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MONKFISH REVIEW PANEL SUMMARY

The Panel reviewed the 2013 monkfish operational stock assessment on April 8-9, 2013 in Woods Hole, Massachusetts. The model configuration has not changed substantively since the last peer review by the SARC 50 in 2010. The model has been updated with two years of data and revisions of discard estimates for 1980-2011 based on new methodology (SBRM approach). Changes in the discard estimates resulted in a minor reduction in the number of selectivity blocks in the southern stock model. Projections of biomass and catch are likely over-optimistic due to the retrospective patterns in both stocks. The Review Panel agreed that the assessment team met all Terms of Reference.

Model results indicate that the North and South, monkfish stocks are not over-fished and overfishing is not occurring. Nevertheless, both stocks demonstrate retrospective patterns in fishing mortality and biomass with fishing mortality consistently being under-estimated and biomass being over-estimated. This pattern was stronger for the Northern Management Area stock component. Potential causes of these retrospective patterns include misspecification of growth and natural mortality. The Review Panel recommends that a new benchmark assessment not proceed until new information on age, growth, longevity, and natural mortality is obtained. Potential differences by sex would need to be addressed. Notwithstanding these concerns, regular assessment updates might be needed to meet management requirements.

The panel noted that a number of key uncertainties remain unresolved since 2010 SARC. These include uncertainties in landings, discards, commercial length frequencies, aging methods, life history, growth, and natural mortality. These uncertainties are propagated through the SCALE assessment model and lead to greater uncertainties in estimates of stock size, recruitment, fishing mortality, biological reference points, and stock projections. The compounding nature of these uncertainties implies increased risk of not achieving the biological reference points. Despite these uncertainties, the work presented represents the best available scientific information and modeling approach for assessing the status of monkfish, and is accepted by the Review Panel for determining the stock status and providing catch advice.

The Review Panel examined projections for initial conditions of population sizes with and without correction for retrospective patterns. In both instances, the probability of becoming overfished in the short term is negligible. Considering consistency of retrospective pattern demonstrated in 2010 and 2013 assessments, the Panel agreed that an adjustment for the retrospective pattern should be made. However, the Panel expressed concern that the adjustment to the initial stock size for projections without change to reference points creates an inconsistency in determination of stock status. The Panel agreed that the correction for retrospective pattern did not address fully the sources of unresolved uncertainty detailed above.

The Review Panel discussed and recommended the following research priorities:

1) resolution of age, growth, and natural mortality issues

2) determination of movement patterns in relation to stock areas
3) development of a one stock model given evidence of movement between the two areas and existing genetic information (on-going genetics work may resolve the two stock-area issue)
4) development of a two-sex model depending on the results of aging work (would require estimation of sex ratios in catch and survey data)

2013 Monkfish Assessment Update

Executive Summary

Assessments of the northern and southern management units of monkfish were updated with minmal changes to methodological approaches and data of the previous assessments (NEFSC 2010).

TOR 1. Update catch estimates from all sources including landings and discards. Characterize the uncertainty in these sources of data.

Data for 2010 and 2011 were added to the catch time series in the assessment (complete data for 2012 were not yet available). Due to changes in software and data, the previous time series (1980-2009) of discard estimates for both areas were revised. The revisions resulted in higher estimates of discards in the south and an increase in the proportion of small fish in the discard and catch in the south during 2000-2009. Changes to the historical data in the north were minimal.

Landings and catch during 2010 and 2011 remained at relatively low levels in the north and increased slightly in the south. The catch-length frequency in recent years did not expand to larger sizes, which might have been expected while catches have been relatively low.

Estimation of total catch for monkfish has several sources of uncertainty. Before 1980, fishery removals were primarily bycatch, but most were unreported. Therefore, evaluation of fishery development is difficult, leading to problems interpreting the state of the resource in the early years of the marketed fishery. Since 1980, the quality of landings estimates generally increased, but the series includes under-reporting and difficulties converting landed products to live weight.

There is no information on the magnitude of discards prior to 1989. Recent assessments have assumed that discard rates before 1989 were similar to discard:kept ratios observed in later years; this may be problematic if discard rates were lower in later years because markets had developed. The quality of discard data generally increased in the 1989-2009 observer time series as a result of increasingly greater coverage of fleets and improved protocols, but there were some unsampled portions of the fishery (e.g., some half-year periods in which entire gear-types were not sampled).

TOR 2. Update fishery-independent indices used as inputs in the last assessment model. Characterize uncertainty and any bias in these sources of data.

All survey series used in the assessment models were updated through 2011, which was the most recent year with complete data available.

Within the northern management area, broad trends in stock size were consistent among the five surveys conducted there. Biomass fluctuated without trend from 1963 to the early 1980s, but declined thereafter to near historic lows during the 1990s, when landings reached their peak. Biomass indices increased from 2000 to 2004, but then decreased and have remained at lower levels since then. Abundance indices in the north fluctuated without trend during 1963-1998 but spiked during 2000-2002, reflecting a strong 1999 year class.

General trends in survey indices in the southern area are also consistent among surveys. Survey biomass and abundance indices were high during the mid-1960s, fluctuated around an intermediate level during the 1970s and mid-1980s, then declined to low levels since the late 1980s. Biomass indices increased slightly around 2002 but have returned to lower levels since then.

Size-based indices of abundance indicate relatively strong recruitment in the northern area during the 1990s and in several recent years, and variable but stable recruitment in the south. Length distributions gradually became truncated from the 1960s to early 1990s, and the median size of monkfish in survey catches has remained fairly constant since the early 1990s.

TOR 3. Update the SCALE model for monkfish to estimate fishing mortality, recruitment and stock biomass (total and spawning stock) and their uncertainty. Include a historical retrospective analysis to allow a comparison with previous assessment results.

The SCALE models for both management areas were updated with two additional years of data and the revised catch data for 1980-2009. The basic configuration of the models was not changed. Retrospective patterns were estimated based on 7 peels.

The SCALE models for the north changed little with the revised data and additional years of data. For the north, estimated F in 2011was 0.08 (retrospective bias -54%, corrected F_{2011}=0.16), estimated total biomass was 60,500 mt (retrospective bias +87%, corrected total biomass = 32,390 mt). Estimated age-1 recruitment in 2011 was 11.7 million fish, near the time series low (retrospective bias +23%, corrected age-1 recruitment = 9.5 million). Spawning biomass continued to increase in the northern management area.

The SCALE model for the south changed somewhat with the revised data and additional two years. The increased proportion of small fish in the revised catch data caused a shift in estimated selectivity so that the final model estimated only one selectivity time block (vs. two blocks in the 2010 SAW 50 assessment). The re-estimated time series of F, biomass, and recruitment using the single selectivity block in the south were similar to the estimates from SAW 50. For the south, estimated F in 2011was 0.11 (retrospective bias -22%, corrected F_{2011}=0.14), estimated

total biomass was 111,100 mt (retrospective bias +24%, corrected total biomass=88,806 mt). Estimated age-1 recruitment in 2011 was 23.3 million fish, near the time series low (retrospective bias +50%, corrected age-1 recruitment=15.3 million). Spawning biomass continued to increase through 2010, but in 2011 showed a slight downturn.

The SCALE model results for monkfish continue to be subject to high levels of uncertainty due to weaknesses in input data, such as under-reported landings and unknown discards during the 1980s; incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality, sex ratios and stock structure; and the relatively short reference time frame of the model (no information prior to 1980). Further, both models have difficulty fitting the catch-length frequencies in some years, with substantial overestimates of the numbers of large fish in the stock. The recent retrospective patterns have improved in the north since the 2010 assessment, but optimistic retrospective patterns remain in both areas (F underestimated, biomass overestimated) and are pronounced in the northern area.

TOR 4. Update biological reference points as needed and evaluate stock status to determine if the stock is overfished and if overfishing is occurring. Provide estimates of uncertainty.

Reference points were updated using the revised selectivity estimates from the SCALE models. The following table gives the reference points for each management area. Reference points were not adjusted for retrospective patterns.

North	BRP	Basis	SAW 50 (2010)	2013 Update
	Fmax	Age-based YPR	0.43	0.44
	Bthreshold	0.5*Bmax Projected	26,465	23,037
	Btarget	Bmax Projected	52,930	46,074
	MSY	Fmax Projected	10,745	9,383
South				
	Fmax	Age-based YPR	0.46	0.37
	Bthreshold	0.5*Bmax Projected	37,245	35,834
	Btarget	Bmax Projected	74,490	71,667
	MSY	Fmax Projected	15,279	14,328

In the north, F_{max} (F threshold) changed only slightly (SAW 50 F_{max}=0.43, 2013 update F_{max}=0.44). In the south, F_{max} under the single selectivity block was estimated as 0.37 (SAW 50

F_{max}=0.46). Given the current estimates of F from the SCALE models, overfishing is not occurring in either management area.

Biomass reference points based on long-term projections of total biomass at F_{max} were recommended in the SAW 50 assessment, adopted for management in 2012, and updated in the current assessment. Given the current estimates of biomass from the SCALE models, monkfish are not overfished in either management area.

The BRPs for monkfish are based on output from the SCALE model, which is subject to high levels of uncertainty as discussed under TOR 3; therefore the BRPs are also highly uncertain.

TOR 5. Summarize sources of data, model and reference point uncertainty relevant to setting Acceptable Biological Catch limits.

The SCALE model results for monkfish continue to be subject to high levels of uncertainty due to weaknesses in input data such as: under-reported landings and unknown discards during the 1980s; incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality, sex ratios and stock structure; and the relatively short reference time frame (1980-2011) of the model. Further, both models have difficulty fitting the catch length frequencies in some years, with substantial overestimates of the numbers of large fish in the stock. The retrospective patterns have improved in the north since the 2010 assessment, but optimistic retrospective patterns remain in both areas (F underestimated, biomass overestimated).

The BRPs use output from the SCALE model, which is subject to high levels of uncertainty as discussed under TOR 3; therefore the BRPs are also highly uncertain.

TOR 6. Perform short-term (3 year) projections for stock biomass under alternative harvest strategies.

SCALE model results and AGEPRO projections were used to predict stock trends during 2014-2016 under two scenarios: $F=F_{threshold}$ assuming stochastic long-term recruitment (using both unadjusted and retrospective-adjusted SCALE outputs), and status quo F (unadjusted 2011 F estimated from SCALE) assuming stochastic long-term recruitment.

For both areas, fishing at $F_{threshold}$ led to declines in total stock biomass in the unadjusted and retrospective-adjusted runs. In the north, total stock biomass increased during 2012-2016 under $F_{status\ quo}$, while in the south, total stock biomass decreased during 2012-2016 under $F_{status\ quo}$.

The projections for both areas have a high degree of uncertainty due to uncertainty in the starting conditions (output from the SCALE model).

TOR 7. Should the baseline model fail when applied in the operational assessment, provide guidance on how stock status might be evaluated. Should an alternative assessment approach not be readily available, provide guidance on the type of scientific and management advice that can be. An underlying premise of operational assessment

is to minimize the number of significant changes in methodology that would likely require a more detailed peer review.

The baseline model performed similarly to previously accepted versions of the model; therefore, despite its high uncertainty, it was not considered to have failed.

TOR 8. If feasible, present preliminary results from ongoing research projects and indicate how they could impact future assessments.

Studies are currently underway to investigate growth and migration patterns of monkfish. Results are too preliminary and incomplete to include in depth.

Introduction

Life History

Monkfish (*Lophius americanus*), also called goosefish, are distributed in the Northwest Atlantic, from the Grand Banks and northern Gulf of St. Lawrence south to Cape Hatteras, North Carolina (Collette and Klein-Macphee 2002). Monkfish may be found from inshore areas to depths of at least 900 m (500 fathoms). Seasonal onshore-offshore migrations occur and appear to be related to spawning and possibly food availability (Collette and Klein-MacPhee 2002).

Monkfish rest partially buried on soft bottom substrates, and attract prey using a modified first dorsal fin ray that resembles a fishing pole and lure. Monkfish are piscivorous and commonly eat prey as large as themselves. Despite the behavior of monkfish as a demersal 'sit-and-wait' predator, recent information from electronic tagging suggests seasonal off-bottom movements (Rountree et al. 2006). Growth is rapid at about 10 cm per year, and is similar for both sexes up to age 6 and lengths of around 60 cm (Richards et al. 2008). Few males are found older than age 7, but females can live to 12-14 years or older. Tagging studies underway suggest that growth patterns may differ between males and females (Richards et al. 2012); however, relatively few tags have been recaptured to date, and the information is insufficient to support revising the growth assumptions in the assessment at this time. Monkfish as large as 138 cm have been captured in NEFSC bottom trawl surveys.

Female monkfish begin to mature at age 4 and 50% of females are mature by age 4.7 (about 41 cm). Males mature at slightly younger ages and smaller sizes (50% maturity at age 4.2 or 37 cm (NEFSC 2002; Richards et al. 2008). Spawning takes place from spring through early autumn, progressing from south to north, with most spawning occurring during the spring and early summer. Females lay a buoyant mucoid egg raft or veil which can be as large as 12 m long and 1.5 m wide and only a few mm thick. The eggs are arranged in a single layer in the veil, and the larvae hatch after about 1-3 weeks, depending on water temperature. The larvae and juveniles spend several months in a pelagic phase before settling to a benthic existence at a size of about 8 cm (Collette and Klein-MacPhee 2002).

Stock Identification

The Fishery Management Plan (FMP) defines two management areas for monkfish (northern and southern), divided roughly by a line bisecting Georges Bank (Figure 1). The two assessment and management areas for monkfish were defined in the 1999 FMP based on differences in temporal patterns of recruitment (estimated from NEFSC surveys), perceived differences in growth patterns, and differences in the contribution of fishing gear types (mainly trawl, gill net, and dredge) to the landings.

Genetic studies suggest a homogeneous population of monkfish off the U.S. east coast (Chikarmane et al. 2000). Monkfish larvae are distributed over deep (< 300 m) offshore waters of the Mid-Atlantic Bight in March-April, and across the continental shelf (30 to 90 m) later in the year, but relatively few larvae have been sampled in the northern management area (Steimle et al. 1999). NEFSC surveys continue to indicate different recruitment patterns in the two management units in recent years.

The perceived differences in growth in the two management areas were based on studies about 10 years apart and under different stock conditions (Armstrong et al. 1992: Georges Bank to Mid-Atlantic Bight, 1982-1985; Hartley 1995: Gulf of Maine, 1992-1993). Age, growth, and maturity information from the NEFSC surveys and the 2001, 2004, and 2009 cooperative monkfish surveys indicated only minor differences in age, growth, and maturity between the areas (Richards et al. 2008; Johnson et al. 2008). The recent biological evidence (growth, maturity, and genetic information) suggests that use of a single stock hypothesis in the assessment might be appropriate. However, substantial differences in the fisheries exist, and current management maintains separate regulatory areas to accommodate these differences.

The southern deepwater extent of the range of American monkfish (*L. americanus*) overlaps with the northern extent of the range of blackfin monkfish (*L. gastrophysus*; Caruso 1983). These two species are morphologically similar, which may create a problem in identification of survey catches and landings from the southern extent of the range of monkfish. The potential for a problem, however, is believed to be small. The NEFSC closely examined winter and spring 2000 survey catches for the presence of blackfin monkfish and found none. The cooperative monkfish survey conducted in 2001 caught only eight blackfin monkfish of a total of 6,364 monkfish captured in the southern management area.

Fisheries Management

Commercial fisheries for monkfish occur year-round using gillnets, trawls, and scallop dredges. No significant recreational fishery exists. The primary monkfish products are tails, livers and whole gutted fish. Peak fishing activity occurs during November through June, and value of the catch is highest in the fall due to the high quality of livers during this season.

U.S. fisheries for monkfish are managed in the Exclusive Economic Zone (EEZ) through a joint New England Fishery Management Council - Mid-Atlantic Fishery Management Council Monkfish Fishery Management Plan (FMP). The primary goals of the Monkfish FMP are to end and prevent overfishing and to optimize yield and economic benefits to various fishing sectors involved with the monkfish fisheries (NEFMC and MAFMC 1998; Haring and Maguire 2008). Current regulatory measures vary with type of permit but include limited access, limitations on days at sea, mesh size restrictions, trip limits, minimum size limits, and annual catch limits (Tables 1 and 2).

Biological reference points for monkfish were established in the original Fishery Management Plan (FMP), but were revised after SAW 34 (NEFSC 2002), after the Data Poor Stocks Working Group (DPSWG) in 2007 (NEFSC 2007a), and after SAW 50 in 2010. The overfishing definition is F_{max}. Prior to 2007, $B_{threshold}$ was defined as one-half of the median of the 1965-1981 3-year average NEFSC autumn trawl survey catch (kg) per tow. After acceptance of an analytical assessment in 2007 (NEFSC 2007a), B_{target} was redefined as the average of total biomass for the model time period (1980-2006), and $B_{threshold}$ as the lowest observed value in the total biomass time series from which the stock had then increased (termed "B_{Loss}"). According to the earlier (survey index-based) reference points, monkfish were overfished and overfishing status could not be determined (NEFSC 2005); however, with adoption of the analytical assessment in 2007, monkfish status was changed to no longer overfished and overfishing was not occurring. SAW 50 in 2010 also concluded that both stocks were not overfished and overfishing was not occurring, while recognizing the continuing significant uncertainty in the determination.

2007 DPSWG Assessment

The DPSWG accepted a length-tuned analytical model (SCALE) for monkfish assessment and status determination, and adopted a value for natural mortality (M) of M=0.3. However, the WG emphasized that the assessment was highly uncertain due to under-reported landings; unknown discards during the 1980s; incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality, and stock structure; the shorter reference time frame (1980-2006) than in previous assessments (1963-2006); and the relatively recent development of the assessment model. The WG also concluded that uncertainties in historical catch data precluded application of long-term models that rely on episodes of depletion and recovery to estimate stock size.

2010 SAW 50 Assessment

The 2010 Southern Demersal Working Group (SDWG) updated the SCALE model to assess the status of monkfish using data through 2009. Further developments included examination of retrospective patterns in the SCALE estimates, and development of short-term stochastic age-based projections. Data from a cooperative monkfish survey conducted during winter/spring of 2009 were analyzed and included in the assessment model, along with data collected on the new NEFSC survey vessel, starting in spring 2009, which was adjusted using calibration coefficients developed for monkfish. Length frequency composition data from the 2009 cooperative survey were included in the final SCALE assessment model. The SARC 50 panel decided against making an adjustment for the retospective pattern in the assessment.

TOR 1. Update catch estimates from all sources including landings and discards. Characterize the uncertainty in these sources of data.

Landings

Landings statistics for monkfish are sensitive to conversion from landed weight to live weight, because a substantial fraction of the landings occur as tails only (or other parts). The conversion

of landed weight of tails to live weight of monkfish in the NEFSC weigh-out database is made by multiplying landed tail weight by a factor of 3.32. In 2012, the dealer database for 2005-2011 was corrected because some dealers were reporting 'head-on, gutted' monkfish (conversion factor of 1.14) as 'round' (no conversion). This resulted in a 1.5% overestimate of monkfish landings (live weight) during those years (all years combined), which has now been corrected. Early catch statistics (before ~1980) are uncertain, because much of the monkfish catch was sold outside of the dealer system or used for personal consumption until the mid-1970s. For 1964 through 1989, there are two potential sources of landings information for monkfish: the NEFSC 'weigh-out' database, which consists of fish dealer reports of landings; and the 'general canvass' database, which contains landings data collected by NMFS port agents (for ports not included in the weigh-out system) or reported by states not included in the weigh-out system (Table 3). All landings of monkfish are reported in the general canvass data as 'unclassified tails.' Consequently, some landed weight attributable to livers or whole fish in the canvass data may be inappropriately converted to live weight. This is not an issue for 1964-1981, when only tails were recorded in both databases. For 1982-1989, the weigh-out database contains market category information that allows for improved conversions from landed to live weight. The two data sources produce the same trends in landings, with general canvass landings slightly greater than weigh-out landings. It is not known which of the two measures more accurately reflects landings, but the additional data sources suggest that the general canvass is most reliable for 1964-1981 landings, whereas the availability of market category details suggests that the weigh-out database is most reliable for 1982-1989.

Beginning in 1990, most of the extra sources of landings in the general canvass database were incorporated into the NEFSC weigh-out database. However, North Carolina reported landings of monkfish to the Southeast Fisheries Science Center, and until 1997 these landings were not added to the NEFSC general canvass database. Since these landings most likely come from the southern management area, they have been added to the weigh-out data for the southern management area for 1977-1997 for the landings statistics used for stock assessment.

Beginning in July 1994, the NEFSC commercial landings data collection system was redesigned to consist of vessel trip reports (VTR) and dealer weigh-out records. The VTRs include area fished for each trip, which is used to apportion dealer-reported landings to statistical areas. The northern management area includes statistical areas 511-515, 521-523, and 561; the southern management area includes areas 525-526, 562, 537-543, and 611-636 (Figure 1). Each VTR trip should have a direct match in the dealer data base, but this is not always true. VTR records with no matching dealer landings were excluded, but dealer landings with no matching VTR were included in landings statistics, apportioning the unmatched landings to management area using proportions calculated from matched trips pooled over gear, state, and quarter.

Total U.S. landings (live weight) remained at low levels until the middle 1970s, increasing less than 1,000 mt to around 6,000 mt in 1978 (Table 3; Figure 2). Annual landings remained stable at between 8,000 and 10,000 mt until the late 1980s. Landings increased from the late 1980s to

over 20,000 mt per year 1992-2004, peaking at 28,500 mt in 1997. Landings declined steadily after 2003, and stabilized around an average of 8,300 mt during 2009-2011. During recent years (2008-2011 North; 2009-2011 South), fishing year landings have been below the TAC (Table 2).

By region, landings began to increase in the north in the mid-1970s, and began to increase in the south in the late 1970s. Most of the increase in landings during the late 1980s through mid-1990s was from the southern area. Historical under-reporting of landings shold be considered in the interpretation of this series.

Trawls, scallop dredges and gill nets are the primary gear types that land monkfish (Table 4; Figure 3). Trawls have contributed approximately half of the landings since 1964. Prior to 1994, gillnets contributed less than 10% of total landings, but landings from gillnets have generally increased, and accounted for almost 40% of landings in the past decade (2002-2011). Monkfish landings from the scallop dredge fishery have declined to about 9% in the past decade, primarily due to regulatory changes.

Until the late 1990s, total landings were dominated by landings of monkfish tails. From 1964 to 1980 landings of tails rose from 19mt to 2,302mt, and peaked at 7,191mt in 1997 (Table 5). Landings of tails declined after 1997, but are still an important component of the landings. Landings of gutted whole fish have increased steadily since the early 1990s, and are now the largest market category on a landed-weight basis. On a regional basis, more tails were landed from the northern area than the southern area prior to the late 1970s (Tables 5 and 6). From 1979 to 1989, landings of tails were about equal from both areas. In the 1990s, landings of tails from the south predominated, but since 2000, landings of tails have been greater in the north.

Beginning in 1982, several market categories were added to the system (Table 5). Tails were broken down into large (> 2.0 lbs), small (0.5 to 2.0 lbs), and unclassified categories, and the liver market category was added. In 1989, unclassified round fish were added; in 1991, peewee tails (<0.5 lbs) and cheeks; in 1992, belly flaps; and in 1993, whole gutted fish were added. Monkfish livers have become a very valuable product. Landings of livers increased from 10mt in 1982 to an average of over 600mt during 1998-2000. During 1982-1994, ex-vessel prices for livers rose from an average of $0.97/lb to over $5.00/lb, with seasonal variations as high as $19.00/lb. Landings of unclassified round (whole) or gutted whole fish jumped in 1994 to 2,045mt and 1,454mt, respectively; landings of gutted fish continued to increase through 2003. The tonnage of peewee tails landed increased through 1995 to 364mt and then declined to 153mt in 1999 and 4mt in 2000 when the category was essentially eliminated by regulations.

Foreign Landings

Landings (live wt) from NAFO areas 5 and 6 by countries other than the US are shown in Table 3 and Figure 2. Reported landings were high but variable in the 1960s and 1970s, with a peak in 1973 of 6,818mt. Landings were low but variable in the 1980s, declined in the early 1990s, and

have generally been below 300mt in recent years. There were no updated NAFO data available for monkfish for this assessment update.

Discard Estimates

Catch data from the fishery observer, dealer, and VTR databases were used to investigate discarding frequencies and rates. The number of trips with monkfish discards available for analysis varied widely among management areas and gear types (Tables 7 and 8). As in previous monkfish assessments (NEFSC 2007a; NEFSC 2010), monkfish discards were estimated on a gear, half-year, and management area basis, using observed discard-per-kept-monkfish to expanded to total discards for otter trawls and gillnets, and observed discard-per-all-kept-catch to expanded for scallop dredges and shrimp trawls. As before, discards for 1980-1988 (before observer sampling) were estimated by applying average discard ratios by management area and gear type (trawl, shrimp trawl, gillnet, dredge) from 1989-1991 to landings for 1980-1988 as follows:

Area	**Shrimp Trawls**	**Trawls**	**Gillnets**	**Dredges**
North				
Years included	1989-1991	1989-1991	1989-1991	1992-1997
Number of trips	124	253	1191	54
South				
Years included	n/a	1989-1991	1991-1992	1991-1993
Number of trips		334	177	32

Methods for estimating discards were changed slightly from previous assessments, and the time series of estimates for all gears in both areas were re-estimated. The revisions were prompted by advances in standardized SBRM methodology (Rago et al. 2005; Wigley et al. 2007), which have rendered obsolete the earlier SAS programs used for monkfish. In the current assessment, dealer landings were used to expand the d/k ratios; assessements in 2007 and 2010 used landings from vessel trip reports (VTRs). Some additional program and gear codes were included in the current assessment (e.g. observer training trips, haddock separator trawl, Rhule trawl). The most significant impacts of the changes were the inclusion of more observed trips since 2004 for trawlers, and changes to the estimated kept-all (raising factor) for scallop dredges in both areas (see Figures 1 and 2 in Appendix I). In general, d/k ratios remained similar to previous estimates despite the higher sample sizes. Revised estimates of discards were slightly lower in the north and slightly higher in the south (Appendix I, Figures 3 and 4) during 1980-2002, but were higher thereafter in the south due primarily to the change in raising factor for scallop dredges.

The proportion of discards in the northern area catch was about 13% in the 1980s, 7% during 2002-2006, became slightly higher on average (12%) during 2007-2009, and was 14% for 2010-2011 (Table 9; Figures 4 and 5). The proportion of discards in the southern area catch generally increased since the 1980s (average 16% 1980-1989), with an annual average of 29% during 2002-2006, 24% during 2007-2009, and 28% in 2010-2011 (Table 9; Figures 4 and 5). Gill nets consistently have had the lowest discard ratios. Some of the trends in discarding may reflect imposition of size limits starting in 2000 and decreased trip limits in the south starting in 2002. The DPWG (NEFSC 2007a) noted a potential bias in discard estimates due to increased observer sampling in the multispecies groundfish fishery. Monkfish discard rates may differ between the directed monkfish fisheries and bycatch fisheries. The most frequent discard reasons were that fish were too small for regulations or the market. The estimates of total catch for 1980-2011 are shown in Figure 5 and Table 9.

Size Composition of U.S. Catch

Tail lengths were converted to total lengths using relations developed by Almeida et al. (1995). As in NEFSC (2007a), length composition of landings and discard were estimated from fishery observer samples by management area, year, gear-type (trawls, dredges and gillnets), and catch disposition (kept or discarded). Landings in unknown gear categories were allocated proportionately to the 3 major gear types before assigning lengths. The stratification used for assigning lengths within area and gear type is shown in Table 10. The estimated length composition of landings and discard is shown in Figures 6-9. Size composition was re-estimated for 1994-2009 (all available years) because of the updates to the discard estimates. There were minor changes in the estimated length composition for 1994-2006 due to an error discovered after the SAW 50 review (NEFSC 2007a) (gillnet discard lengths in the south characterized using kept lengths) and because different blocks of years/areas were used in some cases when data were sparse. Length composition was estimated for 2010-2011 using the same methods applied to the earlier data.

Age composition of the catch was not estimated due to uncertainties in the aging method that were highlighted in previous assessments (NEFSC 2007a; NEFSC 2010) and because the operational model for monkfish (SCALE) is length-based.

Effort and CPUE

Evaluating trends in effort or catch rates in the monkfish fishery is difficult for several reasons. Much of the catch is taken in multi-species fisheries, and defining targeted monkfish trips is difficult. There have been programmatic changes in data collection from port interviews (1980-1993) to logbooks (1994-2009), and comparison of effort statistics among programs is difficult. Catch rates may not reflect patterns of abundance, because they have been affected by regulatory changes (e.g., 1994 - closed areas; 2000 - trip limits; 2006 - reductions in trip limits).

CPUE data have not been used in the assessment model for monkfish, therefore they were not examined for this assessment update.

TOR 2. Update fishery-independent indices used as inputs in the last assessment model. Characterize uncertainty and any bias in these sources of data.

Resource surveys used in the 2010 assessment models were updated. Surveys included in the 2013 assessment update were 2001, 2004, and 2009 cooperative monkfish surveys; NEFSC winter, spring, and autumn offshore surveys; NEFSC scallop surveys (SFMA only); Northern Shrimp Technical Committee (NSTC) shrimp surveys (NFMA only); and ME/NH inshore surveys.

The NEFSC survey strata used to define the northern and southern management areas are:

Survey	Northern Area	Southern Area
NEFSC Offshore bottom trawl	20-30, 34-40	1-19, 61-76
NSTC Shrimp	1,3,5-8	
Shellfish		6,7,10,11,14,15,18,19,22-31,33-35,46,47,55,58-61,621,631

NEFSC spring and autumn bottom trawl survey indices for 1963-2008 were standardized to adjust for statistically significant effects of trawl type (Sissenwine and Bowman 1977) on catch rates. The trawl conversion coefficients apply only to the spring survey during 1973-1981.

NEFSC indices derived from surveys on the FSV *Henry B. Bigelow* (starting spring 2009) were adjusted using calibration coefficients estimated during experimental work (Miller et al. 2009). The FSV *Henry B. Bigelow*, which became the main platform for NEFSC research surveys in spring 2009, has significantly different size, towing power, and fishing gear characteristics than the previous survey platform (R/V *Albatross IV)*, resulting in different fishing power and catchability for most species. Calibration experiments to estimate these differences were conducted during 2008 (Brown 2009; NEFSC 2007b), and were peer reviewed by a panel of three non-NMFS scientists during the summer of 2009 (Anonymous 2009). The objective was to develop specific protocols for guidance in the selection and use of appropriate estimators based on the amount of data available and the relative performance of two candidate estimators. The Panel developed general guidance on which estimator to use given sample sizes for each species. Following these guidelines, monkfish catches were converted using a simple ratio estimator without a seasonal (spring vs. fall) or length-specific correction. The low catch rates of monkfish in the *Albatross* series made development of more detailed coefficients infesible. The overall coefficients for monkfish were 7.1295 for numbers and 8.0618 for biomass (kg) (Anonymous 2009; Miller et al. 2009).

Coefficients of variation and confidence intervals for all survey indices are given in the tables for each survey and region discussed below.

Northern Area

Biomass indices from NEFSC autumn research trawl surveys fluctuated without trend between 1963 and 1975, increased briefly in the late 1970s, but declined thereafter to near historic lows during the 1990s (Table 11; Figure 10). From 2000 to 2003 the index increased, reflecting recruitment of a relatively strong 1999 yearclass. Subsequently, the biomass index declined and has remained low since. In the unconverted *Bigelow* time series (2009-2012, Figure 11), biomass and abundance indices in the north have generally increased.

Indices from the NEFSC spring research trawl surveys reflect similar trends of relatively high biomass levels in the mid 1970s (but with possible declines in the late 1970s); a declining trend from the early 1980s to the lowest values in the time series in 1998; an increase to relatively high biomass from 2001 to 2005; and somewhat lower levels since then (Table 12, Figure 10). The spring *Bigelow* indices (Figure 11) increased during 2009-2011, but declined in 2012.

Survey length distributions have become increasingly truncated over time (Figure 12). By 1990, fish greater than 60 cm long were uncommon. The minimum, median, and maximum lengths in the trawl surveys declined during the 1980s and have fluctuated around smaller sizes since ~1990 (Figure 13). Several modes potentially representing strong yearclasses have appeared in survey length distributions in recent years (Figure 12). However, despite relatively low exploitation in recent years, there is little evidence of increased abundance of large individuals in the survey catches.

Abundance indices were estimated for monkfish of lengths corresponding to ages 1 and 2 for input to the assessment model (Figure 19). To the extent that these indices reflect recruitment, recruitment in the northern area has increased in the past decade. Survey abundance at length and at age suggests relatively strong 1993, 1999 and possibly 2006 yearclasses in the northern area. Survey age data are available for 1993-2006 from the autumn trawl survey and for 1995-2006 for the spring trawl survey (NEFSC 2007a).

Other surveys which catch monkfish in portions of the northern area include the ASMFC shrimp survey, the Massachusetts Division of Marine Fisheries fall and spring surveys, and ME/NH inshore surveys (Figures 10, 14, and 15). The shrimp survey samples the western Gulf of Maine during summer and caught more monkfish than the spring or fall surveys prior to 2009 (when the FSV *Bigelow* survey series began) (Table 13; Figure 10). Patterns of abundance and biomass have been relatively consistent among the spring, fall, ME-NH, and shrimp surveys (Figure 10). The Massachusetts surveys catch few monkfish and were not considered to reflect patterns of abundance for the entire management area (NEFSC 2007a); therefore they have not been used in recent assessments.

Southern Area

Biomass indices from the NEFSC autumn research survey were high during the mid-1960s, fluctuated around an intermediate level during the 1970s-mid 1980s, then declined to consistently low levels since the late 1980s (Table 15; Figure 16). NEFSC spring surveys reflect

similar trends as the autumn series: biomass remained fairly high during the mid 1970s-early 1980s, but fluctuated around lower levels thereafter (Table 16; Figure 16). A spike in biomass was observed in 2003, but subsequent indices have returned to lower values. Biomass and abundance indices based on the NEFSC winter flatfish survey (conducted during 1992-2007) fluctuated without trend (Table 17; Figure 16). Although the winter survey series had a short duration, the gear used in the winter survey was more effective for capturing monkfish than the gear used in autumn or spring surveys. Abundance indices based on the NEFSC sea scallop survey have fluctuated widely and have been at relatively low levels since 2007 (Table 18; Figure 16).

Inconsistent geographic coverage should be considered in the interpretation of southern survey indices. For example, the fall survey did not sample southern strata until 1967. The winter survey sampled Georges Bank inconsistently and did not sample deep strata before 1998. The scallop survey does not currently sample the entire southern management area, and the timing of this survey has shifted in recent years from mid-summer to late spring.

Abundance (numbers per tow) shows trends similar to biomass, with a spike in 1972, fluctuations around a relatively low level since the mid-1970s, and a slight increase in 2002 and 2003 followed by a return to lower levels. Length distributions from the southern area showed truncation over time but somewhat less dramatically than in the north (Figure 17). As in the northern area, fish greater than 60 cm have been rare since the 1980s, especially when compared to the 1960s. Any recent strong recruitment does not appear to remain in the system long enough to contribute substantially to increased stock biomass. Survey age data are available for 1993-2006 from the autumn trawl survey, 1995-2006 for the spring trawl survey, and 1997-2007 for the winter trawl survey (NEFSC 2007a). Age samples collected since 2006 survey have not been processed due to uncertainties regarding validity of the aging method (NEFSC 2007a).

TOR 3. Update the SCALE model for monkfish to estimate fishing mortality, recruitment and stock biomass (total and spawning stock) and their uncertainty. Include an historical retrospective analysis to allow a comparison with previous assessment results.

Several modeling approaches were investigated by the Data Poor Stocks Working Group (NEFSC 2007a), but the only approach considered suitable was a relatively new one called SCALE (for Statistical Catch-At-Length Analysis). SCALE models were used in 2007 to estimate fishing mortality, recruitment and stock biomass, and to re-define reference points. The SCALE models were updated for SAW 50 (NEFSC 2010) and are updated again for the current assessment.

Monkfish SCALE Model

Introduction

Incomplete or lacking age-specific catch data and survey indices often limit the application of a full age-structured assessment (e.g. Virtual Population Analysis and many forward-projecting age-structured models). Stock assessments often rely on the simpler size/age aggregated models (e.g. surplus production models) when age-specific information is lacking. However, these models may not utilize all of the available information for a stock assessment. Knowledge of a species growth and lifespan, along with total catch data, size composition of the removals, recruitment indices, and indices on numbers and size composition of the recruited fish in a survey, can provide insights on population status using a simple model framework.

The Statistical Catch At Length (SCALE) model, is a forward-projecting age-structured model tuned with total catch (mt), catch at length or proportional catch at length, recruitment at a specified age (usually estimated from first length mode in the survey), survey indices of abundance of the larger/older fish (usually adult fish), and the survey length frequency distributions. The SCALE model was developed in the AD model builder framework. The model parameter estimates are fishing mortality and recruitment in each year, fishing mortality to produce the initial population (F_{start}), logistic selectivity parameters for each year or blocks of years, and Qs for each survey index.

The SCALE model was developed as an age-structured model that does not rely on age-specific information on a yearly basis. The model is designed to fit length information, abundance indices, and recruitment at age which can be estimated by using survey length slicing. However, the model does require an accurate representation of the average overall growth of the population, which is input to the model as mean lengths at age. Growth can be modeled as sex-specific growth and natural mortality, or growth and natural mortality can be modeled with the sexes combined. The SCALE model will allow for missing data.

Model Configuration

The SCALE model assumes growth follows the mean input length at age with predetermined input error in length at age. Therefore, a growth model or estimates of the average mean length at age is essential for reliable results. The model assumes static growth; therefore, population mean length/weight at age is assumed constant over time.

The SCALE model estimates logistic parameters for a flattop selectivity curve at length in each time block specified by the user for the calculation of population and catch age-length matrices, or the user can input fixed logistic selectivity parameters. Presently the SCALE model cannot accommodate a dome shaped selectivity pattern.

The SCALE model computes an initial age-length population matrix in year one of the model as follows. First, the estimated population numbers at age starting with age-1 recruitment are normally distributed at 1 cm length intervals, using mean length at age with the assumed

standard deviation. Next, the initial population numbers at age are calculated from the previous age at length abundance using the survival equation. An estimated fishing mortality (F_{start}) is also used to produce the initial population. This F can be thought of as the average fishing mortality that occurred before the first year in the model. Now the process repeats itself, with the total estimated abundance at age being redistributed according to the mean length at age and standard deviation in the next age (age+1).

This two-step process is used to incorporate the effects of length-specific selectivities and fishing mortality. The initial population length and age distribution is constructed by assuming population equilibrium with an initial value of F, called F_{start}. Length-specific mortality is estimated as a two-step process in which the population is first decremented for the length specific effects of mortality as follows:

$$N^{*}_{a,len,y_1} = N_{a-1,len,y_1} e^{-\left(PR_{len}F_{start}+M\right)}$$

In the second step, the total population of survivors is then redistributed over the lengths at age *a* by assuming that the proportions of numbers at length at age *a* follow a normal distribution with a mean length derived from the input growth curve (mean lengths at age).

$$N_{a,len,y_1} = \pi_{len,a} \sum_{len=0}^{L_\infty} N^{*}_{a,len,y_1}$$

where

$$\pi_{len,a} = \Phi\left(len+1 \mid \mu_a, \sigma_a^2\right) - \Phi\left(len \mid \mu_a, \sigma_a^2\right)$$

where

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

Mean lengths at age can be calculated from a von Bertalanffy model from a prior study as shown in the equation above, or mean lengths at age can be calculated directly from an age-length key. Variation in length at age a = σ_s^2 can often be approximated empirically from the growth study used for the estimation of mean lengths at age. If large differences in growth exist between the sexes, then growth can be input as sex-specific growth with sex-specific natural mortality. However, catch and survey data are still fitted with sexes combined.

This SCALE model formulation does not explicitly track the dynamics of length groups across age, because the consequences of differential survival at length at age do not alter the mean length of fish at age a+1. However, it does realistically account for the variations in age-specific partial recruitment patterns by incorporating the expected distribution of lengths at age.

In the next step, the population numbers at age and length for years after the calculation of the initial population use the previous age and year for the estimate of abundance. Here, the calculations are done on a cohort basis. As in the previous initial population survival equation, the partial recruitment is estimated on a length vector.

$$N^{*}_{a,len,y} = N_{a-1,len,y-1} e^{-\left(PR_{len} F_{y-1} + M\right)}$$

Second stage:

$$N_{a,len,y} = \pi_{len,a} \sum_{len=0}^{L_{\infty}} N^{*}_{a,len,y}$$

Constant M is assumed along with an estimated length-weight relationship to convert estimated catch in numbers to catch in weight. The standard Baranov's catch equation is used to remove the catch from the population in estimating fishing mortality.

$$C_{y,a,len} = \frac{N_{y,a,len} F_y PR_{len} \left(1 - e^{-\left(F_y PR_{len} + M\right)}\right)}{(F_y \quad PR_{len}) + M}$$

Catch is converted to yield by assuming a time invariant average weight at length.

$$Y_{y,a,len} = C_{y,a,len} W_{len}$$

The SCALE model results in the calculation of population and catch age-length matrices for the starting population and then for each year thereafter. The model is programmed to estimate recruitment in year 1 and estimate variation in recruitment relative to recruitment in year 1 for each year thereafter. Estimated recruitment in year one can be thought of as the estimated average long term recruitment in the population since it produces the initial population. The residual sum of squares of the variation in recruitment $\sum(\text{Vrec})^2$ is then used as a component of the total objective function. The weight on the recruitment variation component of the objective

function (Vrec) can be used to penalize the model for estimating large changes in recruitment relative to estimated recruitment in year one.

The model requires an age-1 recruitment index for tuning, or the user can assume relatively constant recruitment over time by using a high weight on Vrec. Usually there is little overlap in ages at length for fish that are one and/or two years of age in a survey of abundance. The first mode in a survey can generally index age-1 recruitment using length slicing. In addition, numbers and the length frequency of the larger fish (adult fish) in a survey where overlap in ages at a particular length occurs can be used for tuning population abundance. The model tunes to the catch and survey length frequency data using a multinomial distribution. The user specifies the minimum size (cm) for the model to fit. Different minimum sizes can be fit for the catch and survey data length frequencies.

The number of parameters estimated is equal to the number of years in estimating F and recruitment plus one for the F to produce the initial population (F_{start}), logistic selectivity parameters for each year or blocks of years, and for each survey Q. The total likelihood function to be minimized is made up of likelihood components comprising fits to the catch, catch length frequencies, the recruitment variation penalty, each recruitment index, each adult index, and adult survey length frequencies:

$$\mathrm{L}_{\mathrm{catch}} = \sum_{years}\left(l\,\mathrm{n}(Y_{\mathrm{obs,y}} + 1) - \ln\left(\sum_{a}\sum_{len} \mathrm{Y}_{\mathrm{pred,len,a,y}} + 1 \right)\right)^2$$

$$L_{catch_lf} = -N_{eff}\sum_{y}\left(\sum_{inlen}^{L_\infty}\left(\left(C_{y,len}+1\right)\ln\left(1+\sum_{a} C_{pred,y,a,len}\right) - \ln\left(C_{y,len}+1\right)\right)\right)$$

$$\mathrm{L}_{\mathrm{vrec}} = \sum_{y=2}^{Nyears}\left(Vrec_y\right)^2 = \sum_{y=2}^{Nyears}\left(R_1 - R_y\right)^2$$

$$\sum L_{rec} = \sum_{i=1}^{Nrec}\left[\sum_{y}^{Nyears}\left(\ln\left(I_{rec_i,inage_i,y}\right) - \ln\left(\sum_{len}^{L_\infty} N_{y,inage_i,len} * q_{rec\,i}\right)\right)^2\right]$$

$$\sum L_{adult} = \sum_{i=1}^{Nadult} \left[\sum_{y}^{Nyears} \left(\ln\left(I_{adult_i, inlen+_i, y}\right) - \left(\sum_{a} \sum_{inlen_i}^{L_\infty} \ln\left(N_{pred,y,a,len} * q_{adult\,i}\right) \right) \right)^2 \right]$$

$$\sum L_{lf} = \sum_{i=1}^{Nlf} \left[-N_{eff} \sum_{y} \left(\sum_{inlen_i}^{L_\infty} \left(\left(I_{lf_i,y,len} + 1\right) \ln\left(1 + \sum_{a} N_{pred,y,a,len}\right) - \ln\left(I_{lf_i,y,len} + 1\right) \right) \right) \right]$$

In equation L_{catch_lf}, calculation of the sum of length is made from the user input specified catch length to the maximum length for fitting the catch. Input user-specified fits are indicated with the prefix "in" in the equations. LF indicates fits to length frequencies. In equation L_{rec}, the input specified recruitment age; in L_{adult} and L_{lf}, the input survey specified lengths up to the maximum length is used in the calculation.

$$Obj\ \ fcn = \sum_{i=1}^{N} \lambda_i L_i$$

Lambdas represent the weights to be set by the user for each likelihood component in the total objective function.

Monkfish SCALE Model Configuration and Results

The SCALE model was updated in the current assessment using revised catch numbers and catch weights in both management areas for 1994-2009 (see Appendix I for detailed discussion of data revisions), and two additional years of data (2010, 2011). Complete data for 2012 were not available for this assessment update, so the terminal year was 2011.

No conclusive new information on growth and natural mortality was available for this assessment, and assumptions of growth, variation in mean length at age, and natural mortality (M=0.3) were the same as those used in the 2007 and 2010 assessments (NEFSC 2007a; NEFSC 2010). Mean and variance in monkfish length at age were estimated from industry-based surveys (2001 and 2004), and NEFSC winter, spring, and fall surveys for management areas combined (Table 19). No significant differences in growth were observed between the management units in the 2001 and 2004 cooperative surveys. The standard deviation for age 1 was 2.9; for older ages a standard deviation of 4.5 was assumed. The overall standard deviation on mean lengths at age was estimated directly from the age data. The oldest aged fish from surveys and commercial samples was age 12. Mean lengths at age for the older fish (10-12) was supplemented with data collected from a study of large monkfish (Johnson et al. 2008).

Age modes in the predicted length frequencies are seen for most ages, due to the linear nature of monkfish growth and the model structure that uses a single annual growth time step. The absence

of a decline in growth with age in monkfish produces this process error in the SCALE model fits. This can be concealed by increasing the variance on mean lengths at age by increasing the assumed variance on the mean lengths at age. However, as in the previous assessments, an increase in the variance on the mean lengths at age beyond what is supported by the raw growth data was avoided due to concerns about its effect on the estimated selectivity.

Relative abundance trends for recruits (ages 1, 2, and/or 3) and adults (40+ cm) in each management unit were updated and are shown in Figure 20. For both management units, the model was fit to spring, fall, and industry-based survey length frequencies (30+ cm), 40+ cm adult indices, and recruitment indices at age. The northern area had additional inputs from the ASMFC summer shrimp trawl survey (1991-2011) and the ME-NH fall inshore trawl survey (2000-2011). The southern area had additional information from the NEFSC winter trawl (1992-2007) and NEFSC scallop dredge (1984-2011) surveys (Figure 20). Survey abundance indices were scaled using the approximate size (nm^2) of the survey area divided by the average coverage of the survey's tow (Table 20). The survey catchability estimates from the model were used as a diagnostic check for the interpretation of survey efficiencies. Survey indices from the R/V *Bigelow* (2009-2011) were converted to *Albatross* units by dividing *Bigelow* numbers per tow by the conversion coefficient described above (7.2).

For this assessment update, the model configurations used in SAW 50 (NEFSC 2010) were adhered to as closely as possible. For the northern stock, two runs are described (Table 21; Figures 21-26). Run 1 is a repeat of the 2010 model (1980-2009) using the revised catch data to show the impact of the data revisions (Figure 21). Run 2 uses the revised catch data plus an additional two years of data (1980-2011, Figure 22), and is considered the final run for the north. In the south (Table 22; Figures 27-33), run 1 was also a repeat of the 2010 model using revised data and run 2 used the revised data plus two new years of data. Run 3 in the south estimated only one selectivity block (vs. two blocks estimated in 2010) and is considered the final run for the south.

In the past and in the current assessments using SCALE, a single selectivity block was estimated for the northern management unit. In the south, three selectivity blocks were estimated in 2007, two blocks in 2010, and in the current assessment only one block in the final run. The change from three to two blocks in 2010 provided a better fit to the catch length frequency data and corresponded to a shift to more gillnet gear in the southern fishery. One selectivity block was used in the current assessment because the revised data for the catch time series showed more small fish than in the earlier data, and the most recent two years of data (2010, 2011) have relatively high numbers of small fish in the catch (Figure 9) primarily from discards in the scallop dredge sector (Figure 7c). The change in the selectivity due to the revised data is shown in Figure 27, and the change due to revised data plus two additional years of data is shown in Figure 28. The difference in selectivity blocks 1 and 2 in the south was minimal, with the revised data and 2 additional years (Figure 29).

As in previous SCALE-based assessments, models for both the north and south had difficulty in fitting the catch length frequency data, and in particular overestimated the abundance of larger fish. This pattern was seen in the final few years in the north, and more persistently in the south (Figures 25 and 32). A possible explanation for this might be a mis-specification of growth. Currently accepted growth models are linear (Richards et al 2008) and linear growth is used in SCALE. However, there are suggestions from tagging studies that growth may slow at older ages, at least for males (Richards et al. 2012). If growth does slow, using a constant growth increment would lead to overestimation of numbers at length of large fish by SCALE. A further factor may be the recent decline in catches in both areas without a concomitant expansion of length frequencies in the catch or surveys. The model may not have been able to reconcile the effects of a decline in catch with the lack of a corresponding shift in the length distributions.

The final run for the north (run 2) estimated higher F and lower biomass than the SAW 50 assessment did, but very little change in selectivity (Table 22; Figure 22). For 2004-2009, the annual F estimate was on average 47% higher than was estimated in 2010 and total biomass was on average 20% lower. Some of this is likely due to the strong retrospective pattern observed in 2010 (see below for further discussion). Regardless, the model for the north estimates terminal F to be near the lowest in the model time series, and terminal biomass to be increasing from a low point in 2006. The estimates of age 1 recruitment suggest strong recruitment pulses in 1993 and 2000 (1992 and 1999 yearclasses), but no major recruitment events since then. The northern model estimated lower total biomass in the terminal year than was projected for 2011 from the 2010 assessment: 81,900 tons projected in 2010 versus 60,500 tons estimated for 2011 in the current assessment.

The final run for the south (run 3) (Table 23) estimated similar F and biomass as the SAW 50 assessment, despite the revised data and change from two to one selectivity block (Figure 29). For 2006-2009, the annual F estimate was on average 12% higher than estimated in 2010 and total biomass was on average 3% lower. The model for the south estimates terminal F to be increasing slightly and terminal biomass to be decreasing slightly. The estimates of age 1 recruitment have fluctuated widely, but have been near the time series low since 2005, with a slight increase in the 2011 estimate. The southern model estimated lower total biomass in the terminal year than was projected for 2011 from the 2010 assessment: 132,200 tons projected in 2010 versus 111,100 tons estimated for 2011 in the current assessment.

Monkfish SCALE Model Uncertainty

Assessment of monkfish is difficult because of the often poor quality of data or lack of data. Survey data provide a long-term picture, but there is high variability in the survey trends due to the low numbers of fish caught in many of the surveys. Landings were historically under-reported, and discard data were not available until relatively recently, and length composition of discards even more recently. Age samples were not taken in surveys until 1994 and from landings until 2000; landings are sparsely sampled for age because removing vertebrae compromises product quality, and even if there were samples, significant questions have been

raised about the aging method, which has not been validated using known-age individuals. Important aspects of monkfish biology are poorly understood, including stock structure and movement patterns, growth rates, and longevity. Effects of process error within the model due to the linear growth trend are unknown. Uncertainty surrounds the lack of an explanation for the consistent sex ratio patterns that occur with size in multiple surveys (Richards et al. 2008).

Given the litany of data limitations, it is not surprising that most of the assessment approaches applied during the 2007 Data Poor Stocks Working Group assessment were not successful. The SCALE model was considered useful at that assessment because it integrated the available information and the resulting estimates appeared reasonable (e.g. biomass estimates consistent with empirically-estimated biomass from industry-based surveys). This remained true in the 2010 assessment and the current assessment. However, substantial uncertainty remains surrounding the lack of evidence for rebuilding of the size structure with the observed decline in the catch.

Retrospective patterns in the current model for the north are somewhat less severe than in previous assessments (Figure 26), suggesting that the strong 1999 yearclass may have contributed to the retrospective pattern in the north. However, retrospective underestimation of F and overestimation of biomass continues to be severe, based on the average of 7 peels. If the fishing mortality estimated for 2011 is adjusted upward to account for the average retrospective under-estimation of -54% for the 2004-2010 terminal years, the estimate for 2011 changes from 0.08 to 0.16. If the total biomass estimated for 2011 is adjusted downward to account for the average retrospective overestimation of 87% for the 2004-2010 terminal years, the estimate for 2011 changes from 60,485 mt to 32,390 mt.

The model for the southern area exhibits less severe retrospective patterns than the north; however, the retrospective errors in fishing mortality and stock size increased slightly for the south with this model update (Figure 33). If the fishing mortality estimated for 2011 is adjusted upward to account for the average retrospective underestimation of -23% for the 2004-2010 terminal years, the estimate for 2011 changes from 0.11 to 0.14. If the total biomass estimated for 2011 is adjusted downward to account for the average retrospective overestimation of +25% for the 2004-2010 terminal years, the estimate for 2011 changes from 111,100 mt to 88,806 mt. Age-specific retrospective adjustments using seven peels are summarized in Table 24.

As a further diagnostic, estimates of total biomass based on converting SCALE output numbers at length (30+ cm) to biomass using the length-weight relationship were compared with biomass estimated by applying the length composition from NEFSC *Bigelow* fall and spring surveys to the estimated total number (30+) from SCALE and then converting to biomass. In the north, the estimates from the two methods did not diverge greatly (Figure 34A.); however, in the south the biomass estimates derived by applying the survey length were about half that estimated using the model-estimated length composition (Figure 34B). In a similar analysis of the 2011 estimated

and observed catch in the south, 33% of the estimated catch (mt) was over 90 cm, whereas only 3% of the observed catch was over 90 cm (Figure 34C).

Potential explanations for the lack of fit and/or retrospective pattern in the SCALE model were explored in SAW 50. The explanations deemed most likely to cause underlying problems with the model were (1) the growth model being incorrect (i.e., if growth is not linear with age) and (2) setting M=0.3 may be inappropriate (i.e., monkfish longevity may be greater than currently assumed). Although studies are underway to investigate growth and migration of monkfish, there are insufficient results at present to provide further clarification on these issues.

Improvements to the SCALE model since 2007 allow for estimation of within-model uncertainty on fishery selectivity and stock numbers through the MCMC procedure. However, uncertainty in F could not be estimated with the MCMC for monkfish because fishing mortality is set equal to model results in the MCMC. Therefore, all of the within-model uncertainty is not accounted for in the MCMC results. The high uncertainty surrounding this assessment will be largely underestimated by within model uncertainty estimates and probably should not be solely used for the determination of the uncertainty in setting ABCs.

Spawning biomass is not output directly by the SCALE model, but was estimated as the product of population numbers at length (SCALE), maturity at length (Richards et al. 2008), weight at length (SCALE), and fraction female at length (based on data in Richards et al. 2008). Trends in spawning biomass are shown in Figure 35. In the north, estimates of spawning biomass have been increasing since 2006, while in the south spawning biomass had been increasing since the late 1990s but showed a slight downturn in the terminal year of the model (2011).

TOR 4. Update biological reference points as needed and evaluate stock status to determine if the stock is overfished and if overfishing is occurring. Provide estimates of uncertainty.

Overfishing Reference Points

SAW 34 (NEFSC 2002) and Framework 2 of the Monkfish FMP established the overfishing definition as F_{max} and estimated it be equal to 0.2 for both management areas (assuming M=0.2). NEFSC (2007a) examined length-based and age-based YPR models and concluded that the length-based approach was not appropriate as it assumes a von Bertalanffy growth model which does not fit currently understood monkfish growth patterns. NEFSC (2007a and 2010) used the age-based YPR model to update the value of F_{max} assuming M=0.3, and Framework Adjustment 7 of the monkfish FMP adopted this approach for use in management in 2011 (Table 1). The current assessment updates the age-based YPR model using revised selectivity patterns output from SCALE. F_{target} was not defined in the original monkfish FMP or in Framework Adjustments 2 or 7. The DPWSG (NEFSC 2007a) recommended that $F_{40\%}$ be used to define $F_{target;}$ however, this has not yet been formally adopted by management.

Age-based YPR was calculated for each management region using the approach of NEFSC (2007a). This assumed a constant natural mortality M=0.3 and applied selectivity at age approximated from SCALE output selectivity at length for each area. Mean weights at age for the catch and stock were from SCALE output, and maturity ogives were from the 2001 Cooperative Monkfish Survey data (NEFSC 2002), which were very similar to other estimates of maturity (NEFSC 2007a). The estimates from NEFSC (2007a; 2010) and the current assessment are shown in Table 25, and the updated yield curves in Figure 36. The difference in estimates for the two areas reflects differing selectivity estimated for the two areas. The differences between years reflect the changes in selectivity patterns estimated by the SCALE model, especially in the south.

The updated estimates of F_{max} are 0.44 in the northern area and 0.37 in the southern area. Unadjusted estimates of current F (2011) are 0.08 in the northern area and 0.11 in the southern area, both less than the respective overfishing thresholds (Figure 37).

Biomass Reference Points

In the 2010 assessment, recommended biomass reference points were estimated based on long-term projected biomass corresponding to F_{MSY} or its proxy (= F_{max} for monkfish). The recommended reference points were subsequently adopted in Framework Adjustment 7 to the Monkfish FMP. Total biomass targets (i.e., B_{max} at F_{max}) and thresholds ($0.5*B_{max}$) calculated in 2010 and from the current assessment are shown in Table 26. Current estimates of B_{target} are 46,074 mt in the northern area and 71,667 mt in the southern area, and estimates of $B_{threshold}$ are 23,037 mt in the northern area and 35,834 mt in the southern area. The total catch produced from the long-term B_{target} at the respective values of F_{max} (i.e., proxy for F_{MSY}) is 9,383 mt for the northern area and 14,328 mt for the southern area.

All of the BRPs are based on results of the SCALE model (including F reference points from the YPR which uses selectivity curves estimated by SCALE); therefore, the BRPs are subject to the same high level of uncertainty that surrounds the SCALE model results. Further, the BRPs based on projected biomass at F_{max} are subject to high uncertainty, due to reliance on projections of SCALE model results and the high estimate of F_{max} due to the assumption of M=0.3 in the YPR model.

Using the biological reference points recommended in the 2010 stock assessment and adopted in 2011, the current assessment indicates that monkfish are not overfished with no overfishing occurring in both the northern and southern stock management areas (Figure 37). These determinations are considered highly uncertain due to the many uncertainties in the assessment model upon which they are based. A comparison of the current estimates of reference points with those estimated in the last two assessments is given in Table 26.

TOR 5. Summarize sources of data, model and reference point uncertainty relevant to setting Acceptable Biological Catch limits.

The assessment results for monkfish continue to be uncertain, due to likely under-reported landings and unknown discards during the 1980s and incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality, and stock structure. The population models for both areas exhibit retrospective patterns which are stronger in the north than the south (Figures 26 and 33); however, there appears to be stronger bias in the results in the south than in the north (Figure 34). The BRPs are based on output from the SCALE model; therefore, the BRPs are also highly uncertain.

TOR 6. Perform short-term (3 year) projections for stock biomass under alternative harvest strategies.

SCALE model results and AGEPRO projections were used to predict stock trends during 2013-2016 under two scenarios: $F=F_{threshold}$ assuming stochastic long-term recruitment (using both unadjusted and adjusted SCALE outputs); and status quo F (unadjusted 2011 F estimated from SCALE) assuming stochastic long-term recruitment (Table 27).

For both areas, fishing at $F_{threshold}$ led to declines in total stock biomass in the unadjusted and retrospective-adjusted runs. In the north, total stock biomass increased during 2013-2016 under $F_{status\ quo}$, while in the south, total stock biomass decreased during 2013-2016 under $F_{status\ quo}$.

The projections for both areas have a high degree of uncertainty due to uncertainty in the starting conditions (output from the SCALE model).

Table 28 compares the projected biomass from the SAW 50 models (ACT scenarios) to the current model estimates of biomass for 2010 and 2011 for both management areas.

TOR 7. Should the baseline model fail when applied in the operational assessment, provide guidance on how stock status might be evaluated. Should an alternative assessment approach not be readily available, provide guidance on the type of scientific and management advice that can be. An underlying premise of operational assessment is to minimize the number of significant changes in methodology that would likely require a more detailed peer review.

The baseline models for both monkfish management areas performed similarly to the two previous assessments that were accepted for use in management. Therefore the baseline model was expected to be adequate to guide management under the same terms as in previous years (i.e., with great caution considering the uncertainties underlying the model).

TOR 8. If feasible, present preliminary results from ongoing research projects and indicate how they could impact future assessments.

Figure 40 shows preliminary results from an ongoing archival tagging study of monkfish in which recaptured fish with tags are returned to the investigators (Richards et al. 2012). The

recaptured monkfish were 54-77 cm TL at release. The tag returns are few, but suggest the possibility of differences in growth between male and female monkfish. The two females that 'shrank' were at large for a short time and the apparent shrinkage may represent measurement error on the live fish (and poor health of one of the recaptured fish).

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