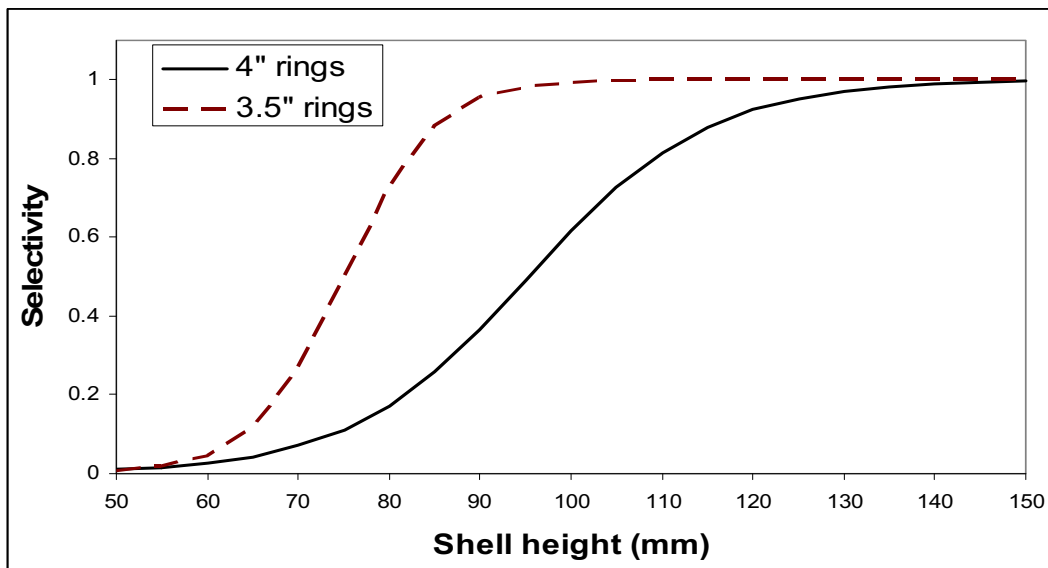


Figure B4-11. Selectivity of commercial scallop dredges with 3.5" and 4" rings (Yochum 2006, Appendix 5). The 3.5" ring selectivity is from NEFSC (2004).

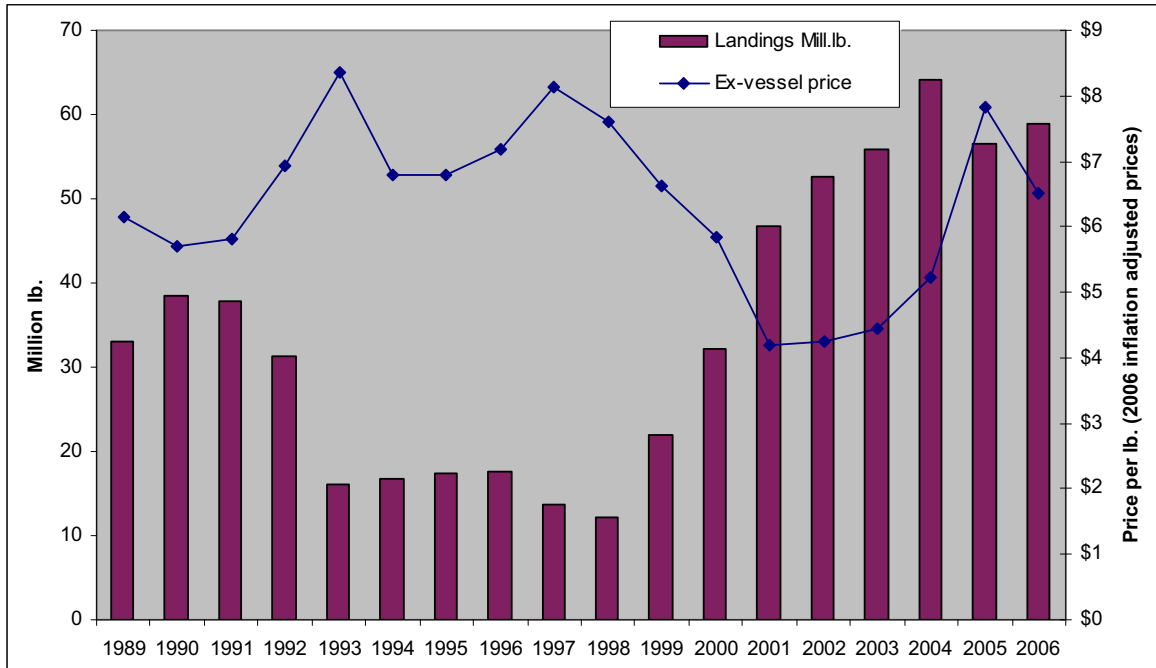


Figure B4-12. Trends in landings and ex-vessel prices in the U.S. sea scallop fishery.

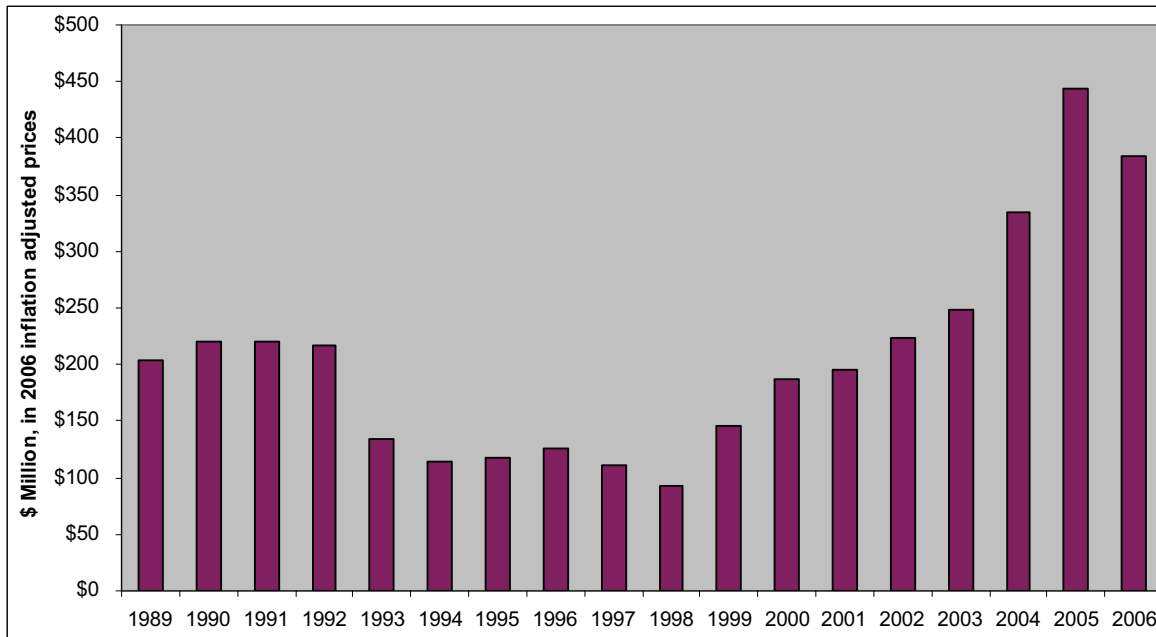


Figure B4-13. Trends in U.S. ex-vessel sea scallop revenues (adjusted for inflation to 2006 equivalent prices)

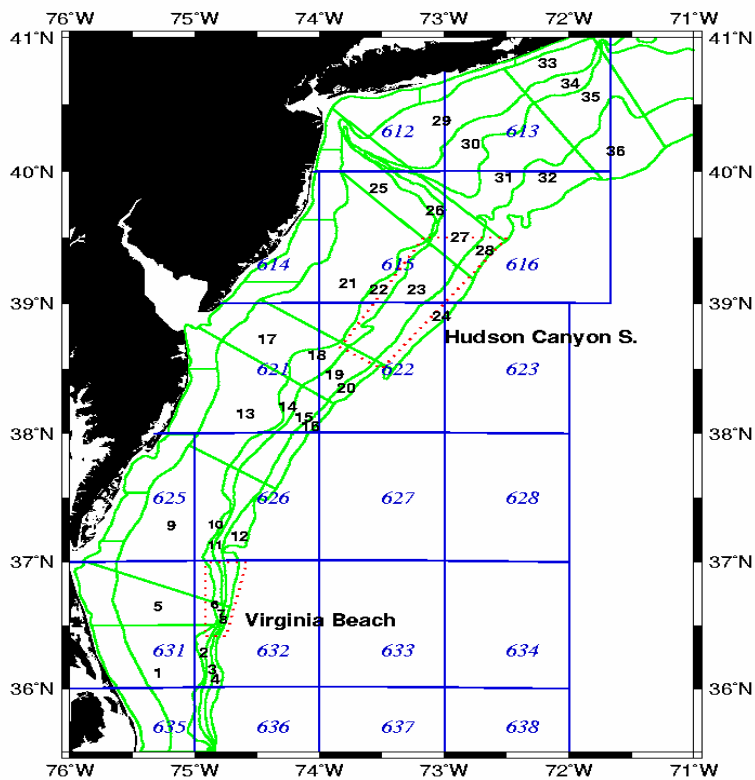
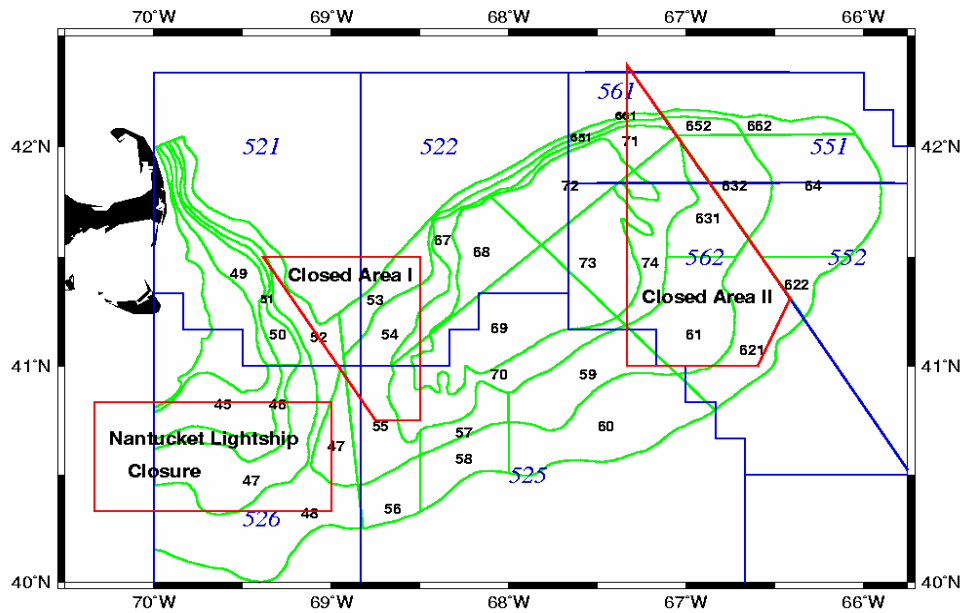


Figure B5-1. Georges Bank and Mid-Atlantic shellfish strata, statistical areas, groundfish closed areas and the original Mid-Atlantic rotational areas.

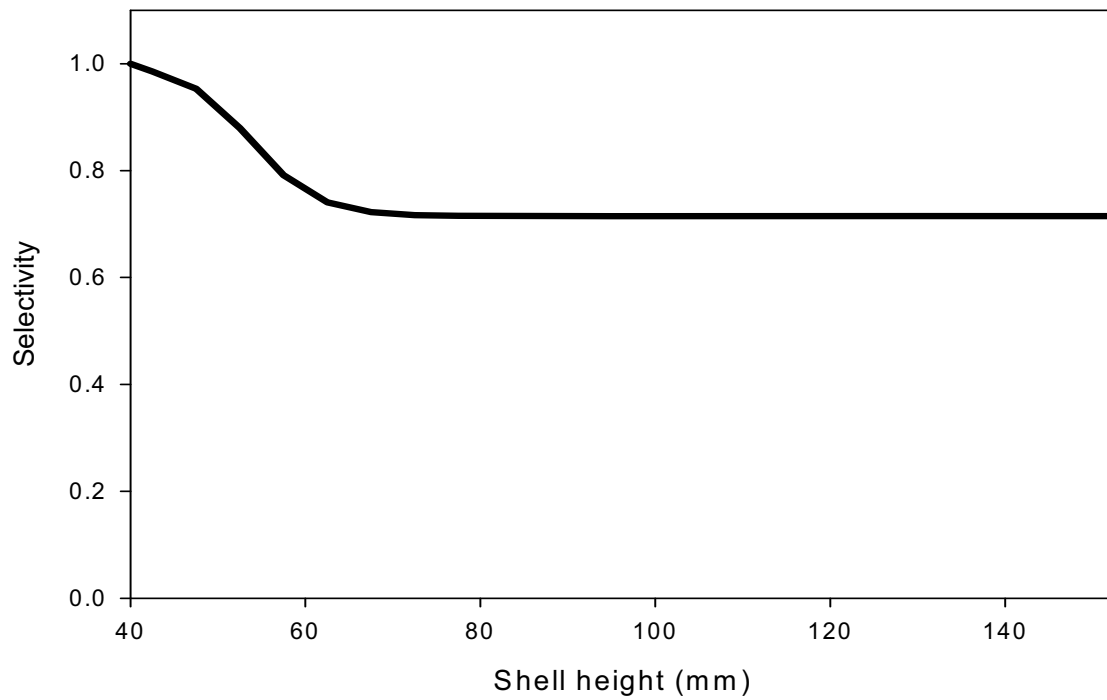
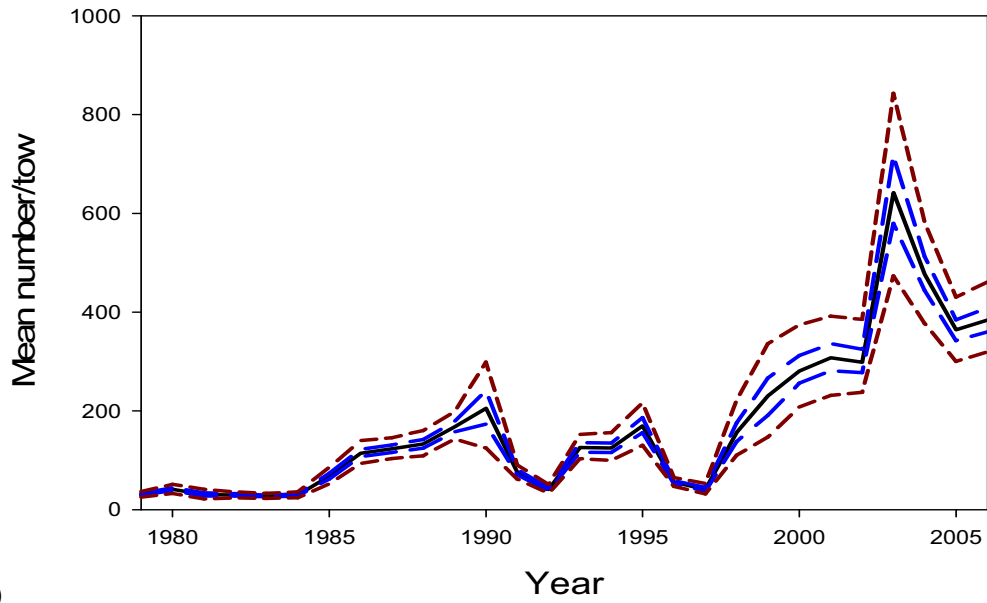


Figure B5-2. Selectivity of the lined dredge assumed in previous assessments (e.g., NEFSC 2001, 2004).

(a)



(b)

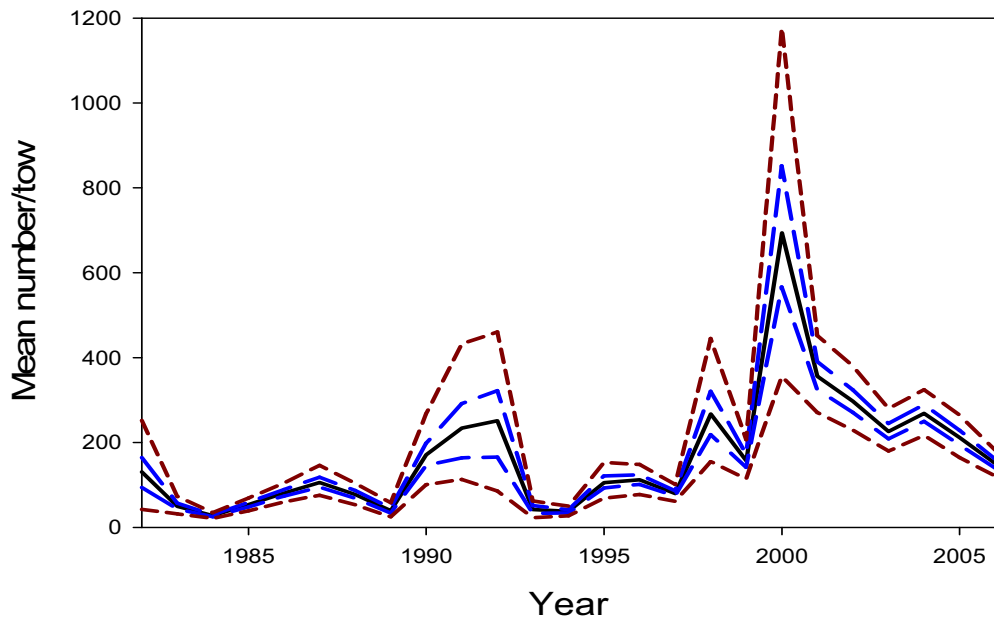


Figure B5-3. Bootstrapped estimates of abundance from NEFSC sea scallop survey for (a) Mid-Atlantic and (b) Georges Bank, showing median (solid line), 1st and 3rd quartiles (long dash), and 95% confidence interval (short dashed lines).

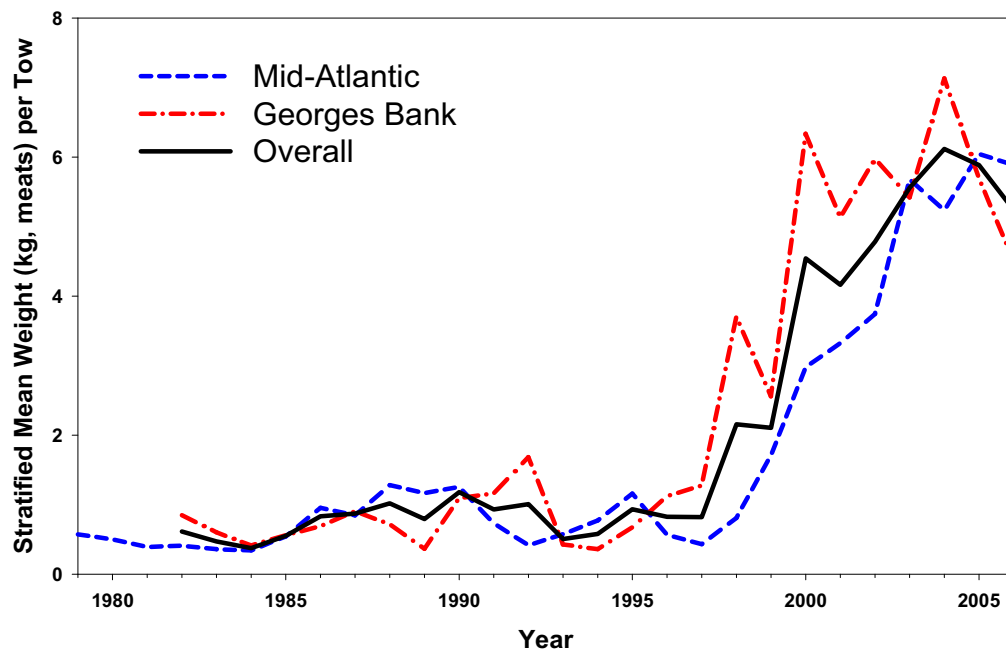
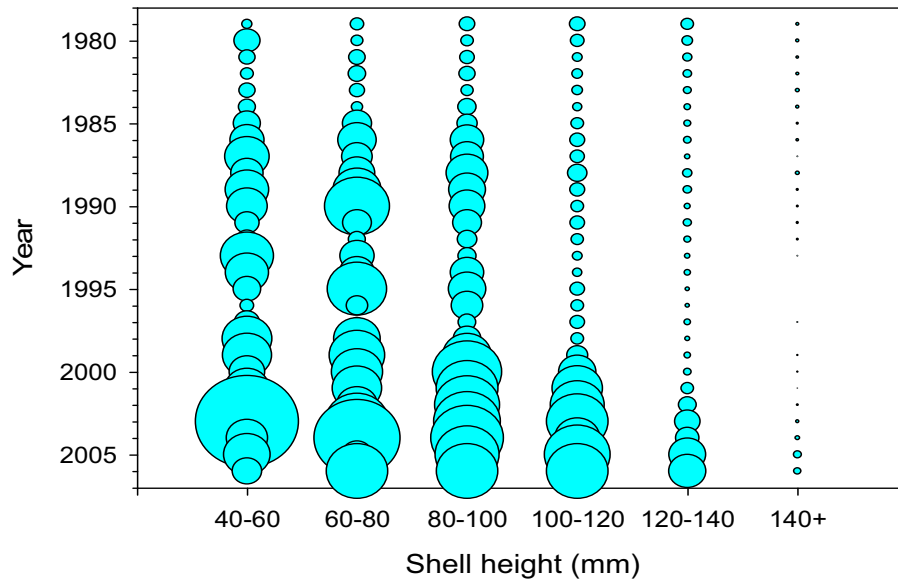


Figure B5-4. NEFSC sea scallop biomass indices.

(a)



(b)

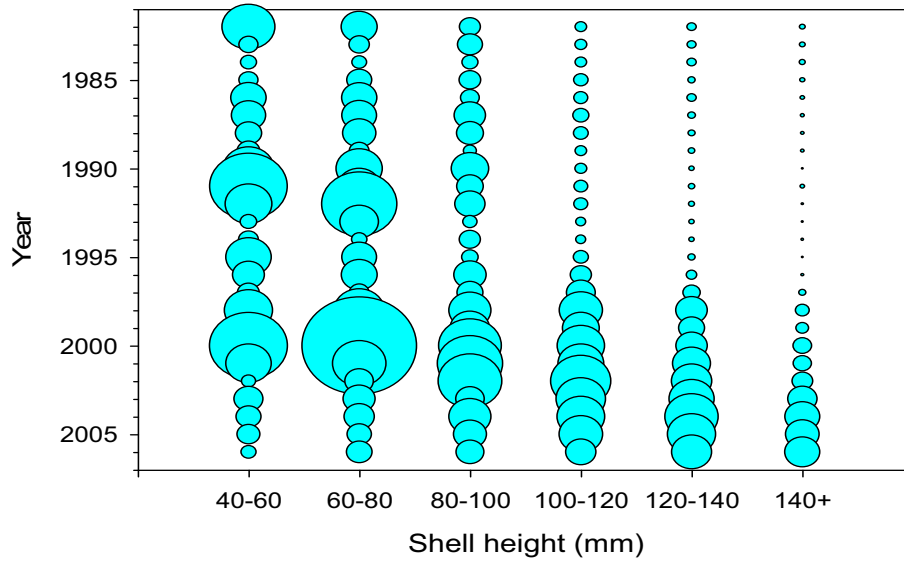
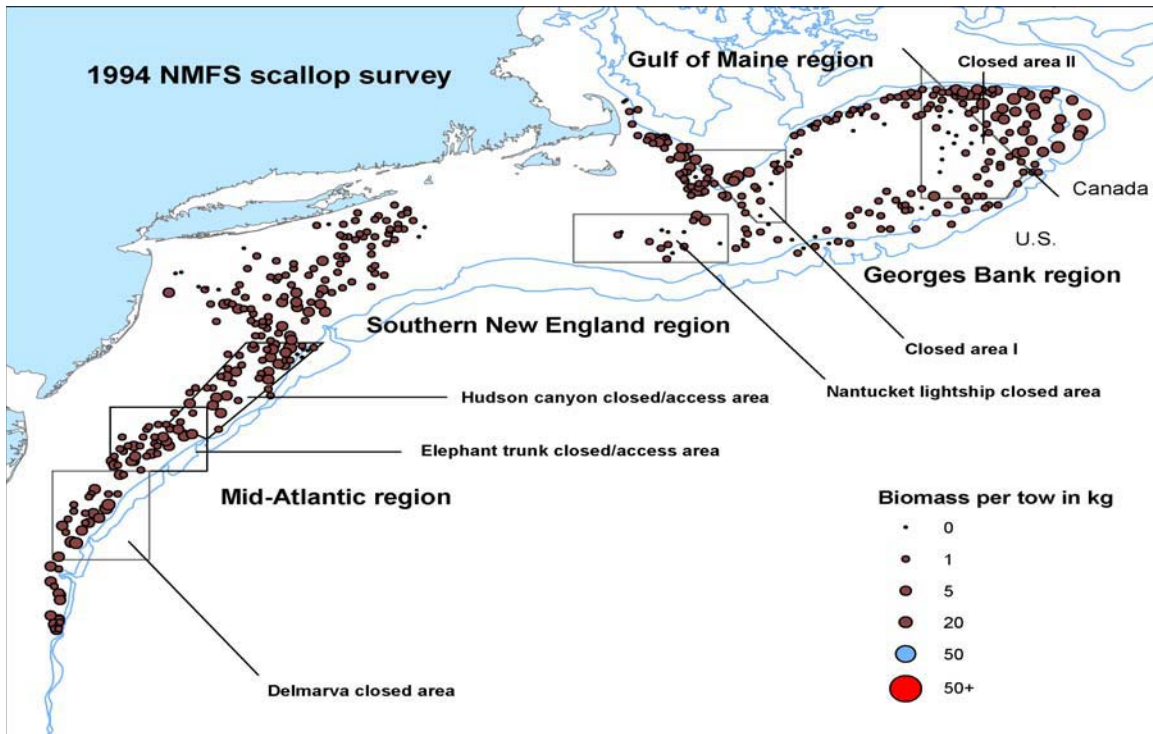


Figure B5-5. NEFSC sea scallop survey numbers at shell heights, in 20 mm intervals for (a) Mid-Atlantic Bight and (b) Georges Bank

(a)



(b)

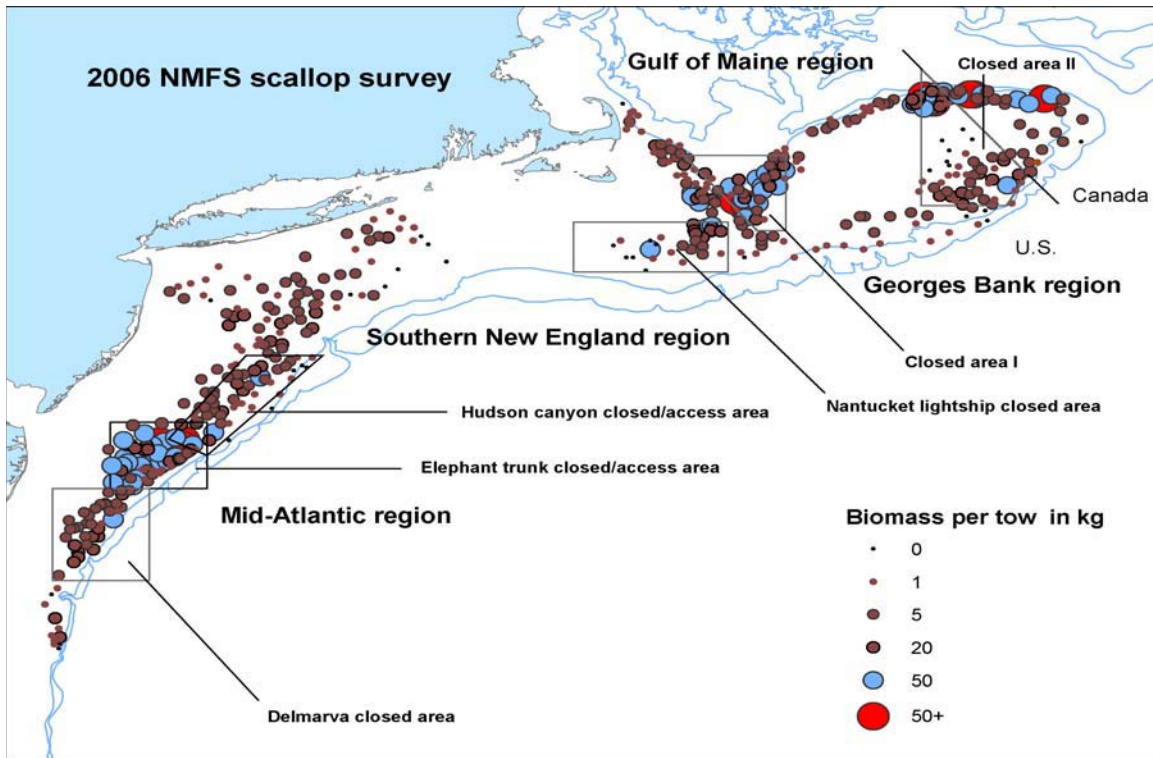


Figure B5-6. Distributions of sea scallops from the NEFSC sea scallop survey in (a) 1994 and (b) 2006.

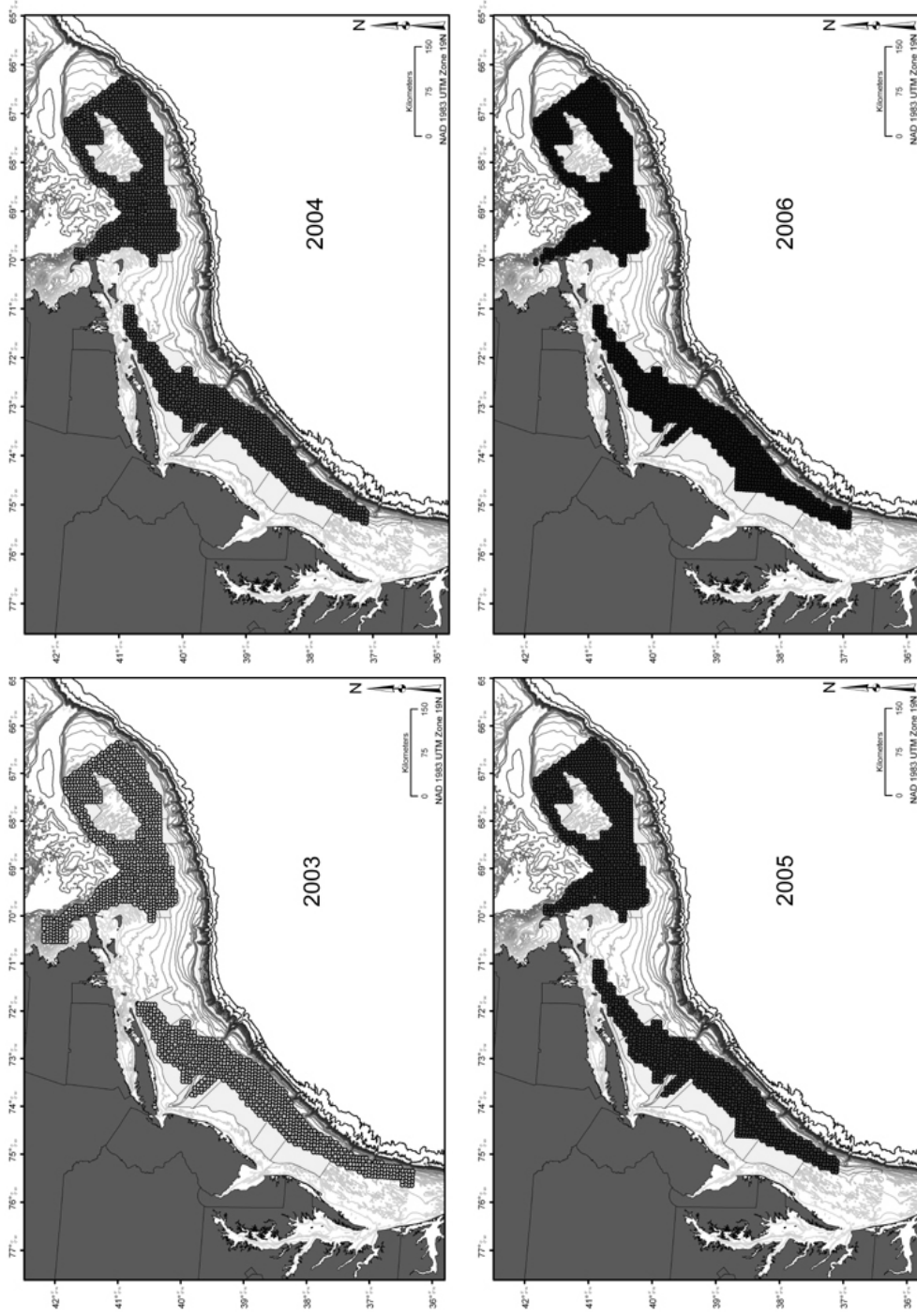


Figure B5-7. Stations occupied in SMAST video surveys during 2003-2006. NEFSC shellfish strata are shown for comparison.

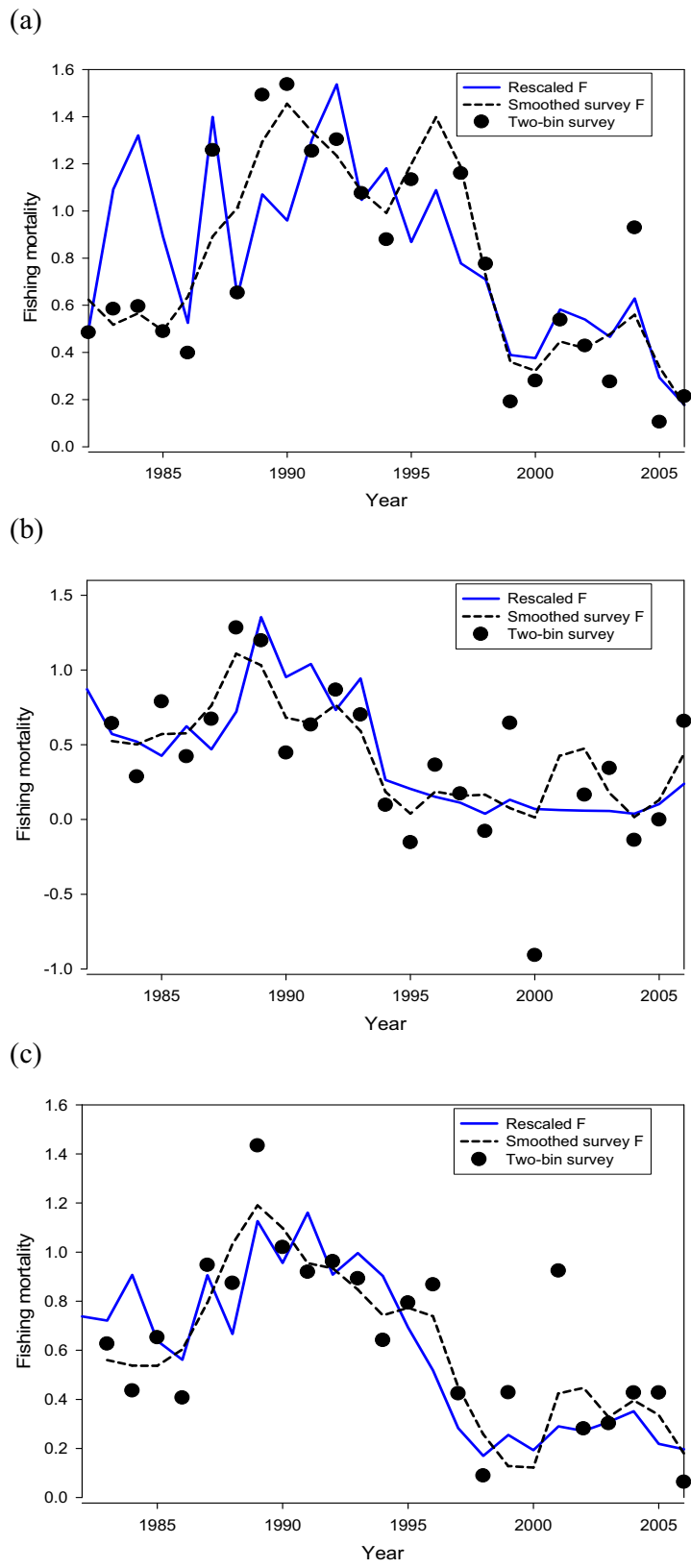


Figure B5-8. Survey-based (with moving average smoother) and rescaled-F estimates of fishing mortality for sea scallops (a) in the Mid-Atlantic, (b) on Georges Bank, and (c) overall.

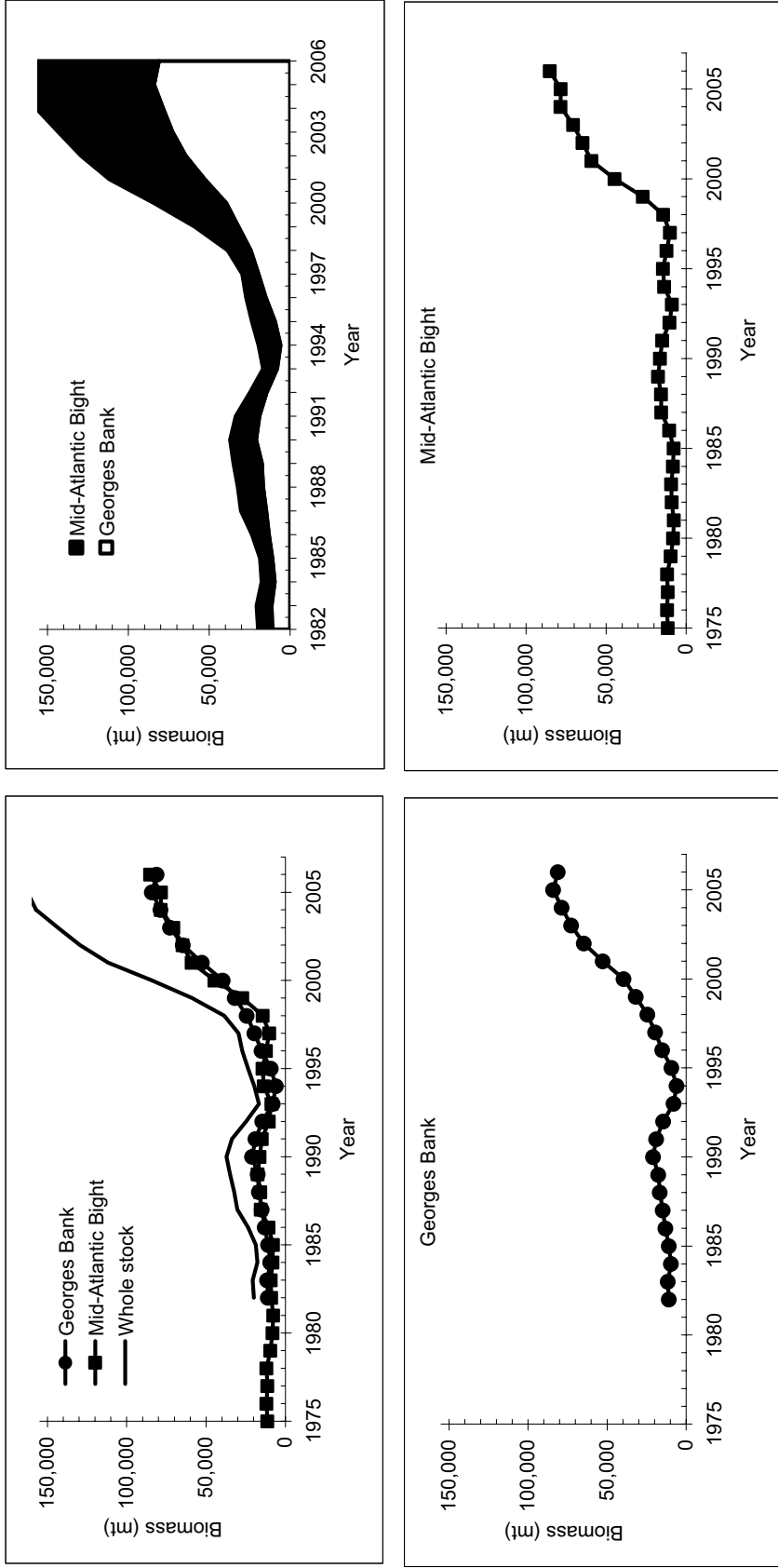


Figure B5-9. Biomass estimates (40+ mm on January 1) from the CASA model for sea scallops on Georges Bank during 1982-2006, in the Mid-Atlantic Bight during 1975-2006 and for the whole stock during 1982-2006.

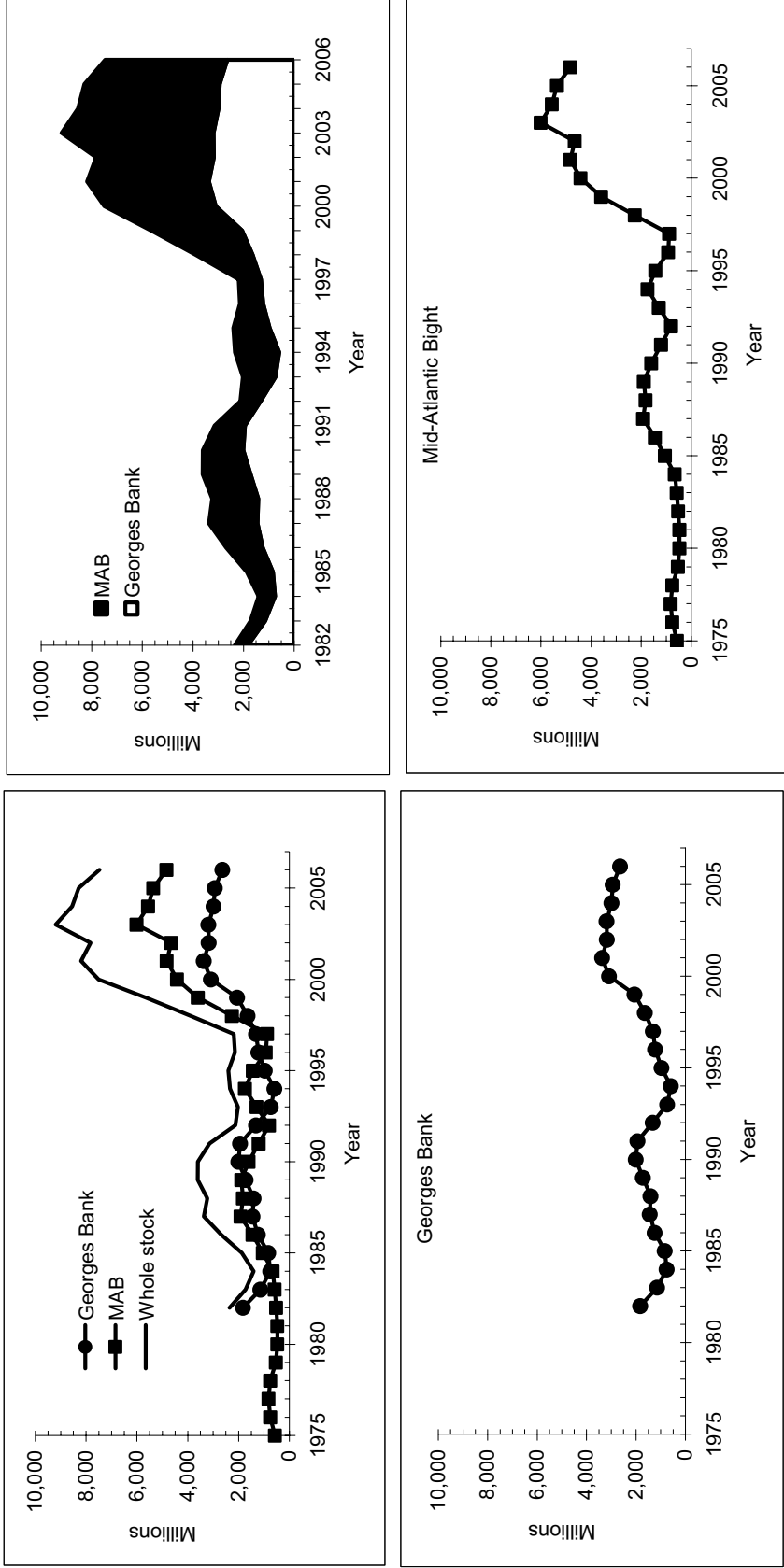


Figure B5-10. Abundance estimates (40+ mm on January 1) from the CASA model for sea scallops on Georges Bank during 1982-2006, in the Mid-Atlantic Bight during 1975-2006 and for the whole stock during 1982-2006.

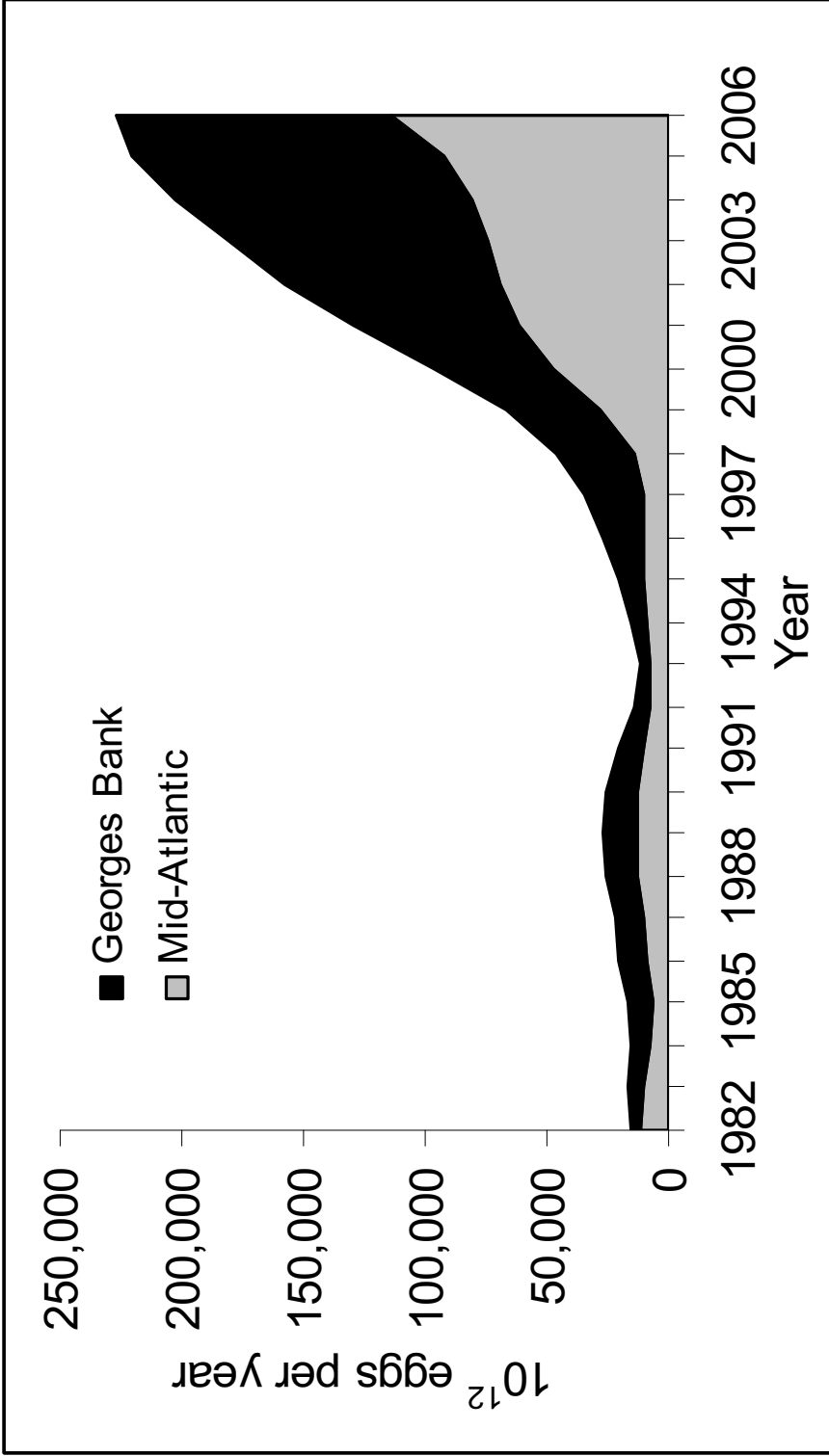


Figure B5-11. Estimated annual egg production by the sea scallop stock during 1982-2006.

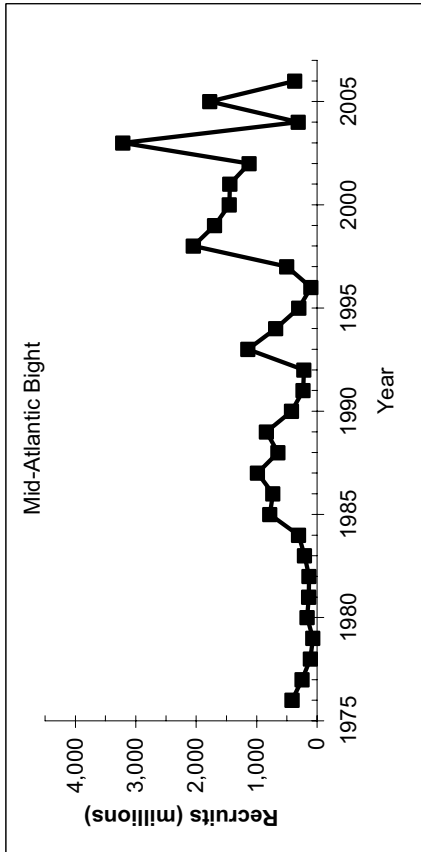
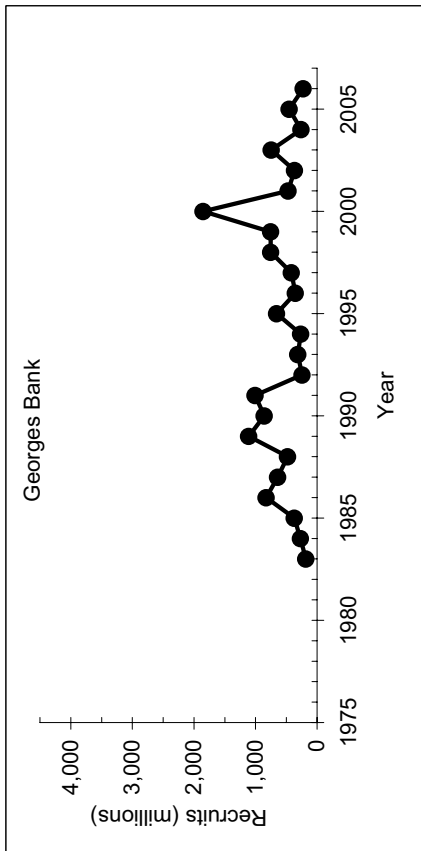
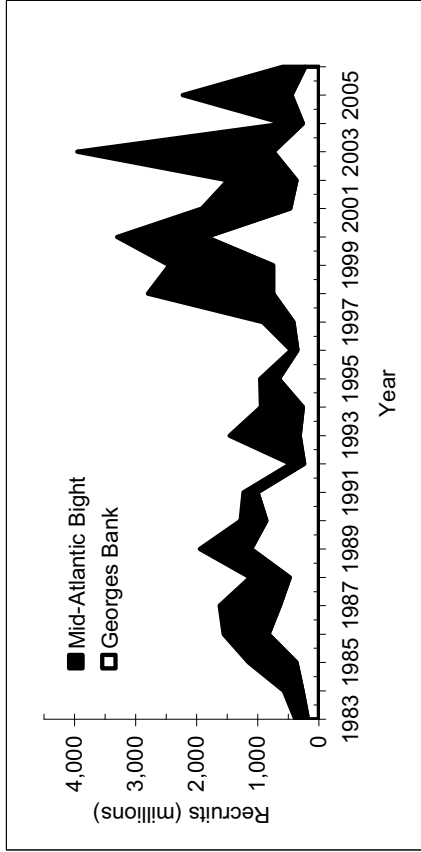
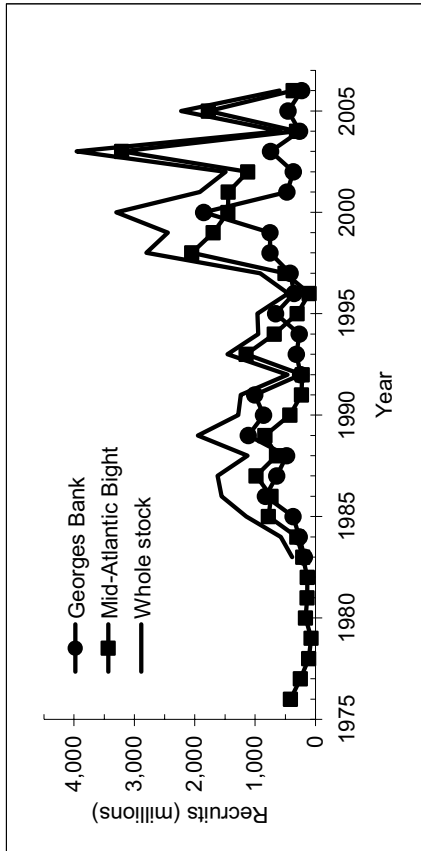


Figure B5-12. Recruitment estimates (40+ mm on January 1) from the CASA model for sea scallops on Georges Bank during 1983-2006, in the Mid-Atlantic Bight during 1976-2006 and for the whole stock during 1983-2006.

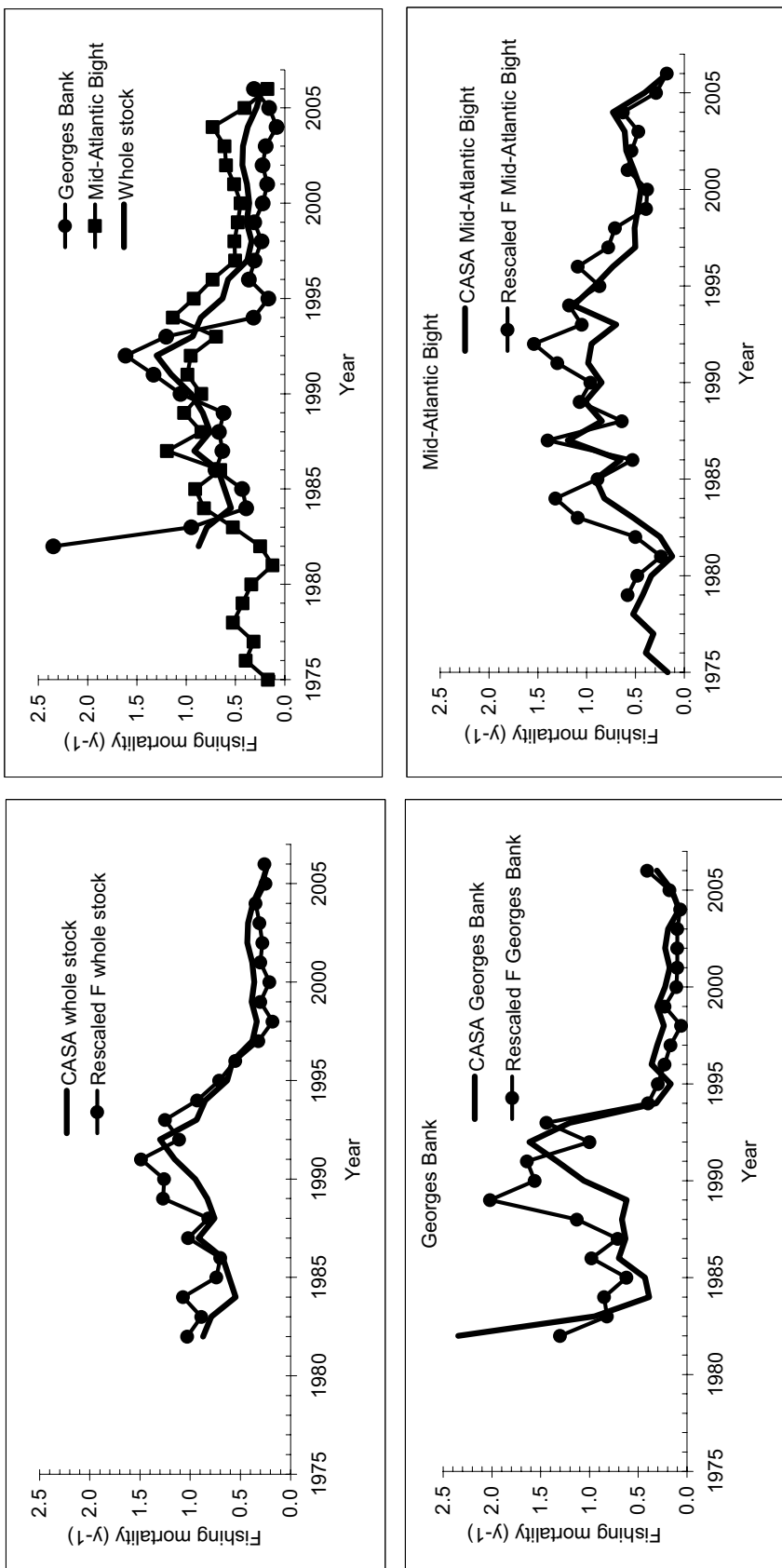


Figure B5-13. Fishing mortality estimates (y^{-1}) from the CASA model for sea scallops on Georges Bank, in the Mid-Atlantic Bight and for the whole stock during 1975-2006.

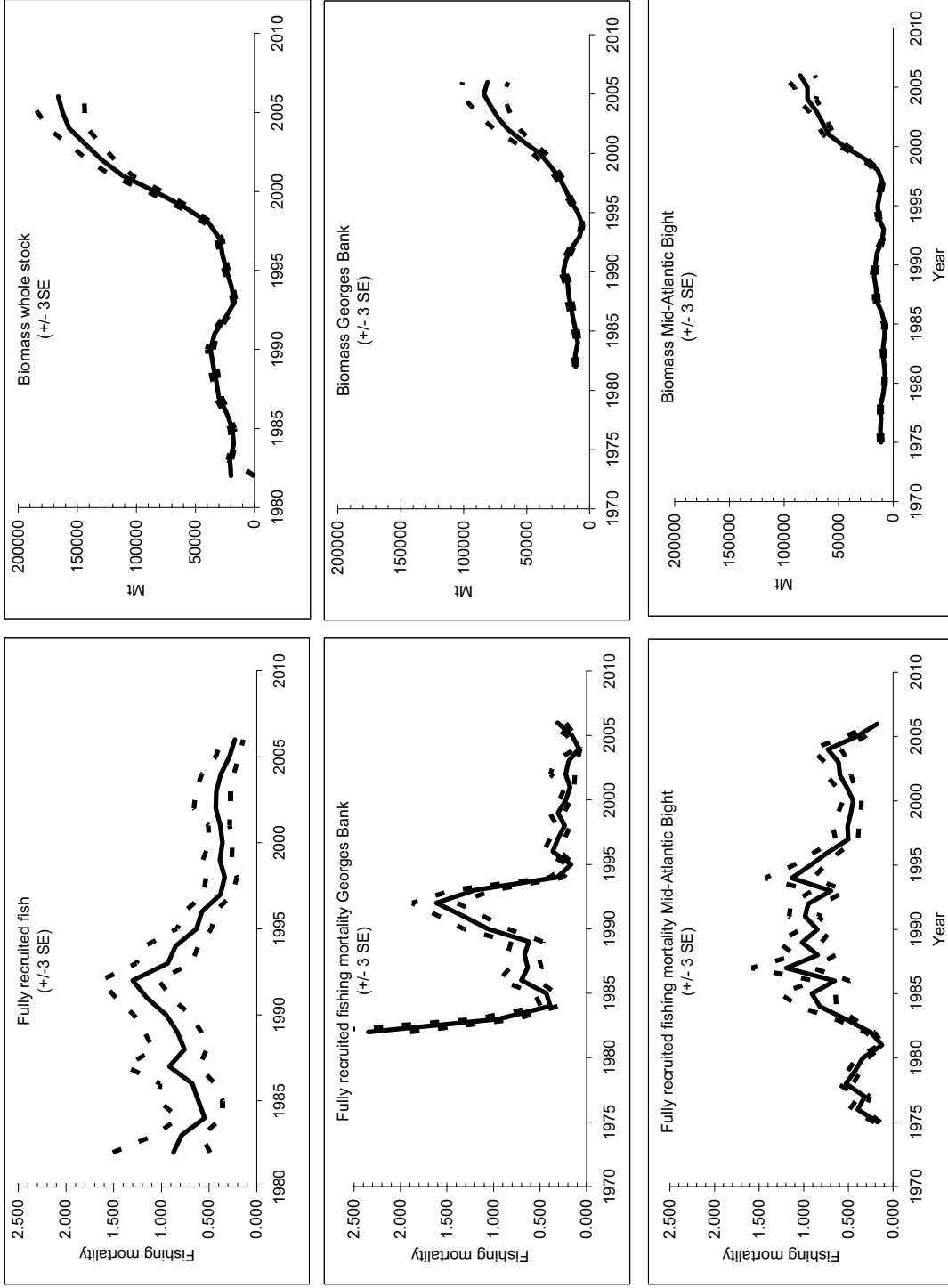


Figure B5-14. CASA model estimates of fishing mortality and biomass (± 3 standard errors) for sea scallops on Georges Bank, the Mid-Atlantic Bight and for the stock as a whole.

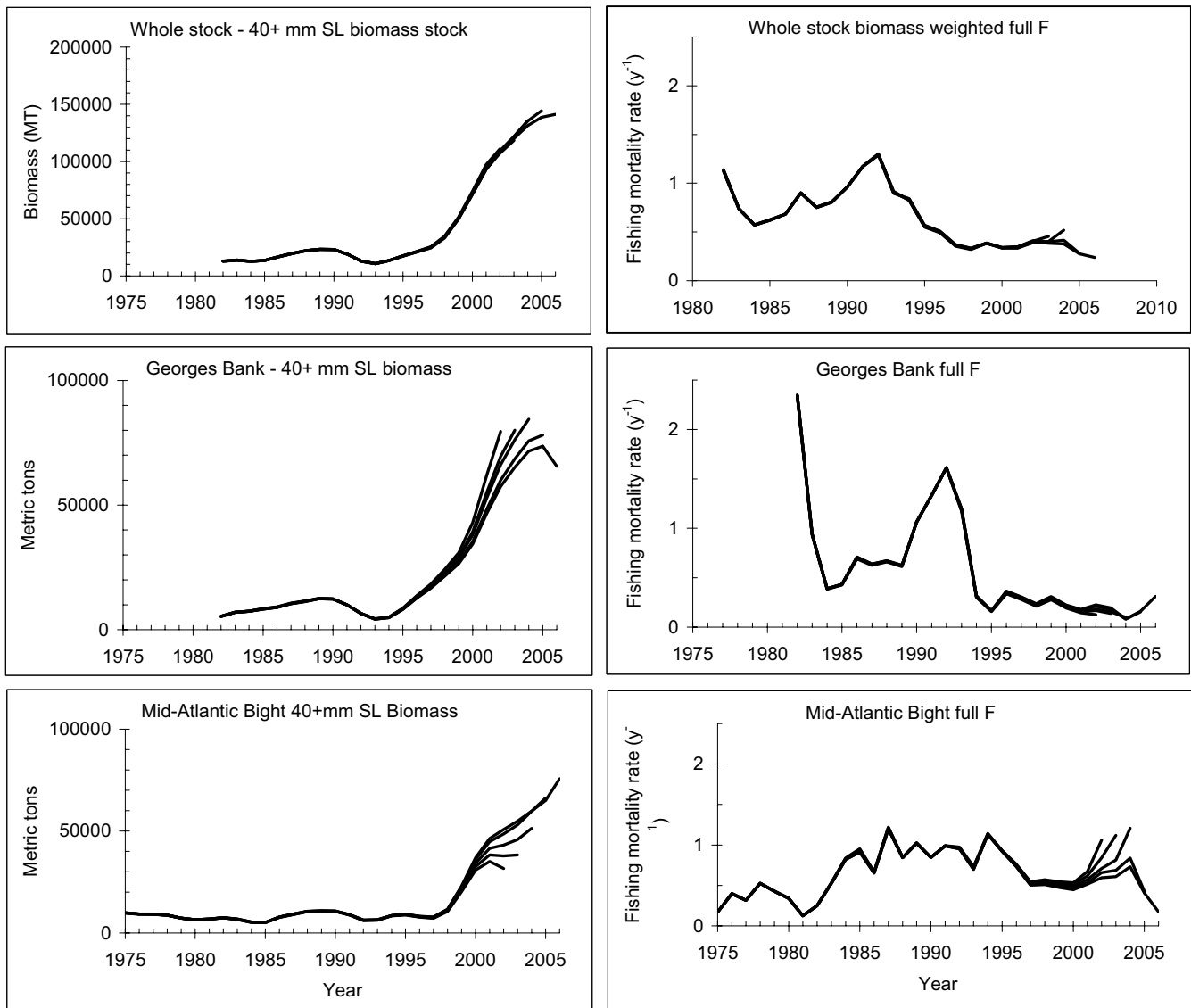


Figure B5-15. Retrospective analysis for basecase CASA model estimates of sea scallop stock biomass on Georges Bank, in the Mid-Atlantic Bight and for the stock as a whole. Fishing mortality rates for the whole stock are the biomass weighted fully recruited fishing mortality rates for Georges Bank and the Mid-Atlantic Bight, rather than catch numbers divided by mean fishable abundance, because biomass weighted values were easier to compute in retrospective analyses and should give approximately the same result.

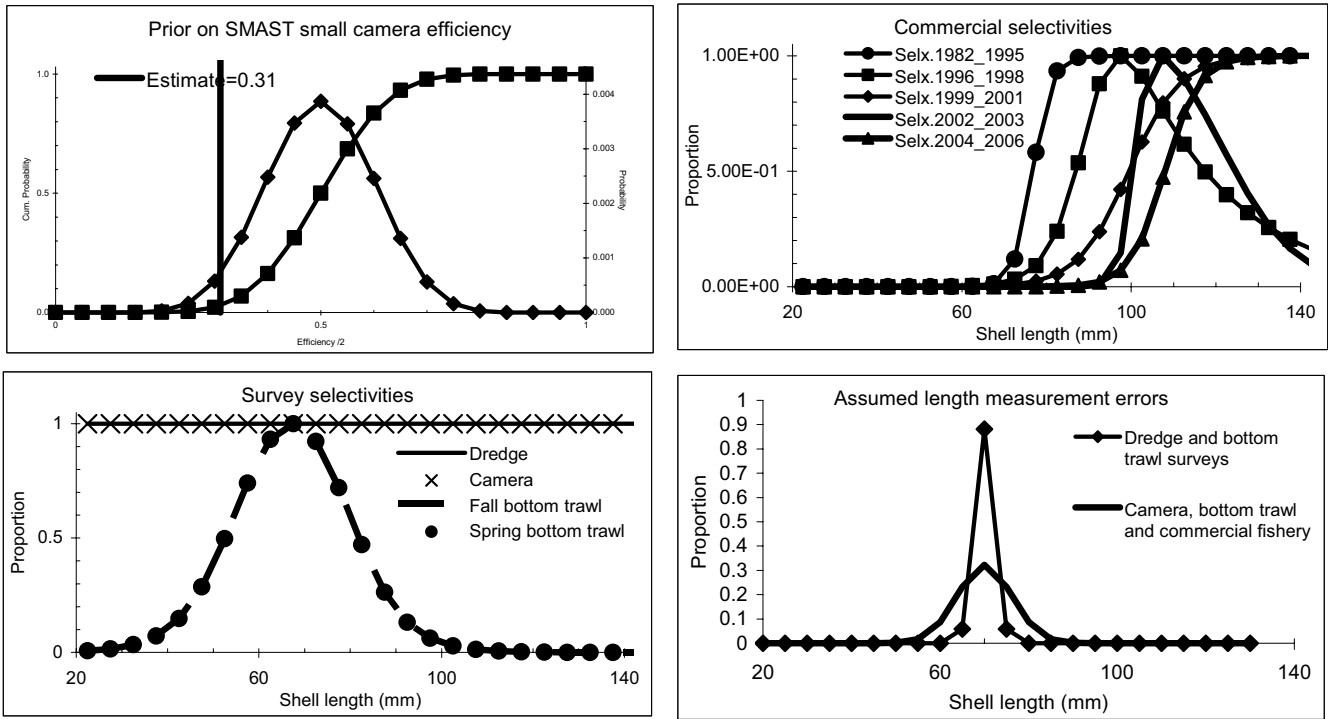


Figure B5-16. Diagnostics from basecase CASA model for sea scallops on Georges Bank.

Dredge survey

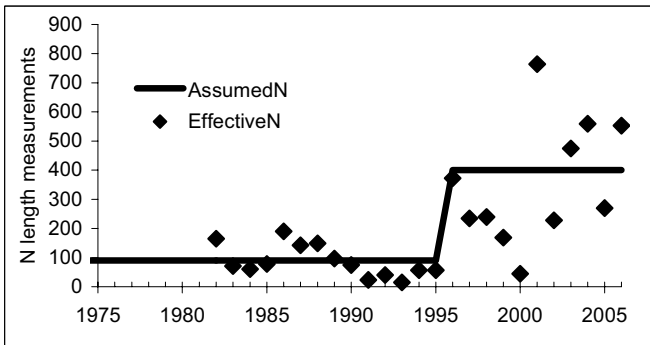
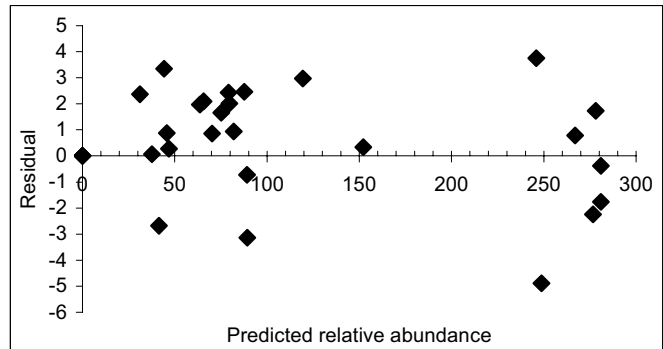
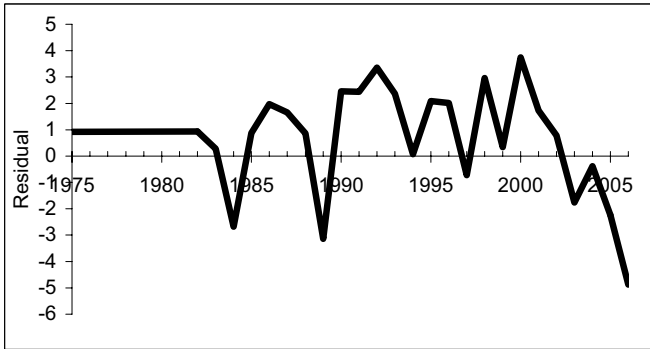
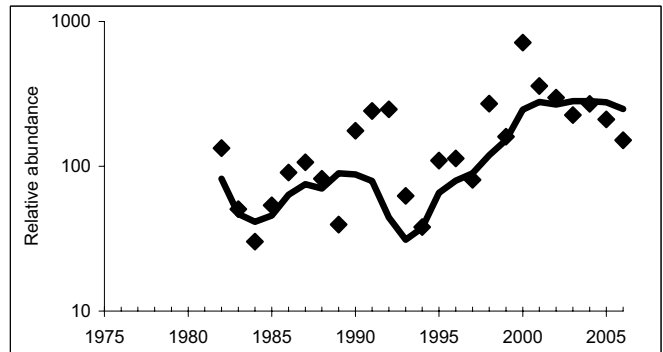
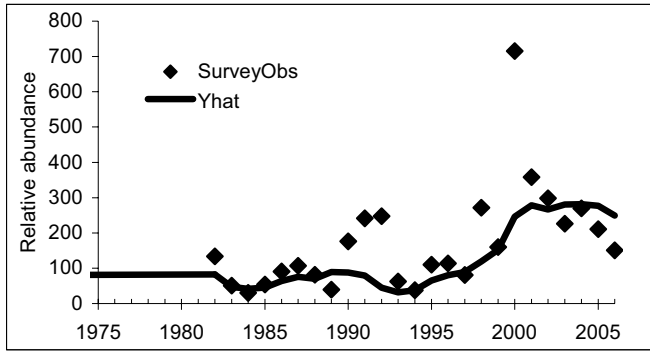


Figure B5-16 continued.

Small camera survey

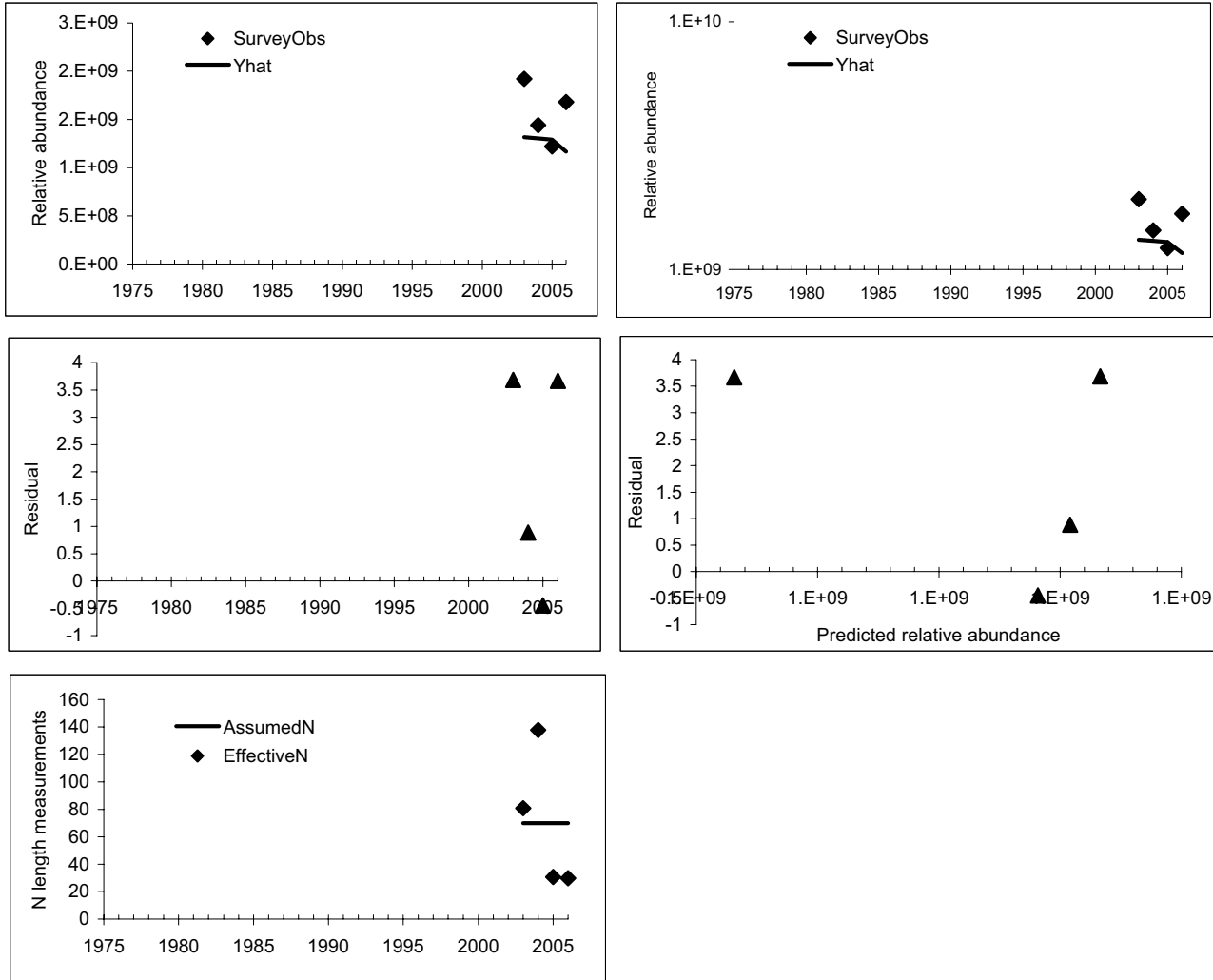


Figure B5-16 continued.

Landings per unit effort (LPUE)

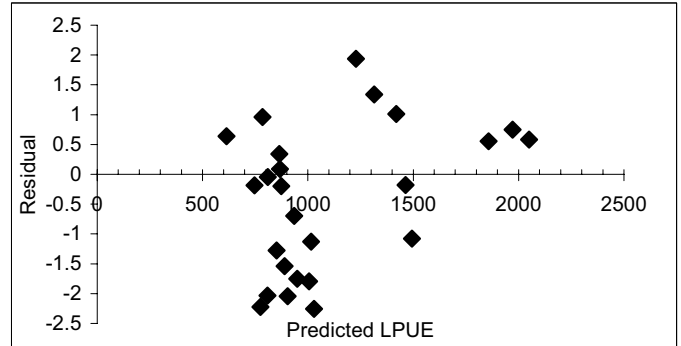
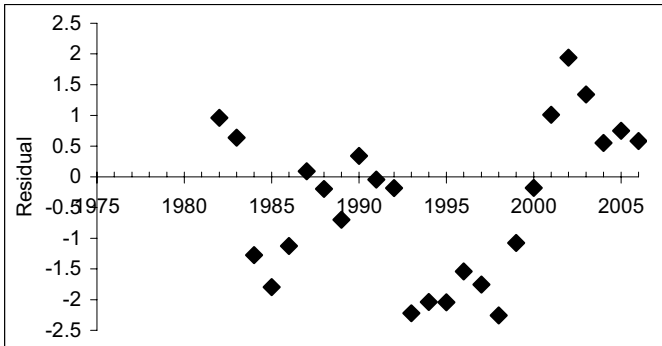
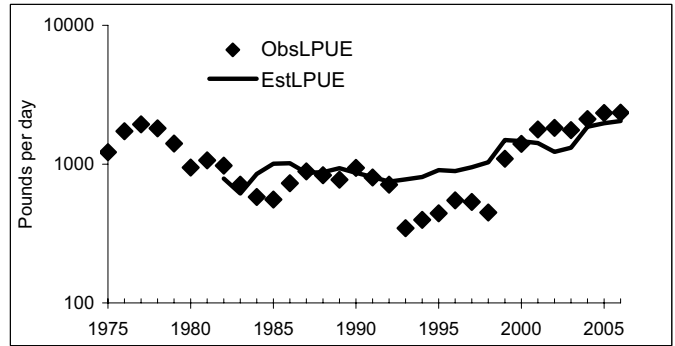
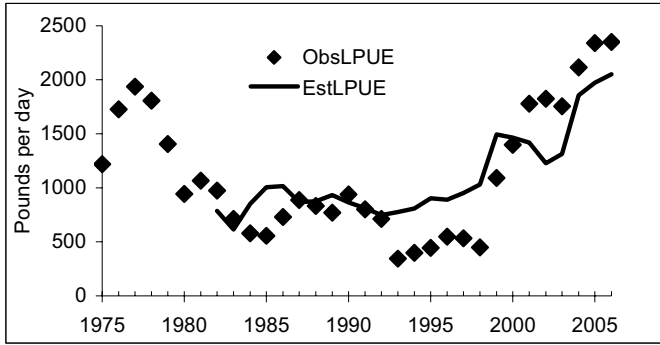


Figure B5-16 continued.

Landings

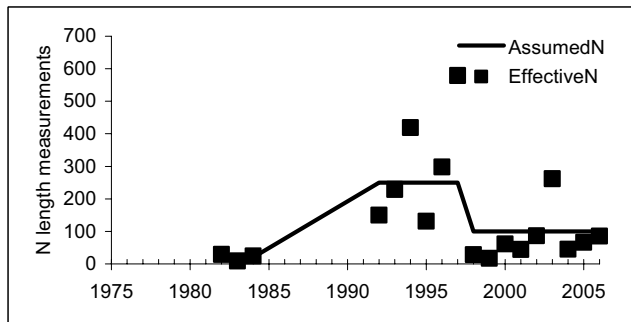
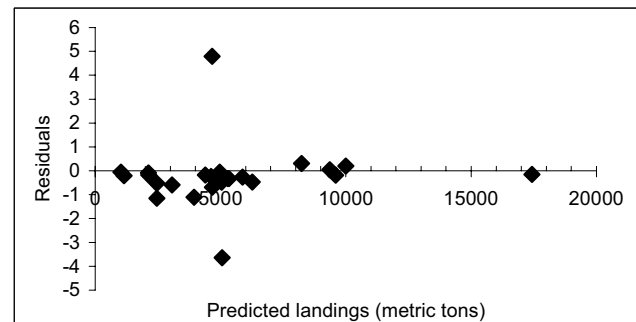
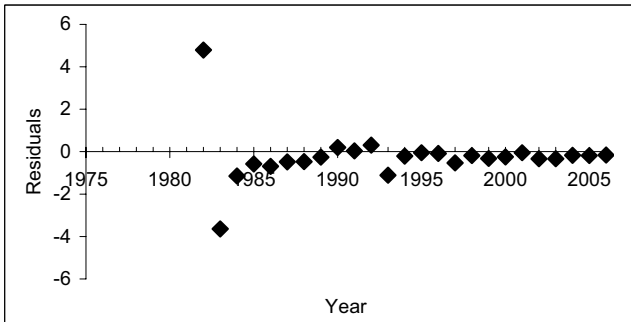
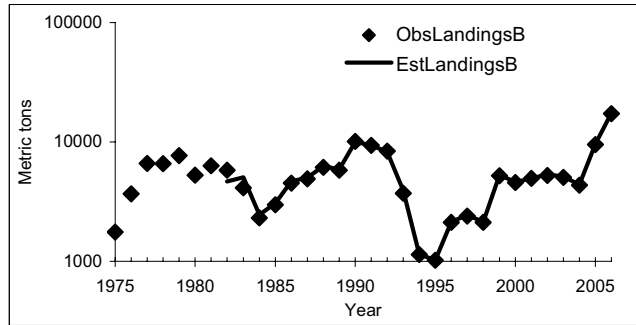
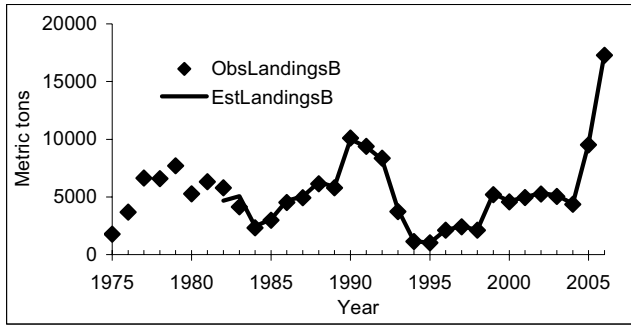


Figure B5-16 continued.

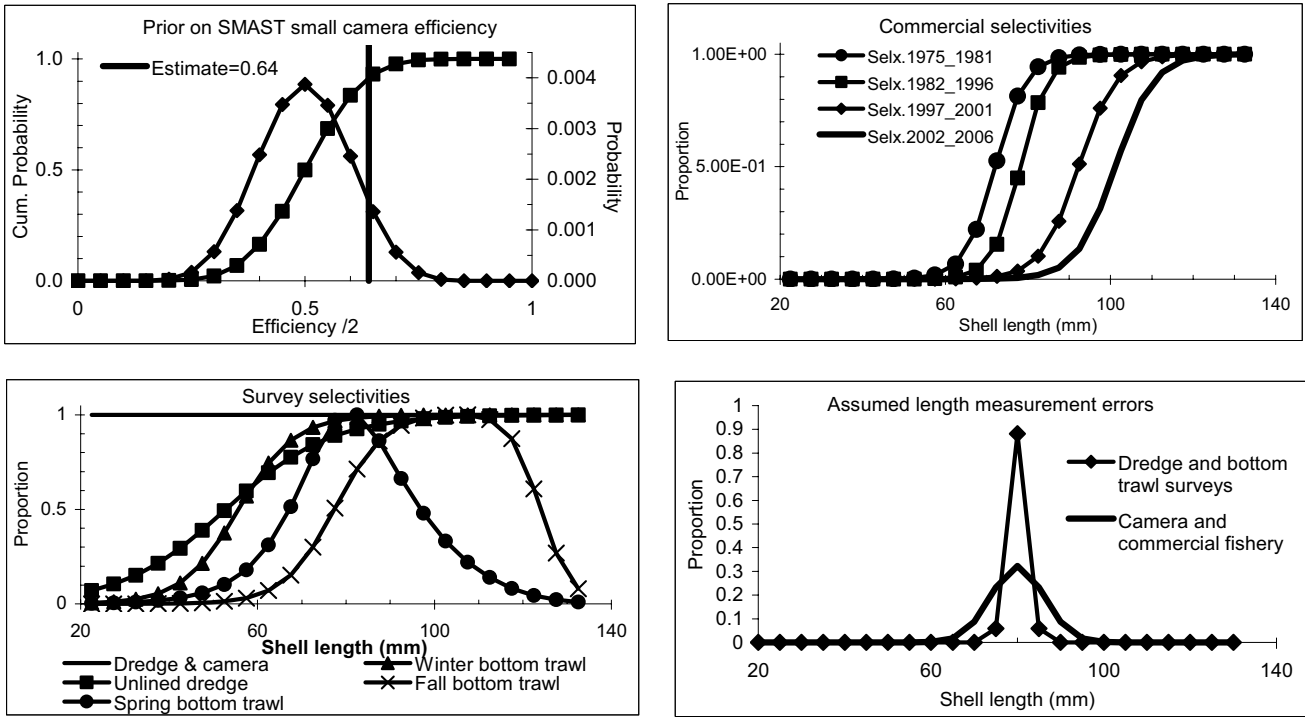


Figure B5-17. Diagnostics from basecase CASA model for sea scallops in the Mid-Atlantic Bight.

Dredge survey

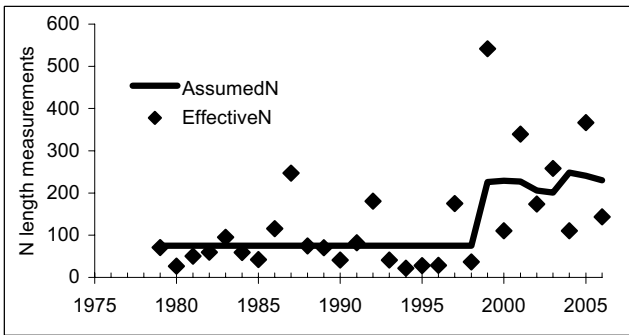
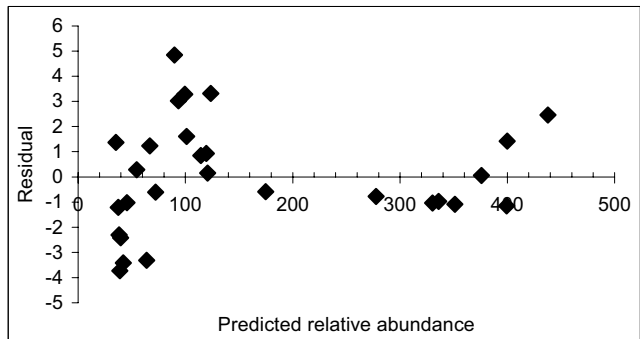
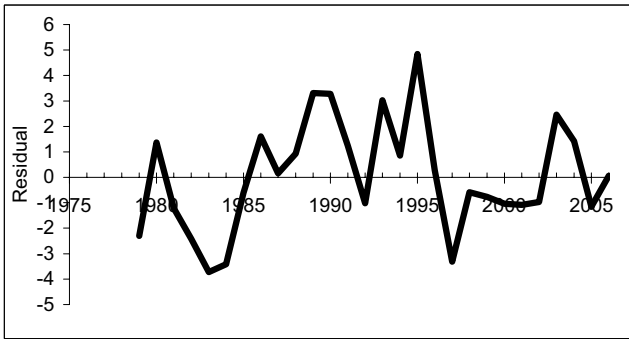
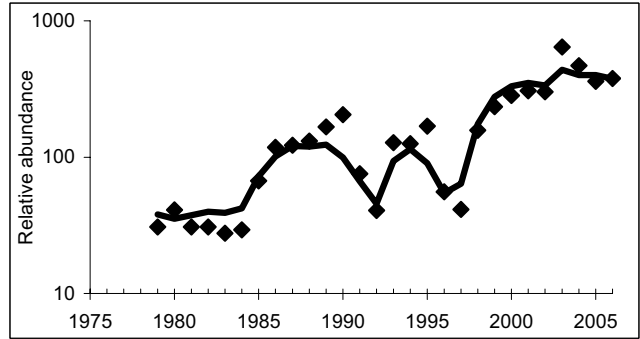
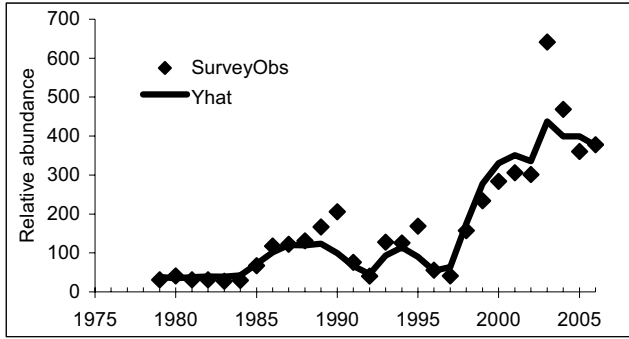


Figure B5-17 continued.

Small camera survey

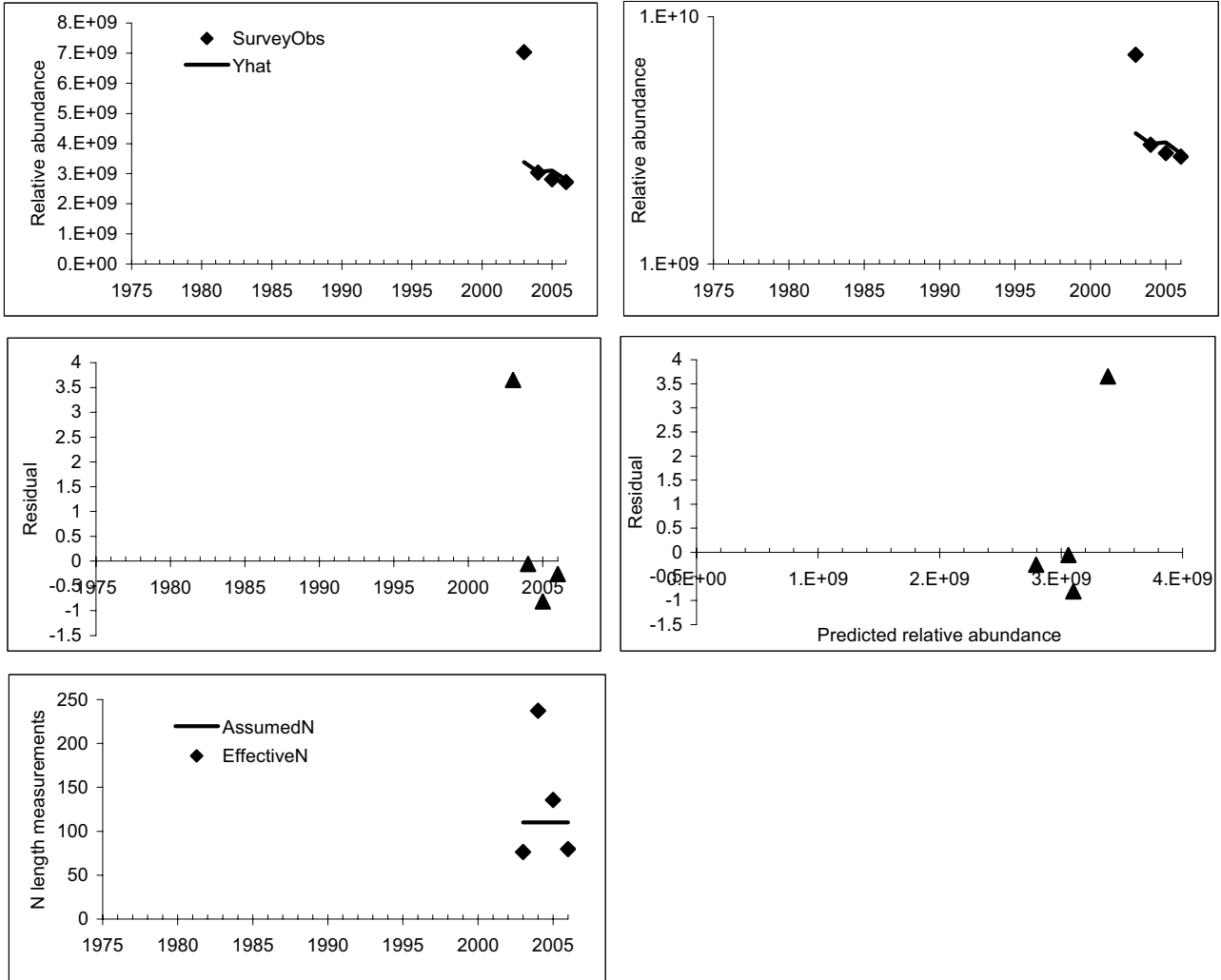


Figure B5-17 continued.

Winter bottom trawl survey

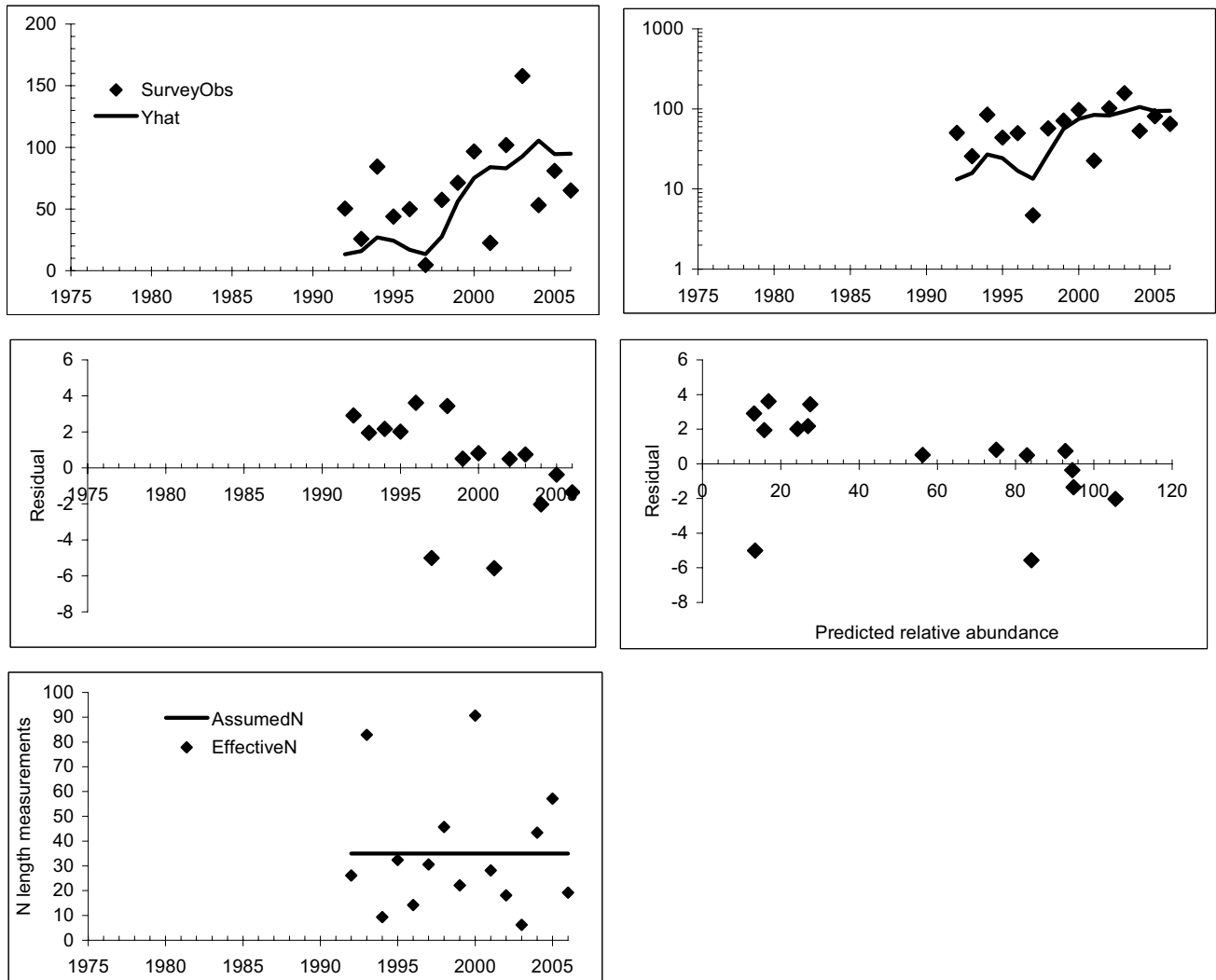


Figure B5-17 continued.

Unlined dredge

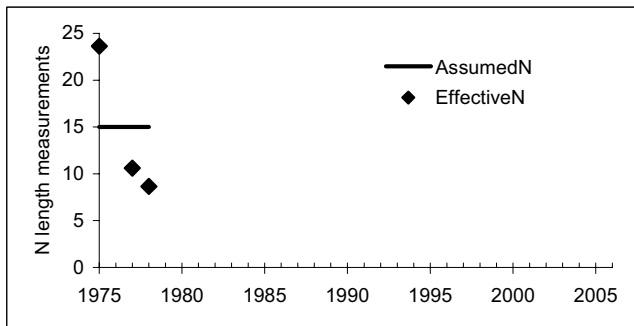
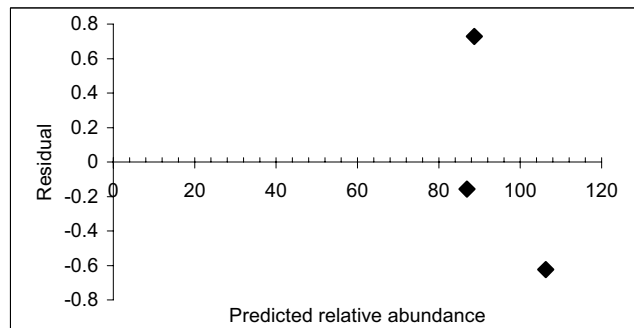
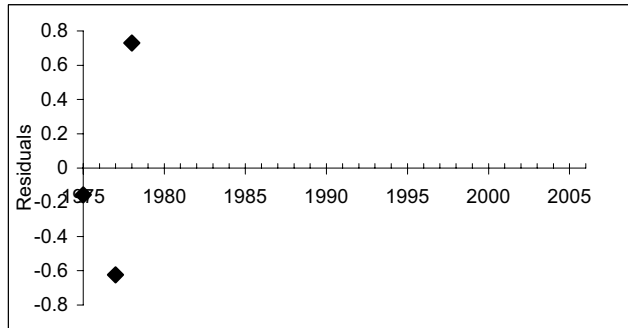
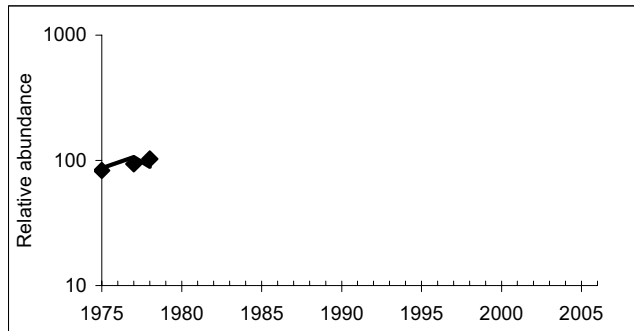
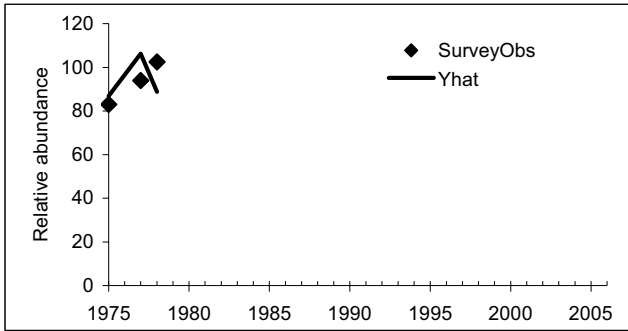


Figure B5-17 continued.

LPUE

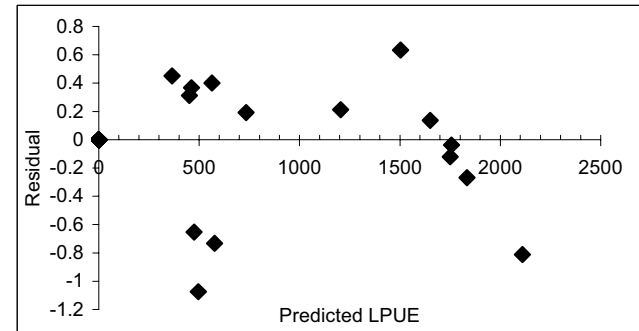
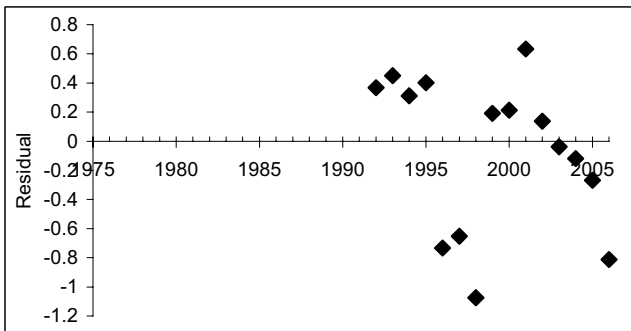
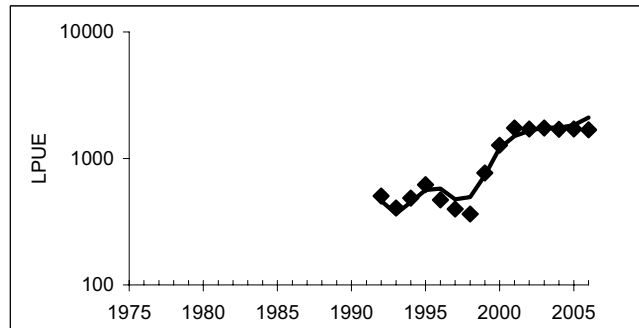
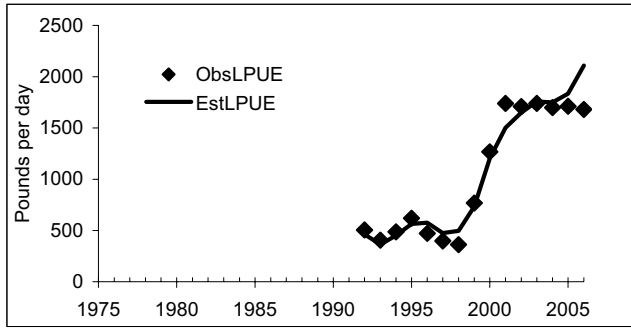


Figure B5-17 continued.

Landings

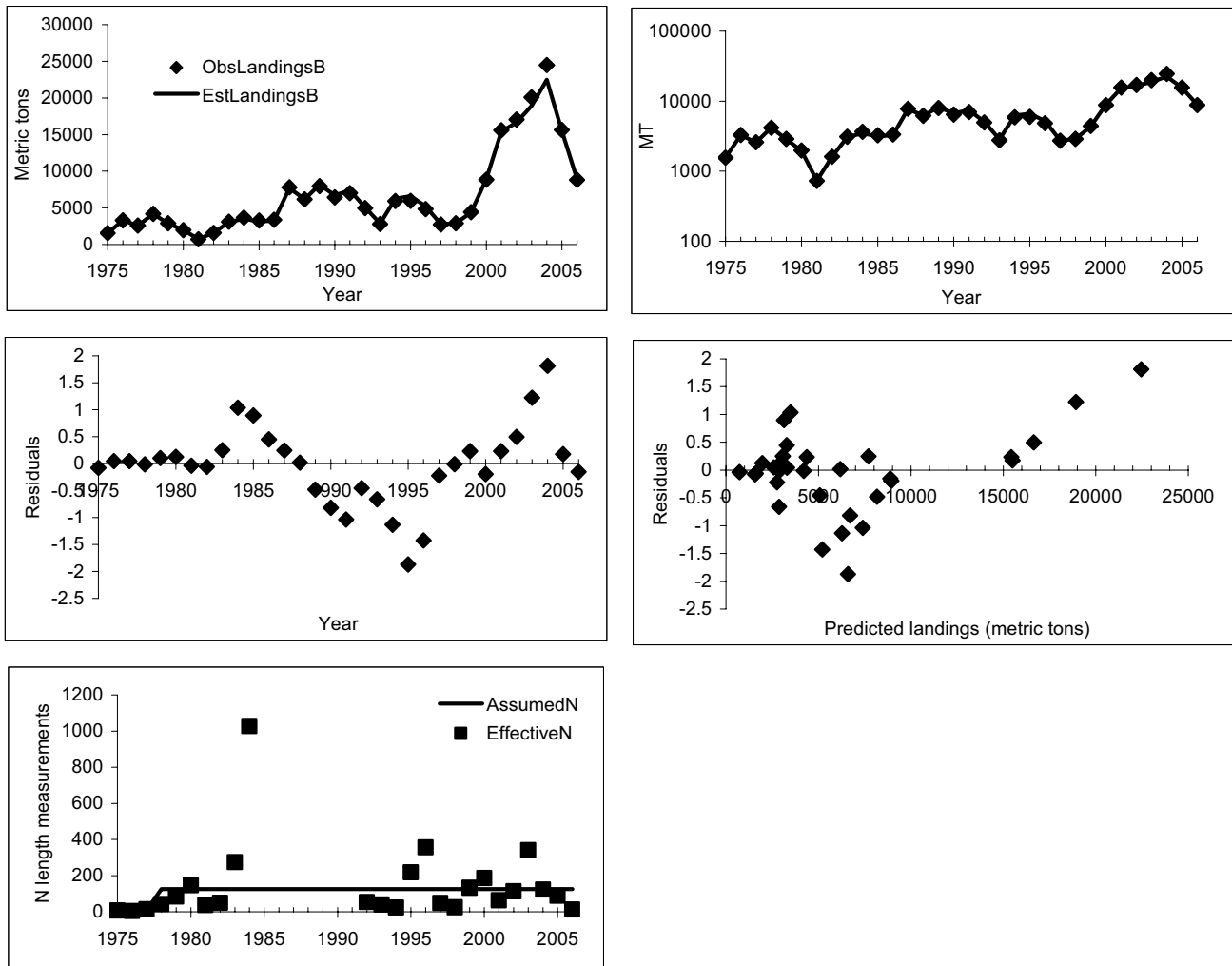


Figure B5-17 continued.

Fall bottom trawl (for illustration only, does not affect CASA estimates)

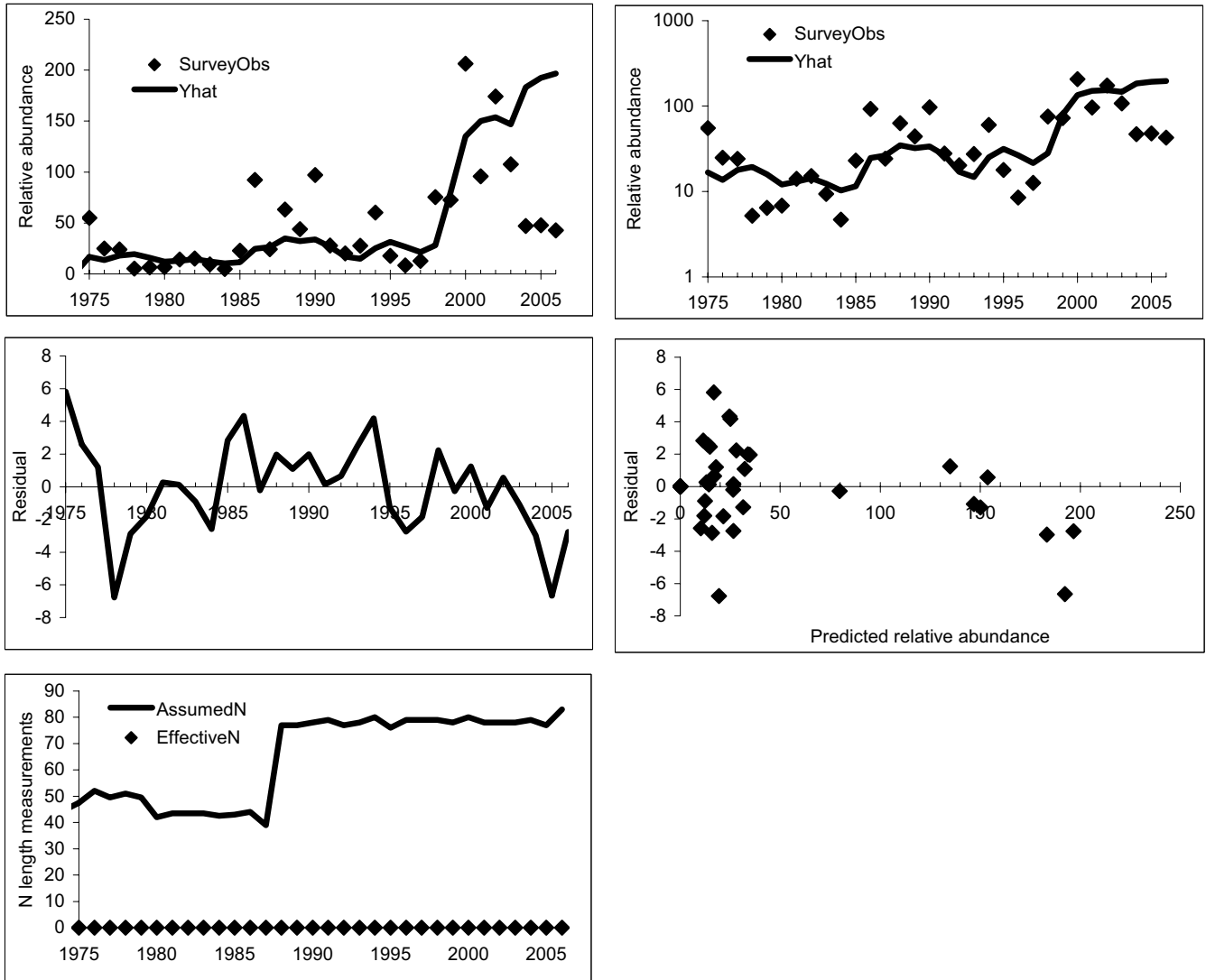


Figure B5-17 continued. Fall and spring bottom trawl data were used in the CASA model for sea scallops in the Mid-Atlantic for comparison only. Survey trend and shell height data from the fall and spring bottom trawl survey did not affect model estimates.

Spring bottom trawl (for illustration only, does not affect CASA estimates)

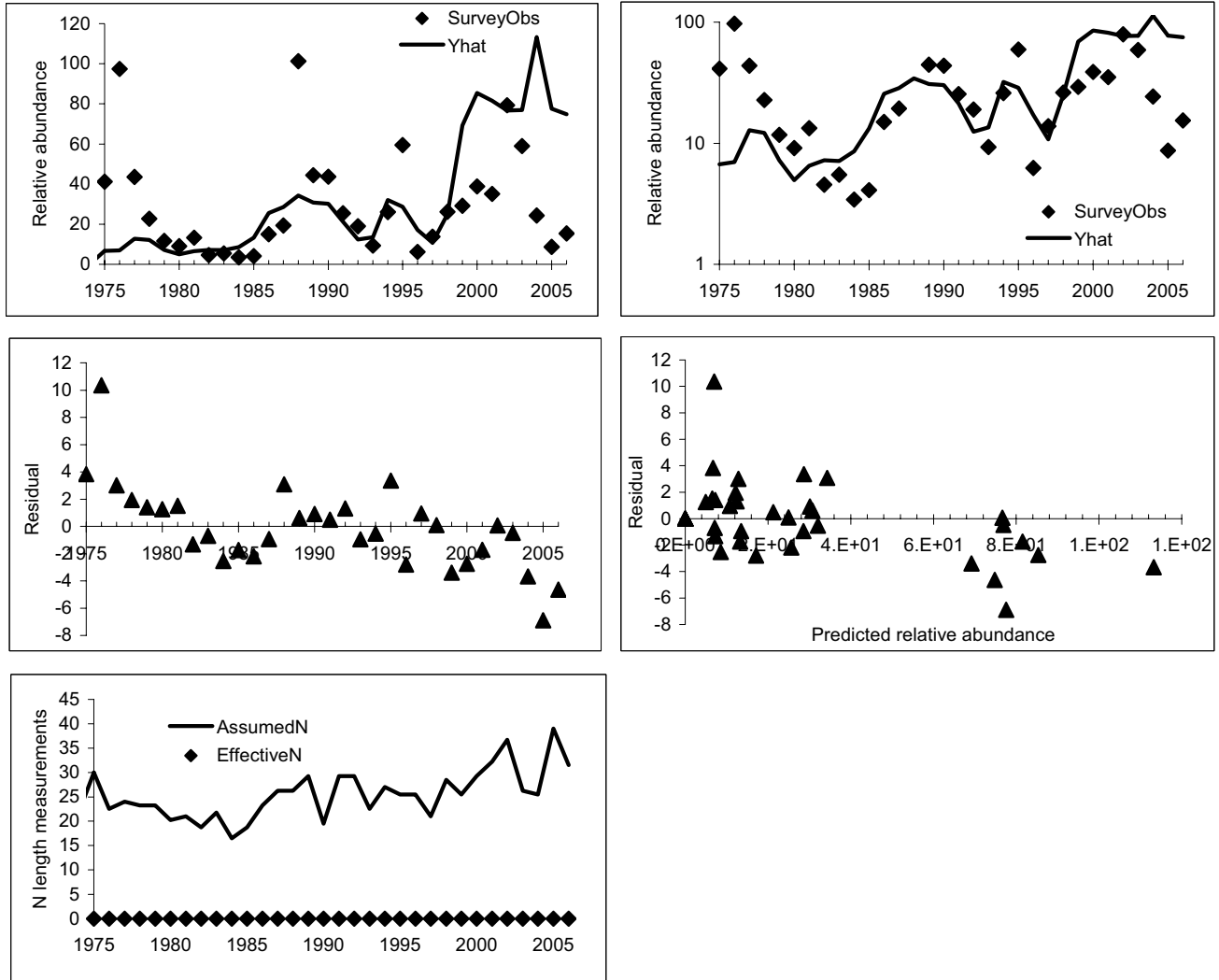
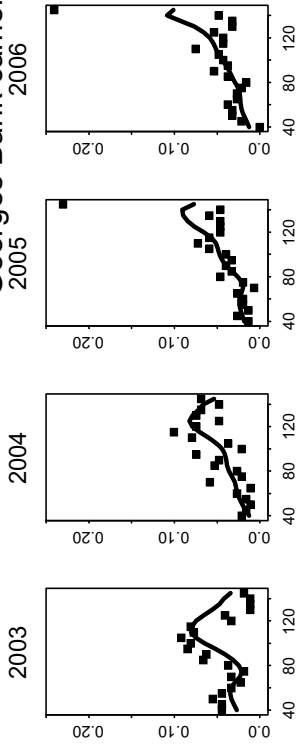


Figure B5-17 continued. Fall and spring bottom trawl data were used in the CASA model for sea scallops in the Mid-Atlantic for comparison only. Survey trend and shell height data from the fall and spring bottom trawl survey did not affect model estimates.

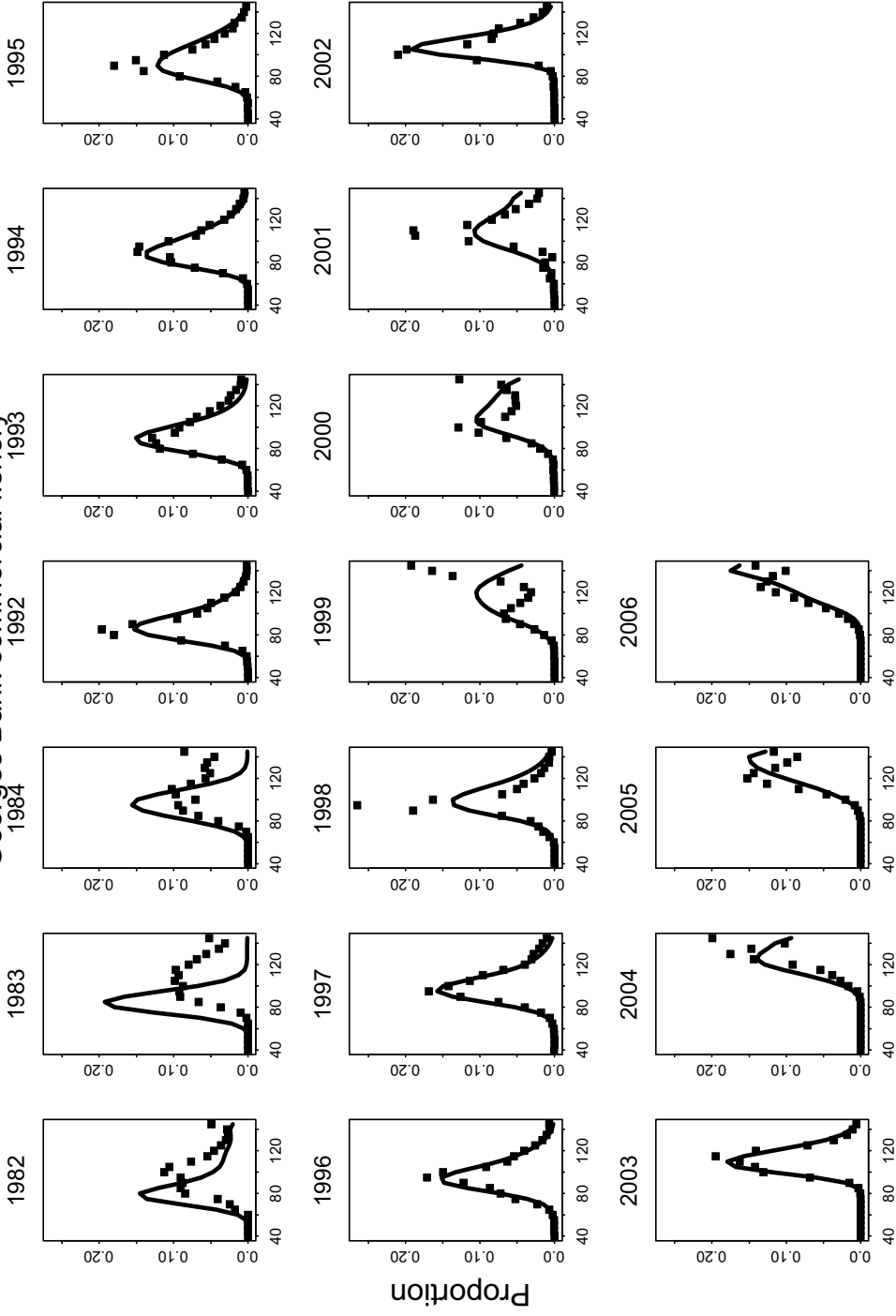
Georges Bank camera survey



Shell length (mm)

Figure B5-18. Observed and predicted shell height composition data from the basecase CASA model for sea scallops on Georges Bank.

Georges Bank commercial fishery



Shell length (mm)

Figure B5-18 continued.

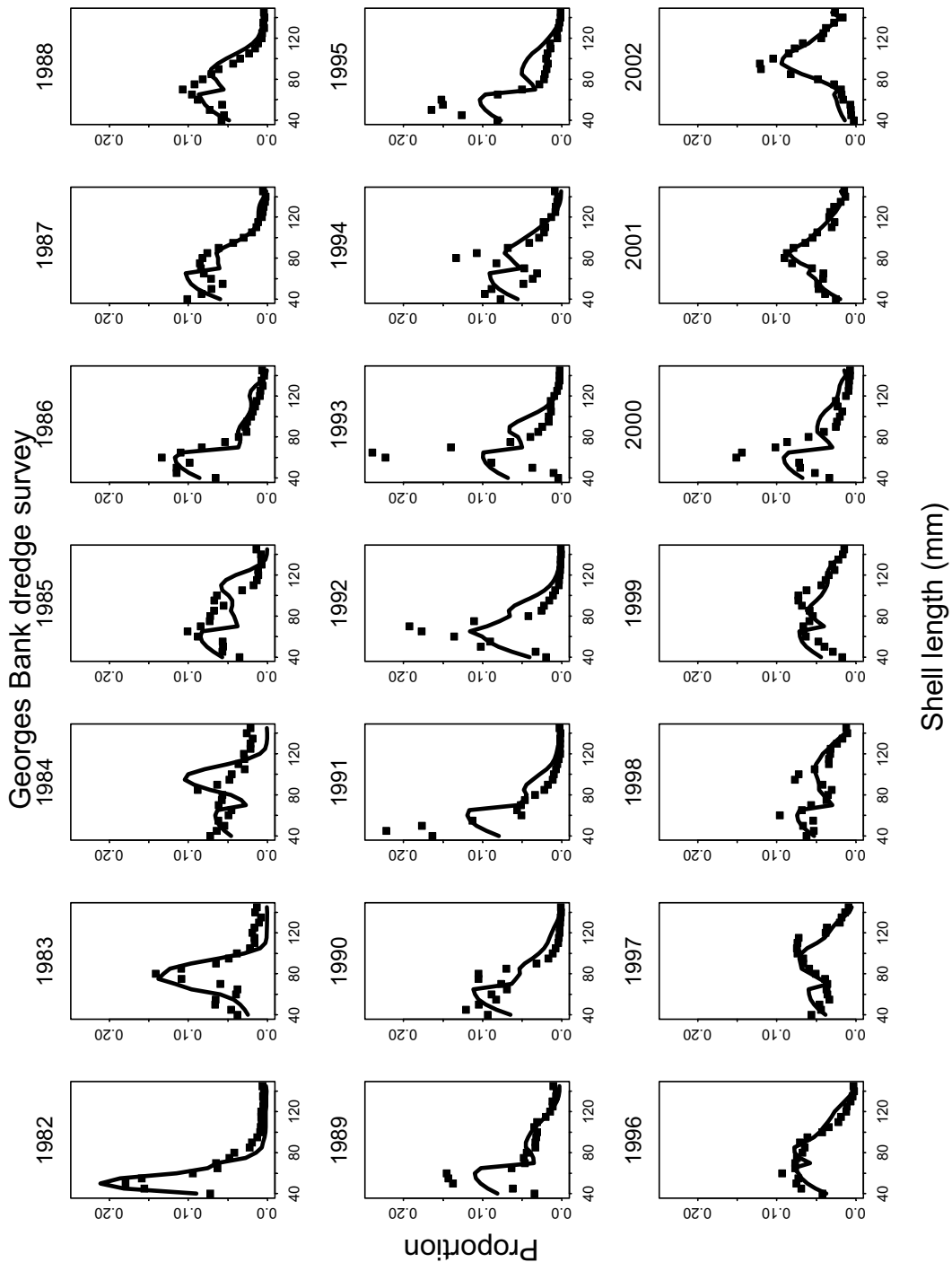
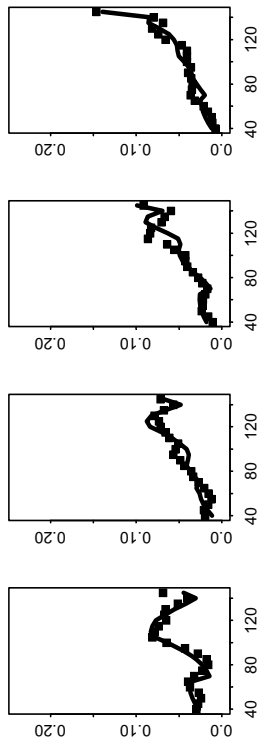


Figure B5-18 continued.

Georges Bank dredge survey
2003 2004 2005 2006

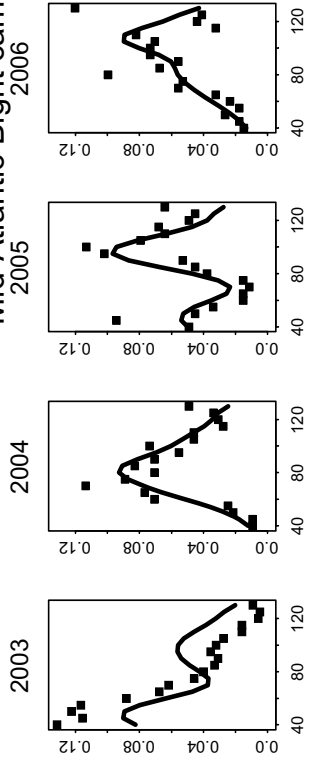


Proportion

Shell length (mm)

Figure B5-18 continued.

Mid-Atlantic Bight camera survey



Proportion

Shell length (mm)

Figure B5-19. Observed and predicted shell height composition data from the basecase CASA model for sea scallops in the Mid-Atlantic.

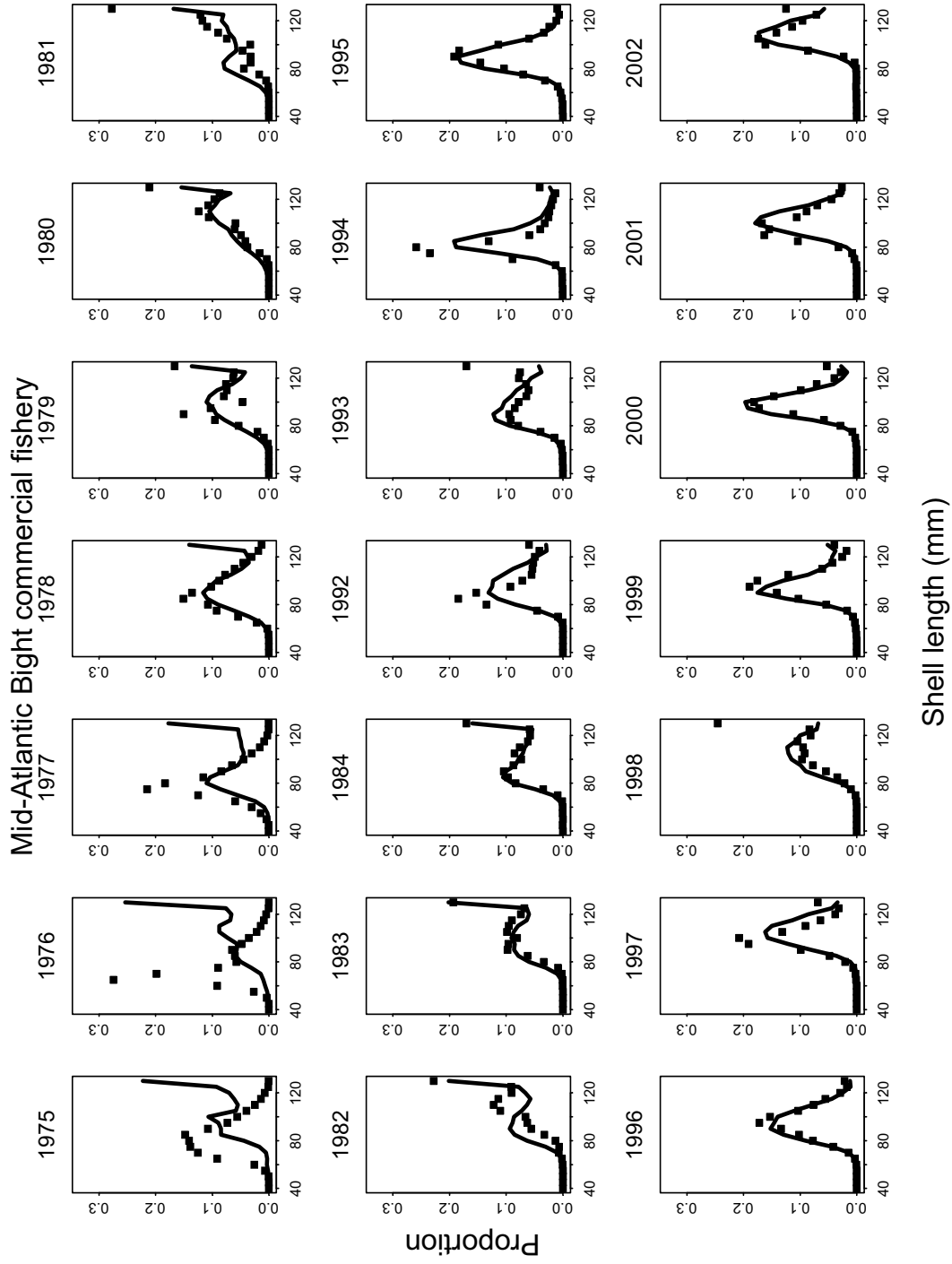
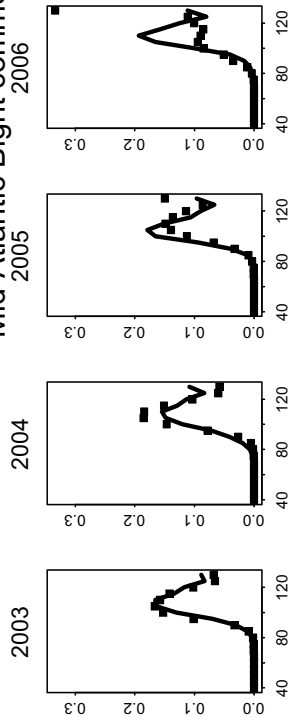


Figure B5-19 continued.

Mid-Atlantic Bight commercial fishery



Proportion

Shell length (mm)

Figure B5-19 continued.

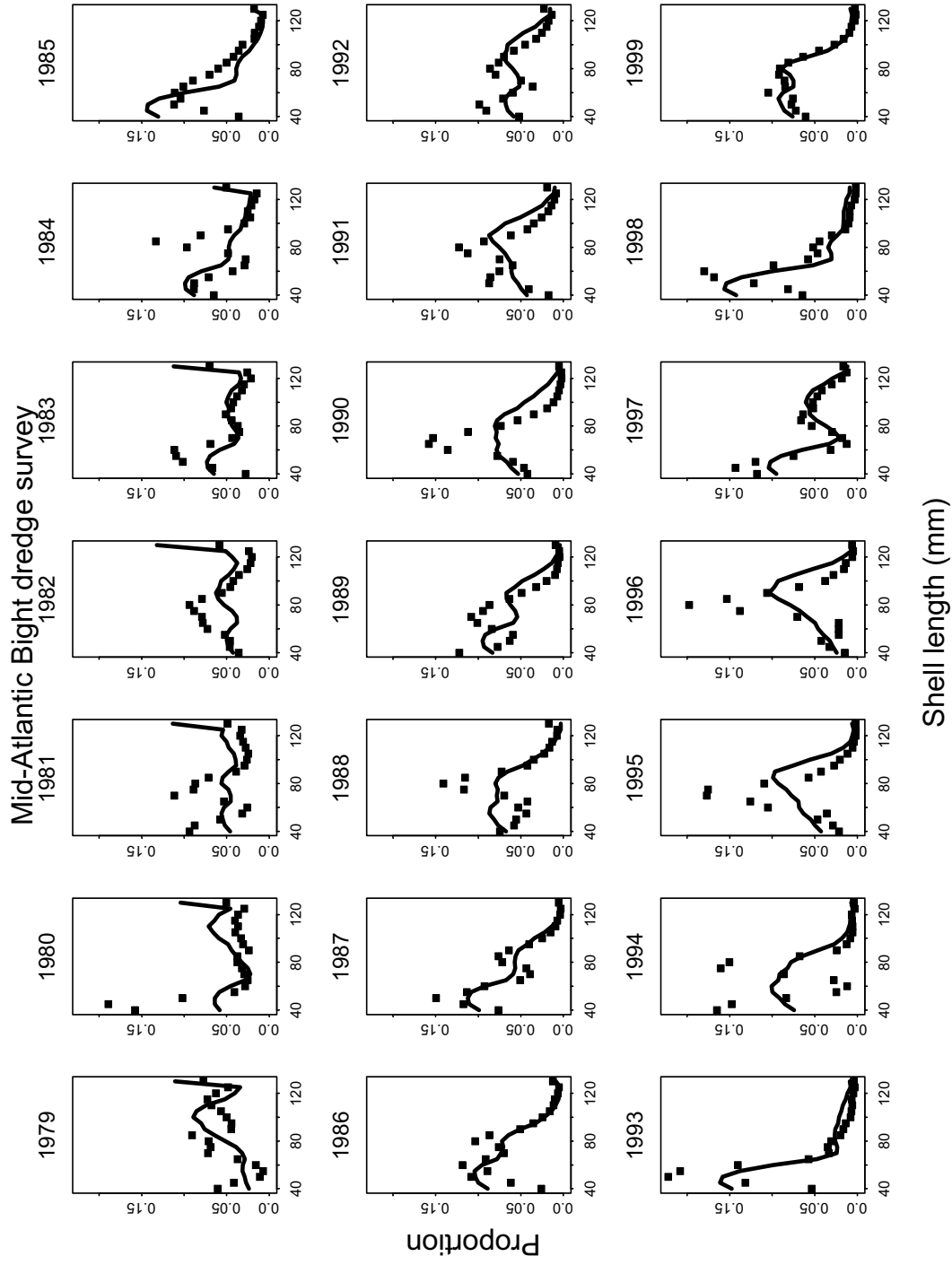
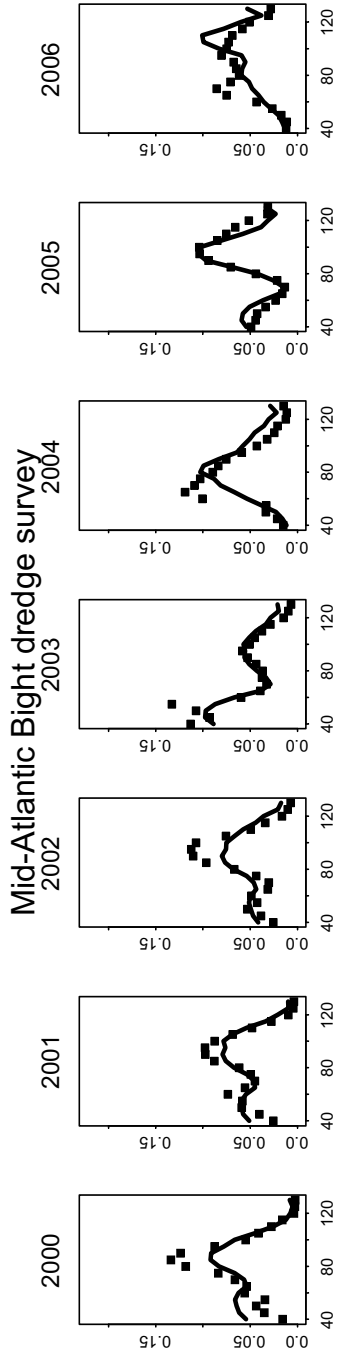


Figure B5-19 continued.

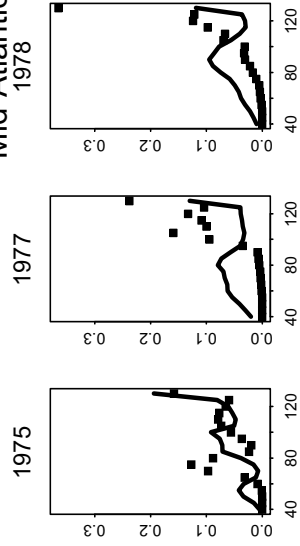


Proportion

Shell length (mm)

Figure B5-19 continued.

Mid-Atlantic Bight 10 ft unlined dredge



Proportion

Shell length (mm)

Figure B5-19 continued.

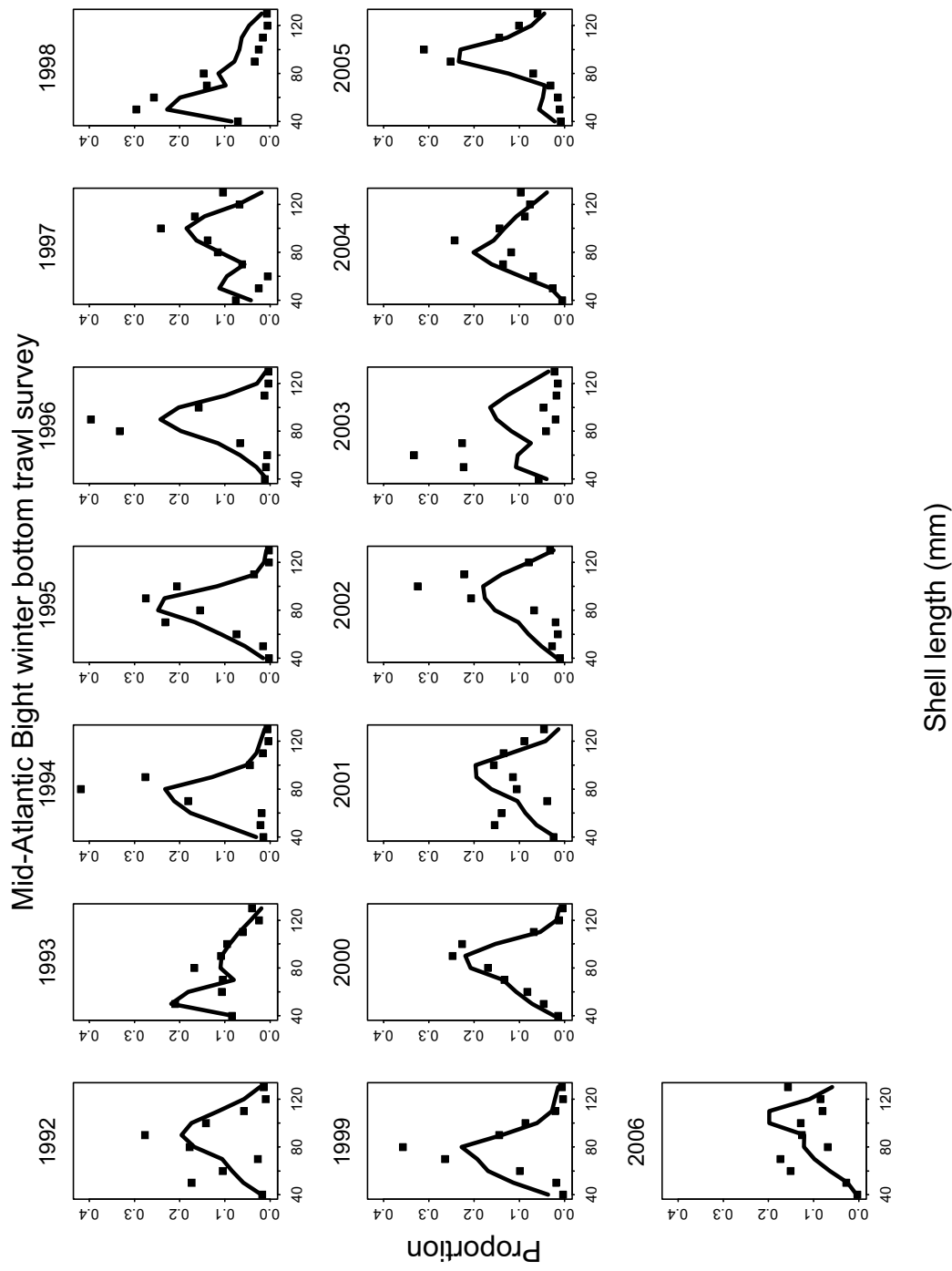


Figure B5-19 continued.

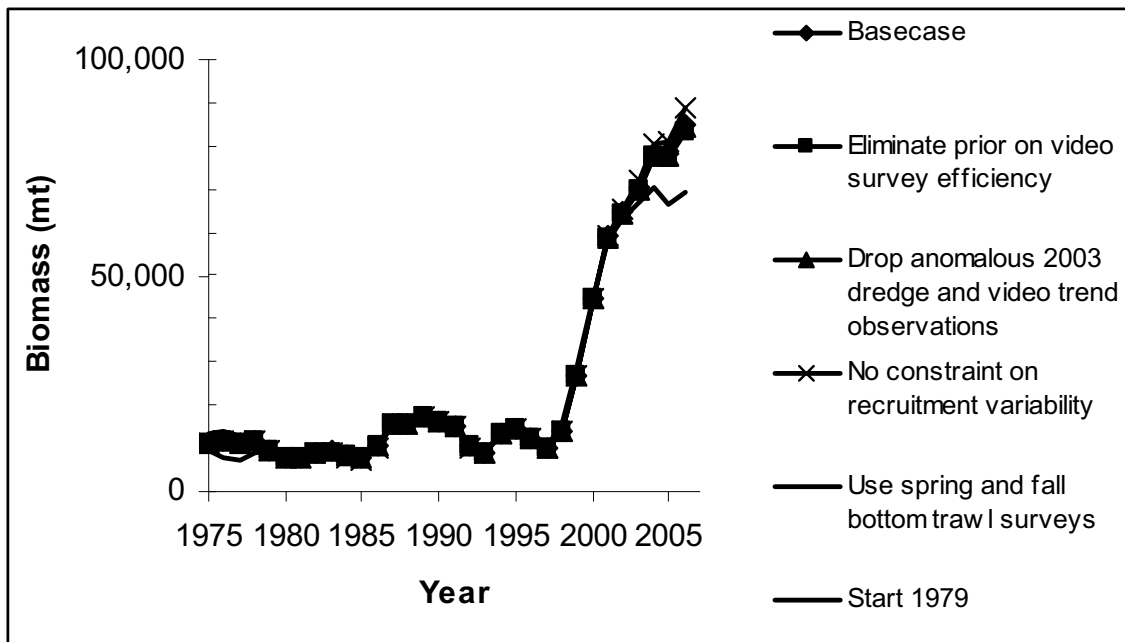


Figure B5-20. Biomass estimates for sea scallops in the Mid-Atlantic Bight from the basecase and a variety of sensitivity analysis runs.

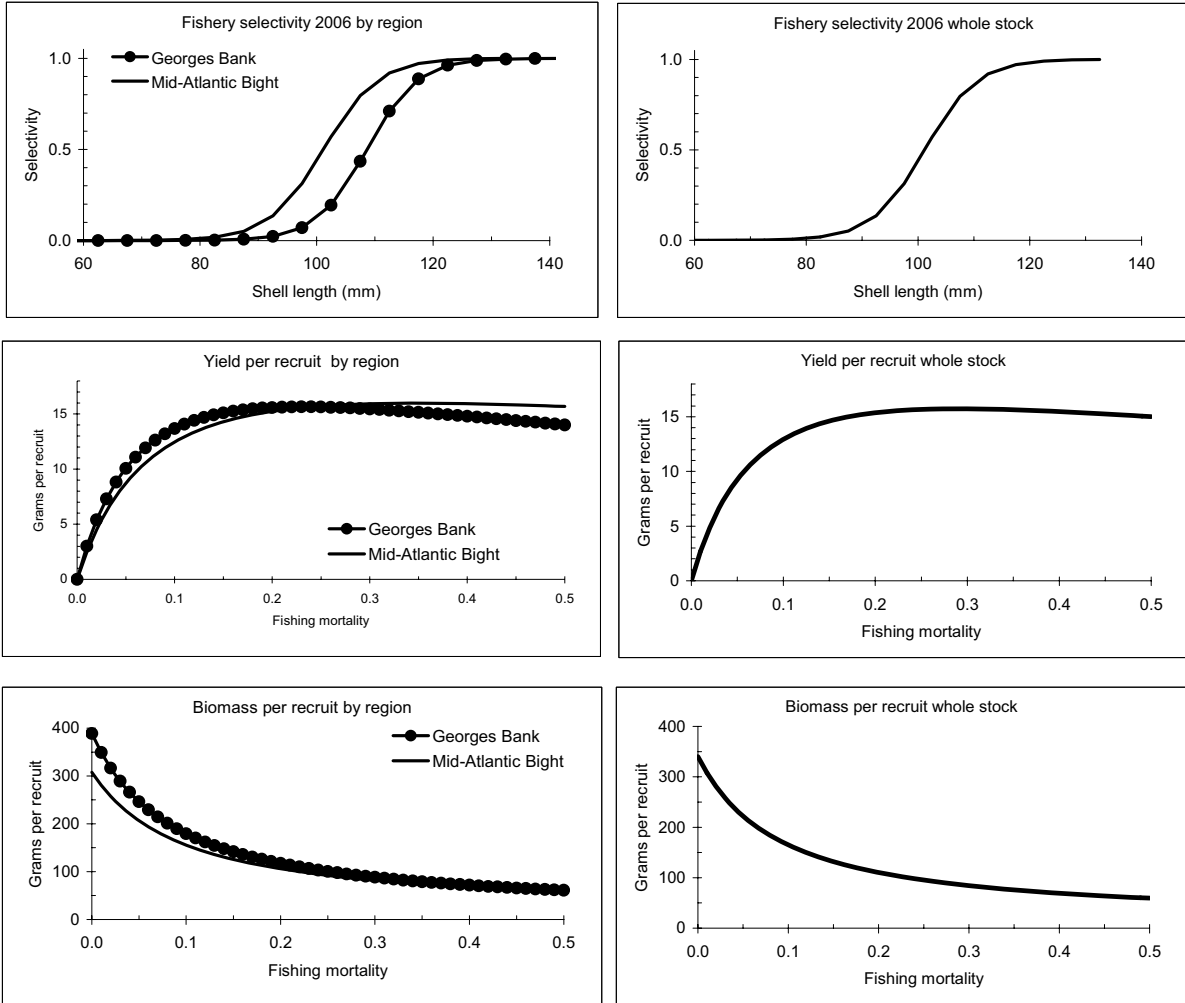


Figure B6-1. Fishery selectivity, yield per recruit, and biomass per recruit curves for the Mid-Atlantic Bight, Georges Bank and for the whole stock. Estimates for the whole stock are averages weighted by median recruitment during 1983-2006 in the Mid-Atlantic Bight and Georges Bank. The whole stock selectivity curve was not used in calculations and is shown as information only.

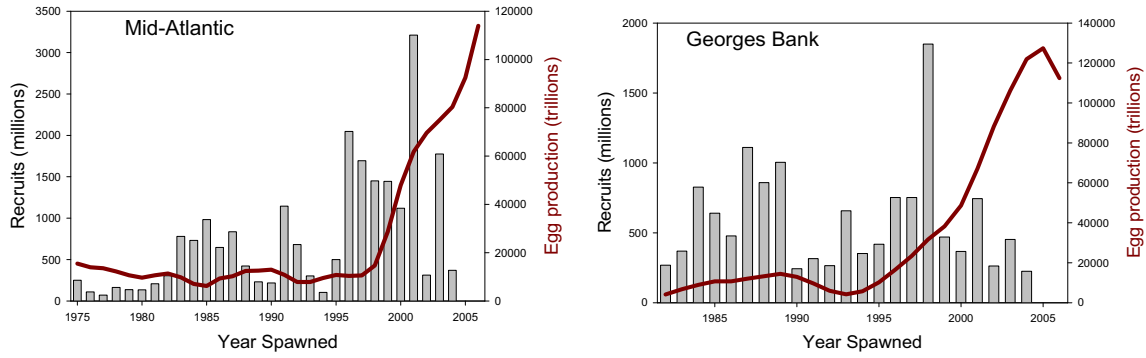


Figure B6-2. Sea scallop recruitment (bars) and egg production (lines) in the Mid-Atlantic and Georges Bank.

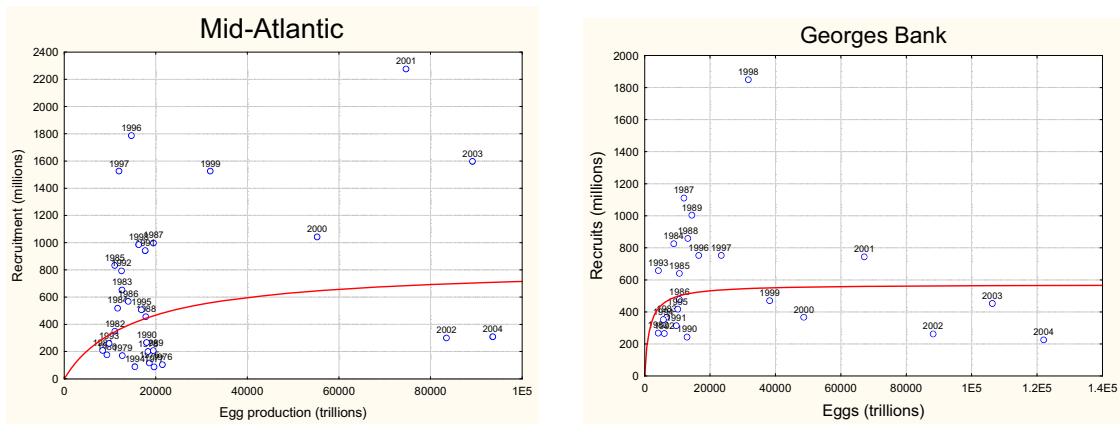


Figure B6-3. Stock-recruit plots and estimated Beverton-Holt stock-recruitment fits for the Mid-Atlantic and Georges Bank.

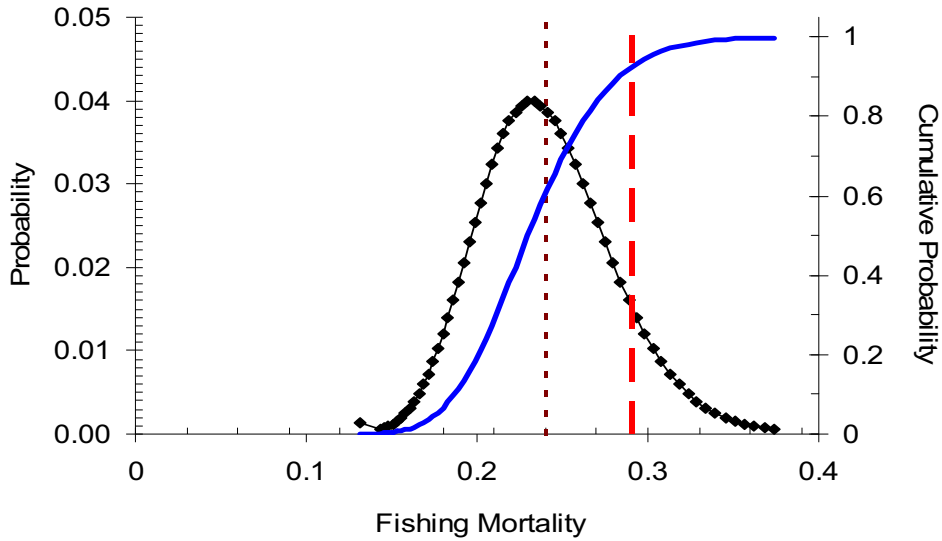


Figure B7-1. Uncertainty in estimated fishing mortality during 2006 from the (baseline) CASA model run. The proposed (dashed line) and current (dotted line) overfishing thresholds are shown.

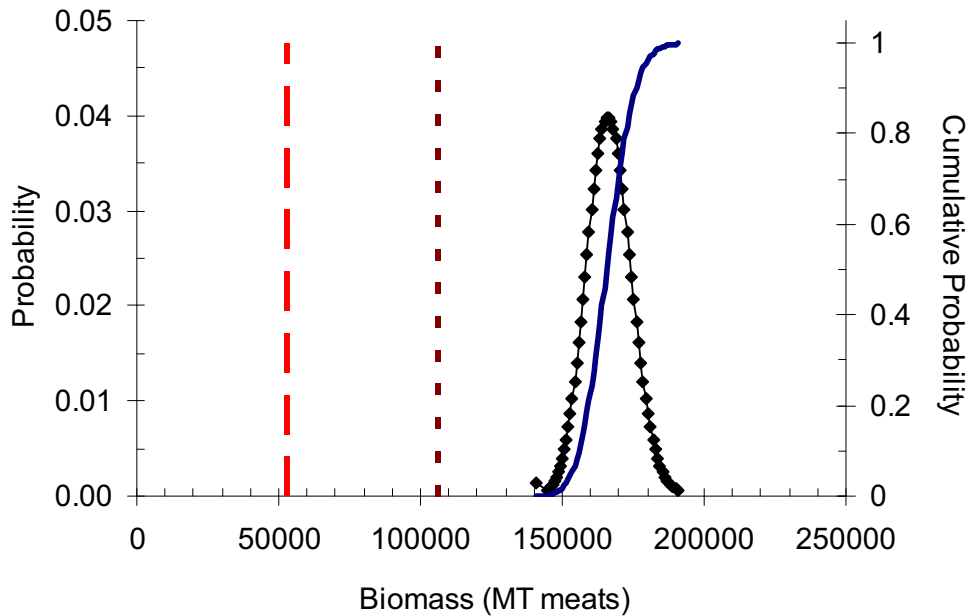


Figure B7-2. Uncertainty in estimated biomasses from the (baseline) CASA model run. The new proposed biomass target (dotted line) and threshold (dashed line) are also shown.

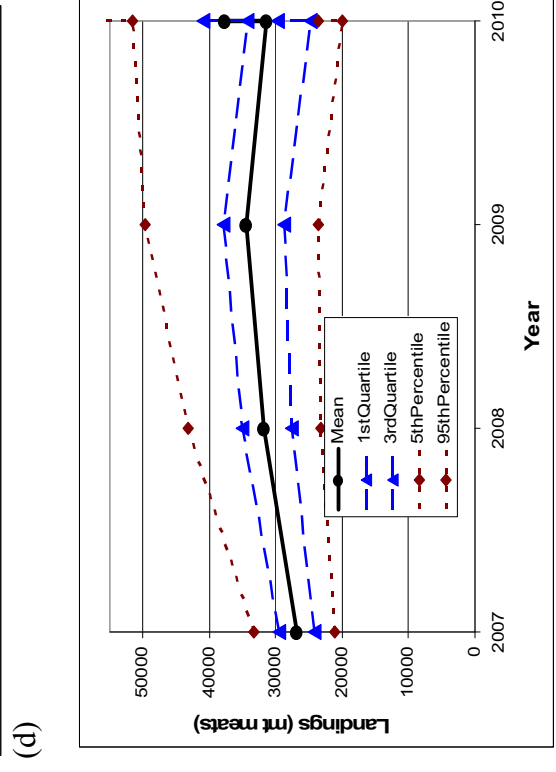
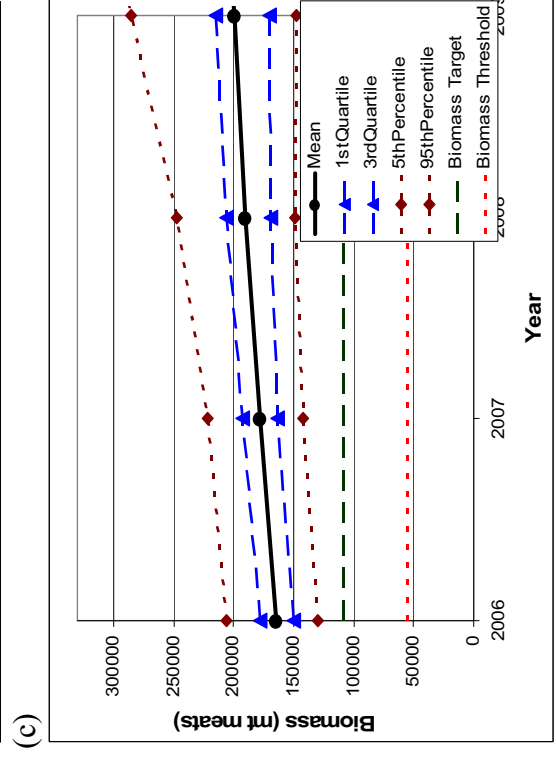
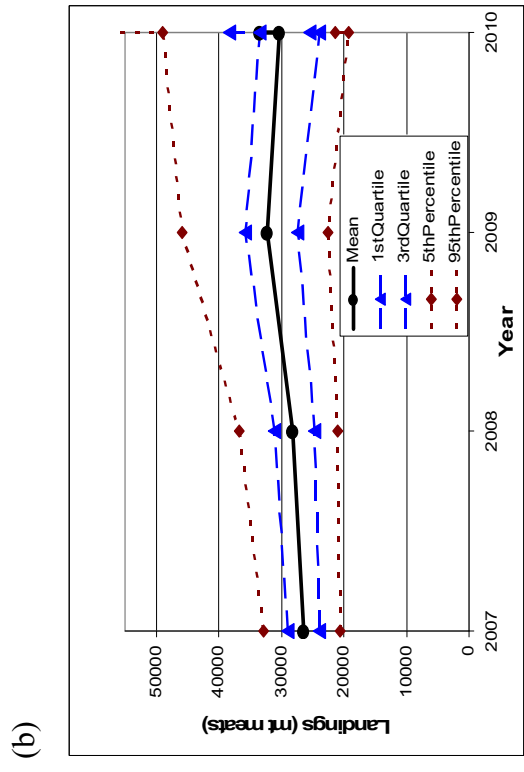
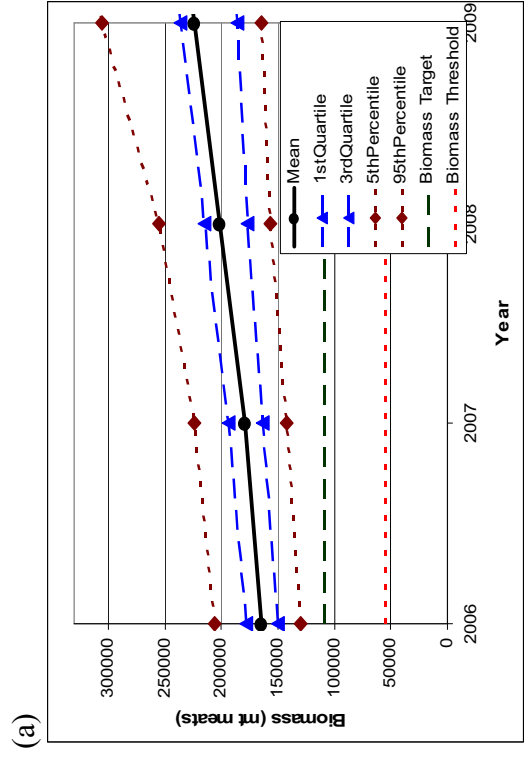


Figure B8-1. Short-term projections of the US sea scallop biomass (a, c) and landings (b, d) assuming 2007-2009 fishing mortalities are $F = 0.20$ (a, b) and $F = 0.24$ (c, d). The mean of 400 stochastic runs is shown, together with the 5th, 25th, 75th, and 95th percentiles. The simulations were initialized in July 2006, so that biomass estimates are for July, and landings in a given year represent landings from July of the previous year until June of that year.

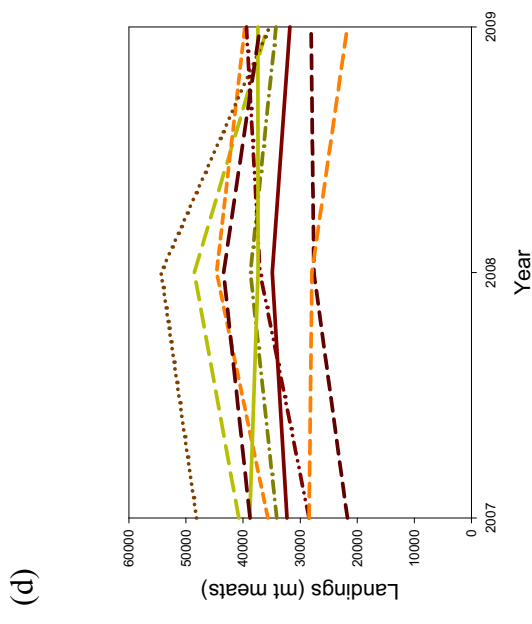
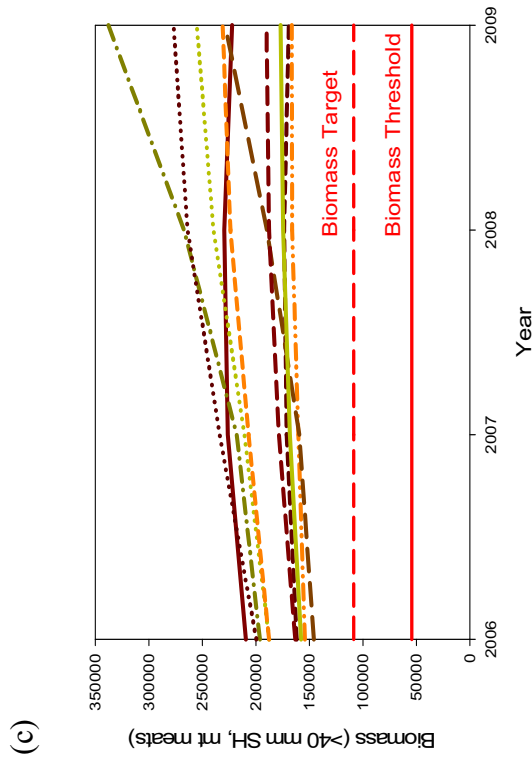
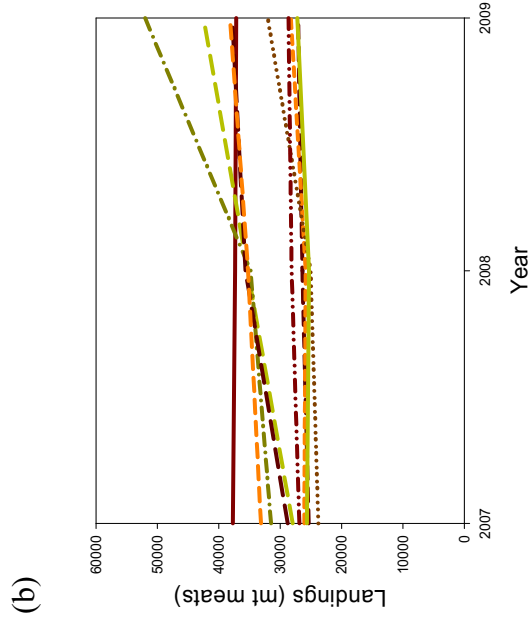
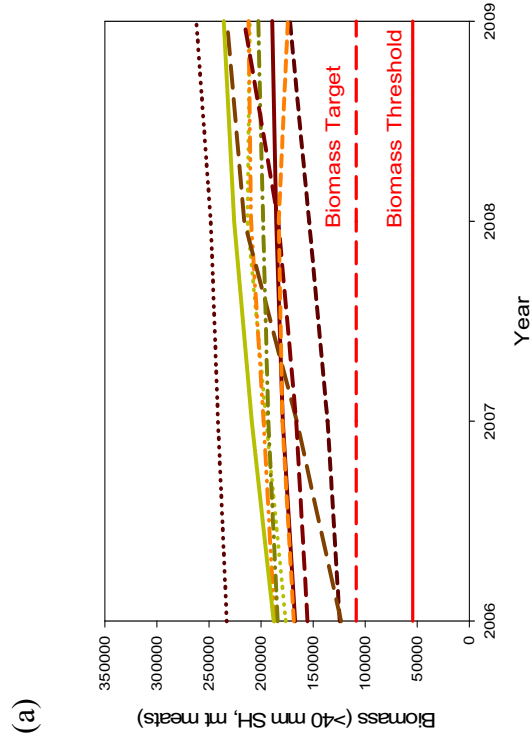


Figure B8-2. Ten example trajectories of the short-term sea scallop projections with differing initial population bootstraps and stochastic recruitment streams. Sea scallop biomass (a, c) and landings (b, d) assume 2007-2009 fishing mortalities are $F = 0.20$ (a, b) and $F = 0.24$ (c, d). The simulations were initialized in July 2006, so that biomass estimates are for July, and landings in a given year represent landings from July of the previous year until June of that year.

APPENDIX B1: Invertebrate subcommittee meetings and participants

The Invertebrate Subcommittee held four meetings during 2007 on March 8-9, April 9-11, April 30-May 1, and May 8-9 to work on the sea scallop stock assessment for SAW/SARC-45. All of the meetings were held in the Stephen H. Clark Conference Room at the Northeast Fisheries Science Center in Woods Hole. Participating members and affiliations are listed below.

Larry Jacobson (Invertebrate Subcommittee Chair, NEFSC, Woods Hole)
Dvora Hart (Sea Scallop Assessment Lead Scientist, NEFSC, Woods Hole)
Chuck Adams (School for Marine Science and Technology, UMASS Dartmouth)
Andrew Applegate (New England Fishery Management Council)
Deirdre Boelke (New England Fishery Management Council)
Danielle Brezinski (University of Maine)
Antonie Chute (Rapporteur, NEFSC, Woods Hole)
Chad Demarest (Massachusetts Marine Fisheries Institute)
Bill Du Paul (Virginia Institute of Marine Science)
Demet Haksever (New England Fishery Management Council)
Brad Harris (School for Marine Science and Technology, UMASS Dartmouth)
Chad Keith (NEFSC, Woods Hole)
Chris Legault (NEFSC, Woods Hole)
Michael Marino (School for Marine Science and Technology, UMASS Dartmouth)
Bob Mohn (Invited outside expert, Department of Fisheries and Oceans, Canada)
Paul Nitschke (NEFSC, Woods Hole)
Victor Nordahl (NEFSC, Woods Hole, MA)
Cate O'Keefe (School for Marine Science and Technology, UMASS Dartmouth)
Paul Rago (NEFSC, Woods Hole)
Stacy Rowe (NEFSC, Woods Hole)
David Rudders (Virginia Institute of Marine Science)
Chris Sarro (School for Marine Science and Technology, UMASS Dartmouth)
David Simpson (NEFMC and Connecticut Marine Fisheries Division)
Stephen Smith (Invited outside expert, Department of Fisheries and Oceans, Canada)
Ron Smolowitz (Fisheries Survival Fund)
Kevin Stokesbury (School for Marine Science and Technology, UMASS Dartmouth)
Jim Weinberg (NEFSC, Woods Hole)
Noelle Yochum (Virginia Institute of Marine Science)

APPENDIX B2: Verification of annual shell growth increments

This appendix will examine the question of whether the growth increments obtained from shell rings are truly annual, and whether the growth matrices obtained from shell growth increment data gives appropriate predictions of growth. Early work examining monthly shell samples (Stevenson and Dickie 1954), or comparing growth from shell rings to tagging (Merrill et al. 1966) concluded that shell growth rings are laid down annually. Kranz et al. (1984) used stable isotope analysis to age two shells in the Mid-Atlantic Bight, and suggested that sea scallops lay down two shell rings a year. However, this conclusion is only really supported by one of their two shells. Stable isotope analysis of two shells from Brown's Bank was supportive of the 1 ring per year hypothesis (Tan et al. 1988).

Here, we followed the growth of large cohorts found in sites in the closed areas, to test whether the shell increments collected from these cohorts matched the observed growth. Four stations where large sets of small scallops were observed were selected for this study, two in Closed Area II, one in Nantucket Lightship Closed Area, and one in the Elephant Trunk Closed Area (Figure 1). These stations were revisited in subsequent years to obtain size-frequency information. Starting in 2003 (2004 for sites #2 and #4) between 60-100 shells were saved at each station for growth analysis, as described in Appendix B3. Growth increment matrices were constructed for each site based on shells collected there. Growth from one year could then be projected to the next year and compared to the observed size frequency for that year to evaluate whether the growth matrix gave accurate predictions. In some cases, size-frequencies were not available for some years, in which case a multiyear projection was made by applying the matrix to the original size frequency the appropriate number of times.

Site #1 (Closed Area II)

This site was repeatedly sampled after a large set of small scallops (1998 year class) was observed there in 2000. In 2004 and 2006, the number of scallops caught at this site was small (141 in 2004, 81 in 2006), either because the dredge missed the main bed, or (in 2006), because of heavy fishing after the area was reopened. These years were therefore dropped from the analysis. The growth of scallops at this site during the remaining years is shown in Figure 2. Figure 3 compares the observed (normalized) size-frequency with that predicted from the previous observation and the site-specific growth matrix. In all cases the fit was very good.

Site #2 (Closed Area II)

This site, which is close to Station #1, was also selected because a large set of small scallops was observed there in 2000. The site was resampled in 2002, 2004, 2005, and 2006 (Figure 4). Comparison of the observed size-frequencies to that projected using the growth matrix were good with the exception of the projection from 2002-2004, where the projected sizes were somewhat greater than that observed (Figure 5).

Site #3 (Nantucket Lightship Area)

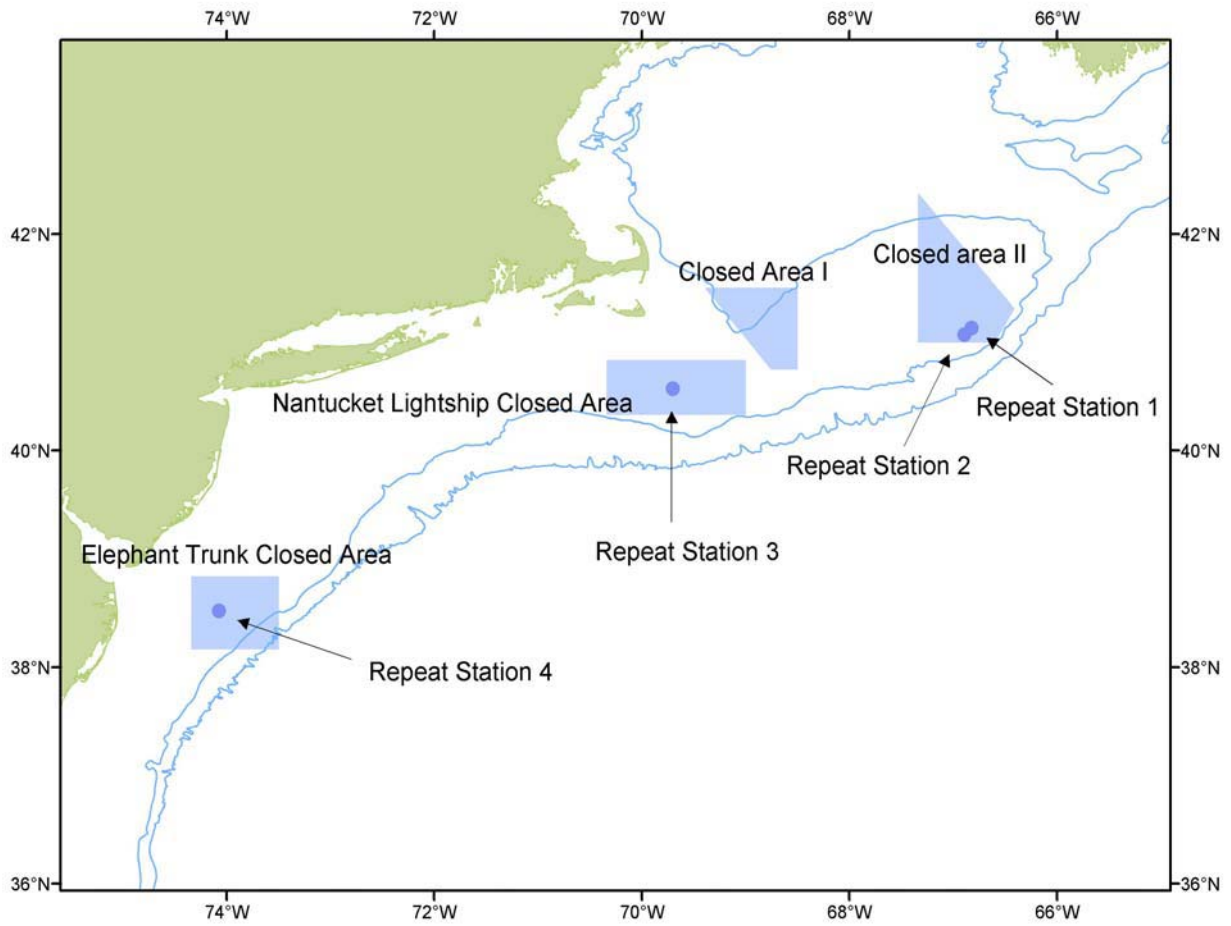
This site was originally sampled in 1999, and was revisited in 2000, 2003, 2004, 2005, and 2006 (Figure 6). In 2000, a second strong cohort was observed in addition to the one observed in 1999. Comparisons between predicted and observed growth was always quite good (Figure 7).

Site #4 (Elephant Trunk Closed Area)

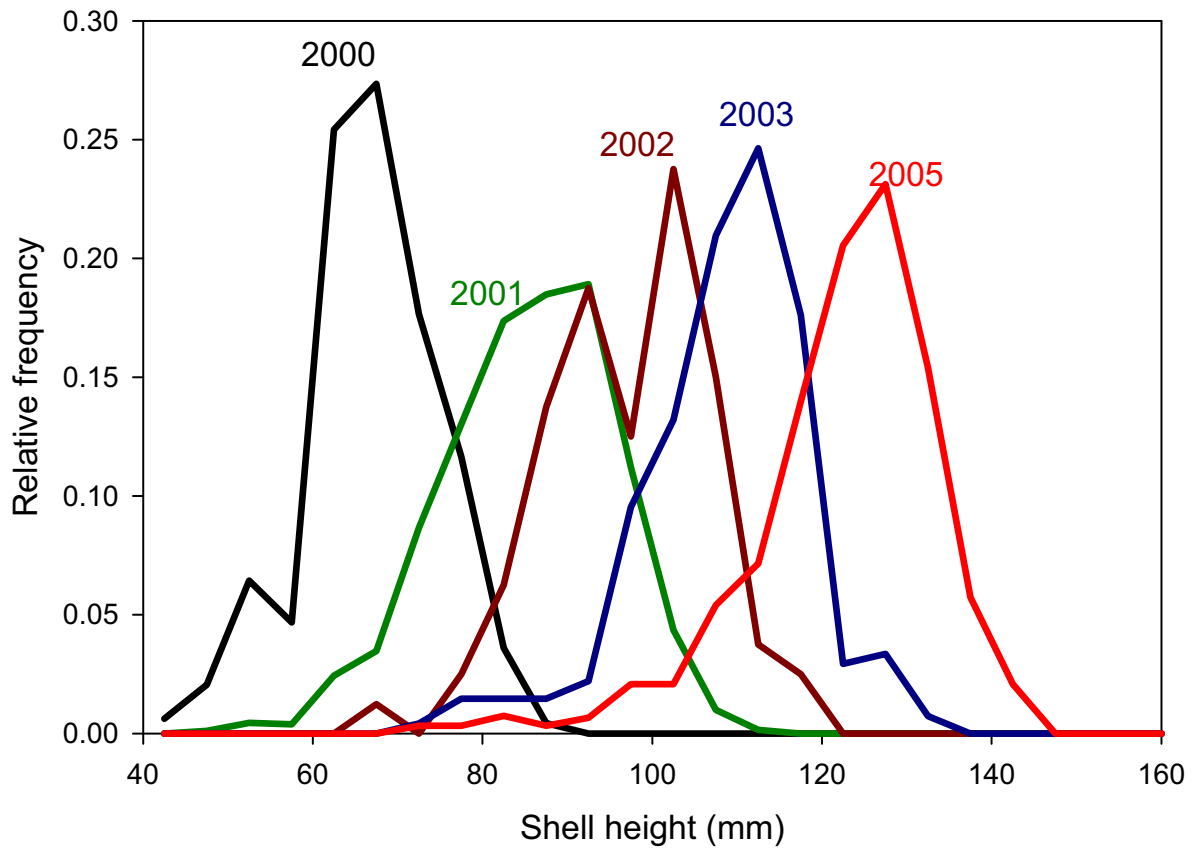
This site was first sampled in 2003, a year before this area was closed. However, nearly all of the scallops observed in 2003 were well below commercial size, so that the fishing that occurred in this area until it was closed in July 2004 should not have affected growth or mortality at this site. It was resampled each year thereafter (2004, 2005, 2006, Figure 8). There was little growth between 2005 and 2006, which was also observed in the Elephant Trunk as a whole. The growth between these years was inconsistent with that observed between 2004 and 2005. Comparisons between observed and projected size-frequencies showed good agreement in 2004 and 2005, but the projection from 2005 to 2006 predicted considerably greater growth than actually occurred (Figure 9). Projections were also made based on the Kranz et al. (1984) hypothesis that two growth rings are laid down each year, so that the growth matrix was applied twice to obtain the predicted shell heights in the next year (Figure 10). The observations do not support Kranz et al.'s hypothesis of semi-annual rings.

Discussion and Conclusions

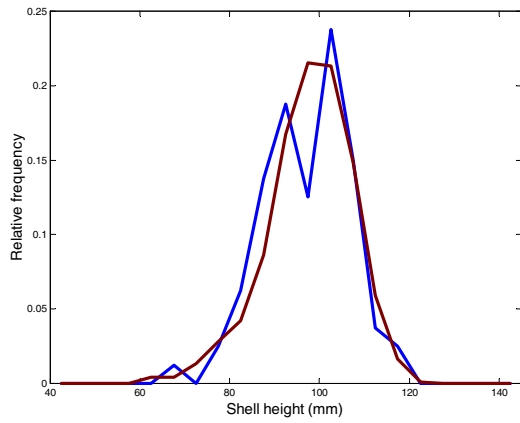
In all but two of the 16 comparisons made here, size-frequencies predicted from growth matrices were in good agreement with observations. One case was a modest deviation at site #2 for a two-year projection between 2002 and 2004. The other was a stronger deviation in site #4 between 2005 and 2006. None of the shells collected at this site would have reflected any growth since the last shell ring (probably in the fall of 2005) was laid down, since the partial increment from the last ring to the edge of the shell was not used. Thus, the projected sizes reflect what would have occurred if growth during 2005-6 was the same as in previous years. The deviation between observed and predicted growth does not imply that the shell rings are not annual. Rather, they indicate a change in growth between 2005 and 2006, probably related to environmental conditions (e.g., food supply). None of the data are consistent with the Kranz et al. hypothesis of semi-annual rings, since that would predict much faster growth than was observed. It can be concluded that growth matrices derived from shell ring data, under the assumption that the growth lines are laid down annually, generally give good predictions for growth, and are appropriate for use in this assessment.



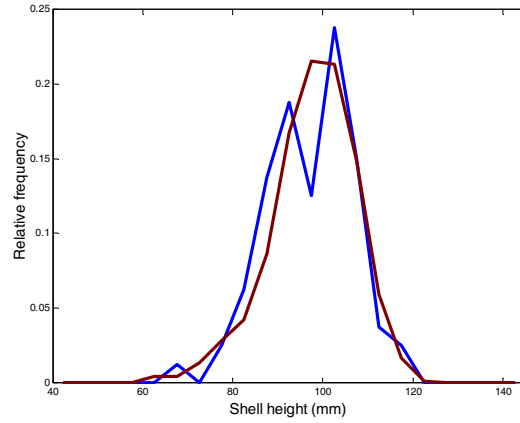
APPENDIX B2 Figure 1. Locations of the four repeat sites in this study.



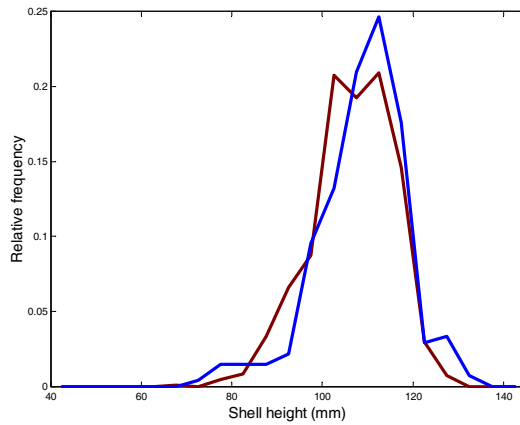
APPENDIX B2 Figure 2. Normalized size-frequencies by year at site #1 (Closed Area II).



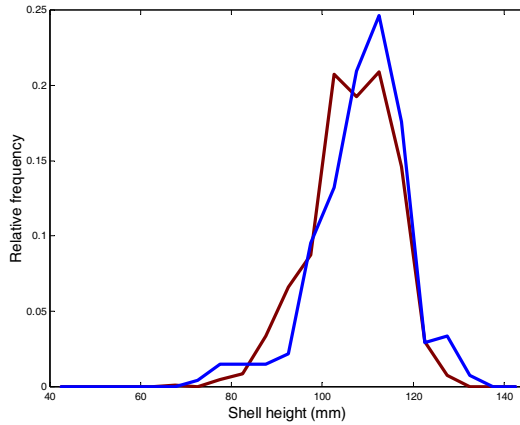
2000 to 2001



2001 to 2002

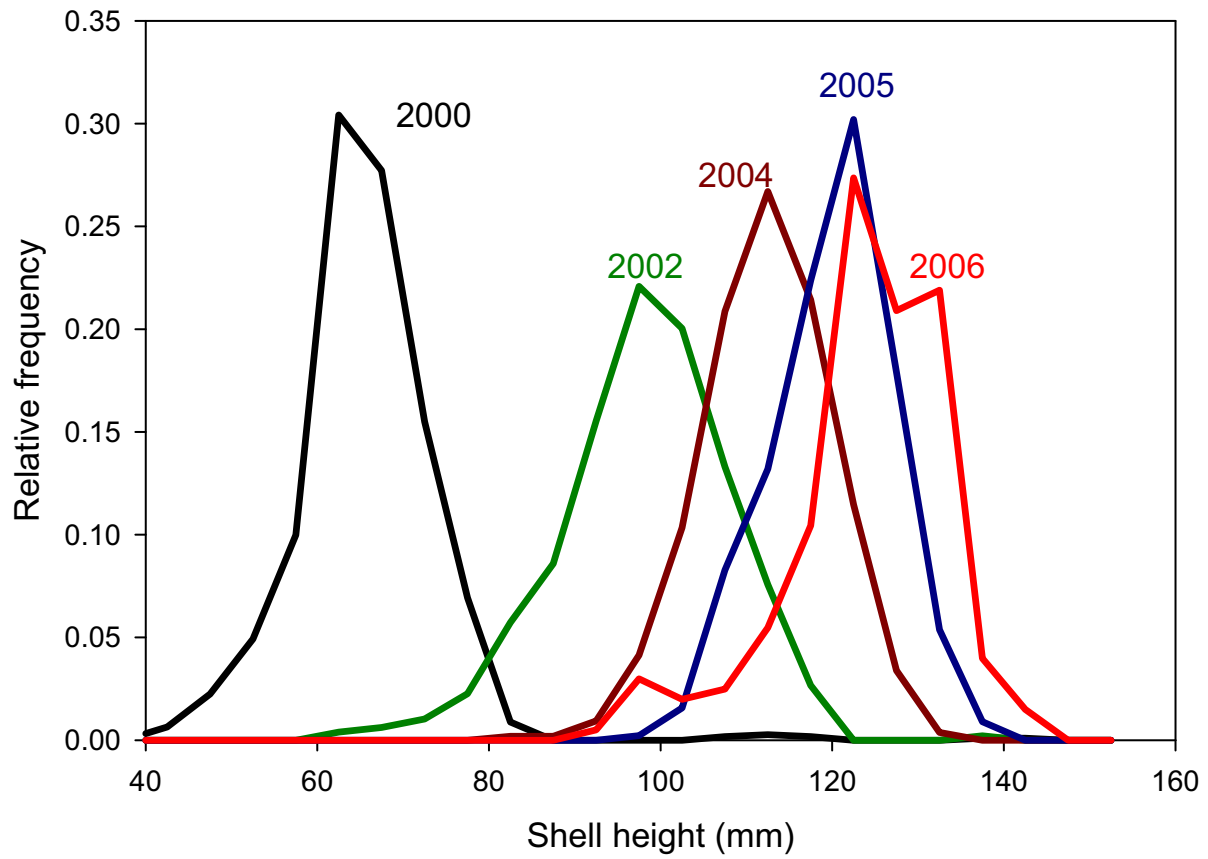


2002 to 2003

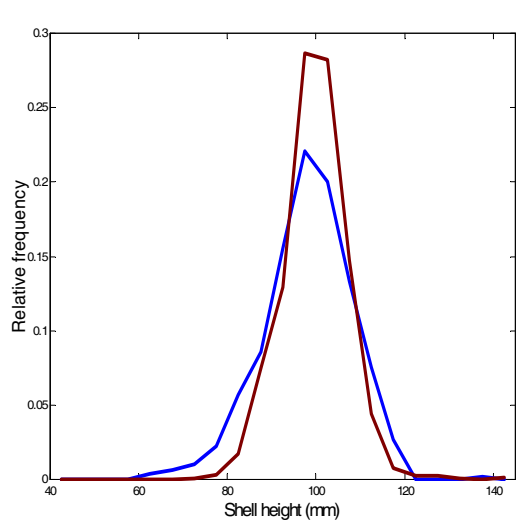


2003 to 2005

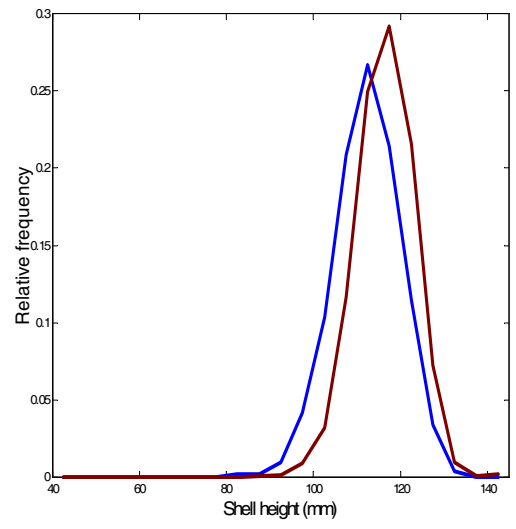
APPENDIX B2 Figure 3. Comparison between observed (solid blue line) and projected (dashed-dotted brown line) normalized size-frequencies at site #1 (Closed Area II).



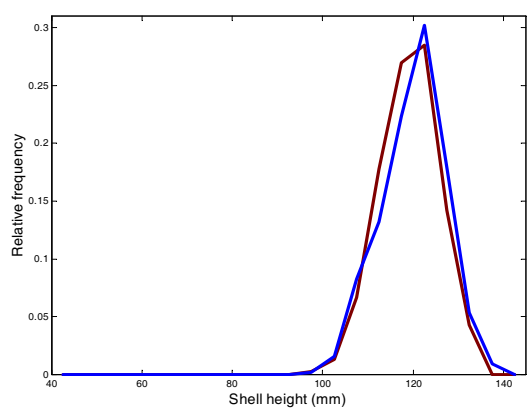
APPENDIX B2 Figure 4. Normalized size-frequencies at site #2 (Closed Area II).



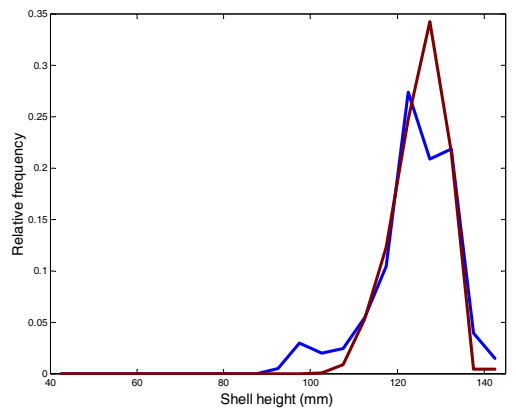
2000 to 2002



2002 to 2004

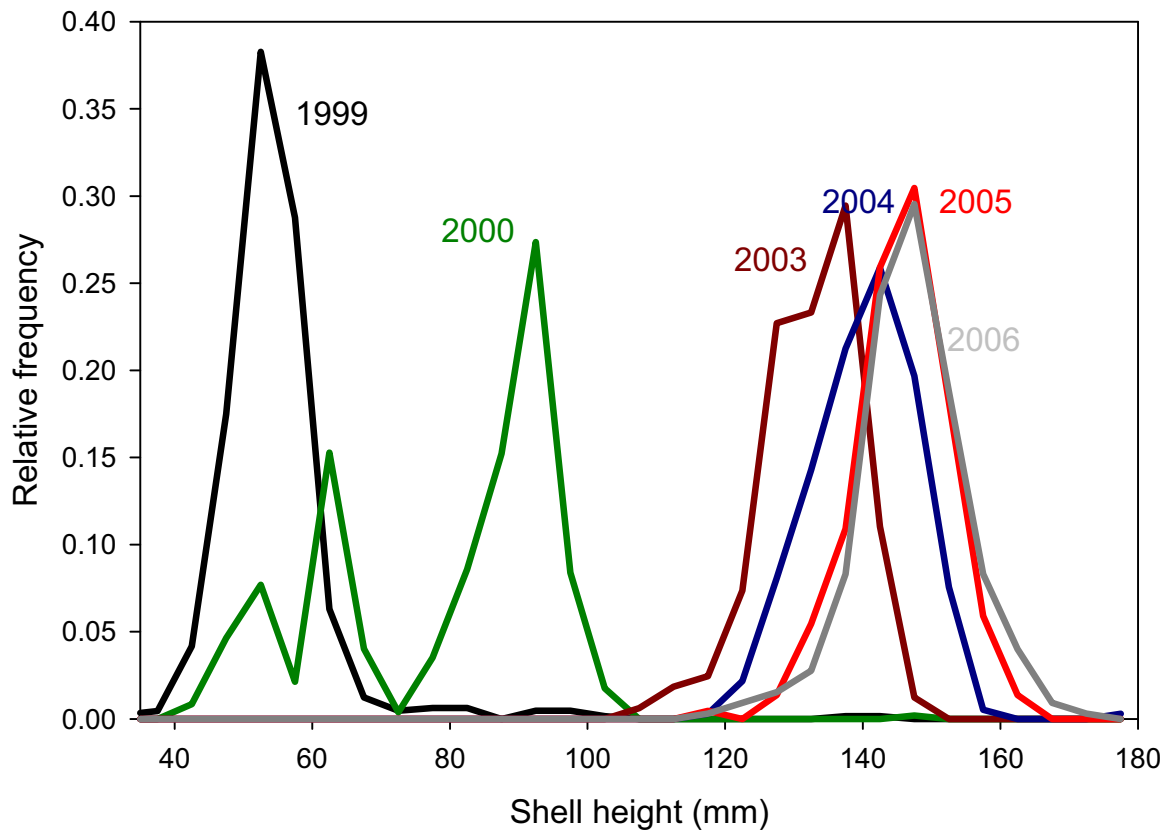


2004 to 2005

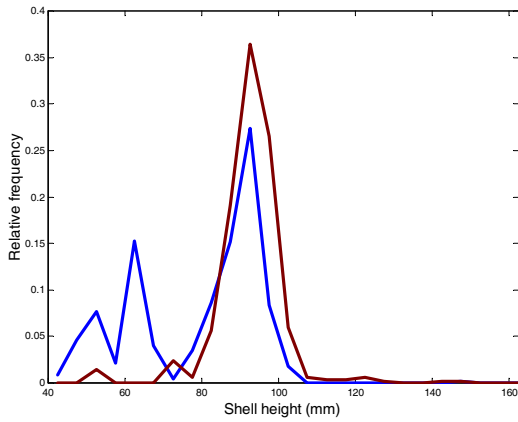


2005 to 2006

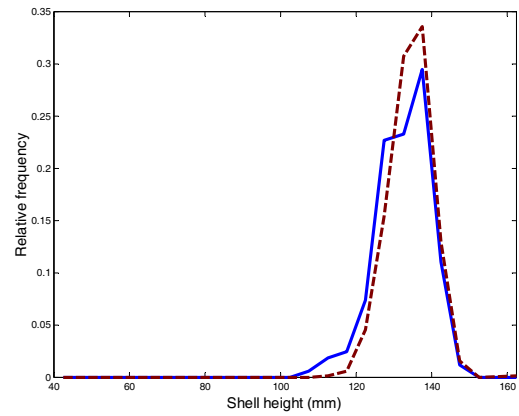
APPENDIX B2 Figure 5. Comparison between observed (solid blue line) and projected (dashed-dotted brown line) normalized size-frequencies at site #2 (Closed Area II).



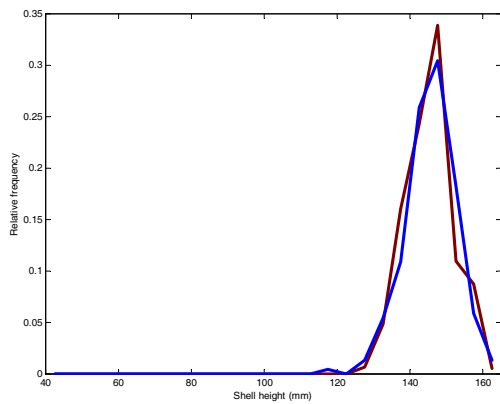
APPENDIX B2 Figure 6. Normalized size-frequencies at site #3 (Nantucket Lightship Closed Area).



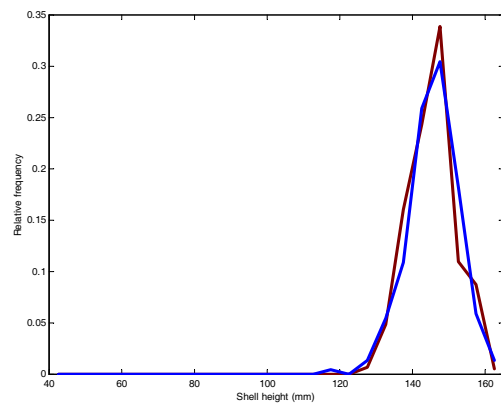
1999 to 2000



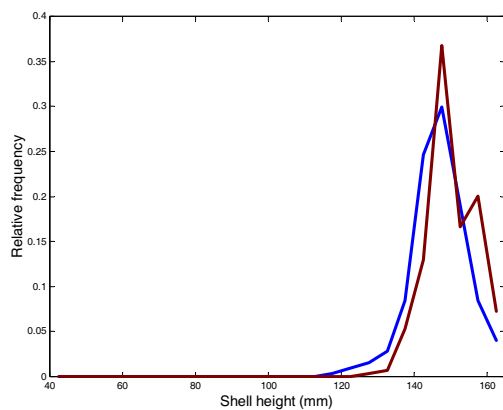
2000 to 2003



2003 to 2004

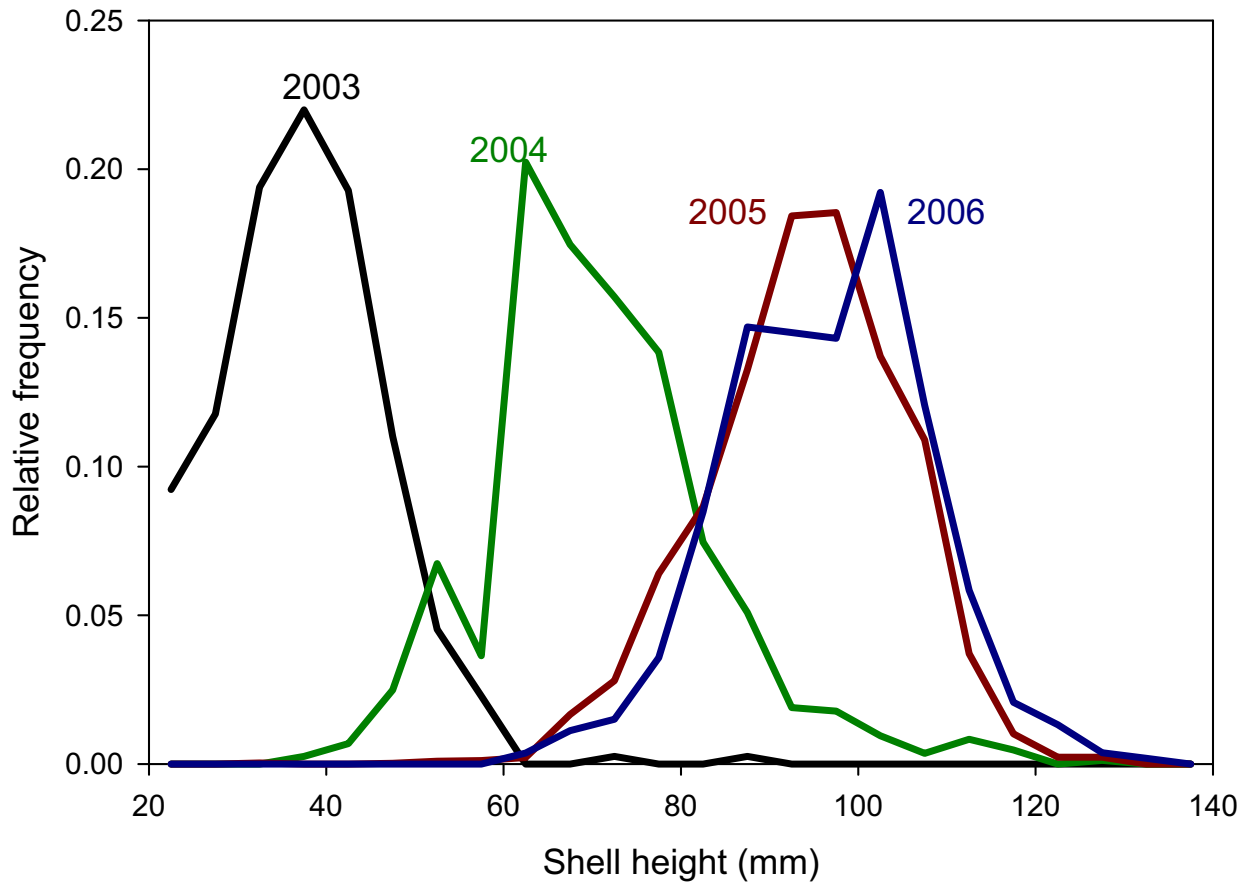


2004 to 2005

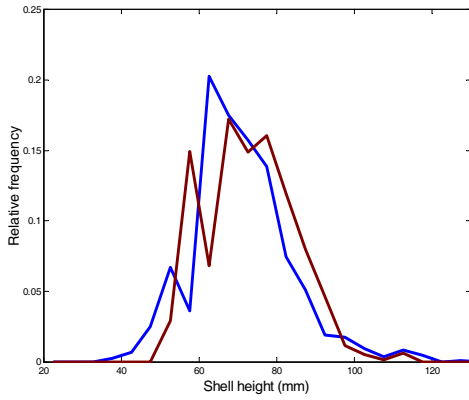


2005 to 2006

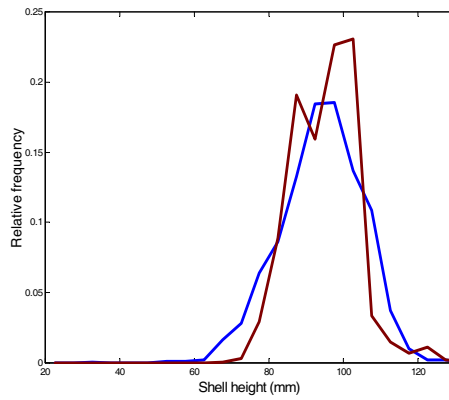
APPENDIX B2 Figure 7. Comparison between observed (solid blue line) and projected (dashed-dotted brown line) normalized size-frequencies at site #3 (Nantucket Lightship Closed Area).



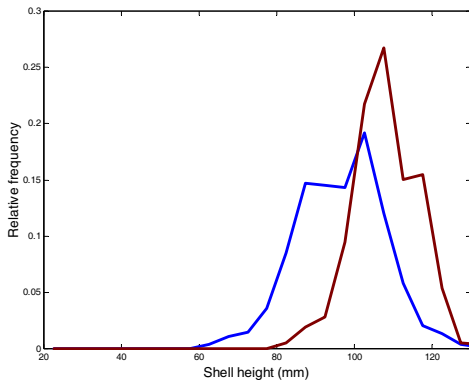
APPENDIX B2 Figure 8. Normalized size-frequencies at site #4 (Elephant Trunk Closed Area).



2003 to 2004

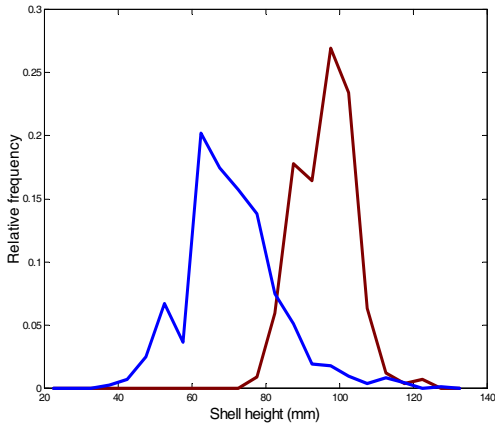


2004 to 2005

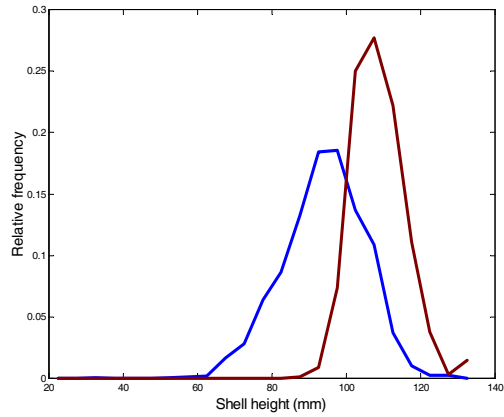


2005 to 2006

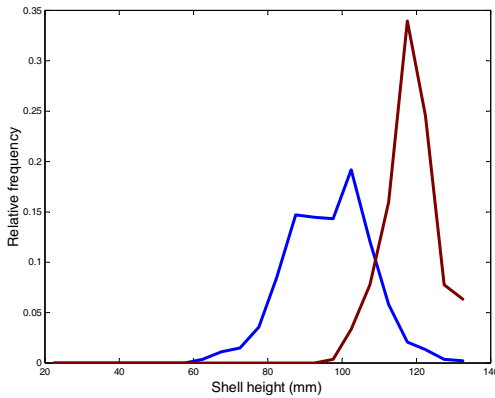
APPENDIX B2 Figure 9. Comparison between observed (solid blue line) and projected (dashed-dotted brown line) normalized size-frequencies at site #4 (Elephant Trunk Closed Area).



2003 to 2004



2004 to 2005



2005 to 2006

APPENDIX B2 Figure 10. Comparison between observed (solid blue line) and projected (dashed-dotted brown line) normalized size-frequencies at site #4, under the assumption that two shell growth lines are laid down annually, as suggested in Kranz et al. (1984).

(Appendix B3)

APPENDIX III: Methodology for estimation of growth from shell rings

Shells were collected on the 2001-2006 NEFSC surveys. At about half the valid tows in the survey, scallops were chosen randomly (averaging about 6 per station) to be used for the growth analysis. The scallops were scrubbed with a wire brush, shucked, and both valves were frozen and transported back to shore for later analysis. After the shells were thawed and cleaned, rings on the top valve of each shell that represented annuli were marked with a pencil. On some shells, one or more “shock marks” were evident. These were distinguishable from annuli by their irregular nature and because a point of injury was usually evident in the form of a crack or deformation of the shell. After the shells were marked, the distance in millimeters from the umbo to each of the ring marks was measured with calipers. Since the first ring is often very small and difficult to discern, the data only include the measurements to the second ring and above. Growth increments were calculated as the distance between the rings (in mm). The partial increment from the last ring to the edge of the shell was not used in the analysis.

Growth matrices were calculated by binning the growth ring shell heights into 5 mm classes (e.g., 40-44 mm shell height), and labeling the bins 1, 2, . . . , n , where the last bin represents a plus group. The ij th entry of the matrix represents the fraction of scallops that started in bin i that grew in a year’s time to bin j . Growth matrices based on the shell increments for Georges Bank and the Mid-Atlantic are given in Table App3-1.

To estimate growth parameters from the increment data, we used the growth increment form of the Von Bertalaffy equation:

$$\Delta L = (L_{\infty} - L)[1 - \exp(-K\Delta t)], \quad (1)$$

where L is the starting length, ΔL is the growth increment that occurred over time Δt , and L_{∞} and K are the two growth parameters to be estimated. Equation (1) predicts that a plot of the increments (ΔL) vs. starting length (L) will be a straight line with slope $m = -[1 - \exp(-K\Delta t)]$, x -intercept L_{∞} and y -intercept $b = -mL_{\infty}$. Thus, one could estimate K and L_{∞} from a plot of increment vs. starting length, with

$$K = -\frac{1}{\Delta t} \ln(1 + m) \quad (2)$$

and

$$L_\infty = -b/m \quad (3)$$

L_∞ and K may vary considerably among individuals in real populations. If this is the case, the naive estimation using equations (2) and (3) that ignores this variability can be seriously biased (Sainsbury 1980). We will derive here approximately unbiased estimates of L_∞ and K when these parameters have individual variability.

The growth increment of the i th individual, ΔL_i depends on a fixed effect (the starting length L) and random effects depending on the individual:

$$\Delta L_i = (m + m_i)L + (b + b_i) + \epsilon, \quad (4)$$

where m and b are the mean slope and intercept (averaging over all individuals), m_i and b_i are deviations from the mean slope and intercept for the i th individual, ϵ is a random independent error, and $E(\epsilon) = E(m_i) = E(b_i) = 0$. Note that the slope and intercept obtained from a simple linear regression of ΔL_i vs. L will not necessarily be the same as m and b .

The parameters associated with the i th individual can be calculated as:

$$K_i = -\ln(1 + m + m_i) \quad (5)$$

and

$$L_{\infty,i} = -(b + b_i)/(m + m_i). \quad (6)$$

We define $K = E(K_i)$, i.e., the mean of the individual K_i s in the population. We have

$$K = E(K_i) = E(-\ln(1+m+m_i)) \geq -\ln(E(1+m+m_i)) = -\ln(1+m). \quad (7)$$

Thus, estimating $E(K_i)$ as $-\ln(1+m)$ using the mean slope only will result in an estimate that is biased low.

Approximating $\ln(1 + m + m_i)$ by a second order Taylor polynomial,

$$\ln(1 + m + m_i) \simeq \ln(1 + m) + \frac{1}{1 + m}m_i - \frac{1}{2(1 + m)^2}m_i^2. \quad (8)$$

Taking expectations in the above equation gives:

$$K = -E(\ln(1 + m + m_i)) \simeq -\ln(1 + m) + \frac{\text{Var}(m_i)}{2(1 + m)^2} \quad (9)$$

An approximately unbiased estimate of $L_\infty = E(L_{\infty,i})$ can be computed similarly:

$$E(L_{\infty,i}) = -E\left(\frac{b + b_i}{m + m_i}\right) \simeq -\frac{b}{m} - \frac{1}{m^2}\left[\frac{b}{m}\text{Var}(m_i) - \text{Cov}(b_i, m_i)\right] \quad (10)$$

Approximate formulas for the standard errors of K and L_∞ , σ_K and σ_{L_∞} , are

$$\sigma_K \simeq \frac{\sigma_m}{(1 + m)} \quad (11)$$

and

$$\sigma_{L_\infty}^2 \simeq L_\infty^2 \left(\frac{\sigma_b^2}{b^2} + \frac{\sigma_m^2}{m^2} - \frac{2\sigma_b\sigma_m\rho(b, m)}{bm} \right) \quad (12)$$

where σ_b and σ_m are the standard errors of b and m respectively, and $\rho(b, m)$ is the correlation coefficient of b with m (see e.g., Rice 1987).

All analysis were conducted using the statistical program *R* (v2.3.1), using the mixed-effects (Pineiro and Bates 2000) package *lme4*. The above techniques require shells to have at least two increments, in order to estimate the two parameters $L_{\infty,i}$ and K_i . The increments included in the analysis included all shells collected at random stations with at least two growth increments.

Numerical simulations

As a verification technique of the above formulas, increments were simulated using the statistical program *R*, assuming L_∞ and K are gamma random variables, with means 140 and 0.5, respectively, and a specified CV. 1000 animals were simulated, with each contributing 4 increments. The simulated growth increments were subject to a 10% CV. Naive and mixed-effects estimates were made for various CVs (Fig App3-1). As expected, growth variability caused fairly considerable biases in the naive estimates, with K underestimated and L_∞ overestimated (Sainsbury 1980). Mixed-effect estimates were always within 3% of the true values.

Results

In Georges Bank, 15685 increments were measured from 3656 shells (Fig App3-2). In the Mid-Atlantic, 5706 increments were measured from 2098 shells (Fig App3-2). Parameter estimates, with standard errors, are given in

the table below, together with some previous estimates. Comparison of the new mean growth curve with previous curves indicate that the new Georges Bank parameters give similar growth to that of Serchuk et al. (1979) until about 120 mm, and then predict slower mean growth (Fig A3-4). The new Mid-Atlantic curve predicts somewhat faster growth for small scallops (< 80 mm), but slower growth at larger sizes, with a considerably smaller asymptotic size. Further analysis, demonstrating that growth depends on such factors as depth and closure status, will be detailed in a forthcoming publication (Hart and Chute in prep.).

Growth Parameter Estimates

Source	L_{∞}	sd	K	sd
Georges Bank				
New	146.5	0.3	0.375	0.002
Harris and Stokesbury (2006)	140.0	2.1	0.51	0.04
Harris and Stokesbury (2006)	148.6	4.0	0.36	0.04
Harris and Stokesbury (2006)	121.1	6.2	0.27	0.09
Thouzeau et al. (1991)	144.87		0.2814	
Serchuk et al. (1979)	152.46		0.3374	
Posgay (1979)	143.6		0.37	
Merrill et al. (1966)	143.3		0.2324	
Merrill et al. (1966)	145.1		0.2258	
Mid-Atlantic				
New	131.6	0.4	0.495	0.004
Serchuk et al. (1979)	151.84		0.2997	

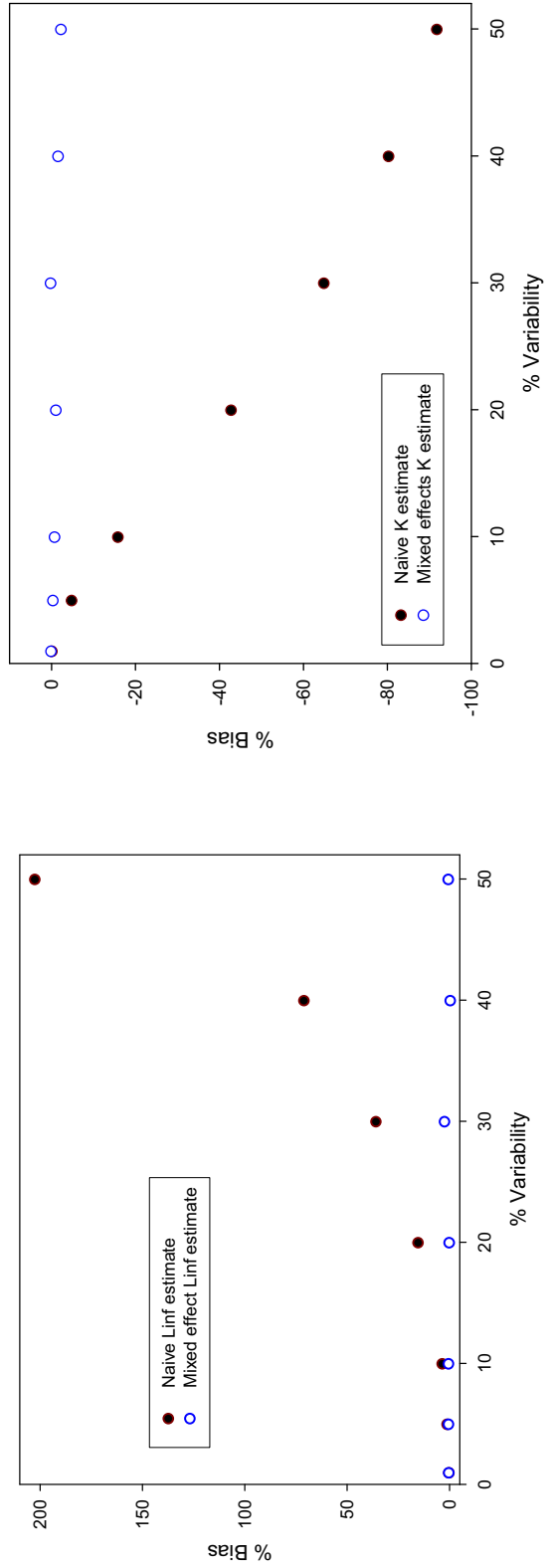


Fig App3-1. Percentage bias in simulated data for naive and mixed-effect estimates of growth parameters, where both growth parameters are assumed to vary.

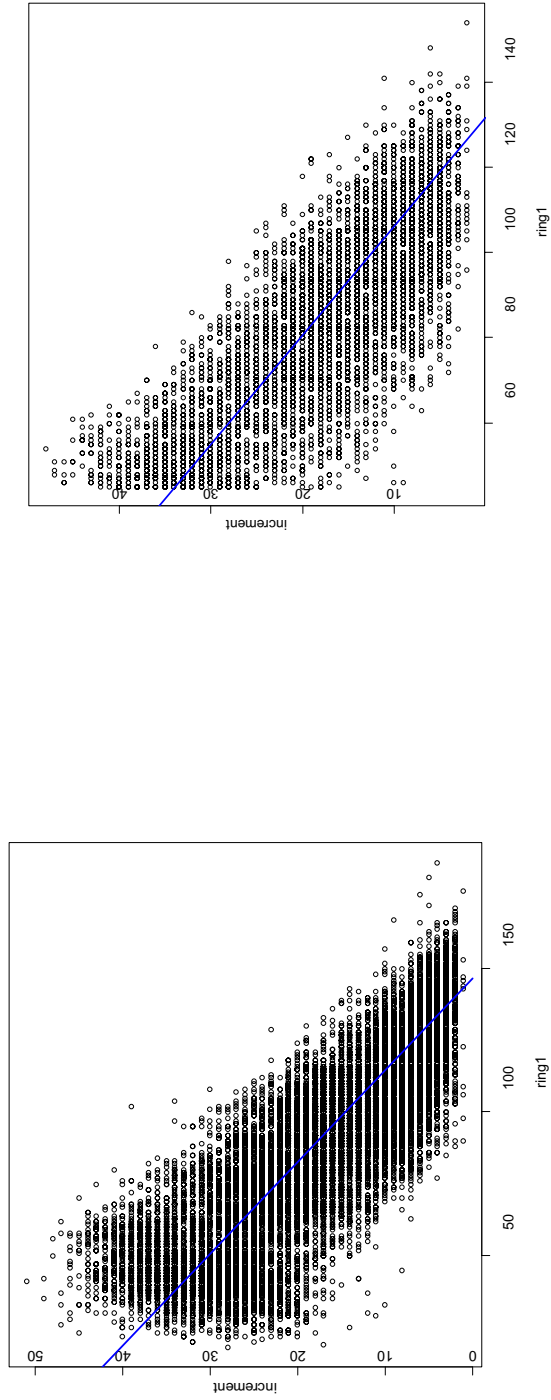


Fig App3-2. Georges Bank (left) and Mid-Atlantic (right) growth increments with mixed-effect regression lines.

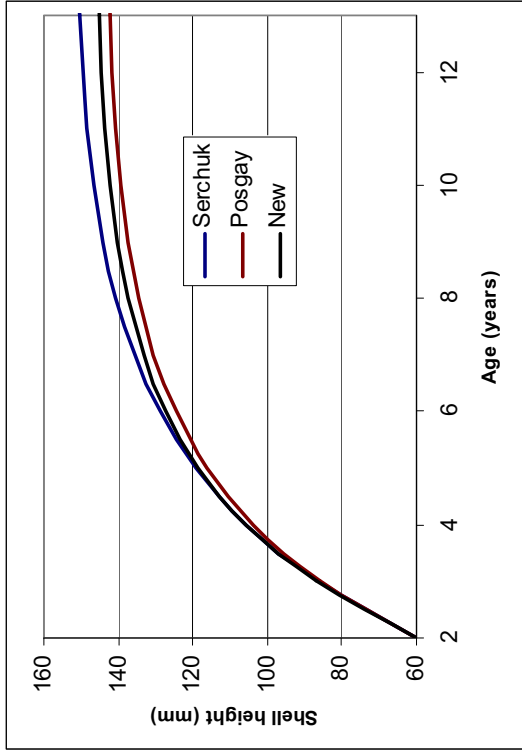
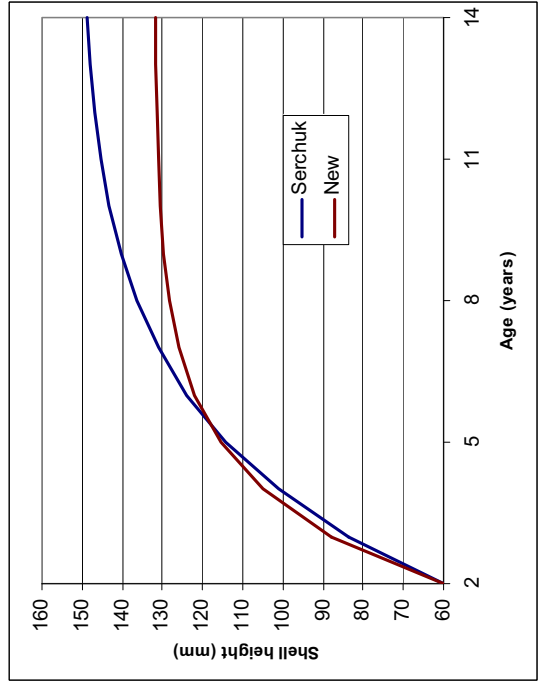


Fig App3-3. Comparison of the new growth curves (left is those for Georges Bank and right are the two for the Mid-Atlantic) with those of Serchuk et al. (1979) and Posgay (1979), for a scallop starting (at age 2) at 60 mm.

APPENDIX B4: Shell Height/Meat Weight Relationships

New shell height/meat weight data was collected on the annual NMFS sea scallop survey during 2001-2006. This appendix will present and analyze these data.

Methods

Sea scallops (averaging about 6 per station) were selected for analysis on roughly half of all stations (511 stations in the Mid-Atlantic, 592 stations on Georges Bank). The scallops were measured to the nearest millimeter, carefully shucked, excess water was removed from the meat, and the meat was weighed to the nearest gram. Data was also collected in 2003, but there was partial data loss when the data was transferred from ship to shore, so these data will not be used. In 2004-2006, whole and gonad weights were also recorded, but these data will not be presented here. The data here was separated into two regions (Mid-Atlantic and Georges Bank); further separation into subareas is possible, but will not be presented here.

Preliminary analysis indicated a residual pattern for those scallops with shell height less than 70 mm due to the small weights of these scallops (1-3 g) combined with the fact that meat weight could only be measured to the nearest gram. For this reason, the analysis was restricted to scallops that are at least 70 mm shell height. Scallops less than this height are below commercial size and thus their meat weight has no influence on CASA model calculations.

A generalized linear mixed-effects (GLMM) model was used to fit the equations

$$W = \exp(\alpha + \beta \ln(L)) \quad (\text{A4-1})$$

and

$$W = \exp(\alpha + \beta \ln(L) + \gamma \ln(D)), \quad (\text{A4-2})$$

where W is meat weight (grams), L is shell height (mm), and D is depth (meters), to the data. The GLMM used a gamma likelihood with a log link, appropriate for data (such as these) with “constant CV” error (McCullagh and Nelder 1989). This method avoids log-transforming the response variable (meat weight) that can lead to biased estimates when back-transformed. Because samples collected at the same station may be more similar than those from other stations, “station” was used as a random effect, and this random effect was weighted by the total number of scallops caught on that station so that stations at high abundances would be appropriately represented. The results were compared to those using a simple log-log regression and a GLM with just fixed effects. Both of these gave nearly identical results after applying a bias correction to the log-log regression, and differed only slightly from the GLMM presented here. All data analysis was conducted using the R statistical program (v2.3.1), with the lme4 mixed-effects package.

Results

Mid-Atlantic

A total of 2945 observations were sampled from 511 stations (Figure 1). Parameters (Table App4-1) were well estimated with no evidence of a residual pattern (Table 2, Figure 2).

Predictions from the new estimates are similar to most previous estimates, with the exception of Lai and Helser (2004) (Figure 3). Compared to the estimates used in previous assessments, the new estimates predict slightly heavier meats at small shell heights, but lighter meats at very large shell heights, but these differences are very small. The relationship that includes a depth effect indicates that sea scallops have considerably heavier meats at shallower depths (Figure 4).

Georges Bank

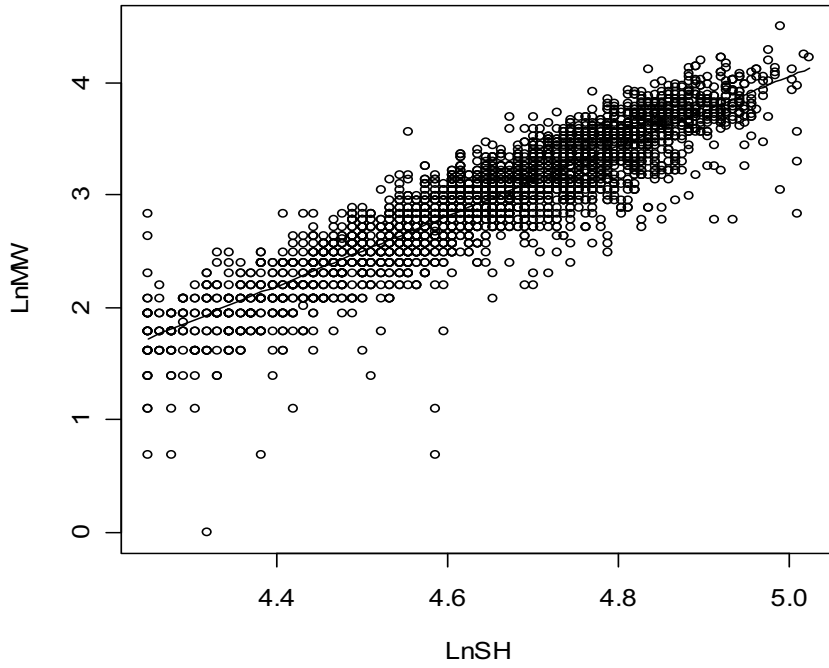
Based on 3824 scallops at 592 stations, model fits appeared good with little or no residual pattern (Figures 5-6). Parameters reasonably precise (Tables 1-2), and, as was the case for the Mid-Atlantic relationships, predict slightly greater meat weights at small shell heights, and slightly lower meat weights at large shell heights than does the relationship used in the previous two assessments (Figure 7). Predictions from the new relationship fall about in the middle of other estimates. Meat weights were substantially greater at shallower depths (Figure 8).

APPENDIX B4 Table 1. New shell height/meat weight parameters, with those from other studies for comparisons

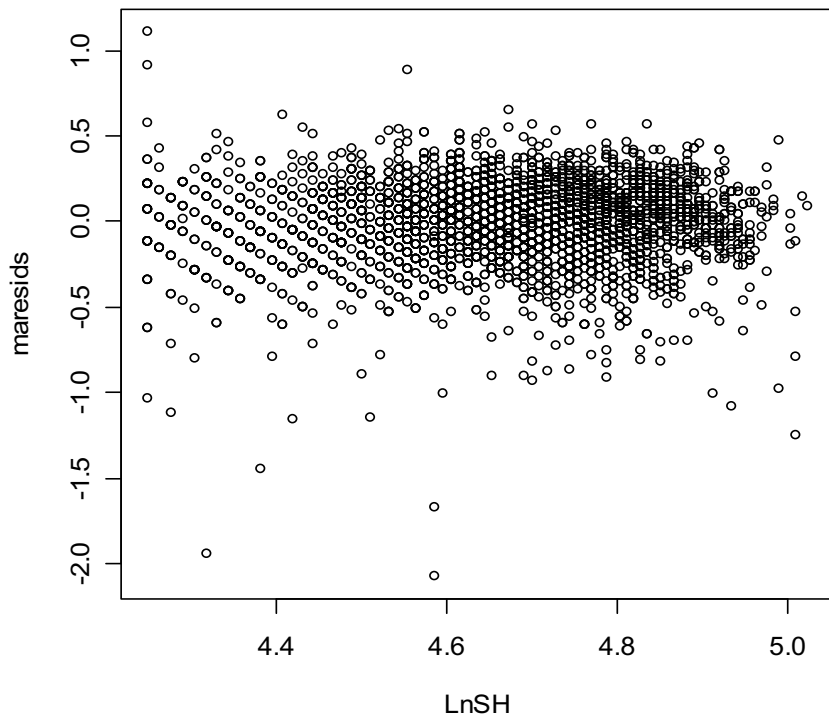
	α	β	γ
Mid-Atlantic Bight			
Haynes (1966)	-11.0851	3.0431	
Serchuk & Rak (1983)	-12.1628	3.2539	
NEFSC (2001)	-12.2484	3.2641	
Lai and Helser (2004)	-12.3405	3.2754	
New	-12.01	3.22	
New with depth effect	-9.18	3.18	-0.65
Georges Bank			
Haynes (1966)	-10.8421	2.9490	
Serchuk & Rak (1983)	-11.7656	3.1693	
NEFSC (2001)	-11.6038	3.1221	
Lai and Helser (2004)	-11.4403	3.0734	
New	-10.70	2.94	
New with depth effect	-8.62	2.95	-0.51

APPENDIX B4 Table 2. Standard errors for the new parameter estimates

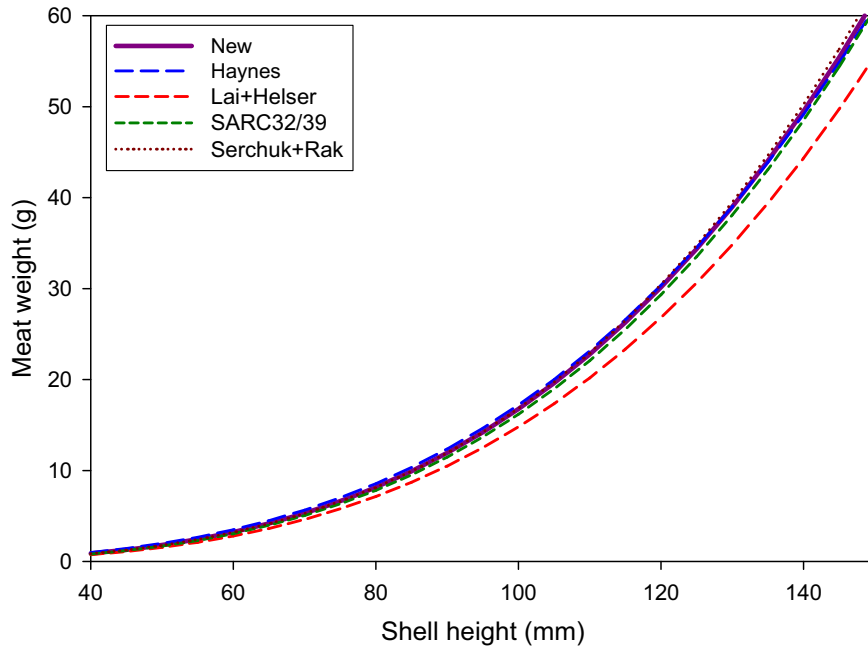
	α	β	γ
Mid-Atlantic Bight			
New	0.15	0.05	
New with depth effect	0.39	0.05	0.08
Georges Bank			
New	0.27	0.06	
New with depth effect	0.17	0.05	0.05



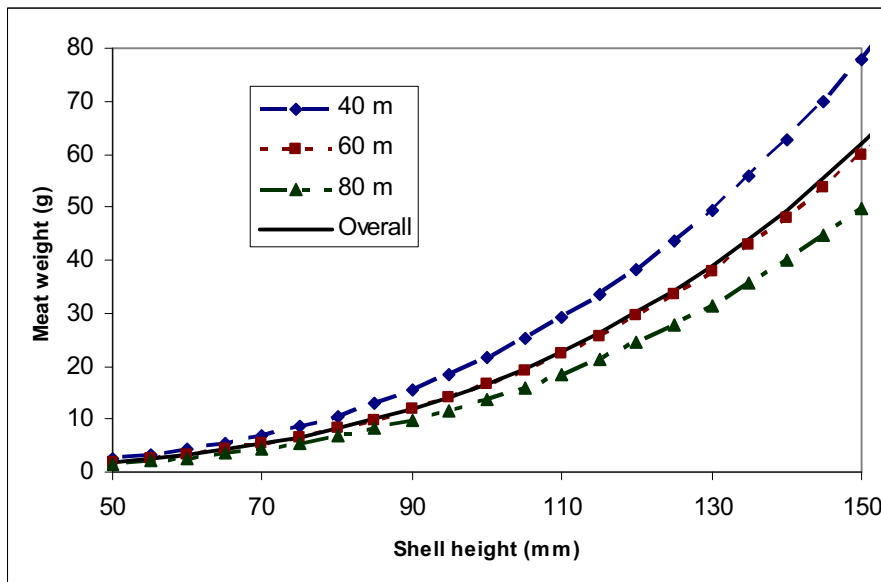
APPENDIX B4 Figure 1. Mid-Atlantic shell height/meat weight data



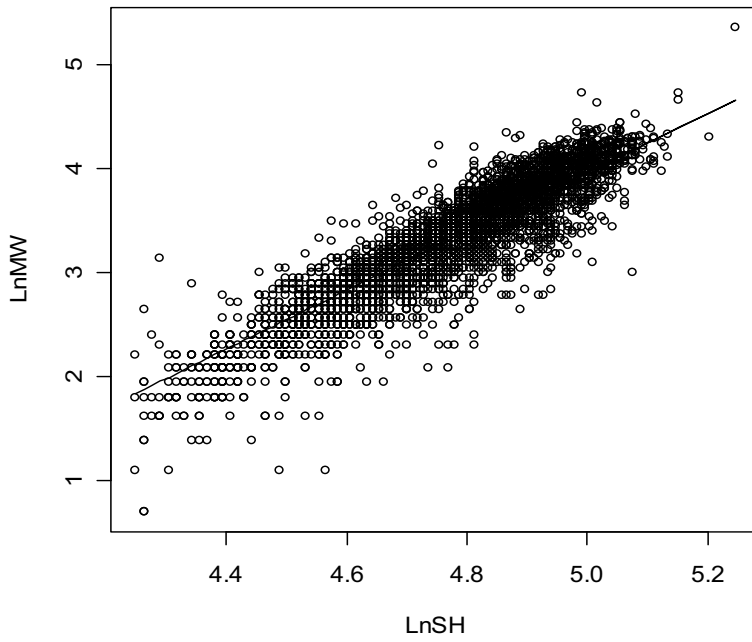
APPENDIX B4 Figure 2. Residual plot of Mid-Atlantic SH/MW data



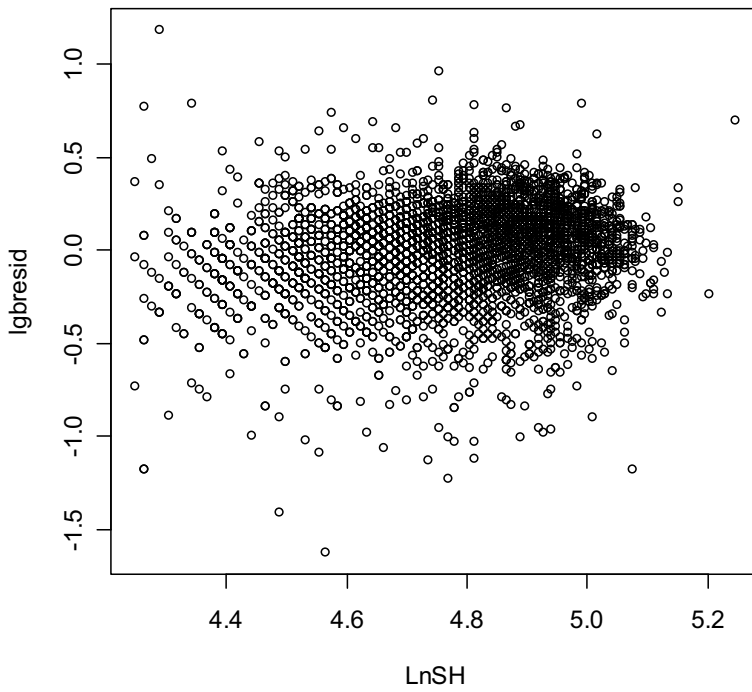
APPENDIX B4 Figure 3. Comparison of shell height/meat weight in the Mid-Atlantic



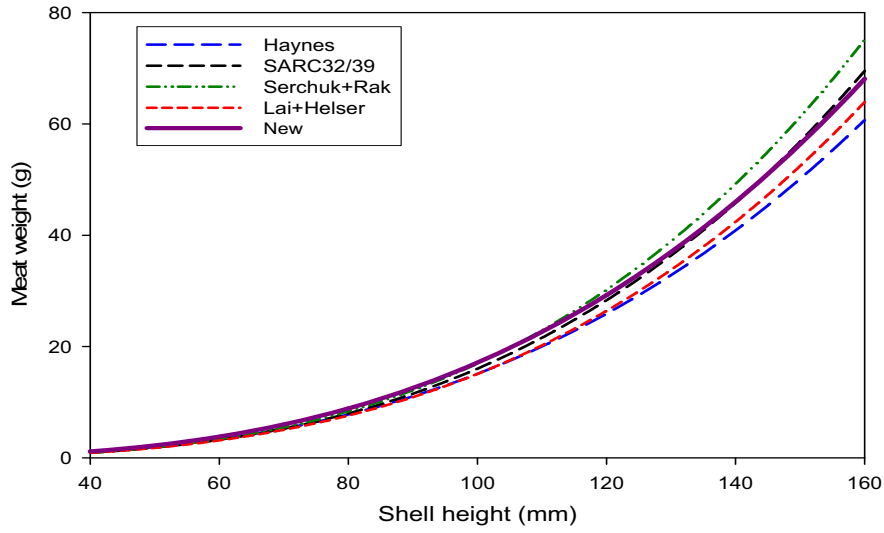
APPENDIX B4 Figure 4. Shell height/meat weight relationships at relationships 40, 60, and 80 m depth, and overall



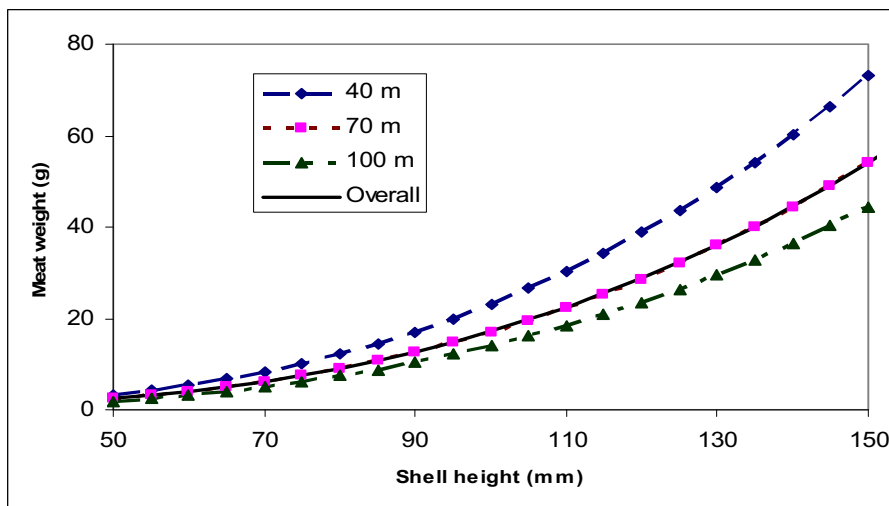
APPENDIX B4 Figure 5. Georges Bank shell height/meat weight data



APPENDIX B4 Figure 6. Residual plot of Georges Bank SH/MW data



APPENDIX B4 Figure 7. Comparison of SH/MW relationships in Georges Bank



APPENDIX B4 Figure 8. Georges Bank SH/MW relationships at 40, 70, 100 m depth and overall

APPENDIX B5: Selectivity of commercial sea scallop dredges with 4" rings

A size-selectivity curve was constructed to characterize the performance of the commercial New Bedford style sea scallop (*Placopecten magellanicus*) ("commercial") dredge, configured to meet the requirements of Amendment #10 to the Sea Scallop Fishery Management Plan. In order to construct an absolute size-selectivity curve, the commercial (experimental) gear must be compared to a non-selective (control) gear. The National Marine Fisheries Service (NMFS) survey dredge ("survey") served as the control gear in this study. The survey dredge is assumed to be non-selective because there is a liner sewn into the dredge bag which prohibits scallops from escaping. With the catch-at-length data from the two dredges, the Share Each Length's Catch Total (SELECT) model developed by Millar (1992) was used to generate the curve.

Data Collection and Analysis

The catch-at-length data needed to generate the selectivity curve was gathered during three cruises aboard commercial sea scallop vessels between 2005 and 2006. One cruise was completed in Georges Bank (in the Groundfish Closed Area II (CA2 2005)) and two in the mid-Atlantic (both in the Elephant Trunk Closed Area (ETCA 2005 and 2006)). Within each area, pre-determined stations, selected within a systematic random grid, were sampled. At each station, a standard NMFS survey dredge was towed simultaneously with a New Bedford style commercial sea scallop dredge. Simultaneously towing the two dredges from the same vessel allowed for similar type of substrate and population of scallops to be sampled. The survey dredge was 8-feet (2.4 m) in width, was configured with 2-inch (51 mm) rings, a 3.5-inch (89 mm) diamond mesh twine top, and a 1.5-inch (3.8 cm) diamond mesh liner and the commercial dredges were 15-feet (4.6 m) in width, had 4-inch (102mm) rings, a 10-inch (25.4 cm) mesh twine top and no liner. Rock chains and chafing gear were used on both dredges as dictated by the area surveyed and current regulations.

Each tow, from all cruises, was evaluated and deemed invalid if any of the following conditions were observed: hangs, flips, crossing or tangling of the gear, the tow was not deemed "good" in the comments section of the deck or bridge log, the inclinometer indicated that the gear was not fishing correctly, no scallops were caught or there were fewer than 20 scallops caught in either dredge. A catch of less than 20 suggests that there were actually no scallops present at the station; rather, scallops from a preceding tow may have been lodged in the dredge or left on deck.

The number of scallops caught per each 5 mm length class (evaluated as the mid-point of the length class, i.e., length "7.5 mm" represents the length class 5-10 mm) from each gear, was multiplied by an expansion factor equal to the number of baskets of scallops caught divided by the number of baskets measured. The tows were then combined by cruise, closed area, year and all tows together. For each tow and combination of tows, a plot was made of the ratio of the number of scallops in each length class in the commercial dredge to the total in both dredges (Commercial/Total) in order to determine if the commercial gear was behaving selectively. This assessment validated proceeding with the analysis.

The catch-at-length data for each tow combination were then analyzed with the Share Each Length's Catch Total (SELECT) model developed by Millar (1992). The SELECT model generates the parameters needed to create the selectivity curve as well as a parameter that denotes relative fishing intensity between the two gears (experimental and control). This is the

split parameter, p_j , which accounts for how catch among gears ($j=1, \dots, n$) will vary due to affects such as differential fishing effort, fish avoidance behavior and localized fish concentrations.

Due to variation in wind speed, water depth, sea state, scallop density and other factors that cannot be controlled, there is variation in selectivity from one tow to the next. This must be considered when tows are combined. A test for overdispersion (variation exceeding that which is predicted by the model) was completed using the replication estimate of between-haul variation (REP) combined hauls approach discussed in Millar et al. 2004. In order to avoid over-inflating the degrees of freedom for this analysis, only length classes where, when all tows are combined, one dredge has caught at least 20 scallops were used. In order to determine if this affected the estimated parameters, the model was run under this criterion as well as under the criteria that, for each length class, at least one dredge had more than: 1) zero scallops, 2) 60 scallops and 3) 1,000 scallops. In general, with fewer length classes used in the analysis, the 50% retention length, selection range, split parameter and log likelihood values all increased; however, these changes were not substantial.

In order to create a selectivity curve that is representative of the offshore commercial fleet, sampling was conducted aboard commercial scallop vessels, under conditions that mimicked commercial practices and the experiments were performed during different months and in different areas, which contained a variety of substrates. The only aspect of this study that is not representative of commercial practices is tow duration; however, an assessment of how the number of baskets of scallops and trash caught in the commercial dredge affects the parameters of the selectivity curve was made. This served as a proxy for how tow duration might affect the selection process. It must be noted, though, that tow duration does not predict the size of the catch. For this assessment, tows from all three cruises were grouped into five categories based on the number of baskets of scallops caught in the commercial dredge: 1) fewer than three, 2) three to six, 3) six to twelve, 4) twelve to twenty-four, and 5) more than twenty-four. These increments were chosen because there were a similar number of tows that fit into each group. A selectivity curve was generated for each category, using the same length classes that were used to evaluate all tows combined. A Spearman's rank correlation coefficient analysis was then completed on the resulting 50% Retention Length (l_{50}), Selection Range (SR) and split parameter (p_c) values. This procedure was repeated with increasing baskets of trash. Categories for this analysis were based on the number of baskets of trash in the commercial dredge: 1) less than 0.25, 2) 0.25 to one, 3) one, 4) one to two, and 5) more than two.

Results and Discussion

The catch-at-length data obtained during this study were evaluated with the SELECT model using the logistic as well as Richards, log-log and complementary-log-log curves in order to determine the most appropriate model for the data. The deviance residuals from the logistic fit showed no considerable trends and the curve adequately fit the data. The other three curves did not significantly improve the fit, based on AIC values, and, therefore, the results will only be presented for the logistic SELECT model. Also, the REP assessment for combining multiple tows indicated that there was extra variation for all tow combinations (by cruise, year, area and all combined) and, therefore, the standard errors for the estimated parameters were multiplied by the square root of REP.

The logistic parameters estimated for each combination of tows were inserted into the selectivity curve equation. The range of l_{50} values from the different combinations of data was

98.1-105.2 mm and of selection range values was 18.6-28.7. However, the final results are those that were estimated for all valid tows for the CA2 and ETCA cruises combined since an evaluation of the resulting parameters and confidence intervals from all combinations of data (by cruise, area and year) revealed little significant difference. Additionally, by including tows from multiple cruises on different vessels, during different times of the year and in different areas and substrates the selectivity curve becomes more representative of the commercial fleet. The resulting SR for this analysis is 23.6 mm, the l_{50} is 100.1 mm and the estimated split parameter is 0.77.

The next assessment evaluated how increasing number of baskets of trash and scallops caught in the commercial dredge might affect the estimated selectivity parameters. This served as an indication of whether the results were affected by the reduced tow duration used in this study. The Spearman's rank correlation coefficient significantly indicated that with increasing number of scallops the selection range and the split parameter values increase. While the results for the 50% retention length appear to show a similar trend, the results were not significant. In contrast, none of the evaluated parameters showed a significant relationship with increasing number of baskets of trash; however, the l_{50} values show a decreasing trend with increasing baskets of trash. It can be assumed that the selectivity curve generated in this study does represent commercial practices since there is not a significant difference in the l_{50} values with increasing baskets of scallops or trash. Additionally, during the survey cruises, the dredge bag ranged from being empty to completely full, which mirrors the range observed during commercial operations.

Lastly, the final results for this study were compared to those obtained from an additional cruise in the Nantucket Lightship Closed Area (NLCA). This cruise was conducted under the same conditions and during the same time period as the aforementioned cruises; however, the survey and commercial dredges used in the NLCA were not configured as they were in the other areas. For this reason, data from the NLCA were not combined with the other cruises. The estimated parameters for the NLCA cruise yielded a 50% retention length of 101.6 mm, a selection range of 17.63 mm and a split parameter value of 0.76. Standard errors for the estimated parameters were multiplied by the square root of REP because the data were overdispersed. Results from the NLCA are comparable to the results from the other cruises combined. An assessment of these parameters with confidence intervals reveals that there is no significant difference between the two 50% retention lengths and split parameters, but that there is between the selection ranges. Regardless, the similarity of the results for the NLCA cruise and for the other cruises combined indicates that the selection curve generated for this study is robust to changes in gear configuration. Additionally, the length frequency distribution in the NLCA is different from the other closed areas. This implies that the selection curve is also robust to differences in length frequency distribution.

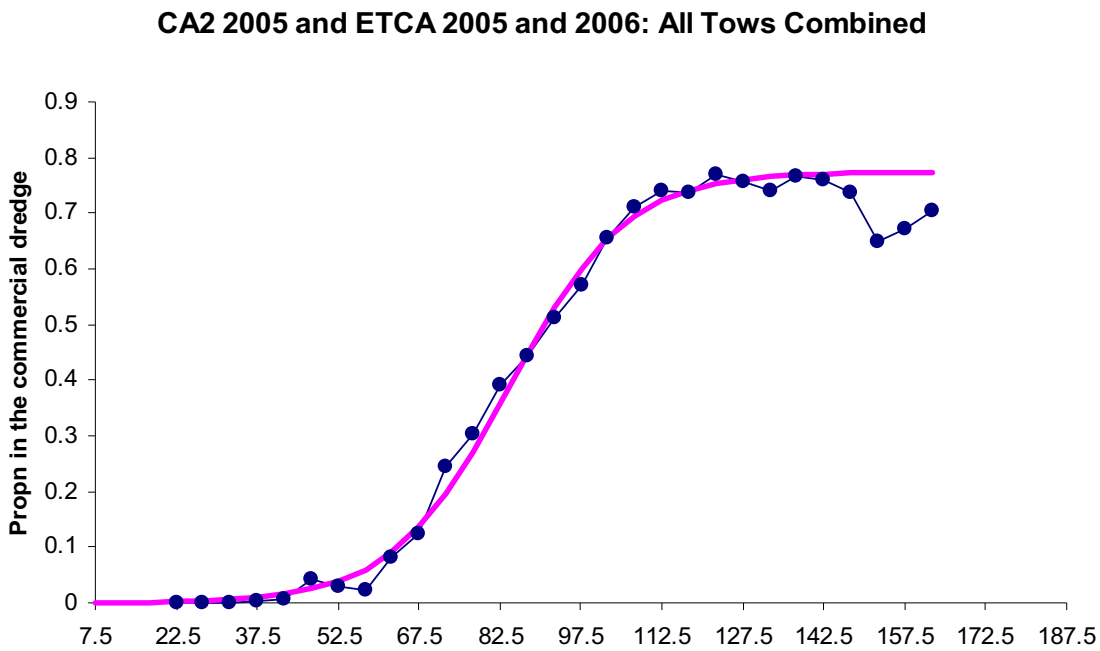
To maximize the effectiveness of the resulting curve from this study, more information is required regarding incidental mortality and the fate of scallops that interact with or escape from the commercial dredge and of the scallops that are landed on deck but are not harvested.

APPENDIX B5 Table 1. Estimated parameters from the logistic SELECT analyses on catch-at-length data for all length classes with at least 20 scallops in one of the dredges. Listed are lengths used in the analyses and the starting values to estimate the parameters in both R and Excel. The estimated values (left column) for logistic parameters a and b , as well as the 50% retention length (l_{50}), the selection range ($SR = l_{75} - l_{25}$) and the relative efficiency split parameter (p_c) are given. The number of tows (No. Tows) used for each analysis, log likelihood (L) and the replication estimate of between-haul variation (REP) are specified as well as the standard errors (right column), which have been multiplied by the square root of REP.

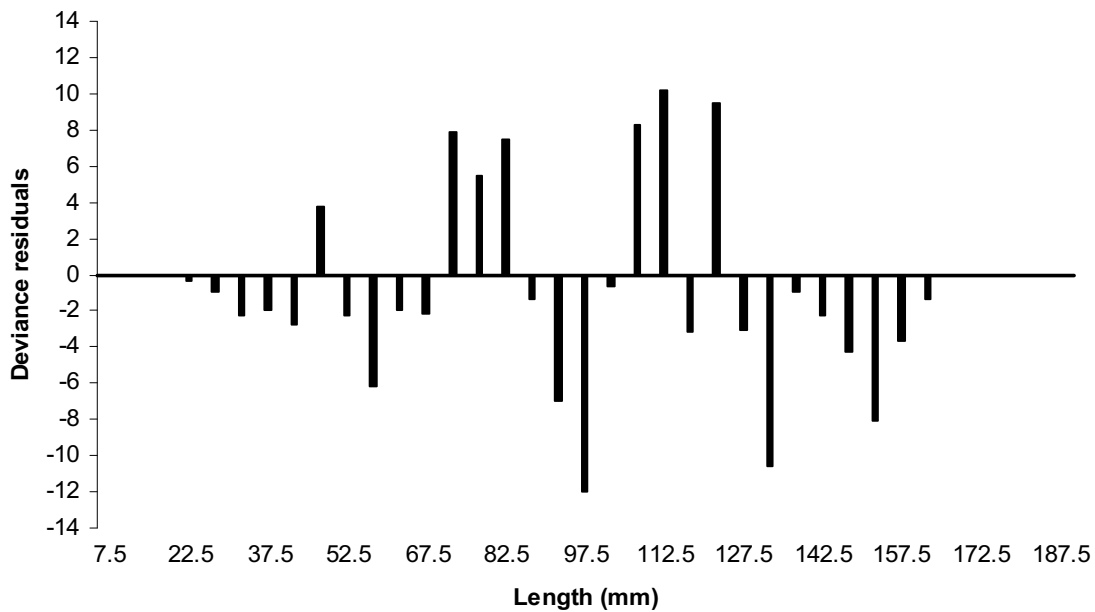
	NLCA 2005		CA2 2005, ETCA 2005 & 2006	
Lengths	42.5-172.5		22.5-162.5	
Start values	(-12, 0.12, 0.8)		(-12, 0.12, 0.8)	
a	-12.6700		-9.32	
b	0.12		0.09	
p_c	0.76	0.005	0.77	0.004
l_{50} (mm)	101.63	1.42	100.11	0.60
SR (mm)	17.63	1.85	23.61	0.59
L	-50672		-311035	
REP	8.01		7.98	
No. Tows	35		1052	

APPENDIX B5 Figure 1. (A) Logistic SELECT curve fitted to the proportion of the total catch in the commercial gear and (B) deviance residuals for CA2 2005, ETCA 2005 and ETCA 2006 cruises combined.

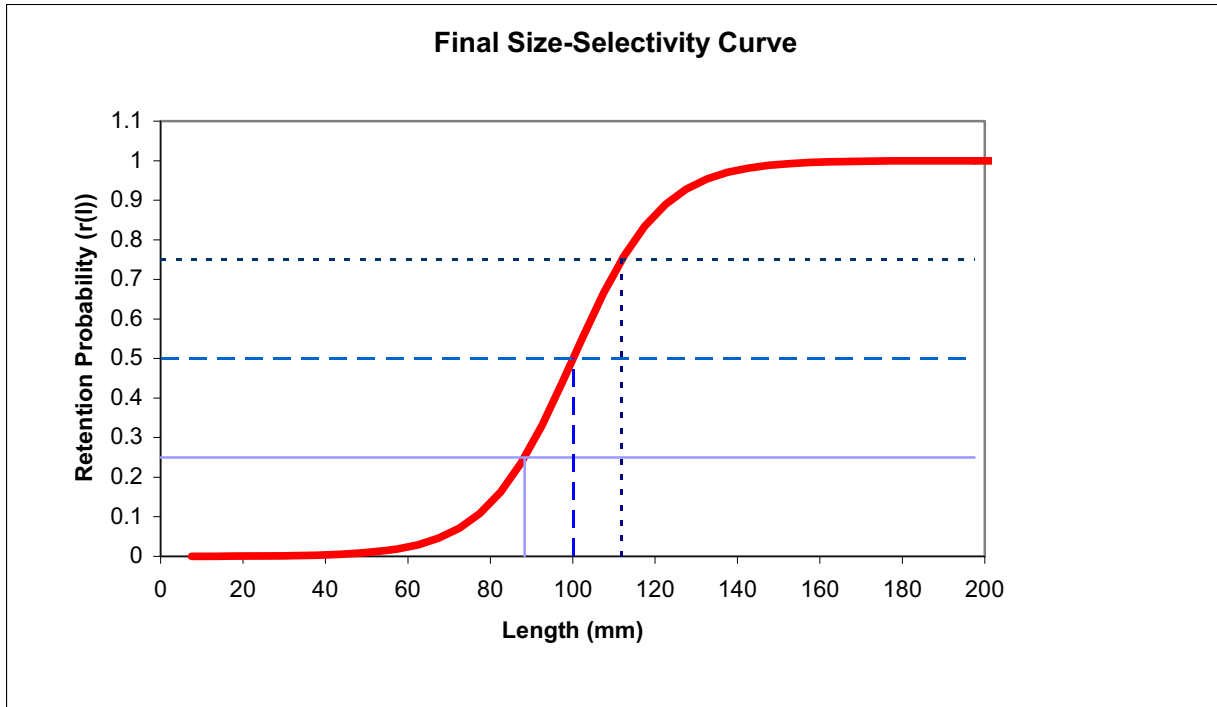
A



B



APPENDIX B5 Figure 2. Logistic selection curve for the New Bedford style dredge which incorporates all valid tows from the three cruises. The lengths at 25%, 50% and 75% probability of retention are shown. The selection range is the difference between the 75% and 25% retention lengths ($l_{75} - l_{25}$).



APPENDIX B6: Imputed NEFSC scallop survey data for unsampled strata

Some strata were unsampled during 1979-2006 NEFSC scallop surveys, particularly in the Georges Bank region (Tables 1 and 2). In NEFSC (2004), these “holes” in the survey data for a particular year (y) were filled automatically in database retrieval software by borrowing data from the same survey strata collected during the previous ($y-1$) and/or next ($y+1$) annual surveys. Borrowed data were used to compute means for survey holes and stratified random means for larger areas in the normal manner. Borrowing was one-sided in cases where data from $y-1$ or $y+1$ were lacking, and in the most recent survey year in particular where data for year $y+1$ are never available.

The borrowing procedure and variance calculations are ad-hoc but have a number of advantages: 1) survey indices for year y do not change after year $y+1$; 2) a minimum of programming and staff time is required; 3) the most relevant data are used, and 4) the calculations (linear interpolation between adjacent surveys) are simple, objective and make few assumptions about spatial patterns in population dynamics. No allowance is made for measurement errors in borrowed data. However, scallop survey data are relatively precise and important strata with high scallop abundance were generally not missed.

A more complicated statistical model based procedure was used in this assessment to fill all of the holes in NEFSC scallop survey data. However, data for Georges Bank during 1979-1981 were not used in the assessment, even after holes were filled, because the number of unsampled strata was relatively high (Figure 1).

The new statistical model was fit to tow-by-tow survey data (number of 40+ mm SH sea scallops per tow) by maximum likelihood using the `glm.nb()` function in `Splus` with a log link and assuming that measurement errors in the survey data were from a negative binomial distribution. Years and “newstrata” (see below) were categorical variables in the model and separate models were used for each subregion and post-stratification scheme. Residuals plots indicated that the model used to predict strata means fit the data reasonably well (Figure 1).

Subregions and newstrata are specific to the post-stratification scheme employed in a particular database run. Newstrata are original survey strata split into open and closed management areas. Subregions are contiguous groups of newstrata that define areas of particular interest. Data used in models that fill holes and in calculating abundance indices are from random stations within the original survey strata so that statistical assumptions are not violated in splitting strata into newstrata. Post-stratification exacerbates problems with holes because sections of a stratum assigned to newstrata might not have been sampled during a particular survey even if the larger stratum was sampled.

After fitting, the statistical model was used to calculate and store predicted values for every combination of subregion, year and newstrata. Predicted survey length composition for each subregion, year and newstrata was calculated by applying the shell height composition (total numbers in each 5 mm bin) from tows in the same subregion during the same year to predicted total numbers per tow from the model. Survey database software automatically retrieves predicted values for each shell height group to fill holes, as required. Predicted biomass per tow was calculated in the survey database software in the normal manner by applying a shell height/meat weight relationship.

The standard error for predicted number or biomass per tow is used in database variance calculations for larger subregions and regions. The standard error for predicted catch per tow in

a particular size bin was $\sqrt{p_L^2 s^2}$ where s is the standard error for predicted mean number per tow from the model (all sizes) and p is the observed proportion of mean numbers per tow for shell height bin L . Variances in the proportion p are not considered because the number of shell height measurements in a subregion is normally high.

The major benefit of the new modeling approach is that secondary holes in newstrata that occur after poststratification are automatically filled and that variance calculations have a better statistical basis. Differences in abundance and biomass indices between the complicated model based- and simple borrowing procedures were modest for Georges Bank as a whole and almost identical for the Mid-Atlantic Bight as a whole. Differences between model based and borrowing estimates were more substantial, however, for some subregions on Georges Bank.

APPENDIX Table B6-1. Numbers of random tows in NEFSC scallop surveys on Georges Bank by survey stratum and year (including tows by the F/V Tradition during 1999). Black areas indicate strata that were not sampled.

Year	Stratum									
	46	47	49	50	51	52	53	54	55	58
1979	4	9	5	4	7	3	3	5	7	2
1980		4	5	5	7	5		4	10	2
1981	5	9	5	5	8	5	5	6	9	2
1982	6	9	6	8	8	6	6	6	6	3
1983	6	9	6	12	11	6	6	6	6	4
1984	6	9	7	12	12	6	6	6	5	4
1985	6	10	9	11	12	7	7	7	7	4
1986	6		9	16	12	11	7	7	1	8
1987	6	12	9	16	11	11	7	7	9	8
1988	6	12	9	16	12	12	7	7	10	8
1989	6	12	8	15	12	12	7	6	10	8
1990	6	12	9	15	13	12	7	7	10	8
1991	6	12	9	16	12	12	7	7	10	8
1992	6	12	9	16	11	11	7	7	10	8
1993	6	12	9	13	9	10	7	7	10	8
1994	6	12	9	16	12	12	7	7	10	8
1995	6	12	9	16	11	12	7	7	10	8
1996	6	12	5	16	12	11	7	7	10	8
1997	6	13	7	16	12	14	9	10	10	8
1998	15	22	9	16	11	12	7	7	10	8
1999	6	15		5	6	14	11	15	14	8
2000	6	12	7	13	9	9	6	7	10	8
2001	6	14	9	15	14	14	15	11	12	6
2002	6	14	6	13	14	13	16	11	12	6
2003	6	13	9	14	10	14	15	13	10	6
2004	4	18	9	12	12	11	15	20	10	4
2005	5	20	10	11	12	12	12	19	10	4
2006	4	18	7	14	10	16	13	17	14	4

APPENDIX Table B6-1 continued

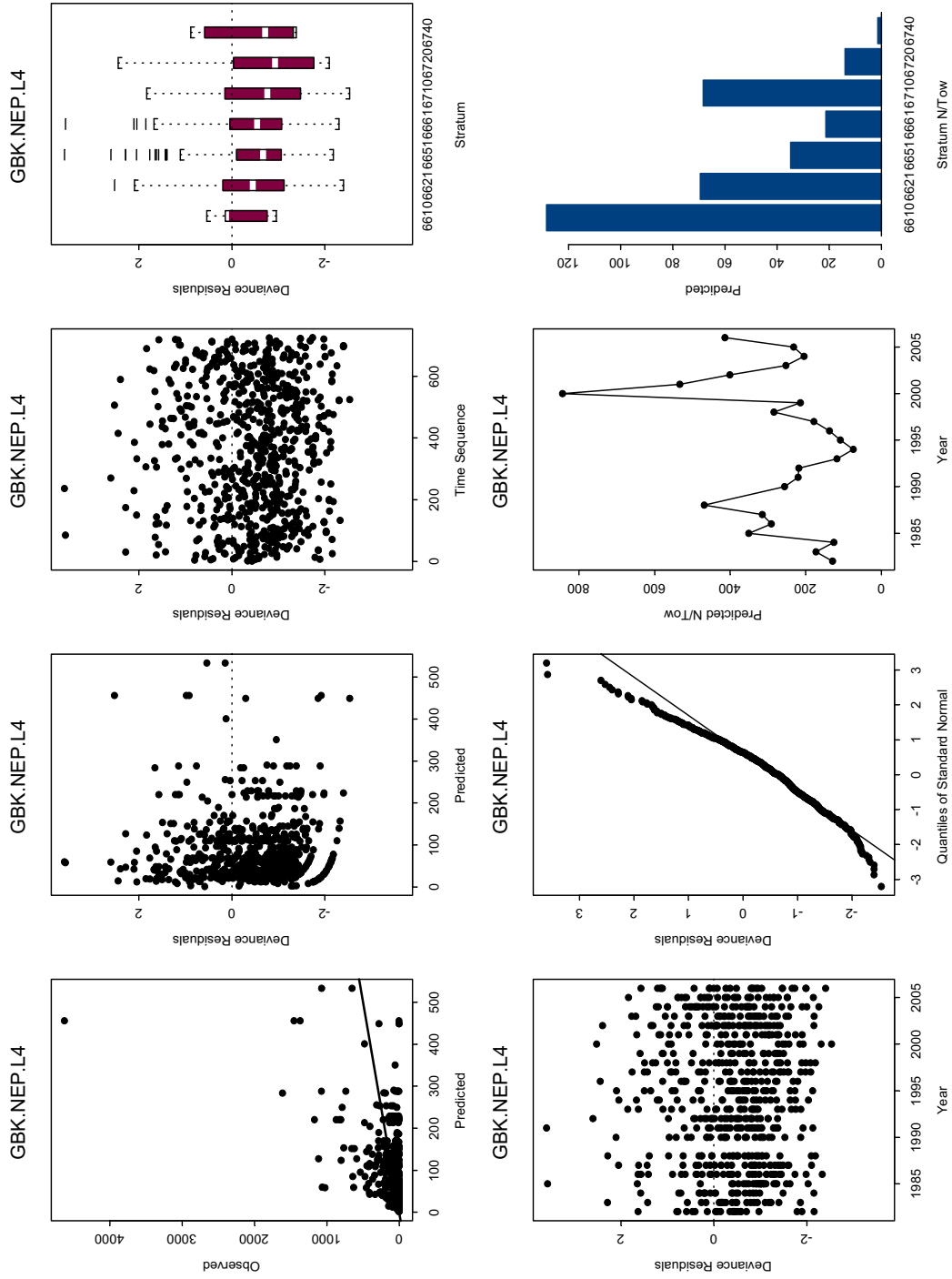
Year	Stratum								
	59	60	61	62	63	65	66	71	72
1979	10	8							
1980	10	8							
1981	9	8							
1982	10	9	7	9	4	6	6	4	5
1983	8	8	7	8	5	9	8	4	4
1984	8	8	7	8	3	9	8	5	4
1985	12	12	8	12	7	10	10	6	6
1986	12	12	8	13	7	12	12	6	6
1987	12	12	8	12	7	12	12	5	6
1988	12	12	8	12	6	11	12	6	6
1989	12	12							
1990	12	12	8	12	7	12	12	6	
1991	12	12	8	12	7	12	12	6	6
1992	12	12	8	12	7	11	12	6	6
1993	12	12	8	12	7	10	10	6	6
1994	12	12	8	12	7	12	12	6	6
1995	12	12	8	12	7	12	12	6	6
1996	12	12	8	12	7	12	12	6	6
1997	12	12	8	12	7	15	14	8	5
1998	11	11	8	12	7	12	10	6	6
1999	12	12	8	14	6	11	11	4	2
2000	12	12	7	12	7	11	12	6	
2001	10	12	18	23	6	10	11	5	
2002	10	10	18	24	4	12	14	8	5
2003	8	9	16	21	4	12	12	8	
2004	7	6	24	24	3	12	10	12	3
2005	8	7	22	24	3	11	9	12	
2006	6	7	24	17	3	12	9	19	

APPENDIX Table B6-2. Numbers of random tows in NEFSC scallop surveys in the Mid-Atlantic Bight by survey stratum and year (including tows by the F/V Tradition during 1999). Black areas indicate strata that were not sampled.

Year	Stratum										
	6	7	10	11	14	15	18	19	22	23	24
1979	2	1	5	7	7	12	7	5	12	20	3
1980	1	2	5	7	7	12	7	5	12	20	3
1981	2	1	5	6	7	12	7	5	12	20	3
1982	4	3	8	6	6	12	7	5	12	20	6
1983	4	4	8	8	8	12	6	7	8	16	6
1984	5	4	8	8	10	12	6	8	8	16	6
1985	5	5	8	8	10	12	8	8	8	16	6
1986	5	5	8	8	12	12	10	13	8	16	6
1987	5	5	8	8	12	11	10	12	8	16	4
1988	6	4	8	8	12	12	10	12	8	16	6
1989	5	5	8	8	12	12	10	12	8	16	6
1990	3	3	8	8	12	12	10	12	8	16	5
1991	5	5	8	8	12	12	10	11	8	16	6
1992	5	5	8	8	12	12	10	12	8	16	6
1993	5	5	8	8	12	12	8	10	8	16	6
1994	5	5	8	8	12	12	10	12	8	16	5
1995	5	5	8	8	12	12	10	12	8	16	6
1996	5	5	8	8	12	12	8	10	8	16	6
1997	5	5	8	8	11	12	9	12	8	16	6
1998	5	5	8	8	12	12	10	12	8	16	6
1999	5	5	8	8	12	12	10	12	8	16	6
2000	5	5	8	8	12	12	10	13	8	16	6
2001	5	5	9	14	10	12	8	12	10	22	8
2002	5	5	9	12	10	12	8	11	12	22	8
2003	5	5	8	12	10	12	10	12	10	20	6
2004	3	2	8	12	14	16	24	21	14	25	10
2005	2	3	7	10	15	16	26	22	14	26	8
2006	3	2	6	10	14	20	20	25	14	25	5

APPENDIX Table B6-2 continued.

Year	Stratum									
	25	26	27	28	29	30	31	33	34	35
1979	4	8	12	2	8	14	24	2	4	7
1980	4	9	11	2	8	14	24	4	4	6
1981	5	8	12	2	8	14	24	4	4	6
1982	7	9	12	3	8	14	24	7	7	5
1983	6	13	10	7	6	15	24	10	10	5
1984	7	14	10	6	8	15	24	10	14	5
1985	4	14	12	6	6	15	24	10	10	6
1986	4	14	20	10	6	15	24	7	13	10
1987	4	14	20	10	6	15	24	10	14	10
1988	4	14	19	10	6	15	23	10	14	10
1989	4	14	20	10	6	15	24	10	29	10
1990	3	12	17	10	5	14	24	10	14	10
1991	5	14	20	10	6	15	24	10	14	10
1992	4	14	20	10	6	15	24	10	14	10
1993	4	14	20	10	6	15	22	7	10	8
1994	4	14	20	10	6	15	23	10	14	10
1995	4	12	20	10	6	15	24	10	14	10
1996	4	13	19	10	6	15	20	8	10	8
1997	4	14	20	10	6	14	24	10	13	10
1998	4	14	19	9	6	14	23	6	14	10
1999	4	14	20	10	6	15	24	7	14	10
2000	4	13	20	10	6	15	24	10	14	10
2001	8	14	20	8	6	12	18	8	10	8
2002	6	10	19	7	6	10	16	6	6	6
2003	6	10	20	8	4	9	16	6	6	6
2004	5	8	20	8	4	6	20	5	5	18
2005	5	7	21	7	4	6	21	5	6	10
2006	6	7	16	5	5	9	20	5	5	8



APPENDIX Figure B6-1. Model residuals and diagnostics for a negative binomial model used to impute NEFSC scallop survey mean numbers per tow (40+ mm SH) based on tow by tow survey data from the Northeast Peak subregion of Georges Bank. Goodness of fit results for the Northeast Peak were typical although spatial and temporal patterns in predicted abundance varied among subregions.

APPENDIX B7: Selectivity and efficiency of large camera video data from the SMAST video survey during 2003-2006⁵

Selectivity curves were estimated for sea scallops in the SMAST video (“large” camera) survey using the Millar’s maximum likelihood SELECT model (Millar and Fryer, 1999) and “small” camera video data as a standard measure of sea scallop length composition and density at study sites. The small camera is believed to be fully efficient (100% detection probability) for sea scallops about 35+ mm SL. The data were ideal because large and small camera data were collected at each station so that stations can be analyzed as replicate “paired tow” experiments. Estimates for Georges Bank and Mid-Atlantic Bight combined during 2003-2006 indicate that the large camera system has an increasing logistic selectivity pattern for sea scallops with selectivity $\geq 50\%$ at 48+ mm, $\geq 90\%$ at 71+ mm SL, and $\geq 95\%$ at 79+ mm (approximate SE 1.7 mm for all estimates). The selectivity range for the large camera (L_{75} - L_{25}) was 22 mm (SE 2.4 mm). The SELECT model was configured so that the estimated split parameter p measured the ratio of total catches of sea scallops large enough to be fully selected by both cameras. Estimates of the split parameter p averaged 0.84 (SE 0.003 mm), which is approximately the same as the ratio expected based on assumed sample areas (A) for the two cameras, i.e. expected $p = A_{large} / (A_{small} + A_{large}) = 3.235 / (3.235 + 0.788) = 0.80$. This suggests that the large camera also has 100% detection probability for large fully selected scallops in its sample area.

Introduction / Methods

The primary purpose of the SMAST video survey camera selectivity comparisons was to identify the shell height at which the large camera was fully selective, assuming that the small camera was 100% selective at 35+mm shell height. SMAST camera survey selectivity curves were estimated by comparing large camera to small camera data from Georges Bank and the Mid-Atlantic Bight area combined during 2003-2006. Only stations where data was available for both cameras were included; any stations that were missing data from more than 2 quadrats were excluded. The number of stations varied each year with survey coverage (Figure 1).

Because the large and small cameras simultaneously collect data from the same locations, they can be directly compared for selectivity estimates. The large camera effective field of view is 3.235 m² at each quadrat and the small camera effective field of view is 0.788 m² (Stokesbury et al., 2004). The large camera’s view field allows for a larger number of scallops to be identified and measured, whereas the small camera with higher resolution allows for detection of smaller scallops (Figure 2).

Selectivity comparisons were based on shell height measurements from the large and small cameras by year and area (Table 1). Shell height measurements were binned in 10 mm increments to minimize potential effects of imprecise shell height measurements. Increment mid-points were used in all calculations (e.g. 5 mm for the 1-9.99 mm bin). Millar’s SELECT model (EXCEL Solver Version⁶) was used to fit an increasing logistic shape curve of selectivity for the large camera using the small camera as a standard. The model is:

⁵ Michael C. Marino III, Catherine O’Keefe (School of Marine Science and Technology (SMAST, University of Massachusetts Dartmouth, 706 South Rodney French Boulevard, New Bedford, MA 02744-1221), and Larry D. Jacobson (Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA, 02543)

⁶ <http://www.stat.auckland.ac.nz/~millar/selectware/code.html>

$$s_L = \left[\frac{e^{a+bL}}{1 + e^{a+bL}} \right]$$

where s_L is selectivity at length and a and b are parameters (Millar and Fryer, 1999). A third “split” parameter p represents relative sampling intensity between the two gears and was initially estimated by taking the average of the ratio of the sample in the large camera to the total sample (large / large + small) at each shell height bin. The model was used to estimate the shell heights with selectivity values of 50% (L_{50}), 90% (L_{90}) and 95% (L_{95}) as well as the selectivity range (SR = L_{75} – L_{25}).

Results / Discussion

The estimated selectivity curves for all years in both Georges Bank and the Mid-Atlantic showed a similar pattern of low selectivity at small sizes, increasing between approximately 35mm to 80 mm and reaching an asymptote of 1.0 around 85 mm (Figures 3-6). Parameter estimates (Table 2) were generally similar although L_{50} and related statistics were relatively high and imprecise for 2004. Simple averages were used to calculate “best” overall selectivity parameters for sea scallops in the large camera (Table 3). Similar results were obtained when means were computed using inverse variance weights.

Deviance residuals indicate generally good model fit (Figure 7). There were some runs of positive and negative residuals in 2003 and 2004. In 2005 and 2006 the model seemingly overestimated selectivity for the very large scallop size bins but this is most likely a result of low sample sizes for large scallops due to their low abundance.

Appendix B7 Table 1. Numbers of sea scallops measured and counted used in this analysis from video surveys during 2003-2006 in the Mid-Atlantic Bight and Georges Bank.

	LARGE			SMALL		
	MA+GB	MA all	GB all	MA+GB	MA all	GB all
	2003			2003		
Measured	4001	3018	993	1322	1041	281
Total Counted	6860	5043	1817	2014	1554	460
	2004			2004		
Measured	2216	1363	853	528	330	198
Total Counted	3902	2430	1472	917	564	353
	2005			2005		
Measured	1866	1196	670	430	276	154
Total Counted	3696	2333	1363	839	555	284
	2006			2006		
Measured	2265	1528	737	535	344	191
Total Counted	3549	2218	1331	940	536	404

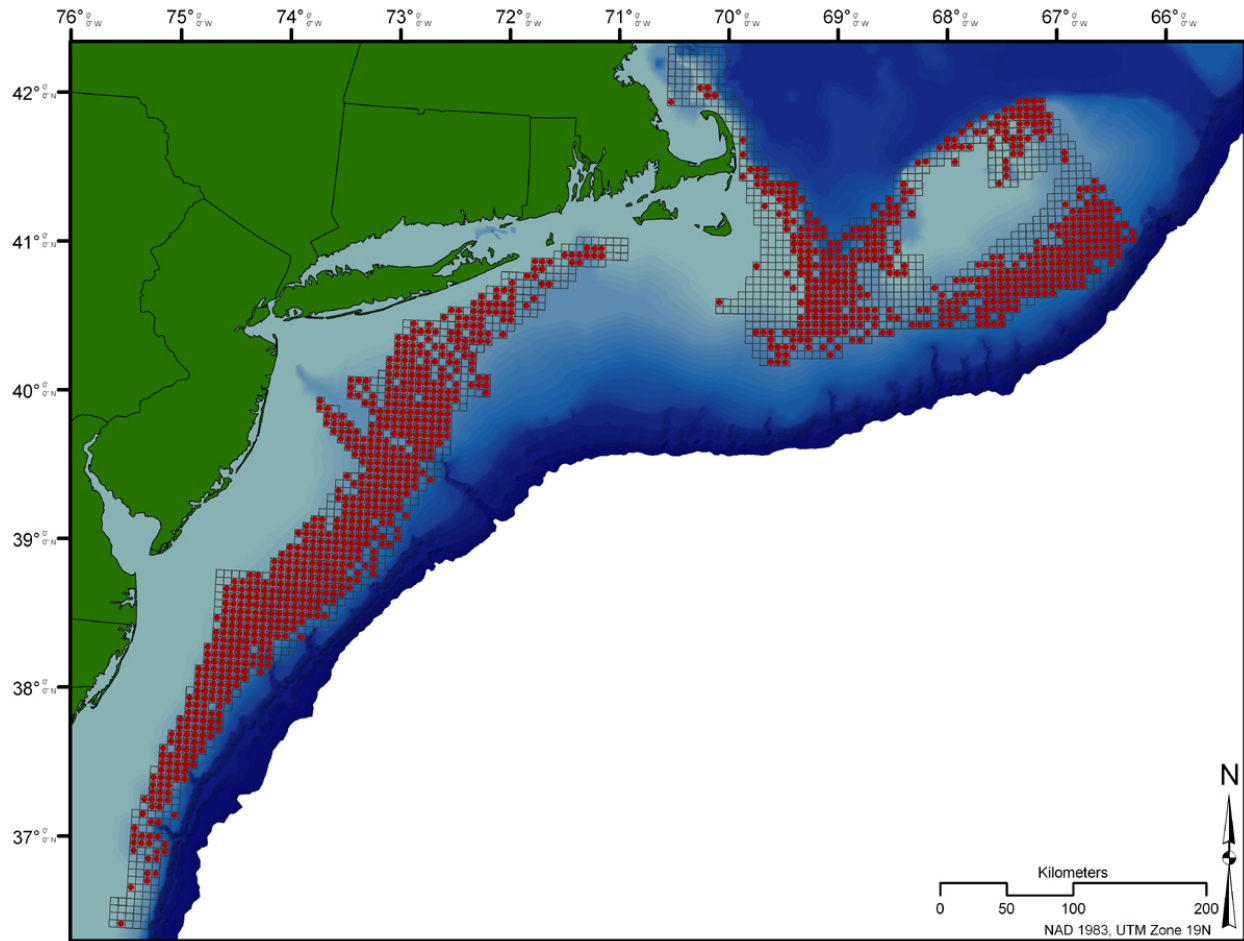
Appendix B7 Table 2. Estimated selectivity parameters p , a , b , L_{95} , L_{90} , L_{50} and SR with standard errors and variances from SELECT models fit to large and small camera video data collected during 2003-2006 on Georges Bank and in the Mid-Atlantic.

Year	2003	2004	2005	2006
Split (%)	88.5	83.8	82.5	81.8
SE(Split)	0.005	0.012	0.008	0.008
Var(Split)	2.75E-05	1.44E-04	6.40E-05	6.40E-05
weights	0.364	0.159	0.238	0.238
L95(mm)	85.71	103.07	63.99	64.96
SE(L90)	1.720	5.070	3.080	2.780
Var(L90)	2.959	25.705	9.486	7.728
weights	0.397	0.135	0.222	0.246
L90(mm)	77.62	90.62	57.43	59.98
SE(L90)	1.720	5.070	3.080	2.780
Var(L90)	2.959	25.705	9.486	7.728
weights	0.397	0.135	0.222	0.246
L50(mm)	54	54	38	45
SE(L50)	1.720	5.070	3.080	2.780
Var(L50)	2.959	25.705	9.486	7.728
weights	0.397	0.135	0.222	0.246
SR(mm)	24	36	19	15
SE(SR)	2.709	9.430	7.980	4.400
Var(SR)	7.341	88.925	63.680	19.360
weights	0.446	0.128	0.151	0.275
a	-4.98	-3.24	-4.35	-6.8
SE(a)	0.470	0.730	1.740	1.880
Var(a)	0.221	0.533	3.028	3.534
weights	0.462	0.297	0.125	0.115
b	0.09	0.06	0.11	0.15
SE(b)	0.011	0.016	0.047	0.045
Var(b)	1.11E-04	2.56E-04	0.002	0.002
weights	0.473	0.311	0.106	0.111

Appendix B7 Table 3. Average values for selectivity parameters p , a , b , L_{95} , L_{90} , L_{50} and SR with standard errors, variances, CVs and 90% confidence intervals from SELECT models fit to large and small camera video data collected during 2003-2006 on Georges Bank and in the Mid-Atlantic.

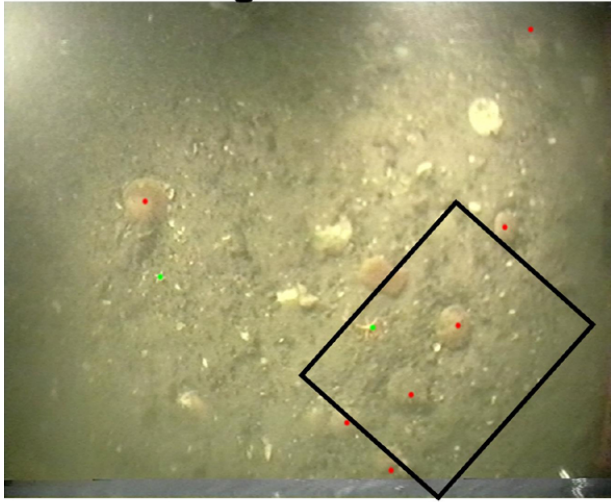
n=4 for experiment from 2003- 2006

	Split (%)	L95(mm)	L90(mm)	L50(mm)	SR(mm)	a	b
Average	84.15	79.43	71.41	47.71	23.44	-4.84	0.10
Var	1.87E-05	2.867	2.867	2.867	11.207	0.457	0.000
SE	0.004	1.693	1.693	1.693	3.348	0.676	0.017
CV	5.14E-05	0.021	0.024	0.035	0.143	-0.140	0.163
CI90	0.008	3.319	3.319	3.319	6.561	1.325	0.033
Upper	84.16	82.75	74.73	51.03	30.01	-3.52	0.14
Lower	84.14	76.11	68.09	44.39	16.88	-6.17	0.07



Appendix B7 Figure 1. SMAST video stations during 2003-2006. Stations where scallops were detected by both cameras in at least two quadrats were used to estimate selectivity curves and are highlighted in red.

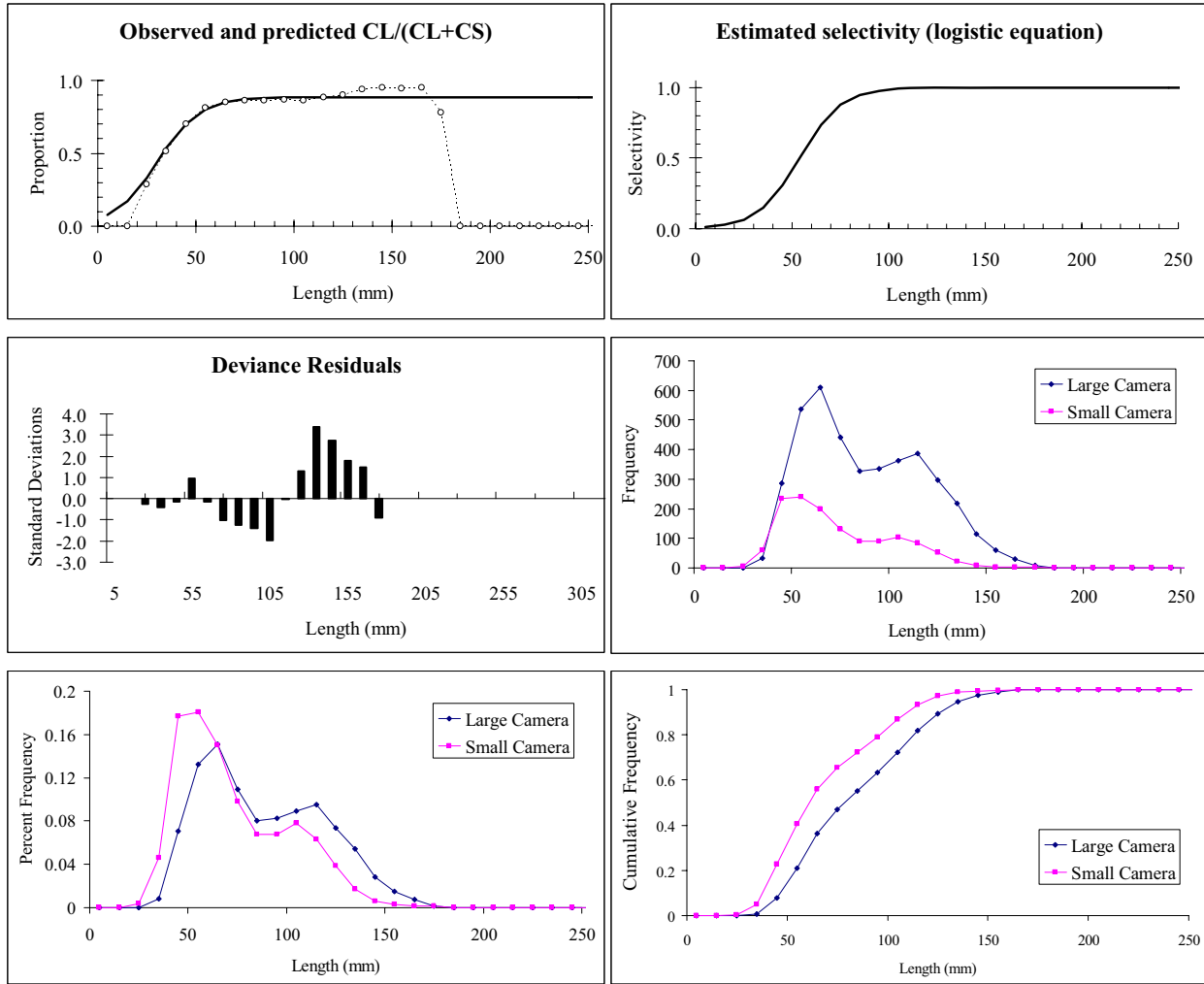
Large Camera



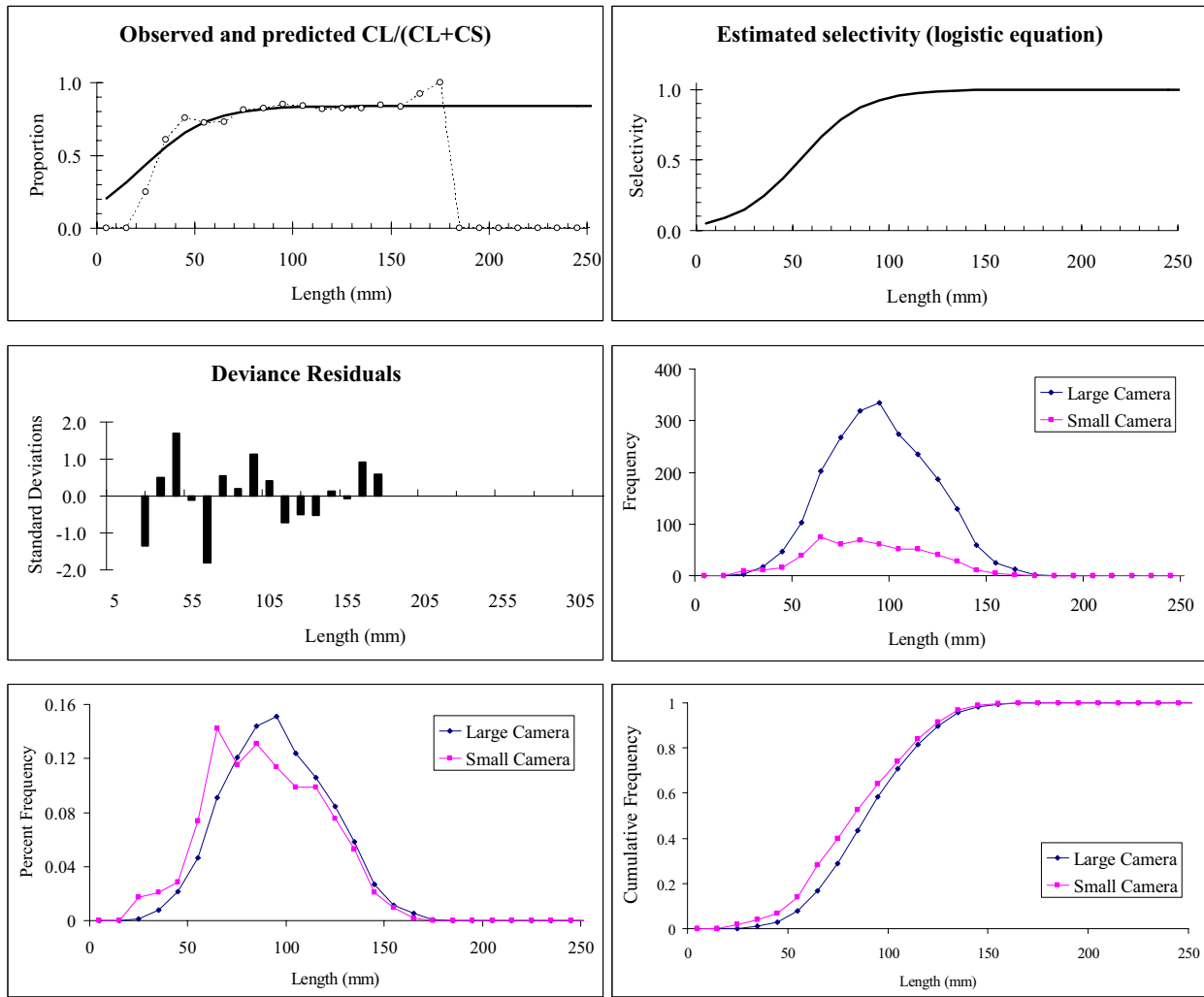
Small Camera



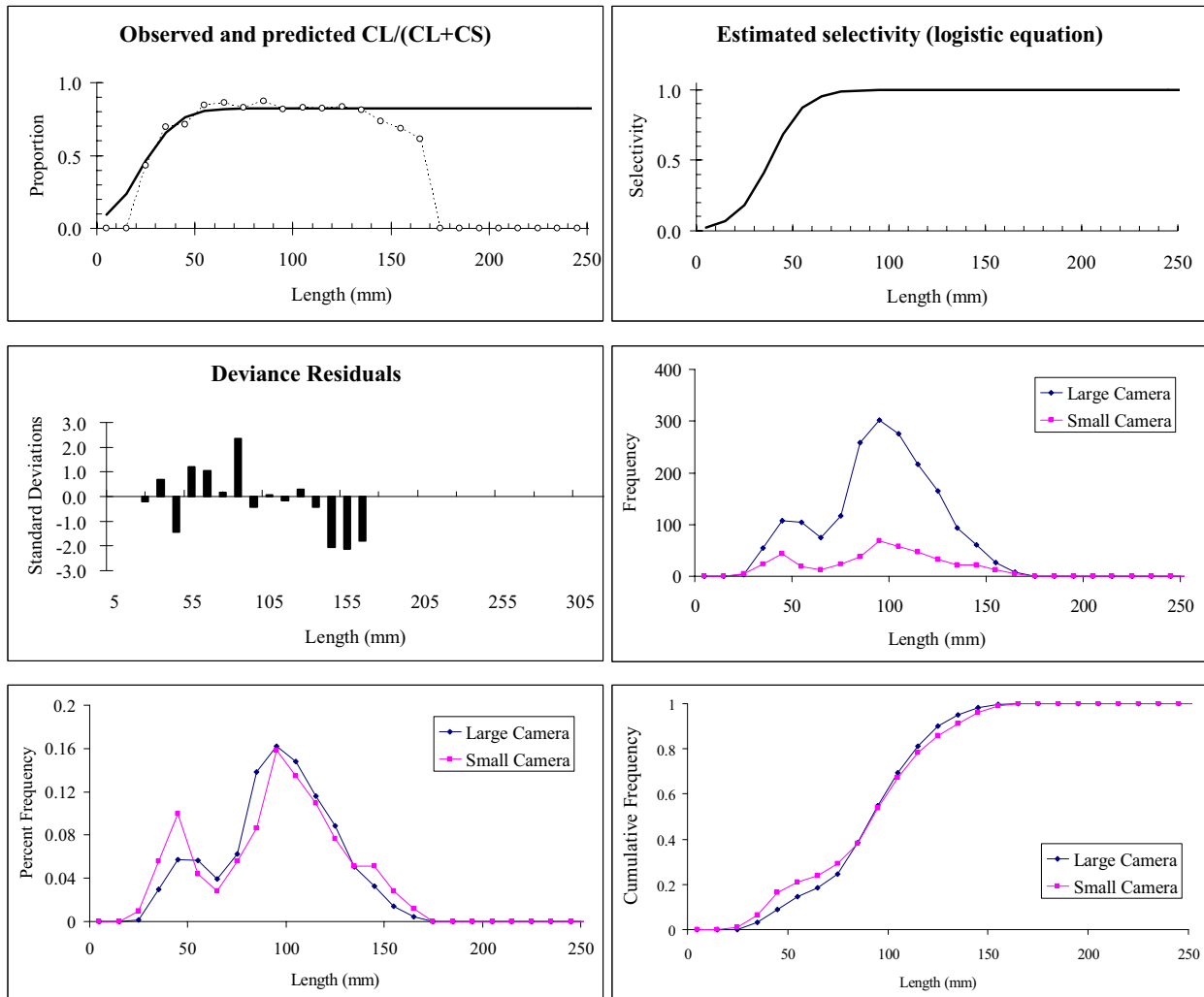
Appendix B7 Figure 2. *Left:* Large camera image with small camera inset. *Right:* Small camera inset enlarged



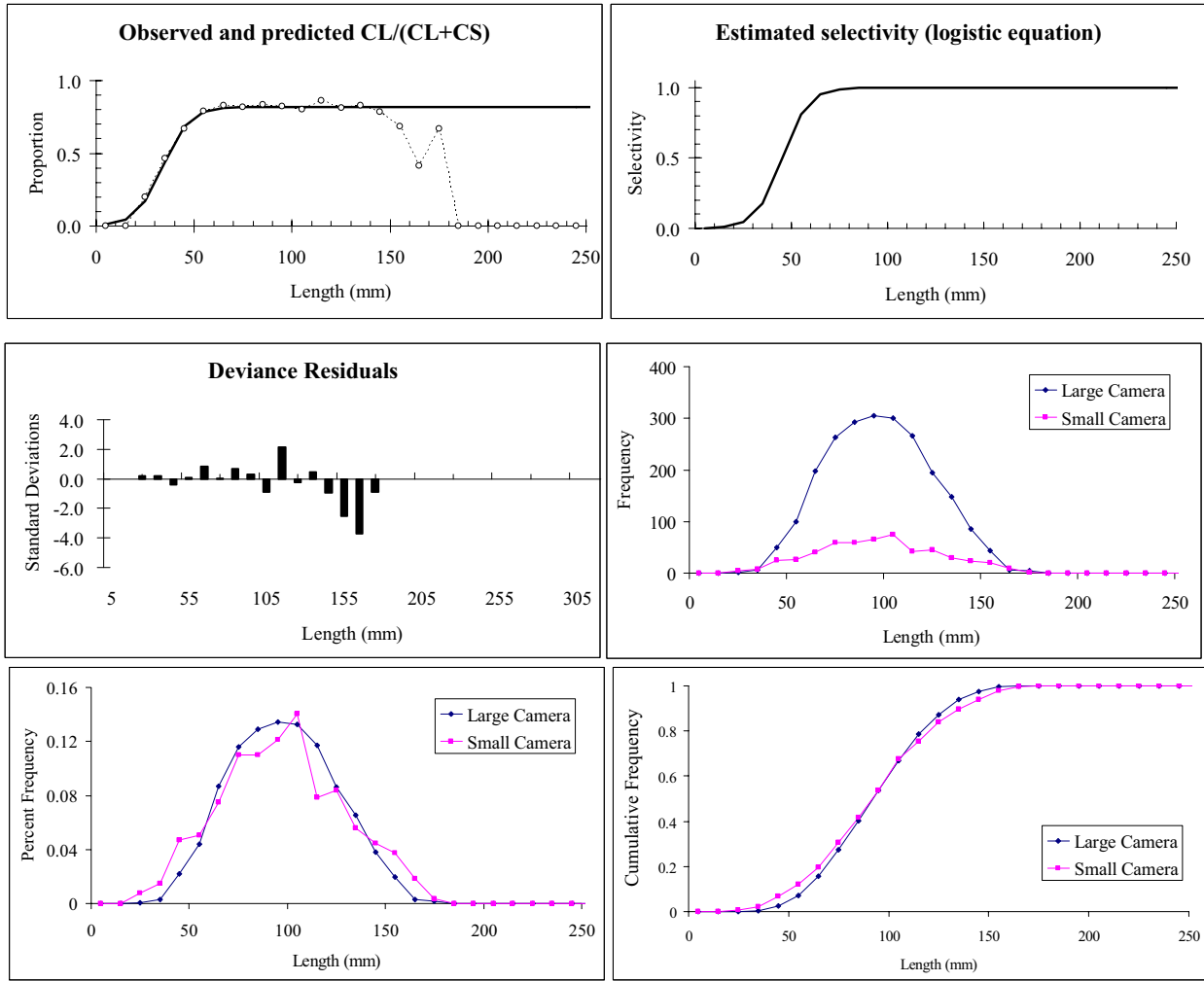
Appendix B7 Figure 3. Observed and predicted shell height measurements, Millar SELECT estimated selectivity logistic curve, deviance residuals for SELECT model, original shell height composition data (frequencies, percent, and cumulative frequencies) for video survey data from the Mid-Atlantic Bight and Georges Bank during 2003.



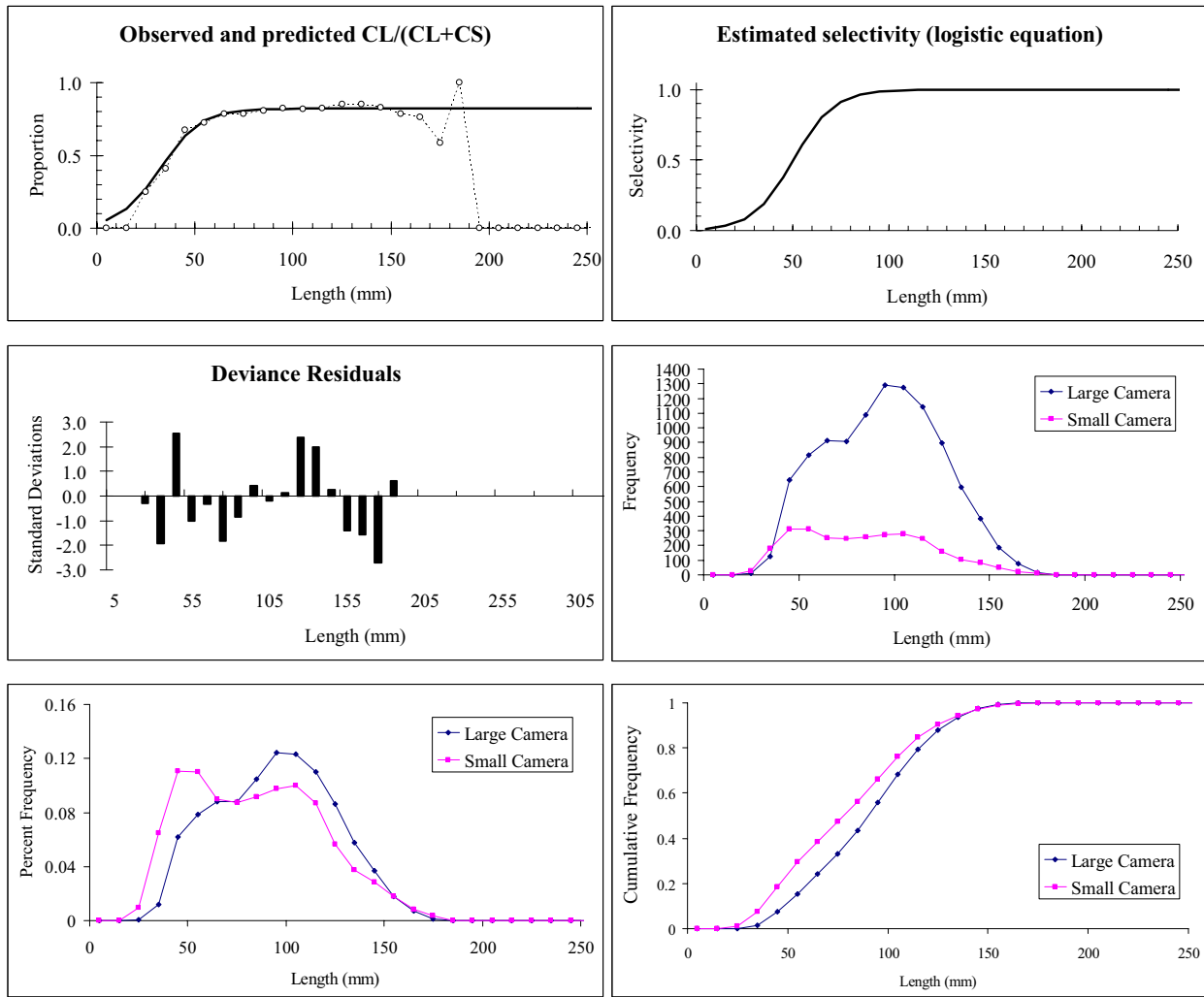
Appendix B7 Figure 4. Observed and predicted shell height measurements, Millar SELECT estimated selectivity logistic curve, deviance residuals for SELECT model, original shell height composition data (frequencies, percent, and cumulative frequencies) for video survey data from the Mid-Atlantic Bight and Georges Bank during 2004.



Appendix B7 Figure 5. Observed and predicted shell height measurements, Millar SELECT estimated selectivity logistic curve, deviance residuals for SELECT model, original shell height composition data (frequencies, percent, and cumulative frequencies) for video survey data from the Mid-Atlantic Bight and Georges Bank during 2005.



Appendix B7 Figure 6. Observed and predicted shell height measurements, Millar SELECT estimated selectivity logistic curve, deviance residuals for SELECT model, original shell height composition data (frequencies, percent, and cumulative frequencies) for video survey data from the Mid-Atlantic Bight and Georges Bank during 2006.



Appendix B7 Figure 7. Observed and predicted shell height measurements, Millar SELECT estimated selectivity logistic curve, deviance residuals for SELECT model, original shell height composition data (frequencies, percent, and cumulative frequencies) for video survey data from the Mid-Atlantic Bight and Georges Bank during 2003-2006 (combined).

APPENDIX B8: NEFSC survey dredge selectivity and efficiency estimates for sea scallops on Georges Bank and in the Mid-Atlantic Bight during 2003-2006, based on SMAST video survey data⁷

Selectivity curves and sampling efficiency were estimated for the NEFSC sea scallop dredge survey by using a statistical model to compare length composition data from the dredge survey to length composition data from the large camera and small camera SMAST video surveys. In comparisons, the video data were assumed to sample a range of size groups with full efficiency and selectivity. Selectivity curves for the NEFSC survey dredge based on SMAST video small camera survey data indicate that the survey dredge has constant selectivity for sea scallops 40+ mm SH (Figure 1). Curves based on SMAST large camera survey data show the same general pattern but are not as useful for characterizing dredge selectivity for sea scallops less than 70+ mm SH (Figure 2).

Overall, survey dredge efficiency averaged 0.38 (CV 10%). Averaging estimates from large and small camera comparisons, survey dredge efficiency was 0.40 (CV 7%) for the Mid-Atlantic Bight and 0.37 (CV 18%) for Georges Bank. Based on small camera comparisons for scallops 45+ mm SH, survey dredge efficiency averaged 0.43 (CV 9%) in Mid-Atlantic Bight and 0.38 (CV 32%) on Georges Bank during 2003-2006 (Table 1). Based on large camera comparisons for scallops 70+ mm SH, dredge survey efficiency averaged 0.36 (CV 11%) in the Mid-Atlantic Bight and 0.36 (CV 18%) on Georges Bank during 2003-2006 (Table 2). The CV calculated using the standard deviation of all eight dredge efficiency estimates was 19%.

Assumptions about measurement errors in length data from the video survey did not appreciably affect results.

Introduction and Methods

In this analysis, NEFSC scallop dredge survey selectivity curves and efficiency were estimated using SMAST video survey data for Georges Bank and the Mid-Atlantic Bight during 2003-2006. Efficiency estimates for the NEFSC survey dredge from this analysis should be more accurate than previous estimates based on SMAST video data (NEFSC 2004) because they are based on a wider range of sea scallop shell height data, data from additional surveys, and refined assumptions about survey dredge selectivity. Efficiency estimates in NEFSC (2004) were for subregions while estimates from this analysis are for Georges Bank and the Mid-Atlantic Bight as a whole.

The assumed survey dredge selectivity curve used in previous sea scallops assessments (NEFSC 2004) indicates survey dredge survey selectivity is highest between 40 and 50 mm SH, declines rapidly and is relatively constant after 60 mm SH (Figure 3). One hypothesis used to explain this selectivity pattern is that the small mesh liner in the survey dredge generates a pressure wave in front of the dredge that differentially reduces catches of large scallops. Results from this analysis suggest that the liner probably affects catches over a wider range of shell heights to the same extent. The selectivity curve used in previous assessments was estimated by

⁷ Larry D. Jacobson (Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543), Catherine O'Keefe, Michael C. Marino III (School for Marine Science and Technology, University of Massachusetts Dartmouth, 706 South Rodney French Boulevard, New Bedford, MA 02744-1221), and Antonie Chute (Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA, 02543)

comparing catch at shell height data from the current survey dredge, which has a small mesh liner, to catch at length data from a similar unlined dredge (Serchuk and Smolowitz 1980).⁸ Based on these selectivity assumptions, efficiency estimates for the NEFSC survey dredge in the last assessment (NEFSC 2004) were for sea scallops 90+ mm SH.

Data used in the analysis were for NEFSC shellfish strata sampled randomly by the dredge survey and sampled completely by the video survey (Figure 4).⁹ Only a few dredge survey tows were available for some strata in most years because the dredge survey has a stratified random design with sampling roughly proportional to stratum area in most cases. The video survey uses a fixed survey design with a relatively large number of stations across the entire area of each stratum.

The dredge and video surveys do not constitute paired tow experiments, which would be ideal for estimating selectivity and efficiency. Therefore, the underlying population length composition sampled in the dredge and video surveys is the same only in expectation across a large area and large number of samples. Histograms of numbers per tow in the dredge survey and numbers counted per tow in each video image indicate skewed and highly variable distributions for catch in both surveys (Figures 5-10).

Video survey data are available from both “large” and “small” cameras, which are both used at each station. Marino et al.’s (2007; see Appendix B6) results indicate that the survey dredge has ≥90% selectivity for sea scallops 70+ mm SH. The large camera samples a larger number of scallops and is therefore better for estimating densities of medium to large scallops. The effective sampling area for the small camera (0.788 m²) is a portion of the effective area (3.235 m²) for large camera. However, small camera resolution and probability of detection are higher for small scallops.

To scale video data for analysis, densities at size were calculated

$$\eta_L = n_L \frac{N}{n} \frac{1}{4AK} 100$$

where N is the total number counted (but not necessarily measured), n is the total number measured, n_L is the number measured for length group L , and K is the number of video stations. Data were collected from 4 images per station and the effective area of the video camera is A ($A=3.235$ m² for the large camera and 0.788 m² for the small camera, including adjustments for the scallops seen on the edge of the sampling area). Densities as numbers per m² were scaled for analysis to numbers per 100 m² for convenience. N and n include all size groups.

To scale dredge survey data for analysis, densities at size were calculated

$$\delta_L = \frac{d_L}{8(0.3048)1853} 100$$

⁸ Parameters for the dredge selectivity curve used in the previous assessment (NEFSC 2004) are: $a=14.3322$, $b=0.266807$ and $c=0.714879$ (see below).

⁹ NEFSC shellfish strata used for the Georges Bank region in each year were: 46, 47, 49, 50, 51, 52, 53, 54, 55, 59, 61, 621, 631, 651, 661, 71 and 74 except that stratum 74 was not used for 2005 because it was not sampled during the 2005 dredge survey. Strata used for the Mid-Atlantic Bight region in each year were: 6, 7, 10, 11, 14, 15, 18, 19, 22, 23, 24, 26, 27, 28, 30, 31, 33 and 34.

where the survey data was d_L in units of mean numbers per standard tow, the survey dredge is 8 ft or 8(0.3048) m wide and the standard tow is 1 nm=1,853 m.

Selectivity

Length measurements are less precise in the video survey than in the dredge survey, with standard deviations for measurement errors of about 6.1 mm (Stokesbury et al., in prep). To make dredge and video length data as comparable as possible, selectivity curves were fit with and without adding simulated measurement errors to the dredge survey data. The idea was to generate measurement errors in the dredge survey data that were of similar magnitude to the measurement errors in the video survey. It was not possible to remove measurement errors from the video survey, although the latter approach might be seem ideal intuitively. Based on Stokesbury et al. (in prep.), simulated length measurement errors were additive and from a truncated normal distribution with a standard deviation of 6.1 mm.

Millar's (1992) SELECT model was modified and used to fit a three parameter declining logistic selectivity curve with a right hand offset. The model is:

$$s_L = \left[1 - \frac{1}{1 + e^{a-bL}} \right] (1 - c) + c$$

and

$$S_L = s_L / \max(s_L)$$

where a , b and c are parameters and SH is the final estimate. Note that the curve is scaled to a maximum value of one in contrast to Millar's original approach, which did not rescale selectivity curves. Rescaling makes the curves more flexible, easier to interpret and enhances estimability.

It was difficult to calculate effective sample size for data from either survey in this analysis, particularly after the data were scaled to units of numbers per 100 m². Uncertainty about effective sample size prevented calculation of variances for selectivity parameter estimates within the SELECT model used to fit the selectivity curves but had no effect on estimates or general results. Bootstrapping or Bayesian procedures for estimating variance are a topic for future research.

The choice of curve was based on precedent and preliminary analysis of dredge and video survey data. The selectivity curves used in this analysis for the NEFSC dredge are the same general type and shape as the curve used for the NEFSC survey dredge in recent assessments (Figure 3). The most important feature of this type of curve is that selectivity decreases towards an asymptote selectivity as sea scallop shell height increases. The general shape of the selectivity curve used in this analysis was reasonable (see below). In retrospect, it may have been possible to use a simpler, 2 parameter curve with some statistical advantages but there would be no appreciable effect on conclusions.

The primary purpose of the analysis with large camera comparisons was to determine the general shape of the dredge selectivity curve and efficiency for 70+ mm SH. Large camera comparisons may be particularly useful for estimating dredge survey efficiency because the large camera samples more scallops (over a narrow range of full selectivity) than the small camera. Small camera comparisons were used to include sea scallops < 70 mm SH, at the expense of lower numbers of samples, particularly for larger sizes.

Based on preliminary analysis and available data, size groups included in the analysis were 35-140 mm for large camera comparisons and 20-135 mm for small camera comparisons. Use of smaller or larger size groups complicated parameter estimation, possibly because the smallest and largest size groups were poorly sampled. In contrast, the lower bound for dredge survey data in the previous assessment was 40 mm SH. Forty mm is approximately the same as the spacing of mesh in the liner of the dredge (38 mm). As described in Marino et al. (2007), the large camera video survey has an increasing logistic shaped selectivity curve that reaches 90% at about 70 mm SH. For large scallops, the dredge survey selectivity is thought to be low and constant while the large camera video survey selectivity is known to be high and constant. For small scallops, selectivity is low and changing with size in the large camera survey and uncertain but thought to be relatively high and changing with size in the dredge survey. For small scallops, the ratio of catch in the dredge gear to total catch (dredge + video gear), which is used to estimate selectivity, is variable and selectivity estimates for small scallops are likely to be imprecise and biased.

Dredge efficiency

Dredge efficiency in this analysis is the probability of capture for scallops above a certain minimum size in the path of the survey dredge. This definition differs from conventional definitions (and the definition used in the CASA model) that define efficiency in terms of capture efficiency for sizes that are fully selected by the gear. However, the definitions are basically the same if sea scallops are all above the size at which the dredge selectivity curve is flat.

When estimating selectivity curves with typical ascending logistic selection patterns surveys, the split parameter in the SELECT model can be used to estimate gear efficiency. This is not possible for sea scallops using dredge and video survey because the sizes at 100% selection may not overlap and because the flat portion of the selectivity curves occurred at minimum selectivity values.

Based on Marino et al. (2007) efficiency was calculated for scallops 70+ mm based on large camera comparisons because the selectivity curves for both gears appear to be flat by about 70 mm SH. Based on selectivity curve results shown below, efficiency was calculated for scallops 45+ mm based on small camera comparisons.

Results

Selectivity curves were reasonably easy to fit once the poorly sampled largest and smallest sea scallop size groups were eliminated from the analysis. Large camera comparisons generally indicate that selectivity curves for the NEFSC survey dredge (Table 3) is flat for scallops 70+ mm SH (Figure 2). The curve for Mid-Atlantic Bight during 2004 from the large camera comparison was the notable exception (Figure 9). Small camera comparisons consistently indicate that survey dredge selectivity curves (Table 4) are flat or nearly flat for scallops 40+ mm SH (Figure 1).

Diagnostics indicate reasonable SELECT model fit in most cases (Figures 11-14), although runs of positive and negative residuals occurred in many cases. Assumptions about length measurement errors had minor effect on estimated selectivity curves (Figure 15).

Selectivity curve estimates appear to be robust to measurement errors in length data. The shapes of selectivity curves for small scallops based on large camera comparisons were variable

for reasons described above. In particular, the apparently steep increases in dredge selectivity below 70 mm SH based on large camera comparisons are artifacts due to possibly increasing selectivity in the dredge survey and declining selectivity in the large camera video survey. The apparently high selectivity at sizes less than 60 mm SH in the survey dredge selectivity curve used in the last assessment (Figure 3) was probably due to constant selectivity in the lined dredge and declining selectivity in the unlined dredge, which was used as the standard in comparisons (Serchuk and Smolowitz 1980).

Dredge efficiency

Dredge efficiency estimates were relatively consistent (Tables 1-2) and similar to estimates from the last assessment (NEFSC 2004). Based on large camera comparisons, dredge survey efficiency for scallops 70+ mm SH averaged 0.36 (CV 11%) in the Mid-Atlantic Bight (Mid-Atlantic Bight) and 0.36 (CV 18%) on Georges Bank (Georges Bank) during 2003-2006. Based on small camera comparisons, survey dredge efficiency for scallops 45+ mm SH averaged 0.43 (CV 9%) in Mid-Atlantic Bight and 0.38 (CV 32%) on Georges Bank during 2003-2006. Averaging large and small camera results, survey dredge efficiency was 0.40 (CV 7%) for Mid-Atlantic Bight and 0.37 (CV 18%) for Georges Bank. Overall, survey dredge efficiency averaged 0.38 (CV 10%). The consistency in efficiency estimates from the large and small camera comparisons is additional support for the hypothesis that survey dredge efficiency is flat above 35 mm SH.

References

- Marino MC, O'Keefe C, Jacobson LD. 2007. Selectivity of the SMAST camera video survey. (Appendix to this assessment)
- Millar RB. 1992. Estimating the size-selectivity of fishing gear by conditioning on the total catch. *J Amer Stat Assoc.* 87:962-968.
- Serchuk FM, Smolowitz RJ. 1980. Size selection of sea scallops by an offshore scallop survey dredge. *ICES C.M.* 1980/K:24.
- Stokesbury KDE, Jacobson LD, Allard M, Chute A, Harris BP, Jaffarian T, Marino MC II, Nogueira JI, Rago P. In prep. Quantification, effects and stock assessment modeling approaches for dealing with measurement error in body size data using Atlantic sea scallops as an example.

APPENDIX B8 Table 1. Efficiency estimates for sea scallop 70+ mm SH in the NEFSC survey dredge based on SMAST video survey (large camera) data.

Region	2003	2004	2005	2006	Average	CV
Mid-Atlantic Bight Georges Bank	0.308	0.353	0.402	0.374	0.359	0.110
	0.304	0.445	0.369	0.314	0.358	0.181

APPENDIX B8 Table 2. Efficiency estimates for sea scallop 45+ mm SH in the NEFSC survey dredge based on SMAST video survey (small camera) data.

Region	2003	2004	2005	2006	Average	CV
Mid-Atlantic Bight Georges Bank	0.467	0.424	0.456	0.382	0.432	0.088
	0.295	0.554	0.350	0.314	0.378	0.316

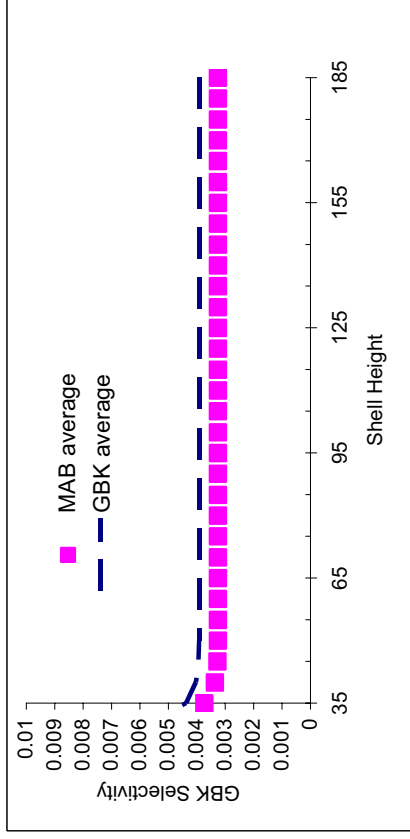
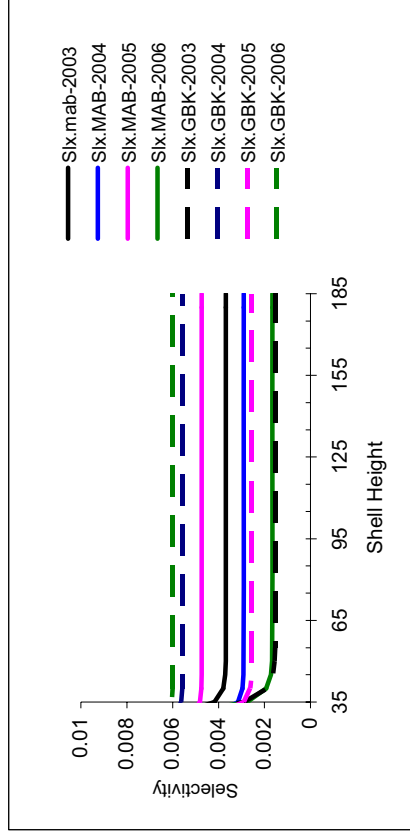
APPENDIX B8 Table 3. Selectivity curve parameter estimates for sea scallop 70+ mm SH in the NEFSC survey dredge based on SMAST video (large camera) comparisons (assuming length measurement errors with standard deviation = 6.1 mm). Estimates assuming no length measurement errors were similar.

Parameter	2003	2004	2005	2006
Mid-Atlantic Bight				
<i>a</i>	0.00006	0.00005	0.00006	0.00006
<i>b</i>	0.22548	0.03905	0.00010	0.07868
<i>c</i>	0.00010	0.00676	1.02865	0.01839
Split parameter <i>p</i>	0.99625	0.81521	0.30149	0.86735
Log likelihood	-43.5	-25.2	-23.1	-21.0
Georges Bank				
<i>a</i>	0.00006	0.00006	0.00006	0.00006
<i>b</i>	0.34678	0.88794	0.99988	0.76524
<i>c</i>	0.11557	0.78192	0.99988	0.57430
Split parameter <i>p</i>	0.03842	0.68330	0.99989	0.42654
Log likelihood	-11.9	-14.0	-11.0	-7.7

APPENDIX B8 Table 4. Selectivity curve parameter estimates for sea scallop 35+ mm SH in the NEFSC survey dredge based on SMAST video (small camera) comparisons (assuming length measurement errors with standard deviation = 6.1 mm). Estimates assuming no length measurement errors were similar.

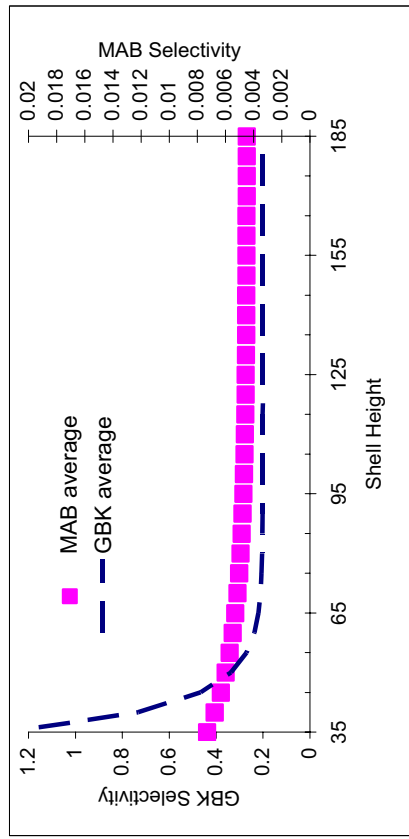
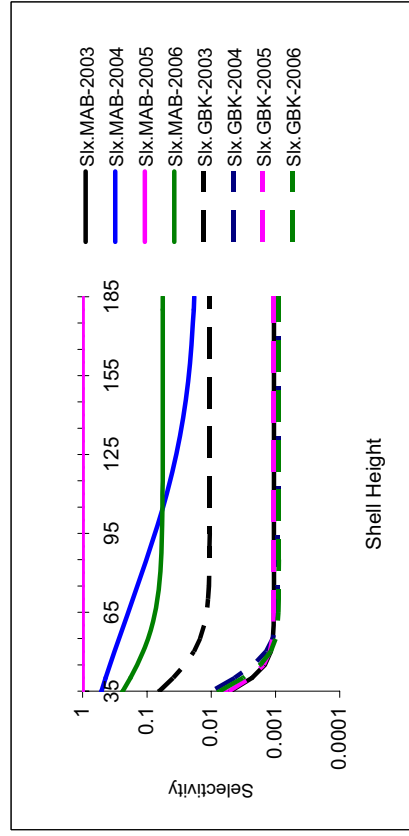
Parameter	2003	2004	2005	2006
Mid-Atlantic Bight				
<i>a</i>	0.00006	0.00006	0.00006	0.00006
<i>b</i>	0.30574	0.33378	0.38423	0.27451
<i>c</i>	0.00017	0.00010	0.00010	0.00010
Split parameter <i>p</i>	0.98729	0.99423	0.98980	0.99622
Log likelihood	-55.5	-26.9	-23.0	-20.8
Georges Bank				
<i>a</i>	0.00006	0.00006	0.00006	0.00006
<i>b</i>	0.22758	0.19620	0.15315	0.26664
<i>c</i>	0.05262	0.03953	0.02653	0.06963
Split parameter <i>p</i>	0.01431	0.00982	0.00793	0.01894
Log likelihood	-12.7	-13.3	-10.1	-7.0

Dredge survey selx based on small camera survey assuming 7% CV for length measurement errors

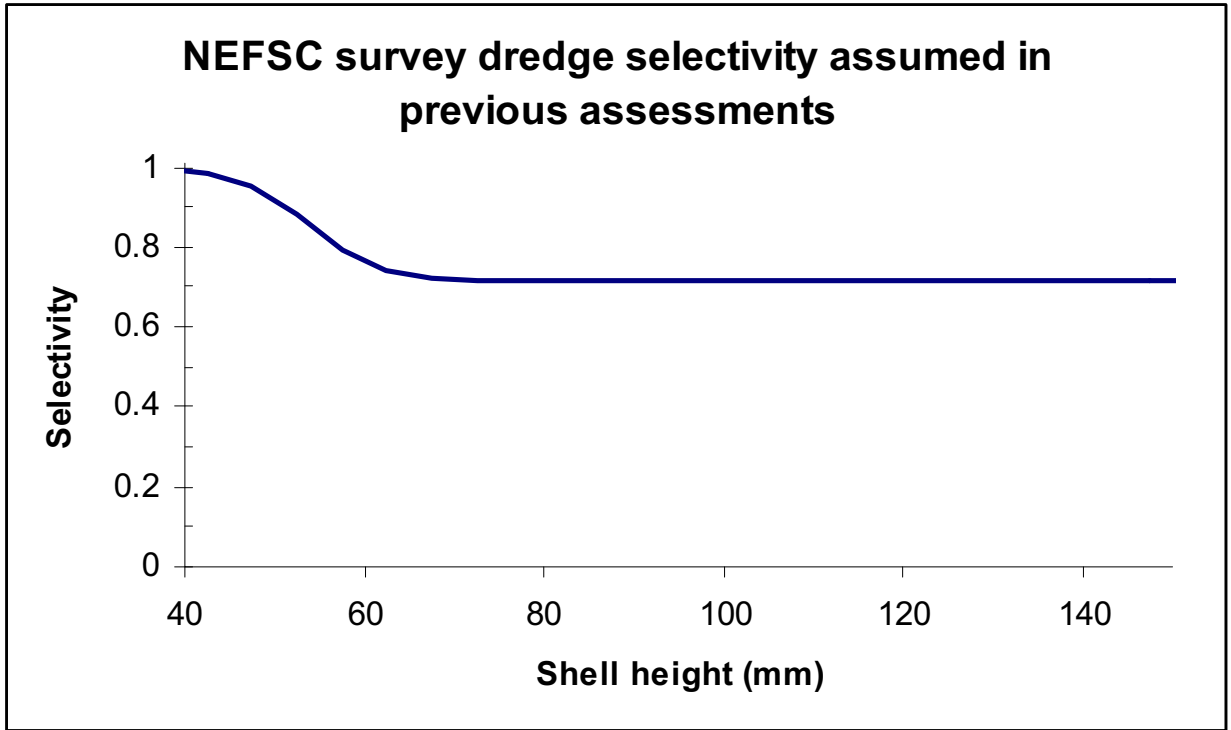


APPENDIX B8 Figure 1. Estimated selectivity curves based on small camera comparisons.

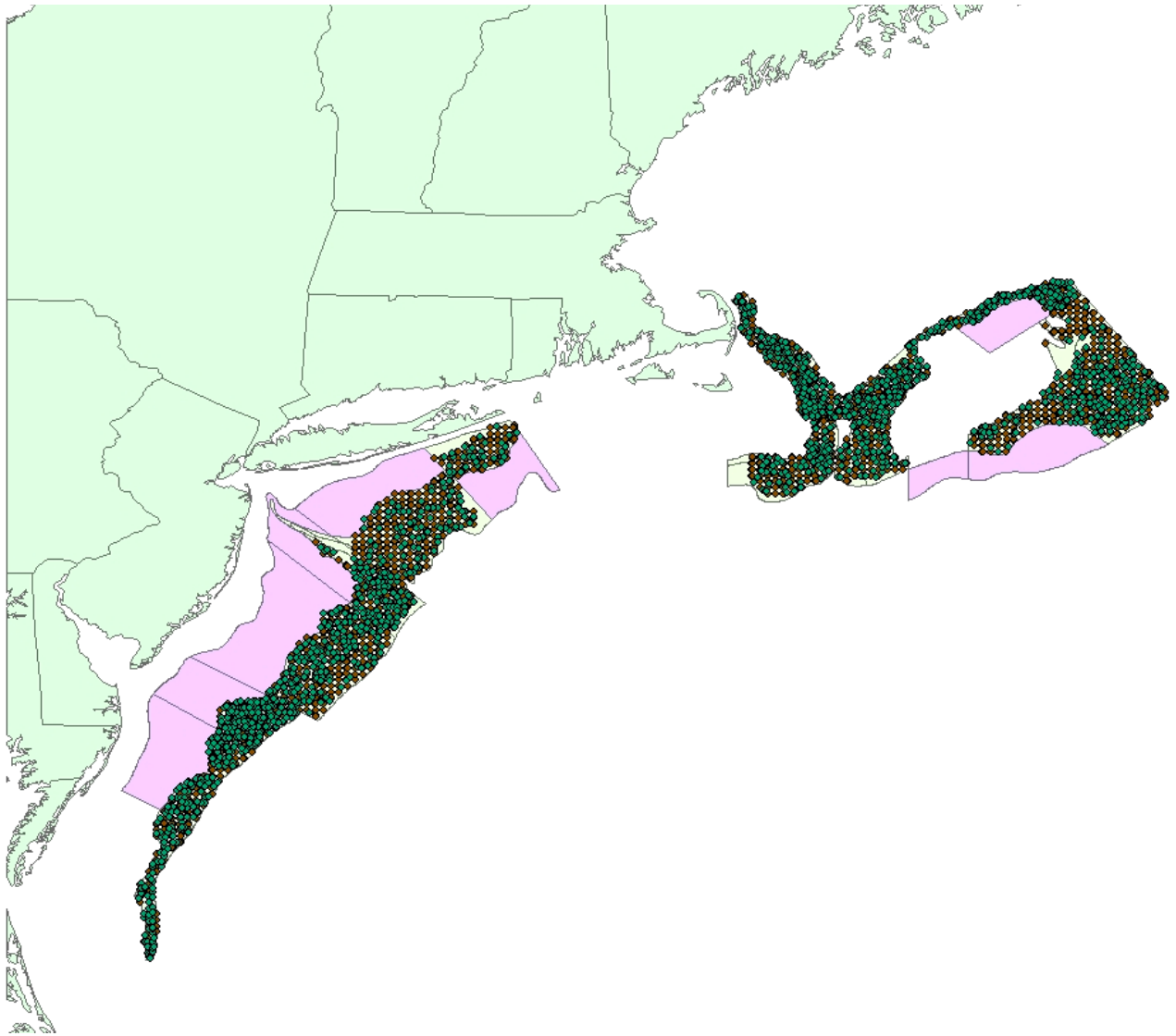
Dredge survey selx based on large camera survey assuming 7% CV for length measurement errors



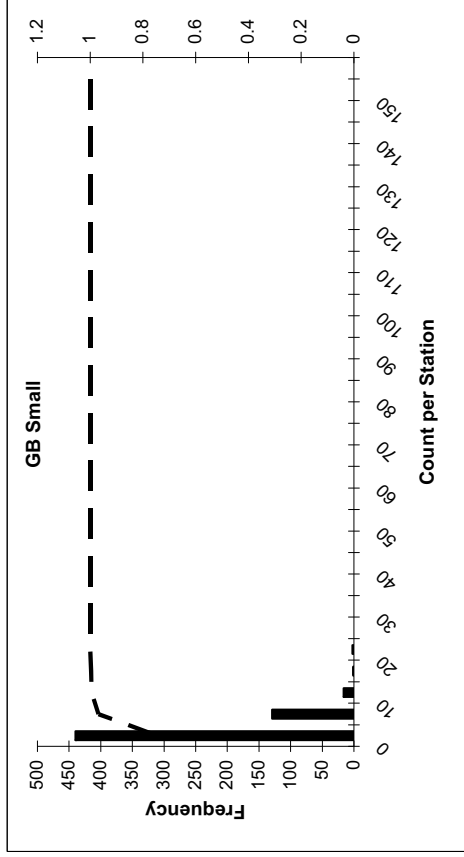
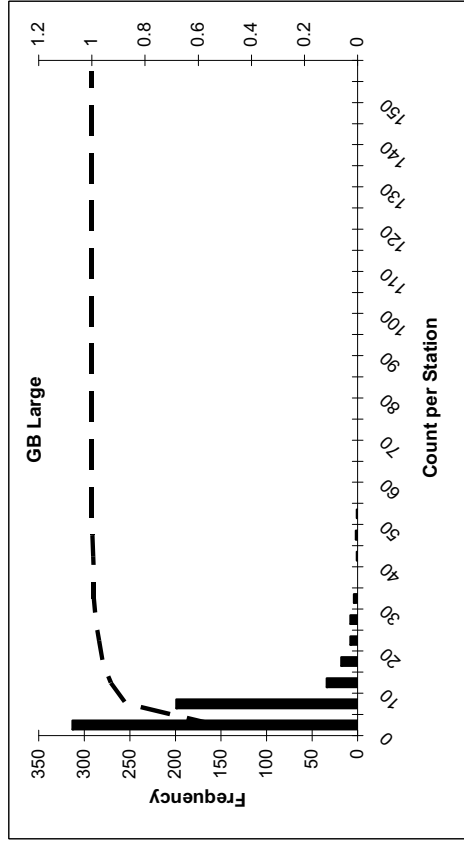
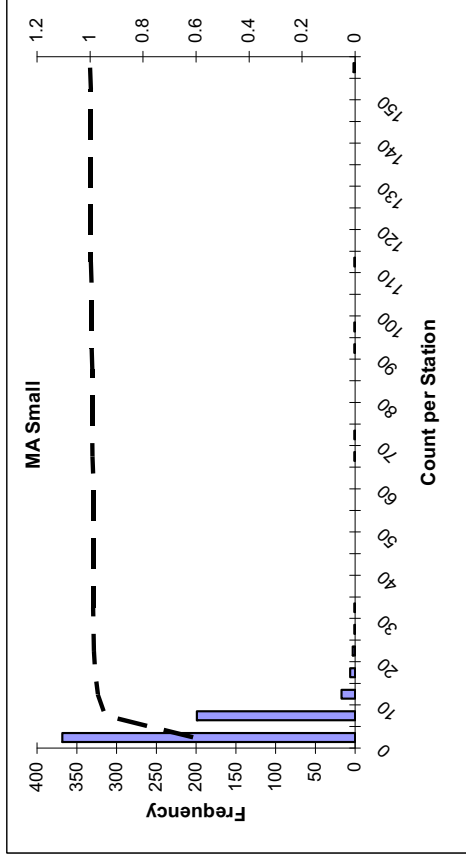
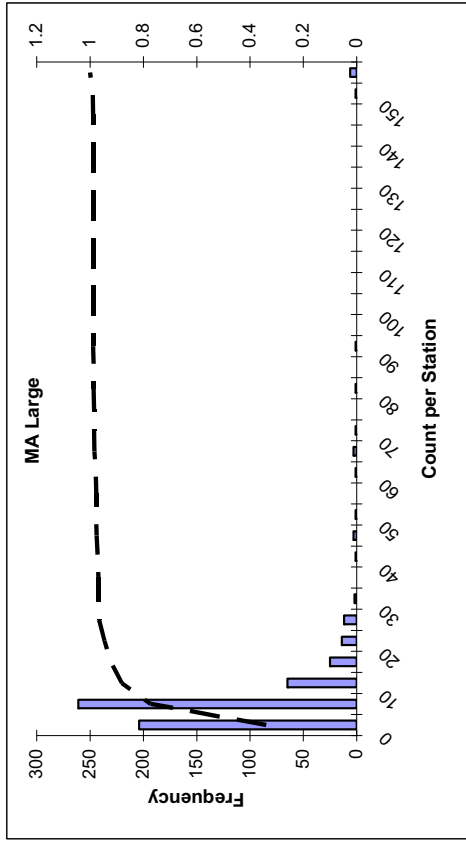
APPENDIX B8 Figure 2. Estimated selectivity curves based on large camera comparisons.



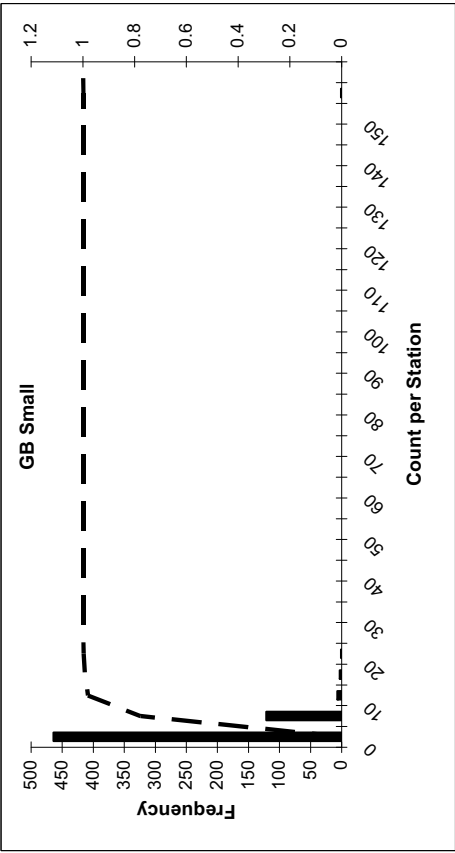
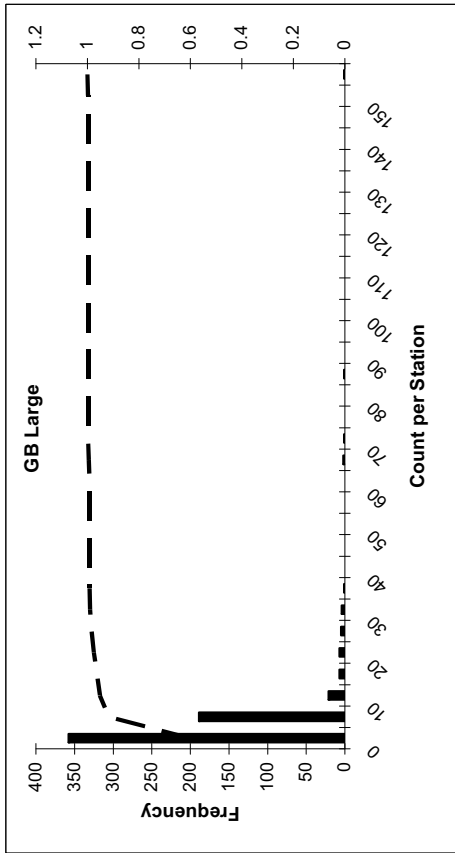
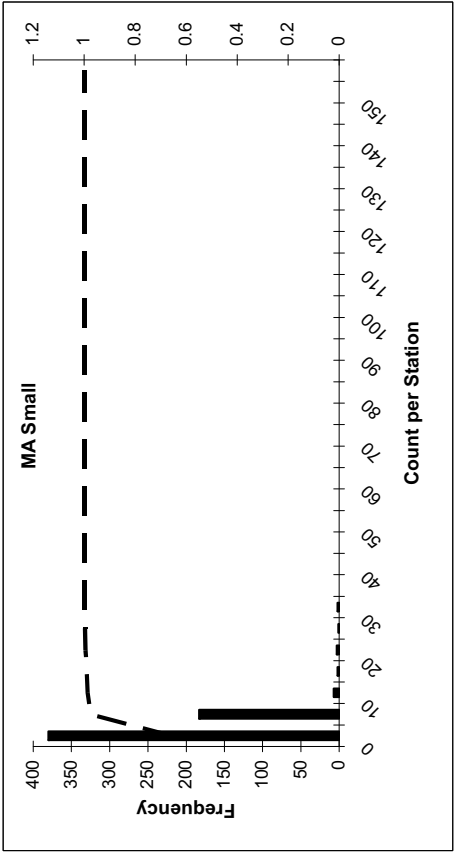
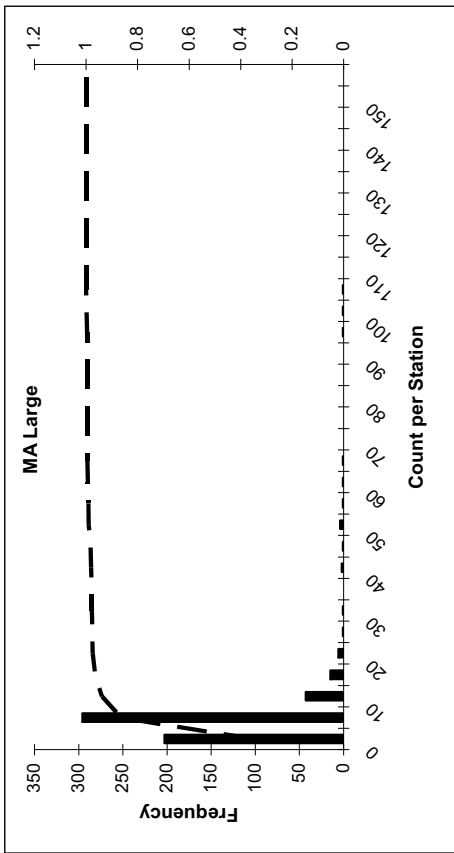
APPENDIX B8 Figure 3. Survey dredge selectivity curve for sea scallops assumed in previous assessments.



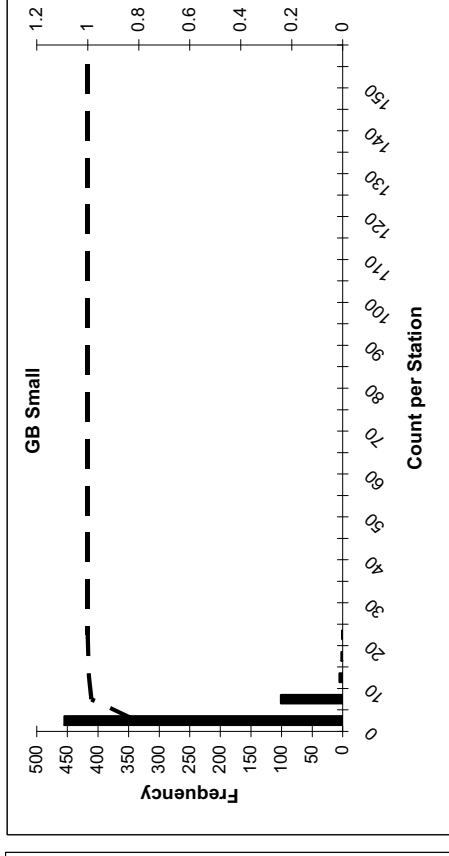
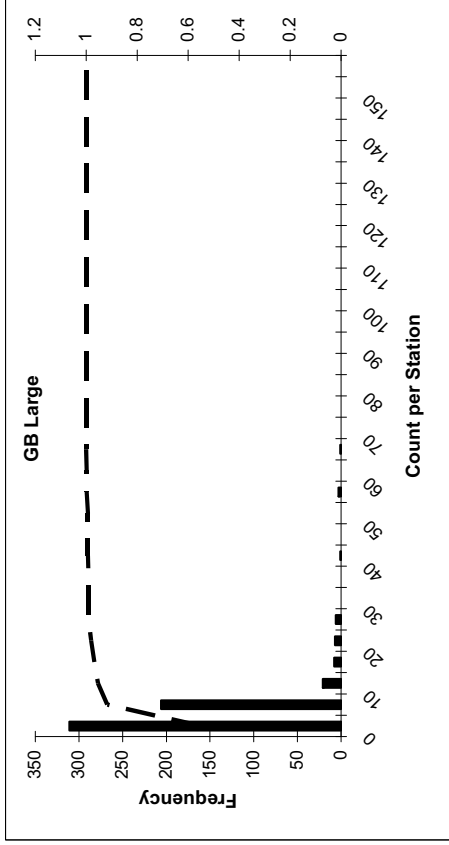
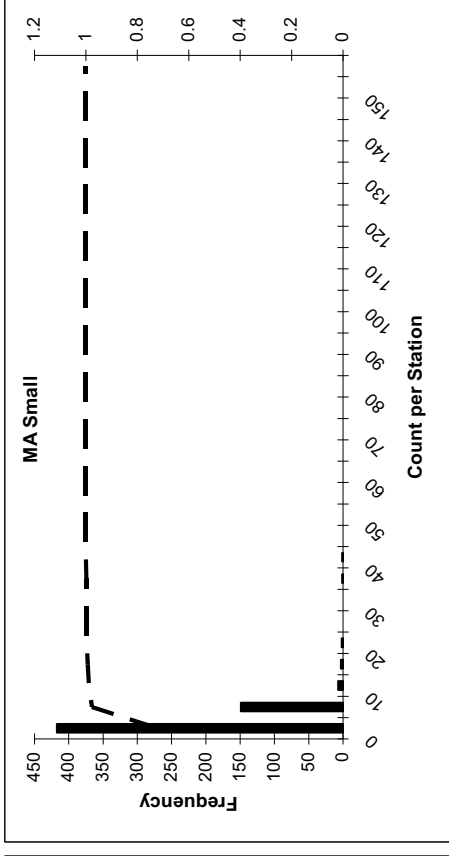
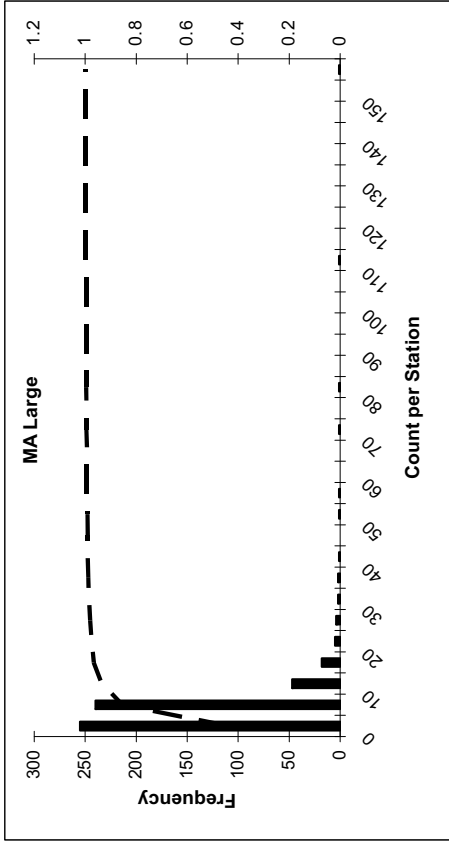
APPENDIX B8 Figure 4. Location of NEFSC shellfish strata and video stations for data used to estimate dredge survey selectivity and efficiency.



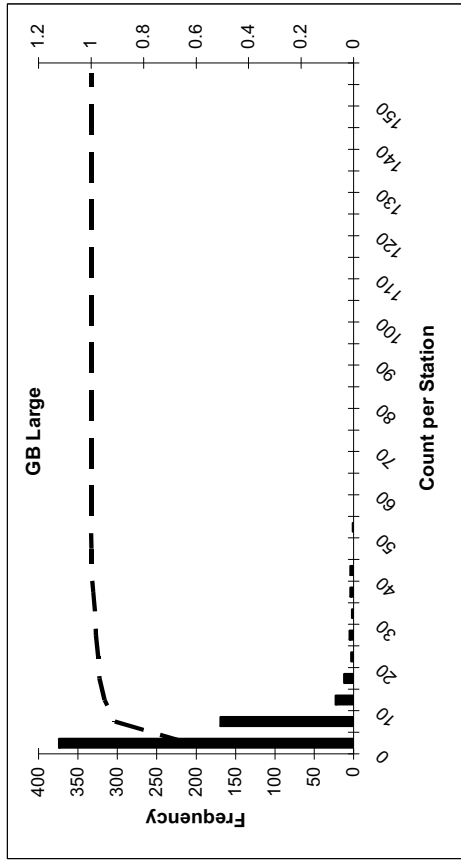
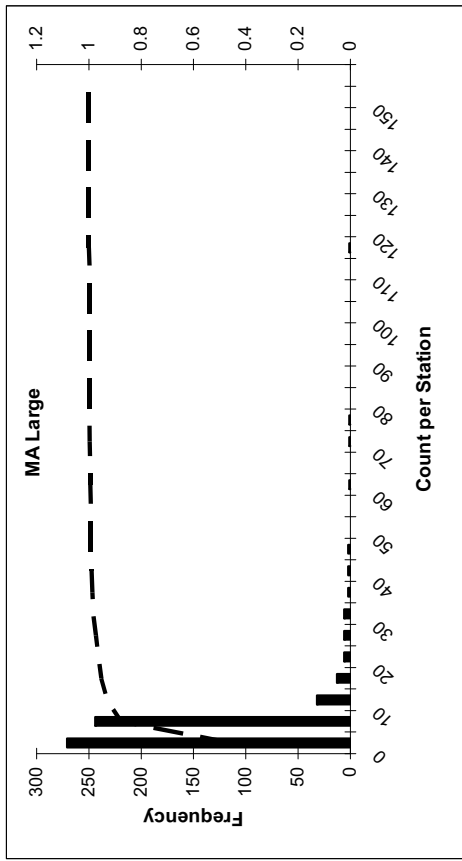
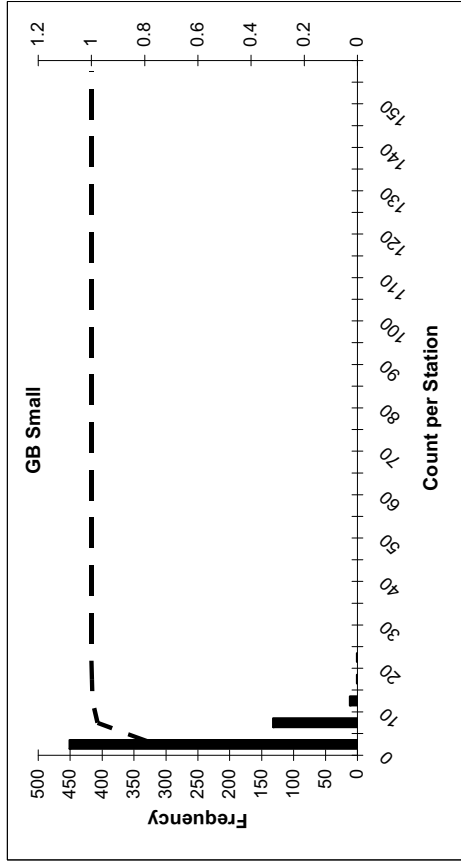
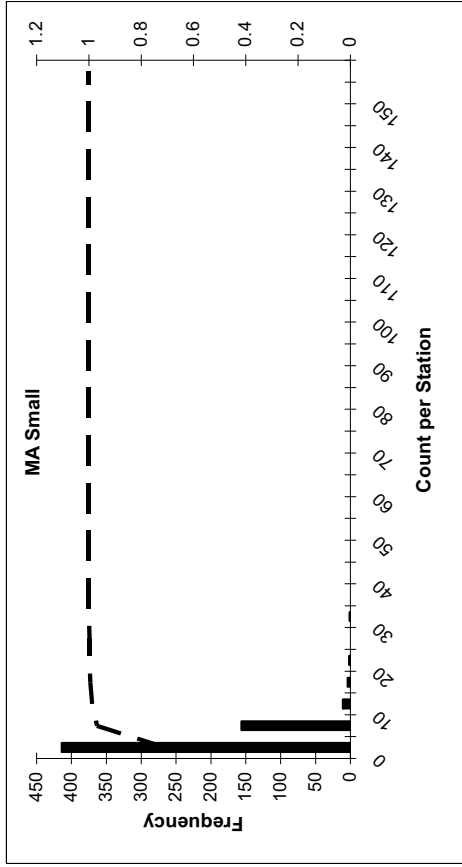
APPENDIX B8 Figure 5. Frequency distributions (bars) and cumulative distributions (broken line) for sea scallops counted in video images for video stations during 2003 used in this analysis.



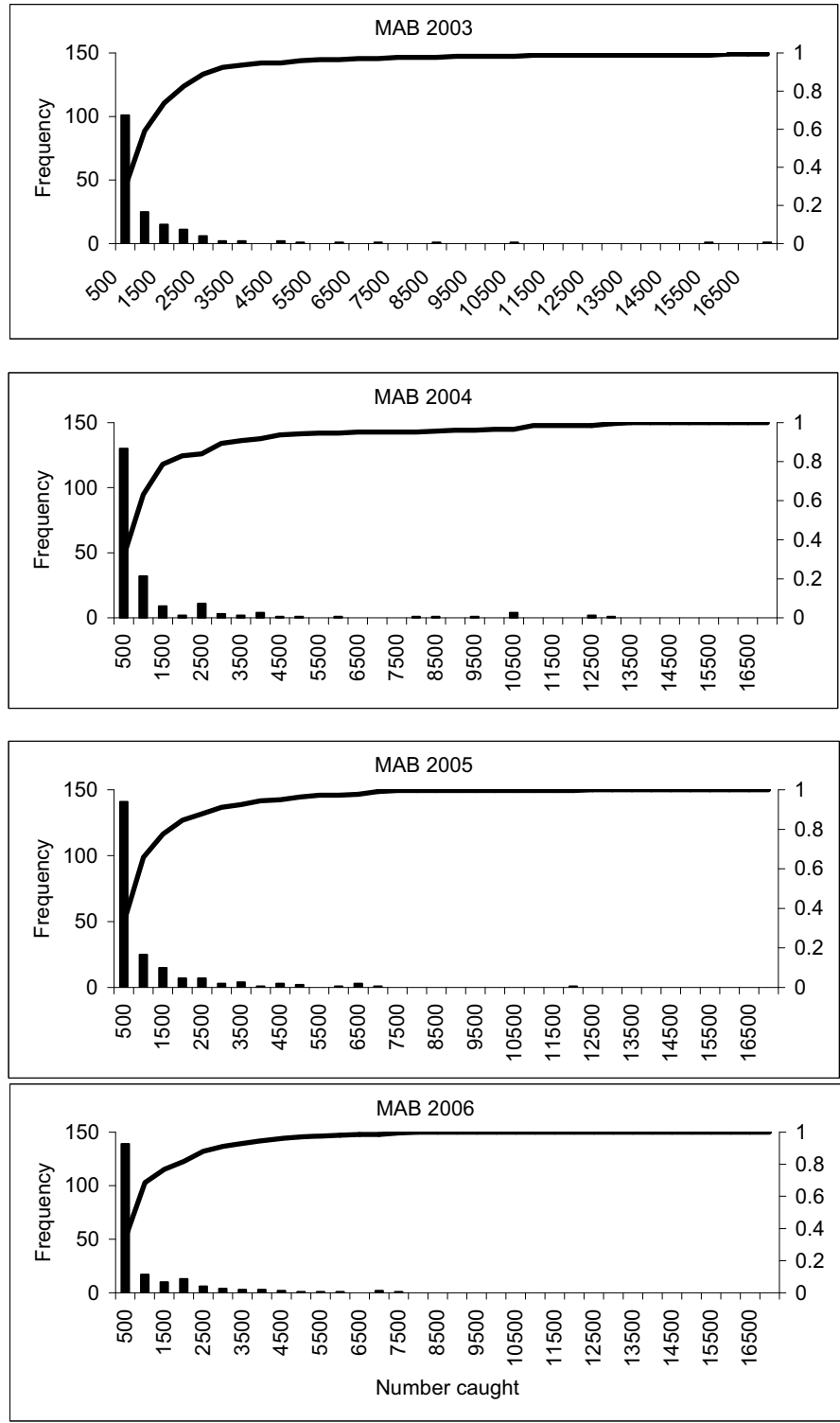
APPENDIX B8 Figure 6. Frequency distributions (bars) and cumulative distributions (broken line) for sea scallops counted in video images for video stations during 2004 used in this analysis.



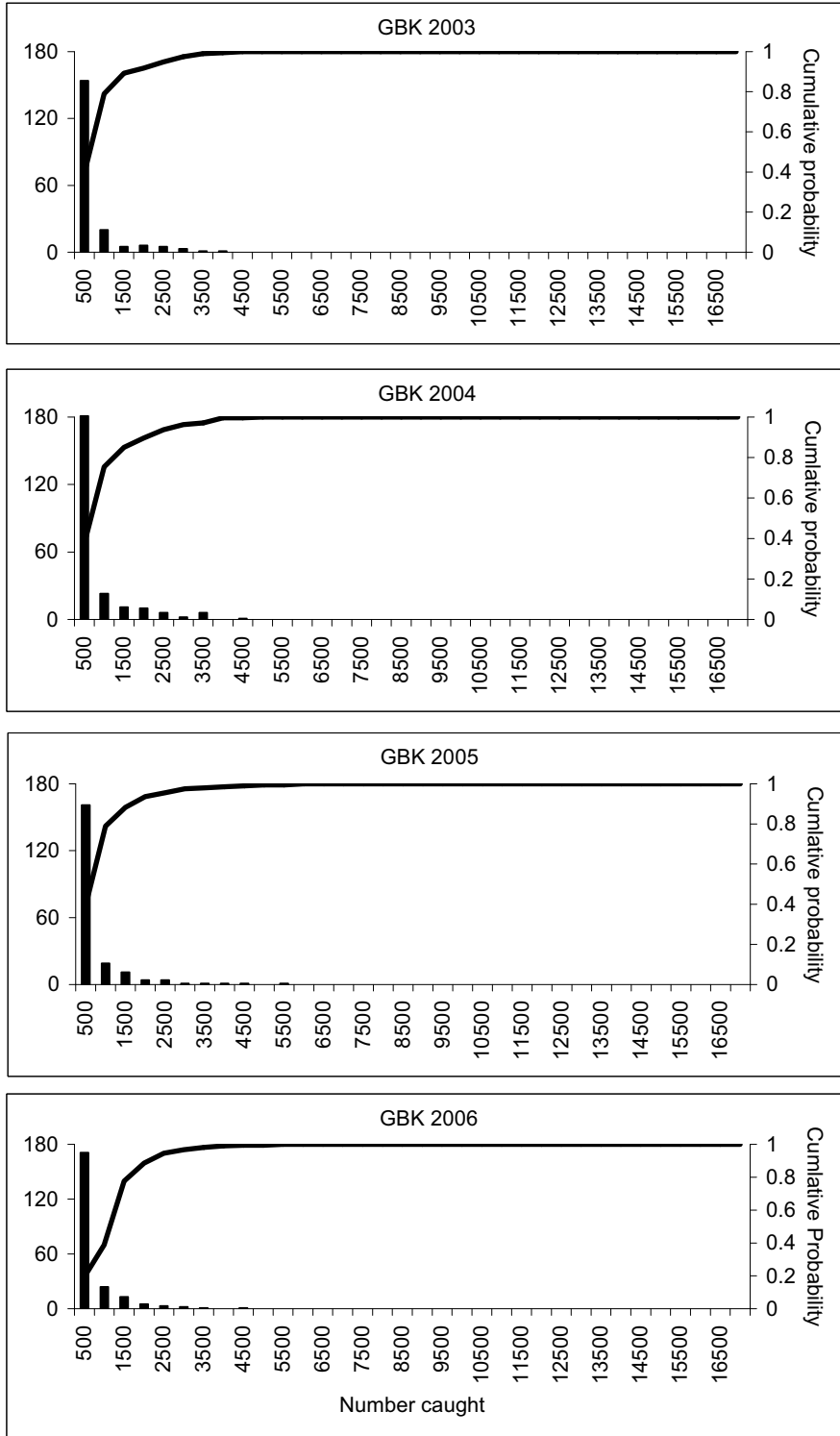
APPENDIX B8 Figure 7. Frequency distributions (bars) and cumulative distributions (broken line) for sea scallops counted in video images for video stations during 2005 used in this analysis.



APPENDIX B8 Figure 8. Frequency distributions (bars) and cumulative distributions (broken line) for sea scallops counted in video images for video stations during 2006 used in this analysis.

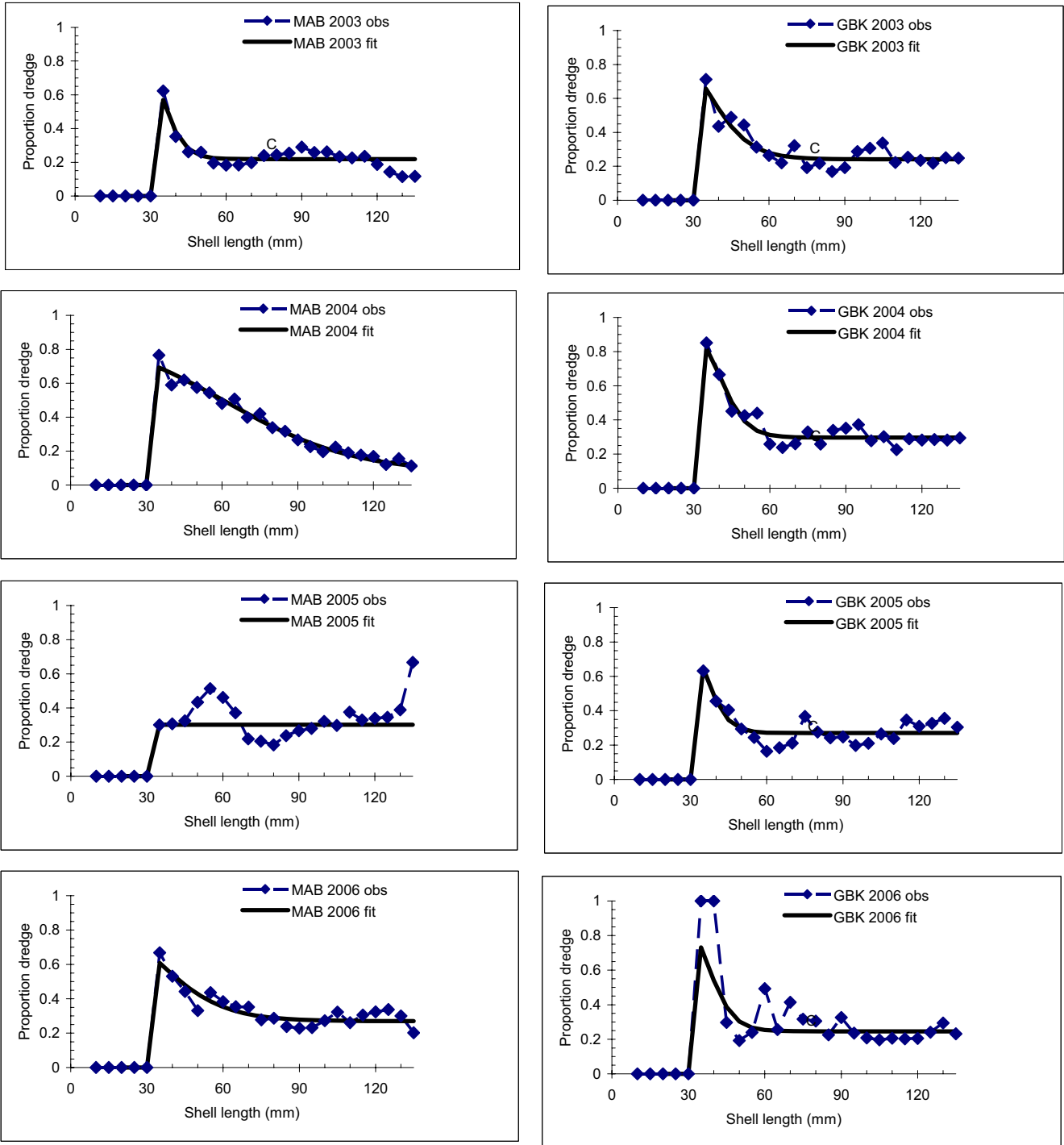


APPENDIX B8 Figure 9. Frequency distributions (bars) and cumulative distributions (solid lines) for sea scallops numbers per tow in dredge survey catches in the Mid-Atlantic Bight during 2003-2006.



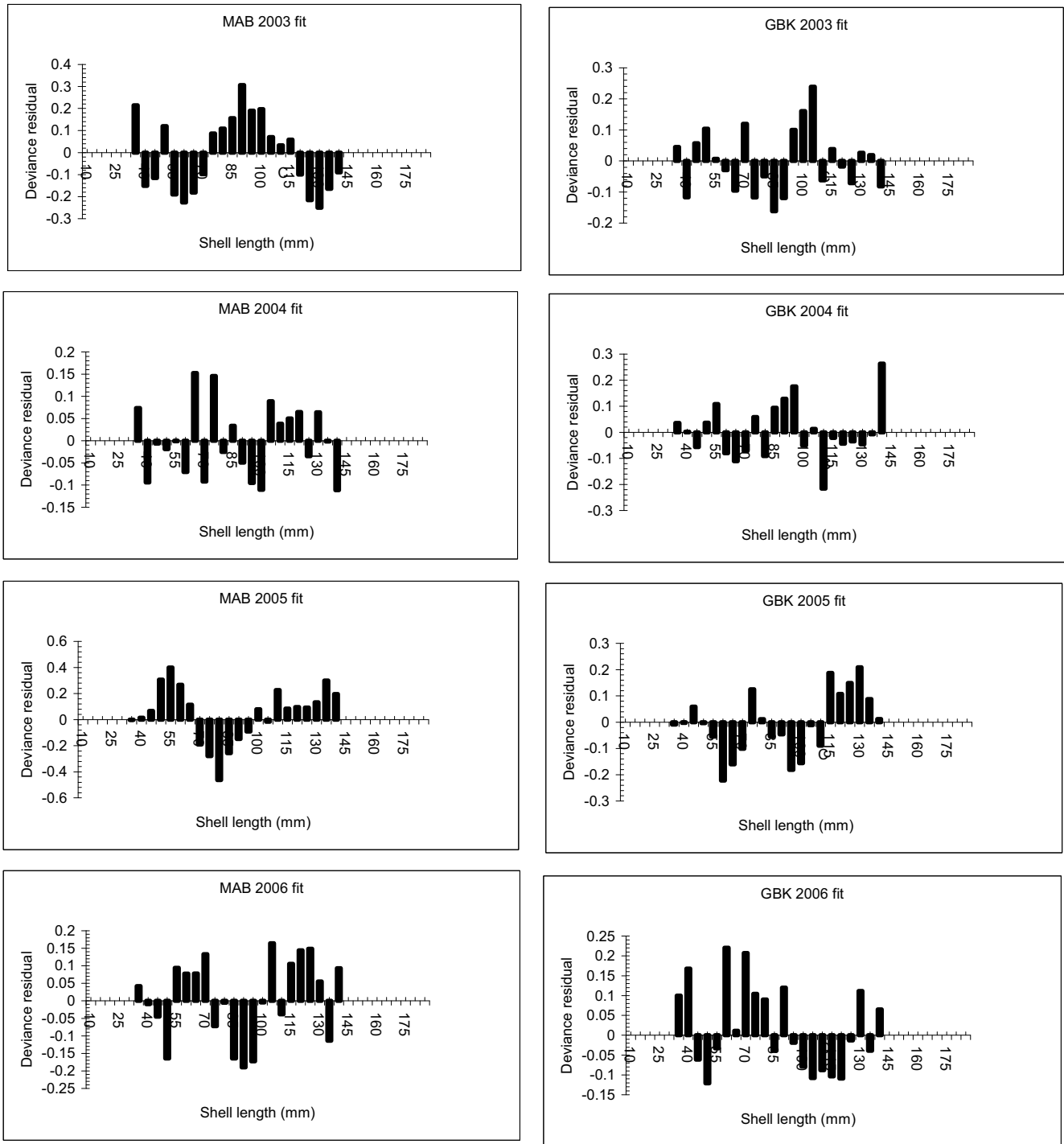
APPENDIX B8 Figure 10. Frequency distributions (bars) and cumulative distributions (solid lines) for sea scallops numbers per tow in dredge survey catches on Georges Bank during 2003-2006.

Goodness of fit plots for dredge survey selectivity models (large camera data)



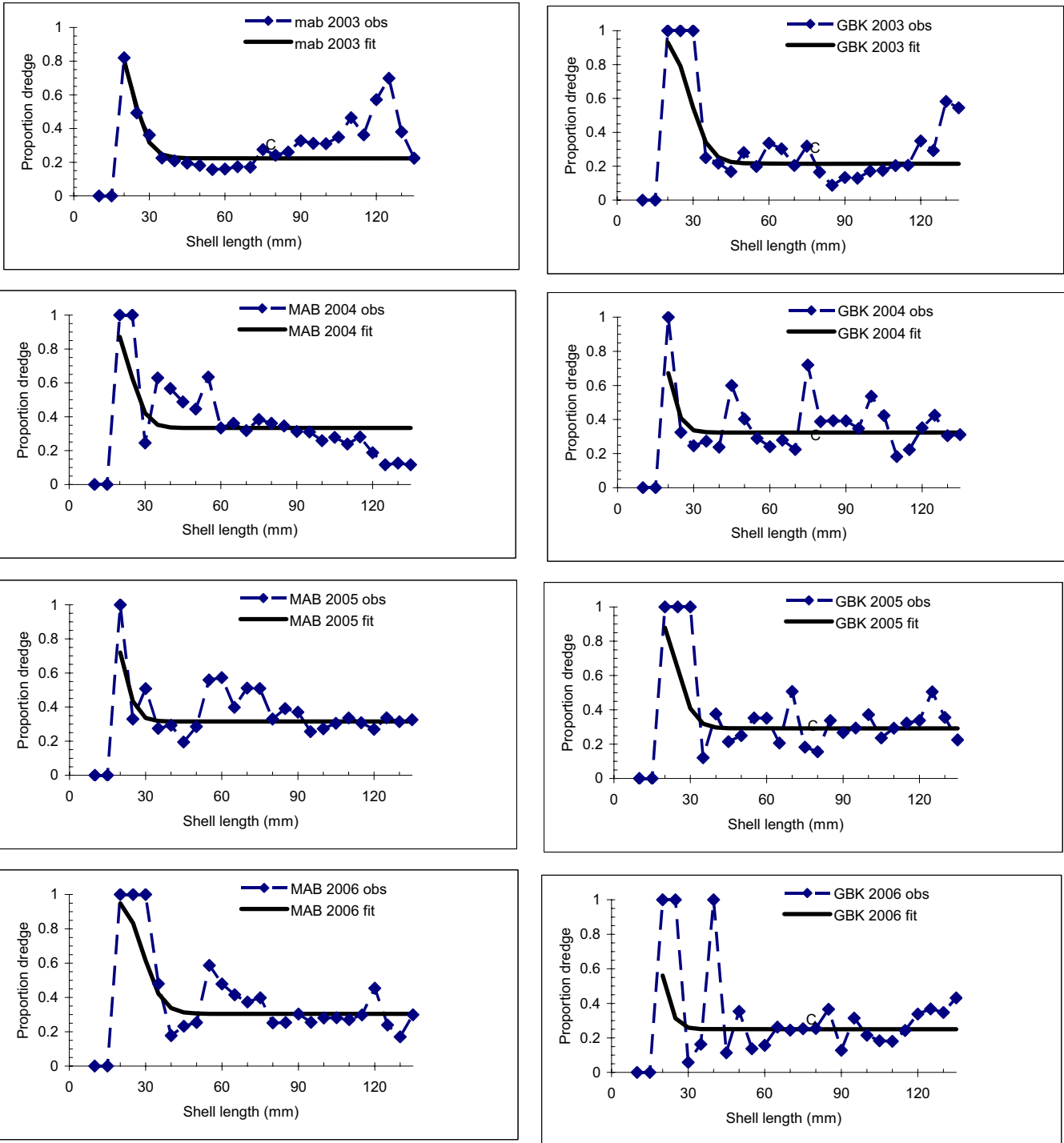
APPENDIX B8 Figure 11. Observed and predicted plots for selectivity estimates from large camera comparisons.

Deviance residual plots for dredge survey selectivity models (large camera data)



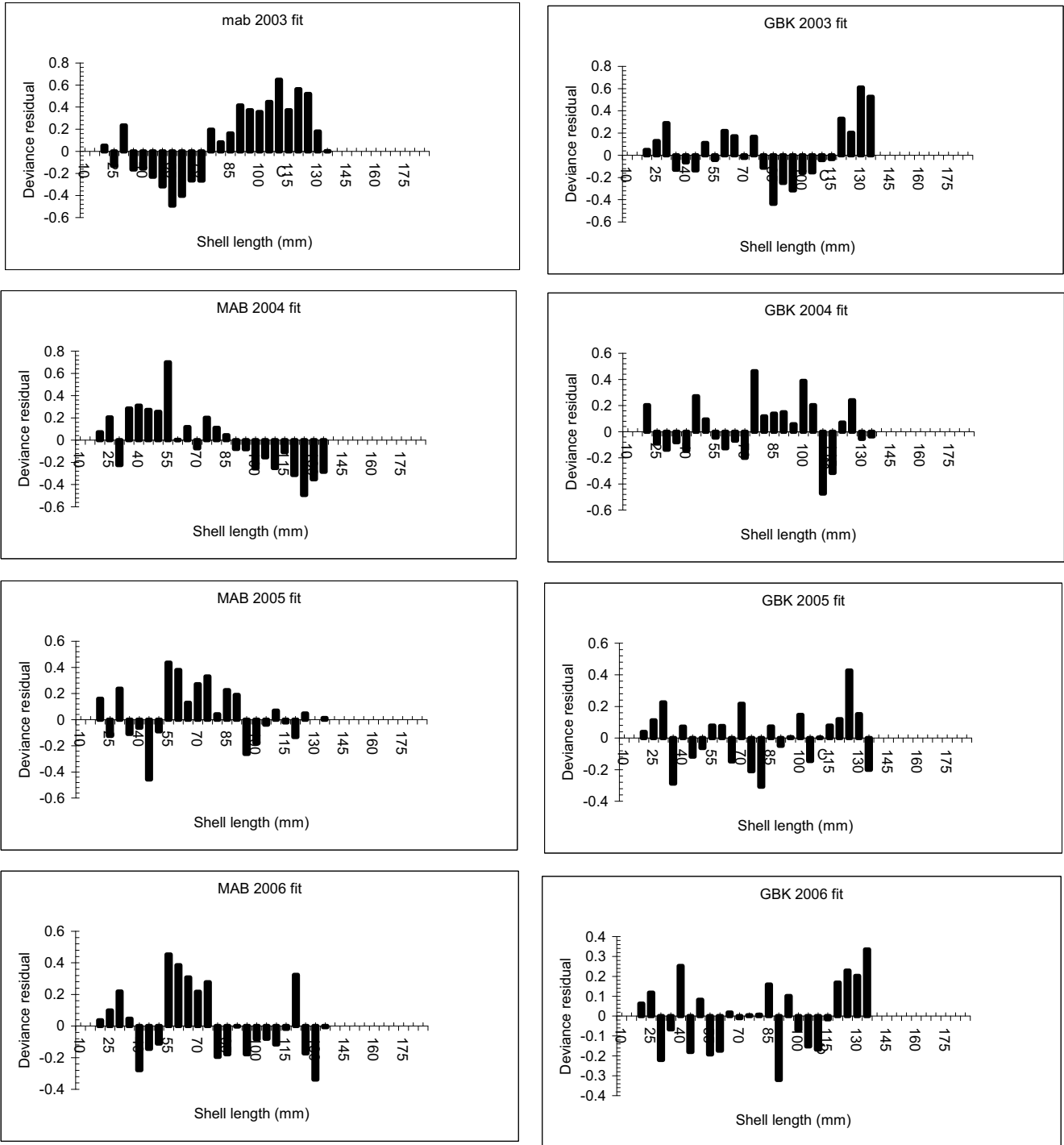
APPENDIX B8 Figure 12. Deviance residuals for selectivity estimates from large camera comparisons.

Goodness of fit plots for dredge survey selectivity models (small camera data)



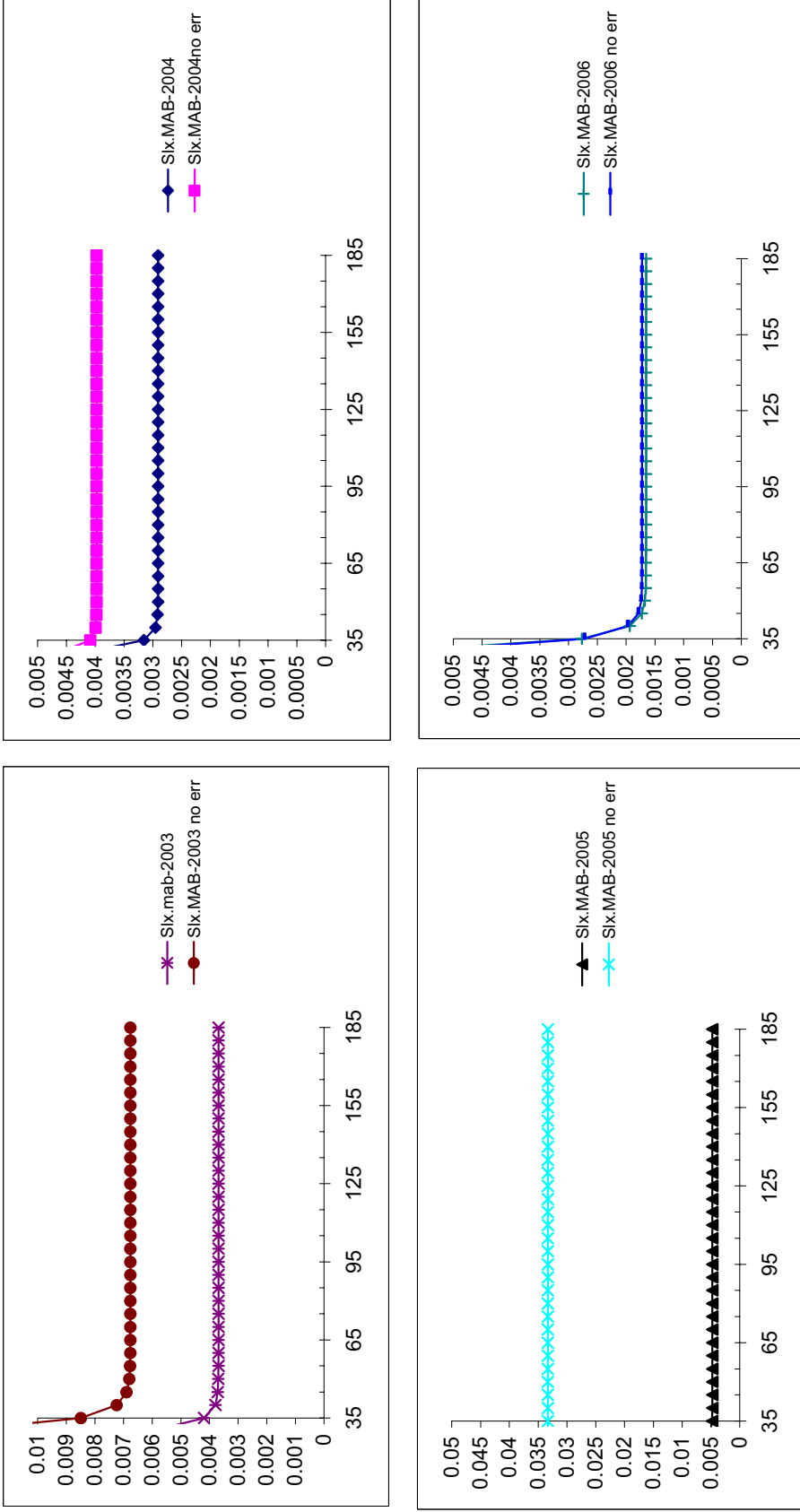
APPENDIX B8 Figure 13. Observed and predicted plots for selectivity estimates from small camera comparisons.

Deviance residual plots for dredge survey selectivity models (small camera data)



APPENDIX B8 Figure 14. Deviance residuals for selectivity estimates from small camera comparisons.

Dredge survey selx based on small camera survey with and without length measurement errors



APPENDIX B8 Figure 15. Comparison of selectivity curves for Mid-Atlantic Bight based on small camera comparisons with and without length measurement errors.

APPENDIX B9: Scallop Dredge Rock Chain Analysis and Calibration

It is believed that the capture of large rocks during standard scallop survey dredge hauls reduces scallop dredge performance. In addition, the interception of large rocks can cause delays to the standard survey, reduce effective strata sampling in marginal habitat (rocky), can be a safety issue and more often than not, result in gear damage. To resolve this issue in the past, an attempt was made to repeat dredge hauls at all random sites that captured large rocks. Because of the uncertainty, the following study was conducted.

Starting in 2001, NEFSC collected annual comparative paired dredge hauls during the standard summer survey. The comparison dredge hauls were between the standard 8 foot wide New Bedford style scallop dredge and another of the same design but rigged with rock excluding chains. The rock chains are laid across and vertically over the dredge mouth opening to create smaller windows in order to exclude rocks but still catch scallops in strata where there is a prevalence of rocks. Paired tows were conducted at random sites within the Great South Channel (GSC) strata set (49, 50, 51, and 52) aboard the R/V Albatross IV (Figure 1). These 4 channel strata were the only strata considered for comparison due to the rugged habitat (Figure 2). The purpose of the study was to identify a statistical difference in terms of catch between the standard dredge and the rock chain dredge configuration and then produce a calibration coefficient to apply to historical catches from the study strata set.

NEFSC conducted 79 paired dredge hauls in the hard habitat site (GSC) for the survey years of 2001, 2002, 2004, 2005, and 2006 (Figure 1). No comparative tows were conducted in 2003. See Table 1 for a year by year breakdown of pairs per sampling year. Presented below are the results of 6 comparisons. The first three tests were conducted using raw scallop catches, while the last three tests had an adjustment to the catch based on longer tow distances. Tow distances were determined by a dredge angle recording device to calculate total bottom time. All catch values were log transformed for each comparison and pairs with zero catch in either both or one were excluded from the analysis. See Table 2 for a listing of catch by dredge type, year, and pair.

The first set of three comparisons (A, B, and 1) were conducted to look at just the raw catch numbers without any tow distance effect. See Figure 3 for a catch distribution by pair. A parametric t-test and a non-parametric Wilcoxon Rank Sum test were conducted for all tests (Table 3).

Test A was comprised of the 39 pairs from 2001 and 2002; test B was the 40 pairs from 2004, 2005, and 2006, while test 1 was all 79 pairs (all years).

The results of test A produced a significant difference for the parametric test ($p=0.006$) between the two dredge types. The non parametric test was the same result ($p=0.005$). The mean difference (0.504) back transformed was 1.655, a bias correction yielded 1.794 and approximate correction was 2.969. The bias correction was performed to compensate for the transformation of normal random variable to a log transformed one. [Calculation $\exp(S.D.^2/2)$]. The approximate correction was calculated by multiplying the bias correction by the mean difference

Test B (40 pairs from 2004 – 2006) was not significant for both parametric ($p=0.126$) and non-parametric ($p=0.102$). If a calibration was needed, the approximate correction for Test B was 1.099 (mean difference = -0.185) and would be a negative adjustment to the rock chain catches, which is opposite of Test A.

Test 1 (79 pairs all years) was not significant for both parametric ($p=0.166$) and non-parametric ($p=.188$). If a calibration was needed, the approximate correction for Test 1 was 1.896 (mean difference = 0.155) and would be a positive adjustment to the non rock chain catches, which is opposite of Test A as well

The second set of comparisons was C, D, and test 6. These comparisons are set up the same way as the three described above, except that the catch data has been standardized by

tow distance. Also, the tow distances were a combination of calculated distances from the inclinometer exercise and 7 regression predicted tow distances referred to as the “combo”. The catches were then standardized to (.95nm/new trackline) ratio before being analyzed. The attempt was to reduce the affect of the tow distances on the mean difference

Test C (39 pairs from 2001 to 2002 with combo tow distance adjustment) yielded a significant result (p=0.006) for the parametric test and for the nonparametric test (p=0.006). Even with the tow distance adjustment to catch, the statistical results were the same as test A. The approximate correction for the calibration from test C. was 2.958 to positive adjust the non rock chain tows. Test D (40 pairs from 2004 to 2006 with combo tow distance adjustment) yielded non-significant parametric results (p=0.109) and non-parametric results (p=0.097). The approximate correction for the calibration for test D. was 1.096 but in a negative adjustment to the rock chain catches. Test 6 (79 pairs from 2001 to 2006 with combo tow distance adjustment) yielded non-significant parametric results (0.189) and non-parametric results (p=0.198). The approximate correction for the 79 pairs was 1.892 to positive adjust the non rock chain catches

The same result occurs whether the tow distance adjustment is included or not. The approximate correction (1.896) for Test 1 (unadjusted catches) is almost the same as approximate correction (1.892) for Test 6 (adjusted catches by tow distance combo). This seems to indicate that a correction factor could be made for historical catches by just using the un-adjusted catches and the approximate calibration from them

A third comparison was conducted that separated the catches by strata groupings rather than years. One test compared strata (49, 50, and 51). The results were significant (p=0.042) for the parametric, but not significant for the non-parametric (p=0.061). The other test was not significant for both parametric and non-parametric.

Because the catch differences seemed to shift by period (2001/2002 vs. 2004-2006) and the direction of the differences between periods, an additional analysis was performed to look at the affect of strata set and year. A generalized linear model approach was chosen to test for year and strata differences using a unified approach. A gamma likelihood was used for the data to avoid the log transformation and incorporate the linear relationship between the mean and variance (Figure 4)¹⁰. In addition, an identity link was used as the catches from the rock chain tows appeared to be linearly related to the catches from the tows made without rock chains (Figure 4). A full factorial model with factors Year.Period (2001, 2002 vs. 2004, 2005, and 2006) and Strata.group (49, 50, 51 vs. 52) was fit to the data (Annex 1). The resulting analysis of deviance indicates that only the coefficient for the non-rock chain catch covariate and terms containing Year.Period were significant (Table 4). Model selection using Akaike’s information criteria resulted agreed with this and the final model was of the form (Table 5):

$$\text{Catch}_{\text{RC}} = \text{Year.Period} + \text{Catch}_{\text{NRC}} + \text{Year.Period}:\text{Catch}_{\text{NRC}}$$

The implications of this result are that for the period 2001/2002, non-rock chain catches would be converted to rock chain catches as:

$$\text{Catch}_{\text{RC}} = 6.755303 + 1.43794 \times \text{Catch}_{\text{NRC}}$$

while for the experiments run in 2004 to 2006:

$$\text{Catch}_{\text{RC}} = (6.755303 - 4.661788) + (1.43794 - 0.4364523) \times \text{Catch}_{\text{NRC}}$$

¹⁰ SPLUS was used to conduct analysis of these data.

These results are not useful for converting non-rock chain catches to rock chain equivalent catches for the time series given the differences found between years. Dredge loading differences between time periods will be investigated from the existing dataset for the next SARC.

Annex 1.

SPLUS commands used in this analysis:

Fit full factorial model:

```
>vics.data.corrected.full.glm<-glm(formula = RC.Test.1 ~ NRC.Test.1 * Year.Period * Strata.group, family = Gamma(link = identity), data = vicsdata.corrected)
```

Analysis of deviance:

```
>anova(vics.data.corrected.full.glm,test="F")
```

Model selection using Akaike Information criteria (AIC):

```
>vics.data.corrected.red.glm<-step.AIC(vics.data.corrected.full.glm)11
```

²Step.AIC is available in the MASS library.

APPENDIX B9 Table 1: Distribution of Pairs Among Years and Strata.

Year	Pairs	Strata 49	Strata 50	Strata 51	Strata 52
2001	21	0	10	3	8
2002	18	1	8	5	4
2004	23	6	5	7	5
2005	3	1	2	0	0
2006	14	0	3	4	7
Total:	79	8	28	19	24

APPENDIX B9 Table 2: Raw Catches by Year

Year	Pair #	RC	NRC	Year	Pair #	RC	NRC
2001	1	1809	1917	2004	40	1391	1408
2001	2	27	8	2004	41	80	30
2001	3	104	19	2004	42	10	30
2001	4	618	159	2004	43	47	54
2001	5	100	13	2004	44	17	81
2001	6	2701	2012	2004	45	503	454
2001	7	117	37	2004	46	32	38
2001	8	1756	1860	2004	47	302	662
2001	9	99	45	2004	48	303	723
2001	10	310	395	2004	49	2	1
2001	11	279	244	2004	50	550	815
2001	12	19	5	2004	51	83	180
2001	13	21	18	2004	52	275	172
2001	14	872	411	2004	53	56	57
2001	15	300	567	2004	54	18	29
2001	16	75	273	2004	55	2	3
2001	17	27	15	2004	56	48	23
2001	18	124	286	2004	57	14	9
2001	19	41	81	2004	58	141	246
2001	20	12	2	2004	59	3191	2923
2001	21	3	5	2004	60	468	78
2002	22	573	346	2004	61	31	10
2002	23	12	96	2004	62	56	110
2002	24	367	41	2005	63	39	275
2002	25	170	45	2005	64	454	670
2002	26	38	7	2005	65	368	180
2002	27	384	437	2006	66	1296	2127
2002	28	219	402	2006	67	361	1065
2002	29	173	96	2006	68	179	218
2002	30	223	53	2006	69	7	6
2002	31	24	250	2006	70	283	267
2002	32	5	2	2006	71	112	380
2002	33	419	108	2006	72	65	49
2002	34	35	19	2006	73	17	15
2002	35	59	20	2006	74	20	40
2002	36	1142	927	2006	75	18	17
2002	37	29	16	2006	76	722	1572
2002	38	1384	306	2006	77	244	154
2002	39	12	2	2006	78	267	486
				2006	79	1389	1968

APPENDIX B9 Table 3. Statistical Results and Calibration Coefficients

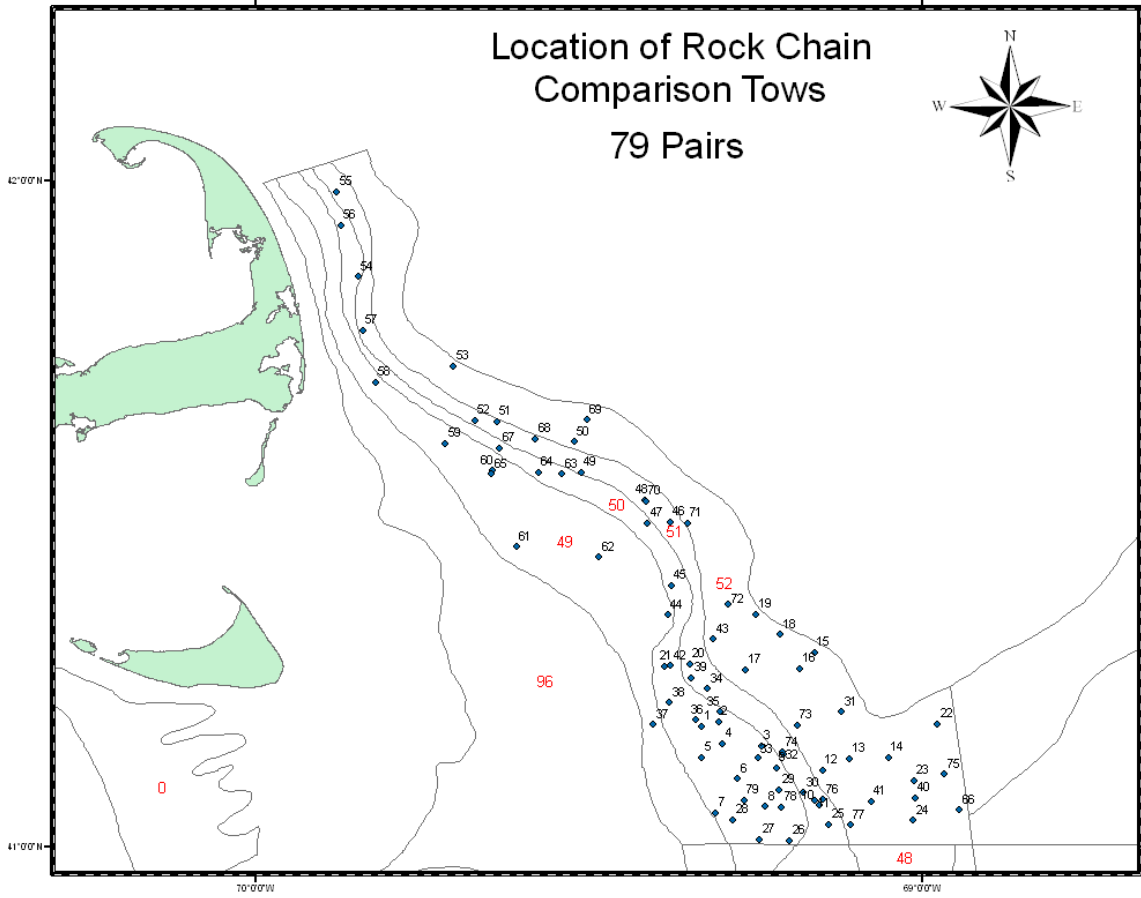
Paired Sample Comparisons										
Test	Mean Dif.	S.D.	S.E.	t stat	df	Sign.	Wilcox.	Calibration EXP(Mean Diff)	Bias Correc.	Approx. Correc.
A	0.504	1.081	0.173	2.91	38	0.006	0.005	1.655	1.794	2.969
B	-0.185	0.748	0.118	-1.57	39	0.126	0.102	0.831	1.323	1.099
1	0.155	0.985	0.111	1.399	78	0.166	0.188	1.168	1.624	1.896
C	0.501	1.080	0.173	2.895	38	0.006	0.006	1.650	1.792	2.958
D	-0.197	0.759	0.120	-1.64	39	0.109	0.097	0.821	1.334	1.096
6	0.148	0.990	0.111	1.325	78	0.189	0.198	1.159	1.632	1.892
39 pairs = 2001 and 2002										
40 pairs = 2004 to 2006										
79 pairs = all years										

APPENDIX B9 Table 4. Analysis of deviance for full factorial model

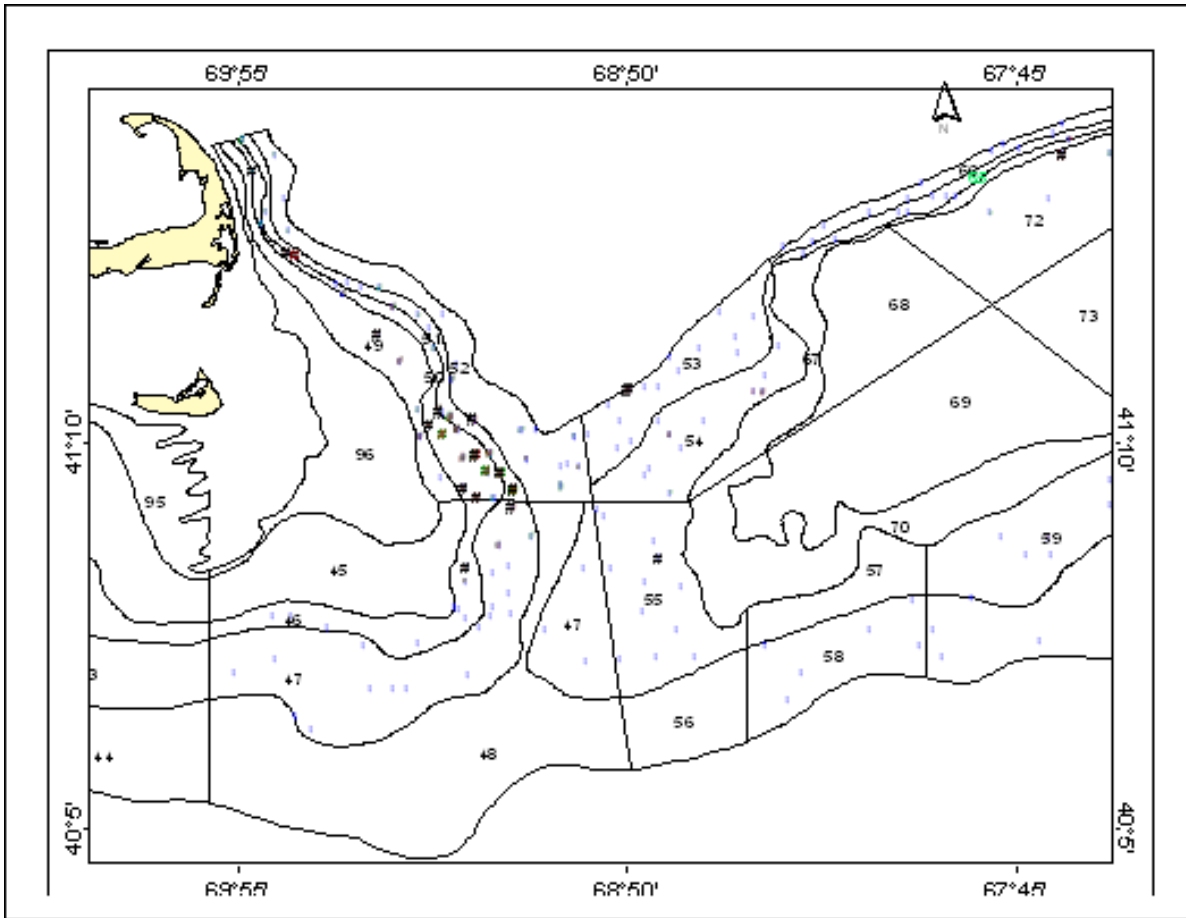
Terms added sequentially	Df	Deviance	Residual Df	Residual Deviance	F- Value	Pr(F)
NULL			78	189.781		
+NRC.Test.1	1	125.792	77	63.988	128.271	0.000
+Year.Period	1	7.081	76	56.908	7.220	0.009
+Strata.group	1	0.033	75	56.875	0.033	0.855
+NRC.Test.1:Year.Period	1	3.382	74	53.493	3.449	0.067
+NRC.Test.1:Strata.group	1	0.428	73	53.065	0.436	0.511
+Year.Period:Strata.group	1	0.014	72	53.051	0.014	0.905
+NRC.Test.1:Year.Period:Strata.group	1	1.928	71	51.123	1.966	0.165

APPENDIX B9 Table 5. Analysis of deviance for reduced model

Terms added sequentially	Df	Deviance	Residual Df	Residual Deviance	F Value	Pr(F)
NULL			78	189.781		
+NRC.Test.1	1	125.792	77	63.988	141.816	0.000
+Year.Period	1	7.081	76	56.908	7.983	0.006
+NRC.Test.1:Year.Period	1	3.411	75	53.497	3.846	0.054

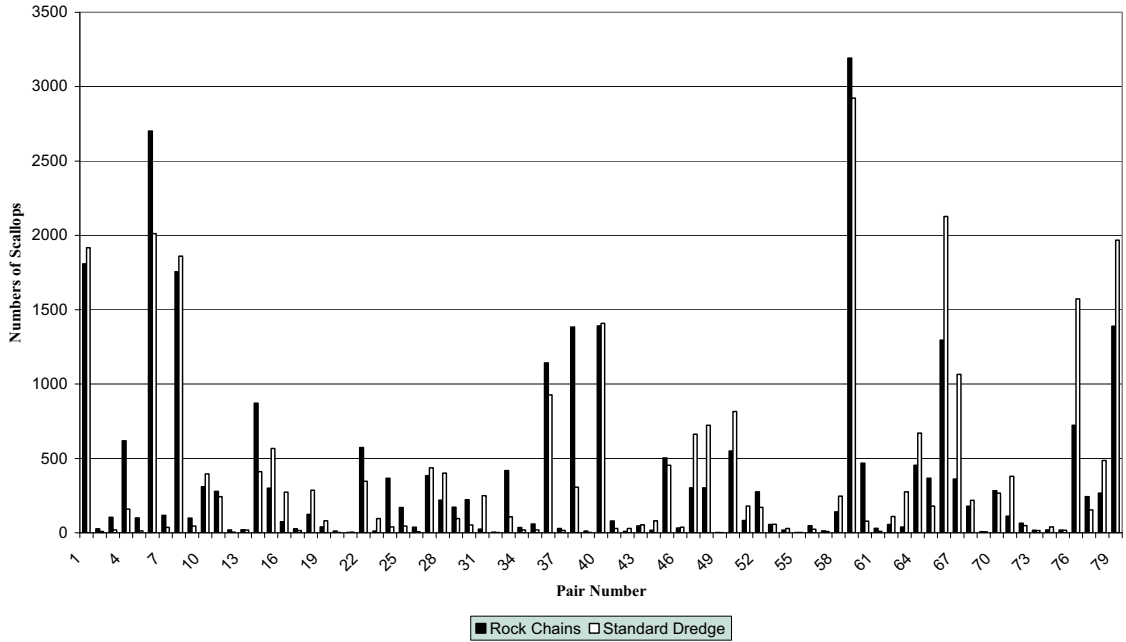


APPENDIX B9 Figure 1. Location of 79 Paired Tows between the Rock Chain and the Standard Dredge

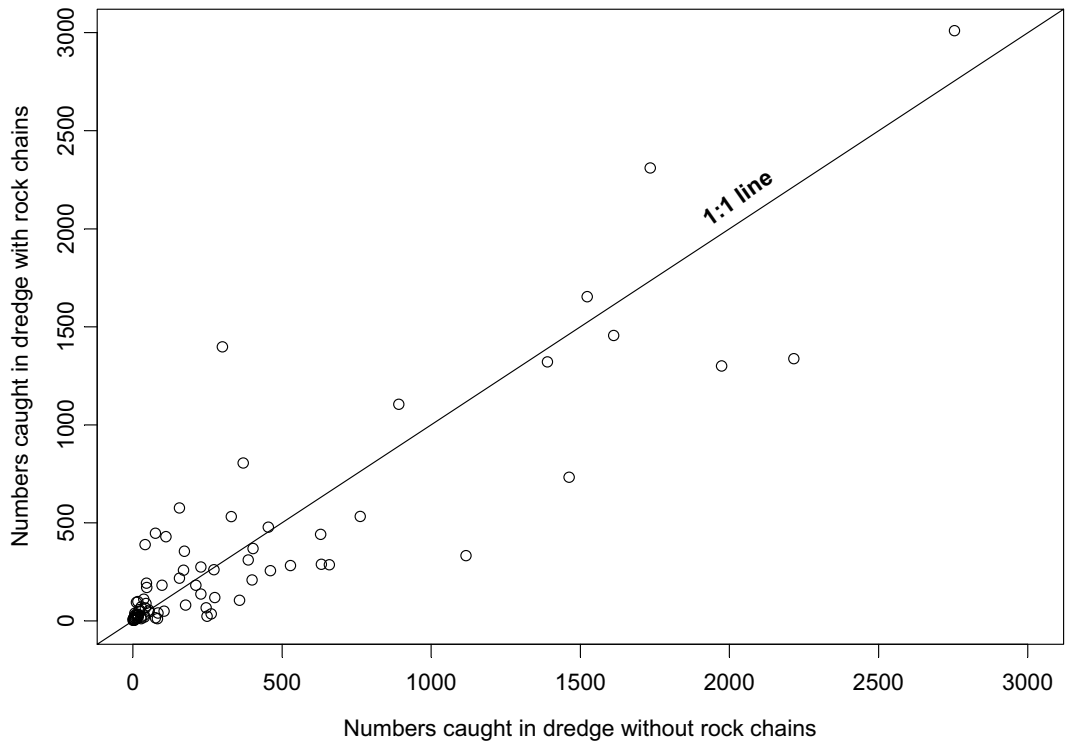


Blue Triangles = dredge hauls with no large rocks

APPENDIX B9 Figure 2. A Sample of the Distribution of Large Rocks on a Typical Scallop Survey



APPENDIX B9 Figure 3. Raw catches over all years for both dredge types.



APPENDIX B9 Figure 4. Catches from dredge with and without rock chains; 1:1 line added for reference.

APPENDIX B10: Technical documentation for the CASA Length Structured Stock Assessment Model¹²

The stock assessment model described here is based on Sullivan et al.'s (1990) CASA model¹³ with a number of additional features. Many aspects are similar to aspects of the Stock Synthesis Model (Methot 2000) although CASA assumes a single set of life history characteristics within a single stock area. CASA is entirely length-based with population dynamic calculations in terms of the number of individuals in each length group during each year. Age is largely irrelevant in model calculations although “effective age” (years since recruitment to the model) calculations have been implemented experimentally. Unlike many other length-based stock assessment approaches, CASA is a dynamic, non-equilibrium model based on a forward simulation approach. CASA incorporates a very wide range of data with parameter estimation based on maximum likelihood. CASA can incorporate prior information and constraints on parameters such as survey catchability in a quasi-Bayesian fashion. The implementation described here was programmed in AD-Model Builder (Otter Research Ltd.).¹⁴

Population dynamics

Time steps in the model are years, which are also used to tabulate catch and other data. Recruitment occurs at the beginning of each time step. If time steps are years, then instantaneous rates have units y^{-1} . The number of years in the model n_y is flexible and can be changed easily (e.g. for retrospective analyses) by making a single change to the input data file. Millimeters are the units for length data. Length-weight relationships should generally convert millimeters to grams. The units for catch and biomass are usually metric tones. Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt).

The definition of length groups (or length “bins”) is a key element in the CASA model and length-structured stock assessment modeling in general. Length bins are identified in CASA by their lower bound. Calculations requiring information about length (e.g. length-weight) use the mid-length ℓ_j of each bin. The user specifies the first length included in model calculations (L_{min}) and the size of length bins (L_{bin}). Based on these specifications, the model determines the number of length bins to be used in modeling as $n_L = 1 + \text{int}[(L_\infty - L_{min})/L_{bin}]$, where L_∞ is maximum asymptotic size based on a von Bertalanffy growth curve supplied by the user and $\text{int}[x]$ is the integer part of x . The last length bin in the model is always a “plus-group” containing individuals L_∞ and larger. Specifications for length data used in tuning the model are entirely separate (see below).

Growth

Although age is not considered, Von Bertalanffy growth models are implicit in several of the configurations of the CASA model. The growth parameter L_∞ is not estimable because it

¹² Documentation last updated on May 11, 2007 as file CASA-Appendix-NC-describe57.doc.

¹³ Original programming in AD-Model Builder by G. Scott Boomer and Patrick J. Sullivan (Cornell University), who bear no responsibility for errors in the current implementation.

¹⁴ AD-Model Builder can be used to calculate variances for any estimated or calculated quantity in a stock assessment model, based on the Hessian matrix with “exact” derivatives and the delta method. Experience with other models (e.g. Overholtz et al., 2004) suggests that variances estimates from AD-Model Builder, which consider the variance of all model parameters, are similar to variances calculated by the common method of bootstrapping survey abundance data.

is used in defining length bins prior to the parameter estimation phase and in determining the largest “plus-group”.¹⁵ The von Bertalanffy growth parameter t_0 is not estimable because it is irrelevant in length-based models that predict growth during a year based on the von Bertalanffy growth parameter K , L_∞ and size at the beginning of the year.

At the beginning of the year, scallops in each size group grow (or not) based on terms in the growth transition matrix $P(b,a)$ which measures the probability that a surviving individual that starts in bin a will grow to bin b by the beginning of the next year (columns index initial size and rows index subsequent size). Growth probabilities do not include any adjustments for mortality. In the CASA model, growth occurs immediately at the beginning of each year and the model assumes that no growth occurs during the year.

Growth probabilities depend on growth increments because:

$$L_2 = L_1 + \iota$$

where L_1 is the starting length, L_2 is length after one year of growth and ι is the growth increment. When growth increments are based on parametric probability distributions (e.g. gamma distributions following Sullivan et al. 1990), probability calculations assume that individuals start at the middle of their original length bin ℓ_a , and then grow to sizes that cover the whole range of each possible subsequent size bin. Thus:

$$P(b,a) = \int_{j=\ell_b-L_{bin}/2}^{\ell_b+L_{bin}/2} P(j|\ell_a) \partial j = \aleph(\ell_b + L_{bin}/2 | \ell_a) - \aleph(\ell_b - L_{bin}/2 | \ell_a)$$

where $P(j|\ell_a)$ is the probability of increment j for an individual originally in bin a (at mid-length ℓ_a). $\aleph(a|\ell_a)$ is the size-specific cumulative distribution function for growth increments. In parametric growth model calculations, cumulative distributions for growth increments are computed by numerical integration based on Simpson’s rule (Press et al., 1990) and a user-specified number of steps per bin. The user can change the number of steps to balance the accuracy of the calculation against time required for growth calculations.

Growth probabilities $P(b,a)$ are calculated in CASA by one of four options. Option 1 is similar to Sullivan et al.’s (1990) approach in that growth probabilities are calculated by numerical integration assuming that increments follow parametric gamma distributions. The gamma distributions for growth increments are initial size dependent and are specified in terms of mean increments and CV’s. Mean increments $\bar{\iota}_a$ are from the von Bertalanffy growth curve:

$$\bar{\iota}_a = (L_\infty - \ell_a)(1 - e^{-K})$$

where $K=e^\chi$ is the von Bertalanffy growth coefficient and χ is an estimable parameter.¹⁶ Under Option 1, CVs are a log-linear function of length:

$$CV_L = e^{\kappa+\lambda L}$$

¹⁵ “Estimable” means a potentially estimable parameter that is specified as a variable that may be estimated in the CASA computer program. In practice, estimability depends on the available data and other factors. It may be necessary to fix certain parameters at assumed fix values or to use constraints of prior distributions for parameters that are difficult to estimate, particularly if data are limited.

¹⁶ Most intrinsically positive or intrinsically negative parameters are estimated in log scale to ensure estimates do not change sign, and to enhance statistical properties of estimates.

where κ and γ are estimable parameters. Sullivan et al. 1990 assumed constant CV's for growth. This implementation of the CASA model includes the special case of constant CV's when $\lambda=0$.

Option 2 is nonparametric and constructs a transition matrix directly from size-specific annual growth data (i.e. data records consisting of starting length, length after one year and number of observations). Under Option 2:

$$P(b,a) = \frac{n(b|a)}{\sum_{j=a}^{n_L} n(j|a)}$$

where $n(b|a)$ is the number of individuals that started at size a and grew to size b after one year.

Under option 3, mean increments are from the von Bertalanffy growth curve as in option 1, but with length-specific CVs (and other model parameters) estimated in the model based on growth increments and other data (see below for goodness of fit calculations). Under option 3, the von Bertalanffy growth parameter K , which describes mean growth, and parameters for variance in growth (κ and γ) are estimable. Option 4 uses a constant, user-specified transition matrix provided as data to the model.

Growth calculations based on assumed gamma distributions (Sullivan et al. 1990) might be unrealistic for some species because the gamma distribution predicts growth increments of zero to infinity. Therefore, with options 1-3, the user may specify minimum and maximum growth increments for each size. Probabilities from truncated gamma distributions for growth increments between the minimum and maximum values are normalized to sum to one before use in population dynamics calculations. Size bins outside those specified are ignored in all model calculations.

Abundance, recruitment and mortality

Population abundance in each length bin during the first year of the model is:

$$N_{1,L} = N_1 \pi_{1,L}$$

where L is the size bin, and $\pi_{1,L}$ is the initial population length composition expressed as proportions so that $\sum_{L=1}^{n_L} \pi_L = 1$. $N_1 = e^\eta$ is total abundance at the beginning of the first modeled year and η is an estimable parameter. It is not necessary to estimate recruitment in the first year because recruitment is implicit in the product of N_1 and π_L . The current implementation of CASA takes the initial population length composition as data supplied by the user.

Abundance at length in years after the first is calculated:

$$\vec{N}_{y+1} = P(\vec{N}_y \circ \vec{S}_y) + \vec{R}_{y+1}$$

where \vec{N}_y is a vector (length n_L) of abundance in each length bin during year y , P is the matrix ($n_L \times n_L$) of growth probabilities $P(b,a)$, \vec{S}_y is a vector of length-specific survival fractions for year y , \circ is for the element-wise product, and \vec{R}_y is a vector holding length-specific abundance of new recruits at the beginning of year y .

Survival fractions are:

$$S_{y,L} = e^{-Z_{y,L}} = e^{-(M_{y,L} + F_{y,L} + I_{y,L})}$$

where $Z_{y,L}$ is the total instantaneous mortality rate and $M_{y,L}$ is the instantaneous rate for natural mortality (see below). Length-specific fishing mortality rates are $F_{y,L} = F_y s_{y,L}$ where $s_{y,L}$ is the size-specific selectivity¹⁷ for fishing in year y (scaled to a maximum of one at fully recruited size groups), F_y is the fishing mortality rate on fully selected individuals. Fully recruited fishing mortality rates are $F_y = e^{\phi + \delta_y}$ where ϕ is an estimable parameter for the log of the geometric mean of fishing mortality in all years, and δ_y is an estimable “dev” parameter.¹⁸ The instantaneous rate for “incidental” mortality ($I_{y,L}$) accounts for mortality due to contact with the fishing gear that does not result in any catch on deck (see below).¹⁹ The degree of variability in dev parameters for fishing mortality, natural mortality and for other variables can be controlled using variance constraints described below.

Natural mortality rates $M_{y,L} = u_L e^{\zeta + \xi_y}$ may vary from year to year and by length. Variability among length groups is based on a user-specified vector \bar{u} that describes the relative natural mortality rate for each length group in the model. The user supplies a value for each length group which the model rescales so that the average of all of the values is one (i.e. \bar{u} is set by the user and cannot be estimated). Temporal variability in natural mortality rates are modeled in the same manner as temporal variability in fishing mortality. In particular, ζ is an estimable parameter measuring the mean log natural mortality rate during all years and ξ_y is an estimable year-specific dev parameter. Several approaches are available for estimating natural mortality parameters (i.e. natural mortality covariates and surveys that measure numbers of dead individuals, see below).

Incidental mortality $I_{y,L} = F_y u_L i$ is the product of fully recruited fishing mortality (F_y , a proxy for effective fishing effort, although nominal fishing effort might be a better predictor of incidental mortality), relative incidental mortality at length (u_L) and a scaling parameter i , which is supplied by the user and not estimable in the model. Mortality at length is supplied by the user as a vector (\bar{u}) containing a value for each length group in the model. The model rescales the relative mortality vector so that the mean of the series is one.

Given abundance in each length group, natural mortality, and fishing mortality, predicted fishery catch-at-length in numbers is:

$$C_{y,L} = \frac{F_{y,L} (1 - e^{-Z_{y,L}}) N_{L,y}}{Z_{y,L}}$$

Total catch number during each year is $C_y = \sum_{j=1}^{n_L} C_{y,L}$. Catch data (in weight, numbers or as length composition data) are understood to include landings (L_y) and discards (d_y) but to exclude losses to incidental mortality (i.e. $C_y = L_y + d_y$).

Discard data are supplied by the user in the form of discarded biomass in each year or a discard rate for each year (or a combination of biomass levels and rates). It is important to remember that discard rates in CASA are defined the ratio of discards to landings (d/L). The user may also specify a mortal discard fraction between zero and one if not discards are expected to die. If the discard fraction is less than one, then the discarded biomass and

¹⁷ In this context, “selectivity” describes the combined effects of all factors that affect length composition of catch or landings. These factors include gear selectivity, spatial overlap of the fishery and population, size-specific targeting, size-specific discard, etc.

¹⁸ Dev parameters are a special data type for estimable parameters in AD-Model Builder. Each set of dev parameters (e.g. for all recruitments in the model) is constrained to sum to zero. Because of the constraint, the sums $\phi + \delta_y$ involving $n_y + 1$ terms amount to only n_y parameters.

¹⁹ . See the section on per recruit modeling below for formulas used to relate catch, landings and incidental mortality.

discard rates in the model are reduced correspondingly. See the section on per recruit modeling below for formulas used to relate catch, landings and incidental mortality.

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

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

where ρ is an estimable parameter that measures the geometric mean recruitment and the γ_y are estimable dev parameters that measure interannual variability in recruitment. As with natural mortality devs, a variance constraint can be used to help estimate recruitment deviations (see below).

Proportions of recruits in each length group are calculated based on a beta distribution $B(w,r)$ over the first n_r length bins that is constrained to be unimodal.²⁰ Proportions of new recruits in each size group are the same from year to year. Beta distribution coefficients must be larger than one for the shape of the distribution to be unimodal. Therefore, $w=1+e^\omega$ and $r=1+e^\rho$, where ω and ρ are estimable parameters. It is probably better to calculate the parameters in this manner than as bounded parameters because there is likely to be less distortion of the Hessian for w and r values close to one and parameter estimation is likely to be more efficient.

Surplus production during each year of the model can be computed approximately from biomass and catch estimates (Jacobson et al., 2002):

$$P_t = B_{t+1} - B_t + \delta C_t$$

where δ is a correction factor that adjusts catch weight to population weight at the beginning of the next year by accounting for mortality and growth. The adjustment factor depends strongly on the rates for growth and natural mortality and only weakly on the natural mortality rate. In the absence of a direct estimate, useful calculations can be carried out assuming $\delta=1$. In future versions of the CASA model, surplus production will be more accurately calculated by projecting populations at the beginning of the year forward one year assuming only natural mortality. ***[NOTE: surplus production calculations are being updated and were not available for the 2007 sea scallop stock assessment.]***

Population summary variables

Population summary variables described above are calculated for the entire stock (all length groups) and two user specified ranges of length bins. One set of bins is typically used for “stock” statistics that may, for example, exclude the smallest size groups. The other set of bins is typically used for exploitable sizes that may be vulnerable to the fishery. Several statistics are calculated for the beginning (January 1) and middle (July 1) of the year.

Estimated total abundance at the beginning of the year is the sum of abundance at length $N_{y,L}$ at the beginning of the year. Average annual abundance is:

$$\bar{N}_{y,L} = N_{y,L} \frac{1 - e^{-Z_{y,L}}}{Z_{y,L}}$$

²⁰ Standard beta distributions used to describe recruit size distributions and in priors are often constrained to be unimodal in the CASA model. Beta distributions $B(w,r)$ with mean $\mu = w/(w+r)$ and variance

$\sigma^2 = wr / [(w+r)^2(w+r+1)]$ are unimodal when $w > 1$ and $r > 1$. See http://en.wikipedia.org/wiki/Beta_distribution for more information.

CASA assumes that weight-at-length relationships for the stock (on January 1) and the fishery may differ and that mean fishery weight-at-length may change interannually. For example, total stock biomass is:

$$B_y = \sum_{L=1}^{n_L} N_{y,L} w_L$$

where w_L is weight at length for the population on January 1 computed at the midpoint of each length bin using the length-weight relationship for the fishery specified by the user. Total catch weight is:

$$W_y = (1 + \omega_y) \sum_{L=1}^{n_L} C_{y,L} w'_L$$

where w'_L is weight at length in the fishery and ω_y is an annual anomaly input by the user to describe changes in fishery length weight that may occur from year to year due, for example, to changes in seasonal distribution of fishing. Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt).

F_y estimates for two years are comparable only if the fishery selectivity in the model was the same in both years. A set of simpler exploitation indices may be more useful when fishery selectivity changes over time. For example:

$$U_y = \frac{C_y}{\sum_{j=x}^{n_L} \tilde{N}_{y,L}}$$

where x is a user-specified length bin (e.g. just below the first bin that is fully selected during all fishery selectivity periods) and the term \tilde{N} is predicted abundance at the middle (July 1st) of the year. Similar statistics are calculated based on stock and catch weights and for January 1st as well as July 1st. Exploitation indices from different years with different selectivity patterns may be readily comparable if x is chosen carefully.

Spawner abundance in each year is (T_y) is computed:

$$T_y = \sum_{L=1}^{n_L} N_{y,L} e^{-\tau z_y} g_L$$

where $0 \leq \tau \leq 1$ is the fraction of the year elapsed before spawning occurs (supplied by the user). Maturity at length (g_L) is from an ascending logistic curve:

$$g_L = \frac{1}{1 + e^{-a-bL}}$$

with parameters a and b supplied by the user.

Spawner biomass or egg production (S_y) in each year is computed:

$$S_y = \sum_{L=1}^{n_L} T_{y,L} x_L$$

where:

$$x_L = cL^v$$

Using parameters (c and v) for fecundity- or body weight at size supplied by the user.

Fishery and survey selectivity

The current implementation of CASA includes six options for calculating fishery and survey selectivity patterns. Fishery selectivity may differ among “fishery periods” defined by the user. Selectivity patterns that depend on length are calculated using lengths at the midpoint of each bin (ℓ). After initial calculations (described below), selectivity curves are rescaled to a maximum value of one.

Option 1 is a flat with $s_L=1$ for all length bins. Option 2 is an ascending logistic curve:

$$s_{y,\ell} = \frac{1}{1 + e^{A_Y - B_Y \ell}}$$

Option 3 is an ascending logistic curve with a minimum asymptotic minimum size for small size bins on the left.

$$s_{y,\ell} = \left(\frac{1}{1 + e^{A_Y - B_Y \ell}} \right) (1 - D_y) + D_y$$

Option 4 is a descending logistic curve:

$$s_{y,\ell} = 1 - \frac{1}{1 + e^{A_Y - B_Y \ell}}$$

Option 5 is a descending logistic curve with a minimum asymptotic minimum size for large size bins on the right:

$$s_{y,\ell} = \left(1 - \frac{1}{1 + e^{A_Y - B_Y \ell}} \right) (1 - D_y) + D_y$$

Option 6 is a double logistic curve used to represent “domed-shape” selectivity patterns with highest selectivity on intermediate size groups:

$$s_{y,\ell} = \left(\frac{1}{1 + e^{A_Y - B_Y \ell}} \right) \left(1 - \frac{1}{1 + e^{D_Y - G_Y \ell}} \right)$$

The coefficients for selectivity curves A_Y , B_Y , D_Y and G_Y carry subscripts for time because they may vary between fishery selectivity periods defined by the user. All options are parameterized so that the coefficients A_Y , B_Y , D_Y and G_Y are positive. Under options 3 and 5, D_y is a proportion that must lie between 0 and 1. All selectivity curves are rescaled to a maximum value of one before used in further calculations.

Depending on the option, estimable selectivity parameters may include α , β , δ and γ . For options 2, 4 and 6, $A_Y = e^{\alpha_Y}$, $B_Y = e^{\beta_Y}$, $D_Y = e^{\delta_Y}$ and $G_Y = e^{\gamma_Y}$. Options 3 and 5 use the same conventions for A_Y and B_Y , however, the coefficient D_Y is a proportion estimated as a logit-transformed parameter (i.e. $\delta_Y = \ln[D_Y/(1-D_Y)]$) so that:

$$D_Y = \frac{e^{\delta_Y}}{1 + e^{\delta_Y}}$$

The user can choose, independently of all other parameters, to either estimate each fishery selectivity parameter or to keep it at its initial value. Under Option 2, for example, the user can estimate the intercept α_Y , while keep the slope β_Y at its initial value.

Per recruit modeling

A complete per recruit output table is generated in all model runs that can be used for evaluating the shape of YPR and SBR curves, including the existence of particular reference points. The output table summarizes a wide range of per recruit information in terms of fully recruited fishing mortality F and a number of exploitation indices (U) specified by the user. Per recruit calculations in CASA use the same population model and code as all other model

calculations under conditions identical to the last year in the model. It is a standard length-based approach except that discard and incidental mortality are accommodated in all calculations.

In per recruit calculations, fishing mortality rates and associated yield estimates are understood to include landings and discard mortality, but to exclude incidental mortality. Thus, landings per recruit is:

$$L = \frac{C}{(1 + \Delta)}$$

where C is total catch (yield) per recruit and Δ is the ratio of discards D to landings in the last year of the model. Discards per recruit are calculated:

$$D = \Delta L$$

Losses due to incidental mortality (G) are calculated:

$$G = \frac{I(1 - e^{-Z})B}{Z}$$

$$= IK$$

where $I = Fu$ is the incidental mortality rate, u is a user-specified multiplier (see above) and B is stock biomass per recruit. Note that $C = FK$ so that $K = C/F$. Then,

$$G = \frac{FuC}{F}$$

$$G = uC$$

In addition to generating a per recruit output table, the model will estimate key ($F_{\%SBR}$, F_{max} and $F_{0.1}$) per recruit model reference points as parameters. For example,

$$F_{\%SBR} = e^{\theta_j}$$

where $F_{\%SBR}$ is the fishing mortality reference point that provides a user specified percentage of maximum SBR. θ_j is the model parameter for the j^{th} reference point.

Per recruit reference points are time consuming to estimate and it is usually better to estimate them after other more important population dynamics parameters are estimated. Phase of estimation can be controlled individually for %SBR, F_{max} and $F_{0.1}$ so that per recruit calculations can be delayed as long as possible. If the phase is set to zero or a negative integer, then the reference point will not be estimated. As described below, estimation of F_{max} always entails an additional phase of estimation. For example, if the phase specified for F_{max} is 2, then the parameter will be estimated initially in phase 2 and finalized the last phase (phase ≥ 3). This is done so that the estimate from phase 2 can be used as an initial value in a slightly different goodness of fit calculation during the final phase.

Per recruit reference points should have no effect on other model estimates. Residuals (calculated – target) for %SBR, $F_{0.1}$ and F_{max} reference points should always be very close to zero. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with reference point calculations turned on and then off to see if biomass and fishing mortality estimates change.*

The user specifies the number of estimates required and the target %SBR level for each. For example, the target levels for four %SBR reference points might be 0.2, 0.3, 0.4 and 0.5

to estimate $F_{20\%}$, $F_{30\%}$, $F_{40\%}$ and $F_{50\%}$. The user has the option of estimating F_{max} and/or $F_{0.1}$ as model parameters also but it is not necessary to supply target values.

Tuning and goodness of fit

There are two steps in calculating the negative log likelihood (NLL) used to measure how well the model fits each type of data. The first step is to calculate the predicted values for data. The second step is to calculate the NLL of the data given the predicted value. The overall goodness of fit measure for the model is the weighted sum of NLL values for each type of data and each constraint:

$$\Lambda = \sum \lambda_j L_j$$

where λ_j is a weighting factor for data set j (usually $\lambda_j=1$, see below), and L_j is the NLL for the data set. The NLL for a particular data is itself usually a weighted sum:

$$L_j = \sum_{i=1}^{n_j} \psi_{j,i} L_{j,i}$$

where n_j is the number of observations, $\psi_{j,i}$ is an observation-specific weight (usually $\psi_{j,i}=1$, see below), and $L_{j,i}$ is the NLL for a single observation.

Maximum likelihood approaches reduce the need to specify *ad-hoc* weighting factors (λ and ϕ) for data sets or single observations, because weights can often be taken from the data (e.g. using CVs routinely calculated for bottom trawl survey abundance indices) or estimated internally along with other parameters. In addition, robust maximum likelihood approaches (see below) may be preferable to simply down-weighting an observation or data set. However, despite subjectivity and theoretical arguments against use of *ad-hoc* weights, it is often useful in practical work to manipulate weighting factors, if only for sensitivity analysis or to turn an observation off entirely. Observation specific weighting factors are available for most types of data in the CASA model.

Missing data

Availability of data is an important consideration in deciding how to structure a stock assessment model. The possibility of obtaining reliable estimates will depend on the availability of sufficient data. However, NLL calculations and the general structure of the CASA model are such that missing data can usually be accommodated automatically. With the exception of catch data (which must be supplied for each year, even if catch was zero), the model calculates that NLL for each datum that is available. No NLL calculations are made for data that are not available and missing data do not generally hinder model calculations.

Likelihood kernels

Log likelihood calculations in the current implementation of the CASA model use log likelihood “kernels” or “concentrated likelihoods” that omit constants. The constants can be omitted because they do not affect slope of the NLL surface, final point estimates for parameters or asymptotic variance estimates.

For data with normally distributed measurement errors, the complete NLL for one observation is:

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

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

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

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

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

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

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

The second approach is equivalent but used when the weights for each observation (w_i) may differ:

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

In the latter case, the maximum likelihood estimator:

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

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

If data x have lognormal measurement errors, then $\ln(x)$ is normal and L is calculated as above. In some cases it is necessary to correct for bias in converting arithmetic scale means to log scale means (and *vice-versa*) because $\bar{x} = e^{\bar{\chi} + \sigma^2/2}$ where $\chi = \ln(x)$. It is often convenient to convert arithmetic scale CVs for lognormal variables to log scale standard deviations using $\sigma = \sqrt{\ln(1 + CV^2)}$.

For data with multinomial measurement errors, the likelihood kernel is:

$$L = n \sum_{i=1}^n p_i \ln(\theta_i) - K$$

where n is the known or assumed number of observations (the “effective” sample size), p_i is the proportion of observations in bin i , and θ_i is the model’s estimate of the probability of an observation in the bin. The constant K is used for convenience to make L easier to interpret. It measures the lowest value of L that could be achieved if the data fit matched the model’s expectations exactly:

$$K = n \sum_{i=1}^n p_i \ln(p_i)$$

For data x that have measurement errors with expected values of zero from a gamma distribution:

$$L = (\gamma - 1) \ln \left(\frac{x}{\beta} \right) - \frac{x}{\beta} - \ln(\beta)$$

where $\beta > 0$ and $\gamma > 0$ are gamma distribution parameters in the model. For data that lie between zero and one with measurement errors from a beta distribution:

$$L = (p-1)\ln(x) + (q-1)\ln(1-x)$$

where $p > 0$ and $q > 0$ are parameters in the model.

In CASA model calculations, distributions are usually described in terms of the mean and CV. Normal, gamma and beta distribution parameters can be calculated mean and CV by the method of moments.²¹ Means, CV's and distributional parameters may, depending on the situation, be estimated in the model or specified by the user.

Robust methods

Goodness of fit for survey data may be calculated using a “robust” maximum likelihood method instead of the standard method that assumes lognormal measurement errors. The robust method may be useful when survey data are noisy or include outliers.

Robust likelihood calculations in CASA assume that measurement errors are from a Student's t distribution with user-specified degrees of freedom d_f . Degrees of freedom are specified independently for each observation so that robust calculations can be carried out for as many (or as few) cases as required. The t distribution is similar to the normal distribution for $d_f \geq 30$. As d_f is reduced, the tails of the t distribution become fatter so that outliers have higher probability and less effect on model estimates. If $d_f = 0$, then measurement errors are assumed in the model to be normally distributed.

The first step in robust NLL calculations is to standardize the measurement error residual $t = (x - \bar{x})/\sigma$ based on the mean and standard deviation. Then:

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

Catch weight data

Catch data (landings plus discards) are assumed to have normally distributed measurement errors with a user specified CV. The standard deviation for catch weight in a particular year is $\sigma_Y = \kappa \hat{C}_Y$, where “^” indicates that the variable is a model estimate and errors in catch are assumed to be normally distributed. The standardized residual used in computing NLL for a single catch observation and in making residual plots is $r_Y = (C_Y - \hat{C}_Y)/\sigma_Y$.

Specification of landings, discards, catch

Landings, discard and catch data are in units of weight and are for a single or “composite” fishery in the current version of the CASA model. The estimated fishery selectivity is assumed to apply to the discards so that, in effect, the length composition of catch, landings and discards are the same.

²¹ Parameters for standard beta distributions $B(w,r)$ with mean $\mu = w/(w+r)$ and variance

$\sigma^2 = wr/[(w+r)^2(w+r+1)]$ are calculated from user-specified means and variances by the method of moments. In particular, $w = \mu[\mu(1-\mu)/\sigma^2 - 1]$ and $r = (1-\mu)[\mu(1-\mu)/\sigma^2 - 1]$. Not all combinations

of μ and σ^2 are feasible. In general, a beta distribution exists for combinations of μ and σ^2 if $0 < \mu < 1$ and $0 < \sigma^2 < \mu(1-\mu)$. Thus, for a user-specified mean μ between zero and one, the largest feasible variance is $\sigma^2 < \mu(1-\mu)$. These conditions are used in the model to check user-specified values for μ and σ^2 . See

http://en.wikipedia.org/wiki/Beta_distribution for more information.

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

$$D_t = L_t \Delta_t$$

where $\Delta_t = D_t/L_t$ is the ratio of discard and landings (a.k.a. d/K ratios) for each year. If $d_t < 0$ then the data are treated as discard in units of weight:

$$D_t = \text{abs}(d_t)$$

In either case, total catch is the sum of discards and landed catch ($C_t = L_t + D_t$). It is possible to use discards in weight $d_t < 0$ for some years and discard as proportions $d_t > 0$ for other years in the same model run.

If catches are estimated (see below) so that the estimated catch \hat{C}_t does not necessarily equal observed landings plus discard, then estimated landings are computed:

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

Estimated discards are:

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

Note that $\hat{C}_t = \hat{L}_t + \hat{D}_t$ as would be expected.

Fishery length composition data

Data describing numbers or relative numbers of individuals at length in catch data (fishery catch-at-length) are modeled as multinomial proportions $c_{y,L}$:

$$c_{y,L} = \frac{C_{y,L}}{\sum_{j=1}^{n_L} C_{y,j}}$$

The NLL for the observed proportions in each year is computed based on the kernel for the multinomial distribution, the model's estimate of proportional catch-at-length (\hat{c}_y) and an estimate of effective sample size ${}^c N_y$ supplied by the user. Care is required in specifying effective sample sizes, because catch-at-length data typically carry substantially less information than would be expected based on the number of individuals measured. Typical conventions make ${}^c N_y \leq 200$ (Fournier and Archibald, 1982) or set ${}^c N_y$ equal to the number of trips or tows sampled (Pennington et al., 2002). Effective sample sizes are sometimes chosen based on goodness of fits in preliminary model runs (Methot, 2000; Butler et al., 2003).

Survey index data

In CASA model calculations, "survey indices" are data from any source that reflect relative proportional changes in an underlying population state variable. In the current version, surveys may measure stock abundance at a particular point in time (e.g. when a survey was carried out), stock biomass at a particular point in time, or numbers of animals that dies of natural mortality during a user-specified period. For example, the first option is useful for bottom trawl surveys that record numbers of individuals, the second option is useful for bottom trawl surveys that record total weight, and the third option is useful for survey data that track trends in numbers of animals that died due to natural mortality (e.g. survey data for sea scallop "clappers"). Survey data that measure trends in numbers dead due to natural mortality can be useful in modeling time trends in natural mortality. In principle, the model will estimate model natural mortality and other parameters so that predicted numbers dead and the index data match in either relative or absolute terms.

In the current implementation of the CASA model, survey indices are assumed to be linear indices of abundance or biomass so that changes in the index (apart from measurement error) are assumed due to proportional changes in the population. Nonlinear commercial catch rate data are handled separately (see below). Survey index and fishery length composition data are handled separately from trend data (see below). Survey data may or may not have corresponding length composition information.

In general, survey index data give one number that summarizes some aspect of the population over a wide range of length bins. Selectivity parameters measure the relative contribution of each length bin to the index. Options and procedures for estimating survey selectivity patterns are the same as for fishery selectivity patterns, but survey selectivity patterns are not allowed to change over time.

NLL calculations for survey indices use predicted values calculated:

$$\hat{I}_{k,y} = q_k A_{k,y}$$

where q_k is a scaling factor for survey index k , and $A_{k,y}$ is stock available to the survey. Scaling factors are calculated $q_s = e^{\varpi_s}$ where ϖ_s is estimable and survey-specific.

Available stock for surveys measuring trends in abundance or biomass is calculated:

$$A_{k,y} = \sum_{L=1}^{n_l} s_{k,L} N_{y,L} e^{-Z_{y,L} \tau_{k,y}}$$

where $s_{k,L}$ is size-specific selectivity of the survey, $\tau_{k,y} = J_{k,y}/365$, $J_{k,y}$ is the Julian date of the survey in year y , and $e^{-Z_{y,L} \tau_{k,y}}$ is a correction for mortality prior to the survey. Available biomass is calculated in the same way except that body weights w_L are included in the product on the right hand side.

Available stock for indices that track numbers dead by natural mortality is:

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

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

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

where $\tilde{N}_{y,L} = N_{y,L} e^{-Z\Delta}$ is abundance at elapsed time of year $\Delta = \tau_{k,y} - \nu_k$, $\nu_k = j_k / 365$, and j_k is the user-specified duration in days for the period of availability. The instantaneous rates for total $\tilde{Z}_{y,L} = Z_{y,L} (\tau_{k,y} - \nu_k)$ and natural $\tilde{M}_{y,L} = M_{y,L} (\tau_{k,y} - \nu_k)$ mortality are also adjusted to correspond to the period of availability. In using this approach, the user should be aware that the length based selectivity estimated by the model for the dead animal survey ($s_{k,L}$) is conditional on the assumed pattern of length-specific natural mortality (\bar{u}) which was specified as data in the input file.

NLL calculations for survey index data assume that log scale measurement errors are either normally distributed (default approach) or from a t distribution (robust estimation approach). In either case, log scale measurement errors are assumed to have mean zero and log scale standard errors either estimated internally by the model or calculated from the arithmetic CVs supplied with the survey data.

The standardized residual used in computing NLL for one survey index observation is $r_{k,y} = \ln(I_{k,y} / \hat{I}_{k,y}) / \sigma_{k,y}$ where $I_{k,y}$ is the observation. The standard deviations $\sigma_{k,y}$ will vary

among surveys and years if CVs are used to specify the variance of measurement errors. Otherwise a single standard deviation is estimated internally for the survey as a whole.

Survey length composition data

Length bins for fishery and survey length composition data are flexible and the flexibility affects goodness of fit calculations in ways that may be important to consider. The user specifies the starting size (bottom of first bin) and number of bins used for each type of fishery and survey length composition. The input data for each length composition record identifies the first/last length bins to be used and whether they are plus groups that should include all smaller/larger length groups in the data and population model when calculating goodness of fit. Goodness of fit calculations are carried out over the range of lengths specified by the user. Thus length data in the input file may contain data for size bins that are ignored in goodness of fit calculations. As described above, the starting size and bin size for the population model are specified separately. In the ideal and simplest case, the minimum size and same length bins are used for the population and for all length data. However, as described below, length specifications in data and the population model may differ.

Care is required in some cases because the implicit definitions of plus groups in the model and data may differ. If the first bin used for length data is a plus group, then the first bin will contain the sum of length data from the corresponding and smaller bins of the original length composition record. However, the first bin in the population model is never a plus group. Thus, predicted values for a plus group will contain the sum of the corresponding and smaller bins in the population. The observed and predicted values will not be perfectly comparable if the starting sizes for the data and population model differ. Similarly, if the last bin in the length data is a plus group, it will contain original length composition data for the corresponding and all larger bins. Predicted values for a plus group in the population will be the sum for the corresponding bin and all larger size groups in the population, implicitly including sizes $> L_{\infty}$. The two definitions of the plus group will differ and goodness of fit calculation may be impaired if the original length composition data does not include all of the large individuals in samples.

In the current version of the CASA model, the size of length composition bins must be $\geq L_{bin}$ in the population model (this constraint will be removed in later versions). Ideally, the size of data length bins is the same or a multiple of the size of length bins in the population. However, this is not required and the model will prorate the predicted population composition for each bin into adjacent data bins when calculating goodness of fit. With a 30-34 mm population bin and 22-31 and 32-41 mm population bins, for example, the predicted proportion in the population bin would be prorated so that 2/5 was assigned to the first data bin and 3/5 was assigned to the second data bin. This proration approach is problematic when it is used to prorate the plus group in the population model into two data bins because it assumes that abundance is uniform over lengths within the population group. The distribution of lengths in a real population might be far from uniform between the assumed upper and lower bounds of the plus group.

The first bin in each length composition data record must be $\geq L_{min}$ which is the smallest size group in the population model. If the last data bin is a plus group, then the *lower* bound of the last data bin must be \leq the upper bound of the last population bin. Otherwise, if the last data bin is not a plus group, the *upper* bound of the last data bin must be \leq the upper bound of the population bin.

NLL calculations for survey length composition data are similar to calculations for fishery length composition data. Surveys index data may measure trends in stock abundance or biomass but survey length composition data are always for numbers (not weight) of individuals in each length group. Survey length composition data represent a sample from the true stock which is modified by survey selectivity, sampling errors and, if applicable, errors in recording length data. For example, with errors in length measurements, individuals

belonging to length bin j , are mistakenly assigned to adjacent length bins $j-2, j-1, j+1$ or $j+2$ with some specified probability. Well-tested methods for dealing with errors in length data can be applied if some information about the distribution of the errors is available (e.g. Methot 2000).

Prior to any other calculations, observed survey length composition data are converted to multinomial proportions:

$$i_{k,y,L} = \frac{n_{k,y,L}}{\sum_{j=L_{k,y}^{first}}^{L_{k,y}^{last}} n_{k,y,j}}$$

where $n_{k,y,j}$ is an original datum and $i_{k,y,L}$ is the corresponding proportion. As described above, the user specifies the first $L_{k,y}^{first}$ and last $L_{k,y}^{last}$ length groups to be used in calculating goodness of fit for each length composition and specifies whether the largest and smallest groups should be treated as “plus” groups that contain all smaller or larger individuals.

Using notation for goodness of fit survey index data (see above), predicted length compositions for surveys that track abundance or biomass are calculated:

$$A_{k,y,L} = \frac{s_{k,L} N_{y,L} e^{-Z_{y,j} \tau_{k,y}}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{last}} s_{k,j} N_{y,j} e^{-Z_{y,j} \tau_{k,y}}}$$

Predicted length compositions for surveys that track numbers of individuals killed by natural mortality are calculated:

$$A_{k,y} = \frac{s_{k,L} \tilde{M}_{y,L} \bar{N}_{y,L}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{last}} s_{k,L} \tilde{M}_{y,L} \bar{N}_{y,L}}$$

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

$$\bar{A}'_{k,y} = \bar{A}_{k,y} E_k$$

where E_k is an error matrix that simulates errors in collecting length data by mapping true length bins in the model to observed length bins in the data.

The error matrix E_k has n_L rows (one for each true length bin) and n_L columns (one for each possible observed length bin). For example, row k and column j of the error matrix gives the conditional probability $P(k|j)$ of being assigned to bin k , given that an individual actually belongs to bin j . More generally, column j gives the probabilities that an individual actually belonging to length bin j will be recorded as being in length bins $j-2, j-1, j, j+1, j+2$ and so on. The columns of E_k add to one to account for all possible outcomes in assigning individuals to observed length bins. E_k is the identity matrix if there are no structured measurement errors.

In CASA, the probabilities in the error matrix are computed from a normal distribution with mean zero and $CV = e^{\pi_k}$, where π_k is an estimable parameter. The normal distribution is truncated to cover a user-specified number of observed bins (e.g. 3 bins on either side of the true length bin).

The NLL for observed proportions at length in each survey and year is computed with the kernel for a multinomial distribution, the model's estimate of proportional survey catch-at-length ($\hat{i}_{k,y,L}$) and the effective sample size lN_y supplied by the user. Residuals are not used in computing NLL for length composition data but are available for use in checking model fit.

Residuals for goodness of fit to length data

Three types of residuals are calculated automatically for all of the length composition data used in the model and are written to a special output file that can be used to make residual plots and other diagnostics. The output file contains one record for set of length composition data and length bin used in goodness of fit calculations. Each record contains the name of the survey, survey id number, length, length bin id number, observed proportion, predicted proportion and three types of residuals (simple, Pearson and deviance).

For length composition type t , in year y and length L , the simple residual is

$$r_{t,y,L} = p_{t,y,L} - \hat{p}_{t,y,L}$$

where $p_{t,y,L}$ and $\hat{p}_{t,y,L}$ are observed and predicted proportions at length. The Pearson residual is

$$\rho_{t,y,L} = \frac{r_{t,y,L}}{\hat{p}(1-\hat{p})/{}^lN_{t,y,L}}$$

where the denominator is the expected standard deviation given the predicted proportion and the effective sample size used in goodness of fit calculations. The deviance residual is basically the contribution of the length composition observation to the total likelihood:

$$\delta_{t,y,L} = \text{sign}(r_{t,y,L}) {}^lN_{t,y,L} [p_{r,y,L} \ln(\hat{p}_{r,y,L}) - p_{r,y,L} \ln(p_{r,y,L})]$$

Note that the deviance residual is zero if the observed and predicted proportions match exactly and that the deviance and simple residuals have the same sign.

Effective sample size for length composition data

Effective sample sizes that are specified by the user are used in goodness of fit calculations for survey and fishery length composition data. A post-hoc estimate of effective sample size can be calculated based on goodness of fit in a model run (Methot 1989). Consider the variance of residuals for a single set of length composition data with N bins used in calculations. The variance of the sum based on the multinomial distribution is:

$$\sigma^2 = \sum_{j=1}^N \left[\frac{\hat{p}_j(1-\hat{p}_j)}{\varphi} \right]$$

where φ is the effective sample size for the multinomial and \bar{p}_j is the predicted proportion in the j^{th} bin from the model run. Solve for φ to get:

$$\varphi = \frac{\sum_{j=1}^N [\hat{p}_j(1-\hat{p}_j)]}{\sigma^2}$$

The variance of the sum of residuals can also be calculated:

$$\sigma^2 = \sum_{j=1}^N (p_j - \hat{p}_j)^2$$

This formula is approximate because it ignores the traditional correction for bias. Substitute the third expression into the second to get:

$$\varphi = \frac{\sum_{j=1}^N [\hat{p}_j(1 - \hat{p}_j)]}{\sum_{k=1}^N (p_j - \hat{p}_j)^2}$$

which can be calculated based on model outputs. The assumed and effective sample sizes will be similar in a reasonable model when the assumed sample sizes are approximately correct. Effective sample size calculations can be used iteratively to manually adjust input values to reasonable levels (Methot 1989).

Variance constraints on dev parameters

Variability in dev parameters (e.g. for natural mortality, recruitment or fishing mortality) can be limited using variance constraints that assume the deviations are either independent or that they are autocorrelated and follow a random walk. When a variance constraint for independent deviations is activated, the model calculates the NLL for each log scale residual γ_y / σ_γ , where γ_y is a dev parameter and σ is a log-scale standard deviation. If the user supplies a positive value for the arithmetic scale CV, then the NLL is calculated assuming the variance is known. Otherwise, the user-supplied CV is ignored and the NLL is calculated with the standard deviation estimated internally. Calculations for autocorrelated deviations are the same except that the residuals are $(\gamma_y - \gamma_{y-1}) / \sigma_\gamma$ and the number of residuals is one less than the number of dev parameters.

LPUE data

Commercial landings per unit of fishing effort (LPUE) data are modeled in the current implementation of the CASA model as a linear function of average biomass available to the fishery, and as a nonlinear function of average available abundance. The nonlinear relationship with abundance is meant to reflect limitations in “shucking” capacity for sea scallops.²² Briefly, tows with large numbers of scallops require more time to sort and shuck and therefore reduce LPUE from fishing trips when abundance is high. The effect is exaggerated when the catch is composed of relatively small individuals. In other words, at any given level of stock biomass, LPUE is reduced as the number of individuals in the catch increases or, equivalently, as the mean size of individuals in the catch is reduced.

Average available abundance in LPUE calculations is:

$${}^a\bar{N}_y = \sum_{L=1}^{n_L} s_{y,L} \bar{N}_{y,L}$$

and average available biomass is:

$${}^a\bar{B}_y = \sum_{L=1}^{n_L} s_{y,L} w_L^f \bar{N}_{y,L}$$

where the weights at length w_L^f are for the fishery rather than the population. Predicted values for LPUE data are calculated:

$$\hat{L}_y = \frac{{}^a\bar{B}_y \eta}{\sqrt{\varphi^2 + {}^a\bar{N}_y^2}}$$

²² D. Hart, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA, pers. comm.

Measurement errors in LPUE data are assumed normally distributed with standard deviations $\sigma_y = CV_y \hat{L}_y$. Standardized residuals are $r_y = (L_y - \hat{L}_y) / \sigma_y$.

Per recruit (SBR and YPR) reference points

The user specifies a target %SBR value for each reference point that is estimated. Goodness of fit is calculated as the sum of squared differences between the target %SBR and %SBR calculated based on the reference point parameter. Except in pathological situations, it is always possible to estimate %SBR reference point parameters so that the target and calculated %SBR levels match exactly. Reference point parameters should have no effect on other model estimates and the residual (calculated – target %SBR) should always be very close to zero.

Goodness of fit for $F_{0.1}$ estimates is calculated in a manner similar to %SBR reference points. Goodness of fit is calculated as the squared difference between the slope of the yield curve at the estimate and one-tenth of the slope at the origin. Slopes are computed numerically using central differences if possible or one-sided (right hand) differences if necessary.

F_{max} is estimated differently in preliminary and final phases. In preliminary phases, goodness of fit for F_{max} is calculated as $(1/Y)^2$, where Y is yield per recruit at the current estimate of F_{max} . In other words, yield per recruit is maximized by finding the parameter estimate that minimizes its inverse. This preliminary approach is very robust and will find F_{max} if it exists. However, it involves a non-zero residual $(1/Y)$ that interferes with calculation of variances and might affect other model estimates. In final phases, goodness of fit for F_{max} is calculated as (d^2) where d is the slope of the yield per recruit curve at F_{max} . The two approaches give the same estimates of F_{max} but the goodness of fit approach used in the final phases has a residual of zero (so that other model estimates are not affected) and gives more reasonable variance estimates. The latter goodness of fit calculation is not used during initial phases because the estimates of F_{max} tend to “drift down” the right hand side of the yield curve in the direction of decreasing slope. Thus, the goodness of fit calculation used in final phases works well only when the initial estimate of F_{max} is very close to the best estimate.

Per recruit reference points should have little or no effect on other model estimates. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with and without reference point calculations to ensure that reference points do not affect other model estimates including abundance, recruitments and fishing mortality rates.*

Growth data

Growth data in CASA consist of records giving initial length, length after one year of growth, and number of corresponding observations. Growth data may be used to help estimate growth parameters that determine the growth matrix P . The first step is to convert the data for each starting length to proportions:

$$P(b,a) = \frac{n(b,a)}{\sum_{j=n_L-b+1}^{n_L} n(j,a)}$$

where $n(b,a)$ is the number of individuals starting at size a that grew to size b after one year. The NLL is computed assuming that observed proportions $p(a|b)$ at each starting size are a sample from a multinomial distribution with probabilities given by the corresponding column in the models estimated growth matrix P . The user must specify an effective sample size $^P N_j$ based, for example, on the number of observations in each bin or the number of individuals contributing data to each bin. Observations outside bin ranges specified by the

user are ignored. Standardized residuals for plotting are computed based on the variance for proportions.

Survey gear efficiency data

Survey gear efficiency for towed trawls and dredges is the probability of capture for individuals anywhere in the water column or sediments along the path swept by the trawl. Ideally, the area surveyed and the distribution of the stock coincides so that:

$$I_{k,y} = q_k B_{k,y}$$

$$q_k = \frac{a_k e_k u_k}{A}$$

$$e_k = \frac{A q_k}{a_k u_k}$$

$$K_t = \frac{A}{a_k u_k}$$

$$e_k = K_t q_t$$

where $I_{k,y}$ is a survey observation in units equivalent to biomass (or numerical) density (e.g. kg per standard tow), $B_{k,y}$ is the biomass (or abundance) available to the survey, A is the area of the stock, a_k is the area swept during one tow, $0 < e_k \leq 1$ is efficiency of the survey gear, and u_k is a constant that adjusts for different units.

Efficiency estimates from studies outside the CASA model may be used as prior information in CASA. The user supplies the mean and CV for the prior estimate of efficiency, along with estimates of A_k , a_k and u_k . At each iteration if the model, the gear efficiency implied by the current estimate of q_k is computed. The model then calculates the NLL of the implied efficiency estimate assuming it was sampled from a unimodal beta distribution with the user-specified mean and CV.

If efficiency estimates are used as prior information (if the likelihood weight $\lambda > 0$), then it is very important to make sure that units and values for the survey data (I), biomass or abundance (B), stock area (A), area per tow (a), and adjustments for units (u) are correct (see Example 1). The units for biomass are generally the same as the units for catch data. In some cases, incorrect specifications will lead to implied efficiency estimates that are ≤ 0 or ≥ 1 which have zero probability based on a standard beta distribution used in the prior. The program will terminate if $e \leq 0$. If $e \geq 1$ during an iteration, then e is set to a value slightly less than one and a penalty is added to the objective function. In some cases, incorrect specifications will generate a cryptic error that may have a substantial impact on estimates.

Implied efficiency estimates are useful as a model diagnostic even if very little prior information is available because some model fits may imply unrealistic levels of implied efficiency. The trick is to down weight the prior information (e.g. $\lambda = 1e^{-6}$) so that the implied efficiency estimate has very little effect on model results as long as $0 < e < 1$. Depending on the situation, model runs with e near a bound indicate that estimates may be implausible. In addition, it may be useful to use a beta distribution for the prior that is nearly a uniform distribution by specifying a prior mean of 0.5 and variance slightly less than $1/12 = 0.083333$.

Care should be taken in using prior information from field studies designed to estimate survey gear efficiency. Field studies usually estimate efficiency with respect to individuals on the same ground (e.g. by sampling the same grounds exhaustively or with two types of gear). It seems reasonable to use an independent efficiency estimate and the corresponding survey index to estimate abundance in the area surveyed. However, stock assessment models are usually applied to the entire stock, which is probably distributed over a larger area than the area covered by the survey. Thus the simple abundance calculation based on efficiency and

the survey index will be biased low for the stock as a whole. In effect, efficiency estimates from field studies tend to be biased high as estimates of efficiency relative to the entire stock.

Maximum fishing mortality rate

Stock assessment models occasionally estimate absurdly high fishing mortality rates because abundance estimates are too small. The NLL component used to prevent this potential problem is:

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

where:

$$d_t = \begin{cases} Ft - \Phi & \text{if } Ft > \Phi \\ 0 & \text{otherwise} \end{cases}$$

and

$$q_t = \begin{cases} \ln(Ft/\Phi) & \text{if } Ft > \Phi \\ 0 & \text{otherwise} \end{cases}$$

with the user-specified threshold value Φ set larger than the largest value of F_t that might possibly be expected (e.g. $\Phi=3$). The weighting factor λ is normally set to a large value (e.g. 1000).

APPENDIX B11: Forecasting methodology (SAMS model)

The model presented here is a modified version of the SAMS (Scallop Area Management Simulator) model used to project abundances and landings as an aid to managers since 1999. Subareas were chosen to coincide with current management. Thus, Georges Bank was divided into three open areas (South Channel, Northern Edge and Peak, and Southeast Part), the three access portions of the groundfish closures, and the three no access portions of these areas. The Mid-Atlantic was subdivided into six areas: Virginia Beach, Delmarva, the Elephant Trunk Closed Area, the Hudson Canyon South Access Area, New York Bight, and Long Island.

Methods

The model follows, for each area i and time t , population vectors $\mathbf{p}(i,t) = (p_1, p_2, \dots, p_n)$, where p_j represents the density of scallops in the j th size class in area i at time t . The model uses a difference equation approach, where time is partitioned into discrete time steps t_1, t_2, \dots , with a time step of length $\Delta t = t_{k+1} - t_k$. The landings vector $\mathbf{h}(i,t_k)$ represents the catch at each size class in the i th region and k th time step. It is calculated as:

$$h(i,t_k) = [I - \exp(\Delta t H(i,t_k))] p(i,t_k),$$

where I is the identity matrix and H is a diagonal matrix whose j th diagonal entry h_{jj} is given by:

$$h_{jj} = 1 / (1 + \exp(s_0 - s_1 * s))$$

where SH is the shell height of the mid-point of the size-class. The parameters s_0 and s_1 are derived in Appendix V.

The landings $L(i,t_k)$ for the i th region and k th time step are calculated using the dot product of landings vector $\mathbf{h}(i,t_k)$ with the vector $\mathbf{m}(i)$ representing the vector of meat weights at shell height for the i th region:

$$L(i,t_k) = A_i \mathbf{h}(i,t_k) \bullet \mathbf{m}(i) / (w e_i)$$

where e_i represents the dredge efficiency in the i th region, and w is the tow path area of the survey dredge (estimated as $8/6076 \text{ nm}^2$).

Even in the areas not under special area management, fishing mortalities tend to not be spatially uniform for poorly mobile stocks such as sea scallops (Hart 2001). Fishing mortalities in open areas were determined by a simple “fleet dynamics model” that estimates fishing mortalities in open areas based on area-specific exploitable biomasses, and so that the overall DAS or open-area F matches the target. Based on these ideas, the fishing mortality F_i in the i th region is modeled as:

$$F_i = k * f_i * B_i$$

where B_i is the exploitable biomass in the i th region, f_i is an area-specific adjustment factor to take into account preferences for certain fishing grounds (due to lower costs, shorter steam times, ease of fishing, habitual preferences, etc.), and k is a constant adjusted so that the total DAS or fishing mortality meets its target. For these simulations, $f_i = 1$ for all areas.

Scallops of shell height less than a minimum size s_d are assumed to be discarded, and suffer a discard mortality rate of d . Discard mortality was estimated in NEFSC (2004) to be

20%. There is also evidence that some scallops not actually landed may suffer mortality due to incidental damage from the dredge. Let F_L be the landed fishing mortality rate and F_I be the rate of incidental mortality. For Georges Bank, which is a mix of sandy and hard bottom, we used $F_I = 0.15F_L$. For the Mid-Atlantic (almost all sand), we estimated $F_I = 0.04F_L$.

Growth in each subarea was specified by a growth transition matrix G , based on area-specific growth increment data (see Appendix III).

Recruitment was modeled stochastically, and was assumed to be log-normal in each subarea. The mean, variance and covariance of the recruitment in a subarea was set to be equal to that observed in the historical time-series between 1979-2006 (Mid-Atlantic) and 1982-2006 (Georges Bank). New recruits enter the smallest nine size bins in proportions (1/7,1/7,1/7,1/7,1/7,4/35,3/35,2/35,1/35) at a rate r_i depending on the subarea i , and stochastically on the year. Area-specific recruitment rates are given in Table 1. These simulations assume that recruitment is a stationary process, i.e., no stock-recruitment relationship is assumed (NEFSC 2004). At the current high biomass levels, it is likely that any stock-recruitment relationship would have asymptoted, so that this assumption is reasonable provided that biomass remain at or above the target level.

The population dynamics of the scallops in the present model can be summarized in the equation:

$$p(i, t_{k+1}) = \rho_i + G \exp(-M\Delta t H) p(i, t_k),$$

where ρ_i is a random variable representing recruitment in the i th area. The population and harvest vectors are converted into biomass by using the shell-height meat-weight relationship:

$$W = \exp[a + b \ln(s)],$$

where W is the meat weight of a scallop of shell height s . For calculating biomass, the shell height of a size class was taken as its midpoint. A summary of model parameters is given in Table 2.

Commercial landing rates (LPUE) were estimated using an empirical function based on the observed relationship between annual landing rates, expressed as number caught per day (NLPUE) and survey exploitable numbers per tow. At low biomass levels, NLPUE increases roughly linearly with survey abundance. However, at high abundance levels, the catch rate of the gear will exceed that which can be shucked by a seven-man crew. This is similar to the situation in predator/prey theory, where a predator's consumption rate is limited by the time required to handle and consume its prey (Holling 1959). The original Holling Type-II predator-prey model assumes that handling and foraging occur sequentially. It predicts that the per-capita predation rate R will be a function of prey abundance N according to a Monod functional response:

$$R = \frac{\alpha N}{\beta + N},$$

where α and β are constants. In the scallop fishery, however, some handling (shucking) can occur while foraging (fishing), though at a reduced rate because the captain and one or two crew members need to break off shucking to steer the vessel during towing and to handle the gear during haulback. The fact that a considerable amount of handling can occur at the same time as foraging means that the functional response of a scallop vessel will saturate quicker than that predicted by the above equation. To account for this, a modified Holling Type-II model was used, so that the landings (in numbers of scallops) per unit effort (DAS) L (the predation rate, i.e., NLPUE) will depend on scallop (prey) exploitable numbers N according to the formula:

$$L = \frac{\alpha N}{\sqrt{\beta^2 + N^2}}. \quad (*)$$

The parameters α and β to this model were fit to the observed fleet-wide LPUE vs. exploitable biomass relationship during the years 1994-2004 (previous years were not used because of the change from port interviews to logbook reporting). The number of scallops that can be shucked should be nearly independent of size provided that the scallops being shucked are smaller than about a 20 count. The time to shuck a large scallop will go up modestly with size. To model this, if the mean meat weight of the scallops caught, g , in an area is more than 20 g, the parameters α and β in (*) are reduced by a factor $\sqrt{20/g}$. This means, for example, that a crew could shuck fewer 10 count scallops per hour than 20 count scallops in terms of numbers, but more in terms of weight.

An estimate of the fishing mortality imposed in an area by a single DAS of fishing in that area can be obtained from the formula $F_{DAS} = L_a/N_a$, where L_a is the NLPUE in that area obtained as above, and N_a is the exploitable abundance (expressed as absolute numbers of scallops) in that area. This allows for conversion between units of DAS and fishing mortality.

Initial conditions for the population vector $\mathbf{p}(i,t)$ were estimated using the 2006 NMFS research vessel sea scallop survey, with dredge efficiency chosen so as to match the 2006 CASA biomass estimates. The initial conditions from the 2006 survey were bootstrapped using the bootstrap model of Smith (1997), so that each simulation run had both its own stochastically determined bootstrapped initial conditions, as well as stochastic recruitment stream.

APPENDIX B11 Table 1 – Mean and covariance of area specific log-transformed recruitment

Mid-Atlantic	HC	VB	ET	DMV	NYB	LI			
<i>Means</i>	4.14	3.88	4.41	4.01	3.39	3.14			
<i>Covariance Matrix</i>									
HC	1.48	0.54	1.14	0.97	0.93	0.65			
VB	0.54	2.04	0.58	1.32	0.06	-0.20			
ET	1.14	0.58	1.96	1.20	0.75	0.74			
DMV	0.97	1.32	1.20	1.84	0.70	0.34			
NYB	0.93	0.06	0.75	0.70	1.17	0.81			
LI	0.65	-0.20	0.74	0.34	0.81	0.98			
Georges Bank	CL1-NA	CL1-Acc	CL2-NA	CL2-Acc	NLS-NA	NLS-Acc	Sch	NEP	SEP
<i>Means</i>	3.67	3.51	2.87	3.34	-2.15	3.41	4.62	3.16	2.38
<i>Covariance Matrix</i>									
CL1-NA	2.92	0.03	0.34	0.32	-1.03	-0.45	0.75	-0.22	-0.47
CL1-Acc	0.03	1.83	0.94	0.77	2.24	0.58	0.61	0.52	0.38
CL2-NA	0.34	0.94	1.98	0.89	-0.40	0.27	0.53	0.33	0.34
CL2-Acc	0.32	0.77	0.89	2.63	2.22	1.34	0.76	1.00	0.77
NLS-NA	-1.03	2.24	-0.40	2.22	11.03	1.22	0.18	2.09	2.52
NLS-Acc	-0.45	0.58	0.27	1.34	1.22	5.07	0.25	0.72	0.39
Sch	0.75	0.61	0.53	0.76	0.18	0.25	1.27	0.20	0.01
NEP	-0.22	0.52	0.33	1.00	2.09	0.72	0.20	0.82	0.57
SEP	-0.47	0.38	0.34	0.77	2.52	0.39	0.01	0.57	1.42

APPENDIX B11 Table 2. Model parameters

Parameter	Description	Value
Δt	Simulation time step	1 y
M	Natural mortality rate	0.1 y ⁻¹
A	Shell height/meat wt parameter	-10.70 (GB), -12.01 (MA)
B	Shell height/meat wt parameter	2.94 (GB), 3.22 (MA)
s_0	Logistic selectivity parameter	9.692
s_1	Logistic selectivity parameter	0.1016
s_d	Cull size	90 mm
D	Mortality of discards	0.2
E	Dredge efficiency	0.311 (GB), 0.394 (MA)
α	LPUE/biomass relationship	43183
β	LPUE/biomass relationship	30626

APPENDIX B12: Simulator testing of CASA and rescaled F models

We conducted simulation testing to detect potential bugs, check accuracy and assess robustness of the CASA, rescaled F , and Beverton and Holt (1956) stock assessment models for sea scallops. CASA is a new and relatively complex stock assessment approach for sea scallops that uses a wide range of data, involves a number of assumptions, and estimates fishing mortality, abundance, biomass and other population characteristics by maximum likelihood (Appendix B10). The rescaled F model is a very simple approach used in previous sea scallop assessments (NEFSC 2004) that estimates fishing mortality based on survey data shell height composition, landings data and some information about growth and natural mortality. The Beverton-Holt (1956) model is a simple, equilibrium approach often used for “data poor” stocks. It uses survey size (e.g., shell height) composition data to estimate fishing mortality.

Software

Four independently coded programs were used in testing: a simulator program, the CASA and rescaled F /Beverton-Holt estimation programs, and an interface program to link them. The first program (SAMS model, Appendix B11) simulates a potentially realistic (e.g. spatially structured) population and saves “true” simulated population information (e.g. abundance at size and catch at size without observation errors) for use by the estimation programs.

The interface program links SAMS output to the three assessment models and summarized test results. The interface constructed data files required to run each assessment model with user specified amounts of observation errors in simulated landings, fishing effort, survey records, LPUE observations and survey and fishery length composition data. All models use the same data (same observation errors) in each iteration.

The interface program runs each assessment model with simulated data, and collects and stores biomass, fishing mortality and other estimates from each model. After a specified number of iterations, the interface summarizes information from each model and iteration. Output from the interface program includes tables that compare estimates of biomass and fishing mortality from each model to the “true” values based on a number of statistics that measure model performance.

The statistics used to measure model performance include the CV, %bias (bias/true value), and %RMSE (root means squared error/true value) for biomass and fishing mortality. CV measures the relative precision of estimates (variability around their mean). The %bias statistic measures the relative difference between the truth and the average estimate. The %RMSE statistic measures relative accuracy, considering both precision and bias. The three measures are related because mean squared error $MSE = \text{bias}^2 + \text{variance}$.

Simulated landings and survey abundance data were assumed in simulations to be gamma random variables, with mean equal to their true values and a specified variance. Simulated shell height composition data were multinomial random variables based on a user specified number of samples from the true shell height composition. LPUE data were a nonlinear function of stock biomass and abundance calculated from simulated landings and fishing effort data assuming that observation errors for landings and fishing effort were independent

Simulations for sea scallops

Results are presented below for example simulations of particular relevance to this assessment. Similar to patterns in the real scallop fishery, simulations were for 30 years with true fishing mortality starting at a moderate level ($F = 0.5$), increasing to a high level of $F = 1.0$,

and then fell decreasing to a relatively low level ($F = 0.3$) near the end of the simulation. The simulated population assumed some variability in growth among six areas within a single region.

Simulated data ranged from very precise to imprecise. A single survey abundance index with a flat selectivity curve and LPUE data as a nonlinear index of average fishable abundance was available in each simulated year. There were three sets of sets of simulated data with CVs and multinomial sample sizes listed in the table below. The assessment models were all run 20 times for each set of observation errors.

Scenario (magnitude of observation errors)	CV survey & landings data	CV for effort data	Sample size survey and fishery shell height data
Low	10%	2%	800
Medium	20%	2%	400
High	30%	2%	200

Other than observation errors in simulated data, all of the assessment models were generally configured for optimal model performance. In particular, assumptions about natural mortality and growth assumed in modeling were accurate. Size ranges assumed in tabulating survey data for the rescaled F model and the assumed critical length in Beverton-Holt model were reasonable choices. The growth transition matrix supplied to CASA was the average transition matrix for all area in the simulations. In CASA modeling, assumptions about the survey selectivity pattern (flat) and the general shape of the fishery selectivity pattern (logistic) were correct. There were no changes in fishery selectivity patterns that might have complicated interpretation of results from any of the models.

Results

For convenience, model performance statistics were averaged over all years for each model and level of observation error (Table 1). In terms of average percent bias, fishing mortality estimates from the rescaled F model were consistently biased low (-11 %). CASA model fishing mortality estimates were consistently biased high to a modest extent (< 5%). CASA model abundance and biomass estimates were biased high, usually by less than 10%. CASA estimates of landings were relatively unbiased (-0.3 to -2%). More simulations with larger numbers of iterations are required to make definite conclusions, but %bias was not strongly dependent on the magnitude of observation errors.

CASA model F estimates were most precise (lower CV, Table 1) than estimates from alternative models unless observation errors were high. CVs for CASA model fishing mortality, abundance, biomass and landings estimates increased almost proportionally with CVs for simulated observation errors assumed in survey and landings data.

Results for %RMSE (Table 1) were similar to results for CVs because bias was modest in all cases and changes in accuracy were due primarily to differences in precision.

Comparison of the mean fishing mortality estimates for each year from the three models gives insights into their performance (Figure 1). The negative bias in the rescaled F mortality estimates was due to underestimation of fishing mortality during years when true fishing mortality rates were highest. The positive bias of the CASA model was due to a consistent overestimation of mortality during the first four years of the simulation. CASA estimated fishing mortalities that were essentially unbiased after the initial years. The strong

oscillations in the Beverton-Holt estimator are due to recruitment variability in the underlying simulation.

In comparing results for individual years, fishing mortality estimates from the rescaled F seem more variable than from CASA (Figure 2 to 3). In addition, CASA estimates seem to track trends in true fishing mortality better than estimates from other models. CASA estimates appear to track abundance and biomass with a reasonably well (Figure 3 to 4).

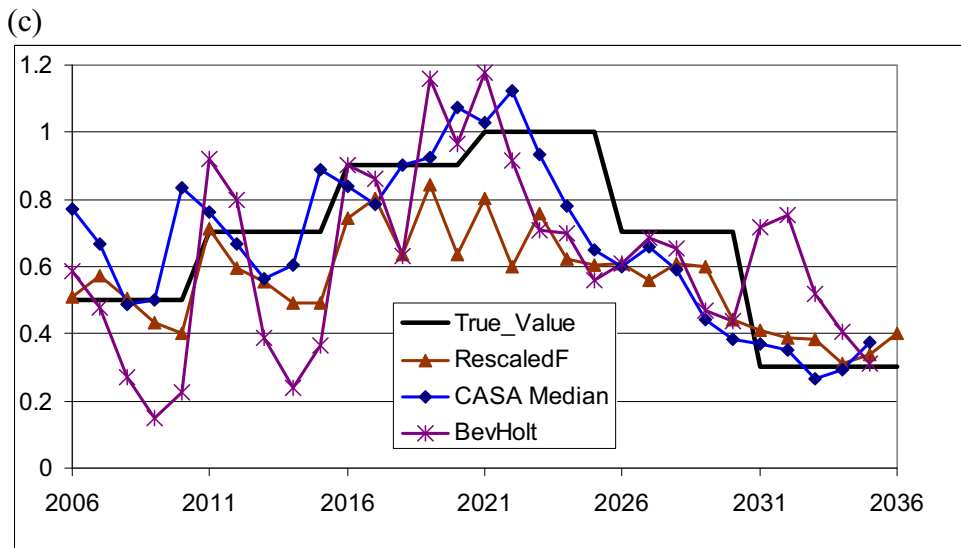
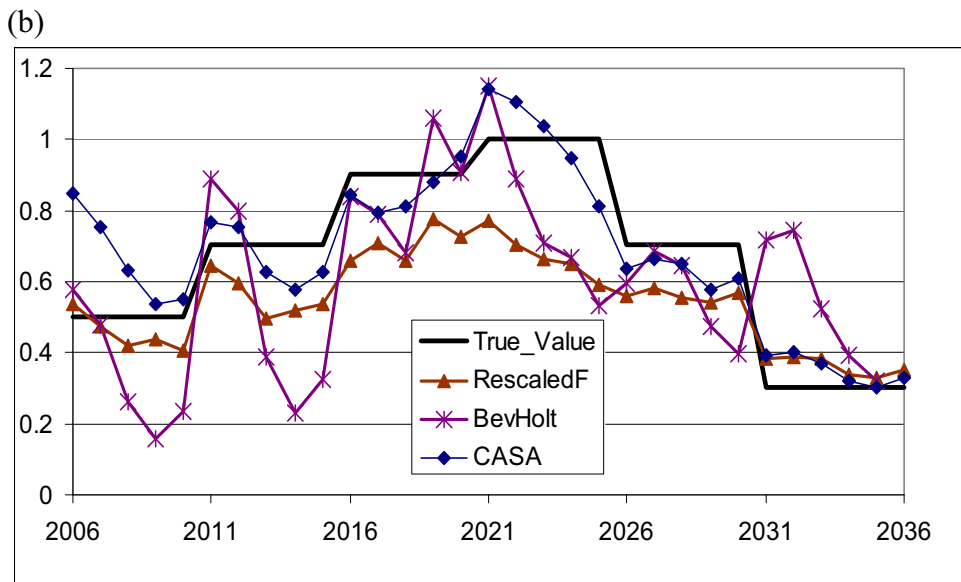
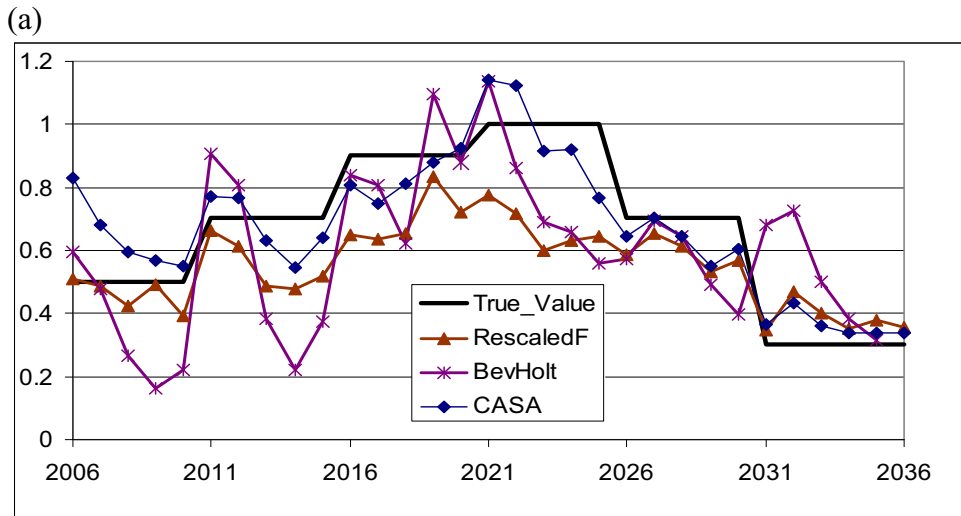
Conclusions

More testing is required, but simulation tests support use of CASA in this assessment for sea scallops. Results indicate that the CASA model is working properly and estimating abundance and biomass reasonably well. The CASA model generally performed better than the rescaled F and Beverton-Holt models. With the exception of the first few years, fishing mortality estimates from CASA was nearly unbiased.

CASA estimates were the most precise and accurate, except at the highest (30%) observation error levels. For sea scallops, low to medium (10-20%) observation errors in survey data are probably more realistic because the dredge and video surveys are relatively precise.

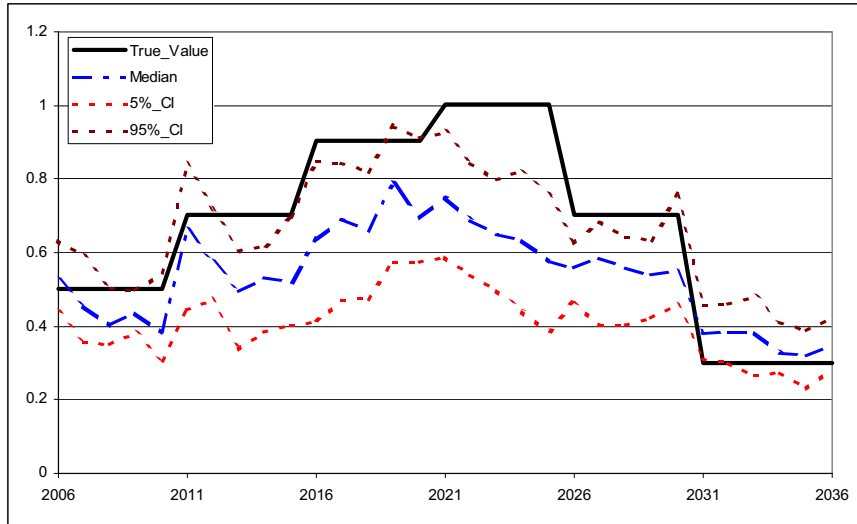
APPENDIX B12 Table 1. Performance measures (%bias, CV and %RMSE) for fishing mortality, abundance, biomass and landings estimates based on simulation testing (20 iterations each). Figures for each model are averages performance measures averaged over 30 simulated years. Performance during individual years may have been better or worse than indicated in the table. The CASA model failed to converge in one iteration with high observation errors. Effects of this run on performance measures for CASA with high levels of observation error were minimized by using medians, instead of means, in the table. When all runs converged, means and medians were similar.

Model / estimate	%Bias			CV			%RMSE		
	Low	Medium	High	Low	Medium	High	Low	Medium	High
Rescaled F	-0.13	-0.11	-0.11	0.15	0.28	0.46	0.26	0.35	0.49
Beverton-Holt F	-0.07	-0.01	0.05	0.11	0.12	0.14	0.23	0.40	0.42
CASA-F	0.05	0.05	0.03	0.20	0.20	0.30	0.20	0.28	0.32
CASA-Abundance	0.08	0.07	0.12	0.04	0.09	0.49	0.17	0.20	0.58
CASA-Biomass	0.04	0.03	0.07	0.05	0.10	0.57	0.09	0.13	0.61
CASA-Landings	0.00	0.00	-0.02	0.10	0.20	0.29	0.10	0.20	0.29

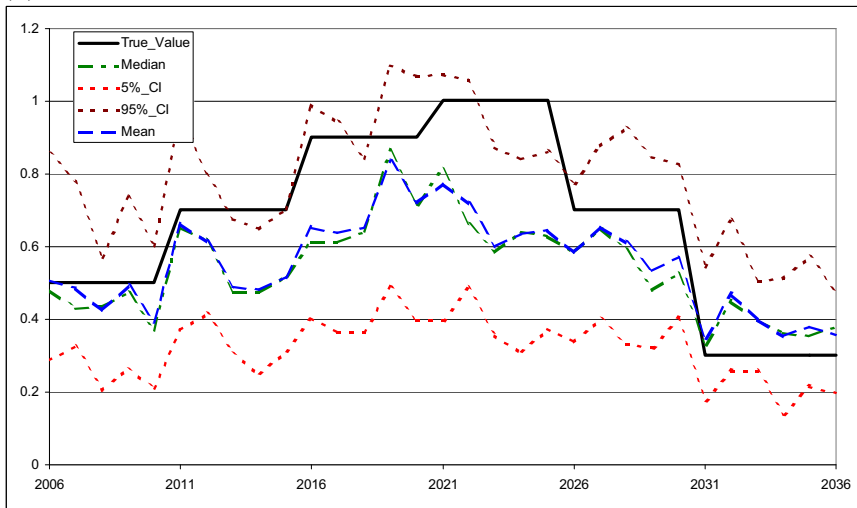


APPENDIX B12 Figure 1. Mean annual fishing mortalities for fishing mortality estimates from three models using data with (a) low, (b) medium, and (c) high observation errors.

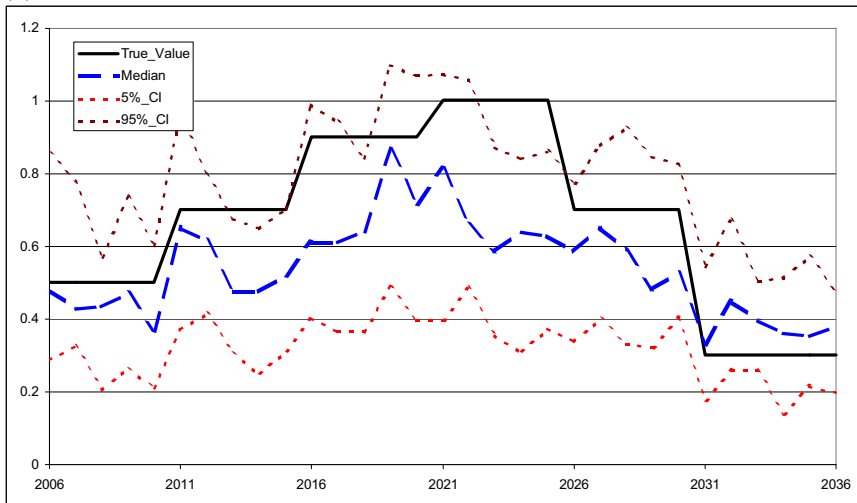
(a)



(b)

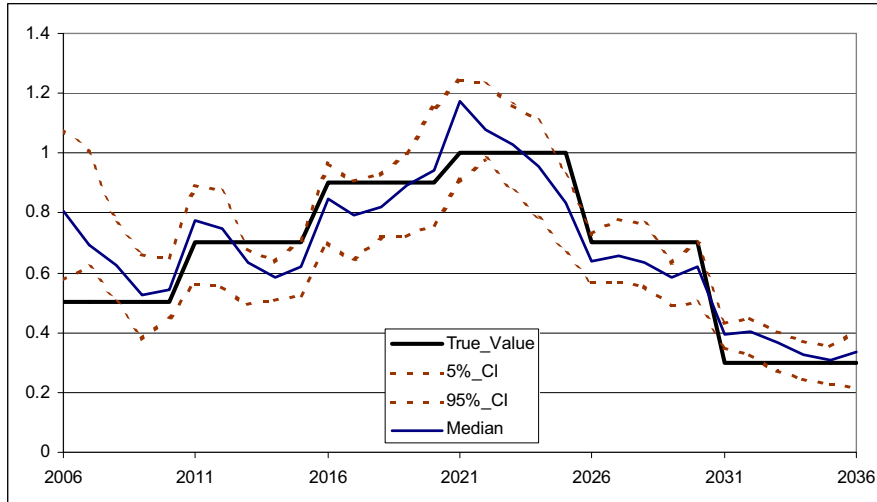


(c)

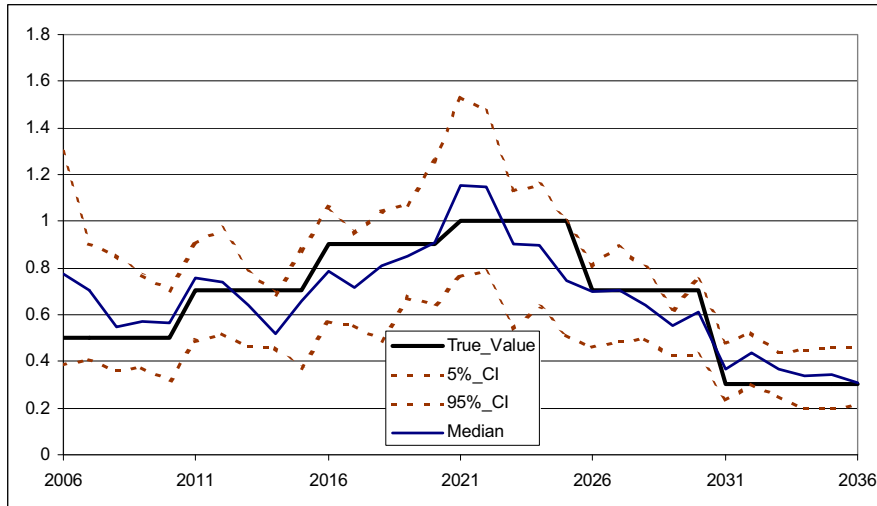


APPENDIX B12 Figure 2. Median, 5th and 95th percentiles for rescaled F estimates of annual fishing mortality using data with (a) low, (b) medium, and (c) high observation errors.

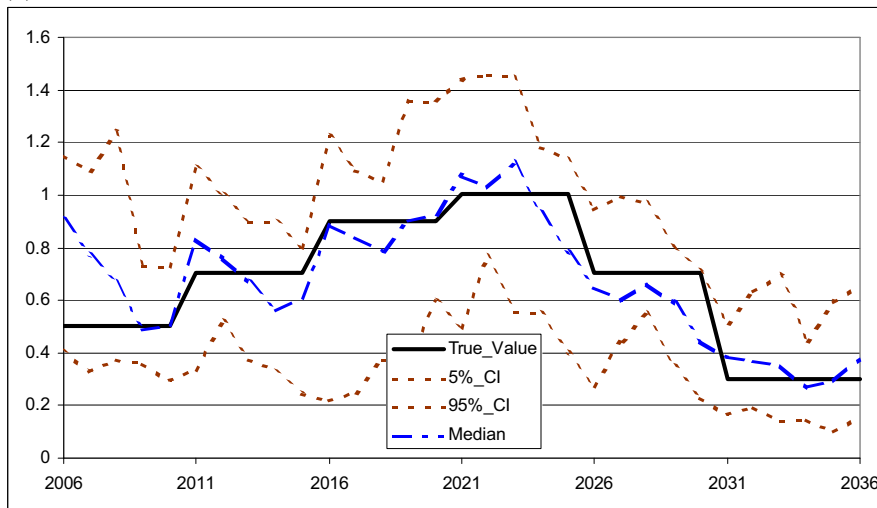
(a)



(b)

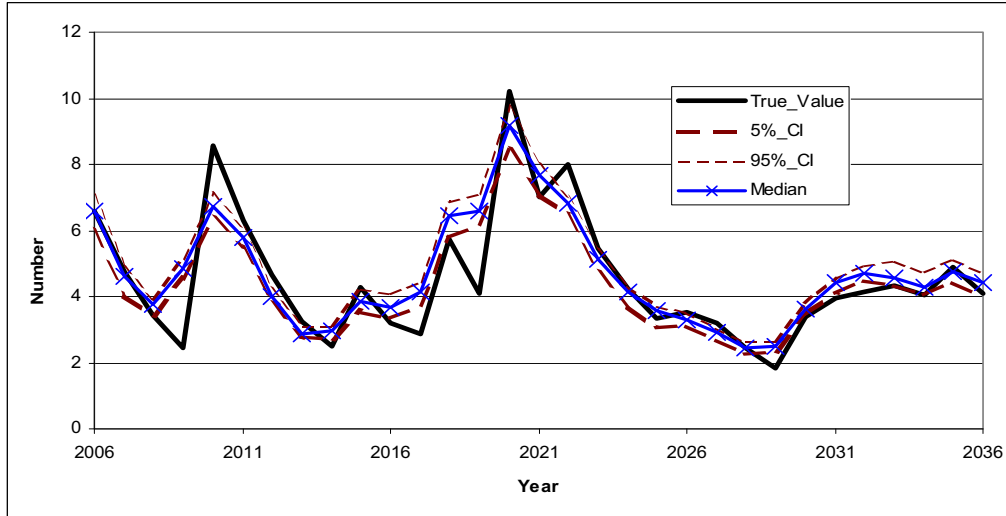


(c)

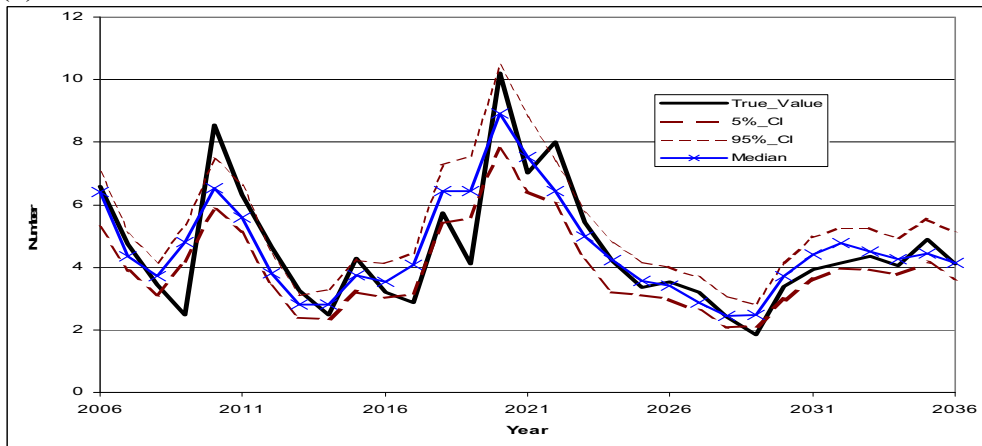


APPENDIX B12 Figure 3. Median, 5th and 95th percentiles for CASA annual fishing mortality estimates using data with (a) low, (b) medium, and (c) high observation errors.

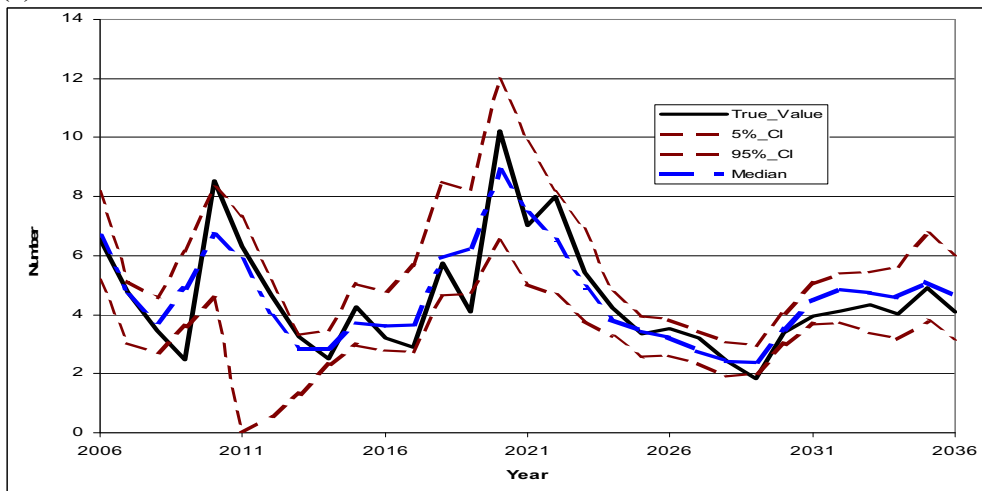
(a)



(b)

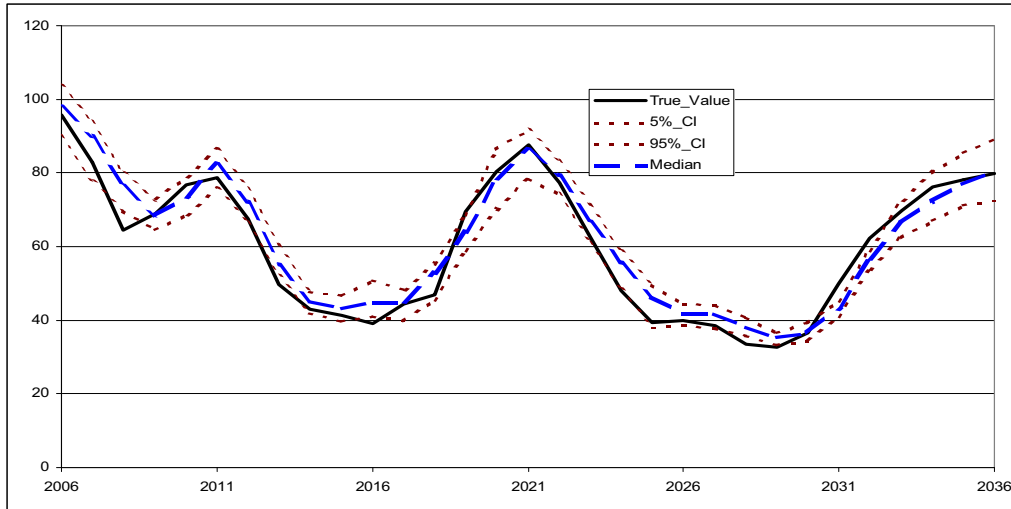


(c)

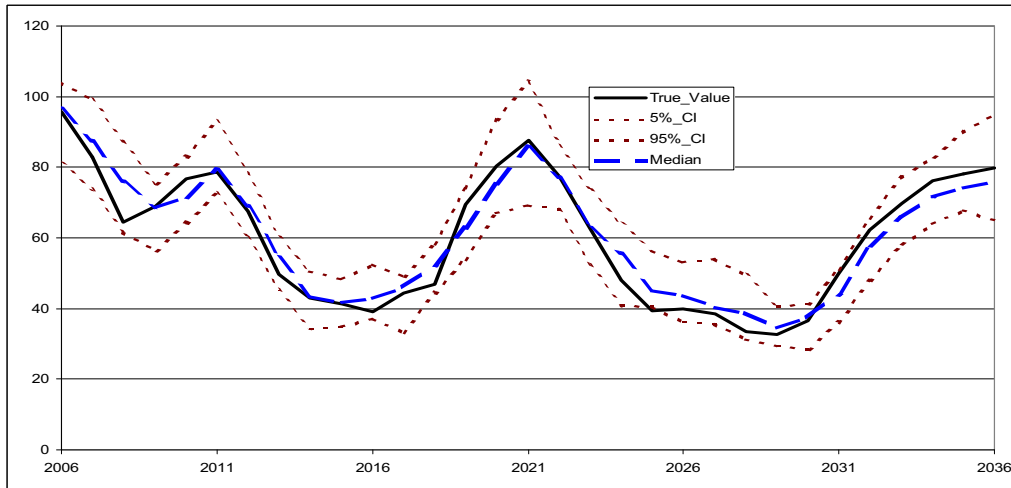


APPENDIX B12 Figure 4. Median, 5th and 95th percentiles of CASA annual abundance estimates using data with (a) low, (b) medium, and (c) high observation errors.

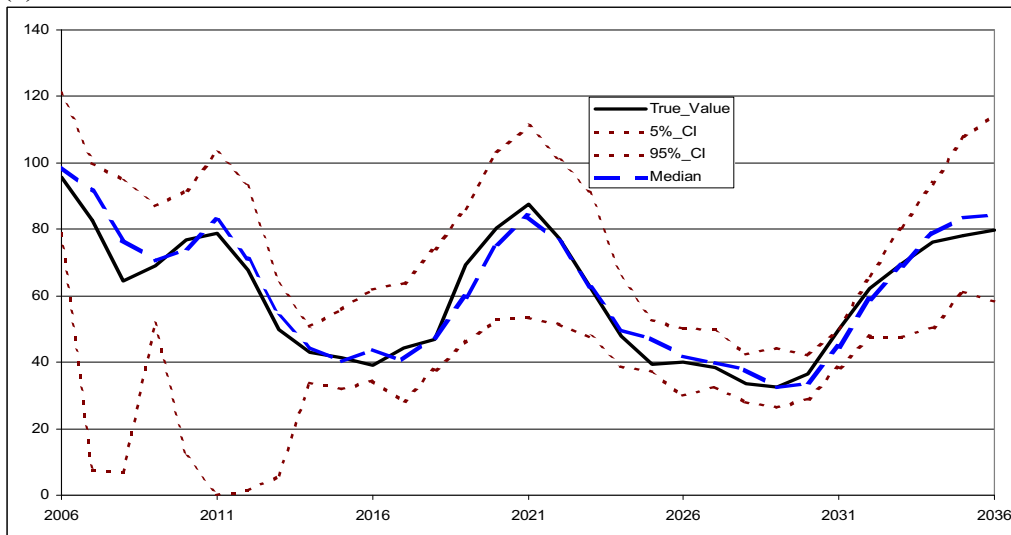
(a)



(b)



(c)



APPENDIX B12 Figure 5. Median, 5th and 95th percentiles for CASA annual biomass estimates using data with (a) low, (b) medium, and (c) high observation errors.