Atlantic States Marine Fisheries Commission

2019 Atlantic Menhaden Ecological Reference Point Stock Assessment Report

Vision: Sustainably Managing Atlantic Coastal Fisheries

Atlantic States Marine Fisheries Commission

2019 Atlantic Menhaden Ecological Reference Point Benchmark Stock Assessment

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EXECUTIVE SUMMARY

The impact of forage species harvest on predator species and the larger ecosystem has received increasing attention in recent years. Much of this work has concluded that forage fisheries should be managed more conservatively than single-species reference points would suggest. However, much of that work has also been conducted outside the North West Atlantic Continental Shelf ecosystem. The North West Atlantic Continental Shelf ecosystem is complex, with numerous predators and prey overlapping spatially, temporally, and trophically.

Atlantic menhaden have supported one of the largest fisheries in the U.S. since colonial times. The vast majority of landings are turned into fish meal and fish oil for use in a variety of products, and a smaller component is used as bait for other commercial and recreational fisheries. Atlantic menhaden are also an important food source for a wide range of species in the North West Atlantic Continental Shelf ecosystem, including larger fish such as striped bass and bluefin tuna, birds such as bald eagles and osprey, and marine mammals like humpback whales and bottlenose dolphin. Many of these predators support valuable commercial and recreational fisheries or ecotourism industries, in addition to having cultural value.

Managers and stakeholders have expressed strong interest in managing Atlantic menhaden in an ecosystem context. In 2015, the Atlantic States Marine Fisheries Commission held an Ecosystem Management Objectives (EMO) Workshop with managers, scientists, and stakeholders to identify fundamental ecosystem management objectives for Atlantic menhaden. The objectives included sustaining Atlantic menhaden to provide for directed fisheries, sustaining Atlantic menhaden to provide for predators, providing stability for all types of fisheries, and minimizing the risk to sustainability due to a changing environment.

Models and Data

The Commission's Ecological Reference Point Workgroup (ERP WG) was tasked with developing reference points for management use that could account for Atlantic menhaden's role as a forage fish throughout its range. To accomplish this, the ERP WG explored a suite of models to develop ecological reference points and estimate population parameters for Atlantic menhaden. These approaches ranged from simple, with minimal data requirements and few assumptions, to complex, with extensive data needs and detailed assumptions. The approaches included: a time-varying intrinsic growth rate (*r*) surplus production model, a Steele-Henderson surplus production model, a multispecies statistical catch-at-age model, a moderate complexity Ecopath with Ecosim (EwE) model with a limited predator/prey field, and a full ecosystem EwE model.

A suite of five key predator and prey species were identified from diet data and other considerations (referred to as ERP focal species). Atlantic striped bass, bluefish, spiny dogfish, and weakfish were identified as key predator species of Atlantic menhaden. Weakfish was included as both an Atlantic menhaden predator and a prey item for the other predators. Atlantic herring was included as a key alternative prey to Atlantic menhaden for the predators identified. The Steele-Henderson surplus production model explored each of the ERP focal predators, resulting in a base model that included only Atlantic menhaden and striped bass. The multispecies statistical catch-at-age and the two EwE models included all of the ERP focal species. The intermediate complexity EwE included a few additional trophic groups, while the full EwE incorporated a large number of additional species and groups.

The ERP models were parameterized with the best available data for Atlantic menhaden and the ERP focal species. For Atlantic menhaden, data from the single-species benchmark assessment conducted in parallel with this assessment were used. All ERP focal species had recently undergone benchmark assessments or assessment updates which included the time series of new Marine Recreational Information Program (MRIP) estimates of recreational catch. All ERP focal species had life history, landings, and index data available through 2017, as well as estimates of fishing mortality and population size. Newer data were not available for all of the groups included in the full EwE; as a result, inputs for those groups were extrapolated from the previously published full EwE model, which had a terminal year of 2013.

In addition to the single-species assessment inputs, the ERP WG examined a range of diet datasets – from individual, small-scale studies to larger scale, long-term monitoring programs – to parameterize the multispecies models. The proportion of Atlantic menhaden in the diets of key predators varied by season, location, and age class of predators sampled. The main sources of diet data included the Northeast Area Monitoring and Assessment Program (NEAMAP), the Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP), and the Northeast Fisheries Science Center Food Habits Database (NEFSC FHD). These programs covered a fairly large proportion of the Atlantic coastal shelf and provided ten to thirty years of diet data collected with consistent methodologies. However, sample sizes often precluded analyses on finer spatial or temporal scales. These databases focused on finfish and shellfish species, not birds or marine mammals. Smaller scale studies were used to supplement the data from these long-term programs for some of the modeling approaches, especially for species that were not well represented in the long-term programs.

Model Results and Comparisons

The ERP WG evaluated the performance of these models, their strengths and weaknesses, and their ability to inform the fundamental ecosystem management objectives identified by the EMO Workshop. To meet the ecosystem management objectives, the models needed to be able to assess both top-down effects of predation on Atlantic menhaden and bottom-up effects of Atlantic menhaden biomass levels on predators in order to quantify tradeoffs between management objectives. The EwE models were the only models that were able to evaluate both factors. The surplus production model with time-varying *r* only estimated changes in productivity without attributing them to a particular cause. The Steele-Henderson model included the effect of striped bass predation on Atlantic menhaden, but did not have a feedback mechanism to predict the effect of Atlantic menhaden harvest on striped bass biomass. Similarly, the current implementation of the multispecies statistical catch-at-age explored here lacked the bottom-up feedback necessary to explore trade-offs between Atlantic menhaden harvest and predator biomass.

The ERP models agreed about the overall trend of Atlantic menhaden population size and exploitation rates over the last 30 years, indicating biomass was increasing and exploitation rate was decreasing. These trends and the magnitude of the estimates were also consistent with the estimates from the single-species assessment. This was not surprising, as all of the ERP models used the same time-series of total removals, life history parameters, and indices of abundance as the single-species model. In addition, the EwE models used some outputs from the single-species model directly as inputs.

All of the ERP models produced MSY- or MSY-proxy reference points. Those reference points were calculated from the current ecosystem conditions, i.e., the estimate of productivity or predator consumption levels from the terminal year of each model. However, these reference point estimates may not meet the management objectives for the ecosystem, because several of the predators included in the ERP models were in an overfished state in the terminal year of the models.

ERP Targets and Thresholds

To establish reference points for Atlantic menhaden that take into account their role as forage fish, the ERP WG recommended using the intermediate complexity EwE model in conjunction with the Atlantic menhaden single-species assessment model.

This approach combined the individual strengths of each model. The single-species model provided the best information on Atlantic menhaden population size and fishing mortality, as it included more detail on size and age structure, fishery selectivity, and recruitment variability than the EwE models. The EwE models provided an evaluation of the impact of proposed harvest scenarios on important predator species in the long-term, which the single-species model could not do.

The intermediate complexity EwE was chosen over the full EwE because the full EwE model results suggested that the reduced predator set of the intermediate complexity EwE model captured the dynamics of the more responsive predators from the full ecosystem model. Striped bass and nearshore piscivorous birds were the most sensitive species in the full EwE models, as they showed larger changes in biomass than other species did in response to increases or decreases in fishing pressure on Atlantic menhaden. The Atlantic menhaden harvest scenarios that sustain the biomass of predators included in the intermediate complexity EwE were thus expected to not cause large declines for other predators that were only included in the full EwE model. In addition, it would be feasible to update the intermediate complexity EwE model on a timeframe suitable for management. The full EwE model required extensive data from stock assessments and other sources for the large number of species and groups included in the model; as a result, updating the model would be a significant effort.

While the final values for the ERP target and threshold will be a management decision that takes into account the management objectives of both Atlantic menhaden and their predators, the tradeoffs between those management objectives can be evaluated with the ERP approach outlined here. To illustrate the potential use of the combined single-species assessment and intermediate complexity EwE model, the ERP WG put forward example values of an ERP target and an ERP threshold based on existing management objectives for striped bass. Striped bass was the focal species for this analysis because it was the most sensitive fish species to Atlantic menhaden *F*, and focusing on one key predator provided a more tractable example for evaluating tradeoffs among management strategies. Example ERPs based on striped bass biomass should not cause significant declines for other species that were less sensitive to levels of Atlantic menhaden removals.

Multiple combinations of *F* on striped bass and *F* on Atlantic menhaden could keep striped bass populations at their biomass target or threshold (Figure 144). The example ERP target was defined as the maximum *F* on Atlantic menhaden that would sustain striped bass at their biomass target when striped bass were fished at their *F* target. The example ERP threshold was defined as the maximum *F* on Atlantic menhaden that would keep striped bass at their biomass threshold when striped bass were fished at their *F* target. For the example analysis, all other species were fished at their current *F* rates.

The example ERP target and threshold were lower than the current single-species target and threshold (Figure 148). The example ERP target was estimated at a full *F* (i.e., maximum *F*-at-age from the intermediate complexity EwE model) of 0.188, compared to a full *F* of 0.314 for the single-species target. The example ERP threshold was estimated at a full *F* of 0.573, compared to a full *F* of 0.856 for the single-species threshold. The current estimate of full *F* from the single-species model is 0.157, below both the example ERP target and threshold.

This example was based on the *F* and *B* targets laid out in the striped bass fishery management plan. Higher or lower reference points for striped bass will result in higher or lower reference points for Atlantic menhaden. Similarly, this example maintained the other species at their current *F* rates; higher or lower *F* rates on other species would also result in different reference point values for Atlantic menhaden. Managers and stakeholders can evaluate the tradeoffs between Atlantic menhaden harvest, predator harvest, and resulting biomass for all modeled species quantitatively and transparently with this combination of models in order to set the final reference point values and total allowable catch.

Next Steps

This approach represents the first step towards a practical application of an ecosystem approach to fishery management. The ERP WG identified a number of research recommendations dealing with data collection, modeling, and the management process in order to improve the ERP assessment and move the ecosystem approach to management forward.

The ERP models developed for this assessment did not include spatial or seasonal dynamics. Incorporating finer scale dynamics would be possible for some of the models, but would require both additional work on model development and better data. Spatially and seasonally resolved data were lacking, making it difficult to parameterize and calibrate the models on that scale. The ERP WG recommended expanding the collection of diet and condition data along the

Atlantic coast to provide seasonally and regionally stratified annual, year-round monitoring of key predator diets. This would provide information on prey abundance and predator consumption. In addition, ERP WG recommended improving the collection of diet data and monitoring of population trends for non-finfish predators (e.g., birds, marine mammals) and data-poor prey species (e.g., bay anchovies, sand eels, benthic invertebrates) to better parameterize the full ecosystem models.

The ERP WG also recommended further development of the multispecies statistical catch-at-age and the EwE models. In addition to spatial and seasonal dynamics, further development of bottom-up feedback into the multispecies statistical catch-at-age model and stochastic recruitment dynamics into the EwE models would improve the understanding of the relative importance of fishing, trophic interactions, and recruitment dynamics on ecosystem dynamics.

The ERP WG recommended that the intermediate complexity EwE model should be updated in conjunction with the next single-species assessment update in approximately three years and that the next benchmark be conducted in six years in conjunction with the single-species benchmark stock assessment. The other models should be updated and reevaluated as part of the next benchmark assessment if sufficient progress has been made on the modeling research recommendations.

Currently, the timing of individual assessments or updates for Commission-managed species are set independently of each other. The ERP WG in conjunction with other technical groups can develop a timeline for Commission assessments to ensure the most up-to-date data are available for timely ERP assessment updates.

The ERP WG also requested to be tasked by the Atlantic Menhaden Management Board or the Commission's Policy Board with the development of a timeline and framework for continued deployment of ecosystem-based fishery management by the Commission. Atlantic menhaden and their key predators are currently managed by separate Boards within the Commission (and in some cases, in collaboration with NOAA Fisheries). This means that management objectives, including *F* and *B* targets for each species, are set independently of each other. For successful ecosystem-based fishery management, the discussion of trade-offs between Atlantic menhaden and their predators should occur across Boards in order to develop consistent management objectives for individual species and the ecosystem. This will require changes to the way the Commission has historically operated. The Commission also does not have explicit management objectives for species like marine mammals and seabirds. The development of clear, quantitative management objectives for this ecosystem and the evaluation of the trade-offs between Atlantic menhaden harvest and other species need to be a holistic process that engages all managers and stakeholders. The ERP WG recommended that a formal management strategy evaluation be part of this process to identify harvest strategies that will maximize the likelihood of achieving these ecosystem management objectives.

The ERP WG recognized that implementing reference points and tools to address ecosystem issues is a complex and multifaceted problem. The full implementation of ecosystem-based fisheries management will require significant process and cultural changes to fishery management beyond simply new reference points for Atlantic menhaden. However, these new reference point methods for Atlantic menhaden are a critical first step in that implementation. While the Commission continues to refine the ERP models, collect better data, and consider changes to its management structure and process, managers can set harvest strategies for Atlantic menhaden that take into account their role as forage fish in a transparent and quantitative way.

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TOR 1. Review and evaluate the fishery-dependent and fishery-independent data used in the Atlantic menhaden single-species assessment, and justify inclusion, elimination, or modification of those data sets.

The Atlantic menhaden data were thoroughly vetted by the Atlantic Menhaden Technical Committee (TC) and Stock Assessment Subcommittee (SAS), and the available datasets are described in the single-species assessment report. The fishery-dependent data for Atlantic menhaden were robust. The reduction fishery, which accounted for the majority of landings, was well-sampled and both total landings and age composition information were considered precise and reliable. The bait fisheries and the recreational fisheries were not as adequately sampled, and there was a higher degree of uncertainty in the total landings and the age composition information; however, as these fisheries made up only about 10% of total landings, they did not significantly increase the uncertainty of the overall fishery-dependent data used in the assessment.

The fishery-independent data for Atlantic menhaden were more limited and had more uncertainty. Several data sets were available for young-of-year (YOY) abundance indices, but few were long time series. The few long-term YOY indices of abundance that were available were all from a single region, the Chesapeake Bay; however, the Chesapeake Bay is one of the major nursery grounds for Atlantic menhaden. Additionally, several data sets were available to characterize age-1+, or adult, Atlantic menhaden relative abundance. Most surveys that encountered Atlantic menhaden were geographically limited (i.e., occurred in a single state or river/bay) and were not designed to capture menhaden specifically. The hierarchical method of combining multiple separate surveys into a single index of abundance helped overcome some of the geographical limitations. In addition, no SAS-accepted age data were available from the fishery-independent data sources, which increased uncertainty since several indices captured Atlantic menhaden outside the range of sizes seen in the fisheries.

The Ecological Reference Point Working Group (ERP WG) considered the data collected and developed through the single-species assessment to be the best available data for Atlantic menhaden, and used all datasets in the ecological reference point models, with two exceptions. The Southern Adult (SAD) was not used in the biomass dynamic models. Length analysis of the SAD index indicated the index was dominated by age-1 fish, which made it inappropriate for that type of model. The Northern Adult (NAD) and Mid-Atlantic Adult (MAD) indices had a broader size structure and were used in the biomass dynamic models. In addition, the WG accepted the reduction fishery CPUE (RCPUE) index as an index of abundance for use in the surplus production models. Although the WG recognized the SAS's concerns about the index, the long time series and the contrast it provided, which the surplus production models required, outweighed the potential biases.

TOR 2. Characterize precision and accuracy of additional fishery-dependent and fishery-independent data sets, including diet data, used in the ecological reference point models.

The ERP WG relied on the most recent single-species stock assessments wherever possible to provide fishery-dependent and fishery-independent datasets for non-menhaden species. The key predator and prey species identified for the intermediate complexity models (Atlantic herring, Atlantic striped bass, bluefish, spiny dogfish, and weakfish) all had data available through 2017 that had been prepared by the TC or SAS responsible for the single-species assessment. The full ecosystem model included the most recent data for the key predator and prey species, but used the older time series of data from the previously published version of the model for other species.

The key predator and prey species were chosen in part because of the quality of the data available for them. Four of the five species had peer-reviewed statistical catch-at-age models that include fishery-dependent and fishery-independent indices of abundance and reliable estimates of total removals. Spiny dogfish was the one exception; the spiny dogfish assessment was a swept-area biomass estimate from a trawl survey but did include reliable estimates of total catch. For other species, the data were less robust. Important prey items like bay anchovy and sand eels and important predators like birds and whales lacked traditional stock assessments and often did not have reliable estimates of total removals or population abundance or biomass.

The ERP WG examined a range of diet datasets, from individual small-scale studies to larger-scale, long-term monitoring programs. The proportion of Atlantic menhaden in the diets of key predators varied by season, location, and age-class of predators sampled. The main sources of diet data included the Northeast Area Monitoring and Assessment Program (NEAMAP), the Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP), and the Northeast Fisheries Science Center Food Habits Database (NEFSC FHD). These programs covered a fairly large proportion of the Atlantic coastal shelf, and provided ten to thirty years of diet data collected with consistent methodologies. The key predator and prey species were moderately well-represented in these databases, but sample sizes often precluded analyses on finer spatial or temporal scales. In addition, these databases focused on finfish and shellfish species, not birds or marine mammals. Smaller scale studies were used to supplement the data from these long-term programs for some of the modeling approaches, especially for species that were not well represented in the long-term programs.

TOR 3. Develop models used to estimate population parameters (e.g., F, biomass, abundance) of Atlantic menhaden that take into account Atlantic menhaden's role as a forage fish and analyze model performance.

The ERP WG explored a suite of models to develop ecological reference points and estimate population parameters for Atlantic menhaden, ranging from very simple with minimal data requirements and few assumptions about population drivers to very complex with extensive data needs and detailed assumptions about the mechanisms of population dynamics. These included two surplus production models (one that estimated a time-varying intrinsic growth rate and one that implemented the Steele-Henderson approach of including predator biomass as part of the modeling process), a multispecies statistical catch-at-age model, and two formulations of Ecopath with Ecosim (EwE), one of intermediate complexity with a limited predator/prey field and one with a full ecosystem.

TOR 4. Develop methods to determine reference points and total allowable catch for Atlantic menhaden that account for Atlantic menhaden's role as a forage fish.

To develop reference points and estimates of total allowable catch that meet the ecosystem management objectives, the models needed to be able to assess both top-down effects of predation on Atlantic menhaden and bottom-up effects of Atlantic menhaden biomass levels on predators. The EwE models were the only models that were able to evaluate both factors; the other explored here models only captured the effects of predation on Atlantic menhaden. Therefore, the ERP WG recommended an approach that combined the single-species assessment model with the intermediate complexity EwE model. The single-species model represented the best information on current Atlantic menhaden population dynamics, including estimates of abundance and fishing mortality, while the intermediate complexity EwE model provided a way to evaluate harvest strategies for Atlantic menhaden in an ecosystem context while still being tractable to update on a management timeline.

The final reference point values used in management will be set by the Atlantic Menhaden Management Board based on their evaluation of the tradeoffs between Atlantic menhaden harvest and predator management objectives; however, the ERP WG developed example ecological targets and thresholds for Atlantic menhaden as a proof-of-concept. Striped bass were found to be one of the most sensitive species across several models, so the ERP WG developed the example target and threshold based on the current striped bass management objectives, as laid out in the striped bass fishery management plan. The ERP target was defined as the maximum fishing mortality rate on Atlantic menhaden that would sustain striped bass at their biomass target when striped bass were fished at their F target; the ERP threshold was defined as the maximum fishing mortality rate on Atlantic menhaden that would keep striped bass at their biomass threshold when striped bass were fished at their *F* target. The single-species projection model would then be used to calculate a TAC based on the ERP target.

The example ERP target and threshold were lower than the current single-species target and threshold. The ERP target was estimated at a full *F* of 0.188, compared to a full *F* of 0.314 for the single-species target. The ERP threshold was estimated at a full *F* of 0.573, compared to a full *F* of 0.856 for the single-species threshold. This example is based on the F and B targets laid out in the striped bass fishery management plan. Higher or lower reference points for striped bass would result in higher or lower reference points for Atlantic menhaden. In addition, other species in the model were fished at their F_{2017} values; increasing or decreasing *F* on the other species would also result in different reference points for Atlantic menhaden.

TOR 5. State assumptions made for all population and reference point models and explain the likely effects of assumption violations on synthesis of input data and model outputs.

Each of the models explored had a different set of assumptions about population and ecosystem dynamics.

The surplus production model with time-varying *r* did not make any explicit assumptions about what was causing changes in productivity: potential factors like changes in M from predation or other sources and variability in recruitment were all combined into changes in *r.* The Steele-Henderson surplus production model assumed that all changes in productivity were driven by the fishery and the key predator species in the model; other sources of mortality were included in the estimate of *r*, but the estimate of *r* was not time-varying. Changes in productivity that result from variability in recruitment or changes in M due to other factors could be attributed to predation by modeled species.

The multispecies statistical-catch-at-age model assumed that changes in M over time are due to changes in predation mortality from modeled predators (M_2); M_2 is a function of predator biomass, diet composition, and consumption-to-biomass ratios. To simplify the ecosystem structure, a limited suite of predator and prey species was used. A pool of other, non-modeled prey biomass was assumed to be constant to allow for diverse predator diets, and an age-varying but time-constant level of M from other sources (M_0) was assumed to account for non-modeled predators. Unlike the surplus production models, the multispecies statistical catch-at-age model was able to track observed variability in recruitment by fitting to observed recruitment indices and age composition information. This implementation of the multispecies statistical catch-at-age model was focused on predator impacts on Atlantic menhaden abundance; it did not include bottom-up effects of Atlantic menhaden abundance on predator biomass.

The EwE models are comprised of two model frameworks: the Ecopath model, a static, mass-balance representation of the ecosystem, and Ecosim, where change in biomass is predicted as consumption minus losses to predation, fishing, and migration, with consumption modeled using foraging arena theory. The two formulations of EwE differed in how detailed the trophic structure of the models were; the intermediate complexity model included 17 trophic groups, while the full model included 61 trophic groups. Both models allowed for unexplained, non-modeled mortality in addition to explicit loss to predation and fishing. The EwE models allowed for both top-down impacts of predators on prey species, and bottom-up effects of prey availability on predator biomass. As configured, the EwE models assumed a stock-recruitment relationship existed for all species, and as a result, may overstate the impact of adult mortality on future population abundance for species where recruitment is more environmentally driven.

Modeling of environmental factors was limited by the poor understanding of the relationship between specific environmental drivers and recruitment and mortality. None of the models included explicit environmental drivers in the base model run.

None of the models included spatial dynamics; all data were pooled to a coastwide stock level. As a result, nuances of population dynamics at the regional scale may be lost.

TOR 6. Characterize uncertainty of model estimates and reference points.

Estimates of uncertainty for model parameters and reference points could not be directly compared across models because of differences in the way each model calculated and propagated uncertainty through the estimates. The major source of uncertainty for these models was from the input data and model structure, and these were explored through sensitivity analyses.

The surplus production models were both sensitive to the starting year of the model and the indices used. The magnitude of the estimates of population size and exploitation rate varied significantly between different runs; however, the overall trend and relative stock status (e.g., B/B_{MSY}) were similar across runs. This is a common result with surplus production models.

For the multispecies statistical catch-at-age model, uncertainty about diet data had the greatest effect on the prey species, while the run with the alternative indices had the greater effect on the predator species. The estimate of unexplained M (M_0) used in the model was also a source of uncertainty.

The most sensitive parameters in the EwE models were the vulnerability parameters, which describe the exchange rates of prey from non-vulnerable states into vulnerable foraging "arenas," where they can be consumed by predators. The diet data used as input also had an effect on model results, as with the multispecies statistical catch-at-age, especially in identifying the major predators on Atlantic menhaden. The implementation of EwE used for this assessment did not include the ability to propagate uncertainty in input data such as species or species group biomasses and exploitation rates through to the final population and reference points estimates, so that source of uncertainty has an unknown impact.

TOR 7. Evaluate stock status for Atlantic menhaden from recommended model(s) as related to the respective reference points (if available).

All of the models explored by the ERP WG agreed on the current status of Atlantic menhaden: in 2017, overfishing was not occurring and the stock was not overfished, even when Atlantic menhaden's role as a forage fish was taken into consideration. Current levels of Atlantic menhaden removals were unlikely to cause a decline in predator populations.

TOR 8. Compare trends in population parameters and reference points among proposed modeling approaches, including the results of the single-species benchmark assessment. If outcomes differ, discuss potential causes of observed discrepancies.

All of the ERP models explored here agreed with the single-species assessment model about the overall trend of Atlantic menhaden population size and exploitation rates over the last 30 years: a generally increasing trend in biomass and a decreasing trend in exploitation rate. This consistency in findings is not surprising, since all the ERP models used the same time-series of total removals, life history parameters, and indices of abundance as the single species model, and in some cases (the EwE models) used output from the single-species model directly.

The ERP models produced similar assessments of stock status to the single-species assessment results, which determined that Atlantic menhaden were not overfished and were not experiencing overfishing in 2017. Current levels of Atlantic menhaden removal were not projected to cause declines in predator biomass. However, the ERP models were also consistent in the finding that fishing Atlantic menhaden at the single-species threshold would cause declines in predator biomass or condition.

The example ERP target and threshold developed based on management objectives for striped bass were lower than the single-species *F* target and threshold, but the current *F* for Atlantic menhaden was below the ERP target and threshold as well.

TOR 9. If a minority report has been filed, explain majority reasoning against adopting approach suggested in that report. The minority report should explain reasoning against adopting approach suggested by the majority.
No minority report was filed.

TOR 10. Develop detailed short and long-term prioritized lists of recommendations for future research, data collection, and assessment methodology. Highlight improvements to be made by next benchmark review.
The ERP WG endorsed the research recommendations laid out in the single-species assessment to improve the understanding of Atlantic menhaden population dynamics, especially the recommendations to develop an Atlantic menhaden-specific coastwide fishery-independent index of adult abundance and to continue to investigate environmental covariates related to productivity and recruitment on a temporal and spatial scale.

In addition, the ERP WG identified several short- and long-term research recommendations to improve the ERP assessment in the future. This included enhanced collection of diet and condition data through geographically widespread, annual, year-round monitoring of selected predator diets stratified seasonally and regionally, as well as enhanced collection of abundance and life history data on species such as birds, marine mammals, and non-commercially important finfish and shellfish. Incorporating bottom-up effects into the multispecies statistical catch-at-age model would improve the utility of that model for management use. Better incorporation of stochastic recruitment dynamics into the EwE models would improve the understanding of the relative importance of fishing, trophic interactions, and recruitment dynamics on ecosystem dynamics.

In addition to data and modeling recommendations, the ERP WG also recommended socioeconomic research and management strategy evaluation be conducted. Establishing appropriate ecological reference points requires understanding the tradeoffs between species and stakeholders from a socioeconomic as well as biological standpoint.

TOR 11. Recommend timing of next benchmark assessment and intermediate updates, if necessary relative to biology and current management of the species.

The ERP WG recommended that the moderate complexity EwE model should be updated in conjunction with the next single-species assessment, and that the other models should be updated and reevaluated as part of the next benchmark assessment. The ERP WG recommended the next benchmark be conducted in six years if sufficient progress has been made on the modeling research recommendations.

TERMS OF REFERENCE

For the 2019 ASMFC Atlantic Menhaden
Ecological Reference Point Benchmark Stock Assessment and Peer-Review

Board Approved May 2018

Terms of Reference for the Ecological Reference Point Assessment

1. Review and evaluate the fishery-dependent and fishery-independent data used in the Atlantic menhaden single-species assessment, and justify inclusion, elimination, or modification of those data sets.

2. Characterize precision and accuracy of additional fishery-dependent and fishery-independent data sets, including diet data, used in the ecological reference point models.
    a. Provide descriptions of each data source (e.g., geographic location, sampling methodology, potential explanation for outlying or anomalous data)
    b. Describe calculation and potential standardization of abundance indices.
    c. Discuss trends and associated estimates of uncertainty (e.g., standard errors)
    d. Justify inclusion or elimination of available data sources.
    e. Discuss the effects of data strengths and weaknesses (e.g., temporal and spatial scale, gear selectivities, ageing accuracy, sample size) on model inputs and outputs.

3. Develop models used to estimate population parameters (e.g., F, biomass, abundance) of Atlantic menhaden that take into account Atlantic menhaden's role as a forage fish and analyze model performance.
    a. Briefly describe history of model usage, its theory and framework, and document associated peer-reviewed literature. If using a new model, test using simulated data.
    b. Justify choice of ecological factors (e.g., predator species, other prey species, environmental factors) as appropriate for each model
    c. Describe stability of model (e.g., ability to find a stable solution, invert Hessian)
    d. Justify choice of CVs, effective sample sizes, or likelihood weighting schemes as appropriate for each model.
    e. Perform sensitivity analyses, model diagnostics, and retrospective analyses as appropriate for each model.
    f. Clearly and thoroughly explain model strengths and limitations, including each model's capacity to account for environmental changes

4. Develop methods to determine reference points and total allowable catch for Atlantic menhaden that account for Atlantic menhaden's role as a forage fish.

5. State assumptions made for all population and reference point models and explain the

likely effects of assumption violations on synthesis of input data and model outputs.

6. Characterize uncertainty of model estimates and reference points.

7. Evaluate stock status for Atlantic menhaden from recommended model(s) as related to the respective reference points (if available).

8. Compare trends in population parameters and reference points among proposed modeling approaches, including the results of the single-species benchmark assessment. If outcomes differ, discuss potential causes of observed discrepancies.

9. If a minority report has been filed, explain majority reasoning against adopting approach suggested in that report. The minority report should explain reasoning against adopting approach suggested by the majority.

10. Develop detailed short and long-term prioritized lists of recommendations for future research, data collection, and assessment methodology. Highlight improvements to be made by next benchmark review.

11. Recommend timing of next benchmark assessment and intermediate updates, if necessary relative to biology and current management of the species.

Terms of Reference for the Ecological Reference Point External Peer Review

1. Evaluate the justification for the inclusion, elimination, or modification of data from the Atlantic menhaden single-species benchmark assessment.

2. Evaluate the thoroughness of data collection and the presentation and treatment of additional fishery-dependent and fishery-independent data sets in the assessment, including but not limited to:
    a. Presentation of data source variance (e.g., standard errors).
    b. Justification for inclusion or elimination of available data sources,
    c. Consideration of data strengths and weaknesses (e.g., temporal and spatial scale, gear selectivities, aging accuracy, sample size),
    d. Calculation and/or standardization of abundance indices.

3. Evaluate the methods and models used to estimate Atlantic menhaden population parameters (e.g., F, biomass, abundance) that take into account Atlantic menhaden's role as a forage fish, including but not limited to:
    a. Evaluate the choice and justification of the recommended model(s). Was the most appropriate model (or model averaging approach) chosen given available data and life history of the species?
    b. If multiple models were considered, evaluate the analysts' explanation of any differences in results.
    c. Evaluate model parameterization and specification as appropriate for each

model (e.g., choice of CVs, effective sample sizes, likelihood weighting schemes, calculation/specification of M, stock-recruitment relationship, choice of time-varying parameters, choice of ecological factors).

4. Evaluate the methods used to estimate reference points and total allowable catch.

5. Evaluate the diagnostic analyses performed as appropriate to each model, including but not limited to:
    d. Sensitivity analyses to determine model stability and potential consequences of major model assumptions
    e. Retrospective analysis

6. Evaluate the methods used to characterize uncertainty in estimated parameters. Ensure that the implications of uncertainty in technical conclusions are clearly stated.
7. If a minority report has been filed, review minority opinion and any associated analyses. If possible, make recommendation on current or future use of alternative assessment approach presented in minority report.

8. Recommend best estimates of stock biomass, abundance, exploitation, and stock status of Atlantic menhaden from the assessment for use in management, if possible, or specify alternative estimation methods.

9. Review the research, data collection, and assessment methodology recommendations provided by the TC and make any additional recommendations warranted. Clearly prioritize the activities needed to inform and maintain the current assessment, and provide recommendations to improve the reliability of future assessments.

10. Recommend timing of the next benchmark assessment and updates, if necessary, relative to the life history and current management of the species.

11. Prepare a peer review panel terms of reference and advisory report summarizing the panel's evaluation of the stock assessment and addressing each peer review term of reference. Develop a list of tasks to be completed following the workshop. Complete and submit the report within 4 weeks of workshop conclusion.

1 INTRODUCTION

1.1 Brief Overview

The importance of Atlantic menhaden as a forage fish has long been recognized. As far back as 2004, managers, stakeholders, and the public have had an interest in Atlantic menhaden's role as forage in the ecosystem. Atlantic menhaden are a food source for a variety of species including larger fish such as striped bass (Hartman and Brandt 1995b) and bluefin tuna (Butler et al. 2010), birds such as bald eagles (Mersmann 1989) and osprey (Glass and Watts 2009), and marine mammals like bottlenose dolphin (Gannon and Waples 2004). Many of these predators support valuable commercial and recreational fisheries or ecotourism industries, in addition to having cultural value.

The single-species assessments in 2004 and 2010 used estimates of natural mortality from multispecies virtual population analyses (MSVPA) as input to the single-species model to better quantify the effects of predation on Atlantic menhaden populations (ASMFC 2004, 2010). However, there was still a strong interest in accounting for not only the effects of predation on Atlantic menhaden population dynamics, but also the effects of Atlantic menhaden removals on important predator species.

After an Ecosystem Management Objectives Workshop in 2015 (ASMFC 2015; see also Section 1.4), the Atlantic Menhaden Management Board formally tasked the Commission's Ecological Reference Point Workgroup (ERP WG) with developing reference points for management use that could account for Atlantic menhaden's role as a forage fish. The ERP WG developed a suite of models to provide ecological reference points (ERPs) and parameterized them with the best available data for Atlantic menhaden and key predator species. The ERP WG evaluated the performance of these models, their strengths and weaknesses, and their ability to inform the fundamental management objectives identified by the Board in order to determine the best tool for ecosystem-based management of Atlantic menhaden (Table 1).

Given the results, the ERP WG recommends a hybrid approach combining the current single-species assessment model with an EwE model of intermediate complexity to quantitatively evaluate trade-offs between Atlantic menhaden harvest and biomass levels of key managed predators. The final balance between the level of Atlantic menhaden harvest and maintaining predator biomass levels will be a management decision, but this approach will allow managers and stakeholders to evaluate those tradeoffs both quantitatively and transparently.

1.2 Need for Ecological Reference Points

The impact of fishing forage species on predator species and the larger ecosystem has received increasing attention in recent years. Much of this work has concluded that forage fisheries should be managed more conservatively than single-species reference points would suggest, to both ensure the sustainable harvest of forage fish and to reduce ecosystem impacts from their removal. For example, Smith et al. (2011) recommended maintaining forage fish populations at target biomass of 75% of unexploited biomass to prevent negative consequences to predators, compared to the approximately 60% level implied by fishing at F_{MSY}. Pikitch et al. (2012) recommended a precautionary approach for forage fish management in order to sustain both predator and prey species, including fishing at 50-75% of F_{MSY} and using a biomass threshold of 30-40% of unexploited biomass, depending on the quality of data available. Hilborn et al. (2017) pushed back on these conclusions, pointing out that the models used to develop those recommendations did not include consideration of environmental drivers of forage fish recruitment, the weak stock-recruitment relationship observed for most forage species, or the differing selectivities of predators and fisheries. As a result, some of the ecosystem models may overstate the ecosystem impact of fishing on forage fish abundance and predators. Despite those conclusions, there remains a general consensus that ecosystem services should be considered when managing forage fisheries.

All stock assessments account for some level of predation mortality in their estimates of *M*. Those that use age-varying natural mortality (such as Lorenzen 1996) incorporate the idea that natural mortality rates are higher at the youngest and smallest age or size classes, which is driven at least in part by higher predation rates on those groups. Some assessments have incorporated time-varying *M*, with approaches like an *M* vector scaled by annual key predator biomasses (northern shrimp, ASMFC 2018), or a random-walk process without an explicit driver (weakfish, ASMFC 2016). Generally, however, most assessments do not capture changes in natural mortality in direct response to predator demand. They also generally do not consider the effects of prey availability on the growth or survival of predators when establishing biological reference points for prey species.

Atlantic menhaden stock assessments have included an age- and time-varying natural mortality component since 2004, but there has been increasing interest from stakeholders and managers in explicitly managing Atlantic menhaden to account for their ecosystem services and changing predator demand. In 2017, when the Board was considering changing the management plan for Atlantic menhaden, ASMFC received 127,698 comments from the public in favor of some form of ecological reference points, compared to 7 comments in favor of single-species reference points.

Ecological reference point models are needed to quantify the effects of Atlantic menhaden harvest on their predators, to examine the impact of predators on Atlantic menhaden removal targets, and to quantitatively evaluate the tradeoffs between Atlantic menhaden harvest and predator biomass. Non-species-specific “rule-of-thumb” advice provided by meta-analyses like Smith et al. (2011) and Pikitch et al. (2012) are based on ecosystems that are not representative of the Atlantic coastal shelf and estuaries. More importantly, such “rules-of-thumb” reference points do not allow for the evaluation of specific trade-offs between forage fishery removals and abundance of important predator species. In order to provide the best management advice for this species and this ecosystem, ecological reference point models developed specifically for the coast-wide Atlantic menhaden stock are needed.

1.3 Regulatory History

See the single-species benchmark stock assessment for a more thorough discussion of Atlantic menhaden regulatory history. Sections from that assessment have been abbreviated below.

Atlantic menhaden management authority is vested in the states because the vast majority of landings come from state waters. All Atlantic coast states and jurisdictions, with the exception of the District of Columbia, have a declared interest in the Atlantic menhaden management program.

The first coastwide fishery management plan (FMP) for Atlantic menhaden was passed in 1981 (ASMFC 1981). The FMP did not recommend or require specific management actions, but provided a suite of options should they be needed. The FMP has undergone a series of revisions and amendments in the subsequent years.

In 1988, the ASMFC concluded that the 1981 FMP had become obsolete and initiated a revision to the plan. The 1992 Plan Revision included a suite of objectives to improve data collection and promote awareness of the fishery and its research needs (ASMFC 1992). Amendment 1, approved in 2001, provided specific biological, social, economic, ecological, and management objectives (ASMFC 2001). Amendment 2, approved in 2012, established a 170,800 metric ton total allowable catch (TAC) for the commercial fishery beginning in 2013 (ASMFC 2012a).

Amendment 3 (2017a) completely replaced Amendment 2 and currently sets the management program for Atlantic menhaden. The Amendment continues to manage the stock via single-species biological reference points until the review and adoption of menhaden-specific ecological reference points as part of the 2019 benchmark stock assessment process. In the interim, the Board used an *ad hoc* approach to set the TAC at 216,000 mt, an increase from the previous years' TACs, but less than what would be recommended if the stock were fished at the single-species target *F* rate, in order to provide a qualitative buffer for ecosystem services.

1.4 Ecological Management Objectives

In 2015, the Commission established the Ecosystem Management Objectives (EMO) Workgroup to identify potential ecosystem management objectives for menhaden-specific ecological reference points. To provide a range of perspectives on Atlantic menhaden management, the multi-disciplinary workgroup included representatives from the Atlantic Menhaden Management Board, stakeholder Advisory Panel, and Technical Committee.

At the EMO Workshop, the Workgroup identified potential ecosystem management objectives, as well as their associated performance measures, through a structured decision-making process (ASMFC 2015). Two types of objectives were identified: fundamental and means. Fundamental objectives are the end goals the group would like to achieve and represent what the group values. Means objectives are intermediary goals necessary to achieve the fundamental objectives, i.e., they represent "means to the ends" of achieving the fundamental objectives. A comprehensive list of fundamental and means objectives was created and then distilled into a more concise list. The Workgroup developed performance metrics for the refined list of fundamental objectives.

EMO Workshop Fundamental Management Objectives and Performance Measures

Fundamental Objectives	Performance Measures
Achieve broad public support for management	- Unanimous vote of the Atlantic Menhaden Management Board - Positive press releases from all stakeholders - "Informed consent" or acknowledgement that the decisions made were "fair and reasonable" - Participation in the fishery benefits - Absence of legal action - Strong compliance with management measures
Sustain menhaden to provide for fisheries	- Meeting or exceeding (positively) reference points - Non-truncated age distribution - Historical distribution maintained - Avoid unintended economic consequences of management - Employment in fishery - Achieving yield objectives for all fisheries - Achieving abundances that exceed "depleted" status - Reduce regulatory discards
Sustain menhaden to provide for predators	- Same as for fishery, assuming reference points are ecological reference points - Predators in a healthy nutritional state - Distribution of menhaden related to predator requirements (prey availability)
Sustain menhaden to provide for historical and cultural values	- Maintaining "historical" (meaning existing and recent past infrastructure rather than distant past) patterns of employment (spatial, demographic, gear use, etc.)
Sustain menhaden to provide for ecosystem services	- Same as above; represented in the other menhaden "services"
Minimize risk to sustainability due to changing environment	- Analysis would explicitly consider uncertainty about future environmental conditions
Provide stability for all types of fisheries	- Variability for employment and yield - Frequency of substantive management action
Sustain ecosystem resiliency or stability	- Covered by metrics above; if successful in providing for a viable fishery and other food web components that are related to menhaden

The EMO Workgroup also developed the following list of means objectives, which support achieving the fundamental objectives:

- Science
    - Increase knowledge base
    - Better communication of science
    - Account for variation
- Management
    - Define clear objectives
    - Provide timely advice
- Ecosystem
    - Ensure adequate supply of menhaden for:
        - Individual predator groups
        - Food web as a whole
- Account for spatial/temporal variation when using trade-offs
- Minimize the risks of collapse for:
    - Menhaden – the metric of collapse would be a certain level of biomass or fecundity relative to unfished spawning stock biomass or fecundity
    - Fishery – the metric for fishery collapse would depend on the fishery; it would indicate that the fishery is no longer economically viable to fish
    - Irreversible ecosystem change – changes to the food web such that it would not recover to a previous state even with the relaxation of fishing pressure

1.5 Model Selection

As part of the 2015 Benchmark Stock Assessment for Atlantic Menhaden, the ERP WG presented a suite of preliminary ERP models and ecosystem monitoring approaches for feedback (SEDAR 2015). The ERP WG used the peer review recommendations from that assessment and the outcomes of the EMO Workshop to assess the ability of various ERP models to address management objectives and performance measures. The ERP WG focused on those fundamental objectives and performance measures that could be addressed using ecological models. Some objectives, such as “sustain Atlantic menhaden to provide for historical and cultural values” or “achieve broad public support for management,” fell outside the purview of the ERP WG. Table 1 summarizes the fundamental objectives and associated performance measures that each ERP model can address.

To best address the management objectives identified at the EMO Workshop (Table 1), the approach selected needed to:

- explicitly examine the trade-off between fishery removal of menhaden and resulting changes in biomass or abundance among important predators;
- provide quantitative and understandable advice on removal levels of Atlantic menhaden under various predator biomass or fishing levels;
- examine the implications and consequences of Atlantic menhaden harvest strategy on important predators, either through predator growth rates and condition or mortality rates;
- be updatable on a timeframe consistent with Atlantic menhaden management.

Approaches were then selected based on: (1) the ability to address multiple management objectives; (2) the ability to predict and monitor performance measures in response to management action; (3) technical merits; and (4) consistency with the advice from the 2015 Peer Review.

Based on this evaluation, the ERP WG placed emphasis on models of intermediate complexity (a multispecies statistical catch-at-age model and an Ecopath with Ecosim model with limited predator and prey components) in developing ecological reference points. However, more complex (a full Ecopath with Ecosim model) and simpler (a Steele-Henderson surplus production model and a surplus production model with time-varying *r*) models were also developed, in order to provide context for the results of the intermediate complexity models and evaluate the tradeoffs between model assumptions, data availability, and the ability to meet management objectives.

In the end, the intermediate complexity Ecopath with Ecosim model was best able to meet the ecosystem management objectives in a timeframe suitable for management, while providing information consistent with the more complex model.

2 ASSESSMENT HISTORY

2.1 Previous Stock Assessments

Since the stock assessment peer review process was adopted by the ASFMC in 1998, Atlantic menhaden have been assessed several times as a single species (ASMFC 1999, 2004, 2010, 2012b; SEDAR 2015; ASMFC 2017b). The most recent peer-reviewed benchmark stock assessment was SEDAR 2015, which was updated in 2017.

Explicit multispecies considerations have been a part of the single-species assessments since 2004. To better quantify the effects of predation on Atlantic menhaden the single-species assessments in 2004 and 2010 used the *M*-at-age estimates from MSVPA as input to the single-species model. Issues with MSVPA model performance and the effort to develop explicit ecological reference point models resulted in moving away from the time-varying *M*-at-age to a time-constant *M*-at-age in the 2015 assessment (SEDAR 2015). The process of developing ecological reference points for Atlantic menhaden began as part of the 2015 single-species assessment, but the work was not ready to be peer-reviewed at that time.

2.2 Summary of Previous Assessment Models

The Beaufort Assessment Model (BAM) was used to provide management advice during the 2015 benchmark stock assessment (SEDAR 2015) and the 2017 update. BAM is a statistical catch-at-age model that estimates population size-at-age and recruitment, using 1955 as the start year, and then projects the population forward in time. The model estimates trends in the population, including abundance-at-age, recruitment, spawning stock biomass, egg production, and fishing mortality rates. BAM was configured to be a fleets-as-areas model with each of the fleets broken into areas to reflect differences along the coast. This means that both reduction and bait fleets were split into north and south regions because the fisheries operated differently along the coast and through time.

In 2001, ASMFC began developing the Expanded Multispecies Virtual Population Analysis model (MSVPA-X), an extension of the ICES MSVPA, which was peer-reviewed in 2006 (NEFSC 2006, Garrison et al. 2010). The MSVPA-X model, like the original MSVPA, was a set of single-species VPA models that were linked by a feeding model, which allowed for the calculation of M_2, predation mortality on Atlantic menhaden. The extended version allowed for the use of tuning indices and improved the consumption, feeding, and size-selectivity models. The MSVPA-X model explicitly modeled Atlantic menhaden, striped bass, bluefish, and weakfish, and included a pool of “other prey”, which could be broken down into more specific groups if necessary.

The MSVPA-X was intended to better quantify predator and prey interactions and to account for these effects on Atlantic menhaden, specifically through the development of time-varying *M* estimates for use in single-species assessments. It was not intended to replace the single-species assessments, set reference points, or set harvest limits for the modeled species, but rather to inform the single species assessment for Atlantic menhaden. Estimates of *M* for Atlantic menhaden from the MSVPA-X were used in BAM for the 2004 and 2010 assessments. The MSVPA-X was updated for the 2015 assessment, but the estimates of *M* were not used in the base run of BAM. This was due to concerns about the MSVPA-X performance (SEDAR 2015) not matching the biomass trajectory of important predators. More importantly, the MSVPA-X could not match the trajectory of BAM biomass estimates with the more complex and detailed BAM parameterization and was sensitive to small changes in predator/prey overlap and prey preference parameters. The uncertainty from the MSVPA-X was used to set the scale of the uncertainty surrounding *M* in the Monte Carlo bootstrap runs done for the base run. The resulting *M*-at-age from the MSVPA-X was also used as a sensitivity analysis during the 2015 benchmark for the single species assessment.

The 2015 assessment also began work on the task of developing ecological reference points for Atlantic menhaden. A suite of ERP and ecosystem monitoring approaches were identified and characterized by the timeframe for completion, the type of ERPs they would provide, and what management objectives they would meet. The 2015 Peer Review Panel recommended: 1) fully engaging managers and stakeholders in a Management Strategy Evaluation process, and 2) placing emphasis on models of intermediate complexity as potential tools for examining trade-off among predators and prey. The 2015 assessment and the EMO Workshop report (Section 1.4) formed the basis of the 2019 ERP Assessment.

It is important to note that all the approaches examined were based on the unit stocks for both predators and prey. While regional approaches are possible, both data needs and the desire to provide stock-level advice for Atlantic menhaden made regional approaches unviable at this time. Rates of production, fishery removals, predator removals, and changes in predator/prey abundance can be different at the regional level than the dynamics on a stock-wide scale. Despite this and given the above constraints, the methods and approaches developed provide management advice on a stock-wide level only.

2.3 Biological Reference Points

Atlantic menhaden are currently managed with single-species reference points, based on the historical performance of the population during 1960 to 2012, a period during which the Technical Committee (TC) considers the population to have been sustainably fished. The F_{TARGET} is defined as the median geometric mean F on ages 2-4 from 1960-2012, and the $F_{THRESHOLD}$ is the maximum geometric mean F for ages 2-4 during that period. To determine overfished status, a fecundity target and threshold are used (rather than a spawning stock biomass target and threshold). The fecundity target and threshold are defined as the mature egg production one would expect when the population is being fished at the target or threshold fishing mortality rates, respectively. Based on the assessment update (ASMFC 2017), Atlantic menhaden were neither overfished nor experiencing overfishing under these reference points.

After the 2015 assessment, ASMFC considered using interim ecological reference points for Atlantic menhaden until this assessment could be completed. These interim reference points would have been based on generic or "rule-of-thumb" guidelines proposed in the literature such as a biomass target of 75% unfished biomass (Smith et al. 2011) or F=50%M (Pikitch et al. 2012). In the end, the Board decided not to change the definitions of the reference points until Atlantic menhaden specific ERPs could be developed, and instead applied an *ad hoc* buffer to the quota, setting the TAC lower than what the single-species target F rate would have allowed (ASMFC 2017b).

3 PREDATOR AND PREY SPECIES

3.1.1 Diet Data Sources

The ERP WG examined a range of diet datasets, from large-scale, long-term monitoring programs to individual small-scale studies. The proportion of Atlantic menhaden in the diets of key predators varied by year, season, location, and age class of predators sampled, making the selection of diet data sources important in model parameterization.

Fish stomach-content data were obtained from three main sources: the Northeast Fisheries Science Center (NEFSC) Food Web Dynamics Program, the North East Area Monitoring and Assessment Program (NEAMAP), and Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP). The NEFSC program has systematically sampled predator food habits since 1973 (Link and Almeida 2000). The food-habits data are structured by predator species and length, but prey lengths and ages are not routinely measured. A subset of the database is structured by both predator and prey lengths, which was used for part of the following analyses. NEAMAP and ChesMMAP also collect stomach-content data under similar protocols to the NEFSC program; NEAMAP has collected data since 2008 and ChesMMAP since 2002. These data were used to supplement the stomach-content data and have an added benefit of increasing the coastal area covered for this dataset (NEAMAP and ChesMMAP sample areas further inshore than the NEFSC sampling program).

Both datasets have strengths (e.g., the NEFSC data has a long time-series and the NEAMAP data are more inshore so is better able to acquire many of the species used in this study) and weaknesses (e.g., the NEFSC data are from further offshore and the NEAMAP data timeseries is short) but taken together they offer a fairly comprehensive snapshot of the populations. However, sample sizes often precluded analyses on finer spatial or temporal scales. These databases focused on finfish and shellfish species, not birds or marine mammals. Smaller scale studies were used to supplement the data from these long-term programs for some of the modeling approaches, especially for species that were not well represented in the long-term programs.

3.1.2 Identification of Key Predator and Prey Species

Two of the ERP models presented in this report are models of intermediate complexity, which focus on a limited number of key predator and prey species. To identify this suite of key predator and prey species, the ERP WG considered a number of factors, including: the importance of a species' role as a Atlantic menhaden consumer (as indicated by the diet data), the importance of a species' role as an alternative prey to Atlantic menhaden (as indicated by the diet data), the quality and availability of life history and fishery data for the species, and the relevance of the species to ASMFC management.

Predator Species

Diet data were used to identify key predators during the last benchmark assessment as part of the update to the MSVPA-X (SEDAR 2015). The methods and conclusions from that assessment were reviewed by the ERP WG and used to inform the choice of key predators used in this assessment. The NEFSC Food Habits Database (FHDB) (1981-2012) was queried for all species with Atlantic menhaden recorded in their gut contents. Only twelve species had records of Atlantic menhaden in their gut contents: striped bass, bluefish, spiny dogfish, weakfish, smooth dogfish, spiny butterfly ray, clearnose skate, goosefish, Atlantic angel shark, dusky shark, sandbar shark, and Atlantic herring. Of the twelve predators whose diets contained Atlantic menhaden, there were some notable outliers, such as Atlantic herring, which does not typically feed on Atlantic menhaden, and spiny butterfly ray, which had one individual stomach that contained 86% of the total prey weight for that species and 100% of that stomach was Atlantic menhaden. The ERP WG decided to remove these outliers from the list of key predators, along with Atlantic angel shark, dusky shark, and sandbar shark, all of which had less than 50 stomachs sampled throughout the entire time series (the next lowest sample number of stomachs was about 800; species with notably fewer samples would not provide an accurate representation of diets when compared to the rest of the data available). The remaining predators were considered by the group for inclusion into the models.

The annual Atlantic menhaden consumption (C) of each predator was estimated using the methodology from Butler et al. (2009), defined as:

$$C = B \times P \times DR \times W \times T \tag{3.1}$$

Where:

B = Biomass of predators (B) calculated from scaled up swept area biomasses from the NEFSC Survey 1981-2012. This calculation assumed that catchability is equal to 1.0 and that the survey covers the inshore and offshore extent of each species' range.

P = the proportion of each predator stock in the model domain calculated using swept area biomass from the NEFSC Survey and scaled up to the full range of the species to estimate total biomass. For offshore strata, a GIS program was used to pare out tows that were offshore of the model domain. All strata with at least one tow in the model domain were then divided by the total tows conducted in that stratum to get the proportion of tows in that domain. Model domain biomass divided by expanded total biomass by range was calculated to get the proportion of each predator in the model domain.

DR = Daily ration (in kg prey per kg predator per day) generated using direct estimates from literature and calculations using parameters from the literature. Direct estimates for similar species or overall average of other species that were not as similar were used when necessary.

W = the proportion of total prey in weight that is Atlantic menhaden generated using data from the NEFSC FHDB, ChesMMAP survey, and NEAMAP survey.

T = the portion of the year (in days) that predator and prey are both in the model domain calculated using the NJ Ocean Trawl Survey. It was assumed Atlantic menhaden were always present somewhere in the model domain throughout the year. The NJ Ocean Trawl Survey catches all predators, so it was used as a proxy of when predators were in the model domain. Only 2% of the stations fell outside domain so all of them were used. The average biomass per season across years 1990-2012 was used to calculate when predators were present in the domain. All proportions were standardized to 1.0 then divided by the maximum. The NJ Ocean Trawl Survey occurs 5 months out of each year so biomass for months in which sampling did not occur was linearly interpolated based on the closest surrounding months' biomass. For any month with less than 1% of the max, the predator was assumed not present. Time (days) in the model domain was then finally calculated from months where the predator was present in the model domain.

Spiny dogfish, striped bass, and bluefish had the highest Atlantic menhaden consumption (Table 2). In addition, those species also had reliable data on catch and indices of abundance, as well as recently updated assessments with estimates of biomass and fishing mortality from peer-reviewed stock assessments. All three are managed either solely (striped bass) or cooperatively (spiny dogfish and bluefish) by the Commission, so providing quantitative information on these species would be relevant to management. All three of these predators were included in the group of key predators.

Weakfish and smooth dogfish alternated between the fourth and fifth most important Atlantic menhaden predator, depending on the ranking system, but weakfish more consistently ranked as the fourth. The ERP WG debated including smooth dogfish and/or weakfish given their relatively low menhaden consumption rates compared to the top three predator species. The ERP WG decided not to include smooth dogfish because of data availability challenges, including the lack of age data to support an age-structured model. The ERP WG decided to include weakfish due to the decline in population through the years which could provide important contrast, given that it is the only one of the predator species that has shown significant declines in population size over the time series. Predation mortality and/or increased competition for Atlantic menhaden from striped bass have been proposed as a factor in weakfish population declines (NEFSC 2009). Weakfish also had more robust data to support modeling efforts, and are solely managed by the Commission, so information on the ecosystem effects of Atlantic menhaden fishing on weakfish would be more relevant to management.

Marine Mammals

Overall lack of data and taxonomic resolution in marine mammal diet data limits incorporation of marine mammals as predators for multispecies/food web/ecosystem models of Atlantic menhaden. A paper by Smith et al. (2015) is the only broad, systematic review of marine mammal diets (i.e., consumption rates) for the US Atlantic Coast; note that it also includes some studies outside of the area. The paper develops annual consumption rates of marine mammals on key marine species. In the paper, marine mammal diet compositions were allocated to 12 standard prey groups of similar taxonomy (squid, mesopelagic fish, clupeids, scombrids, small gadids, large gadids, shrimp, zooplankton, benthic invertebrates, sandlance, flatfish, and miscellaneous fish). Because the data for diet composition were from a wide array of references using multiple sampling types (mostly scat and stomach analysis from bycaught and stranded animals), finer taxonomic resolution was not possible for this systematic review of marine mammal diets that included ~110 papers and reports. A literature review in web of science showed no additional research papers (from 2008-2018) with information on Atlantic menhaden in marine mammal diets. Of the 110 articles reviewed by Smith et al. (2015), only 3 studies specifically identified Atlantic menhaden in the diet. All 3 studies were on bottlenose dolphin. Bottlenose dolphin are the only species of marine mammal with adequate taxonomic resolution in the diet data to support inclusion of dolphins as a predator in a multi-species model; however, the proportion of Atlantic menhaden in bottlenose dolphin diets (4% or less) suggests that they are not important predators of Atlantic menhaden.

Prey Species

The key ERP predators identified here are generalists, consuming a wide range of other prey items in addition to Atlantic menhaden. The ERP models of intermediate complexity include a pool of “other prey biomass,” but also allow for the modeling of other, specific prey species in addition to Atlantic menhaden. To identify an additional key prey species to be modeled explicitly, the ERP WG used similar criteria to what was used for key predator identification. Atlantic herring was chosen as an alternate prey species because it was a major component of the diets of the key predators. In addition, unlike several other prey species – such as bay anchovy, sand eels, and benthic invertebrates – Atlantic herring was recently assessed with an age-structured model. As a result, reliable catch data, indices of abundance, age structure, biomass, and fishing mortality were available.

4 LIFE HISTORY

4.1 Atlantic Menhaden

See the single-species benchmark stock assessment for a more thorough discussion of Atlantic menhaden life history. Sections from that assessment have been abbreviated below.

Stock Definitions

Atlantic menhaden inhabit nearshore and inland tidal waters from Florida to Nova Scotia, Canada. Atlantic menhaden are considered a single stock. Historically there was considerable debate relative to stock structure of Atlantic menhaden on the US East Coast, with a northern and southern stock hypothesized based on meristics and morphometrics (Sutherland 1963; June 1965). Based on size-frequency information and tagging studies (Nicholson 1972 and 1978; Dryfoos et al. 1973), the Atlantic menhaden resource is believed to consist of a single unit stock or population. Genetic studies (Anderson 2007; Lynch et al. 2010) support the single stock hypothesis.

Migration Patterns

There have been several studies examining Atlantic menhaden migration patterns (Roithmayr 1963; Dryfoos et al. 1973; Nicholson 1978; ASMFC 2004). Adults begin migrating inshore and north in early spring following the end of the major spawning season off the Carolinas during December-February. The oldest and largest fish migrate farthest, reaching southern New England by May and the Gulf of Maine by June. Adults that remain in the south Atlantic region for spring and summer migrate south later in the year, reaching northern Florida by fall. In the fall, Atlantic menhaden begin a migration to the Carolinas and spawn as a population in the winter months, although spawning occurs along the migration route earlier in the year (Ahrenholz 1991; Berrien and Sibunka 1999).

Historical tagging data from 1966-1969 was recently reanalyzed by Liljestrand et al. (2019a, 2019b), which indicated that while the pattern of Atlantic menhaden’s movement was similar to previous findings, the magnitude of movement during the winter in the northern region differed. For example, previous literature (Roithmayr 1963; Nicholson 1971) stated that the majority of Atlantic menhaden in the north migrate south to overwinter in North Carolina whereas Liljestrand et al. suggested about 55% of Atlantic menhaden in the northern region migrates southward. Therefore, there may be less southward movement of Atlantic menhaden in the winter than previously described by the literature and more residency in the northern area throughout the year.

Age and Growth

In 1955, the NOAA Laboratory at Beaufort, North Carolina, began monitoring the Atlantic menhaden purse-seine fishery for size and age composition of the catch (June and Reintjes 1959). Scales were selected as the ageing tool of choice for Atlantic menhaden due to ease of processing and reading and an age validation study confirming reliable age marks on scales (June and Roithmayer 1960). The Beaufort lab to date still ages all the reduction and bait fishery samples. The maximum age used in this assessment is 10 years, although Atlantic menhaden over age 6 are rarely found in the fisheries.

In the single-species assessment, a time-invariant relationship for length-weight was used. Annual estimates of fork length-at-age were interpolated from the annual, cohort-based von Bertalanffy growth fits with a bias correction in order to represent the population at the start of the fishing year (March 1) for use in estimating population fecundity. Age-6 fish average around 375 mm in fork length and 600 grams in weight over the time series.

Maturity and Fecundity

Using data from the NEAMAP Southern New England/Mid-Atlantic Nearshore Trawl Survey to evaluate maturity-at-age, it was determined that maturity is a length-based process as opposed to an age-based process. A logistic regression was fit to the maturity and length data from the commercial reduction fishery database. Time-varying lengths-at-age for the population were used along with the logistic regression to provide time-varying maturity at age for 1955-2017 for the single-species assessment. Generally, 5-15% of age-1 fish were mature, approximately 50% were mature by age-2, and 95-100% were mature by age-5 (Figure 1).

Since SEDAR 40 (2015), work has been completed by VIMS (R. Latour and J. Gartland, unpublished data) to address a single-species research recommendation and update fecundity values for use in BAM. Based on the analysis of the study, Latour and Gartland concluded that Atlantic menhaden are indeterminate batch spawners. Additionally, estimates of age-specific annual fecundity for Atlantic menhaden spanning age-0 to age-6+ were provided for SEDAR 69 (2019). Female fecundity-at-age for each year was fixed in BAM and was based on a function of mean weight by age for the population. The annual fecundity-at-age in year *i* (AF_{ai}) was estimated as:

$$AF_{ai} = RBF * WT_{ai} * SF * PM_{ai} \tag{4.1}$$

where *RBF* (relative batch fecundity) was 236.92 eggs/g ovary-free body weight, *SF* (spawning frequency) was 11.70 spawns/season, and where WT_{ai} (weight-at-age) and PM_{ai} (maturity-at-age) were the weight-at-age *a* and proportion of fish mature at age *a* for a given *i* at the start of the fishing year (i.e., March 1). The updated fecundity values from Latour and Gartland resulted in higher estimated fecundity from SEDAR 2015. Refer to the single-species assessment Section 2.6 and Appendix 14.1 for more details.

Natural Mortality

In the previous Atlantic menhaden stock assessment (SEDAR 2015), *M* was determined using the method of Lorenzen (1996), which was scaled to an historical analysis done on historical tagging data. Since SEDAR 40 (2015), the historical tagging data have been digitized and a new analysis was conducted by Liljestrand et al. (2019a, 2019b), which provided updated values. The new analysis uses methods that were not available during the original collection of the data set.

For the 2019 single-species benchmark assessment (SEDAR), several methods for estimating *M* were explored, including several age-constant *M* estimates and age-varying *M* approaches. Ultimately, an age-varying but time-invariant approach using the methods of Lorenzen (1996) and scaled to the new tagging estimates from Liljestrand (2019a, 2019b) was used. This resulted in estimates of *M* ranging from 1.76 for age-0 fish to 0.72 for age-6 fish (Figure 1). See SEDAR 69 (2019) for further details.

Habitat

Estuarine and nearshore waters along the Atlantic coast from Florida to Nova Scotia serve as important habitat for juvenile and/or adult Atlantic menhaden. Adult Atlantic menhaden spawn in oceanic waters along the continental shelf, as well as in sounds and bays in the northern extent of their range. Winds and tides transport larvae shoreward from the shelf toward nursery grounds in the estuaries. After hatching from buoyant eggs, the larvae are transported by ocean currents to fresh and brackish-water estuaries where much of the early development takes place. Juvenile habitat is unconsolidated bottom consisting mostly of sand and mud, with various mixtures of organic material. In more northerly areas, juveniles can be found in rocky coves, with mixtures of cobble, rock, and sand bottoms. Sub-adult habitat is found in temperate, nearshore marine and estuarine areas that have a bottom composition of sand and mud, and more organic material than in marine areas. Adult habitat ranges from a bottom composition of sand, mud, and organic material to marine sand and mud with increasing amounts of rocks in the more northerly areas. Adults appear to prefer water temperatures near 18°C; adult migrations and movement may be attributed to seeking waters within a certain temperature range.

4.2 Atlantic Herring

Stock Definitions

Atlantic herring (*Clupea harengus*) is a schooling pelagic clupeid which ranges from North Carolina to Labrador in the Western Atlantic. In US waters the Georges bank-Gulf of Maine stock are fall spawners that range from NC through the Gulf of Maine (GOM) and out to Georges Bank (GB). There are two main spawning components for this meta-stock, one centered on GB, and the other in coastal portions of the GOM (Shepard et al. 2009; NEFSC 2012; NEFSC 2018a).

Migration Patterns

When not spawning, these sub-components intermingle in the summertime along the Maine coast with the GB component located both in the inshore GOM and offshore on GB. Sometime after spawning in their respective areas, both sub-components travel south to overwinter from Block Island Sound to the Virginia Capes. Return migration back to their summertime feeding grounds occurs in early to mid-spring. There is thought to be some mixing between the GOM-GB stock and the adjacent Canadian 4WX stock. While the rate of mixing is unknown, the magnitude is thought to be rather small (NEFSC 2018).

Age and Growth

Life span is generally thought to be 14 years for Atlantic herring in the absence of fishing (NEFSC 2018). The average size-at-age of Atlantic herring has declined over time, most notably for older ages; the average weight at age of an age-8 fish from 1965-1986 was 0.35 kg, while the average weight at age of an age-8 fish from 1995-2017 was 0.2 kg. The time-series average was 0.28 kg (Figure 2).

Maturity and Fecundity

Atlantic herring are 65% mature at age-3, 90% by age-4 and 100% mature by age-5 (Figure 2; NEFSC 2018a). Atlantic herring lay sticky sinking eggs over gravel or sand in shallow portions of the GOM and GB in the fall with larval settlement and recruitment to Age 1 occurring in the early spring. As such the birthdate for all cohorts occurs January 1st in any given year (NEFSC 2018a).

Natural Mortality

Atlantic herring are important prey items for a variety of fish, birds, mammals, and other predators (NEFSC 2018a). Some of these predators, such as striped bass and bluefish, are also important predators of menhaden. Despite this, the most recent assessment for Atlantic herring assumed a 0.35 natural mortality static across age and year based largely on model diagnostics and a lack of change in consumption by important predators (Figure 2) (NEFSC 2018a).

Habitat

Atlantic herring are a pelagic species found in the open ocean, but the benthic zone is especially important for their reproduction. In U.S. waters, herring spawn mainly in two areas: the Gulf of Maine and Georges Bank/Nantucket Shoals. Spawning grounds are located in high-energy environments with strong tidal currents and high salinity. Eggs require water temperatures ranging from 7 to 15°C and depths from 5 to 90 m, and will not survive if covered by mud or fine sand.

Larvae have been observed in depths up to 1,500 m, but are generally found in depths in the 41 to 220 m range and temperatures below 12.5°C in the Gulf of Maine, Georges Bank, and southern New England. Juveniles are commonly found in waters with temperatures from 2.5 to 14.5°C, depths between 4-300 m, and salinities ranging from 20 to 32 ppt. Adults occupy the same geographic range and similar habitats as juveniles, but typically prefer more saline (> 28 ppt) waters.

4.3 Striped Bass

Stock Definitions

Atlantic striped bass (*Morone saxatilis*) are found along the eastern coast of North America from the St. Lawrence River in Canada to the St. Johns River in Florida (ASMFC 1990). Atlantic striped bass are anadromous, returning to their natal rivers to spawn. As a result, the Atlantic striped bass population includes multiple biologically distinct stocks. Stocks which occupy coastal rivers from the Albermarle Sound/Roanoke River system in North Carolina south to the St. Johns River in Florida are believed to be primarily endemic and riverine, as historical tagging data suggest they do not presently undertake extensive Atlantic Ocean migrations as the more northern stocks do.

The habitat of the coastal migratory striped bass population includes the coastal and estuarine areas from Maine through Virginia and the coastal waters of North Carolina. The coastal migratory striped bass population is assessed and managed as a single stock, although it is known to be comprised of multiple biologically distinct stocks, predominantly the Chesapeake Bay stock, the Delaware Bay stock, and the Hudson River stock.

Migration Patterns

Atlantic migratory striped bass exhibit two types of migration: a spawning migration in late winter to early spring where mature adults move from ocean waters to the spawning grounds at the heads of estuaries and in their tributaries (Shepherd 2007; Zurlo 2014), and a north-south migration in coastal ocean waters during the rest of the year, with fish moving northward into New England and Gulf of Maine waters during the summer and southward to waters off of Virginia and North Carolina during the winter (Kneebone et al. 2014). Juveniles remain in their natal estuaries until they are about three years old, when they begin to leave the estuaries and join the coastal migratory population (Nichols and Miller 1967). The extent of the migration that individual striped bass undertake varies depending on the sex, size, and stock of the fish (Hill et al. 1989; Secor and Piccoli 2007; Callihan et al. 2014).

Age and Growth

Generally, longevity of striped bass has been estimated as approximately 30 years, with a maximum observed age of 31 years based on otoliths (Secor 2000). Striped bass are capable of attaining moderately large size, reaching as much as 125 pounds (57 kg) (Tresselt 1952), and fish weighing 50-60 pounds (23-27 kg) are not exceptional (Figure 3). Growth rates and maximum size are significantly different for males and females. Both sexes grow at the same rate until 3 years old; beginning at age-4, females grow faster than males. Females grow to a considerably larger size than males; striped bass over about 30 pounds (14 kg) are almost exclusively female (Bigelow and Schroeder 1953).

Maturity and Fecundity

Female striped bass begin to mature at age-4. They are 45% mature by age-6 and 100% mature by age-9 (Figure 3; NEFSC 2019). Males mature at younger ages, reaching 100% maturity by age-4 (NEFSC 2013).

The number of mature ova in female striped bass varies by age, weight, and fork length. Jackson and Tiller (1952) found that fish from Chesapeake Bay produced from 62,000 to 112,000 eggs/pound of body weight, with older fish producing more eggs than younger fish. Raney (1952) observed egg production varying with size, with a 3-pound (1.4 kg) female producing 14,000 eggs and a 50-pound (23 kg) specimen producing nearly 5,000,000.

Natural Mortality

Striped bass are a long-lived species, suggesting natural mortality is relatively low. In the 2013 benchmark assessment, age-specific *M* estimates for ages 1-6 were derived from a curvilinear model fitted to tag-based *Z* estimates (assuming *Z*=*M*) for fish younger than age 3 from New York and tag-based *M* estimates (Jiang et al. 2007) for age 3-6 striped bass from Maryland calculated for years prior to 1997 (NEFSC 2013). This resulted in a maximum *M*-at-age of 1.13 for age 1 fish declining to *M*=0.19 for age-6 fish (Figure 3). *M* for ages 7+ was assumed equal to 0.15, consistent with Hoenig's (1983) regression on maximum age.

An increasing prevalence of mycobacteriosis in the Chesapeake Bay since 1997 could be causing increases in natural mortality (Ottinger and Jacobs 2006). Although fish who are infected with the disease show overall decreased health (Overton et al. 2003), the slow progression of the disease may take years to become lethal in infected fish, thus allowing for multiple spawning opportunities, making determination of the population level impacts of the disease difficult (Jacobs et al. 2009). Various hypotheses have been put forward to explain the increasing prevalence of mycobacteriosis, including lack of forage and increasing water temperatures in Chesapeake Bay (Jacobs et al. 2009).

Habitat

Atlantic striped bass move between a variety of habitats in their life cycle. Generally, spawning and early development occurs at the heads of estuaries and in their tributaries, fish mature in estuaries, and move into the ocean as adults. Habitat selection and migratory behavior in striped bass is influenced by temperature and photoperiod (Able and Grothues 2007; Wingate and Secor 2007; O'Connor et al. 2012; Manderson et al. 2014). Striped bass are not usually found more than 6 to 8 km offshore (Bain and Bain 1982). Fishery-independent and fishery-dependent data suggest striped bass distribution on their overwintering grounds during December through February has changed significantly since the mid-2000s, with the migratory portion of the stocks moving well offshore in the U.S. Exclusive Economic Zone (EEZ, >3 miles offshore) (NEFSC 2018).

4.4 Bluefish

Stock Definitions

Bluefish (*Pomatomus saltatrix*) are a coastal, pelagic species found in temperate and tropical marine waters throughout the world (Goodbred and Graves 1996; Juanes et al. 1996). Bluefish in the western North Atlantic are managed as a single stock (NEFSC 1997; Shepherd and Packer 2006). Genetic data support a unit stock hypothesis (Graves et al. 1992; Goodbred and Graves 1996; Davidson 2002). The management unit is defined as the portion of the stock occurring along the Atlantic Coast from Maine to the east coast of Florida.

Migration Patterns

Bluefish spawn offshore, and juveniles settle in estuarine and nearshore shelf habitat (Kendall and Naplin 1981; Marks and Conover 1993; Able et al. 2003). Traveling in loose groups of fish aggregated by size, bluefish typically migrate north as far as Maine in the spring/summer and south as far as Florida in the fall/winter (Wilk 1977; Klein-MacPhee 2002; Shepherd et al. 2006).

Age and Growth

The maximum observed age for bluefish is 14 years (NEFSC 2015). Bluefish grow nearly one-third of their maximum length in their first year (Richards 1976; Wilk 1977). Estimates of L_∞ from the literature range from 87 cm – 128 cm (Lassiter 1962; Barger 1990; Terceiro and Ross 1993; Salerno et al. 2001; Robillard et al. 2009). Bluefish average weight is 5-6 kg at ages 6+ (Figure 4). There is no evidence of sexual dimorphism in growth.

Maturity and Fecundity

Bluefish mature quickly, with approximately half of the population mature at age-1 and close to one hundred percent mature (97%) by age-2 (Figure 4; NEFSC 2015). Bluefish are characterized as iteroparous spawners with indeterminate fecundity and spawn continuously during their migration (Robillard et al. 2008). This results in distinctive spring and summer cohorts within a year.

Natural Mortality

In past stock assessments, a value of 0.2 has been assumed as the instantaneous natural mortality (*M*) for bluefish over all ages and years (Figure 4; NEFSC 2015). This is in the range of estimates from age-constant methods based on maximum age or growth parameters such as Hoenig (1983), Jensen (1996), Hewitt and Hoenig (2005), and Then et al. (2014).

Habitat

Bluefish larvae occur near the edge of the continental shelf in the south Atlantic Bight, in open oceanic waters in the mid-Atlantic Bight, and over mid-shelf depths farther north (Shepherd and Packer 2006). Spring-spawned larvae are subject to advection to northern waters by the Gulf Stream (Shepherd and Packer 2006). Adult and juvenile bluefish are found primarily in waters less than 20m deep along the Atlantic coast (Shepherd and Packer 2006). Adults use both inshore and offshore areas of the coast and favor warmer water temperatures although they are found in a variety of hydrographic environments (Ross 1991; Shepherd and Packer 2006). Bluefish can tolerate temperatures ranging from 11.8°-30.4°C, however they exhibit stress, such as an increase in swimming speed, at both extremes (Olla and Studholme 1971; Klein-MacPhee 2002). Temperature and photoperiod are the principal factors directing activity, migrations, and distribution of adult bluefish (Olla and Studholme 1971).

4.5 Spiny Dogfish

Stock Definitions

Spiny dogfish (*Squalus acanthias*) are a small shark species that inhabit both sides of the North Atlantic and North Pacific Oceans, mostly in the temperate and subarctic areas. Spiny dogfish are considered a unit stock in the Northwest Atlantic Ocean (US and Canadian waters), ranging from Labrador to Florida, and are most abundant from Nova Scotia to Cape Hatteras (Rago et al. 1998).

Migration Patterns

Spiny dogfish are highly migratory (Compagno 1984) and migrate north in the spring and summer and south in the fall and winter. In the winter and spring, they congregate primarily in

Mid-Atlantic waters but also extend onto the shelf break of southern Georges Bank. In the summer, they are located farther north in Canadian waters and move inshore into bays and estuaries. By autumn, spiny dogfish have migrated north with high concentrations in Southern New England, on Georges Bank, and in the Gulf of Maine. They remain in northern waters throughout autumn until water temperatures begin to cool and then return to the Mid-Atlantic. Juvenile spiny dogfish school by size until sexually mature and then aggregate by both size and sex.

Age and Growth

Spiny dogfish are long-lived. The maximum recorded age for this species was 35 years for males and 40 years for females in the northwest Atlantic (Nammack et al. 1985). Female spiny dogfish are larger than males and can reach up to 125 cm in length (NEFSC 2006). L_∞ has been estimated at 100.5 cm for females (Nammack et al. 1985), corresponding to a weight of 5 kg at the oldest ages (Figure 5).

Maturity and Fecundity

Spiny dogfish mature late and have low fecundity. Female spiny dogfish reach sexual maturity at 12 years (~75 cm) (Figure 5), while males reach sexual maturity at six years (~60 cm). Mating occurs in the winter months and the pups are delivered on the offshore wintering grounds. Females give birth every two years with litters ranging from 2 to 15 pups. While carrying one litter, the female will begin developing eggs for the fertilization of her next litter. After an 18- to 24-month gestation period, pups are released live and fully formed at about 20-33 cm (Burgess 2002).

Natural Mortality

Natural mortality for spiny dogfish has been estimated at 0.092, based on a maximum expected age of 50 years (Rago et al. 1998) (Figure 5).

Habitat

Spiny dogfish are predominately epibenthic species, with no known associations to any particular substrate, submerged aquatic vegetation, or any other structural habitat (McMillan and Morse 1999). Data from fishery independent surveys can be used to define habitat based on water temperature and depth on the Atlantic coast. Juvenile and adult spiny dogfish showed similar patterns in habitat preference. Both life stages are most commonly caught in waters with bottom temperature ranges from 6-17^{o}C, and bottom depth ranges from 10m – 150m (ASMFC 2002).

4.6 Weakfish

Stock Definitions

Weakfish (*Cynoscion regalis*) can be found along the Atlantic coast from Florida through Massachusetts, but the core of their distribution is from North Carolina to New York. Genetic data suggest weakfish are a single stock (Graves et al. 1992; Cordes and Graves 2003), but tagging data and meristic/life history information suggest there may be spatial structure or sub-stock structure in the population (Crawford et al. 1988). However, since stock boundaries could not be determined with confidence from the available literature, weakfish continued to be assessed and managed as a single species within this range (ASMFC 2016). Tringali et al. (2011) found that there was an active zone of introgressive hybridization between weakfish and sand seatrout (*C. arenarius*) in Florida, centered in the Nassau and St. Johns Rivers, with the genome proportions of “pure” weakfish estimated at 48% in Nassau County and 17% in Duval County, and that “pure” weakfish were rare southward.

Migration Patterns

Weakfish exhibit a north-inshore/south-offshore migration pattern, although in the southern part of their range they are considered resident. Shepherd and Grimes (1983) observed that migrations occur in conjunction with movements of the 16-24^{o} C isotherms. Warming of coastal waters during springtime triggers a northward and inshore migration of adults from their wintering grounds on the continental shelf from Chesapeake Bay to Cape Lookout, North Carolina (Mercer 1983). The spring migration brings fish to nearshore coastal waters, coastal bays, and estuaries where spawning occurs. Weakfish move southward and offshore in waves as temperatures decline in the fall (Manderson et al. 2014; Turnure et al. 2014).

Age and Growth

The historical maximum age recorded using otoliths is 17 years for a fish collected from Delaware Bay in 1985 (ASMFC 2016). Weakfish growth is rapid during the first year, and age-1 fish typically cover a wide range of sizes, a result of the protracted spawning season. Lowerre-Barbierri et al. (1995) found length at age to be similar between sexes, with females attaining slightly greater length at age than males. Estimates of L_{∞} ranged from 89.3 cm – 91.7 cm depending on study area (Hawkins 1988; Villoso 1990; Lowerre-Barbierri et al. 1995). Weakfish in the catch averaged 5-6 kg at the oldest ages (Figure 6).

Maturity and Fecundity

Weakfish mature early, with 90-97% of age-1 fish estimated to be mature (Figure 6) Lowerre-Barbieri et al. 1996; Nye et al. 2008). Although the majority of age-1 fish were mature, age-1 weakfish spawned less frequently, arrived later to the estuary, and had lower batch fecundity than did older fish (Nye et al. 2008). Batch fecundity ranged from 75,289 to 517,845 eggs/female and significantly increased with both total length and somatic weight (Lowerre-Barbieri et al. 1996). Weakfish have a protracted spawning season and individual fish spawn multiple times in a season; spawning occurs from March to September in North Carolina (peaking from April to June) (Merriner 1976), but the season is shorter (May to mid-July/August) in Chesapeake Bay and Delaware Bay (Shepherd and Grimes 1984; Lowerre-Barbieri et al. 1996).

Natural Mortality

Recent assessments of weakfish indicate natural mortality has increased over time (NEFSC 2009; ASMFC 2016). Catch has declined significantly since the mid-1990s and remained at low levels in recent years under restrictive management, while recruitment indices have been stable over the time series; however, the population has not recovered. ASMFC (2016) used a Bayesian model to estimate time-varying natural mortality, and found that *M* was low (*M*=0.14-0.17) during the 1980s and early 1990s, but began to increase sharply in the late 1990s; it was estimated at 0.92-0.95 from 2003 – 2013 (Figure 6). There are several hypotheses about what caused the increase in *M*, including increasing predation or competition from increasing striped bass and spiny dogfish populations and large-scale environmental drivers like Atlantic Multidecadal Oscillation, but no definitive conclusions can be made (NEFSC 2009).

Habitat

Weakfish are found in shallow marine and estuarine waters along the Atlantic coast. They can be found in salinities as low as 6 ppt (Dahlberg 1972) and temperatures ranging from 17^{o} to 26.5^{o} C (Merriner 1976). Weakfish spawn in estuarine and nearshore habitats throughout their range, and larval and juvenile weakfish generally inhabit estuarine rivers, bays, and sounds, commonly associated with sand or sand/grass bottoms (Mercer 1983). Adult weakfish overwinter offshore on the continental shelf from Chesapeake Bay to North Carolina.

5 FISHERY DEPENDENT DATA SOURCES

5.1 Marine Recreational Information Program (MRIP) Changes

Data on recreational catch for modeled species comes from the Marine Recreational Information Program (MRIP, formerly the Marine Recreational Fisheries Statistics Survey or MRFSS). MRIP uses a combination of effort surveys that are designed to estimate the number of fishing trips taken in various regions of the US and dockside angler intercept surveys that are designed to estimate catch-per-trip and size frequencies of recreationally caught species. Data from these surveys are used to calculate total catch (broken down by harvest and live releases) and the size frequency of landed fish. MRIP estimates are available from 1981 to the present.

Prior to 2018, the estimates of angler effort (i.e., angler trips) used to calculate annual recreational catch and harvest of Atlantic striped bass were derived from the Coastal Household Telephone Survey (CHTS), a random-digit-dial telephone survey. The CHTS was replaced in 2018 by the mail-based Fishing Effort Survey (FES), due to concerns about the inefficient design, coverage bias, and declining response rates of the CHTS. The CHTS and FES were conducted simultaneously for three years (2015-2017), during which the FES produced much higher estimates of fishing effort, and therefore much higher estimates of recreational catch. The results of these years of “side-by-side” surveys were used to develop a calibration model to convert historic CHTS estimates to the scale of the new FES.

All recreational data used in the ERP models has been calibrated to the new FES scale, and the time series of biomass and *F* estimates used as input for some models for these species are from assessments that used the new calibrated MRIP data.

5.2 Atlantic Menhaden

The Atlantic menhaden commercial fishery has two major components, a purse-seine reduction sector that harvests fish for fish meal and oil and a bait sector that supplies bait to other commercial and recreational fisheries. Fishery-dependent data for the Atlantic menhaden purse-seine reduction fishery, including landings, lengths, weights, and ages, have been collected by the Beaufort Laboratory of the National Marine Fisheries Service since 1955. The fishery has changed over the time series from peak landings in the 1950s and several processing plants to lower landings, the implementation of a total allowable catch (TAC), and one remaining processing plant in recent years. Bait landings and biosampling data including lengths and ages were compiled by NOAA Fisheries historically, but have been housed and validated by the Atlantic Coastal Cooperative Statistical Program (ACCSP) since 1985. The Beaufort Laboratory does all the commercial ageing of Atlantic menhaden samples.

There has been a TAC for Atlantic menhaden in place since 2013. Landings in the reduction fishery are currently at their lowest levels in the time series. In contrast, bait landings have increased in recent years as demand has grown because of recent limitations in other species used as bait (e.g., Atlantic herring). In 2017, coastwide landings were comprised of 74% from the reduction fishery and 25% from the bait fishery. Recreational removals comprised 1% of the coastwide landings and are combined with the bait fishery landings for the assessment. Recreational removals are not well captured by MRIP; there is not a known directed recreational fishery for Atlantic menhaden, although they may be caught by recreational anglers for use as bait for other gamefish. A 100% mortality was applied to the reported live recreational releases, so that total recreational removals were equal to the sum of landings and live releases. Total removals have generally declined over time, from a high of 738 thousand mt in 1956 to a time series low of 169 thousand mt in 2013. Total removals rebounded slightly after that, with total removals in 2017 at 175 thousand metric tons (Figure 7).

5.3 Atlantic Herring

Fishery dependent data for Atlantic herring consists of catch and biological sampling for age, length, weight, and spawning condition/fecundity (NEFSC 2018a). Landings are derived from electronic logbooks reported by the harvesters and verified through dealer reports. At-sea observers and portside samples measure both discards and incidentally landed bycatch, respectively. Discards at-sea are generally low for the industrialized fishery for Atlantic herring. Biological samples are also taken from the fishery at the time of off-loading. These samples are processed for length, weight and later aged and staged. Resulting data are then available for the stock assessment process.

Total removals of Atlantic herring peaked at 478 mt in 1968, before declining to a time series low of 44.6 mt in 1983. Total removals were mostly stable from 1990 – 2010, averaging 114 mt, but have declined in recent years to 50.2 million metric tons in 2017 (Figure 8).

5.4 Striped Bass

Striped bass are a predominantly recreationally caught species, with recreational harvest and release mortality making up approximately 90% of total removals in recent years. It is assumed that 9% of striped bass that are released alive die as a result of being caught, so that total recreational removals are equal to the recreational harvest plus 9% of the recreational live releases. Live releases have accounted for 85 to 90% of the total recreational catch in most years, with release mortality comprising 40-50% of the total recreational removals. The size frequency of recreationally landed fish comes from MRIP and is supplemented with state programs such as volunteer angler logbook programs. Data on sizes of striped bass released alive come from state-specific sampling, volunteer angler logbook programs, and the American Littoral Society (ALS) volunteer angler tagging program.

For the commercial sector, strict quota monitoring is conducted by states through various state and federal dealer and fishermen reporting systems, and landings are compiled annually from those sources by state biologists. Biological data (e.g., length, weight, etc.) and age structures from commercial harvest are collected from a variety of gear types through state-specific port sampling programs. Harvest numbers are apportioned to age classes using length frequencies and age-length keys derived from biological sampling. Commercial discards were estimated using tag return data from commercial and recreational sectors; for the Chesapeake Bay and the Delaware Bay these estimates were scaled by estimates of discards from a short-term observer program in the Delaware Bay.

Total removals were low at the beginning of the assessment time series due to the poor condition of the stock and the restrictive management measures put in place to rebuild it (Figure 9). As the stock rebuilt and regulations were eased, removals increased from a low of 1,580 mt in 1987 to a high of 37,391 mt in 2013. Removals were relatively stable from 2003-2013, averaging around 34,000 mt, but began to decline after 2013. From 2014-2017, removals averaged 27,375 mt due to a combination of stock declines and management action.

5.5 Bluefish

Bluefish is a predominately recreational species, with recreational removals making up about 85-92% of the total removals. It is assumed that 15% of bluefish that are released alive die as a result of being caught, so that total recreational removals are equal to the recreational harvest plus 15% of the recreational live releases. The proportion of bluefish released alive has increased over the time series from about 20% in early years to about 65% in recent years. Recreational landings are sampled for length as part of the MRIP program. The MRIP length samples were used to expand recreational landings per half year. Recreational discards were characterized using lengths from bluefish tagged and released in the ALS volunteer tagging program, as well as information provided by volunteer angler programs in Rhode Island, Connecticut, and New Jersey.

Commercial landings data were queried from the ACCSP Data Warehouse, which houses commercial data from state and federal data collection programs, including dealer reports and harvester reports. Biological samples were collected from commercial fisheries by the NEFSC port sampling program and state programs in Virginia, North Carolina, and Florida. Commercial discards were assumed to be negligible.

Bluefish removals were highest at the beginning of the assessment time series, peaking at 84,200 mt in 1987; by 1993 landings had declined to 26,940 mt, and remained relatively stable after that, averaging 27,000 mt from 1996 – 2017 (Figure 10).

5.6 Spiny Dogfish

Commercial fishermen catch spiny dogfish using longlines, trawls, and purse seines. Fishermen target female spiny dogfish because the females grow larger than males and tend to school together. The commercial fishery supplies the European food fish markets that use spiny dogfish for fish and chips.

Spiny dogfish landings are reported in the stock assessment as a total from commercial, recreational, Canadian, and distant water landings, or Northwest Atlantic Fisheries Organization (NAFO) Areas 2-6 (Sosebee and Rago 2017). US landings include those from US and distant water commercial fisheries and recreational landings and discards were obtained from MRIP. Canadian and distant water landings were obtained from the NAFO catch statistics database (Sosebee and Rago 2017). Landings were variable but high in the 1970s and then decreased through the early 1980s. The National Marine Fisheries Service (NMFS) encouraged commercial fishermen to target the bountiful stocks of spiny dogfish in the 1980s and 1990s when stocks of other commercially valuable fish in the Northeast declined. Therefore, landings were high in the 1990s, peaking at 27.8 mt in 1996, and then in the late 1990s, landings declined (Figure 11). In 1998, NMFS determined that spiny dogfish were overfished and implemented stringent harvest restrictions in federal waters to allow the stock to rebound. After federal and state regulations were implemented in the early 2000s, landings declined to a low of 3.2 mt in 2003. As the stock began to improve, landings began to increase in the 2010s. In 2017, commercial landings were estimated at 11.1 mt (Figure 11). Commercial landings are comprised of about 98% female spiny dogfish (Sosebee and Rago 2017).

5.7 Weakfish

For weakfish, the proportion of removals coming from the recreational sector has increased over time, increasing from about 10% of total removals at the beginning of the time series to approximately 50% of total removals in recent years. It is assumed that 10% of weakfish that are released alive die as a result of being caught, so that total recreational removals are equal to the recreational harvest plus 10% of the recreational live releases. The proportion of weakfish released alive has increased over the time series from less than 10% in early years to more than 90% in recent years. Recreational landings are sampled for length as part of the MRIP program. The MRIP length samples were used to expand recreational landings per half year. Recreational discards were characterized using lengths from the MRIP sampling of released fish on headboat vessels; prior to that program, it was assumed that the length frequency of fish released alive was the same as the length frequency of harvested fish.

Weakfish commercial landings data came from state-specific harvest records collected through a mandatory reporting system where available, or from the NMFS commercial landings database. Estimates of commercial discards were developed from the Northeast Fishery Observer Program data. Biosamples were collected through state sampling programs, and pooled length frequencies were developed for sub-regions based on geographic location and commercial size limits. Florida landings for both the commercial and recreational sector were corrected for hybridization using the observed proportion “pure” weakfish in the catch from Tringali et al. (2011).

Weakfish landings have declined significantly over the assessment time series; total landings in 2017 were 391 mt, just 2% of their 1986 value of 19,515 mt (Figure 12).

6 ATLANTIC MENHADEN INDICES OF ABUNDANCE

6.1 Fishery-Independent Indices

6.1.1 Background of Analysis and Model Description

When several population abundance indices provide conflicting signals, hierarchical analysis can be used to estimate a single population trend. The abundance indices for Atlantic menhaden were combined into regional composite indices using hierarchical modeling as described in Conn (2009). This method assumes each index samples a relative abundance but that the abundance is subject to sampling and process errors. It can be used on surveys with different time series, but it does assume that indices are measuring the same relative abundance and that the surveys have similar selectivities. The Conn method was also used to combine individual abundance indices into regional indices in SEDAR 2015 and ASMFC 2017b.

6.1.2 Model Configuration and Results

The Atlantic Menhaden Stock Assessment Subcommittee (SAS) developed an Atlantic menhaden young-of-year (YOY) index from 16 fishery-independent surveys and three regional adult indices from various fishery-independent surveys: a northern adult index (NAD), a Mid-Atlantic adult index (MAD), and a southern adult index (SAD). Refer to the single-species benchmark (2019) for full methods for the indices of relative abundance in numbers to support the BAM and MSSCAA models.

The NAD hierarchical biomass index predicted variable abundance throughout the time series with notable peaks in 1999, 2002, and the mid-2010s (Figure 13). Despite the higher abundance occurring in 2014-2015, the final two years of the index (2016-2017) indicate a decreasing adult abundance. All three of the individual abundance indices used in the NAD indicated a declining abundance in the terminal years. The MAD hierarchical index predicted high abundance in the beginning of the time series followed by low abundance in the early 1990s (Figure 13). From 1985 until the Virginia’s Gill Net (VA GN) began in 1998, the MAD relied on only the Maryland Gill Net survey (MD GN) and thus there are larger errors associated with those years. The index then bounces around from the mid-1990s to the 2010s. Despite high abundance in 2014-2015, the final two years of the index (2016-2017) indicate a decreasing adult abundance just like the

NAD indicated. Both of the individual abundance indices used in the MAD indicated a declining abundance in the terminal years. The SAD hierarchical index predicted high abundance in 1990 followed by low abundance from 1991-2004, followed by an increase to a high in 2006 (Figure 13). The index is variable from 2006-2015 with a low abundance in 2016 and a slight uptick in the terminal year of 2017. All three of the individual abundance indices used in the SAD indicated an increasing or neutral abundance in the terminal year.

To develop biomass indices for the surplus production models, the length frequencies from the individual surveys were converted into weight frequencies using the time-invariant length-weight relationship developed for the single-species benchmark (SEDAR 2019). The individual GLM indices were converted into biomass using the weight frequencies and then combined regionally using the methods of Conn (2009). Biomass Conn indices were very similar in pattern to the Conn indices in numbers.

6.2 Fishery-Dependent Indices

The ERP WG developed two long-term indices of abundance for Atlantic menhaden: a commercial reduction fishery CPUE index (RCPUE index) and a commercial bait fishery catch per unit effort (CPUE) index, the Potomac River Fisheries Commission (PRFC) index. The Atlantic Menhaden SAS considered fishery dependent indices of abundance in past assessments, including the PRFC index, but did not use them in the single-species assessment due to concerns about the reliability of the index as a measure of relative abundance. These concerns included how to define a consistent unit of effort, the limited spatial scale (of the PRFC index), the potential for hyperstability (of the RCPUE index), and other factors. Although the WG acknowledged the SAS's concerns about these indices, the long time series and the contrast they provided, which the surplus production models required, outweighed the potential biases.

The two indices had similar trends since 1990, but showed differing trends from 1970-1990 (Figure 13). The ERP WG decided to use the RCPUE index for ERP model base runs because of its larger spatial coverage, its consistently recorded unit of effort, its known variance structure, support from supplemental analyses that showed relatively strong correlations with other sources of data, and the ability to standardize the data through explanatory covariates (week, factory, vessel size), among other factors. However, sensitivity analyses with the PRFC index were conducted.

6.2.1 Commercial Reduction Catch Per Unit Effort (RCPUE) Index

A long-term index of abundance spanning 1955-2017 was generated for Atlantic menhaden using catch and effort data from dealer reporting in the reduction fishery (RCPUE index). CPUE was defined as landings (1,000 t) per net tonnage-days fished to account for variability over time in fishing effort and size of fishing vessels used. An index of abundance (RCPUE) was generated by estimating the year effects of a lognormal generalized linear model that predicted CPUE as a function of year, week in year, and plant; week and plant were included in the model to account for changes in the location and number of reduction plants over time and seasonality of the fishery. A similar index using more detailed effort data contained in Captain's

Daily Fishing Reports spanning 1985-2017 was generated and found to be highly correlated (r = 0.92) with the long-term RCPUE index.

6.2.2 Potomac River Fishery Commission Commercial Bait Catch Per Unit Effort (PRFC) Index

A long-term index of abundance spanning 1964-2017 was generated for Atlantic menhaden using pound net landings and effort data collected by the Potomac River Fisheries Commission (PRFC). The PRFC index was calculated as annual ratios of total pound net landings (in mt) to total pound net days fished.

Landings with associated effort (pound net days fished) were available, but discontinuous (1976-1980 and 1988-2018). During 1964-1993, the PRFC required a license for each pound net and did not restrict number of pound net licenses sold. Since pound nets were expensive and labor intensive to fish, it was reasonable to assume that each licensee would maintain stable fishing practices and, as a result, number of licenses could approximate effort. When licenses were capped at 100 in 1993, this estimator may have stopped representing effort in the same manner as before the cap (fishermen may have bought more licenses than needed to keep from being excluded from fishing). Prior to the imposition of the cap, licenses had steadily fallen by half between 1985 and 1993 (to 72). After the cap was imposed, 100 licenses were issued every year; however, not all 100 licenses were necessarily fished.

Previous single-species stock assessments (ASMFC 2004, ASMFC 2012b) used a linear regression to fill missing years of effort. Recently, the PRFC obtained and computerized more detailed data on pound net landings and effort, which allowed index values to be calculated for 1964-1975 and 1981-1987 (A. C. Carpenter, PRFC, personal communication).

To generate estimates of pound net days fished (*DF*) for missing years (those with only license effort data), a linear regression was fitted to *DF* as a function of the number of licenses (*L*):

$$DF = 2794.5 + 19.214 \cdot L \tag{6.1}$$

which had an R^2 value of 0.505 and was significant at an α-level of 0.014 (n = 11).

Pound net days fished predicted by this equation were used to convert landings (in mt) per license to landings per pound net days fished for years without pound net days estimates. A trend was not evident for 1976 – 1978, so the regression intercept was used for pound net days fished for years prior to 1979. For all other years (1979 – 1993), the equation was used to estimate pound net days fished.

7 NON-MENHADEN INDICES OF ABUNDANCE

The single-species assessments for all of these species use multiple (often 5 or more) indices of relative abundance. In order to keep the multispecies models tractable, the ERP WG consulted with the other species' TCs to select the most representative subset of indices. The ERP WG limited the non-menhaden species to one index of recruitment and two age-0+ indices of abundance, with one additional age-0+ index chosen for a sensitivity run.

7.1 Atlantic Herring

The Atlantic herring TC recommended using the NEFSC fall bottom trawl survey as an index of age-1+ abundance. This survey catches Atlantic herring across age-classes but does miss some of the youngest Atlantic herring inshore in the GOM (NEFSC 2018a). This survey has been operational since 1963.

Because of the vessel change from the RV Albatross to the RV Bigelow in 2009, the fall index was separated in the most recent assessment. This results in two separate indices for Atlantic herring: Fall Albatross 1985-2008 and Fall Bigelow 2009-present (NEFSC 2018a).

The ASMFC Summer Shrimp survey was selected as a sensitivity run. The Summer Shrimp survey has operated with consistent gear and methodology in the Gulf of Maine since 1984. It uses a combination of fixed and stratified random stations. Although the survey targets northern shrimp, data for other species is also collected.

There is no dedicated YOY index for Atlantic herring.

The NEFSC Fall Albatross and Summer Shrimp surveys showed similar trends, increasing from lower levels at the beginning of the time series and showing peaks in the mid-1990s before declining again (Figure 8). The NEFSC Fall Bigelow has generally varied without trend since 2009 (Figure 8).

7.2 Striped Bass

For the recruitment index, the Striped Bass TC recommended the composite YOY index for the Chesapeake Bay. The composite index was developed from two separate but methodologically similar seine surveys conducted in the Maryland and Virginia waters of the Chesapeake Bay, combined into a single index using the Conn (2009) method. The index represents recruitment for the Chesapeake Bay stock, which is the major contributor to the coastal metapopulation of striped bass. The index showed several strong year classes in the late 1980s and early 1990s, a period of generally below average recruitment from the early 2000s to 2010, and strong year classes in 2011, 2014, and 2015 (Figure 9).

For age-1+ indices, the Striped Bass TC recommended the Connecticut Long Island Sound Trawl Survey (CT LISTS) and the MRIP CPUE index. Both of these indices have long time series with good contrast in the data, and represent the coastal migratory metapopulation of striped bass, unlike the spawning stock surveys, which represent individual stocks. The Maryland Spawning Stock Survey (MD SSN) was selected as a sensitivity run, as it represents the Chesapeake Bay stock and has a relatively long time series.

CT LISTS is a stratified random trawl survey that occurs in Long Island Sound; the fall component of the survey was used to develop the index. Length frequencies were converted to age composition information using regional age-length keys. The MRIP CPUE was developed from the raw intercept data collected by MRIP. Trip records were subset to trips that occurred in ocean waters from Virginia through Maine from May – October. Striped bass trips were identified using a guild approach as trips that caught either striped bass or another similar species. Similar species were identified on a state-by-state basis as the species with the highest Jaccard coefficient, which measures how often any given species is caught with striped bass compared to how often they are caught separately. For most states, bluefish or Atlantic mackerel (*Scomber scombrus*) were the most commonly co-encountered species. A negative binomial GLM was used to develop the index from the trip data. Recreational harvest-at-age for the ocean during those months were combined with the full recreational release-at-age numbers (i.e., not scaled by the discard mortality rate) to develop age structure information for this index. The MD SSN is a multi-panel gillnet survey that occurs on the spawning grounds in the Maryland portion of the Chesapeake Bay during the spawning season. For more details on survey methods and index calculations, see NEFSC (2019).

Both of these indices showed similar trends, starting out low at the beginning of the time series and increasing through the 1990s (Figure 9). They peaked around the early 2000s and have been gradually declining since. The MD SSN has varied without trend over that time period (Figure 9); however, it shows the same expansion of the age structure during the 1990s and the contraction in recent years that the CT LISTS and MRIP CPUE do.

7.3 Bluefish

For a recruitment index, the Bluefish TC recommended the composite YOY index developed from state seine surveys that are conducted in bays and estuaries from Virginia to New Hampshire, using the Conn (2009) method. The composite index showed years of strong and weak recruitment at the beginning of the time series, with less variability in more recent years (Figure 10).

For the age-0+ indices, the Bluefish TC recommended the North Carolina Pamlico Sound Independent Gillnet Survey (NC PSIGNS) and the MRIP CPUE. These are the only two bluefish indices that are not dominated by age-0 fish and are therefore able to provide information on population age structure. In addition, the MRIP CPUE has the longest time series and widest spatial extent of the indices used in the assessment. The TC recommended using the NEFSC Fall Bottom Trawl Survey conducted on the R/V Albatross (NEFSC Fall Albatross) as a sensitivity run, since it had the widest spatial extent of the fishery independent indices.

NC PSIGNS uses a stratified random sampling design, based on area and water depth, to deploy arrays of gillnets with different mesh sizes. Sampling is conducted from mid-February to mid-December, and all months are used in the index. Length frequency data were converted to age composition information with seasonal age-length keys. The MRIP CPUE was calculated from the raw intercept data collected by MRIP. The MRIP data were subset to directed bluefish trips; that is, trips where the angler caught bluefish or reported they were targeting bluefish. Trips from Florida to Maine from all months were included. A negative binomial GLM was used to develop the index from the trip data. MRIP harvest-at-age for the ocean during those months were combined with the full recreational release-at-age numbers (i.e., not scaled by the discard mortality rate) to develop age structure information for this index. The NEFSC has conducted a stratified random bottom trawl survey since 1963 from North Carolina into the Gulf of Maine; in 2009, the survey switched vessels from the R/V Albatross to the R/V Bigelow. This vessel change resulted in changes to the trawl gear and survey protocol. NEFSC fall inshore strata from Cape Hatteras, NC to Cape Cod, MA were used to develop separate indices for bluefish for the Albatross and Bigelow years. For more information on these indices, see NEFSC (2015).

All three indices showed similar trends: a slight decline from 1985 to 1995 then a slight increasing trend to 2005, after which the NC PSIGNS and MRIP CPUE have been mostly stable (Figure 10).

7.4 Spiny Dogfish

The NEFSC calculates a biomass estimate for spiny dogfish based on area swept from their spring bottom trawl survey (Figure 11). The index does not have a value for 2014 due to mechanical problems on the FSV Bigelow that delayed the spring bottom trawl and resulted in the loss of critical strata for the index. The time series indicates that biomass was lower in the late 1960s-1970s and then increased but was variable through the 1980s and 1990s. The index decreased to a low in 2004 and has increased but been variable since then.

7.5 Weakfish

The Weakfish TC recommended using the composite YOY index developed from state trawl surveys for juvenile finfish that occur in bays and estuaries from North Carolina to Rhode Island, using the Conn (2009) method. The composite YOY generally varied without a strong trend, being below average in the 1980s and most recent years, and above average from 1992-2006 (Figure 12).

The Weakfish TC noted that there were differences in trends between indices that occurred offshore and indices that were conducted inshore, with offshore indices being more variable and with weaker trends that were inconsistent with the inshore surveys. This may be due to mismatches between survey timing and inshore/offshore movements of weakfish in some years. Based on input from the Weakfish TC, the ERP WG decided to use the MRIP CPUE and the Delaware Bay 30’ Trawl Survey (DE 30ft Trawl) as the base run age-0+ indices, both of which are inshore indices, and the NC PSIGNS index as an inshore sensitivity run and the New Jersey Ocean Trawl (NJ OT) as an offshore sensitivity run.

The MRIP CPUE for this assessment was calculated from the raw intercept data collected by MRIP for states from North Carolina to New York. Weakfish trips were identified using a guild approach as trips that caught either weakfish or another similar species. Similar species were identified on a state-by-state basis as the species with the five highest Jaccard coefficients, which measures how often any given species is caught with striped bass compared to how often they are caught separately. For most states, Atlantic croaker (*Micropogonias undulates)*, spot, and summer flounder (*Paralichthys dentatus*) were the most commonly co-encountered species. A negative binomial GLM was used to develop the index from the trip data. MRIP harvest-at-age for the ocean during those months were combined with the full MRIP release-at-age numbers (i.e., not scaled by the discard mortality rate) to develop age structure information for this index. ALS volunteer tagging data were used as a proxy for the length frequencies of fish released alive for the period of years between the implementation of coastwide minimum size limits and the implementation of the MRIP at-sea headboat sampling. NC PSIGNS is described above for bluefish. The NJ OT is a stratified random trawl survey conducted five times per year (January, April, June, August and October) in nearshore ocean waters from the entrance of New York Harbor south, to the entrance of the Delaware Bay. A GLM-based index was derived using a negative binomial distribution of the August and October abundance data with mean depth and bottom salinity as the covariates. New Jersey's age length keys were applied to this survey's mean catch at length indices to derive an index-at-age. For more details on these indices, see ASMFC (2016).

The MRIP CPUE and the DE 30ft Trawl showed similar trends, increasing from the late 1980s through the mid-1990s before declining to low levels (Figure 12). For the MRIP CPUE, the peak in the mid-1990s never reached the levels of the index in the early 1980s. The NC PSIGNS index showed a similar declining trend from the start of its time series in 2001 through 2017 (Figure 12). The NJ OT fluctuated without a general trend but did show a similar peak in 1994 (time series high) and 1995, followed by low values for most of the rest of the time series with smaller peaks in 2000, 2004 and 2011 (Figure 12).

8 SINGLE-SPECIES ASSESSMENTS AND STOCK STATUS

For the key predator and prey species, the most recent stock assessments were used to provide estimates of population size, fishing mortality, and reference points. For species where the terminal year of the most recent published stock assessment was prior to 2017 (namely, bluefish and weakfish), preliminary assessment updates were used to provide biomass estimates on the correct scale; the values from those assessment updates may not match the final assessment update values used in management.

The single-species assessments use target and threshold values based on spawning stock biomass, but the EwE models use total biomass. In addition, the scale of biomass and fishing mortality are not the same between the EwE models and the single-species models, so direct comparisons with the target and threshold values are not possible. To address this issue, spawning stock biomass targets and thresholds were converted to total biomass targets and thresholds, and the percent change between terminal year *B* and *F* and target and threshold *B* and *F* was calculated so that the EwE model results could be scaled appropriately (see also the sections on the EwE models for why this was necessary and how these values were used). Reference points, *B* equivalents, and *B* and *F* scalars are shown in Table 3 and Table 4.

8.1 Atlantic Menhaden

Atlantic menhaden are assessed with a statistical catch-at-age model, the Beaufort Assessment Model (BAM). According to the 2019 benchmark stock assessment (see single species assessment document), Atlantic menhaden were not overfished and overfishing was not occurring in 2017, the terminal year of the assessment. The F_{TARGET} was defined as the median of the geometric mean F on ages 2-4 from 1960 – 2012, and the $F_{\mathrm{THRESHOLD}}$ was the maximum value of the geometric mean F on ages 2-4, over that time series. The overfished determination is based on total population fecundity. The spawning potential ratio associated with the F_{TARGET} and $F_{\mathrm{THRESHOLD}}$ are converted into total fecundity values to represent the FEC_{TARGET} and $FEC_{\mathrm{THRESHOLD}}$, respectively.

Total age-1+ biomass has fluctuated over time from an estimated high of over 6.8 million mt in 1959 to a low of 1.4 million mt in 1973 (Figure 14). Biomass was estimated to have been largest during the late-1950s and late-2010s, with lows occurring during the 1960s, 1970s, and 1980s. From 1980 to the present, biomass has increased in trend. Biomass likely increased at a faster rate than abundance because of the increase in the number of older fish at age and an increase in weight-at-age. Biomass in 2017 was 4.7 million mt.

Population fecundity (i.e., total egg production) was the measure of reproductive output used to assess overfished status. Population fecundity (*FEC*, number of maturing ova) was highest in the early 1960s and from the 1990s to the present (Figure 14). The largest values of population fecundity were in 1955, 1961, and 2012. Throughout the time series, age-2 and age-3 fish have produced most of the total estimated number of eggs spawned annually. Fecundity in 2017 was estimated at 2.6 quadrillion eggs, above both the threshold (1.46 quadrillion eggs) and the target (1.94 quadrillion eggs).

Fishing mortality rate over time was reported as the geometric mean fishing mortality rate at ages-2 to -4 to account for changes in selectivity over time. Geometric mean fishing mortality rate was highest in the 1970s and 1980s and has been declining since approximately 1990 (Figure 14). F in 2017 (0.11) was below both the $F_{\mathrm{THRESHOLD}}$ (0.60) and the F_{TARGET} (0.22).

8.2 Atlantic Herring

Atlantic herring are assessed with a statistical catch-at-age model, the ASAP program from the NEFSC Toolbox. According to the 2018 benchmark stock assessment (NEFSC 2018a), Atlantic herring were not overfished and overfishing was not occurring in 2017, the terminal year of the assessment. The F threshold is the F_{MSY} proxy, $F_{40\%SPR}$. The SSB target (the B_{MSY} proxy) is calculated by using AgePro to project the population forward under $F=F_{40\%SPR}$ until it stabilizes, with recruitment drawn from the observed time series; the long-term equilibrium SSB under these conditions is the SSB target. The SSB threshold is 50% of the SSB target. The ratio of SSB to age-1+ biomass over the entire assessment time-series was used to convert the SSB targets and thresholds to age-1+ biomass targets and thresholds for the ERP models that use total biomass.

Total age-1+ biomass ranged from a peak of 2,035,800 mt in 1967 to a low of 169,860 mt in 1982 (Figure 15). Total biomass in 2017 was 239,470 mt. SSB showed a similar pattern, ranging from a high of 1,352,700 mt in 1967 to a low of 53,084 mt in 1982 (Figure 15). SSB in 2017 was 141,473 mt, above the SSB threshold of 94,500 mt.

F was reported as the average *F* over ages 7 and 8, as those ages are fully selected by the mobile gear fishery, which has accounted for the majority of total landings since 1986. *F* ranged from a low 0.13 in 1965 to a high of 1.04 in 1975 (Figure 15). *F* in 2017 equaled 0.45, below the *F* threshold of 0.51.

8.3 Striped Bass

Striped bass are assessed with a statistical catch-at-age (SCA) model. According to the 2018 benchmark stock assessment (NEFSC 2019), Atlantic striped bass were overfished and overfishing was occurring in 2017, the terminal year of the assessment. The reference points currently used for management are based on the 1995 estimate of female SSB. The 1995 female SSB is used as the SSB threshold because many stock characteristics (such as an expanded age structure) were reached by this year and the stock was declared recovered. The SSB target is 125% of the SSB threshold. The *F* target and *F* threshold are defined as the *F* needed to maintain the population at the SSB target and SSB threshold respectively. The estimate of age-2+ biomass in 1995 from the single species model was used as the $B_{THRESHOLD}$ proxy for the ERP models that use total biomass, and 125% of that value was defined as the B_{TARGET} proxy.

Total age-1+ biomass of striped bass increased from a low of 38,149 mt in 1982 and increased to a peak of 334,661 mt in 1999 before beginning to decline (Figure 16). Total biomass was 173,663 mt in 2017. Female SSB started out at low levels and increased steadily through the late-1980s and 1990s, peaking later than total biomass at 113,602 mt in 2003 before beginning to gradually decline; the decline became sharper in 2012 (Figure 16). Female SSB was estimated at 68,476 mt in 2017, below the SSB threshold of 91,436 mt and below the SSB target of 114,295 mt.

Total *F* has been increasing for both the ocean fleet and the Chesapeake Bay fleet since 1990. Total *F* in 2017 was 0.31, above both the *F* threshold of 0.24 and the *F* target of 0.20 (Figure 16).

8.4 Bluefish

Bluefish are assessed with a statistical catch-at-age model, the ASAP program from the NEFSC Toolbox. Bluefish assessment data used for this assessment was from a preliminary assessment update with data through 2017; for the final values, see NEFSC 2019b. The trends are the same, with some small differences in magnitude between the preliminary update and the final 2019 update. In 2017, the preliminary assessment update indicated bluefish were overfished and overfishing was occurring. The SSB target (the B_{MSY} proxy) is calculated by using AgePro to project the population forward under F=$F_{THRESHOLD}$ until it stabilizes, with recruitment drawn from the observed time series; the long-term equilibrium SSB under these conditions is the SSB target. The SSB threshold is 50% of the SSB target. The equilibrium age-1+ biomass from this projection was used as the B_{TARGET} proxy for the ERP models that use total biomass, and 50% of that value was the $B_{THRESHOLD}$ proxy.

Total age-1+ biomass declined from the beginning of the time series until the mid-1990s before beginning to increase; total biomass has trended downward in recent years (Figure 17). The preliminary estimate of total biomass in 2017 was 117,107 mt. SSB has shown a similar trend, with the preliminary estimate of SSB in 2017 at 107,282 mt, below the SSB threshold (Figure 17).

F is reported as *F* at age 2, the age of full selectivity for bluefish. *F* declined over the time series until 2008, when it began to increase (Figure 17). *F* has been above the *F* threshold for the entire time series. The preliminary estimate of *F* in 2017 was 0.34, above the *F* threshold.

8.5 Spiny Dogfish

Spiny dogfish are assessed using a swept-area biomass estimate derived from the NEFSC Spring Bottom Trawl Survey. Biological reference points are derived from a stock-recruitment relationship derived from the survey data and a population projection model. Based on the 2018 updated, spiny dogfish were not overfished and overfishing was not occurring in 2018 (NEFSC 2018b). The SSB target (B_{MSY} proxy) is SSB_{MAX}, the biomass of female spiny dogfish greater than 80cm that results in the maximum projected recruitment based on a Ricker stock-recruitment model derived from NEFSC trawl survey data. The SSB threshold is 50% of the SSB target. The SSB target is converted from the survey SSB CPUE scale (biomass-per-tow of female spiny dogfish greater than 80cm) to total swept area SSB. The ratio of SSB per tow to total biomass-per-tow over the entire time series was used to convert the female SSB target and threshold to a total biomass target and threshold for the ERP models that use total biomass.

Estimates of total biomass have been variable over the time series, showing an increase from the late 1970s to the early 1990s before declining (Figure 18). Total biomass has generally been increasing since 2004, but 2017 was 414,900 mt the lowest value seen in the last 10 years. Survey data by sex are not available prior to 1980, so the female SSB time series is more limited. Female SSB is reported as the three year average of the annual survey estimates, so the trend is smoother, but generally similar to the total biomass trend: declining from the early 1990s to the early 2000s, then increasing again (Figure 18). The year-specific estimate of female SSB in 2017 was 24,400 mt, the lowest in the time series. However, the indices for all size and sex classes decreased, likely indicating a year specific availability issue rather than a major decline in biomass. The 3-year average of the female swept area SSB was 112,000 mt in 2017, lower than in recent years but still above the SSB threshold of 79,644 mt but below the SSB target of 159,288 mt.

F is reported as female catch on exploitable female biomass; males make up a tiny component of the overall fishery. Observer estimates of commercial discards are not available prior to 1990, so the time series of *F* is shorter than the total biomass and SSB time series. *F* has generally been declining since the mid-1990s, but has been increasing in recent years (Figure 18). *F* was 0.20 in 2017, below the *F* threshold of 0.24.

8.6 Weakfish

Weakfish are assessed using a Bayesian statistical catch-at-age model that estimates a time-varying natural mortality rate. Weakfish were found to be depleted in 2015 with total mortality above the *Z* threshold, based on the 2016 benchmark assessment (ASMFC 2016). Biological reference points for total mortality were developed using a SPR-based approach with natural mortality set at the time-series average estimated by the Bayesian model. The SSB threshold was developed by projecting the population forward under average *M* and no fishing mortality. The SSB threshold was defined as 30% of that unfished SSB; 30% of unfished age-1+ biomass was used as the proxy biomass threshold for the ERP models that used total biomass. A preliminary assessment update was conducted in a maximum likelihood framework model (ASAP, from the NEFSC Toolbox), using the previous time-varying estimates of *M*, in order to incorporate the new MRIP estimates of recreational catch. The overall trend in *F* and SSB from the preliminary update was similar to the benchmark assessment trends, but the scale was somewhat different due to the higher recreational catch estimates.

The preliminary update indicates that total age-1+ biomass has declined since the beginning of the time series, from a high of 33,457 mt in 1986 to a low of 1,634 mt in 2014 (Figure 19). The population rebounded somewhat in the mid-1990s, but has been steadily declining since then. The preliminary estimate of total biomass in 2017 was 3,210 mt, an increase since 2014, but still well below the time-series mean. Spawning stock biomass showed very similar trends to age-1+ biomass, since weakfish are 90% mature at age 1 (Figure 19). The preliminary estimate of SSB in 2017 was 3,114 mt, below the SSB threshold of 8,815 mt.

Full *F* for weakfish declined through the early 1990s before increasing again; *F* spiked in 2008, but has been below average since then (Figure 19). *F* in 2017 was 0.23, below the proxy $F_{\text{THRESHOLD}}$ of 0.93.

9 BEAUFORT ASSESSMENT MODEL (BAM) DESCRIPTION AND CONFIGURATION

The Beaufort Assessment Model (BAM) has been used to assess Atlantic menhaden since 2010 (SEDAR 2010; SEDAR 2015). BAM is a statistical catch-at-age model that estimates population size-at-age and recruitment, using 1955 as the start year, and then projects the population forward in time. The model estimates trends in the population, including abundance-at-age, recruitment, spawning stock biomass, egg production, and fishing mortality rates. BAM was configured to be a fleets-as-areas model with each of the fleets broken into areas to reflect differences along the coast.

BAM estimates of age-1+ biomass have fluctuated over time from an estimated high of over 6,794,000 mt in 1959 to a low of 1,379,000 mt in 1973. From 1980 to the present, biomass has been increasing in trend. Population fecundity (number of maturing ova, used as the metric for spawning capacity of the stock) was highest in the early 1960s and from the 1990s to the present. Age-0 recruits of Atlantic menhaden were highest during the 1950s. An extremely large year class was also predicted for 1958. Recruitment has appeared to be rather stable during the late 1970s to the present. Fishing mortality rate over time was reported as the geometric mean fishing mortality rate at ages-2 to -4, the dominant age-classes in the fishery, to account for differences in selectivity patterns over time. Geometric mean fishing mortality rate was highest in the 1970s and 1980s and has been declining since approximately 1990.

For more detailed information on the BAM configuration and results, see the single-species assessment report.

10 SURPLUS PRODUCTION MODEL WITH TIME-VARYING *r* (SPMTVR) (SUPPORTING)

An alternative to explicit incorporation of ecosystem processes in stock assessments is the use of single species models that implicitly estimate changes with time-varying parameters. Age- and length-structured stock assessments often implicitly account for anthropogenic and environmental effects on stock dynamics through the estimation of time-varying parameters such as natural mortality, growth, selectivity, and catchability (Fu and Quinn II 2000; Wilberg et al. 2010; Wilberg et al. 2011; Methot and Wetzel 2013; Nielsen and Berg 2014; Xu et al. 2019). In situations with less reliable data, the use of surplus production models with time-varying parameters may provide an alternative to explicit modeling of ecosystem drivers (Nesslage and Wilberg 2012, 2019). Using only a time series of catch and at least one index of abundance, surplus production models estimate stock biomass, carrying capacity, and the population's intrinsic growth rate (Prager 1994). The intrinsic growth rate of the population encompasses the growth response of the stock to its surrounding ecosystem, including mortality processes such as predation and recruitment processes affected by environmental conditions. If allowed to vary over time, surplus production model parameters can implicitly capture the effects of shifting drivers on fish stocks without having to explicitly model the underlying mechanisms, especially when time series are of sufficient length to cover periods of major anthropogenic or environmental change are available (Nesslage and Wilberg 2019).

A surplus production model with a time-varying intrinsic growth rate (SPMTVr) was fitted to Atlantic menhaden catch and indices of Atlantic menhaden biomass to generate annual estimates of fishery exploitation rate and total Atlantic menhaden biomass. The SPMTVr used is a modified Schaefer surplus production model with observation error (Polacheck et al. 1993; Quinn and Deriso 1999), which follows a logistic population growth process,

$$\hat{B}_{t+1} = \hat{B}_t + \hat{r}_t \hat{B}_t \left(1 - \frac{\hat{B}_t}{\hat{K}}\right) - C_t, \tag{10.1}$$

such that $\hat{B}_t$ is estimated Atlantic menhaden biomass at time t, $\hat{K}$ is carrying capacity, C_t is catch at time t, and $\hat{r}$ is the intrinsic population growth rate estimated annually according to a random walk on the log scale:

$$\log_e \hat{r}_{t+1} = \log_e \hat{r}_t + \omega_t, \tag{10.2}$$

with annual deviations, ω_t, from a normal distribution with a mean of zero and an SD of 0.1. A random walk was selected to generate annual deviations in *r* because random walk estimation processes have been shown to perform well under a variety of circumstances with trends over time, whereas other forms of annual deviations such as white noise are more limited in their application (Wilberg and Bence 2006).

The estimated index of biomass, $\hat{I}_t$, was the product of catchability and biomass,

$$\hat{I}_t = \hat{q}\hat{B}_t, \tag{10.3}$$

where $\hat{q}$ was survey catchability. Total catch was assumed known without error. Parameter estimates were obtained by minimizing the concentrated negative log likelihood function,

$$-LL_1 = \frac{n}{2}\log_e\left(\sum\left(\log_e(I_t) - \log_e\left(\hat{I}_t\right)\right)^2\right). \tag{10.4}$$

Multiplicative lognormal observation errors were assumed for the index of biomass. A normal (on the loge scale) prior, LL_2, was included on initial biomass,

$$-LL_2 = 0.5\left(\frac{\log_e(B_{t=1}) - \log_e(prior)}{sd}\right)^2 \tag{10.5}$$

such that $\hat{B}_{t=1}$ was the estimated biomass in the first year, prior was the prior point estimate, and *sd* was the standard deviation of the lognormal prior distribution. An additional term, $-LL_3$,

$$-LL_3 = \frac{1}{2\sigma^2}\sum \omega_t^2, \tag{10.6}$$

was included to account for the annual random walk deviations, such that annual deviations were normally distributed with a mean of zero and a known variance.

The SPMTVr estimates dynamic, MSY-based reference points that reflect changing stock productivity (Nesslage and Wilberg 2019). A dynamic overfishing threshold was produced by calculating 75% of annual U_{MSY} estimates (calculated as $\frac{r_t}{2}$). 75% of U_{MSY} was selected because it has been suggested as a general overfishing limit for forage species (Pikitch et al. 2012). Use of 75% of U_{MSY} in the terminal year as a reference point for management assumes that the *r* in the terminal year will continue (i.e., that there is substantial temporal autocorrelation in the population productivity). Biomass at 50% of B_{MSY} (B_{MSY} calculated as $\frac{K}{2}$) was defined as a potential overfished threshold for Atlantic menhaden given its common use in US federal fisheries management.

The SPMTVr was extensively simulation tested using a linked, age-structured, predator-prey model of Atlantic menhaden and striped bass (Nesslage and Wilberg 2019). The SPMTVr generally produced more accurate, less variable estimates of exploitation rate and biomass than traditional Schaefer surplus production models with static intrinsic growth.

10.1 Treatment of Indices & Input Data

The base model configuration of the SPMTVr included total annual landings (1,000t) during 1957-2017 and three adult indices of abundance (RCPUE, NAD, and MAD).

10.2 Parameterization

The intrinsic population growth rate in 1957 was estimated and annual deviations from that rate in each subsequent year. Other estimated parameters included catchability of each of the three indices of abundance, initial biomass, and carrying capacity. All estimated parameters were bounded (Table 5). Estimates of mean fishing mortality across the species' range (F = 0.55yr^{-1}) and natural mortality (M = 1.18yr^{-1}) generated from an historical tagging study conducted in the late 1960s (Liljestrand et al. 2019) were used along with a reported catch of 630,300 t in 1957 to estimate a starting value for initial biomass of 2,424,000 t. In addition, a normal prior was placed on the logarithm of initial biomass with a mean equal to the estimate of biomass in 1957 and a standard deviation of 0.15.

The starting value for the RCPUE index CV was assumed to be 0.5. For fishery-independent indices, CVs were assumed to be the time series average CV for the NAD (CV = 0.71) and MAD (CV = 0.70). Log-scale standard deviations for each index were adjusted iteratively to determine the final weights applied to each index (Francis 2011). The SD of the annual r deviations, ω_t, was set at 0.1 based on previous research in the use of random walk processes (Wilberg and Bence 2006; Nesslage and Wilberg 2012); however, model performance with random walk SDs of 0.05 and 0.2 were explored as well (Section 10.4.2).

10.3 Results

10.3.1 Diagnostics

The base run of the SPMTVr converged on a stable solution and parameter estimates did not approach bounds. The SPMTVr fit overall trends in abundance indices but overestimated RCPUE at the beginning of the time series and underestimated RCPUE at the end of the time series. Also, the model underestimated MAD at the beginning of the time series (Figure 20).

10.3.2 Population Estimates

The model estimated that biomass in 1957 (2,182,820 t) was near carrying capacity (2,182,790 t). The estimated trend in population intrinsic growth rate ranged from values of 0.76 to 0.88 at the beginning of the time series then dropped to values in the range of 0.55 to 0.68 from the 1990s to present (Figure 21). Estimated biomass declined sharply in the late 1950s to mid-1960s then increased through the end of the time series with a small period of decline in the early 1990s (Figure 22). Estimated exploitation rate increased through the 1950s, then largely declined from the 1960s through the end of the time series (Figure 23).

The SPMTVr produced a static biomass reference point (50% of B_{MSY} = 546,000 t), suggesting the stock was overfished from 1963-1969 but has remained above that reference point for the remainder of the time series (Figure 22). The model also produced a dynamic exploitation rate reference point (75% of annual U_{MSY}) that ranged from 0.21-0.33 yr^{-1}, suggesting that the exploitation rate exceeded 75% U_{MSY} prior to mid-1980s but that the stock was not experiencing overfishing from 1999 onward (Figure 23). The model estimated a time-varying TAC (Figure 24) with a 2017 estimate of 443,662 t.

10.3.3 Uncertainty

CVs of model parameter estimates based on asymptotic standard errors are reported in Table 5.

10.4 Sensitivity Analyses

Sensitivity analyses were conducted to examine the effects of model start year, alternative model configuration (alternate starting values for B_{1957} and the SD of r deviations), and alternative abundance indices (Figure 25 - Figure 26). In summary, all model configurations agreed that the stock was not overfished and was not experiencing overfishing in the terminal year of the model (Figure 27 top); however, the models differed substantially in estimated stock status trajectory over the time series (Figure 27 bottom).

10.4.1 Sensitivity to Input Data

Model sensitivity to the exclusion of early years with relatively high landings was examined by starting the model in 1964 instead of 1957. This alternative model estimated the magnitude of r was much lower (0.3-0.4), but r declined over time as in the base model (Figure 21). Biomass estimates were similar in trend but about double the magnitude (Figure 25). Similar to the base model, exploitation rate exhibited a decline over time since 1964, but was approximately half the magnitude (Figure 26).

Model sensitivity to an alternate primary index of abundance was also examined by exchanging the RCPUE with the PRFC fishery-dependent index and starting the model in 1964. This alternative model estimated a large spike in r in the 1970s that was not as pronounced in the RCPUE-based model, indicating a period of very high productivity during that time that is not evident in the RCPUE index (Figure 21). Use of the PRFC index resulted in estimates of biomass and exploitation rate which were largely similar to that of the base run with the exception of the 1970s during which biomass doubled and exploitation rate halved (Figure 25 - Figure 26).

A sensitivity run was also conducted in which the MARMAP/EcoMon ichthyoplankton indices of spawning biomass were added; however, the model did not exhibit good convergence criteria (results not shown).

10.4.2 Sensitivity to Configuration

Sensitivity of the base model to the SD of annual r deviations, ω_t, was examined using lower (0.05) and higher (0.2) values. Estimates of biomass and exploitation rate were largely similar to the base model despite an expected slight flattening in the trend in r with a lower SD (0.05) and slightly more exaggerated trend in r with a higher SD (0.2).

Sensitivity to the starting value and mean of the prior on initial biomass was also examined. First, use of a likelihood penalty was explored in which exploitation rates in 1967-1969 were penalized for straying too far from the estimated values from an historical tagging study (Liljestrand et al. 2019). This model did not exhibit good convergence criteria, but produced similar estimates to that of another alternative model in which the starting value and mean of the prior on initial biomass was set equal to the BAM estimate of biomass in 1957 (Figure 21, Figure 25, Figure 26 & Figure 27). This model estimated very low, declining exploitation rates and high, increasing biomass compared with all other model configurations.

11 STEELE-HENDERSON SURPLUS PRODUCTION MODEL (SUPPORTING)

Steele-Henderson models are surplus production models with additional sigmoidal type III predation functions that estimate predation losses from one or more predators (Collie and Spencer 1993). They quantify the extent that modeled predators and fishing influence a prey species. When applied generally, the Steele-Henderson model reproduced rapid shifts in productivity exhibited by marine populations (regime shifts; Steele and Henderson 1984; Spencer 1997). Steele-Henderson models have been used to explore the role of predation on management of haddock (Spencer and Collie 1997), weakfish (NEFSC 2009), and Atlantic menhaden (Crecco 2010; Uphoff and Sharov 2018).

Steele-Henderson models represent an increase in mechanistic specificity over the time-varying *r* surplus production model: where the SPMTVr model did not specify a cause for time-varying productivity, the Steele-Henderson models assume that changes in productivity beyond what a traditional surplus production model would predict are driven by predator biomass. A Steele-Henderson model has the same data requirements as a surplus production model, plus it needs predator biomass estimates or indices to generate estimates of predation losses through a type III functional response (Collie and Spencer 1993; Crecco 2010). A Steele-Henderson model could be considered a "minimal realistic model" and the key feature of this approach is that only predators likely to have important impacts on the prey of interest are considered (Punt and Butterworth 1995; Yodzis 2001).

The Haddon (2001) version of a Schaefer surplus production model was adapted to the Steele-Henderson model formulation. An observation error model was used that assumed all residual errors were in the index observations and the equation used to describe the time-series was deterministic and without error (Haddon 2001). Biomass dynamics of Atlantic menhaden with losses from harvest and major predators were described by the following discrete time-step equation:

$$B_t = B_{t-1} + r \cdot B_{t-1}\left(1 - \frac{B_{t-1}}{K}\right) - H_{t-1} - \sum_j D_{j,t-1} + \varepsilon \qquad (11.1)$$

where B_t was age-1+ Atlantic menhaden biomass in year t; B_{t-1} = age-1+ biomass in the previous year, r = intrinsic rate of population increase; K = carrying capacity (unfished biomass); H_{t-1} = harvest in the previous year; $\sum D_{j,t-1}$ = the sum of estimated predation losses from predators in the previous year (estimated for each predator j by Equation 11.2, below); and ε = observation error (Collie and Spencer 1993; Spencer and Collie 1995). Initial biomass was estimated directly for the starting year as a separate parameter and then projected forward by the Steele-Henderson model. A fishing-only version of the model (i.e., traditional Schaefer biomass dynamic model) was created from equation 11.1 by excluding predation loss terms ($\sum D_{j,t-1}$).

Annual consumption of Atlantic menhaden biomass by each candidate predator in the Steele-Henderson model was estimated by a type III functional response as

$$D_{j,t-1} = \frac{d_j \cdot P_{j,t-1} \cdot (B_{t-1})^2}{A_j^2 + (B_{t-1})^2} \tag{11.2}$$

where d_j was estimated maximum per biomass consumption for predator j; $P_{j,t-1}$ was predator j biomass at time t-1; A_j was estimated Atlantic menhaden biomass where predator j satiation begins (Collie and Spencer 1993; Spencer and Collie 1995) and B_{t-1} represented age-1+ Atlantic menhaden biomass. Predator biomass was an input and the remaining three terms were estimated by the model.

This configuration of the Steele-Henderson model was tested on the same simulated dataset used to evaluate the time-varying r model. See Appendix B for more a detailed description of the Steele-Henderson model configuration and results.

11.1 Treatment of Indices & Input Data

The base model configuration of the Steele-Henderson included three age 1+ Atlantic menhaden biomass indices: fishery-dependent RCPUE (1985-2017), and fishery-independent MAD (1985-2017) and NAD (1990-2017). The MAD and NAD indices were scaled into RCPUE units using a ratio of averages approach based on years in common (1990-2017).

The Atlantic menhaden single-species assessment time-series of landings in weight was used to characterize removals. Estimates of biomass from the most recent single-species assessments (see Section 8) for the candidate predators were used to characterize predator population trends. Biomass for each predator was defined as the sum of biomass in each age or size class capable of eating age-1+ Atlantic menhaden. Based on an examination of diet length composition data by Uphoff and Sharov (2018), this was age-3+ for striped bass and age-1+ for bluefish; for spiny dogfish, it was sizes 36 cm+ (Scharf et al. 2000).

11.2 Parameterization

Model Parameters

The Steele-Henderson model was implemented in an Excel spreadsheet, and a genetic algorithm plug-in (Evolver; Palisade Corporation 2010) was used to estimate model parameters that minimized the difference between the observed and predicted indices of relative abundance for Atlantic menhaden:

$$\sum_{i,t} \left(\ln\left(I_{i,t}\right) - \ln\left(q_i \cdot B_t\right) \right)^2 \tag{11.3}$$

where $I_{i,t}$ is the value of index *i* in year *t*, q_i is a catchability coefficient for index *i*, and B_t is the age-1+ biomass of Atlantic menhaden in year *t*.

The genetic algorithm used by the Evolver software continuously introduces novel parameter values (i.e., "mutations") during the model fitting procedure. As such, model optimization was concluded after a set time limit (3 minutes), as opposed to numerical convergence criteria. If the progress optimization summary graph indicated the sums of squares converged on an asymptote, the run was used. If the graph indicated it was not reached, then another run of 3-minutes was made and progress was evaluated again. None of the runs required more than 6 minutes for the sums of squares to converge on an asymptote.

The model fitting algorithm required bounds for each parameter (Palisade Corporation 2016) and the ranges used were broad. Parameter *r* varied from 0.1 (very low) to 3.0 (a value associated with chaotic behavior of populations described by logistic equations; May 1974). The range of *K* fell between 100,000 and 10,000,000 mt, a range that fell below lowest observed landings to about 23-times the highest landings. Initial biomass ranged from 50,000 to 2,500,000 mt. The same ranges of estimates of *d* (0 – 17-times predator weight) for striped bass and bluefish estimated by Uphoff and Sharov (2018) were used for the three candidate predators. The range in parameter *A* was set equal to the range for *K*. Mid-range values were used as starting values for all models, excluding sensitivity analyses (described below).

Akaike information criterion for small sample sizes (AICc) were used to evaluate fishing-only and Steele-Henderson models with different predators that related changes in Atlantic menhaden biomass to fishing alone or to fishing and predation (Burnham and Anderson 2001)

Predator Selection

The ERP focal species of striped bass, bluefish, and spiny dogfish were screened for consideration as major predators using correlation analyses of Atlantic menhaden indices (RCPUE, MAD, and NAD) and predator biomass estimates from single-species assessments. Correlation analysis provided weak evidence of potential predator-prey interactions, and Steele-Henderson models were developed with each predator separately, and with combinations of predators. The fit of the index time-series and the AIC_c from the Steele-Henderson models with different predators and the fishing-only surplus production model were used to determine which predators to include in the base model, along with evaluation of the magnitude of M_2 from the Steele-Henderson model. Striped bass was determined to be the sole major predator for the base Steele-Henderson model based on these criteria (Table 6).

Estimates of *ad libitum* consumption of prey at optimal temperature as grams of prey per gram of striped bass per day derived from Hartman and Brandt (1995a; 1995b) bioenergetics models (C_{MAX}) provided a means to judge a maximum value for parameter *d* for striped bass in the initial parameterization of the Steele-Henderson model.

Reference Points

Moustahfid et al. (2009a) explored the use of biomass dynamic models that included predation losses and applied the concept of maximum useable production (MUP; Overholtz et al. 2008; Moustahfid et al. 2009b) instead of maximum sustainable yield (MSY). MUP reference points were generated for Steele-Henderson models using the formula developed by Moustahfid et al. (2009a):

$$B_{MUP} = \frac{K}{2} \tag{11.4}$$

MUP represents the surplus production available to modeled predators and the fishery. The surplus production available to the fishery (*SF*) can be partitioned out as maximum usable production minus recent average predator consumption:

$$SF = MUP - D \tag{11.5}$$

Instantaneous annual fishing mortality at MUP (F_{MUP}) was estimated as:

$$F_{MUP} = \frac{SF}{B_{MUP}} \tag{11.6}$$

The Steele-Henderson model also calculated *F* and time-varying natural mortality from modeled predators (M_2) based on annual harvest and consumption:

$$F_t = \frac{H_{t-1}}{\frac{1}{2}(B_{t-1}+B_t)} \tag{11.7}$$

$$M_{2t} = \frac{D_{t-1}}{\frac{1}{2}(B_{t-1}+B_t)} \tag{11.8}$$

A time-varying total mortality (Z_2) could be calculated from *F* and M_2:

$$Z_{2t} = F_t + M_{2t} \tag{11.9}$$

Estimates of Z_2 / Z_{MUP} and F / F_{MUP} greater than 1.0 would exceed the mortality thresholds, while B / B_{MUP} ratios less than one would indicate the stock is overfished.

Patterson (1992) established a general relationship of biomass of exploited small pelagic fishes to F / Z and proposed that F / Z higher than 0.4 would lead to declines in stock size, so for this analysis, 0.4 was used as a threshold to evaluate F / Z_2.

Direct feedback from prey to predator is not a feature of a Steele-Henderson model and an empirical approach was employed to develop a threshold based on major predator condition. Indicators of condition were not routinely estimated for major predators, so annual weights-at-age were used as a condition metric for major predators assessed by catch-at-age models (bluefish and striped bass). Changes in striped bass weight-at-age may have been a coarse indicator of condition since fasting striped bass replace lipids (the energy currency in marine fish; Rose and O'Driscoll 2002) with water in a linear fashion (Jacobs et al. 2013).

Correlation analysis (Pearson correlation coefficients, ρ; $P \leq 0.05$) was used to estimate strength of associations of D_t / P_t estimated by the base Steele-Henderson model with weight-at-age in the same year, and one, two, and three years before (i.e., an immediate response in weight to feeding vs. lagged responses). Correlations with weight-at-age were considered biologically significant if they occurred over continuous blocks of ages rather than sporadically.

If a major predator had a block of ages with D_t / P_t correlated with weights-at-age, the series of weights for a given age within the block were standardized to their age-specific time-series mean. Then a linear regression of D_t / P_t from the base Steele-Henderson model and standardized weight-at-age for all ages within the time block was used to predict the point where D_t / P_t results in average weight (standardized weight-age-age = 1.0). This point was considered a threshold consumption level for predator condition. Data were further examined to determine the risk that below average weight would occur when D_t / P_t was at or below the threshold and to see if a potential D_t / P_t target was suggested where the chance of a predator being below average weight was substantially less. For striped bass, this analysis resulted in a potential target for D_t / P_t of 2.2, which was consistently met or exceeded once Z_2 / Z_{MUP} fell below 0.87.

11.3 Results

11.3.1 Diagnostics

Based on AIC_c, a fishing biomass dynamic model and a Steele-Henderson model featuring striped bass were equally likely the best models given the data; both models had an AIC_c of -156. Neither the base fishing-only model nor the base striped bass Steele-Henderson model (base Steele-Henderson model) fit the individual indices well. The r^2 for the striped bass Steele-Henderson model was 0.18 for the fit of the observed and estimated RCPUE indices, 0.12 for NAD, and 0.33 for MAD. Residuals appeared normally distributed with a mean near zero and serial patterning was not evident. Trends in estimated indices were similar between the fishing only and striped bass Steele-Henderson models, but the fishing only model trend was smoother (Figure 28). The Steele-Henderson model was able to account for some year-to-year variability (Figure 28), but the model does not include any process error (i.e. recruitment deviations) and so it was not expected to fit the fine scale inter-annual variability in the observed indices.

Bioenergetics-based annual C_{MAX} estimates derived by Uphoff and Sharov (2018) for striped bass ranged from 12.7 to 15.6-times striped bass body weight and the median equaled 14.6; the Steele-Henderson model estimated the striped bass-specific parameter d to be 11.0. Buckel et al. (1999) estimated annual C_{MAX} for bluefish (17.8), while the Steele-Henderson model estimated d at 5.1. Similar information for spiny dogfish was not available.

11.3.2 Population Estimates

Estimates of r and K were quite different between the fishing-only and the base Steele-Henderson model which included striped bass a predator (Table 6). The estimate of r was higher for the Steele-Henderson model (2.27) than the fishing only model (0.32) and K was about 3-times lower for the Steele-Henderson model (Table 6). Adding striped bass predation to harvest resulted in a general shift in depiction from a stock with low productivity and high biomass to one with high productivity and low biomass. The estimate of MSY from the fishing only model was 273,184 mt, while the estimate of MUP from the Steele-Henderson model was 608,517 mt. It is not uncommon with biomass dynamic models that data can be well explained as coming from a small, productive stock or a large, unproductive one since estimates of r and K are often highly negatively correlated (Walters and Martell 2004).

The base Steele-Henderson model indicated that biomass was initially high (B / B_{MUP} ~ 1.5), then declined steadily into the late 1990s (B / B_{MUP} ~ 0.7), increased sharply to near 1.0 by 2000, and finally increased to about 1.25 in 2014 and remained there through 2017 (Figure 29). Biomass was below its threshold during 1990-2001.

Base Steele-Henderson model estimates of landings as a proportion of annual surplus production available to the fishery indicated that the ratio exceeded 1.0 seven times between 1990 and 2010; it has been between 0.69 and 0.86 since 2013 (Figure 30). Relative F was above the F_{MUP} threshold intermittently during 1990-2010 with the base Steele-Henderson model; the model identified 1995-1998 as the period of highest F (Figure 31). Relative M_2 (M_2 / Z_{MUP}) rose from less than 0.20 in 1985 to a plateau of 0.60-0.70 that was maintained from the mid-1990s to 2010. It then dropped to approximately 0.50 by 2013 and remained there through 2017 (Figure 32). Relative Z_2 (Z_2 / Z_{MUP}) was below the threshold during 1985-1989. Relative Z_2 estimated by the base Steele-Henderson model consistently exceeded the threshold from 1990 to 1997 and intermittently through 2010, but declined below the threshold and remained at about 0.80 after 2012 (Figure 33).

Estimates of F / Z_2 indicated that F was the major influence on the stock until the late 1990s (base Steele-Henderson model; Figure 34). Estimates of F / Z_2 from the base Steele-Henderson model were generally below the threshold after 1998, but were at or near it during 2001-2002 and 2011-2012.

Striped bass consumption

The range of D_t / P_t was 1.1 – 3.8 for the base Steele-Henderson model. Estimates from the base Steele-Henderson model started high during 1985-1987 (3.5-3.8), fell to a nadir (1.1) during 1997-1998, quickly rose to near 2.0 by 2001, dipped to 1.6 in 2002, rose again, remained near 2.0 during 2003-2011, climbed to about 2.8 in 2014, and remained there through 2017. D_t / P_t estimates were lower than the annual consumption estimates derived from bioenergetics models, which ranged from 4.2 – 6.3 (Uphoff and Sharov 2018).

Forage Status of Ages 1+ Atlantic Menhaden for Striped Bass in 2017

Based on the base Steele-Henderson model, ages 1+ Atlantic menhaden were at low risk of not maintaining their forage role for striped bass in 2017. Atlantic menhaden harvest was low relative to historic levels and estimated striped bass biomass in 2017 was at its lowest since recovery. This combination led to relatively low predatory and fishery demand. None of the proposed indicators (B / B_{MUP}, Z_2 / Z_{MUP}, D_t / P_t, F / Z_2, F / F_{MUP}, and H_t / SF) exceeded threshold conditions in 2017 and the risk that they did (based on jackknifed distributions for 2017) was estimated as 0% (Table 7). None of the 90% intervals overlapped a threshold. If the suggested target conditions ($D_t / P_t \geq 2.2$ and $Z_2 / Z_{\mathrm{MUP}} < 0.87$) in the previous section are considered, then the risk of not meeting these targets was also zero.

11.3.3 Uncertainty

Bounds of the 90% intervals of r, K, B_{1985}, A, and MUP were within 5% of the estimated values, while d was less precise (9%). High precision of model parameters lead to precise estimates of ages 1+ Atlantic menhaden biomass (Figure 35), D (total biomass consumed by striped bass; Figure 36), M_2 (Figure 37), F (Figure 38), Z_2 (Figure 39), and D_t / P_t (Figure 40).

11.3.4 Simulation Testing

The Steele-Henderson model using Evolver underwent limited testing with the simulated data set used to test the time-varying r surplus production model (25 runs for each scenario). The results indicated that the Steele-Henderson model could fit the simulated index fairly well and had a low relative error for prey biomass and exploitation rate, but overestimated M_2 (Figure 41). The magnitude of M_2 overestimation varied over the time series and depended on the scenario (e.g., increasing, decreasing, or constant predator time series). Because the Steele-Henderson model estimates a time-constant r value, annual variability in productivity caused by variability in recruitment (as in the simulated data) or other factors may be subsumed into the estimates of M_2. The simulation testing suggested that the magnitude of the estimates of M_2 from this model could be positively biased. Estimates of consumption-related parameters can be evaluated for plausibility with estimates from other studies of predator consumption (as was done with the base runs) or M of forage species. Better starting values for consumption might have resulted in better simulation model performance and further work on this with the simulated data is needed.

11.4 Sensitivity Analyses

Runs were made with initial values 20% higher or lower than the midpoints used as the common starting value. The PRFC index was substituted for the RCPUE index (MAD and NAD indices were standardized to PRFC index units using the same approach used for RCPUE). Retrospective bias of the base run was investigated by sequentially removing up to the last four years (2014-2017) from analysis. Additional runs were made that removed one of the indices from analysis to investigate an individual index's influence. Index pairs considered were RCPUE and NAD, RCPUE and MAD, and NAD and MAD.

Since the genetic algorithm did not provide a defined endpoint for convergence, the base Steele-Henderson model was run for one hour to look at run time sensitivity. Three runs were made with different limits on Steele-Henderson model parameter *d*: *d* was confined to a range estimated from bioenergetics; *d* was allowed a higher maximum (starting range for *d* was 0-20.0); and the default penalty function in Evolver was imposed for *d* if estimates exceeded the maximum (17.0). These three runs were prompted by concerns about parameter *d* being at its maximum constraint for one of the predators in Uphoff and Sharov (2018).

Sensitivity runs resulted in a "population" of base Steele-Henderson models with well-correlated parameters that produced the same general depiction of Atlantic menhaden (high *r* and low *K*) as the base Steele-Henderson model (Table 8). Estimates of *r* ranged from 1.66 to 2.56; *K*, $9.7^5 – 1.4 \cdot 10^6$; mt B_{1985}, $7.1 \cdot 10^5 – 1.1 \cdot 10^6$ mt; *d*, 7.8 – 17.0 (two were at maximum constraint) and *A*, $7.8 \cdot 10^5 – 2.0 \cdot 10^6$ mt (Table 8). Correlations among Steele-Henderson model parameters of the base Steele-Henderson model and six sensitivity runs were high for *r*, *K*, B_{1985}, and *A*, and for *d* and *A* ($\rho \geq 0.90$ or ≤ -0.91; $p \leq 0.0064$; Table 9). Parameter *d* was modestly correlated with *r* ($\rho = 0.67$, $p = 0.10$) and *K* ($\rho = 0.63$, $p = 0.15$); High correlation of *r* and *K* led to estimates of MUP among the seven runs with a maximum difference of 4% from the base run (Table 8).

Steel-Henderson model estimates of B / B_{MUP} (Figure 42), Z_2 / Z_{MUP} (Figure 43) and D_t / P_t during 1985-2017 (Figure 44) were very similar across sensitivity runs. Substantially different conclusions about status were unlikely among the Steele-Henderson model runs. Differences among annual estimates of B / B_{MUP}, Z_2 / Z_{MUP}, and D_t / P_t from the sensitivity runs were small (Table 10).

A striped bass Steele-Henderson model using the PRFC index (PRFC Steele-Henderson model) fit the data similarly well to the base model (Figure 45). The r^2 for the PRFC Steele-Henderson model was higher for the fishery-dependent index (PRFC: 0.35), but lower for the fishery-independent indices (NAD: 0.01; MAD: 0.25) Residuals appeared normally distributed and serial patterning was not evident.

Different conclusions about stock status were not likely if the PRFC index was substituted for the RCPUE index during 1985-2017. Trends in B / B_{MUP}, Z_2 / Z_{MUP}, and D_t / P_t were very similar between the base run and the PRFC Steele-Henderson model. Estimates of B / B_{MUP} from the PRFC Steele-Henderson model were generally higher than for the base run (Figure 46), while estimates of Z / Z_{MUP} from the PRFC Steele-Henderson model were generally lower than for the base run (Figure 47). Estimates of D_t / P_t were similar (Figure 48).

Removal of a single index from the time-series increased variability (SSQ / N) by about a third over the base run for the two pairings that included RCPUE and the variability of the run featuring only fishery-independent indices was nearly double that of the base run (Table 11). All pairing combinations resulted in parameters that would generalize Atlantic menhaden as a highly productive stock. If the MAD index was included in the time-series, *r* ranged between 2.01 and 2.27; *r* equaled 1.26 for the run without the MAD index (RCPUE and NAD; Table 11).

Estimates of K were similar for runs that included RCPUE (~1.1 to 1.4 · 10^6 mt) and higher (~1.8 · 10^6 mt) for the run with only fishery-independent indices (Table 11).

Removal of the RCPUE index (i.e., fishery-independent indices remained) resulted in D_t / P_t estimates that exceeded the maximum estimated for striped bass from bioenergetics models during 1985-1989 (Uphoff and Sharov 2018) and fell between the minimum and maximum during 1990-1993 and 2013-2017 (Figure 49). These estimates were considered unlikely and assessment based on the two fishery-independent indices alone would be biased. Estimates of D_t / P_t from the remaining three runs with RCPUE were considered plausible. The RCPUE and NAD run indicated that fewer ages 1+ Atlantic menhaden were consumed by Ages 3+ striped bass (low D_t / P_t), while the base run and run with RCPUE and MAD were similar and D_t / P_t was about 2-3 times that of the RCPUE and NAD run (Figure 49).

The RCPUE and MAD Steele-Henderson model estimates of B / B_{MUP} were very close to the base run (Figure 50). All four runs indicated that B / B_{MUP} was above the threshold during 1985-1990. The base run, RCPUE and MAD run, and the RCPUE and NAD run fell below the B / B_{MUP} threshold during the 1990s. The base and two-index runs that included RCPUE remained near the threshold through 2011 and then climbed above 1.20 and remained there. The RCPUE and NAD run diverged from the base run in the early 1990s and for the remainder of the time-series provided a more optimistic view of B / B_{MUP} that was an additional 0.10 - 0.20 greater than the base run. Estimates of B / B_{MUP} from the MAD and NAD run reflected the unrealistically high estimates of consumption and were 0.15 to 0.25 lower than the base run after striped bass recovered in 1995 (Figure 50). Trends in Z_2 / Z_{MUP} were the converse of those described for B / B_{MUP} (Figure 51).

11.5 Retrospective Analyses

Removal of up to four years from the end of the time-series in retrospective runs had minimal impact. Variability (SSQ / N) remained close to that of the base run (Table 11). Most parameters estimated were well correlated ($\rho > 0.90$ or < -0.90 for r, K, and B_{1985}, K and A, and A and d) among the retrospective runs. All combinations resulted in parameters that would generalize Atlantic menhaden as a small, productive stock (Table 11). Retrospective bias was not apparent in B / B_{MUP} (Figure 52), Z_2 / Z_{MUP} (Figure 53) and D_t / P_t (Figure 54) when up to four years were removed from the end of the 1985-2017 time series.

11.6 Projections

Stochastic projections using the base Steele-Henderson model were made for 2018-2041. They explored four scenarios: (1) continuation of 2017 harvest with major predators at 2017 levels (status quo projection), (2) major predator biomass increases to recovered status (Table 3) and Atlantic menhaden are fished at one half their target F (major predator recovery, half-target F projection); (3) major predator biomass increases to recovered status and Atlantic menhaden are fished at their target F (major predator recovery, target F projection); and (4) predator biomass increases to a point where a proposed consumption threshold is met and Atlantic menhaden are fished at their current harvest (predator consumption threshold, current harvest projection). For Steele-Henderson model projections featuring predator recovery, a ten-year period was arbitrarily chosen for recovery and then predator biomass was held steady for another ten years. Terminal estimates represented "equilibrium" conditions for each projection.

Two distributions provided the best depiction of jackknifed base Steele-Henderson model parameters. A Laplace distribution (also known as a double exponential distribution) fit *K, d, A*, and Atlantic menhaden ages 1+ biomass on January 1, 2018, best. The distribution of *r* was best described by a log logistic distribution. Jackknifed estimates of the four Steele-Henderson parameters needed for projections (*r*, *K*, *d*, and *A*) were weakly to moderately correlated (Table 12). Graphs of distributions are presented in Figure 55 – Figure 57 . Table 13 provides a summary of location, scale, and shape values assigned to distribution functions for each simulated parameter.

Biomass of ages 3+ striped bass in 2018 was set at the estimate for 2017 (134,796 mt). Striped bass biomasses for 2018 and subsequent years were assumed to be normally distributed and were assigned a CV of 6% based on variation of biomass estimates in the recent assessment. Striped bass recovery is based on an SSB target. Target ages 1+ striped bass biomass at target SSB for projections was estimated for ages 1+. The median proportion of ages 1+ striped bass biomass that was comprised of ages 3+ (0.84) during the period the stock has been considered recovered (1995-2017) was multiplied by the target estimate for ages 1+ to approximate target biomass of ages 3+ striped bass (260,685 MT) capable of eating ages 1+ Atlantic menhaden at target SSB.

The status quo projection indicated very low risk that ages 1+ Atlantic menhaden's forage role would not be maintained (Table 14). At "equilibrium", the 90% CI's of B / B_{MUP}, Z_2 / Z_{MUP}, and D_t / P_t did not overlap their proposed thresholds and estimated risk of breaching these thresholds was 0%. Projected D_t / P_t averaged 2.89 (45% higher than the threshold), a value associated with higher than average weights (i.e., better condition) of ages 6+ striped bass (Table 14). Maintaining the forage role of ages 1+ Atlantic menhaden for striped bass was likely.

The projection where striped bass biomass increased to recovered status (ages 3+ biomass nearly doubles) and Atlantic menhaden are fished at status quo F (F_{2017}) represented a high-risk strategy (Table 14). Substantial portions of 90% intervals of all three metrics overlapped their thresholds. Risk of breaching the B / B_{MUP} threshold was 80%; risk of breaching the Z_2 / Z_{MUP} threshold, 55%; and the risk of breaching the D_t / P_t threshold, 85%. Average yield would be 26% less than in 2017 and average D_t / P_t was 10% less than the threshold, indicating consumption was not sufficient to maintain striped bass individual weight at or above the time series average (Table 14). Maintaining the forage role of ages 1+ Atlantic menhaden for striped bass was unlikely with this strategy.

The projection where striped bass biomass increased to recovered status and Atlantic menhaden were fished at their target F had the highest risk (Table 14). Ninety percent intervals of all three metrics came close to completely overlapping their thresholds. Risk of breaching the B / B_{MUP} threshold was 100%; breaching the Z_2 / Z_{MUP} threshold, 95%; and the risk of breaching the D_t / P_t threshold was 100%. Yield was 26% greater than in 2017 and average D_t / P_t was 30% less than the threshold (Table 14). Maintaining the forage role of ages 1+ Atlantic menhaden for striped bass was unlikely.

The projection with predator biomass increasing to a point where their consumption threshold is met and Atlantic menhaden harvested at their current level represented a high risk option, but not as risky as the previous two (Table 14). Ninety percent intervals of all three metrics overlapped their thresholds near the interval midpoint. Risk of breaching the B / B_{MUP} threshold was 45%; risk of breaching the Z_2 / Z_{MUP} threshold, 60%; and the risk of breaching the D_t / P_t threshold was 50%. Striped bass biomass was 83% of the target to maintain D_t / P_t at its threshold (Table 14). Risk that the forage role of ages 1+ Atlantic menhaden for striped bass would not be met was high. Atlantic menhaden harvest in this projection would be considered low by historical standards and striped bass biomass had to be below its current target in order to meet threshold (not target) D_t / P_t.

12 MULTISPECIES STATISTICAL CATCH-AT-AGE MODEL (VADER) (SUPPORTING)

Some of the earliest multispecies modeling work connected virtual population analysis (VPA) models together with predation functions (Helgason and Gislason 1979; Gislason and Helgason 1985; Sparre 1991; Livingston and Jurado-Molina 2000). This modeling approach can be helpful in a complex fisheries modeling environment because the strong assumptions on certain parameters aid in the estimation of the remaining parameters. From this more deterministic modeling technique, statistical approaches were then developed using either age-based, or length-based statistical models. These statistical approaches are more comparable to some of the single-species assessment methods that are now used and have the added benefit of allowing the estimation of uncertainty around the estimated population parameters (Lewy and Vinther 2004; Van Kirk et al. 2010; Curti et al. 2013). The goal of these multispecies approaches is to create more realistic information for fisheries management (Gislason 1999; Moustahfid et al. 2009a). The multispecies statistical catch-at-age model (hereafter referred to as Virtual Assessment for the Description of Ecosystem Responses, or VADER) developed for this assessment adopted the more progressive statistical approach for its modeling methodology and was built on the foundational work of Curti et al. (2013) and McNamee (2018).

12.1 Treatment of Indices & Input Data

The VADER model was constructed around the six ERP focal species: Atlantic menhaden, striped bass, bluefish, weakfish, Atlantic herring, and spiny dogfish. The species were selected based on a review of important predator diet information, the availability of age-structured data for the species, and knowledge of the migratory patterns of the species (see also Section 3.1.2). The confounding factor of temporal and spatial overlap was mitigated to some degree by the fact that the ERP focal species all have similar seasonal migratory patterns (Section 4). However, Atlantic herring and spiny dogfish do not overlap as significantly as the other species – an important consideration when interpreting the output from this assessment model, which does not explicitly account for spatiotemporal overlap in predators and their prey.

In the model, striped bass, bluefish, and spiny dogfish were top predators of both Atlantic menhaden and Atlantic herring. Both Atlantic menhaden and Atlantic herring were strictly forage species. Weakfish served as both a predator of Atlantic menhaden and Atlantic herring, as well as a prey species for striped bass, bluefish, and spiny dogfish. Cannibalism by any species was not accounted for in this study. All symbols and likelihood components for the multispecies model are indicated in Table 15 and Table 16 respectively.

As in Curti et al. (2013), there were six types of input data needed for each species in the model: total fishery removals in weight, fishery-independent indices of abundance, age proportions for both fishery removals and fishery-independent survey catches, average weight-at-age by year, and age-specific predator diet information. With the exception of spiny dogfish, single-species statistical catch-at-age models are used for management of all of the ERP focal species. Spiny dogfish are assessed using a swept-area biomass approach. Unless otherwise noted, all data inputs used were taken directly from recent stock assessment documents and from direct communication with the stock assessment scientists that work on these species (Section 8).

For all species, total fishery removals represent landings plus dead discards from both the commercial and recreational fisheries in weight (thousands of metric tons). Assumptions about discard mortality for this study were consistent with assumptions from the reviewed assessments for each species. This model used a single fleet for each species for removals.

Annual catch-at-age in millions of fish for the entire time series were used to calculate age proportions from the catch. The information used to construct age-specific catch from the recreational fishery is generally believed to be more reliable in numbers than it is in weight. Again, for all species used in this study, this time series of information was obtained from the most recent stock assessment. In contrast to the single-species assessments for these species, which usually model recreational and commercial catches as separate fleets with separately estimated selectivities, all removals were modeled as a single fleet for each species with one selectivity pattern for each fleet. This is not a poor assumption for the ERP focal species: for each species, there is one predominant fishery and gear type that prosecutes the fishery (i.e., striped bass is predominately a rod and reel fishery when considering both the commercial and recreational fishery, while Atlantic menhaden is predominately a commercial purse seine fishery); therefore, the age structure of the removals for each species is most likely governed by one predominant selectivity.

In contrast to the work done by Curti et al. (2013), many of the ERP focal species have multiple surveys – with differing time series and gear types – that are used to estimate stock abundance over time. To accommodate multiple surveys while keeping the model structure as simple and computationally efficient as possible, a subset of the available surveys for each species was used. For each species, one YOY survey was selected along with two adult indices, when available (Table 17). This subset was identified using ASMFC TC guidance on the most appropriate indices of abundance for their respective species. Data for these indices – including the number per tow in each year, uncertainty around the index values, and age composition data – was taken from the stock assessment documents and most recent assessment information. Given that Atlantic menhaden are the focus of this work, the YOY index and all three of the adult indices from the single-species benchmark assessment were used. Only two indices were used for Atlantic herring and one index for spiny dogfish, with no YOY indices for either species. See Section 7 for more details on the surveys used.

Average individual weight-at-age was needed to convert from numbers to biomass units. The weight-at-age information was introduced in the model as a matrix, so the information varies not only by age, but by time as well. This is an important consideration as several of the ERP focal species have significant shifts in weight-at-age through time.

For the trophic interactions of the multispecies runs, data were needed on species food habits, consumption estimates, and information on biomass throughout the ecosystem. These data included consumption-to-biomass estimates for each species (consumption: biomass or *C*/*B*), an estimate of the biomass of "other food" in the ecosystem, and average predator diet information.

Age-specific *C*/*B* ratios were obtained by the methodology from Garrison et al. (2010) as developed for the MSVPA model previously developed for Atlantic menhaden, which included this suite of species. Food consumption rates in fish can vary strongly, particularly between seasons as a function of changing temperatures and metabolic demands. To account for these processes, a modified consumption model was implemented using the Elliot & Persson (1978) evacuation rate approach. Total yearly (*y*) consumption for a predator species (*i*), age (*a*) during a year is:

$$C_y^{i,a} = 24 * E_s^{i.a} * \overline{SC}_y^{i,a} * D_y * w_y^{i,a} * \overline{N}_y^{i,a} \tag{12.1}$$

Where SC_y is the mean stomach-content weight relative to predator body weight in a year (*y*), D_y is the number of days in the year, $w_y^{i,a}$ is the average yearly weight at age for the predator species, and $\overline{N}_y^{i,a}$ is the abundance of the predator age class during the year. The predator-specific evacuation rate $E_{i,a}$ (hr^{-1}) is given as:

$$E_{i,a} = a_{i,a} * \exp\left(b_{i,a} * T_y\right) \tag{12.1.1}$$

Where T_y is the average yearly temperature (°C) and $a_{i,a}$ and $b_{i,a}$ are fitted parameters based upon laboratory feeding experiments, field studies, or other sources (Elliot & Persson 1978). The evacuation rate reflects the temperature-dependent metabolic rates of the predator.

These data were updated through 2017 as these species-specific data were available. As noted above, the *C*/*B* ratios were developed for the MSVPA with more resolution (i.e. daily *C*/*B* ratios by season), but were aggregated across the whole year for this model to create a matrix of age-specific *C*/*B* ratios by species that varied through time based on the annual temperature. The method for calculating the Mean Annual Sea Surface Temperature (SST) for the US East Coast

Shelf for 1985-2017 was calculated using daily mean SST on a 0.25° spatial grid from the NOAA Optimum (OI) SST V2 High Resolution Dataset. OISST is also known as Reynolds' SST. OISST provides global fields that are based on a combination of ocean temperature observations from satellite, Advanced Very High Resolution Radiometer (AVHRR) infrared satellite SST data, and in situ platforms (i.e., ships and buoys). The input data are irregularly distributed in space and must first be placed on a regular grid. Then, statistical methods (optimum interpolation, OI) are applied to fill in where there are missing values. The methodology includes a bias adjustment step of the satellite data to in situ data prior to interpolation (Reynolds 2007).

The High Resolution SST data were provided by the NOAA/Ocean and Atmospheric Research/ Earth System Research Laboratory/Physical Sciences Division from their website at https://www.esrl.noaa.gov/psd/. Specifically, netcdf files with global mean daily SST for 1985-2017 were downloaded from:
https://www.esrl.noaa.gov/psd/cgi-bin/db_search/DBListFiles.pl?did=132&tid=68373&vid=2423

A spatial polygon covering the US East Coast Shelf was used to clip regional data from the global data sets. The daily mean values for the region were used to calculate an annual mean for each year. All analyses were performed using the R software environment for statistical computing and graphics (R Core Team 2018 – specific packages used are included in the references). Algorithms for clipping OISST data and calculating mean annual SST were adapted from algorithms used for the NOAA\Northeast Fisheries Science Center's Ecosystem Status Report (Ecosystem Assessment Program 2012 and Sean Hardison, pers. comm).

As assumed in Curti et al. (2013) and based on previous work (Sparre 1980; Tsou and Collie 2001), a constant, time-invariant total ecosystem biomass was assumed. As a result, the total ecosystem biomass was constant over time, but the biomass of the individual modeled species could vary annually. Prior studies have confirmed that the total biomass in large marine ecosystems can remain relatively stationary through time (Link et al. 2008; Auster and Link 2009; Byron and Link 2010). There were no direct measurements found indicating what this overall biomass estimate should be, so a total biomass estimate from the MSVPA was used as a starting point. To supplement and support the MSVPA derived total ecosystem biomass value, information derived from an Atlantic coast Ecopath model was also investigated (Buchheister et al. 2017). Both values were close in magnitude. Testing with the multispecies assessment model indicated that performance was best for the value derived from the Ecopath model (94,800,000 mt) and therefore this value was the one selected for the base case run of the model.

Stomach-content data were obtained from three main sources: the NEFSC Food Web Dynamics Program, NEAMAP, and ChesMMAP also collect stomach-content data under similar protocols to the NEFSC program (Section 3.1.1). These length-based data for predator and prey from stomach-content information were converted to weight through the use of length-weight relationships as collected in Wigley et al. (2003). Age-specific predator diet habits, input to the model as proportion by weight for each age class, were averaged over 3-year periods to reduce the inherent variability in the dataset, as well as to reduce the amount of missing data and increase the sample size being used for any year (Van Kirk et al. 2010), while still capturing the temporal trends.

Even with binning, there are still gaps and sample size issues in the data for this portion of the model. A Bayesian technique was used to account for this. A multinomial probit model was developed for the diet data, using the implementation in the MNP package in R (Imai and van Dyk 2005). Under the multinomial probit model, a multivariate normal distribution on the latent variables is assumed, $W_i = (W_{i1}, \dots, W_{ip-1})$.

$$W_i = X_i\beta + \epsilon_i, \epsilon_i \sim N(0, \Sigma), for\ i = 1, \dots, n \quad (12.2)$$

where X_i is a $(p-1)$ x k matrix of covariates, β is k x 1 vector of fixed coefficients, ϵ_i is $(p-1)$ x 1 vector of disturbances, and Σ is a $(p-1)$ x $(p-1)$ positive definite matrix. For the model to be identified, the first diagonal element of Σ was constrained, $\sigma_{11} = 1$. The response variable, Y_i, is the index of the prey choice of predator i among the alternatives in the choice set (here it was the prey items Atlantic menhaden, weakfish, Atlantic herring, and "other food") and was modeled in terms of this latent variable, W_i , via

$$Y_i(W_i) = \begin{cases} 0, & if\ max(W_i) < 0 \\ j, & if\ max(W_i) = W_{ij} > 0 \end{cases}, for\ i = 1, \dots, n, and\ j = 1, \dots, p-1 \quad (12.3)$$

where Y_i equal to 0 corresponded to a base category.

The matrix X_i may include both choice-specific and predator-specific variables. A choice specific variable is a variable that has a value for each of the p choices (in our case this is 4 choices for each prey species in the model plus an "other" category), and these p values may be different for each predator. Choice-specific variables are recorded relative to the baseline choice (in this case weakfish was used as the base case) and thus there are $(p-1)$recorded values for each predator. In this way a choice-specific variable is tabulated as a column in X_i. Predator-specific variables, on the other hand, take on a value for each individual predator, but are constant across the choices, e.g., the age of the individual predator. These variables are tabulated via their interaction with each of the choice indicator variables. Thus, a predator-specific variable corresponds to $(p-1)$ columns of X_i and $(p-1)$components of β .

The prior distribution follows the methods of Imai and van Dyk (2005). The prior distribution for the multinomial probit model is

$$\beta \sim N\left(0, A^{-1}\right) \text{ and } p(\Sigma) \propto |\Sigma|^{-\frac{(\nu+p)}{2}} \left[trace\left(S\Sigma^{-1}\right)\right]^{-\frac{(\nu-p)}{2}} \quad (12.4)$$

where A is the prior precision matrix of β, ν is the prior degrees of freedom parameter for Σ, and the $(p-1)$ x $(p-1)$ positive definite matrix A is the prior scale for Σ; the first diagonal element of S is assumed to be one. The prior distribution on Σ is proper if $\geq$ $(p-1)$, the prior mean of Σ is approximately equal to S if $\nu \geq (p-2)$, and the prior variance of Σ increases as ν decreases as long as this variance exists. An improper prior on β was allowed, which was $p(\beta) \propto 1$ (i.e. $A = 0$).

This model was run on the existing dataset, and then used to predict for each combination of predator, year-bin, and predator age group. In some cases, no data existed for certain combinations (a certain predator, in a certain year-bin, at a certain age class), and therefore a prediction was not possible. In these cases, a simpler model without year-bin was also run, and when a combination was missing from the dataset, the global preference of prey for the predator was used instead of the year-bin specific preference, meaning the preference of the predator at that age class across the entire dataset was used to fill this data gap.

Spiny Dogfish

Spiny dogfish input data were a special case. Spiny dogfish inputs were constructed from several sources, due to the fact that spiny dogfish do not have an age-structured assessment. Separate male and female indices from the spring NEFSC Survey were obtained from NEFSC (personal communication, Katherine Sosebee). These were standardized in number per tow and adjusted for any gear or vessel changes. Separate sexed age-length keys were constructed using von Bertalanffy relationships found in Nammack et al. 1985 and following methodology from Curti 2012. Sexed landings were retrieved from Sosebee and Rago 2018 (2018 spiny dogfish status update) and were from the U.S., Canadian, and Foreign fisheries. Since the single species assessment used these values and there was a period of time in the early 2000s when Canadian fisheries had high landings, all of these sources were included (see Table 5 in Sosebee and Rago 2018). Landings were added to sexed dead discards from NMFS port sampling but only back to 1991. Dead discard numbers from Table 3 in the 2018 status update were used for 1985-1990 (this table indicates that this time period was hindcast from SARC 43). These numbers were broken down by sex using the formulas in Table 6 from the 2018 status update using averages of 1991-1993.

Spiny dogfish catch-at-age was calculated using survey index as a length frequency (which assumed that survey length frequencies were the same as the commercial/recreational catch for the landings in the U.S., Canada, and foreign fisheries), age-length keys from Nammack et al. (1985) (time-invariant, the same age-length key was used for each year), and catch in numbers. A plus group was decided after review of Nammack et al. (1985), where growth curve plateaus for males at 20+. Sexed spiny dogfish weight-at-age was also calculated separately using this information. Male and female catch-at-age and weight-at-age were combined to generate a weighted total catch-at-age and weight-at-age using a sex ratio.

The maturity-at-age matrix was assumed to be knife-edge at 12 years old, consistent with Curti (2012) and Nammack et al. 1985. A static and time-invariant natural mortality of 0.092 was used based on a life span of 50 years from TRAC (2010) and Curti (2012). Initial year one biomass estimates in 1985 were from the TRAC (2010) and adjusted for catchability based on Sagarese et al. (2016). These were converted to numbers-at-age using the catch-at-age to partition the biomass into age bins and the weight-at-age to convert to numbers.

12.2 Parameterization

The VADER model followed a traditional statistical catch-at-age structure as used for many single-species stock assessments. These traditional catch-at-age equations were then linked and interacted through a set of trophic interactions. All model equations will not be presented in this document as they followed the equations as developed in Quinn and Deriso (1999), but some of the main equations used will be described for the catch-at-age portions of the model, and the trophic calculations will be presented in detail.

Progression of year class abundance was implemented by the equation:

$$N_{i,a+1,t+1} = N_{i,a,t}e^{-Z_{i,a,t}} \tag{12.5}$$

where N = species abundance in millions of fish, Z = total mortality, i = species, a = age class, and t = year. As there were plus groups for each species used in this project, the final age class modeled (i.e. when $a = a_{\max}$) needed to be adjusted using the equation:

$$N_{i,a,t+1} = N_{i,a-1,t}e^{-Z_{i,a-1,t}} + N_{i,a,t}e^{-Z_{i,a,t}} \tag{12.6}$$

Fishery catch-at-age was calculated using Baranov's catch equation:

$$C_{i,a,t} = \frac{F_{i,a,t}}{Z_{i,a,t}}N_{i,a,t}(1 - e^{-Z_{i,a,t}}) \tag{12.7}$$

where C = fishery catch (recreational, commercial, and dead discards for each) and F = fishing mortality. Fishing mortality-at-age (assuming separable fishing mortality) followed the equation:

$$F_{i,a,t} = s_{i,a}F_{i,t} \tag{12.8}$$

where s = fishery selectivity. Fishery-independent survey catch ($FIC_{i,t}$) was related to species-specific abundances through the following equation:

$$FIC_{i,t} = q_i r_{i,a} N_{i,t} e^{-\frac{m}{12}Z_{i,t}} \tag{12.9}$$

This mathematical configuration assumes an age and time-invariant catchability (q_i), age-specific survey selectivity coefficients ($r_{i,a}$), and also accounts for the time of year during which the survey was conducted (m) so total mortality can be applied to the index appropriately. Species-specific catchabilities (q_i) were calculated from the entire time series deviations between the model predicted absolute abundance and model predicted relative abundance (Walters and Ludwig 1994).

Finally, age-specific fishery and survey selectivity coefficients were estimated for each species for all age classes through the choice of either a logistic or double logistic selectivity function, depending on the choices made by the single-species stock assessment teams. YOY surveys assumed age specific selectivity, with selection being 1 for the first age class and 0 for all other age classes. This formulation departed from previous work (Curti et al. 2013) and was reconfigured to better simulate the selectivities for the modeled species by allowing doming in the selectivity-at-age, which provided more consistency with the selectivity shapes used in the single-species assessments. The four-parameter double logistic equation used for both the fishery selectivity and the fishery-independent survey selectivity was:

$$Sel_{x,i,a} = \left(\frac{1}{1+e^{-(a-\alpha_1)/\beta_1}}\right)\left(1-\frac{1}{1+e^{-(a-\alpha_2)/\beta_2}}\right) \qquad (12.10)$$

And the two-parameter logistic equation used was:

$$Sel_{x,i,a} = \left(\frac{1}{1+e^{-(a-\alpha_1)/\beta_1}}\right) \qquad (12.11)$$

where $Sel_{x,i,a}$ is the species-specific selectivity at age, *x* = fleet or survey, *i* = species, *a* = age class, and α_{1or2} and $\beta_{1\text{or}2}$ are the ascending or descending inflection point and slope parameters, respectively.

Predation mortality (M_2) is a sub component of total mortality (Z), but more specifically a sub-component of the natural mortality component in Z. The simplest equation to describe this is:

$$Z = F + M_0 + M_2 \qquad (12.12)$$

where Z is total mortality, F is fishing mortality, M_0 is residual natural mortality (natural mortality attributed to all other factors except predation by species included in the model), and M_2 is predation mortality from the species included in the model (Helgason and Gislason 1979). Species that were modeled as predators only (e.g. striped bass, bluefish, and spiny dogfish) only had M_0 operating on their populations, while species that were modeled as prey (e.g. Atlantic menhaden, Atlantic herring, and weakfish) had both M_0 and M_2 operating on their populations.

The M_0 value was an important source of uncertainty in the model. Initial values for M_0 were taken from the MSVPA information on the ERP focal species, where available, to determine the portion of natural mortality that was occurring from predation. The assumed total natural mortality from the single-species benchmark assessments for the prey species in this model were prorated downward based on this proportion. Additional analyses looked at the objective function values under different M_0 selections, as well as the difference between the VADER biomass outputs and the single-species biomass outputs. These methods were used to identify the best choice for this parameter in VADER, which was determined to be a 20% decrease from the single species total natural mortality assumptions for the prey species.

There is a recursive property in this formulation of M_2 in that the biomass data element needed for calculating M_2 has total mortality as an element of its calculation, therefore an approximation was used. To approximate the instantaneous rate of M_2, the biomass of the predator and the prey items were assumed to come from the beginning of each year, prior to being subject to these various forms of mortality (Van Kirk et al. 2010). The equation for the instantaneous M_2 is:

$$M_{2i,a,t} = \frac{1}{N_{i,a,t}W_{i,a,t}} \sum_j \sum_b CB_{j,b} B_{j,b,t} \frac{\phi_{i,a,j,b,t}}{\phi_{j,b,t}} \tag{12.13}$$

where $N_{i,a,t}$ = mean number of prey *i* at age *a* and at time *t*, $W_{i,a,t}$ = the weight of prey *i* at age *a* at time *t*, $CB_{j,b}$ = the age-specific (*b*) consumption-to-biomass ratio for predator species *j*, $B_{j,b,t}$ = age-specific biomass of predator *j*, and $\frac{\phi_{i,a,j,b,t}}{\phi_{j,b,t}}$ = the proportion of prey *i* at age *a* in all food available to predator *j* at age *b* in year *t*, which was assumed equal to the proportion of food within the stomach of predator *j* at age *b* in year *t* composed of prey *i* at age *a* (Lewy and Vinther 2004). Under this formulation, a type-II functional response was assumed, where the predator satiates at a high prey biomass, and the satiation reaches an asymptote (i.e., does not decline at higher densities) (Sparre 1980).

The next steps for the predation calculation were to develop the various components of the above equation. Availability (ϕ) of prey *i* at age *a* to predator *j* at age *b* is the product of a suitability coefficient v of prey *i* at age *a* to predator *j* at age *b* and the prey's age and year specific biomass ($B_{i,a,t}$):

$$\phi_{i,a,j,b,t} = \tilde{\nu}_{i,a,j,b,t} B_{i,a,t} \tag{12.13.1}$$

There were also species included in the model that are not explicitly modeled via the statistical catch-at-age equations in the formulation. These species interactions are described through the equation:

$$\phi_{other,t} = \tilde{\nu}_{other,t} B_{other,t} \tag{12.13.2}$$

where B_{other} refers to the biomass of the non-modeled prey with the modeled prey biomasses subtracted out (Sparre 1980):

$$B_{other,t} = B_{\mathrm{ECO}} - \sum_i \sum_a B_{i,a,t} \tag{12.13.3}$$

which is added to the summation of the explicitly modeled prey biomasses after being multiplied by their suitability coefficients. The parameter B_{ECO} is the total biomass of all of the species in the ecosystem. This component is constant over time and across species and age. The inclusion of this component allowed all of the modeled species to be estimated relative to other prey items in the ecosystem. This led to efficiencies because the ERP focal predators have a diverse diet, modeling all potential prey items (including other fish as well as invertebrates) would be a large and time intensive task, and adequate data to make inferences about the population dynamics were not available for all prey species.

The suitability (v) for each prey item at age is calculated as the product of the size and species-specific preferences of each predator by age class. Here, the size preference and the species preference were assumed independent from each other. The equation for this calculation is:

$$\mathrm{v}_{i,a,j,b} = \rho_{i,j} \mathrm{g}_{i,a,j,b} \tag{12.13.4}$$

where $\rho_{i,j}$ is the vulnerability of prey species i to predator species j, and $\mathrm{g}_{i,a,j,b}$ is the size-preference function of prey i at age a to predator j at age b. The vulnerability, ρ, incorporates all differences in food selection, for example behavioral and spatial differences, that are not attributable to size differences (Gislason and Helgason 1985). As mentioned previously, one of the factors in selecting the ERP focal species was that they have significant spatial overlap during the year, making this a reasonable assumption in this case. Species preference is relative to the "other food" group (i.e., all of the prey species not explicitly modeled). The vulnerability (ρ) and suitability parameters (ν) were set to one for the "other food" category. The main assumption for using these equations was that the size and the species were the main drivers controlling whether a predator species eats that particular food item; the other food category was assumed to be the preferred size for the predator.

Suitability coefficients (ν) were scaled across all prey species and ages to facilitate comparisons between estimated available prey biomass and food-habits data such that the suitabilities for a predator age class sum to one (Sparre 1980):

$$\tilde{\mathrm{v}}_{i,a,j,b,t} = \frac{\mathrm{v}_{i,a,j,b,t}}{\sum_i \sum_a \mathrm{v}_{i,a,j,b,t} + \mathrm{v}_{other}} \tag{12.13.5}$$

The scaling of the suitability coefficients creates a one-to-one direct correspondence between the stomach-contents of the predator and the relative suitable prey biomass.

Size preference ($\mathrm{g}_{i,a,j,b}$) of a predator was modeled as a lognormal function of the ratio between predator and prey weights as shown in the following equation:

$$\mathrm{g}_{i,a,j,b} = exp\left[-\frac{1}{2\sigma_j^2}\left(ln\frac{w_{j,b}}{w_{i,a}} - \eta_j\right)^2\right] \tag{12.13.6}$$

where σ and η are size-preference parameters specific to each predator, and w is the age-specific weight of the prey (i) and predator (j) from a specific food habit sample. Another important assumption implicit in this equation was that a predator has a single size-preference coefficient for all prey of a given size, regardless of species, but g still must differentiate between species and ages given that each prey species has a unique length and weight for a given age (Andersen and Ursin 1977, Helgason and Gislason 1979). As implemented in Curti et al. (2013), the size-preference coefficients were estimated external to the model from empirical food-habit data analysis and were input as known mean and variance parameters.

In this model formulation, the total food available to a given predator in the ecosystem may include species beyond those that are explicitly modeled. One of the benefits of this formulation, as opposed to other formulations that necessitate only using species explicitly modeled in the mathematical framework, is the inclusion of a non-modeled prey component identified as an overarching ecosystem biomass value (B_{ECO}).

The final calculation needed to determine the available prey to a predator is defined by:

$$\phi_{j,b,t} = \phi_{other} + \sum_{i}\sum_{a}\phi_{i,a,j,b,t} \qquad (12.13.7)$$

This is the divisor from Equation 12.13 and completes the steps needed to calculate predation mortality.

Given this formulation, most of the parameters can be derived by interrogating different data sources, which is preferable to making numerous assumptions. The number and weights-at-age for all modeled species were collected from both fishery-independent and dependent sources. These are standard sources of information for many stock assessments. The data elements more unique to a multispecies modeling framework were gathered from diet databases, which are now being routinely (and more systematically) collected in various state, academic, and federal fishery-independent surveys. The diet information (food habits) was derived from stomach-content analysis of the species collected; the consumption-to-biomass ratios, the preferred prey items, and preferred prey size were developed from these data. The most notable parameter that was not estimated from data is the total ecosystem biomass (non-modeled prey items). Additionally, some of the elements above were not internally estimated in the model, namely the size-preference parameters; however, this element was estimated from actual data before being input in to the model and was modeled with estimates of uncertainty.

One of the attributes of this multispecies model is the statistical estimation process. The estimated model parameters included age-specific abundances in the first year $N_{i,\,a,\,t=1}$ (*Yr1*), annual recruitment in subsequent years $N_{i,\,a=1,\,t+1}$ (*Age1*), annual fully recruited fishing mortality rates $F_{i,\,t}$, age-specific fishery ($s_{i,a}$) and survey ($r_{i,a}$) selectivity coefficients, and the vulnerability parameters, $\rho_{i,j}$. Due to the estimation of the population in the first year for all species, the model did not depend on an assumption of equilibrium. Single-species statistical models for all of the ERP focal species provided initial estimates of abundance. For all subsequent years, recruitment was estimated as a mean parameter plus a vector of annual deviation parameters that must sum to zero.

All model parameters were estimated with maximum likelihood techniques, programmed in AD Model Builder (ADMB-IDE ver 10.1 2011). In addition to the likelihood approach, a Bayesian-type approach with priors, implemented through penalized likelihoods and bounded parameters, was also used to supplement some of the statistical estimation. The estimation of model parameters allowed the inclusion of the assumption that fishery catch, survey catch, and food habits data are subject to observation error; this is a critically important expansion relative to previous multispecies formulations, in particular the virtual population analysis approaches that have been used for multispecies modeling (Helgason and Gislason 1979; Gislason and Helgason 1985; Sparre 1991; Livingston and Jurado-Molina 2000; Tsou and Collie 2001; Garrison et al. 2010).

The total likelihood comprises five components as well as three penalty functions (Table 16). The total fishery and total survey catch were assumed to be lognormally distributed. The catch-at-age proportions for both the fishery and the survey information, and predator food habits (average proportions by weight) were assumed to follow a Dirichlet multinomial distribution. These are common error distribution assumptions for fisheries stock assessments in general and were also the assumptions used for the single-species assessments for most ERP focal species.

Weightings for the lognormal components were species-specific (Table 18 - Table 23). The CVs were set such that the uncertainty associated with recreational harvest and discard levels were accounted for and were higher for species with higher recreational catch (i.e. striped bass and bluefish). Additionally, a higher CV was assumed for the survey component due to the interannual variability observed in those datasets, in each case the CV was set consistent with the choice made by the single-species assessment working group. Interannual variability can result from variation in availability of the species to the survey gear, changes in survey methodology through time, or the fact that surveys may be taking place in spatially discrete areas at different times of year; therefore it is not necessarily the case that these observed changes in abundance are real, but rather are due to changes in catchability (Pincin et al. 2014). As a result, it is appropriate to allow some significant statistical inference when predicting the various indices in the model.

For the Dirichlet objective function, sample sizes came from two sources depending on the species (Table 18 - Table 23). In cases where the total samples taken for the composition data were known, those data were used (Atlantic menhaden and Atlantic herring); total samples represented numbers of trips or survey tows sampled, not numbers of fish collected. For the other species, the effective sample sizes as used in the single-species assessments for the various composition data were used; these were generally calculated from number of trips or survey tows/hauls sampled.

Penalty functions were imposed on initial abundances, annual recruitment and age-specific biomasses (Table 16). These penalties were imposed to keep parameter estimates from collapsing to zero or producing estimates that were not biologically feasible. The penalty imposed on initial abundances, $Yr1_{PEN}$, was calculated with two methods. The first method prevented age-specific abundances from deviating substantially from those predicted by exponential decay across ages, assuming a total mortality equal to the age-specific average. The second approach penalized deviation from the initial input abundance (*Yr1*) values taken from the benchmark models for all species. This second approach was used for the final model configuration. The penalty imposed on annual recruitment, *Rpen*, prevented the coefficient of variation for the log recruitment of any species from becoming greater than a pre-defined threshold value ($R_{\mathrm{THRESHOLD}}$). The threshold selected was based on the recruitment and its associated variability from the benchmark models for the species in this study. The penalty imposed on age-specific biomasses, B_{PEN}, prevented any age-specific biomass from falling below a pre-defined threshold ($B_{\mathrm{THRESHOLD}}$) to prevent the calculations from crashing due to the biomass dropping to zero. The weights for each of these penalties and their corresponding threshold values were selected iteratively.

12.3 Results

12.3.1 Diagnostics

Model fits were compared to the observed data as a diagnostic test to show the internal performance of the model. Additionally, the output was also compared with a run that had the trophic calculations turned off (representing multiple simplified single-species assessments). Several diagnostic plots are presented to verify that the model is fitting observed data reasonably well. Model parameter estimates and their associated standard deviations and are reported in Table 24 - Table 28.

The predicted total annual fishery catch closely followed observed catches with only minor differences for all species (Figure 58). Some lack of fit to the catch data for weakfish and spiny dogfish was evident.

The fits were less exact for the indices, but the multispecies output did follow temporal trends in the observed time series fairly well (Figure 59 – Figure 64).

For both fishery (Figure 65 – Figure 70) and survey (Figure 71 – Figure 82) age proportions, the predicted trends captured much of the interannual variability seen in the observed dataset. The model did a good job at capturing the age proportions for the catch; however, the model did not fit Atlantic herring and spiny dogfish as well in a relative sense. The model did poorly at predicting the survey age proportions in some instances. The model predicted more older Atlantic menhaden than were observed in the population for the NAD and SAD surveys. The model overpredicted the youngest ages of striped bass in the MRIP CPUE survey and the fit declined as age increased for the CT LISTS survey. The model did not fit the youngest age class for bluefish in either survey used in the model. The fit to the Albatross and Bigelow surveys for Atlantic herring was poor for the youngest age class. Finally, the fit to the Albatross survey for spiny dogfish decreased with increasing age.

Food-habits data were fit without much statistical weight on the input data. This was done to acknowledge the fact that the food habit data were limited for the species examined in this project. Even with this low weight, there was good correspondence between the input values and predicted data, with the multispecies statistical model predicting smoother curves of increasing proportion of diet for prey items in the food habits of the predators (Figure 83 – Figure 86).

Contributions of the different data elements to the objective function are presented in Table 29. This information indicates that the fishery catch-age composition data contributed the most to the objective function value, followed by the fishery-independent survey age-composition, and then the total fishery-independent survey fit. There was also some contribution from the penalty functions, namely from the initial year penalty function, but these were minor contributions relative to the rest of the information. By species, Atlantic menhaden, followed by striped bass, contributed the most to the objective function value.

12.3.2 Population Estimates

Population abundance produced by the multispecies statistical model followed trends that were in line with the understanding from the current benchmark assessments for the ERP focal species (Figure 87). For Atlantic menhaden, the population began at a high level in the early part of the time series and then declined until the mid-1990s. After this, the population increased and then oscillated up and down without trend until the end of the time series. Striped bass began at a low population abundance. Striped bass population abundance then climbed until the late 1990s and was variable around this higher level until the end of the time series, with a decreasing trend. Bluefish followed a trend similar to that of Atlantic menhaden, beginning at a high level, declining, and then recovering towards the end of the time series. The most recent five years indicated a period of decline for bluefish. Weakfish, according to the multispecies model, began at a middle population size level, increased over a short period of time, and then declined for the majority of the time series. There was a short period of time at the end of the time series that indicated some recovery. Atlantic herring population abundance began at a low level and increased through the time series, with a period of decline in the last decade. The spiny dogfish population abundance began at a median level and increased in to the 1990s. The population then went through a period of decline, with a slight recovery in the final decade of the time series.

Fishing mortality estimates produced by the multispecies statistical model followed trends that are in line with the understanding from the current benchmark assessments for the ERP focal species (Figure 88), though in some instances the magnitudes were different. For Atlantic menhaden, fully recruited fishing mortality began at a high level in the early part of the time series and then decreased until the late-1980s. Fully recruited fishing mortality increased into the early 2000s, but then decreased again until the end of the time series. Striped bass fishing mortality started high, decreased sharply early in the time series, and then increased until the end of the time series. Bluefish followed a trend of decreasing fully recruited fishing mortality throughout the time series, with the exception of a sharp increase in the very beginning of the timeseries. Weakfish fully recruited fishing mortality started off low at the beginning of the time series and then increased to a peak in the mid-2000s. It has been declining since. Atlantic herring fully recruited fishing mortality began at a high level and decreased through the mid-1990s. It increased to a peak in 2010, and then decreased to the end of the timeseries. Spiny dogfish fully recruited fishing mortality began at a high level and decreased through the early-2000s. It then increased to the end of the timeseries.

Population biomass produced by the multispecies statistical model followed trends similar to the current benchmark assessments for the ERP focal species (Figure 89). For Atlantic menhaden, the population began at a high level in the early part of the time series and then declined until the mid-1990s. The population biomass increased and then oscillated up and down without trend until the end of the time series. Striped bass began at a low population biomass. Striped bass population biomass then climbed until the late 1990s and has been variable around this higher level until the end of the time series, with a decreasing trend. Bluefish biomass began at a high level, declining and then recovering towards the end of the time series. Weakfish biomass began at a high level, increased over a short period of time, and then declined for the majority of the time series. There was a short period of time at the end of the time series that indicated some recovery. Atlantic herring population biomass began at a low level and increased through the time series, with a period of decline since the early 2000s. The spiny dogfish population biomass began at a low level and increased in to the mid-1990s. The population then went through a period of decline through the end of the time series.

Recruitment estimates produced by the multispecies statistical model indicated events similar to the current benchmark assessments for the ERP focal species (Figure 90). For Atlantic menhaden, recruitment was high in the beginning of the time series, but then declined and oscillated around a low level; the most recent time period saw some higher than average recruitment events. Striped bass began with low recruitment, and then had a period of high recruitment in the middle of the time series. Recruitment was low since this time period, punctuated by two or three above average recruitment events at the end of the time series. Bluefish had a very high recruitment event early in the time series, followed by a period of lower recruitment with multiple above average recruitment events during this time period. Weakfish had two very large recruitment events in the early part of the time series, but has been in a period of very low recruitment since 2000. Atlantic herring recruitment has been without trend, with some very large events occurring throughout the time series. Spiny dogfish saw high recruitment in the early part of the timeseries, but has been in a period of lower but stable recruitment since 1995.

Estimated predation mortality (M_2) varied between the prey species in this study, by prey age, and through time (Figure 91). The predator-only ERP focal species were not prey nor did they undergo cannibalism, so time- and age-varying predation mortality was only estimated for Atlantic menhaden, weakfish, and Atlantic herring. Predation mortality was highest for age-0 Atlantic menhaden and decreased sequentially as age increased. Predation mortality increased for Atlantic menhaden beginning in the early 1990s, peaking in the mid-2000s, and declined towards the end of the time series. At its peak, the predation mortality on age-0 Atlantic menhaden approached 0.14 in several years. The terminal year estimate of M_2 for Atlantic menhaden was 0.06 for age-0 and was 0.03 on average for all other age classes. The proportion of total mortality (Z) attributed to predation mortality was highest for age-0 and age-1 Atlantic menhaden, peaking at around 10% of total mortality due to predation mortality (Figure 92). The other age classes ranged from only having 1% of total mortality due to predation up to a peak of approximately 4% (Figure 92). It is important to note that these predation rates were much lower than previous studies on Atlantic menhaden predation (Garrison et al 2010). This is in large part due to the diet data and the way it was processed for this model.

Predation mortality was highest for Atlantic herring on age-1, as was the case for Atlantic menhaden, and decreased sequentially as age increased. Predation mortality increased for Atlantic herring beginning in the early 1990s, and decreased from 2007 to the end of the time series. At its peak, the predation mortality on age-1 Atlantic herring approached 0.14. The terminal year estimate of M_2 for Atlantic herring was 0.08 for age-1 and was 0.06 on average (Figure 91). The proportion of total mortality attributed to predation mortality is highest for age-1 Atlantic herring, peaking at above 30% of total mortality due to predation mortality. The other age classes ranged from having close to 4% of the total mortality due to predation up to a peak of approximately 20% (Figure 92).

Predation mortality was highest for weakfish on age-0 as was the case for Atlantic menhaden and Atlantic herring, and decreased sequentially as age increased. Additionally, predation mortality increased for weakfish beginning in the mid-1990s, and generally decreased after 2000 to the end of the time series, though there was variability depending on which age class is being examined. At its peak, the predation mortality on age-0 weakfish approached 0.25. The terminal year estimate of M_2 for weakfish was 0.14 for age-0 and was 0.05 on average for all other age classes (Figure 91). The proportion of total mortality attributed to predation mortality was highest for age-0 weakfish, peaking at above 20% of total mortality due to predation mortality. The other age classes ranged from having close to 2.5% of the total mortality due to predation up to a peak of approximately 15% (Figure 92).

Although predation mortality was always highest on age-0 and age-1 fish, different species showed different patterns in terms of total mortality ($F + M_0 + M_2$) at age (Figure 93). For Atlantic menhaden, total mortality was highest on age-0 fish and decreased at older ages across the time series; for Atlantic herring, the pattern was reversed, with total mortality being the lowest on age-0 fish and increasing with age. For weakfish, the pattern changed over time, with total mortality being highest on age-0 and age-1 in some years and highest on older ages in years with higher *F*.

Food-habit information was queried from the diet information from the NMFS trawl survey, the NEAMAP survey, and the ChesMMAP survey and was processed through a Bayesian multinomial probit model to account for the dearth of data in many instances. The food habits of striped bass predicted by VADER created more consistent proportions across time than the original input data; the food habits showed that prey not explicitly modeled make up the largest proportion of striped bass diet, with Atlantic menhaden making up the next most important proportion (Figure 83). Atlantic herring and weakfish constituted a small proportion of the overall diet for striped bass. The “other food” category (all prey items not explicitly modeled) constituting close to 80% of the remaining diet. This trend held throughout the timeframe examined in this study with small differences in each aggregated year period.

A similar trend in the output from the statistical model from this study was seen for bluefish, weakfish, and spiny dogfish (Figure 84 - Figure 86). The estimation by the model predicted proportionally very little consumption of the prey explicitly included in the model, with the "other food" category constituting over 95% of the remaining diet. This trend= held throughout the timeframe examined in this study with small modifications in each aggregated year period.

Consumption of prey as an output of the multispecies model can be represented as thousands of metric tons, and therefore can be viewed in similar currency to catch and other population biomass information. Striped bass consumption of Atlantic menhaden closely followed the trajectory of population size for Atlantic menhaden and trended upward with the increase in population size for striped bass in the time-series (Figure 94). The proportional amount of Atlantic menhaden in striped bass diets increased as this prey item increased in abundance. When striped bass population size was low, the magnitude of Atlantic menhaden consumption was only 50 thousand metric tons (Figure 95). As the striped bass population size increased through time, consumption of Atlantic menhaden also increased, rising to a maximum value of ~200 thousand metric tons in 2010. Consumption of Atlantic herring by striped bass was relatively low for the entire time series, ranging from close to one thousand metric tons to a maximum of ~30 thousand metric tons in 1996. Striped bass was the predominant predator on weakfish, but overall the magnitude was very low.

Bluefish consumption of Atlantic menhaden remained relatively flat and low for the time series examined in this study (Figure 94). The proportional amount of Atlantic menhaden in bluefish diets decreased in the 1990s, coincident with a low population period for both Atlantic menhaden and bluefish. The magnitude of Atlantic menhaden consumption by bluefish ranged from ~25 to 100 thousand metric tons (Figure 95). Consumption of Atlantic herring by bluefish was relatively low for the entire time series, ranging from 20 thousand metric tons to a maximum of 25 thousand metric tons. The remainder of bluefish consumption was attributed to the other prey items that are not explicitly modeled and ranged from ~750 to 1,800 thousand metric tons, which occurred in 1985.

Weakfish consumption of Atlantic menhaden was variable through the time series examined in this study and correlated well with weakfish and Atlantic menhaden population abundance (Figure 94). The proportional amount of Atlantic menhaden in weakfish diets decreases in the early 1990s, coincident with a low population period for both Atlantic menhaden and weakfish. The magnitude of Atlantic menhaden consumption by weakfish ranged from 0.4 to 8 thousand metric tons (Figure 95). Consumption of Atlantic herring by weakfish was low for the entire time series, ranging from 0.03 thousand metric tons to a maximum of 0.4 thousand metric tons in 2009. The remainder of weakfish consumption was attributed to the other prey items that were not explicitly modeled in this study and ranges from 77 to ~800 thousand metric tons, which occurred in 1993.

Spiny dogfish consumption of Atlantic menhaden was variable and low through the time series (Figure 94). The magnitude of Atlantic menhaden consumption by spiny dogfish ranged from

0.2 to 5 thousand metric tons (Figure 95). Consumption of Atlantic herring and weakfish by spiny dogfish was low for the entire time series, ranging from 0.05 thousand metric tons to a maximum of 4 thousand metric tons in 2016 for Atlantic herring and 0.004 to 0.3 thousand metric tons for weakfish. The remainder of spiny dogfish consumption was attributed to the other prey items that were not explicitly modeled in this study and ranges from 8 to ~120 thousand metric tons, which occurred in 2000.

When viewing consumption by prey item, the importance of each predator in the consumption of each prey species can be seen. Striped bass consumed the most Atlantic menhaden relative to the other predators examined in this study (Figure 95). Bluefish was the next most important predator for Atlantic menhaden; bluefish also consumed more Atlantic menhaden than the other predators in this study in the early portion of the time series. Weakfish was also an important predator of Atlantic menhaden; however, given the low population numbers for weakfish during the time series used for this study, its impact on the Atlantic menhaden population was relatively small.

For Atlantic herring, it was bluefish that consumed the most Atlantic herring relative to the other predators examined in this study, followed by striped bass (Figure 95). As was the case for Atlantic menhaden, bluefish consumed more Atlantic herring than the other predators in this study in the early portion of the time series when bluefish abundance was high. Weakfish did not appear to be an important predator for Atlantic herring, and spiny dogfish only contributed significant amounts of predation in certain years.

For weakfish, it was striped bass that consumed the most weakfish relative to the all other predators examined (Figure 95). Spiny dogfish was the only other predator that appeared to contribute significantly to the predation of weakfish, but this was only in certain years, and was at a much lower magnitude than striped bass.

Estimates of recruitment, total abundance, total biomass, and fishing mortality were virtually indistinguishable for runs with tropic calculations turned on and the runs with the trophic calculations turned off for non-prey species (striped bass, bluefish, and spiny dogfish) (Figure 87 – Figure 90). For prey species (Atlantic menhaden, Atlantic herring, and weakfish), runs with the trophic calculations turned off had higher estimates of recruitment, total abundance, and total biomass, but generally similar estimates of F, although there were some differences early in the time series for Atlantic menhaden and later in the time series for weakfish (Figure 87 – Figure 90). This was most likely due to differences in the estimates of natural mortality used in the model. For runs with the trophic calculations turned off, the single-species assessment value of M was used; for runs with the trophic calculations turned on, a scaled down estimate of the single-species M was used as the input non-modeled-predation natural mortality component (M_0) for prey species, and the model calculated an additional component of natural mortality attributed to the predators in the model (M_2). The estimates of M_2 were a small component of total M, and in effect, the runs with the trophic calculations turned off used a higher M value for prey species than the runs with the trophic calculations turned on, resulting in higher estimates of recruitment and abundance, but relative similar estimates of F, as would be expected. The natural mortality for species that acted only as predators was the same in both sets of runs, as the trophic calculations did not include effects on predators; as a result, the estimates of recruitment, population size, and *F* were very similar between the two runs.

12.4 Sensitivity Analyses

Two main sensitivity analyses were conducted for VADER. The first was to test the model's performance relative to a change in the input surveys. As noted, only a subset of indices used in the single-species assessments were included in the base run of VADER. Because of this, sensitivity runs were conducted to determine the importance of these choices; age-1+ indices in the base run were replaced with an alternate age-1+ index as identified by the single-species TCs (Table 17).

A second sensitivity analysis was conducted to test the sensitivity of the model to the input diet data. Instead of the Bayesian multinomial probit model as described and used for the base run, the food habits data (diet proportion of each prey for each predator) output from the previously performed MSVPA-X model (Garrison et al 2010) was used as the food habits input dataset. The food habits as produced by the MSVPA-X used numerous additional sources that were not incorporated in to this model due to missing datasets, and this in turned changed the food habit information significantly, in particular for striped bass. Given these differences, testing the effect on the output was an important sensitivity to undergo.

Generally, the run with the alternative indices had the greatest effect on the predator only species, while the alternative diet data had the greatest effect on the prey species. Annual total abundance showed some differences relative to the base run in the runs with the alternate indices for striped bass, bluefish, and spiny dogfish; however, neither of the sensitivity runs (alternate indices and alternate food habits) indicates a large effect on the model output (Figure 96). For the case of Atlantic menhaden and Atlantic herring, the alternate diet data had significant impacts on the abundance output, with the alternate diet data run increasing the total abundance for these two species. For spiny dogfish, neither sensitivity run indicated much difference from the base run.

Annual fully recruited fishing mortality indicated some departure from the base run from the alternate indices for striped bass, bluefish, weakfish, and Atlantic herring, while Atlantic menhaden and spiny dogfish showed little difference (Figure 97). Weakfish indicated a very different trend in *F* at the end of the time series. For the alternate diet run the biggest effects occurred on the species that are prey in the model (Atlantic menhaden, Atlantic herring), with Atlantic menhaden indicating higher *F* earlier in the time period and lower in the most recent period, and Atlantic herring showing lower *F* rates from this run for the entire time series. Weakfish indicated pretty good coherence to the base run when examining the alternate diet run.

Annual total biomass showed little effect from the alternate indices run across all species (Figure 98). For the alternate diet run the biggest effects occurred on striped bass, bluefish, weakfish, and Atlantic herring; however, there was no consistency in trend, with some species indicating higher abundance and some lower. The effect was more modest or non-existent for the other species.

Annual recruitment was significantly affected by the alternate diet run for Atlantic menhaden and Atlantic herring, with the alternate diet run showing increased recruitment for these species across the time series (Figure 99). The alternate indices only indicated a modest effect relative to the base run, with the biggest impact occurring in weakfish, which generally showed less biomass over the time period under this model configuration.

Average predation mortality (the average predation mortality (M_2) across all age classes) was examined for the alternate diet run relative to the base run. The alternate indices were not examined in this comparison. There were large effects across all three prey species (Figure 100). Atlantic menhaden and Atlantic herring indicated significantly higher predation mortality under the alternate diet information, while weakfish indicated significantly lower predation mortality. This highlights the importance of the diet information as an input to the multispecies model.

12.5 Retrospective Analysis

A retrospective analysis was done on the multispecies iteration of VADER to look at the stability of the model as years of data are added. A retrospective pattern is a systematic inconsistency among a series of estimates of population size, or related assessment variables, based on increasing periods of data (Mohn 1999). This is a standard analysis performed on many single-species assessments and therefore will be an important test for the VADER model to examine the consistency in output from year to year as more information becomes available to the model.

A three-year peel was performed for the VADER model. Three years was chosen because this was a period where the food habits data did not need to be altered to accommodate the new timeseries length. The food-habit data were binned by three-year periods to allow for some dampening of the inherent variability in the food habit data. The food-habit data bins were a limiting factor for the retrospective analysis because, if the time series was reduced by more than three years, a reconstruction of the food habit data would have been needed, making year-to-year comparisons difficult.

A sequential year was dropped from the terminal year of the assessment (2017) for three years, and the model was rerun for each of those three new datasets. The data changed for each run included the total catch, the weight-at-age, maturity, the catch-at-age, the total survey catch, and the survey catch-at-age for each species, along with uncertainty estimates for each of these elements (sample sizes and CVs). The outputs examined were total fishing mortality, biomass, and recruitment.

The severity and direction of the pattern was determined by using the Mohn's Rho statistic. Mohn's Rho (Mohn 1999) has been commonly used to measure the retrospective patterns for many stocks, including for assessments done on the species examined in this study. The statistic is defined as the sum of relative difference between an estimated quantity from an assessment with a reduced time series and the same quantity estimated from the full time series:

$$\rho = \sum_{t} \frac{X_{t_{new}} - X_{t_{full}}}{X_{t_{full}}} \tag{12.14}$$

where X denotes the variable from the assessment (in this case full fishing mortality, total biomass, or recruitment), t denotes the year of comparison, t_{new} denotes the terminal estimate from an assessment with a reduced time series, and t_{full} denotes the assessment using the full time series. To make the statistic comparable across different numbers of reduced years (i.e. peels), Miller and Legault (2017) reconfigured the estimator to be defined as the average of the peel-specific components:

$$\rho_t = \frac{X_{t_{new}} - X_{t_{full}}}{X_{t_{full}}} \tag{12.14.1}$$

$$\bar{\rho} = \frac{1}{P} \sum_{t=earliest\ year}^{P} \rho_t \tag{12.14.2}$$

Where ρ_t = the peel year specific ρ value and P = the total number of years peeled.

The retrospective analysis performed well and indicated relatively good stability for the species in the main population metrics examined. Fishing mortality indicated a retrospective pattern where the population total fishing mortality was overestimated for Atlantic menhaden, bluefish, and spiny dogfish, and underestimated for the other species. These patterns were generally weak (less than 0.13) as indicated by the Mohn's Rho diagnostic for all six species ($\rho_{menhaden}$ = -0.12, $\rho_{striped\ bass}$ = 0.04, $\rho_{bluefish}$ = -0.002, $\rho_{weakfish}$ = 0.12, $\rho_{herring}$ = 0.03, $\rho_{dogfish}$ = -0.03; Figure 101).

Total biomass indicated a retrospective pattern where the population total biomass was overestimated for weakfish and spiny dogfish and underestimated for the other species. These patterns were weak as indicated by the Mohn's Rho diagnostic for all species ($\rho_{menhaden}$ = -0.03, $\rho_{striped\ bass}$ = -0.10, $\rho_{bluefish}$ = -0.03, $\rho_{weakfish}$ = 0.08, $\rho_{herring}$ = -0.07, $\rho_{dogfish}$ = 0.04). This feature is something often seen in this type of retrospective pattern, namely if fishing mortality is underestimated, biomass is frequently overestimated simultaneously, and vice versa (Figure 102).

Recruitment indicated a retrospective pattern where recruitment was overestimated for bluefish and spiny dogfish and underestimated for the other species. This population metric had more variability than the previous two metrics, and showed different patterns and severity depending on the species. The pattern was fairly strong for striped bass, bluefish, and Atlantic herring, and for the other species, the pattern was weak to modest ($\rho_{menhaden}$ = -0.16, $\rho_{striped\ bass}$ = -0.32, $\rho_{bluefish}$ = 0.28, $\rho_{weakfish}$ = -0.12, $\rho_{herring}$ = -0.59, $\rho_{dogfish}$ = 0.19; Figure 103).

In a qualitative sense, the retrospective patterns found in the analysis were on par with or less than those found in the benchmark assessments for these species. It is difficult to make a direct quantitative assessment of this comparison as not all of these benchmark assessments calculated Mohn's Rho statistics or published data that could be analyzed. However, when reviewing the information provided in the benchmark assessment documents, the retrospective patterns found in this study were generally the same or better in a diagnostic context.

12.6 Projections

Data into and output from the base run of the VADER model, as described above, were used as the basis for these projections, including the data for SSB, recruits, and recruitment deviations. The model outputs were exported from ADMB software (ADMB-IDE ver 10.1 2011) and imported to R statistical software (R Core Team 2016) for the projection calculations.

The starting conditions of the projection analysis include initial numbers at age, which were the estimated numbers at age, N_0, for the terminal year of the multispecies stock assessment model. To allow for variability in the projection starting population, a bootstrap procedure was used for recruitment and for numbers-at-age for ages older than the modeled recruits. The bootstrap procedure added a deviation to the starting numbers-at-age, the deviation was based on sampling from a normal distribution with a mean of 0 and a standard deviation set at the standard deviation seen in the population for the time period examined. This deviation was bounded to prevent very large deviations from occurring randomly through the sampling process.

Numbers at age after the initial year were calculated as:

$$N_{i,a+1,y+1} = N_{i,a,y}e^{-Z_{i,a,y}} \tag{12.15}$$

where Z is age and year specific mortality and equals natural mortality for each age for that year plus the fishing mortality rate times the fishery selectivity at age, $N_{i,a,y}$ is the population by age and year, and the subscript i is the species. Fishery selectivity was a vector as estimated for each species from the multispecies stock assessment.

For the constant-F scenarios used for this model, the landings associated with the chosen F strategy were calculated. These annual landings were calculated using the Baranov catch equation and weight of landings.

$$C_a = \frac{F_a}{F_a + M_a}\left(1 - e^{-(F_a+M_a)}\right)N_a \tag{12.16}$$

Where C is catch, F_a is fishing mortality at age, M_a is natural mortality at age, and N_a is the population at the start of the year. In this case, the Baranov catch equation was used so that F

was the input variable and catch was estimated from the input *F*. The catch and population in numbers were converted into biomass units, and the weight-at-age for each species was assumed to be equal to the species-specific average weight-at-age. This weight-at-age was projected forward in a static fashion.

SSB was calculated for each species and was based on the biomass-at-age, as estimated for each year in the projection, multiplied by the maturity-at-age vector from the terminal year of the multispecies stock assessment model. In this case, all SSB was represented in the estimate and therefore comprised both male and female biomass. Spawning was assumed to occur mid-year for all of the species in the model, therefore the SSB was decreased by total mortality for half a year.

Recruitment was projected without an underlying stock-recruitment function and was based on the median recruitment observed from the entire time series for each species. Recruitment variability was included whereby for each year a deviation in recruitment was selected randomly with replacement from the deviations estimated in the multispecies stock assessment model. This may have been overly restrictive assumption in that it was impossible to have recruitment overfishing in a population, however this strategy was chosen due to the lack of good stock-recruitment information and because this is the standard approach in stock assessments of most of the ERP focal species. The projection methods allowed for the inclusion of a Ricker stock recruitment curve as an option, but this was not used for the projections described here.

Projections were run for 100 years to allow the populations to reach equilibrium. The projections were parameterized as above with the exception of the fishing mortality assumptions that were defined a priori, and these projection runs were done using a dynamic *M* formulation.

The projection was run allowing *M* to be calculated dynamically. The description of the dynamic M_2 calculations followed the procedure as defined in Equations 12.13 – 12.13.7. The projections were run in a stochastic fashion. The projection parameters were bootstrapped for two-hundred iterations for the long-term projections, with the initial population and recruitment bootstrapped with uncertainty based on the timeseries from the multispecies model. Outputs included the median, 5th and 95th percentiles for spawning stock biomass, recruitment, landings, and natural mortality for the prey species.

Fishing mortality (*F*) was set to meet the management goal of maintaining an *F* rate at predetermined scenarios of management interest. Four scenarios were conducted as follows:

1. The projections were run setting *F* for the predators and non-menhaden prey (weakfish and Atlantic herring) at their target *F* rates (striped bass = 0.2, bluefish = 0.14, weakfish = 0.55, Atlantic herring = 0.46, and spiny dogfish = 0.22). Atlantic menhaden in this scenario was set at its status quo *F* rate, meaning the *F* rate in 2017 as calculated by the current single-species model (Atlantic menhaden = 0.11).

2. A second projection was run setting *F* for the predators and non-menhaden prey (weakfish and Atlantic herring) at their target *F* rates as above. Atlantic menhaden in this scenario was set at its target *F* rate (Atlantic menhaden = 0.22).
3. A third projection was run setting *F* for the predators and non-menhaden prey (weakfish and Atlantic herring) at their status quo *F* rates as determined from the current single-species assessments (striped bass = 0.31, bluefish = 0.34, weakfish = 0.23, Atlantic herring = 0.45, and spiny dogfish = 0.15). Atlantic menhaden in this scenario was set at its status quo *F* rate, as defined above.
4. A final projection was run setting *F* for the predators and non-menhaden prey (weakfish and Atlantic herring) at their status quo *F* rates as defined above. Atlantic menhaden in this scenario was set at its target *F* rate, as defined above.

Projection Results

In scenario 1 as defined above (Atlantic menhaden at status quo *F*, other species at their *F* targets), the prey species SSB were flat to declining and the predators increased or were flat, with the exception of spiny dogfish, which declined. Atlantic menhaden started the projection at ~700 tmt and increased to ~800 tmt of SSB by year 100 (Figure 104). Atlantic herring began the projection at ~280 tmt and ended at ~225 tmt of SSB (Figure 105). Striped bass SSB began at ~50 tmt and ended at ~90 tmt (Figure 106). Bluefish began at ~100 tmt and ended at ~290 tmt of SSB (Figure 107). Weakfish began at ~12 tmt and ended at ~10 tmt of SSB (Figure 108). Spiny dogfish began at ~70 tmt and ended at ~20 tmt of SSB (Figure 109).

Natural mortality (*M*) was occurring dynamically on the prey species Atlantic menhaden, Atlantic herring, and weakfish in this projection scenario. There was an initial increase in *M* for Atlantic menhaden, with ages-0 and 1 having the highest *M* occurring on them. After the initial increase, the *M* rates stabilized for the remainder of the projection (Figure 104). For Atlantic herring, *M* increased in the first few years on all ages and then stabilizes. The *M* rate was similar across ages (Figure 105). For weakfish, *M* increased slightly in the first few years on all ages and then stabilized (Figure 108).

In scenario 2 as defined above (all species at their *F* targets), the prey species SSB declined and the predators increased or were flat, with the exception of spiny dogfish, which declined. Atlantic menhaden started the projection at ~700 tmt and decreased slightly to ~650 tmt of SSB by year 100 (Figure 110). Atlantic herring began the projection at ~280 tmt and ended at ~220 tmt of SSB (Figure 111). Striped bass SSB began at ~50 tmt and ended at ~90 tmt (Figure 112). Bluefish began at ~100 tmt and ended at ~290 tmt of SSB (Figure 113). Weakfish began at ~12 tmt and ended at ~9 tmt of SSB (Figure 114). Spiny dogfish began at ~70 tmt and ended at ~20 tmt of SSB (Figure 115).

There was an initial slight increase in *M* for Atlantic menhaden, with ages-0 and 1 having the highest *M* occurring on them. After the initial increase, the *M* rates stabilized for the remainder of the projection (Figure 110). For Atlantic herring, *M* increased in the first few years on all ages and then stabilizes. The *M* rate was similar across ages (Figure 111). For weakfish, *M* increased slightly in the first few years on all ages and then stabilized (Figure 114).

In scenario 3 as defined above (all species at status quo *F*), the prey species SSB increased and the predators increased, with the exception of spiny dogfish, which declined. For the predators, the increase was less than in the previous two scenarios, and spiny dogfish declined more modestly. Atlantic menhaden started the projection at ~700 tmt and increased to ~900 tmt of SSB by year 100 (Figure 116). Atlantic herring began the projection at ~280 tmt, had a short period of decline, and ended at ~300 tmt of SSB (Figure 117). Striped bass SSB began at ~50 tmt and ended at ~60 tmt (Figure 118). Bluefish began at ~100 tmt and ended at ~105 tmt of SSB (Figure 119). Weakfish began at ~12 tmt and ended at ~17 tmt of SSB (Figure 120). Spiny dogfish began at ~70 tmt and ended at ~45 tmt of SSB (Figure 121).

The *M* rate for Atlantic menhaden was flat for the entire time series, with ages-0 and 1 having the highest *M* occurring on them (Figure 116). For Atlantic herring, *M* was stable for the time series. The *M* rate was similar across ages (Figure 117). For weakfish, *M* increased slightly in the first few years on all ages and then stabilized (Figure 120).

In scenario 4 as defined above (Atlantic menhaden at *F* target, others at status quo *F*), the prey species SSB was flat or increasing and the predators increased modestly, with the exception of spiny dogfish, which declined. As in scenario 3, the increase was less than in scenarios 1 and 2 and spiny dogfish declined more modestly as well. Atlantic menhaden started the projection at ~700 tmt and increased to ~700 tmt of SSB by year 100 (Figure 122). Atlantic herring began the projection at ~280 tmt, had a short period of decline, and ended at ~300 tmt of SSB (Figure 123). Striped bass SSB began at ~50 tmt and ended at ~60 tmt (Figure 124). Bluefish began at ~100 tmt and ended at ~105 tmt of SSB (Figure 125). Weakfish began at ~12 tmt and ended at ~17 tmt of SSB (Figure 126). Spiny dogfish began at ~70 tmt and ended at ~45 tmt of SSB (Figure 127).

The *M* rate for Atlantic menhaden was flat for the entire time series, with ages-0 and 1 having the highest *M* occurring on them (Figure 122). For Atlantic herring, *M* was stabile for the time series. The *M* rate was similar across ages (Figure 123). For weakfish, *M* was stabile for the timeseries (Figure 126).

13 INTERMEDIATE COMPLEXITY ECOPATH WITH ECOSIM MODEL (NWACS-MICE) (PREFERRED)

A full Northwest Atlantic Continental Shelf (NWACS) ecosystem model was developed using EwE by Buchheister et al. (2017a, 2017b) to inform Atlantic menhaden management in an ecosystem context and an updated version of this model was produced for this assessment (Section 14). To provide an intermediate level of complexity, a Model of Intermediate Complexity for Ecosystem assessment, or MICE model (Plaganyi et al. 2014; Collie et al. 2016; Punt et al. 2016) based on the full NWACS model was developed using EwE. The NWACS-MICE model was restricted in complexity to focus on key species that interact with one another through food web interactions and are also regularly assessed and managed by ASMFC. As a proof of concept, a simple Atlantic menhaden-striped bass EwE model was first developed and reviewed by the ERP WG in summer 2018. This single predator model was later expanded by the ERP WG to include bluefish, weakfish, spiny dogfish, and Atlantic herring, all of which undergo regular population assessments and/or were determined to be important predators on Atlantic menhaden (see Section 3.1.2: Identification of Key Predator and Prey Species). Anchovies (*Anchoa spp.*) were added to the NWACS-MICE model because they represent a major prey item for bluefish and are prey for other modeled species. Benthic invertebrates, zooplankton, phytoplankton, and detritus were also included in the NWACS-MICE model. Therefore, the NWACS-MICE model strikes a level of complexity slightly above VADER but below the full NWACS model. It also serves to link the dynamics of individual stock assessments with feedbacks to predators so that harvest policies for multiple species can be simulated simultaneously and tradeoffs evaluated.

13.1 Ecopath with Ecosim Modeling Framework

The EwE trophic dynamic modeling package facilitates management of basic biomass and food web data for whole ecosystems and has been widely used for analysis of aquatic resources (Pauly et al. 2000; Christensen and Walters 2004; Colléter et al. 2015). The Ecopath component of EwE is a static, mass-balance view of the ecosystem that allows for age structure representation and provides the initial state for dynamic modeling. One of the main assumptions of the modeling framework is that the system is mass-balanced over the course of the year. Ecopath assumes mass balance between groups based on how production is allocated among fishing, predation, other mortality, and migration. The basic data requirements for Ecopath are biomass, total mortality or production rate, consumption rate, diet composition, landings, and discards for each trophic group. Ecopath relies in part on setting up a system of linear equations in which three of the following four parameters are inputted for each group (solving for the fourth): biomass, production/biomass ratio, consumption/biomass ratio, and ecotrophic efficiency (EE). Typically, EE is estimated for each group, and EE is defined as the proportion of the production that is utilized in the modeled ecosystem and accounted for by fishing, predation, migration, and biomass accumulation. For full details on the underlying theory, assumptions, equations, and model mechanics, see the original sources (Walters et al. 1997; Christensen and Walters 2004; Christensen et al. 2008).

In Ecosim, biomass dynamics are modeled on a monthly time step as a series of differential equations, where change in biomass is predicted as consumption minus losses to predation, fishing, and migration (Walters et al. 1997). In Ecosim, consumption is modeled based on the foraging arena theory, which states that predator–prey interactions are restricted to spatial and temporal arenas (Ahrens et al. 2012). Models can include both environmental forcing functions, which drive long term and seasonal patterns of primary production, and mediation effects, which allow a third-party organism to either facilitate or protect against a predator-prey interaction.

The most sensitive parameters in Ecosim models are the vulnerability parameters, V_{ij}, which describe the exchange rates of prey *i* from not vulnerable states into vulnerable “foraging arenas,” where they can be consumed by predator *j*. The V_{ij} parameters control the amount of prey biomass available for consumption and are input in Ecosim as multipliers on Ecopath base predation mortality rates (M_{2ij}) to represent the maximum possible predation mortality rate (M_{2MAX}) that can be exerted on a prey item at high predator biomasses. The V_{ij} parameters must be greater than or equal to 1. Low values restrict flow into the vulnerable state, which thereby limits consumption and prevents any biomass gains in the predator. High V_{ij} values imply strong top-down effects and can lead to dynamic instability in Ecosim models. To simulate a population, increase of an overexploited or invasive predator species with a low initial biomass and low M_{2ij} on their prey, the V_{ij} parameters must be quite high in order for consumption (and therefore biomass) of the predator to increase.

Ecosim models are typically fit to time series data by first identifying the most sensitive V_{ij} and then searching for the values that minimize the sum of squares between predicted and observed values. A weight may be assigned to each data series used in calibration. The weighting scheme may vary, but usually follows conventional approaches of estimating the variability in observed data (i.e. 1/cv). The weights may be adjusted upwards to emphasize fits to species of particular interest. Examples of model fitting procedures are described in the literature (Buchheister et al. 2017b, Chagaris et al. 2015, Heymans et al. 2016). New sensitivity routines in EwE are under development that allow for propagation of uncertainty in input data through all Ecosim simulation routines. Additionally, the multisim framework can facilitate rapid analysis of alternative vulnerability exchange rate parameters (see Chagaris et al. 2017 for an example using this approach).

13.2 Ecopath Model Description

Spatial Domain

The spatial domain for the model is the NWACS ecosystem, which spans the continental shelf of the Northwest Atlantic Ocean from North Carolina to Maine (Figure 128). The model domain includes four continental shelf subregions, following the regional strata of the NEFSC trawl survey: Mid-Atlantic Bight, Southern New England, Georges Bank, and Gulf of Maine. Our model also represents the estuaries along the coastline, such as the Chesapeake Bay, Delaware Bay, and Long Island Sound (Figure 128), given that diet and biomass data from estuaries were included in the model parameterization. Although the domain does not encompass the entire distributional range of Atlantic menhaden (from Florida to Nova Scotia), it is similar to the range in the MSVPA-X developed for Atlantic menhaden (Garrison et al. 2010) and to existing Ecopath models for the region (Link et al. 2008). This domain relies on the natural faunal and oceanographic break in NC (Longhurst 1998), while also including the bulk of historical Atlantic menhaden fishing effort concentrated in Chesapeake Bay and the Mid-Atlantic (SEDAR 2015). The area of the model domain (used to calculate biomass densities) was 246,662 km^2.

Temporal Structure

The NWACS-MICE Ecopath model base year is 1985, which is the earliest year included in all stock assessments for the ERP focal species.

Trophic Structure

The NWACS-MICE model simulates the dynamics of 17 biomass pools, including striped bass (3 age stanzas), Atlantic menhaden (2 age stanzas), spiny dogfish, bluefish (2 age stanzas), weakfish (2 age stanzas), Atlantic herring (2 age stanzas), anchovies, benthic invertebrates, zooplankton, phytoplankton, and detritus (Table 30). Multiple age stanzas were included for key species to represent trophic ontogeny, fishery selectivity, and size/age dependent predation.

Fishing Fleets

In Ecopath, a separate “fleet” was included for each group, where each fleet only captures one species (i.e. bycatch is not included), total landings are combined over gear types/sectors, and discards are not modeled separately.

13.2.1 Basic Inputs

The basic data requirements for Ecopath are biomass (*B*), production to biomass rate (*P/B*; equivalent to the total mortality rate, *Z*), consumption to biomass rate (*Q/B*), diet composition, and landings for each trophic group. Biomass accumulation rates, which describe the instantaneous rate of change of a functional group’s biomass to account for groups that are not in equilibrium, can also be provided.

Biomass

Biomass inputs (million metric tons) were obtained either directly from stock assessments or by simply adding the biomass of lower trophic level groups from the full NWACS model. For all the assessed species, biomass was taken directly from the single species assessment report files as the mid-year 1985 biomass (when available) or calculated as the mean 1985 biomass-at-age,

$$\overline{B_a} = w_a * N_a * \frac{(1-e^{-Z_a})}{Z_a}, \tag{13.1}$$

and summed over ages for each Ecopath age stanza. For multistanza groups in Ecopath, biomass is only input for a single age stanza (usually the oldest) and then calculated by Ecopath for other stanzas based on input growth and mortality parameters. Details for biomass calculations of each group are provided below.

- Striped bass were last assessed in 2018, with data through 2017, using a statistical catch-at-age model. The 1985 mean biomass of age 6+ striped bass was estimated using Equation 13.1 and the January 1 N-at-age, *Z*-at-age, and Rivards weights from the statistical catch-at-age model. Age 6+ biomass input was estimated to be 18,486 mt, and biomass for age 0-1, and age 2-5 striped bass was calculated by Ecopath as 36,158 mt and 8,415 mt respectively (Table 30).
- Atlantic menhaden biomass was derived from the 2019 BAM assessment model that simulated Atlantic menhaden population dynamics from 1955-2017. The BAM report file provided estimates of 1985 mid-year biomass at age. These estimates were simply summed for adult (age 1+) Atlantic menhaden for a biomass input of 1,704,469 mt. Juvenile (age-0) Atlantic menhaden biomass was calculated by Ecopath to equal 281,721 mt.
- Spiny dogfish are assessed using biomass estimates from density and area swept by the NEFSC spring bottom trawl survey. These estimates were available from 1968-2017 and

exhibit high interannual variability. In 1985, the total biomass of spiny dogfish (all sexes and size classes) was estimated to be 1,056,700 mt from the trawl survey. This value is the second highest observed in the time series and about 4x higher than the average biomass in preceding years. When input to Ecopath, this resulted in severe mass imbalance for spiny dogfish prey. Therefore, the average biomass of 1984 and 1986 was used as the input to Ecopath, which was estimated to be 271,555 mt.

- Bluefish biomass inputs were derived from the 2019 statistical catch-at-age model developed using the age structured assessment program (ASAP). Mean annual biomass in 1985 was calculated using Equation 13.1 and N-at-age, *Z*-at-age, and weight-at-age matrices from the ASAP base run of the preliminary assessment update. Input 1985 biomass for adult bluefish (age 1+) was calculated as 219,654 mt and juvenile bluefish biomass (age-0) was calculated by Ecopath to be 4,325 mt.
- Weakfish biomass was derived from preliminary runs of the 2019 ASAP model using the N-at-age, *Z*-at-age, and weight-at-age matrices to obtain mean annual biomass. Biomass for adult weakfish (age 1+) was estimated by ASAP to be 12,703 mt and the juvenile weakfish stanza (age-0) biomass was estimated at 1,222 mt by Ecopath.
- Atlantic herring were last assessed in 2018 using ASAP. Numbers-at-age in 1985 were converted to mean biomass at age using Equation 13.1. Adult Atlantic herring (age 2+) biomass was estimated to be 149,741 mt and the juvenile biomass (age 0-1) was calculated by Ecopath to be 8,322 mt.
- Anchovy biomass was taken directly from the full NWACS model and converted to units of million metric tons. The biomass of anchovies in the full NWACS model (Buchheister et al. 2017a, 2017b) was reported to be 1.1 mt/km^2 with a model area of 246,662 km^2. This converts to a biomass of 271,328 mt for the NWACS-MICE model.
- Benthic invertebrate biomass was calculated by summing the biomass of polychaetes, crustaceans, molluscs, other macrobenthos, filter feeders, other megabenthos, and shrimp from the NWACS model (groups 9-15 from the full NWACS) and multiplying by model area. Input biomass of benthic invertebrates was estimated to be 14,546,250 mt.
- Zooplankton biomass was calculated as the sum of five biomass groups from the full NWACS and includes microzooplankton, small copepods, large copepods, gelatinous zooplankton, and micronekton. Input biomass of zooplankton in the MICE model was estimated to be 13,558,763 mt.
- Phytoplankton and detritus biomasses were taken directly from the full NWACS and multiplied by the model area. Biomass inputs for phytoplankton and detritus are 8,596,470 mt and 12,974,000 mt, respectively.

Biomass Accumulation Rates

The species included in the NWACS-MICE Ecopath model are not necessarily required to be in steady-state during the Ecopath base year (1985). In fact, it is more reasonable to assume that species biomass is changing during the base year period. To represent non steady-state in Ecopath, biomass accumulation rates were used. The biomass accumulation rate is a flow term, also expressed as a rate of change (i.e. proportion of input biomass), where a negative value signifies biomass depletion during the model period and a positive value indicates biomass gains. If the biomass for a group is known, e.g., at the beginning of the year and at the beginning of the next year, biomass accumulation can be calculated as the difference between these values.

The biomass accumulation rate parameters have several important effects on the model. First, they can be used to adjust the calculated biomass for non-leading (typically younger) age stanzas to better match the population structure in the base year of the stock assessment. In this case, a high biomass accumulation rate will shift the age distribution to younger ages leading to more biomass in those age stanzas and possibly lower ecotrophic efficiencies (EEs). Second, biomass accumulation rate inputs have a clear effect during the first few years of an Ecosim simulation. Higher biomass accumulation rates will lead to initial increases in Ecosim simulations, often leading to better fits when a species is increasing rapidly as a result of rebuilding efforts (or vice versa when a species is being rapidly depleted and the biomass accumulation rate is negative).

Biomass accumulation was entered for all assessed species except weakfish and spiny dogfish (Table 30). For multistanza groups, a single biomass accumulation rate is input for all stanzas (i.e. one biomass accumulation rate parameter for all striped bass stanzas). Typically, the input biomass accumulation rate was calculated from stock assessment model timeseries output as (B_{1986}/B_{1985})-1, where B is the total biomass (mid-year or mean) over all ages. For Atlantic menhaden, the biomass accumulation rate was calculated based on age 1+ biomass only. For bluefish, the biomass accumulation rate was reduced by half from -0.128 to -0.064 to balance the model. Atlantic herring input the biomass accumulation rate was calculated as the 3-yr mean biomass accumulation rate (average over 1984-1986) and reduced by half from 0.275 to 0.137 to provide better estimates of biomass and fishing mortality for the younger age stanza.

Mortality

Mortality rates in Ecopath are entered as annual total instantaneous mortality, Z, where $Z=F+M$. Age-specific M was available from the stock assessments as a function of body size using the Lorenzen equations (Lorenzen 1996) and scaled so that the mean M for fully selected ages equals a target M based on longevity (Hoenig 1983). For multistanza groups, the general approach to estimating natural mortality for each age stanza was to take the average M over all ages in each stanza weighted by the 1985 mean (or mid-year) numbers-at-age (Table 30).

$$M_s = \frac{\sum(M_a \cdot N_a)}{\sum N_a} \tag{13.2}$$

Here, M_s is the natural mortality rate for the Ecopath age stanza s and the summations are over all ages a included in stanza s. In the case of Atlantic herring, the most recent assessment used a constant M and so the age-varying M vector was taken from the previous stock assessment that used the Lorenzen estimator. Spiny dogfish and anchovy Ms were taken directly from the full NWACS model and the M (or production to biomass ratio P/B) of the invertebrate and zooplankton groups were taken as the average P/B of the inclusive groups from the full NWACS model, weighted by the biomass of those groups. Lastly, the P/B ratio for phytoplankton was taken directly from the full NWACS model.

For harvested groups (Atlantic menhaden, striped bass, bluefish, weakfish, spiny dogfish, and Atlantic herring), *F* was calculated from stock assessment output as the sum of landings for each stanza divided by the average (or mid-year) biomass of each stanza. These *F* rates were added to numbers weighted mean *M* to obtain the input *Z* values. For species without landings (anchovy, benthos, zooplankton), the input *Z* was equal to *M*.

Diet Composition

In Ecopath, a diet matrix is required that describes the proportion of each prey *i* in the diet of predator *j*, DC_{ij}. The diet matrix of the full NWACS model was simplified for the MICE model by first summing the DC_{ij} across NWACS-MICE prey groups and then averaging across NWACS-MICE predators, weighted by total consumption (B*Q/B) of each predator. Any DC_{ij} for a prey type not included in the MICE model was assigned to diet import (Table 31). Diet import provides a convenient workaround to modeling all the prey items of every species. Essentially, it allows for some proportion of the diet to be obtained from outside the modeled system and this part of their consumption is held constant over time in Ecosim. For example, striped bass age 6+ have a diet import of 0.269, meaning that 26.9% of their consumption comes from groups not included in the model. In Ecosim, that proportion of their total consumption will remain constant over time, i.e. they will always be able to achieve 26.9% of their base food intake.

Consumption Rates

Consumption rates, Q/B, are input for all consumer groups (Table 30); for multi-stanza species it is entered for the leading stanza only and calculated for other stanzas based on input biomass, mortality, and growth parameters. In all cases, Q/B was taken directly from the full NWACS model. For aggregate groups (inverts and zooplankton) the Q/B was taken as the weighted average Q/B for inclusive groups from the full NWACS model weighted by the biomass of each group.

Unassimilated Food

The unassimilated food parameter, *U*, represents the proportion of consumption that is not assimilated into biomass and therefore becomes part of the detrital pool. The *U* values were obtained from the full NWACS model, which were left at the recommended defaults for fish (0.2), benthic invertebrates (0.5), and zooplankton (0.3).

Landings

Landings were included for striped bass, Atlantic menhaden, spiny dogfish, bluefish, weakfish, and Atlantic herring (Table 30). Landings were derived from stock assessment outputs by summing the landings-at-age across fleets and then summing across ages for each stanza.

13.2.2 Balancing

The process of adjusting parameters in an Ecopath model to ensure mass balance is known as "balancing". One of the key diagnostics is that all groups should have EE values < 1.

13.2.3 Ecopath Outputs

Mortality Rates

Fishing mortality, *F*, in Ecopath is calculated simply as input landings divided by input biomass. Thus, the fishing mortality rates should match exactly those from the stock assessment, albeit converted to $F=C/\overline{B}$ by summing landings over ages and dividing by mid-year or average annual biomass. However, because the biomass of younger (non-leading) age stanzas is calculated in Ecopath assuming a stable age distribution, it is often not possible to obtain the exact *F* from a given year in the age-structured assessment models. Adjusting the biomass accumulation rate parameter in Ecopath allows for a better approximation of the age specific biomass and therefore *F* in the stock assessments, but some divergence is still expected for non-leading stanzas. Fishing mortalities for the Ecopath base year 1985 are provided in Table 32. Fishing mortality rates on fully selected age stanzas were 0.171 for striped bass, 0.193 for Atlantic menhaden, 0.019 for spiny dogfish, 0.148 for bluefish, 0.222 for weakfish, and 0.395 for Atlantic herring.

Predation mortality, M_2, in Ecopath is calculated as the total consumption of prey *i* by predator *j* divided by biomass of the prey.

$$M_{2i} = \frac{\sum B_j QB_j DC_{ij}}{B_i} \tag{13.3}$$

Total predation mortality for Atlantic menhaden in Ecopath (1985 base year) was 0.121 for juveniles and 0.031 for adults (Table 32). These low M_2 rates are due to low predator biomass, exclusion of other predators from the model, modest contributions to predator diets, and high Atlantic menhaden biomass. The result is that a large portion of the total mortality of Atlantic menhaden in the MICE model is unexplained (Figure 129), i.e. not attributable to fishing or predation. This is important because the top-down predation effects on Atlantic menhaden are expected to be muted under such configuration unless predator biomass increases drastically. The partitioning of Atlantic menhaden mortality in the MICE model should be contrasted with that in the full NWACS model (Section 14), which includes a broader suite of predators.

Predation mortality for the other forage group of interest, Atlantic herring, was higher than Atlantic menhaden, with M_2=0.895 for juveniles and M_2=0.377 for adults. Even though Atlantic herring contribute to a smaller portion of the predator diets compared to Atlantic menhaden, their predation mortality rates are higher because biomass is an order of magnitude lower than Atlantic menhaden.

Predation mortality rates were low (<0.002) for the adult age stanzas of predator groups (striped bass, spiny dogfish, bluefish), which is expected for larger individuals but is also due to the exclusion of any potential predators of large bodied fish (sharks, dolphins, larger fish) from the model. Weakfish, which function in the model as both a predator and a prey, had a slightly higher predation mortality in the adult stanza (M_2=0.08) than the aforementioned groups. Predation mortality on juvenile stanzas was generally higher than adults, with juvenile bluefish and weakfish having a high M_2, 1.6 and 1.3 respectively. Predation on striped bass juveniles is poorly explained by the model and represents only about 10% of the total mortality, with virtually no predation on the sub-adults.

Bluefish, spiny dogfish, and striped bass accounted for most of the predation mortality in the Ecopath model (Table 33, Figure 130). In fact, bluefish accounted for the largest percentage of predation mortality on Atlantic menhaden, juvenile bluefish, and weakfish. Predation mortality on Atlantic herring was highest for spiny dogfish, followed by bluefish. Striped bass contributed to at least 20% of the predation mortality on juvenile striped bass, Atlantic menhaden, and juvenile weakfish.

Mixed Trophic Impacts
Mixed trophic impact analysis provides a method to assess the direct and indirect effect that changes in biomass of a group will have on biomass of other groups in the system (Ulanowicz and Puccia 1990). The mixed trophic impact is calculated in Ecopath using a standard matrix inversion routine and shows the net effect that a very small increase in biomass of one group has on other groups, through direct and indirect interactions, in a steady-state system. If diet compositions change over time with predator-prey abundances, the interactions that contribute to mixed trophic impact will also change, and so this analysis should not be used for prediction but rather as a form of sensitivity analysis to identify groups that are expected to have quantitative impacts in the model. The mixed trophic impact should not be interpreted in an absolute sense but are relative and can be compared across groups.

The mixed trophic impact of the NWACS-MICE model illustrates that increases in Atlantic menhaden biomass are expected to have positive effects on striped bass, and to a lesser extent bluefish and weakfish (Figure 131). Relatively speaking, the impact of increasing Atlantic menhaden biomass is more positive than that of Atlantic herring for these predators. Conversely, increases in predator biomass are expected to have negative effects on most species, with bluefish having negative impacts on almost all other species. A counterintuitive result is that increasing striped bass age 6+ is estimated to have a net positive effect on juvenile Atlantic menhaden and juvenile weakfish.

13.3 Ecosim Model Description

The NWACS-MICE Ecosim model was calibrated to time series of observed abundance and catch from 1985-2017 using fishing mortality as a forcing function. The general strategy was to fit several Ecosim models under alternative assumptions about prey switching, feeding time adjustment rates, and upper and lower limits to the vulnerability parameters. After a fitted model was obtained, a series of forward projection scenarios (40 years) were conducted to screen single species reference points for Atlantic menhaden, evaluate tradeoffs between

Atlantic menhaden and striped bass, and develop ecological reference points for Atlantic menhaden.

13.3.1 Treatment of Indices & Time Series Data

Indices of Relative Abundance

A total of 18 indices of abundance were used to calibrate the NWACS-MICE Ecosim model (Table 34). These indices were recommended by each species' respective ASMFC TCs as the most representative and were obtained directly from the stock assessment output files, except for spiny dogfish, which was obtained from the assessment report (NEFSC 2018b). The selected indices were derived from fisheries independent surveys and recreational catch rates. Some species included more than one index and most indices spanned the entire simulation period. Time series weights were derived from the year-specific CV for each survey, which were already available in the stock assessments. The time series weights were calculated as the inverse of the mean CV over all available years (i.e. $1/\overline{cv}$), such that more precise data streams have higher weights and thus more influence on model fit.

Catch Time Series

Catch time series were assembled from the stock assessment report files as the landings in weight, summed over all gears and age classes for each stanza. In most cases, annual CVs for landings were available from the stock assessment and the combined CV for all years and fleets was calculated as the average of all CV, weighted by the landings. The time series weight in Ecosim was taken as the inverse of the combined landings CV, which generally resulted in higher weights than the abundance data. Due to the scaling issues associated with the stable age calculations in multi-stanza groups, the catch time series for juvenile stanzas of Atlantic menhaden, bluefish, and Atlantic herring were treated as relative catch and were scaled (internally by Ecosim) to the Ecopath base landings. Spiny dogfish landings were used as a forcing time series because *F* was unavailable for that species.

Fishing Mortality Time Series

Fishing mortality was used a forcing time series in Ecosim for all harvested species except spiny dogfish, which used catch forcing instead. Fishing mortality time series were derived from the stock assessment as $F_y = C_y/\bar{B}_y$, where C_y is the total landings summed over ages and gears for each species/stanza and $\bar{B}_y$ is the mean (or mid year) biomass for each species/stanza. In Ecosim, it is important that *F* in the first year of the time series is equal to the Ecopath base *F*. As mentioned above, however, this is not always possible for younger ages of multi-stanza groups whose biomass (and therefore *F*) is calculated based on stable age assumption and differs from that in the stock assessment used to derive *F*.

13.3.2 Ecosim Calibration Procedure

General Overview of Fitting Ecosim Models

When fitting Ecosim models, there are three broad types of parameters to consider, state variables, flows, and forcing functions. State variables, e.g. biomasses, are the components that this assessment is predominantly interested in; they are solved for as time derivatives in Ecosim. The Ecopath diet matrix describes the initial flow of energy between state variables, and in Ecosim the flows are expressed using foraging arena theory equations. Forcing functions in Ecosim are external factors that drive the system, such as environmental drivers, fishing effort, or fishing mortality. Forcing functions are calculated external to the model and imposed as a time series.

The Ecopath model represents the initial state for time-dynamic simulations. Initial state parameters (biomass, mortality, consumption, diet, landings) are input to Ecopath by the user and not estimable in Ecosim. Thus, to evaluate the effect of initial state parameters on model fit one must manually adjust the input values or use Ecosim Monte Carlo simulations and provide uncertainty around the input parameters. The Monte Carlo routine will, optionally, save the parameters that improve fits to time series, but it is constrained by mass balance (each Monte Carlo trial is evaluated for mass balance and discarded if not) and does not include a minimization search and so is computationally inefficient for fitting models. It is recommended to thoroughly evaluate the pre-balance diagnostics (Link 2010) in Ecopath before going to Ecosim and then only adjust the initial inputs sparingly and on a case-by-case basis (as a last resort) to improve model stability and fit.

In Ecosim, there are two sets of parameters that describe the consumption model according to foraging arena equations. The first set of Ecosim parameters are the vulnerability exchange rates, V_{ij}. These regulate consumption, and therefore regulate biomass gains. Consumption for a predator is mortality for its prey, and so the V_{ij} also serve as limits on predation mortality at high predator biomass. Ecosim models are sensitive to the V_{ij} values. The V_{ij} can be estimated in Ecosim using the fit to time series interface to reduce the sum of squares differences between predicted and observed time series of biomass and catch. These are the only parameters estimated by Ecosim to minimize a goodness-of-fit measure.

The second set of parameters are found on the group info tab, and these include maximum P/B, foraging time adjustments (FTA), predator effect on foraging times, and prey switching. The FTA parameters are important for allowing compensatory improvements in survival at low stock sizes by allowing groups to spend less time feeding at low densities and thus be exposed to less predation. Prey switching is said to occur when predator diet proportions change more rapidly (or slowly) than relative abundances. Prey switching can occur in two ways in Ecosim. First, predators will switch from prey that are declining in abundance, due to density dependent foraging time of prey [FTA>0], which is implied in NWACS-MICE for all juvenile stanzas. Second, predators may explicitly switch between prey types by modifying the rate of effective search (a_{ij}) in relation to changes in abundance of prey using a power function $a_{ijt} = a_{ij} \cdot B_i^{P_j} \cdot K_{ij}$, where Pj = [0,2], and K_{ij} is a scaling constant.

The group info parameters are set by the user and not estimable in Ecosim fit to time series. A recommended configuration is to set FTA=0.5 for the youngest age of multi-stanza groups and FTA=0 for all other groups, with prey switching turned off (=0). To evaluate the effect of alternative values for FTA or prey switching, one must manually adjust the parameter and run the model. However, the vulnerability parameters are dependent on the group info configuration and so it is advised (but not required) to re-estimate the vulnerabilities each time one of these values is changed (see scenarios below).

Lastly, Ecosim forcing functions are input as either time series multipliers or absolute values. For environmental forcing functions (e.g. chlorophyll, temperature), they are typically applied as mean-scaled multipliers on baseline PB parameters (in the case of primary producers), or as multipliers on V_{ij}, search rate, or arena size parameters – thus allowing a forcing function to modify the predator-prey functional response. Most commonly, Ecosim forcing functions are applied to simulate changes in nutrient loading, chlorophyll production, and fishing mortality. In the case of environmental forcing, variables such as temperature or salinity must be accompanied with a habitat preference function for affected groups. Additionally, fishing mortality may be forced by including a time series of species-specific *F* values, a time series of fleet-specific fishing effort that functions as a multiplier on the Ecopath *F* for each fleet and species, or by forcing removals (i.e. forced catch).

Estimating Vulnerability Parameters

When fitting an Ecosim model, it is important to first determine the appropriate number of vulnerabilities to estimate. As a conservative approach, it has been recommended to only estimate *K-1* parameters (Heymans et al. 2016), where *K* is the number of reference time series (i.e. observed biomass and catch) used to tune the model. Alternatively, estimating fewer parameters may lead to a better model based on AIC criteria and this can be tested by estimating different numbers of parameters in a stepwise fashion (Scott et al. 2016).

Fitting an Ecosim model begins by first identifying the most sensitive V_{ij} parameters and then estimating those parameters to improve the model's goodness-of-fit as assessed by the sum of squares of predicted biomass and catch from observed time series. The sensitivity search proceeds by adjusting each vulnerability slightly, one at a time, to see how much the sum of squares changed. The *K-1* most sensitive vulnerabilities are then selected, i.e. "turned on" for parameter estimation. In the NWACS-MICE model, no more than 27 (*K*-1) vulnerability parameters were estimated during a single tuning iteration.

Ecosim models are prone to local minima in SS, thus requiring repeated vulnerability searches in order to find model convergence. Therefore, a methodology was implemented where the sensitivity and estimation routine was repeated until no further improvement in the sum of squares and AIC was obtained. This was done by searching for and estimating the most sensitive 27 V_{ij}, keeping those estimated values, and then searching for and estimating another set of 27 V_{ij}, and so on until the sum of squares and AIC have stabilized. At each iteration, the model may identify and estimate a different set of 27 V_{ij}, such that the total number of V_{ij} estimated is greater than 27. This approach is analogous to estimating parameters in phases –

whereby the most important parameters are estimated first and when those are in an appropriate parameter space, additional ones are turned on. Typically, convergence on a solution would be obtained after 5-7 iterations (Figure 132) and approximately 70-80 V_{ij} would have been estimated. This amounts to between 2-3 estimated parameters per time series, and around 75% of all possible V_{ij}. The vulnerabilities were reset to their default value of 2 and the repeated search was initiated after any changes were made to Ecopath inputs, group info, or forcing functions.

Vulnerability Bounds

The most sensitive parameters in Ecosim models are the vulnerability parameters, V_{ij}, which control the amount of prey biomass available for consumption. They are input in Ecosim as multipliers on Ecopath base predation mortality rates (M_{2ij}) to represent the maximum possible predation mortality rate ($M_{2\text{MAX}}$) that can be exerted on a prey item at high predator biomasses. For this reason, the V_{ij} parameters are also referred to as "predation rate limits".

It is often the case that Ecosim will estimate extremely high values of V_{ij} ($1\text{x}10^9$) in the fitting process, which may result in theoretical predation rates far above the prey's Z when predator biomass is high. While this may improve the sum of squares measure-of-fit over the period of observed data, the high V_{ij} could lead to dynamic instability, exaggerated top-down effects, and groups crashing entirely under extreme fishing scenarios. To correct for this, vulnerability caps were applied after the repeated search was completed. $M_{2\text{MAX}}$, and therefore the V_{ij}, can be expressed as some proportion of the prey's M, such that $V_{ij} = (M_{2\text{MAX}} * M)/M_{2\text{BASE}}$. For example, an $M_{2\text{MAX}}$ of $0.5*M$ means that a single predator will not account for more than 50% of the natural mortality of that single prey. Through an iterative approach, it was found that values of $M_{2\text{MAX}}$ around $0.75*M$ to $1.0*M$ provided the best fit to the data (i.e. compared sum of squares across scenarios where $M_{2\text{MAX}}$ varied from $0.25M$ to $2M$ for all V_{ij}). Additionally, V_{ij} estimated at the lower bound of 1.0 can be problematic in projections scenarios and often causes species to be unresponsive to fishing. Small increases (going from 1.0 to 1.1) can have noticeable effects in projections scenarios that apply high F rates (see Section 13.4.3: Equilibrium MSY).

Applying the vulnerability caps will increase the sum of squares and result in a poorer fit to the data by that measure of fit. However, the vulnerability caps may provide a model with better dynamics in the projection scenarios and more comparable productivity patterns relative to the stock assessments. Additionally, a search procedure that included penalized bounds on the V_{ij} might result in a lower sum of squares with values not on the bound. The decision of what constitutes the "best fit" model should not be based solely on the sum of squares measure of fit but rather the full suite of diagnostics including MSY curves and stock-recruit plots.

13.3.3 MICE Model Simulations

Over 30 different Ecosim configurations were fit during development and testing of the NWACS-MICE model. Those scenarios evaluated model fit under different inputs for foraging time adjustments, prey switching, vulnerability caps, primary production anomalies, and recruitment deviations. Not all of the 30 scenarios are presented in this report; instead, seven alternative scenarios that represent the best fit and most parsimonious model configurations are summarized (Table 35). Only the first four scenarios in Table 35 were intended to be used for management purposes (default configuration and prey switching), whereas the last three (primary production anomaly and recruitment deviations) served as robustness tests during model development and parameter estimation. The last three scenarios are listed here only as a record of their existence in case future model iterations might wish to include primary production drivers or recruitment anomalies. Further work is needed to a) compare and validate the PP anomalies against actual changes in primary production observed through coastal and ocean monitoring systems (e.g. satellites, river gauges); and b) properly adjust for M_0 forcing effect when EE is high (and M_0).

The prey switching scenario sim3.5 is the preferred base run to be used for development of ERPs. While sim3 had a lower sum of squares than the other three, sim3.5 was preferred because the vulnerabilities estimated at the upper bound (1e10) are replaced with the vulnerability caps ($M_{2MAX}=M$) and those estimated at the lower bound (1.0) were replaced with values ranging from 1.02 to 1.5 to remove dynamic instability in projections with high F rates.

Baseline configuration (sim1 and sim1.1)
The baseline configuration (sim1) has FTA set to 0.5 for all of the youngest age stanzas and 0 for all others, with no prey switching (P_j=0). An alternative baseline run (sim1.1) applies the upper vulnerability cap of ($M_{2MAX}=M$).

Prey Switching (sim3 and sim3.5)
The model was fit under prey switching P_j values of 0, 0.5, 1, and 1.5 applied to all Atlantic menhaden predators. Of the values considered, P_j=1 (sim3) resulted in the lowest sum of squares at the end of the repeated search. An alternative prey switching model (sim3.5) was tested that includes the upper and lower vulnerability caps ($M_{2MAX}=M$) as well as changes to foraging time adjustments for striped bass so that F_{TARGET} projections were more comparable to the stock assessments.

Primary Production Anomalies (sim9 and sim9.1)
Ecosim can be invoked to search for time series values of annual relative primary productivity in order to further improve the fits to observed data. The underlying assumption is that primary production is variable over time and causes changes in relative abundance throughout the food web. The NWACS-MICE model was fit with primary production anomalies estimated using a 3-year smoothing spline function (sim9) and with annual primary production anomaly estimates (sim9.1). The primary production anomaly scenarios should not be considered for base run or management advice because the estimated historical primary production pattern may be a spurious trend with no relation to known primary production patterns. Rather, these scenarios were produced to examine whether management advice generated by the model is robust if bottom-up drivers are explicitly included.

Recruitment Deviations (sim12.3)

Assuming that recruitment deviations estimated in the stock assessment models can be interpreted as years of good or poor survival, those deviations can be included as a forcing function on mortality of juvenile age stanzas in Ecosim. Doing so allows the model to represent year class variability over time, but makes no inference as to the mechanism. Recruitment deviations were available for all assessed species as log transformed deviations. These were converted to an index of relative mortality as the inverse of the back transformed log deviation, scaled to a mean of 1 and applied as multipliers on M_0 for all juvenile stanzas. The vulnerabilities were again fit following the repeated search procedure described above. One caveat when using M_0 forcing is that groups with high EE (and low M_0) might not respond to very large multiples of M_0 because it is still only affecting a small portion of Z. In extreme cases (EE>0.8), the M_0 forcing function may need to be rescaled to obtain the desired response in Z. The last two scenarios are those fitted with recruitment deviations applied as M_0 forcing on juvenile stanzas. Recruitment deviation models with prey switching P_j values of 0, 0.5, 1, and 1.5 applied to all Atlantic menhaden predators were fit. Of the values considered, P_j=1.5 (sim12.3) resulted in the lowest sum of squares at the end of the repeated search.

13.4 Ecosim Outputs

13.4.1 Fits to time series

The NWACS-MICE Ecosim model produced reasonably good fits to the relative abundance time series (Figure 133 and Figure 134), with the exception of juvenile Atlantic menhaden and weakfish. The inability to fit to juvenile Atlantic menhaden and weakfish might be explained by the absence of bottom-up drivers in the model that would describe the decline in abundance of juvenile Atlantic menhaden and the increase in weakfish abundance during the mid-1990s. The NWACS-MICE Ecosim model fit the catch trends very well (Figure 135), for all species except juvenile Atlantic menhaden, which is essentially a scaling issue associated with multistanza calculations of juvenile biomass under stable age distribution assumption.

13.4.2 Emergent Stock Recruit relationships

Ecosim models do not include an explicit stock-recruit equation, rather stock-recruit relationships are an emergent property of Ecosim models with multi-stanza age groups (Walters and Martell 2004). A Beverton-Holt or Ricker type stock-recruit curve is generated when the juveniles have non-zero feeding time adjustment, combined with high EE and/or high proportion of other mortality sensitive to feeding time (set to 1 in all scenarios). This represents density-dependent changes in juvenile mortality rate associated with changes in feeding time and predation risk. The shape of the stock-recruit curve is determined by the degree of compensatory increase in juvenile survival at low densities. Compensatory effects are increased (i.e. higher steepness and constant recruitment across broad range of spawning stock size) by setting the V_{ij} of juvenile prey items close to 1.

The stock-recruitment relationship for Atlantic menhaden was revealed by simulating a severe increase and decrease in fishing mortality so as to generate the paired adult and juvenile abundances across a wide range of stock sizes. Stock-recruit curves were generated for each scenario after all fitting and V_{ij} adjustments were complete (Figure 136). The Ecosim stock-recruitment curves tend to show a positive relationship between recruits and adults at low stock sizes, which begins to level off at high stock sizes. In particular, the scenarios with primary production anomaly (sim9 and 9.1) exhibit more of a Beverton-Holt, or possibly a Ricker, curve. Other scenarios showed low compensatory response in juvenile survival and exhibited more of a straight line out of the origin. This figure is provided to make clear that a stock recruit relationship does exist in Ecosim models. Further research should be conducted to understand how density-dependent processes combined with bottom-up drivers affect our estimates of stock-recruit relationships in both multi species and single species models.

13.4.3 Equilibrium MSY

Estimates of equilibrium MSY and F_{MSY} are obtained by running long term Ecosim simulations over a range of *F* or effort values. Each species is analyzed separately and there are two options when invoking the Equilibrium MSY search in Ecosim, stationary and dynamic (Walters et al. 2005). In the stationary analysis, all predators and prey of the species being evaluated are held constant at their Ecopath inputs and do not respond dynamically to changes in the target species. In the non-stationary, dynamic simulations, predators and prey are allowed to respond to changes in abundance of the target species. This sometimes leads to compensatory responses in the target species that might, for example, allow for maximum yield at higher *F* of forage species when predators respond negatively or switch to other prey.

The equilibrium MSY analysis revealed a dynamic instability in sim3 that was associated with vulnerability parameters estimated on the lower bound of 1.0 (Figure 137). Small increases to those values were made in sim3.5 and this instability was removed. This is a primary justification for choosing sim3.5 as the preferred run over sim3, even though it had a higher SS.

Striped bass F_{MSY} was estimated in all four scenarios and ranged between 0.154-0.171 (excluding sim3) (Figure 137, Table 36). Atlantic menhaden F_{MSY} was approximately 0.65 in the scenarios without prey switching and 0.954 and 0.837 for sims 3 and 3.5 respectively. The higher F_{MSY} with prey switching is obtained because predators will quickly switch away from Atlantic menhaden when they are declining allowing for compensatory reductions in M_2. With the exception of sim3, bluefish F_{MSY} was estimated between 0.72-0.86. Weakfish F_{MSY} estimates are unreliable, but estimated at 0.8 for sim 3.5. Lastly, there was good agreement in F_{MSY} of Atlantic herring, with values ranging between 0.24 and 0.4.

13.5 Projections

13.5.1 Single-species proxy reference points

The biomass and fishing mortality reference points from the stock assessment are defined in a variety of ways with different metrics and currencies, making it impossible to apply those values directly in Ecosim. Therefore, a ratio approach was used to calculate proxy reference points that can be applied and evaluated in Ecosim. This was done by multiplying the single species ratios B_{ref}/B_{2017} or F_{ref}/F_{2017} by the corresponding Ecosim predicted B_{2017} and F_{2017} (forcing). Single species reference points and their Ecosim proxies are provided in Table 37.

13.5.2 *F* target and *F* threshold scenarios

For species with defined target and threshold fishing mortality rates, long-term projections under each *F* reference point were conducted, while holding all other species constant at their 2017 *F*. This was done to test if Ecosim can replicate similar dynamics to the stock assessment with regards to how species respond to changes in fishing pressure. Because a ratio approach based off 2017 values was used to convert single species reference points to Ecosim, biomass from each scenario was scaled to its own predicted 2017 estimate.

For striped bass, the projected biomass for sim1 was far below its associated targets and thresholds, whereas sims 1.1, 3, and 3.5 all approximated the biomass target and threshold; sim3.5 showed a higher biomass under the threshold scenario (Figure 138).

Target and threshold fishing mortality rates were evaluated for Atlantic menhaden, but no biomass reference points were available. The Atlantic menhaden projections under F_{TARGET} were all similar, except for sim3, which was slightly lower with some dynamic instability (Figure 139). The scenarios with prey switching (sims 3 and 3.5) predicted higher biomass under $F_{\text{THRESHOLD}}$ than the non-switching scenarios (sims 1 and 1.1). This is because predators will quickly switch away from Atlantic menhaden to other prey as Atlantic menhaden are declining resulting in less predation mortality.

Bluefish target and threshold projections were similar across all scenarios with the exception of sim3 (Figure 140). Sim3 was the only scenario that reached the biomass target, with all others remaining below the target but above the threshold. By testing additional configurations, it was determined that there is tension in the model between bluefish and striped bass, such that no configuration could be found that allowed both of them to reach their biomass targets simultaneously in these scenarios. Also, there is disconnect between the bluefish *F* and *B* reference points, where the $B_{THRESHOLD}$ is half of the target, but the $F_{\text{THRESHOLD}}$ is only 10% higher than the F_{TARGET}. Therefore, the target and threshold scenarios for bluefish are very similar.

13.5.3 Screening BAM *F* reference points

Short-term projection scenarios (2018-2021) were conducted using the BAM under three scenarios: 1) harvest each year is equal to the current TAC of 216,000 mt; 2) harvest is set equal to a level that has a 50% probability of reaching the single-species *F* target; and 3) harvest is set equal to a level that has a 50% probability of reaching the single-species *F* threshold. The projections were run using the BAM Monte Carlo bootstrap routine to capture the uncertainty associated with *M* and fecundity. For each scenario, a total of 4,864 *F* vectors were provided from BAM representing a distribution of *F* values to be evaluated in Ecosim.

The fishing mortality rates coming out of the BAM projections are equal to the full *F* used in the assessment, and based on population size in numbers. To apply proxy scenarios in Ecosim, the ratio of F_y/F_{2017} for projection years was multiplied by the terminal year F_{2017} in Ecosim (equal to 0.048 for adult Atlantic menhaden, as $C/\overline{\mathrm{B}}$). For each scenario, 500 BAM trials were selected at random and converted to Ecosim *F* input files (one file for each trial) using the multisim plugin to automate the simulations. In the Ecosim projections, all other species were held constant at their 2017 status quo *F* rates. Long term (40 year) projections were run by extending the BAM scenarios to 2057 using the mean *F* from the BAM projection years (2018-2021). Biomass trajectories from these scenarios are shown in Figure 141.

Results of this analysis are summarized for each BAM *F* scenario as the proportion of *F* trials that caused each predator to change by *X*%, where *X* ranged from -50% to 50% in 5% increments. Change in biomass was calculated relative to the status quo scenario (ΔB_{REL}), where $\Delta B_{REL} = (B_{TRIAL}-B_{2017})/B_{2017}$, and *B* for each trial and the status quo is equal to the biomass after 4 or 40 years. Additionally, the median ΔB_{REL} is provided. The analysis provides information on the level of risk of predator declines associated with single-species Atlantic menhaden reference points developed by the BAM. Small changes of less than 10% are deemed to be low risk and within the bounds of measurement uncertainty.

The Ecosim model predicted that harvesting Atlantic menhaden at the current TAC of 216,000 mt is not expected to cause any predators to decline by more than 10% over the short and long term (Table 38, Figure 142 - Figure 143). After 40 years of fishing at the current TAC, striped bass biomass was predicted to decline by 5-10% in 16% of trials (Table 38, Figure 143).

The F_{TARGET} scenario represents an increase in Atlantic menhaden fishing mortality from F_{2017}. In this scenario, the proportion of trials leading to declines in predator biomass increased slightly compared to the TAC scenario (0% column in Table 38). No predators were predicted to decline by 10% over the short term (four years) (Table 38, Figure 142). Over the long-term (40 years), striped bass was still the only predator with predicted negative effects with 10% of trials predicting biomass declines of 10-15%.

Impacts on other predators begin to be observed when Atlantic menhaden are fished at $F_{THRESHOLD}$. In this scenario, nearly all trials led to at least some decline in biomass for all species relative to status quo; however, striped bass was the only predator to exhibit declines greater than 10% over the short term. Under the Atlantic menhaden $F_{THRESHOLD}$ scenarios, striped bass biomass was reduced in the short term by at least 10-15% in 58% of the trials and biomass was reduced by 15%-20% in 12% of the trials (Table 38, Figure 142). Over the long term, striped bass biomass was predicted to decline by 10-15% in 90% of trials, by 15-20% in 75% of the trials, and by 20% or more in 56% of trials (Table 38, Figure 143). In other words, there is a greater than 56% probability that fishing Atlantic menhaden at their *F* threshold will cause striped bass biomass to decline by at least 20%.

This analysis indicates that the current TAC and F_{TARGET} scenarios developed by the BAM are not likely to cause negative effects on predators (biomass declines of greater than 10%) over the short term (4 years). Over the long term, and assuming striped bass are fished at status quo F, the Atlantic menhaden F_{TARGET} scenario showed a low probability (0.10) that striped bass would decline by 10% or more. While the median change in biomass relative to 2017 for all species in the first two scenarios are either zero or negative, they do not fall below -10%. It is not until Atlantic menhaden are fished to their biomass threshold that more substantial impacts on predators are predicted by Ecosim, with the most severe impacts on striped bass. Even in this scenario, the predicted median decline for striped bass is -11% after 4 years and -21% after 40 years. This information is intended to gauge the level of risk associated with single species reference points developed by BAM.

13.5.4 Predator-prey surface plots

Species in Ecosim are connected to one another through food web interactions, such that the predicted biomass of any given species is a function of its own fishing mortality rate as well as that of its predators and prey. Specifically, the biomass of striped bass is a function of striped bass F and Atlantic menhaden F. To elucidate this relationship, series of simulations was run under different combinations of F for striped bass and Atlantic menhaden (all other species held constant at 2017 status quo). In these simulations, striped bass F rates ranged from 0 to 2 times F_{2017} and Atlantic menhaden F rates ranged from 0 to 10 times the current F_{2017}. For striped bass, which has two harvested age stanzas, the F multipliers were applied to each stanza (i.e. an F multiplier of 0.5 would be a 50% reduction in 2017 F for all harvested stanzas).

For each simulation, a biomass ratio for striped bass was calculated as age 6+ biomass in the terminal year divided by the target age 6+ biomass (1.58 x predicted 2017 age 6+ biomass). Thus, B ratios < 1 are below the target, B ratios between 0.75 and 1 are above the threshold and below the target, and B ratios > 1 are above the target. Similarly, the biomass of bluefish was predicted as a function of striped bass and Atlantic menhaden F. For bluefish, the biomass target and threshold were calculated as 2.36 times the current biomass, and for weakfish the terminal year biomass was expressed relative to the threshold biomass that is 3.58 times higher than current biomass B_{2017} predicted by Ecosim.

It is important to note that current striped bass F is above the F threshold and that biomass is below the biomass threshold. The analysis shows that at current striped bass F (where F multiplier = 1 on the y-axis of Figure 144), the stock will remain below the threshold regardless of Atlantic menhaden F rates. This indicates that striped bass fishing mortality is currently above that which would achieve biomass target and any efforts to improve stock status should be focused on reducing F on striped bass.

The estimated striped bass target F from the striped bass stock assessment is about 35% lower than current F. At striped bass target F (F multiplier ≈0.65), striped bass biomass would reach the target under current Atlantic menhaden F rates. Striped bass biomass would remain above the threshold over Atlantic menhaden F rates ranging from zero to approximately 4 times F_{2017} (Figure 144), i.e. if striped bass were fished at F_{TARGET}, Atlantic menhaden harvest could be increased from the 2017 rate by up to 4 fold and striped bass biomass would remain above the threshold but below the target.

The harvest of Atlantic menhaden and striped bass is likely to have effects on other species such as bluefish and weakfish that are preyed upon by striped bass and/or compete with them for prey. Bluefish was below the target across all Atlantic menhaden and striped bass *F* combinations, suggesting (as with striped bass) that bluefish *F* needs to be reduced in order to reach their target (Figure 145). Higher *F* rates on striped bass led to higher biomass of bluefish as a result of reduced predation and competition (striped bass prey on juvenile bluefish and also have diet overlap with bluefish). With striped bass fished out of the system, Atlantic menhaden harvest has very little effect on bluefish biomass. In contrast, when striped bass *F* is reduced, biomass of bluefish declines, with the lowest biomass predicted in scenarios with both reduced striped bass *F* and high Atlantic menhaden *F*. This highlights an important prediction by the model – that effects of Atlantic menhaden harvest on predators are only likely to be observed when predator biomasses are high and there is more competition for food.

Similarly, weakfish biomass is lowest at low striped bass *F* (Figure 146). However, in the low striped bass *F* scenarios, weakfish biomass increases with higher Atlantic menhaden *F*. This peculiar result might indicate that the indirect effects (i.e. lower predation and competition) resulting from the impact of Atlantic menhaden harvest on striped bass biomass (Figure 144) are stronger than the direct effects of Atlantic menhaden harvest on weakfish. That is, when striped bass biomass is high, reducing it by way of increased Atlantic menhaden harvest will result in a net benefit to weakfish. On the other hand, when striped bass biomass is low (high *F* scenarios), increasing Atlantic menhaden harvest has a slight negative effect on weakfish (Figure 146).

13.5.5 NWACS-MICE Ecological Reference Points

Of all the modeled fish species, striped bass was the most responsive to changes in Atlantic menhaden *F*. As a result, striped bass were used as an indicator of the impacts of Atlantic menhaden fishing pressure on the ecosystem for the development of ecological reference points. This is supported by analysis from the full NWACS model that evaluated a broader suite of fish species and found that striped bass was the most sensitive Atlantic menhaden fish predator. The full NWACS model also predicted that piscivorous shorebirds were also sensitive to Atlantic menhaden harvest, and those impacts are not considered in the MICE model.

Analysis to develop ERP *F* target and *F* threshold was based on striped bass biomass responses to changes in Atlantic menhaden *F* while maintaining striped bass at F_{TARGET}. All other modeled species were kept constant at current *F* rates. The proposed ERP target (ERP F_{TARGET}) is the maximum Atlantic menhaden *F* that maintains striped bass biomass at their biomass target, when striped bass are fished at their *F* target. The proposed ERP threshold (ERP $F_{THRESHOLD}$) is the maximum Atlantic menhaden *F* that maintains striped bass biomass at their biomass threshold, when striped bass are fished at their *F* target. Here, the ERP target and threshold apply to Atlantic menhaden *F*.

Projections were run from 2018 to 2057 over a range of Atlantic menhaden *F*s while keeping striped bass fixed at F_{TARGET} (0.635*F_{2017}). Figure 147 shows the simulations over time and Figure 148 shows the terminal year striped bass biomass ratio against Atlantic menhaden *F*. Here, striped bass age 6+ biomass is treated as a proxy for SSB based reference points, since females mature between ages 4-8. Ecosim *F* rates were converted back to BAM units by multiplying the Ecosim *F* ratio (ERP F_{TARGET}/F_{2017}) by the current full *F* of 0.157 from BAM.

The Atlantic menhaden ERP *F* target is 0.06, a 20% increase from current Atlantic menhaden *F*. Conveniently, the BAM equivalent ERP *F* target of 0.188 is equal to current F_{TAC} scenarios from the BAM MC runs (averaged over all years and MC trials). The Atlantic menhaden ERP *F* threshold of 0.183 (=BAM full *F* of 0.573) is about 30% lower than the BAM *F* threshold.

As such, it can be concluded that 1) the proposed current Atlantic menhaden TAC is equal to the ERP *F* target for Atlantic menhaden and should maintain striped bass at target biomass when striped bass are fished at their *F* target; 2) fishing Atlantic menhaden at the proposed BAM *F* target will maintain striped bass above the threshold but below the target; and 3) the current BAM *F* threshold for Atlantic menhaden is too high to maintain striped bass at or above their biomass threshold.

This example was based on the *F* and *B* targets laid out in the striped bass fishery management plan. Higher or lower reference points for striped bass will result in higher or lower reference points for Atlantic menhaden. In addition, this example maintained the other species at their current *F* rates; higher or lower *F* rates on other species would also result in different reference point values for Atlantic menhaden. This equilibrium approach for developing ERPs assumed a constant environment during the projection years. Major changes to biomass of other predators and to bottom-up drivers could alter the productivity of Atlantic menhaden and result in different ERPs.

14 FULL ECOPATH WITH ECOSIM MODEL (NWACS-FULL) (SUPPORTING)

Buchheister et al. (2017a, 2017b) previously developed an NWACS ecosystem model using EwE to inform Atlantic menhaden management within an ecosystem context. This NWACS model simulated 61 trophic groups and eight fishing fleets, using data from 1982 to 2013 (Buchheister et al. 2017a, 2017b). For this assessment, the published model was used to derive a new, updated model (the NWACS-FULL model) to support the evaluation of Atlantic menhaden ERPs. An externally funded research project is currently underway to update time series of all available species in the model; however, the timing of this project did not coincide with the Atlantic menhaden ERP process as it will not be completed for another 1-2 years. As a result, a hybrid approach was developed, where only the time series for the six ERP focal species were updated (1982-2017) and incorporated into the NWACS-FULL model. The NWACS-FULL model provides a holistic ecosystem perspective, addressing the broader impacts of Atlantic menhaden fishing on the ecosystem and all of its predators, including birds, marine mammals, and other fishes not accounted for in the other ERP models.

14.1 Ecopath Model Description

14.1.1 Ecopath with Ecosim Modeling Framework

See Section 13.1 for a description of the EwE modeling framework.

14.1.2 The NWACS Ecosystem Model

Spatial Structure

See Section 13.2 for a description of the NWACS spatial structure.

Temporal Structure

The model was parameterized using available data for the ecosystem from 1982 to 2013. The initial year 1982 was chosen because this is the first year of available catch data for many of the single species stock assessments. All-time series for the ERP focal species (Atlantic menhaden, striped bass, bluefish, weakfish, spiny dogfish, and Atlantic herring) were updated to span 1982-2017 using stock assessment data (See NWACS-MICE Section 13.2.1). For all other groups, forcing time series were extended to 2017 using the values from 2013. A project is underway to update these groups with the best available data through 2017, but the timeline of that project did not align with the Atlantic menhaden stock assessment process.

Trophic Structure

The trophic structure of the model represents the principal groups in the ecosystem – from detritus and phytoplankton to marine mammals and seabirds – using 61 different groups (Table 39). Groups are aggregated taxa based on similar functional or taxonomic characteristics, with a higher degree of aggregation for lower trophic levels (e.g., phytoplankton, zooplankton, and benthic invertebrates) and highest trophic levels (e.g., sharks, marine mammals, seabirds). The degree of taxonomic resolution at lower and higher trophic levels largely followed the structure used for the Energy Modeling and Analysis eXercise (EMAX) models (Link et al. 2006, 2008). Given that the initial application of the NWACS model was for Atlantic menhaden, important Atlantic menhaden predators (e.g., striped bass, bluefish, weakfish) are represented as individual species, as are alternative prey for those predators (e.g., Atlantic herring, Atlantic mackerel, anchovies). Other fish species (e.g., Atlantic cod (*Gadus morhua)*, summer flounder, spiny dogfish) that are of particular management concern or ecological significance were also retained explicitly in the model.

Several fishes were partitioned into multiple age stanzas to account for documented ontogenetic differences in diets (e.g., Garrison and Link 2000; Smith and Link 2010; Buchheister and Latour 2015) or changes in habitat or migration behaviors. Stanzas were defined based on age, but associated length cutoffs were also assigned to allow length-based data to be partitioned appropriately among stanzas (e.g., trawl survey catches and diets based on predator length). Length cutoffs for each age were approximated using length-at-age relationships from scientific trawl surveys or from literature studies. For simplicity and consistency in naming of multi-stanza groups, stanzas were labeled as either small (S), medium (M), or large (L), but they represent different ages and lengths for each species (Table 39).

All groups were modeled using biomass densities (mt/km^2).

Fishing Fleets
Multiple fishing fleets were modelled to account for the dynamics of fishing operations in the region. Modelled fleets were defined as the predominant fishing gears used within the model domain, based on landings data from NOAA. The eight modeled fishing fleets were labeled as dredge, trawl, trap, gill net, purse seine, recreational, longline, and other.

14.1.3 Basic Inputs

The basic data requirements for Ecopath are biomass (*B*), production to biomass rate (*P/B*; equivalent to the total mortality rate, *Z*), consumption to biomass rate (*Q/B*), diet composition, and landings for each trophic group. Biomass accumulation rates (BA_i/B), which describe the instantaneous rate of change of a functional group's biomass to account for groups that are not in equilibrium, can also be provided. A summary of the general approaches and data sources used to parameterize the original NWACS model is provided below. Full details are available in the NWACS documentation (Buchheister et al. 2017b) and paper (Buchheister et al. 2017a). Additional detail on the parameterization for the ERP focal species updated in the NWACS-FULL model, can be found in Section 13.2.1.

The model was developed using several data sources, including fishery-independent surveys, single species stock assessments, primary and gray literature, and existing ecosystem models from the Northwest Atlantic shelf and its estuaries. The NWACS model also adopted many parameters from the EMAX project (Link et al. 2006, 2008). The EMAX project developed four Ecopath models for the Mid-Atlantic Bight, Southern New England, Georges Bank, and the Gulf of Maine; these parameters were used for many of the lower and higher trophic levels that did not have stock assessment data. The NWACS-FULL model has a greater taxonomic resolution for the fish groups (typically in the middle to high trophic levels) than the EMAX models. Stock assessment and fisheries independent survey data were used to parameterize these groups, when possible.

Biomass
When available, biomass estimates for fished groups were obtained from the most recent stock assessment for a given group. Data from multiple assessments were combined in cases where there were multiple stocks within the modeled domain (e.g., Gulf of Maine and Georges Bank Atlantic cod). In the situations with multiple stocks, absolute biomasses (in mt) were summed, whereas *P/B* and *Z* were calculated as biomass-weighted averages. In cases where a stock's distribution extends beyond the modelled domain (e.g., some species inhabit the South Atlantic Bight in addition to the Mid-Atlantic Bight), biomass was apportioned into the model domain based on regional catch or biomass proportions (if available). However, in most of these instances (e.g., Atlantic croaker), the entire stock biomass was used for the model because the contribution of the South Atlantic Bight catch (or biomass) was negligible and would not have a

substantial impact. All absolute biomasses were divided by the model area (246,662 km^2) to obtain the biomass density in mt/km^2. Biomasses for the ERP focal species were obtained from the most recent stock assessments updated through 2017 as described in Section 13.2.1.

Fisheries-independent trawl survey data were obtained from the NEFSC to parameterize the biomasses of non-assessed species. The NEFSC trawl survey is a longstanding fisheries independent monitoring program that has been conducted from 1963 – present, and samples depths from 27-366 m on the continental shelf (Azarovitz 1981). All species captured by the NEFSC trawl were re-classified into the NWACS group definitions, and catchability-corrected biomass estimates were generated following Link et al. (2006). Catchability coefficients (q) were assumed to be constant and were estimated using a Bayesian approach that incorporates information on catchability from previous studies. Details on the estimation of catchability coefficients are available in Link et al. (2006). For multispecies groups (e.g., Demersal benthivores-other) that are composed of multiple individual species with different q values, the median q was used.

Eight species were modeled using multiple stanzas (Table 39), with data obtained from stock assessments. Generally, age-specific biomass estimates were available and summed based on the defined age classes. In the absence of age-specific biomasses, these were calculated from abundance-at-age and weight-at-age data if possible.

Biomass Accumulation Rates

Biomass accumulation rates (BA_i/B) were calculated for all assessed species. Biomass accumulation rates describe the instantaneous rate of change of a functional group's biomass, and they account for groups that are not in equilibrium with their sources of mortality. Negative values indicate a declining biomass and positive values indicate an increasing biomass within the Ecopath model. Biomass accumulation rates were calculated as the rate of change in biomass per year from 1982-1983 [$(B_{1983}- B_{1982})/ B_{1983}$], based on data availability. Biomass accumulation rates were entered as relative rates (yr^{-1}) for all trophic groups, but they can also be expressed in absolute terms (with units in $mt\ km^{-1}\ yr^{-1}$).

Production/Biomass and Mortality

P/B rates for lower and higher trophic levels (i.e., non-assessed species) were primarily obtained from the EMAX models (Link et al. 2006, 2008). For assessed species, instantaneous total mortality rate estimates (Z) for each group or age class were calculated as the sum of fishing mortality rate (F) and natural mortality (M) estimates from the stock assessments. Often M was assumed to be constant in the assessments, but if age-specific M values were available, an average for each age stanza was calculated. F rates were calculated as C/B using time series from stock assessments (Christensen and Walters 2004). For full details on the calculation of mortality estimates for the ERP focal species, see section 13.2.1 .

Diet Composition

Diets for the NWACS-FULL model were taken from the published NWACS model, which were obtained from the previous EMAX models, fisheries survey data, and the literature. Diets from the EMAX models were used for many lower trophic level groups and higher trophic level groups that are not typically captured in fisheries surveys. Diets for nodes 17-52 (Table 39) were obtained primarily from the NEFSC and the NEAMAP trawl surveys, which conduct extensive diet sampling within the model domain in deeper (>27 m) and shallower (<37 m) waters respectively. For multi-stanza groups, predators were defined based on the size-cutoffs for each age class, but prey were not classified by age or size because that information was not available in the databases. Any unidentified material was divided among identified prey based on their relative proportions, for each unidentified group. Further details can be found in Buchheister et al. (2017a, 2017b).

Given the central objectives pertaining to Atlantic menhaden, diet estimates for Atlantic menhaden and three dominant predators (striped bass, weakfish, and bluefish) were augmented with literature studies. Diets for the three predators were obtained from the MSVPA-X diet database (Garrison et al. 2010; SEDAR 2015). In addition to data from 21 literature studies, the database includes the diet data from the NEFSC and NEAMAP surveys (mentioned previously), as well as the ChesMMAP survey (Bonzek et al. 2008). Following the methods of the MSVPA (SEDAR 2015), length- and region-specific diets were calculated as an average from these multiple sources weighted by sample sizes, study area, and number of years. The outputted MSVPA diets were region-specific for the MSVPA regions (Gulf of Maine, Southern New England, Mid-Atlantic Bight, Chesapeake Bay, and North Carolina). These regional diets were averaged using region-specific biomasses of each predator species as determined from the NEFSC trawl survey. Given differences in regional definitions, the MSVPA regions identified as Chesapeake Bay, North Carolina, and Mid-Atlantic Bight were assumed to be equally representative of the NWACS Mid-Atlantic Bight region.

Predator diets were modified to apportion the contribution of any multi-stanza prey groups across age-classes. For example, a 13% contribution of Atlantic menhaden to the diet of medium striped bass was allocated among the three Atlantic menhaden age-classes. These allocations were based on predators' size selectivity information when available (including all ERP focal species) and on general guidelines when size selectivity information was not available (Buchheister et al. 2017b).

Consumption Rates
Consumption rate (*Q/B*) values were primarily obtained from the EMAX models (Link et al. 2006, 2008), other ecosystem models (e.g., Christensen et al. 2009), or empirical relationships (Pauly 1989; Palomares and Pauly 1998).

Unassimilated Food
The ratio of unassimilated material to consumed biomass (*UA/Q*) represents the fraction of consumed biomass that is egested and not used for production or respiration. The assimilation efficiency is 1 – *UA/Q*. A *UA/Q* value of 0.2 was assumed for carnivorous fishes and higher trophic levels (Christensen et al. 2008). For lower trophic levels, estimates of *UA/Q* from the EMAX models were used, although several of these were increased during the balancing process to balance the detritus group.

Landings
Catch data were obtained from NOAA online databases and stock assessments. Commercial landings data by weight were downloaded for the entire east coast of the USA by year, species, state, and gear type (NOAA 2014a, http://www.st.nmfs.noaa.gov/commercial-fisheries/index). State-specific landings from North Carolina to Maine were summed to obtain landings for the NWACS model domain. The 127 unique gear types in the database were classified into seven gear types that were used as fishing fleets in the NWACS model (dredge, trawl, trap, gill net, purse seine, longline, and other). An eighth fleet, representing recreational fisheries, was also included using recreational landings data obtained from NOAA by state, year, and species (NOAA 2014b, http://www.st.nmfs.noaa.gov/recreational-fisheries/index). Recreational data included estimates of catch that was brought back to the dock and could be identified by trained interviewer (Type A) and catch that was used for bait, released dead, or filleted as identified by anglers (Type B1).

For assessed species, the landings data from the assessment reports were preferentially used, as these datasets were more detailed, tended to be larger, and were presumably more accurate. Landings data from assessments included dead discards for a more complete estimate of biomass removal. Total catch for each group was apportioned among the eight fishing fleets based on the fractional catches obtained from the NOAA databases. For all multi-stanza groups (except spiny dogfish), catch-at-age matrices from the stock assessment were used to partition catch among stanzas.

14.1.4 Balancing

Model fitting, stability, and sensitivity to parameters are addressed in the NWACS documentation (Buchheister et al. 2017b) and paper (Buchheister et al. 2017a). As with all ecosystem models, there is no single, objective method for arriving at a final model that best replicates historical trends in relative biomass or catch. To the extent possible, the NWACS model was developed following the general guidelines and best-practices for building, parameterizing, balancing, and calibrating EwE models, as recommended in the literature (Christensen et al. 2008; Heymans et al. 2016).

The process of adjusting parameters in an Ecopath model to ensure mass balance is known as "balancing". One of the key diagnostics is that all groups should have EE values < 1. Given that the balanced version of the published NWACS model was used as the starting point, the NWACS-FULL relies on all of the balancing decisions made previously, which are described by Buchheister et al. (2017a). Additional balancing was required after the data were updated for the ERP focal species. Several small changes were made, including changes to the diet matrix and minor adjustments to P/B, Q/B, and BA/B. There were two changes that were more substantial but deemed justifiable. First, Atlantic herring 1982 biomass was increased from 0.466 mt/km^2 (mid-year biomass calculated from the stock assessment) to 0.8 mt/km^2 (+72%) to account for the large amount of predation on this species by a diverse group of predators.

This amount of increase was only 16% greater than the Jan 1 biomass in 1982. This change can also help account for age-0 fish that are not included in the assessment. Second, the starting *B* of large spiny dogfish was decreased by 51% from 2.45 to 1.2 mt/km^2. Spiny dogfish were an outlier based on a pre-balancing analysis (PREBAL; Link 2010) with high biomasses in the system given their trophic level (Figure 149). This 51% reduction still kept the group at the high end of log(biomass) for their given trophic level. Also, the *B* for large spiny dogfish was very variable with a 3-year mean *B* (1981-1983) of 1.57 mt/km^2, so this change was deemed reasonable.

14.1.5 Ecopath Outputs

The balanced Ecopath model output is presented in Table 40. The food web was highly interconnected and complex, with a total of 970 trophic links in the system and an average of 15.9 links per trophic group (Figure 150, Table 41). Atlantic menhaden were consumed by a total of 22 predator groups (36% of the modeled trophic groups) and they contributed to a substantial portion of the diet of some predators, notably 30% for large striped bass and 33% for nearshore piscivorous birds.

14.2 Ecosim Model

14.2.1 Treatment of Time Series Data

The input data needed for the time-dynamic Ecosim model included time series of relative biomass, catch, fishing mortality, and fishing effort. Time series of catch and relative biomass were used as reference time series, whereas fishing mortality (for all groups with stock assessments) and fishing effort (for groups that are not assessed) were included as forcing time series. Relative biomass time series were obtained from stock assessment reports (for assessed species), or from the NEFSC trawl survey for all other fish groups. For assessed species, data from stock assessment reports were used to obtain catch and fishing mortality time series. Data from NOAA landings databases were used to obtain catch time series for non-assessed fishes and non-assessed, commercially-harvested invertebrate species. Fishing effort by fleet was assumed to be proportional to changes in fleet-specific total catch through time, and it was used to drive non-assessed trophic groups. Fishing mortality was used to drive changes in groups with stock assessment data, which included the groups of greatest commercial importance and of greatest relevance to the research objectives. For all groups (including those with multi-stanzas) that had assessments, fishing mortality rates were calculated as catch divided by biomass (*F*= *C*/*B*). If the baseline Ecopath biomass value for a group was changed during the Ecopath balancing procedure (section 14.1.4), then the *F* was calculated using the biomass time series scaled to the balanced Ecopath biomass. For any fishing mortality time series that did not extend for the full 1982-2017 time period, a 3-4 year mean of the nearest assessed years was used to extrapolate any missing values. This was typically only needed for <5 years; however, Atlantic mackerel, butterfish, and Atlantic croaker had longer periods of missing *F*s, with 6-9 years missing at either the beginning or end of the time series.

For the ERP focal species, all-time series were updated through 2017 using the most recent stock assessment data (see Section 13.3.1). For these species, the mid-year biomass estimates from the assessment models were used as reference time series instead of indices of abundance.

14.2.2 Calibration Steps

The NWACS-FULL model was calibrated to the observed, reference time series using an iterative approach. See Section 13.3.2 for full details on the approach. Briefly, the "Fit to Time Series" utility in Ecopath was used in which the most sensitive vulnerabilities for K-1 different predator-prey interactions were estimated by reducing the sum of squares of the model fits. K refers to the number of observed time series used to fit the model (K=68 for this model). This process was done iteratively until there were no substantial reductions in sum of squares or AIC. No more than 67 vulnerabilities were fitted during each tuning iteration, but different vulnerability values could be estimated during each iteration. As many as 229 different vulnerabilities were estimated using this process. This amounts to between ~3 estimated parameters per time series, or ~23% of all possible vulnerabilities (i.e. the number of predator-prey interactions in the diet matrix).

Scenarios

Eight different versions of the NWACS-FULL model (referred to as simulations, or Sims) were developed to examine the sensitivity of model results to specific decisions (Table 42). The eight versions are combinations of decisions pertaining to three components: the diet matrix, the vulnerability constraints, and manual changes pertaining to model dynamics. Sims 1 and 5 involved re-fitting the model using the iterative calibration procedure. The other 6 model versions did not require calibration to observed time series but instead applied vulnerability caps or included manual tuning adjustments made after model fitting. Sums of squares and AIC values were obtained for Ecosim simulation and used for comparison of model fits.

We explored two different options for the diet matrix. The base diet matrix was taken from the published NWACS model (Buchheister 2017b). A second diet option involved increasing the contribution of Atlantic menhaden in the diets of their predators as a way to increase Atlantic menhaden EE in the model and to examine dynamics when Atlantic menhaden importance is increased. Diets were increased to what was deemed to be the upper range of possible values, as informed by available data. For example, Atlantic menhaden contribution to spiny dogfish diet was increased from <1% (the value from NEFSC trawl survey used in the base diet matrix) to ~16% (which is the value from NEAMAP trawl survey). These dietary differences for spiny dogfish represent a range of possible, realistic diet values from extensive food habits surveys that sample in habitats of different depth (offshore vs. nearshore).

Vulnerability caps were examined to investigate the effect of these parameters on the Ecosim model. After completing the fitting process, vulnerabilities could range from 1 to 10^{10}. Vulnerability caps were established to restrict the upper end by assuming that the maximum M_2 that a predator can exert on any individual prey is equal to 75% of the total M experienced by the prey in the base year of the model (see section 13 for more detail). These changes resulted in all predator-prey v<5300. An arbitrary minimum vulnerability value of v=1.01, instead of v=1, was also employed.

The third consideration for alternative model versions was whether or not manual adjustments were made to the vulnerabilities of Atlantic menhaden and the other ERP focal species. Any vulnerability parameter with Atlantic menhaden as a predator was capped at v=4 to 5 such to make the Atlantic menhaden stock recruitment relationship have a Beverton-Holt shape as opposed to one weak density dependent compensation, which was more typical of the other model versions (Figure 151). This involved changing just four vulnerability parameters in the models. Also, the minimum vulnerabilities for each ERP focal species were also adjusted to generate more reasonable F_{MSY} values. The minimum vulnerability value for a selected ERP focal species was evaluated iteratively using the "F_{MSY}" tool within Ecopath. For example, minimum vulnerabilities (v_{min}) for all age-classes of striped bass would be changed from v=1 to v=1.1, and the F_{MSY} tool would be used to evaluate the relative catch of striped bass at varying levels of F on each of the striped bass groups. Often, with v_{min} =1, the species could sustain unrealistically high levels of fishing without having a decline in relative catch. The expectation was to have a dome shaped relative catch curve that indicates a theoretical F_{MSY} value. The v_{min} values for a given ERP focal species would be iteratively adjusted to obtain a dome-shaped curve. This was done separately for each ERP focal species, yielding v_{min} values between 1.03 and 1.1. A more formal and rigorous analysis could be conducted in the future where all yield curves are evaluated simultaneously instead of individually.

14.3 Ecosim Outputs

14.3.1 Fits to time series

Ecosim predictions from 1982-2017 generally corresponded well to observed historical trends in biomass (Figure 152). The observed time series of biomass were fitted as relative biomass as opposed to absolute biomass for each of the eight simulations, and Ecosim internally scales each relative biomass timeseries for each simulation. On the plot, the observed biomasses are scaled according to Sim 1, therefore the fits for the other simulations are slightly better than depicted (Figure 152). Generally, the different simulations tended to generate similar predictions, albeit with some changes in scale or pattern (e.g., Atlantic herring, butterfish, weakfish, cod, haddock, croaker, summer flounder). Model predictions also typically smoothed over higher-frequency interannual changes (e.g., Atlantic menhaden, squid, spiny dogfish), because no information was provided in the model to capture such variability (e.g., recruitment deviations, primary production anomalies). In the case of some species (often when better F data was not available), predictions remained relatively flat despite trends in the observed time series (e.g., shrimp, Atlantic mackerel, hake, skates, demersal piscivores). Fits for the ERP focal species tended to be good with some exceptions (weakfish-M, spiny dogfish-S) (Figure 153).

There was a greater diversity in model fits to the catch time series (Figure 154). Catches for many groups were predicted well (e.g., shrimp, ERP focal species, cod, croaker, demersal piscivores). In several cases model predictions matched the patterns but not the scale or vice versa (Figure 154). For some groups (e.g., Megabenthos other, Hake, Summer flounder, skates, sharks), both the scale and pattern of simulations deviated from the observed catches. In many of the cases of poor model fit, the lack of fit is partly attributable to absence of detailed information on fishing mortality for these groups and poor catch data; for many of these groups, fishing pressure was forced in the model using patterns of fishing effort from generalized fleets (that capture numerous groups) rather than being forced with a group-specific fishing mortality. Fits to ERP focal species tended to be reasonable at least for some simulations (Figure 155). In some instances, vulnerabilities were adjusted manually in an attempt to improve the catch fit for a species (e.g., Atlantic herring); however, this led to substantial reductions in the quality of the fit for another species (e.g., striped bass) (Figure 155), highlighting that there are tradeoffs in the quality of fit for some groups in complex models. The trophic linkages responsible for these tradeoffs should be examined closer through targeted diet studies and sensitivity analysis.

Results of the fits for all 8 model versions (Sims 1-8) suggested that Sim 1 and 5 were the best fitting models based on AIC (Table 42). However, Sim 2 and Sim 6 were chosen as the best models for evaluation because they included the manual adjustments that generated more realistic stock recruitment dynamics for Atlantic menhaden and more feasible F_{MSY} dynamics. Heymans et al. (2016) recommends sacrificing the overall fit to some groups in order to obtain more biologically reasonable dynamics for focal species.

14.3.2 Mortalities and Diets

Mortality rates for Sim 2 and Sim 6 indicate the relative contributions of fishing (*F*), predation mortality (M_2), and unexplained mortality (M_0) to total mortality (*Z*) (Figure 156). *F* represents a small proportion of total instantaneous mortality for small and medium Atlantic menhaden (<3% in 2017) and ~12% for large, age-3+ Atlantic menhaden, and these patterns did not differ greatly among the eight simulations (Figure 157). The contribution of M_2 to the total mortality differed between sim 2 and sim 6 (Figure 156) because of the increased contribution of Atlantic menhaden to predator diets (Table 42). M_0 (the difference between *Z* and $F + M_2$) was much greater for Sim 2 than Sim 6 particularly for small, age-0 Atlantic menhaden (Figure 156). For both Sim 2 and 6, *Z* has increased over the time series for small age-0 Atlantic menhaden, stayed relatively constant for medium age-1-2 Atlantic menhaden, and declined slightly for large, age-3+ Atlantic menhaden.

A variety of predators contribute to Atlantic menhaden M_2, but the dominant groups differed between Sim 2 and Sim 6. For Sim 2, bluefish, striped bass, miscellaneous demersal omnivores and piscivores, marine mammals, and birds were important sources of predation, depending on the Atlantic menhaden age class (Figure 158). For Sim 6, spiny dogfish became the most important predator (Figure 159) because Atlantic menhaden contribution to spiny dogfish diet was increased from <1% (value from NEFSC trawl survey) to ~16% (value from NEAMAP trawl survey) and spiny dogfish are a biomass-dominant group in the system. These dietary differences for spiny dogfish represent a range of possible, realistic diet values from extensive food habits surveys that sample in habitats of different depth (offshore vs. nearshore).

14.3.3 Emergent Stock Recruit relationships

The emergent stock-recruitment relationship for Atlantic menhaden were evaluated using methods recommended by (Christensen et al. 2008) and described in Section 13.4.2. Ecosim models do not include an explicit stock-recruitment equation, but Beverton-Holt or Ricker type stock-recruitment curve may emerge for multi-stanza groups depending on Ecosim parameters settings, particularly the vulnerabilities. The stock-recruitment relationship for Atlantic menhaden was examined by simulating a severe increase and decrease in fishing mortality that would generate paired biomass estimates of small (age-0) and large (age-3+) Atlantic menhaden across a wide range of stock sizes. Stock-recruitment curves were generated for each scenario. Manual adjustments were made to the vulnerability caps of Atlantic menhaden as a predator (capped at v=4 to 5) to achieve a Beverton Holt shape for Sims 2, 4, 6, and 8 (Figure 151).

14.3.4 Equilibrium MSY

An analysis was conducted to estimate and evaluate F_{MSY} values for Atlantic menhaden using the base (1982) conditions of Sim 2 and Sim 6 of the NWACS-FULL model. The methods employed are similar to those described in Section 13.4.3. However, given that fishing occurs on multiple Atlantic menhaden age-stanzas (e.g., age-1-2 and age-3+), some modifications were necessary. The Sim 2 and 6 models were adjusted by creating a single Atlantic menhaden fishing fleet that only targeted Atlantic menhaden. Using the "F_{MSY}" tool in Ecopath simulations were conducted in which the fishing effort of the Atlantic menhaden fleet was applied for 40 years using an effort multiplier (ranging from 0-5). Simulations allowed for full compensation in the system such that predators and prey would respond dynamically to the changes in Atlantic menhaden fishing. These simulations rely on the base 1982 Ecopath parameterization and the Ecosim parameters from the calibrated simulations (e.g., vulnerabilities), and project forward from 1982 for equilibrium conditions. Thus, any relative and effort values coming out of the Equilibrium MSY analysis are relative to 1982.

Relative biomass and catch for several species groups were affected by different Atlantic menhaden fishing rates (Figure 160). Biomasses of Atlantic menhaden, striped bass, and nearshore piscivorous birds exhibited the strongest declines, whereas some groups had increases in biomass (e.g., alosines, pinnipeds, summer flounder) (Figure 160). Catch trends were similar to those of biomass for non-menhaden species. Atlantic menhaden catch peaked with an effort multiplier (relative to 1982) between 2.5-3.4 for Sim 2 with corresponding F_{MSY} estimates of 0.735 (age-1-2) and 0.926 (age-3+) (Figure 160, Table 43).In Sim 6, Atlantic menhaden maximum sustainable yield occurred with an effort multiplier between 1.3 and 1.9 with F_{MSY} estimates of 0.41 (age-1-2) and 0.48 (age-3+) (Figure 160, Table 43). This analysis shows that if predators are more dependent on Atlantic menhaden, harvest policies of Atlantic menhaden may need to be more conservative to meet the management goals for those species.

14.4 Projections

The NWACS-FULL model was used for two sets of forward projections. The first set of projections examined four different fishing scenarios (at status quo or target *F* for Atlantic menhaden and the ERP focal species). The second set of projections explored numerous Atlantic menhaden fishing scenarios (from no fishing to excessive fishing) under different fishing conditions for the ERP focal species.

14.4.1 Projection Scenarios 1 (at status quo and target *F*).

The first set of projections involved maintaining fishing mortality rates at either status quo or target levels for Atlantic menhaden and for all other ERP focal species (striped bass, bluefish, weakfish, spiny dogfish, and Atlantic herring). All other modeled species were kept at their status quo levels of fishing mortality or fishing effort (from 2013, because their time-series were not updated through 2017). The four scenarios were (1) status quo fishing on Atlantic menhaden and status quo fishing on the ERP focal species (SQ Menh – SQ Others), (2) status quo fishing on Atlantic menhaden and target *F* rates for the ERP focal species (SQ Menh – TARG Others), (3) target *F* rate for Atlantic menhaden and status quo fishing on the ERP focal species (TARG Menh – SQ Others), (4) target *F* rate for Atlantic menhaden and target *F* rates for the ERP focal species (TARG Menh – TARG Others). Projections were run for 50 years.

Due to the differences in what *F* rates represent in the single species stock assessments and Ecosim, proxies for all single species *F* reference points were developed using the proportional change of the target *F* from F_{2017} (Table 4). For example, target *F* rates for Atlantic menhaden from BAM are based on abundance and calculated as the geometric mean *F* for ages 2-4, whereas EwE *F* rates are based on biomass and constrained to the age classes in the model. Based on the BAM, *F* rates from 2017 (F_{2017}=0.11) would need to be doubled to reach the Atlantic menhaden target *F* (F_{TARGET}=0.22). Therefore, the target *F* rate in EwE was scaled by doubling the 2017 *F* rate for each of the modeled Atlantic menhaden age classes. This was done to obtain all target *F* rates for projection scenarios (Table 4).

Results of these projections indicated that, with the exception of bluefish, projected biomasses for Atlantic menhaden and the other ERP focal species are not expected to change dramatically (Figure 161). Scenarios where bluefish (and the other ERP focal species) *F* rates were set at target levels generated a strong recovery of bluefish, greater than the 136% increase needed to reach their target biomass. This indicates that for bluefish, their fishing mortality rate has a strong impact on the population response (Figure 161); however, recovery was not seen for other species like striped bass, which did not recover to their target B level. While there may be some ecological explanation for this, it equally, if not more likely that the inability of striped bass to recover to B_{TARGET} in Ecosim is due to parameter estimates and the unbounded nature of the vulnerability search in Ecosim.

Projection results were also used to isolate the effect that Atlantic menhaden fishing would have on the ERP focal species. This was done by comparing the biomass of species in the “TARG Menh – SQ Others” scenario with the status quo (SQ Menh – SQ Others) scenario. Specifically, the percent biomass difference (B_{DIFF}) was calculated as B_{DIFF} = [($B_{TRIAL,y}$/$B_{2017,y}$)-1]*100, where $B_{TRIAL,y}$ is the biomass for the "TARG Menh – SQ Others" (trial) scenario in year y, and $B_{2017,y}$ is the biomass for the status quo scenario in year y. B_{DIFF} was calculated following a projection of 4 years (y=2021) and 40 years (y=2057), using both Sim 2 and Sim 6. Identical calculations were also made for catch relative to the status quo scenario (C_{DIFF}).

Results of the projection analysis (expressed as B_{DIFF} percentages) indicate that the majority of the modeled groups are not expected to have dramatic changes in either *B* or *C* relative to the status quo scenario if Atlantic menhaden are fished at their target *F* rate (Table 44). For Sim 2, B_{DIFF} and C_{DIFF} were ≤5% for all species (excluding Atlantic menhaden) after 4 years. However, the biomass of striped bass and nearshore piscivorous birds were 7% and 9% lower than the status quo fishing scenario after 40 years (using Sim 2) indicating a greater sensitivity to Atlantic menhaden fishing (Table 44). Atlantic menhaden catch was 82% greater after 40 years in the *F* target scenario relative to status quo. The magnitude of B_{DIFF} and C_{DIFF} were greater using Sim 6 of the NWACS-FULL Model (which had higher importance of Atlantic menhaden to predator diets); B_{DIFF} after 40 years was negative for most species (except Alosines and Atlantic cod), with the greatest impact on bluefish (-10%), striped bass (-11%), and nearshore piscivorous birds (-14%) (Table 44).

14.4.2 Projection scenarios 2 (at various Atlantic menhaden *F* rates)

The second set of projections using the NWACS-FULL model examined a range of Atlantic menhaden fishing mortality rates. Projections were conducted under two alternative conditions for the focal ERP focal species: *F* rates for the ERP focal species were collectively maintained at either the threshold *F* (i.e., limit *F*) ($F_{THRESHOLD}$) or at the target *F* (F_{TARGET}). Under each of these 2 conditions, simulations were run at 14 different Atlantic menhaden *F* rates. Specifically, Atlantic menhaden F_{2017} rates for all three age stanzas were scaled using an *F*-multiplier (i.e., *F*= F_{2017}*F-multiplier) where *F*-multipliers were equal to 0, 0.3, 0.6, 1, 1.5, 2, 3, 4, 7, 10, 15, 20, 30, and 40. This was done to explore a broad range of Atlantic menhaden *F* rates and to ensure that the complete Atlantic menhaden yield curve was obtained (although for reference, the maximum observed *F* from 1982-2017 for medium, age-1-2 Atlantic menhaden was only ~10 times the 2017 value). *F*-multipliers of 1, 2, and 5.5 correspond to F_{2017}, F_{TARGET}, and $F_{THRESHOLD}$ for Atlantic menhaden, respectively. Projections were run for 50 years and the relative equilibrium biomass (B/B_{2017}) was calculated, where B was the equilibrium biomass for a given *F* scenario and B_{2017} was the equilibrium biomass for the status quo Atlantic menhaden fishing scenario (i.e., when the *F*-multiplier=1). Equilibrium catches are presented relative to the maximum equilibrium catch across all Atlantic menhaden fishing scenarios (C/C_{MAX}), which would occur under the *F*=0 or $F_{EXTINCTION}$. Patterns were similar across the $F_{THRESHOLD}$ and F_{TARGET} conditions, therefore only the results for the F_{TARGET} projections are presented.

Results of these projections indicate that only bluefish and spiny dogfish would achieve their biomass targets when the ERP focal species are fished at F_{TARGET} (Figure 162). Bluefish and spiny dogfish had relative biomass values approximately twice their biomass thresholds and approximately equal to their biomass target (Figure 162) under the different Atlantic menhaden fishing mortality scenarios. Atlantic herring, striped bass, and weakfish B were all below their *B* thresholds, even when there was no fishing on Atlantic menhaden (~60%, ~50%, and ~30% of *B* thresholds, respectively) (Figure 162). Aside from Atlantic menhaden, bluefish and striped bass showed the greatest response to Atlantic menhaden fishing rates (Figure 162, Figure 163).

In addition to Atlantic menhaden, striped bass, and bluefish, some other modeled trophic groups also exhibited substantial responses to Atlantic menhaden fishing (Figure 164). Striped bass and nearshore piscivorous birds declined 69% and 77%, respectively, as Atlantic menhaden were fished out of the system (Figure 164, Table 45). Biomass of demersal piscivores, seabirds, haddock, large pelagics (HMS), and coastal sharks decreased between 10 and 15% without Atlantic menhaden in the system (Figure 164, Table 45). Atlantic cod and medium pelagic fishes exhibited biomass increases (51% and 18%) at the highest Atlantic menhaden *F* scenarios due to release from predation or competition (Figure 164, Table 45). It should be noted, however, that the lack of spatial consideration might result in an overestimation of the competitive effects on Atlantic cod.

Trends of relative catch (C/C_{MAX}) under the different Atlantic menhaden projection scenarios (Figure 165, Table 45) generally show similar trends as for biomass. Striped bass is the most sensitive species with maximum declines near 75%, but most negatively affected species have catches declining by <20% at the highest Atlantic menhaden *F* rates. Catches of Atlantic Cod and Medium pelagics were predicted to increase as a result of their biomass increases. Atlantic menhaden catch was obviously the most responsive to Atlantic menhaden *F* with maximum sustainable yield occurring at ~15 times the F_{2017} values for the species (Figure 165). This would equate to approximate F_{MSY} values for age-1-2 and age-3+ Atlantic menhaden of 0.57 and 1.68 respectively.

Projection analyses can be used to compare and evaluate potential effects of alternative Atlantic menhaden reference points on the relative biomass and yield of different trophic groups (Table 45). These analyses also indicate the species most sensitive to Atlantic menhaden fishing, as indicated previously. Compared to status quo Atlantic menhaden fishing, fishing Atlantic menhaden at F_{TARGET} is not anticipated to generate substantial losses to biomass of most other groups (2-5%), except for striped bass and nearshore piscivorous birds, which declined by 8 and 9% (Table 45). Catch of Atlantic menhaden was 84% of the theoretical maximum (if Atlantic menhaden were unfished). Fishing at $F_{THRESHOLD}$ resulted in declines of biomass of 4-8% of some groups, with again greater declines for striped bass (28%) and nearshore birds (32%), relative to F_{2017} (Table 45). Results for the F_{MSY} and $F_{EXTINCTION}$ scenarios are included for comparison to highlight the magnitude of change predicted by the model under these more extreme conditions (Table 45).

It is important to note that the values for the F_{MSY} estimates should be examined cautiously. F_{MSY} values were sensitive to vulnerability parameters, and additional analyses are recommended to examine how Atlantic menhaden F_{MSY} values change with the model's parameterization. Also, the F_{MSY} estimates from this projection analysis using *F*-multipliers (F_{MSY} =0.57 for age-1-2 Atlantic menhaden, and F_{MSY} =1.68 for age-3+ Atlantic menhaden) differ from those of Table 43 because of the slightly different methods employed and the different assumptions regarding fishing rates for the other species. Here, the ERP focal species were assumed to be fished at target F levels; all other groups were fished at 2017 status quo F levels, whereas the values in Table 43 assume 1982 fishing mortality rates. This highlights that F_{MSY} estimates for Atlantic menhaden (or any species) will be conditional on the fishing mortality rates of all other trophic groups within a system.

14.5 Uncertainties and sensitivities

Ecosystem models have inherent uncertainties that are broader than single species models, but they are a valuable tool for addressing ecosystem considerations for managers. Hilborn et al. (2017) criticized ecosystem models that aren't case-specific and that don't address important factors in systems (e.g. size selective predation, environmental variability, high natural variability of forage fishes, weak stock-recruitment relationships, and the spatial dynamics of trophic interactions) (but see Pikitch et al. 2018). The NWACS model (and other model derivatives like the NWACS-MICE model) has (or can) address several of the Hilborn et al. (2017) concerns because 1) it is case-specific, developed specifically with Atlantic menhaden in mind, 2) it accounts for size selective predation, and 3) the model can be used as a foundation to explore the topics of environmental variability, recruitment variability and spatial dynamics that are a challenge for any modeling framework, as can be seen from the explorations using the NWACS-MICE model.

A full uncertainty analysis of this model update remains to be completed. Alternative model parameterizations may lead to different model behavior (Mackinson 2014). However, previous MC simulations with the NWACS model (Buchheister et al. 2017a), in which base Ecopath biomass parameters were allowed to vary with a CV of 0.2, suggested that the general patterns of individual group responses were maintained. Ultimately, a more comprehensive sensitivity analyses should be conducted to evaluate more specific, targeted concerns including, for example, uncertainty in predator diets, vulnerability estimates, and other parameters. These were addressed to some extent here and in the MICE model.

As noted previously, any F_{MSY} estimates derived from EwE projections should be examined cautiously given their sensitivities to the vulnerability parameters. Additional analyses are recommended to examine how F_{MSY} values for Atlantic menhaden and other key species change with the model's parameterization. The fact that a species' F_{MSY} estimates are conditional on F rates for other modeled species (e.g., predators) also reinforces the need to consider tradeoffs when managing fisheries in an ecosystem context.

We noted that some of the ERP focal species (striped bass, Atlantic herring, and weakfish) did not recover in projections when these species were fished at their target F rates and when Atlantic menhaden were not fished (Figure 162). Additional manual modification of Ecosim parameters for these species could lead to projections with recovery for these species, but this would likely be at the expense of goodness-of-fit to historical data.

There remained a substantial amount of unexplained mortality for Atlantic menhaden (Figure 156), which was unexpected given the inclusion of all trophic groups, the inclusion of a directed

Atlantic menhaden fishery, and the broad spatial scale of the model. There are multiple hypotheses that could contribute to this pattern. First, the dietary contribution of Atlantic menhaden could be higher than used in Sim 2 (as explored in Sim 6), but this did not appear to address the unexplained mortality of age-1-2 and age-3+ Atlantic menhaden. Second, the overall biomass of Atlantic menhaden in the system could be lower than the estimate used from the BAM output, as the scale of biomass in single species stock assessment models can be harder to determine as opposed to biomass trends. Third, the mortality rates used for Atlantic menhaden could be too high. Fourth, the model may not be fully capturing the spatiotemporal dynamics of Atlantic menhaden-predator interactions, for example by not representing periods or locations of particularly intensive predation. The truth is likely some combination of these hypotheses, and future work is needed to explore these hypotheses in an attempt to reduce the amount of unexplained mortality (and thus increase the EE) for Atlantic menhaden in the model. A more complete accounting of total Atlantic menhaden mortality could potentially increase the effect of Atlantic menhaden fishing on other species in the system as seen when comparing the projection results of Sim 6 with Sim 2 (Table 44).

14.6 Main findings

The NWACS-FULL model can provide long-term strategic advice that is focused on broad-scale assessments of directions and patterns of change for diverse species groups (Christensen and Walters 2011). This model leverages and integrates single-species stock assessment models to provide a fuller, more thorough description of a complex ecosystem. This is done by explicitly accounting for predator-prey feedbacks that are not possible with many other models and quantifying the tradeoffs among different management decisions and scenarios.

This model identified striped bass and nearshore piscivorous birds as the two groups most strongly linked to the dynamics of Atlantic menhaden, and thus they should be of particular focus for managers. For example, striped bass and nearshore piscivorous bird biomasses were estimated to decline by 8% and 9% (respectively) if Atlantic menhaden F was increased from F_{2017} to F_{TARGET} (while ERP focal species were fished at their target levels). Striped bass and nearshore piscivorous bird biomasses would decrease more substantially by 28 and 32% if Atlantic menhaden F was changed from F_{2017} to $F_{\text{THRESHOLD}}$ (Table 45). Prioritizing consideration of striped bass (over other species) in the multispecies and ecosystem models included in this report is warranted based on the results of the NWACS-FULL model. This finding reinforces the linkage between Atlantic menhaden and striped bass indicated by past research, by the other models in this report, and by managers and stakeholders.

Some other modeled groups (e.g., bluefish, weakfish, demersal piscivores, seabirds, haddock, large pelagics, and coastal sharks) were also shown to be negatively affected by increased Atlantic menhaden fishing but their responses at Atlantic menhaden F_{TARGET} were negligible relative to F_{2017} (Table 45). Negative effects on these species groups tended to be most substantial at higher Atlantic menhaden F rates (e.g., F_{MSY}, $F_{\text{EXTINCTION}}$). The model also indicates that a few groups (e.g., medium pelagic fishes, and Atlantic cod) could benefit from increased Atlantic menhaden fishing, due to indirect ecosystem effects.

Many of the groups negatively affected by increased Atlantic menhaden fishing are those with relatively poor data (e.g., nearshore piscivorous birds, seabirds, large pelagics, coastal sharks, demersal piscivores); thus, the model results can help prioritize research needs for these species. For example, the nearshore piscivorous birds (including blue herons, bald eagles, brown pelicans, cormorants, and osprey) have the capacity to consume large amounts of prey per capita due to higher metabolism and Q/B, but their coastal biomass and diet dependencies have not been well quantified (Bryan Watts, College of William and Mary, personal communication). Other studies have shown sensitivity of seabirds to changes in forage fish abundance and management (Cury et al. 2011; Pikitch et al. 2012), and the NWACS-FULL results suggest that a better understanding of bird-Atlantic menhaden linkages would be an important ecosystem consideration for management.

In quantifying the tradeoffs in biomasses and catches of various species groups and fisheries, this model highlights that outcomes are contingent not only on the fishing rates for Atlantic menhaden, but also on the fishing rates of other species such as their predators. Taking a true ecosystem-based approach to managing Atlantic menhaden would ideally require collaboration and coordination among managers for multiple species (e.g., Atlantic menhaden and striped bass) or the system as a whole. However, the mechanisms and process are not yet in place to do this effectively in the current single-species management framework. Models and analyses like those presented in this report could be used to help advance discussion or implementation of a more comprehensive ecosystem-based management strategy for the ERP focal species or the entire ecosystem (e.g., Essington et al. 2016). Given competing interests and uncertainties, this could be a challenging process but it can be facilitated by structured decision-making approaches (e.g., Miller et al. 2010, Irwin et al. 2011) and formal management strategy evaluation (e.g., Mackinson et al. 2018), which could use the NWACS model to explore management alternatives dynamically.

15 MODEL COMPARISONS

15.1 Biomass

To compare population size estimates across models, total age-1+ biomass was used. For the surplus production models, this was equivalent to the total biomass estimates from each model. For the BAM and the VADER model, this was the sum of the beginning of the year biomass at age for ages 1-6+; the mid-year age-1+ biomass was also calculated for a metric that was more equivalent to the output of the NWACS models. For the NWACS models, total age-1+ biomass was used as the biomass in the “adult” age class from the NWACS-MICE model (which used only two age classes: age-0 and age-1+) and the sum of the “medium” and “large” size/age classes for the NWACS-FULL model (which used three size/age classes, with the “small” size class equivalent to age-0). The MCMC confidence intervals from the single-species assessment were used as a measure of the minimum uncertainty when comparing the single-species assessment estimates to the ERP assessment estimates.

Overall, all models showed similar trends in age-1+ biomass estimates and were on similar scales, both in comparison to each other and to the BAM single-species assessment results (Figure 166).

The magnitude of the estimates from the surplus production models (time-varying *r* and Steele-Henderson) were sensitive to the starting year (Figure 167). This is not surprising, since the strongest contrast in the index time-series was in the earliest year (1955-1965; Figure 13). However, on a relative scale, the surplus production models showed very similar trends to each other and to the BAM trends, regardless of the starting year (Figure 167). All three models showed a decline from the late 1950s to a low in the early 1960s before increasing through the end of the time series, although the surplus production models began increasing sooner than the BAM. The surplus production estimates of biomass were less variable than the BAM estimates, which is consistent with the structure of each of the models.

The multispecies statistical catch-at-age model, VADER, more closely tracked the BAM total biomass output and the biomass estimates were generally within the minimum uncertainty bounds of the MCMC confidence intervals for the BAM estimates (Figure 168). Both models showed an increase from 1985 to the early 1990s followed by a decline into the early 2000s and then recovery to levels higher than 1985. The VADER followed more of the variability in the BAM output than the biomass dynamics models.

The NWACS models followed the overall trend and magnitude of the BAM estimates, but, like the surplus production models, did not show the same variability as the BAM estimates (Figure 169). The NWACS models are biomass dynamic models and therefore do not capture the variability in recruitment that is captured by the statistical catch-at-age model structure. The NWACS-FULL model used biomass estimates from the BAM as input; however, the NWACS-MICE model used fishery independent indices instead of BAM output but was still able to recover similar overall trajectories as the BAM and the other ERP models.

15.2 Mortality

15.2.1 Exploitation Rate

Exploitation rate was used to compare measures of fishing mortality across models with different structures as well as different units. Exploitation rate was calculated as predicted total age-1+ removals in weight divided by beginning of the year age-1+ biomass. Age-0 fish make up approximately 1% of total Atlantic menhaden removals over the entire time series, which is why age-0 biomass was excluded. The MCMC confidence intervals from the single-species assessment were used as a measure of the minimum uncertainty when comparing the single-species assessment estimates to the ERP assessment estimates.

All models showed similar magnitude and the same declining trends since the mid-1980s (Figure 170). However, the surplus production models showed a different trend from the BAM estimates in the earliest part of the time-series, the mid-1950s through the mid-1970s (Figure

171). The surplus production models estimated the highest exploitation rates over the entire time series in the early 1960s, followed by a steady decline through the end of the time series. The BAM estimates peaked in the mid-1970s to the mid-1980s before declining. The BAM estimates were lower relative to both the surplus production model estimates and its own time-series high during the 1960s, although still higher than the estimates from the most recent years. This is due in part to differences in the input data; the surplus production model used the fishery-dependent RCPUE index as well as the fishery independent indices, while the base run of the BAM did not. When the RCPUE was included in the BAM, the trend in estimates of exploitation rate in the early part of the time-series was more similar to the surplus production models, peaking at the same time as the surplus production models and declining consistently through the rest of the time-series, with a smaller peak in the 1980s (Figure 172). The estimate of exploitation rate from this run of the BAM were still lower than the estimates from the surplus production models. The VADER and the NWACS models do not extend back that far, so no comparisons with those models were possible for this time period.

The VADER estimates of exploitation rate were very similar to the BAM estimates and generally within the MCMC confidence intervals of the BAM estimates (Figure 173).

The NWACS estimates of exploitation rate were very similar to the BAM estimates (Figure 174), but this was because exploitation rate or fishing mortality rate from the BAM output were used as input to the NWACS models, so the comparison is not truly meaningful.

15.2.2 Non-Fishing Mortality

15.2.2.1 Modeled Predation Mortality

The Steele-Henderson surplus production model, the VADER model, and the NWACS models estimated natural mortality from the predation of modeled species, referred to as M_2. To compare estimates of M_2 across models, a biomass-weighted average M_2 was calculated for the age-structured models (the VADER and full NWACS models), while the full M_2 from the Steele-Henderson model and the full M_2 on age-1+ Atlantic menhaden was used for the less structured models. The time-varying *r* surplus production model and the BAM did not separate out different components of natural mortality, so comparisons were not possible for those models.

The Steele-Henderson, VADER, and NWACS-MICE model generally showed similar trends over time, with M_2 peaking in the late 1990s to early 2000s before declining (Figure 175). This is likely driven by the trend in striped bass biomass over this time period. In contrast, the full NWACS model showed a gradual increase over the entire time-series. The magnitude of M_2 estimates varied across models, with the Steele-Henderson and VADER models estimating the highest M_2, followed by the full NWACS model, and then the NWACS-MICE model.

15.2.2.2 Total Non-Fishing Mortality

Modeled predation mortality (M_2) is only part of total natural mortality (M) in these models. To compare estimates of M across models, a biomass-weighted average M was calculated for the age-structured models (the VADER and full NWACS models) while the full *M* on age-1+ Atlantic menhaden was used for the NWACS-MICE. A biomass-weighted average *M* on age-1+ was also calculated for the BAM, although this *M* is of course input to the single-species model, not estimated. The time-varying *r* and the Steele-Henderson surplus production models included natural mortality (total or non-modeled predation) in the estimate of the intrinsic growth parameter, so comparisons were not possible for those models.

Overall, modeled predation mortality (M_2) made up a small component of total natural mortality, even for the full NWACS model (compare the scale of Figure 175 to Figure 176). Total natural mortality showed relatively little trend across all models over the last 30 years (Figure 176). Estimates of *M* were more variable for the VADER model than for the NWACS models, and all three ERP models estimated higher *M* than was used as input for the single-species model.

15.2.3 Total Mortality

Total mortality (*Z*) is the sum of natural mortality and fishing mortality. The BAM and VADER models calculated total mortality by age, while the other models calculated total mortality for age classes (full NWACS model) or the entire age-1+ population (Steele-Henderson model and NWACS-MICE models). In order to compare estimates of *Z* across models, biomass-weighted average *Z* across all age-1+ age classes was calculated for BAM, VADER, and the full NWACS model. The time-varying surplus production model did not estimate natural mortality.

The estimates of *Z* from the Steele-Henderson model were much lower than the estimates of *Z* from the other models (Figure 177). This is to be expected, given the differences in model structure: the Steele-Henderson model only estimated predation mortality from striped bass and combined other sources of natural mortality into the estimate of *r*, the intrinsic growth rate. The other models had explicit estimates or input of all sources of natural mortality, so their estimates of total mortality were higher.

In general, the models did not show much contrast in *Z* over the time series. The Steele-Henderson, VADER, and the NWACS-MICE model showed a slight declining trend from about 2000 onward, likely corresponding to declines in the striped bass population and the decline in Atlantic menhaden landings, while the full NWACS and the BAM estimates were relatively steady over this period.

15.3 Model Strengths and Weaknesses

The suite of models explored by the ERP WG resulted in similar estimates of biomass, exploitation rate, and stock status in both a single-species and multispecies context. However, each model varied in the type of advice it was able to provide, and not all models met all the ecological management objectives for Atlantic menhaden. As suggested in Section 1.5, the ERP WG determined that the ERP approach selected needed to:

- explicitly examine the trade-off between fishery removal of Atlantic menhaden and resulting trade-offs among important predators
- provide quantitative and understandable advice on removal levels of Atlantic menhaden under various predator biomass or fishing levels
- examine the implications and consequences of Atlantic menhaden harvest strategy on important predators; either through predator growth rates or survivability
- be updatable on a timeframe consistent with Atlantic menhaden management

The time-varying *r* surplus production model was able to identify changes in productivity over time and adjust the sustainable exploitation rate to take those changes into account. However, the model did not attribute changes in productivity to predation or any other specific cause, and therefore could not be used to evaluate tradeoffs between Atlantic menhaden harvest and ecosystem services.

The Steele-Henderson surplus production model attributed changes in productivity to predation, but only to striped bass as currently configured. It could, however, provide reference points that allowed for sustainable Atlantic menhaden harvest in consideration of changing striped bass dynamics. However, it could not directly capture the consequences of Atlantic menhaden harvest to the predator populations (in this case striped bass). External proxy metrics of predator condition relative to consumption levels would need to be implemented to assess the effects of Atlantic menhaden harvest on predators, which would require additional monitoring, research, and analysis to identify metrics and understand their implications for predator population dynamics.

Similarly, the current implementation of the VADER model lacks bottom-up feedback and cannot fully address the trade-offs. However, it was capable of incorporating changes in productivity due to both predation mortality and variability in recruitment.

The NWACS approaches show promise in meeting the needs of Atlantic menhaden management (Table 46). The NWACS-MICE has the desired level of complexity needed for transparent and quantitative examination of trade-offs. The NWACS-FULL model's reliance on model output from other assessments and the sheer quantity of data (some of it of poor quality) that is required make it unwieldy for providing updated advice in a timeframe suitable for management. However, the NWACS-FULL is the only model that can provide a complete evaluation of ecosystem sensitivities to Atlantic menhaden harvest policies.

Both NWACS-FULL and NWACS-MICE agree across many scenarios and sensitivities. More importantly, the NWACS-FULL model suggested that the reduced predator set of the NWACS-MICE model captured the dynamics of the more responsive predators from the full ecosystem model well. The NWACS-FULL model indicated nearshore piscivorous birds were as sensitive as striped bass to Atlantic menhaden harvest rates, while other predators not included in the NWACS-MICE model such as seabirds and demersal piscivores were less sensitive and more similar to bluefish and spiny dogfish in their response to Atlantic menhaden harvest rates. Harvest scenarios that sustain the biomass of predators included in the NWACS-MICE model were thus expected to not cause large declines for other predators that were only included in the NWACS-FULL model.

Dynamics of variable recruitment, year-class strength, and changes in fleet selectivities are nuances not well captured in the biomass dynamics approach of NWACS models, but these nuances are important in fully understanding Atlantic menhaden population dynamics.

None of the models included explicit environmental drivers in the base model run. The BAM, the surplus production model with time-varying *r,* and the VADER model could account for environmentally-driven variation in productivity or recruitment in the observed data without an explicit mechanism by estimating the annual intrinsic growth rate or recruitment annually. However, without a mechanism, these models had no way to predict changes in productivity or recruitment into the future under different environmental conditions. Modeling of environmental factors was limited by the poor understanding of the relationship between specific environmental drivers and recruitment and mortality. Ecosim models were designed to incorporate information on bottom-up drivers. The NWACS-MICE model considered primary production anomalies in an alternative run, and future versions should aim to develop an independent time series of primary production for this system that could be directly included in the model.

In addition, none of the models included spatial or seasonal dynamics. The available diet data indicate there are seasonal and regional differences in diet composition along the Atlantic coast, but the current data, as well as the understanding of Atlantic menhaden migration patterns, are not currently sufficient to support modeling at such a fine scale. As a result, nuances of population dynamics at these scales may be lost. The EwE modeling software includes a spatially-explicit component, Ecospace. The NWACS-MICE model provides an opportunity to quickly develop and test an Ecospace model to capture the seasonal-spatial dynamics, which would facilitate development of Ecospace in the full NWACS model.

16 REFERENCE POINTS

In the previous benchmark assessment report, the ERP workgroup examined a number of different ecological based reference points approaches (SEDAR 2015). These included ecosystem indicators, nutrition reference points for important predators, BAM based single species reference points coupled with rule-of-thumb harvest control rules (HCR), as well as the approaches examined in the current report. Most of the approaches not developed in the current report were discarded after SEDAR 40 (2015) and the EMO Workshop because they only provided qualitative advice, did not fully address managers concerns, or required extensive research and monitoring programs to be initiated.

One exception to this was the BAM based single species reference points coupled with rule-of-thumb HCRs. This approach used the current single-species assessment for Atlantic menhaden (BAM) with a series of potential HCRs as outlined by Smith et al. (2011) and Pikitch et al. (2012).

Smith et al. (2011) recommended maintaining forage fish populations at target biomass of 75% of unexploited biomass to prevent negative consequences to predators, compared to the approximately 60% level implied by fishing at F_{MSY}. Pikitch et al. (2012) recommended a precautionary approach for forage fish management in order to sustain both predator and prey species, including fishing at 50-75% of F_{MSY} and using a biomass threshold of 30-40% of unexploited biomass, depending on the quality of data available.

At the behest of the Board, the EMO workgroup developed the recommendations of Pikitch et al. (2012) for management consideration (see ASMFC 2017) as an interim step while ERPs were under development. However, the ERP workgroup noted a number of difficulties in applying the rule-of-thumb approaches to the coastwide stock of Atlantic menhaden in their ecosystem context. Chief among the issues was that the Pikitch et al. (2012) rule-of-thumb reference points and harvest control rules were derived from ecosystems or locations/seasons where a majority of the trophic energy passed through a handful of species. They were not well tested in ecosystems like the Mid-Atlantic, which have a diverse forage base and a suite of generalist predators. Additionally, Pikitch et al. (2012) ERPs could not quantitatively examine the tradeoffs and risk to predators resulting from Atlantic menhaden fishery removals, a vital function that both managers and stakeholders were interested in examining. As a result, the ERP WG recommended developing ERP models using data specific to Atlantic menhaden and its ecosystem instead.

16.1 Model Reference Points

The suite of models explored by the ERP WG are capable of producing MSY-based reference points (or MSY-proxy reference points such as %SPR). What sets these reference points apart from the single-species equivalents is the consideration of changing productivity over time, whether that is explicitly modeled as predation or simply estimated.

The time-varying *r* model produced estimates of U_{MSY} that take into account changes in productivity over time. Although productivity is lower in recent years than it was at the beginning of the time-series, declining removals have brought the exploitation rate under the threshold of 75%U_{MSY}, indicating that Atlantic menhaden are not currently experiencing overfishing. Biomass is also above the B_{MSY} target, indicating the stock is not overfished.

The Steele-Henderson model produced reference points in terms of "maximum usable production" (MUP) instead of the traditional "maximum sustainable yield" (MSY) concept. Proxy metrics of striped bass condition relative to consumption were developed as targets to relate levels of consumption to striped bass population health, as there is no bottom-up feedback within this model. The Steele-Henderson surplus production model also indicated Atlantic menhaden are not overfished ($B_{2017}/B_{MUP} > 1$) and are not experiencing overfishing ($F_{2017}/F_{MUP} < 1$). Condition metrics indicated current striped bass consumption of Atlantic menhaden was sufficient to sustain the 2017 striped bass population in a healthy condition.

The VADER model and the two NWACS models produced MSY or MSY-proxy reference points based on the time-varying mortality components of these models. The estimates of F_{MSY} from the NWACS models were higher than the estimates of *F* in 2017 for all age stanzas.

In all, the ERP models produced similar assessments of stock status to the single-species assessment results, which determined that Atlantic menhaden were not overfished and were not experiencing overfishing in 2017, relative to the reference points calculated by each model. However, the values of these reference points are determined by the ecosystem conditions (e.g., productivity levels, predator consumption levels) under which they are calculated. While the models were able to calculate reference point values for different levels of productivity or predation, there is no one "right" answer or reference point value; the sustainable level of Atlantic menhaden mortality depends on the management objectives for the predators and the ecosystem, which is ultimately a decision for managers. Therefore, the ERP WG recommended a method for developing an ERP target and threshold, rather than a specific value, to allow managers and stakeholders to evaluate the tradeoffs between Atlantic menhaden harvest and predator biomass.

16.2 ERP Target and Threshold

The ERP WG recommended using the NWACS-MICE model to develop fishing mortality targets and thresholds for Atlantic menhaden that help account for Atlantic menhaden's role in the ecosystem. The final values for the ERP target and threshold will be a management decision that takes into account the management objectives of both Atlantic menhaden and their predators. However, the ERP WG put forward example values of an ERP target and an ERP threshold based on existing management objectives for striped bass.

Striped bass was the focal species for this analysis because it was one of the most sensitive species to Atlantic menhaden *F*, and it allowed for a tractable description of tradeoffs for key groups in the system. ERPs based on striped bass biomass should also sustain other species in the ecosystem that were less sensitive to levels of Atlantic menhaden removals.

The ERP target was defined as the maximum *F* on Atlantic menhaden that would sustain striped bass at their biomass target when striped bass were fished at their *F* target. The ERP threshold was defined as the maximum *F* on Atlantic menhaden that would keep striped bass at their biomass threshold when striped bass were fished at their *F* target. All other species were maintained at their current *F* rates.

The example ERP target and threshold were lower than the current single-species target and threshold. The single-species assessment reported *F* reference points and annual values as the geometric mean of ages 2-4, rather than the full *F* (i.e., maximum *F*-at-age) values; the equivalent full *F* values for the single-species *F* target, threshold, and 2017 estimate are presented here for comparison with the full *F* values derived from the NWACS-MICE model. The ERP target was estimated at a full *F* of 0.188, compared to a full *F* of 0.314 for the single-species target. The ERP threshold was estimated at a full *F* of 0.573, compared to a full *F* of 0.856 for the single-species threshold. The current estimate of full *F* from the BAM model is 0.157, below both the example ERP target and ERP threshold, indicating Atlantic menhaden are not experiencing overfishing from an ecosystem perspective.

This example was based on the *F* and *B* targets laid out in the striped bass fishery management plan. Higher or lower reference points for striped bass will result in higher or lower reference points for Atlantic menhaden. In addition, this example maintained the other species at their current *F* rates; higher or lower *F* rates on other species would also result in different reference point values for Atlantic menhaden. Managers and stakeholders can select final reference point values after examining the tradeoffs between Atlantic menhaden harvest, predator harvest rates, and predator biomass levels.

17 DISCUSSION

17.1 Synthesis of Findings

The ERP WG explored several different models capable of producing ecological reference points for Atlantic menhaden, ranging from mechanistically very simple with minimal data inputs (time-varying *r* surplus production model) to mechanistically very complex with intensive data needs (full NWACS model). All of the ERP models explored here agreed about the overall trend of Atlantic menhaden population size and exploitation rates over the last 30 years, a generally increasing trend in biomass and a decreasing trend in exploitation rate. These trends and the magnitude of the estimates were also very consistent with the estimates from the single-species assessment. This should not be surprising, since all the ERP models used the same time-series of total removals, life history parameters, and indices of abundance as the single species model, and in some cases (the NWACS models) used output from the single species model directly. However, the true value of these kinds of ecological models is not the ability to recreate a single-species assessment, but their ability to put those dynamics into an ecosystem context and develop reference points that take into account management objectives of other ecosystem components. These models produced reference points and management advice that were consistent across the different approaches as well.

The time-varying *r* model indicated that although productivity is lower in recent years than it was at the beginning of the time-series, declining removals have brought the exploitation rate under the threshold of 75%U_{MSY}, so Atlantic menhaden are not currently experiencing overfishing. Biomass is also above the B_{MSY} target, indicating the stock is not overfished.

The Steele-Henderson surplus production model also indicated Atlantic menhaden are not overfished ($B_{2017}/B_{MUP} > 1$) and are not experiencing overfishing ($F_{2017}/F_{MUP} < 1$). Proxy metrics of striped bass condition relative to consumption indicated striped bass consumption of Atlantic menhaden in 2017 was sufficient to sustain the current striped bass population in a healthy condition.

The VADER model indicated that Atlantic menhaden biomass would increase if fished under status quo *F*, even as predator biomass increased. At the single-species *F* target, Atlantic menhaden biomass decreased by less than 10% under increasing predator levels, but stabilized at that level in the long-term projections. The VADER model did not include bottom-up effects of Atlantic menhaden abundance on predators, so in these scenarios, the increases or decreases in predator biomass is strictly a result of increasing or decreasing predator *F* to the target level. The combination of predation mortality and fishing mortality were sustainable for Atlantic menhaden in these projections, but the VADER model as currently configured could not evaluate the effects of Atlantic menhaden biomass on predator populations.

The NWACS-MICE and NWACS-FULL models projected that current levels of Atlantic menhaden harvest are unlikely to lead to declines in key predator species from their current levels.

The consistency of results across ERP approaches presented here suggest that Atlantic menhaden dynamics are only moderately sensitive to changes in predator dynamics and that minor changes in Atlantic menhaden harvest rates are not expected to have major negative effects on most predators. This is most likely due to current Atlantic menhaden management and aspects of Atlantic menhaden ecology and population dynamics. The ERP results are also consistent with the single-species assessment results, which determined that Atlantic menhaden were not overfished and were not experiencing overfishing. However, the ERP models indicated that fishing Atlantic menhaden at the single-species *F* threshold would cause declines in predator biomass, particularly for striped bass and nearshore piscivorous birds.

Atlantic menhaden are not managed with the traditional single-species approach of maximizing yield via F_{MSY} or an F_{MSY} proxy. Instead, the current single-species reference points for Atlantic menhaden are based on the historical performance of the fishery and the stock from 1960-2012. This approach captured the mean Atlantic menhaden dynamics across a large range of predator and fishery dynamics. Also, the Atlantic Menhaden Management Board has set the TAC lower than what would be allowed if fishing at *F* target, an *ad hoc* buffering approach that was adopted in recognition of the uncertainty surrounding Atlantic menhaden's role as a forage fish. The single-species assessment indicated fishing mortality rates on Atlantic menhaden are low and having been declining over time, and biomass and fecundity have increased. As a result, current levels of Atlantic menhaden harvest are unlikely to cause declines in predator species, as the ERP models indicate.

The impact to predators by fishing Atlantic menhaden is somewhat mitigated by the availability of other prey items in the same ecological niche. The nearshore environment of the Northwestern Atlantic has a very diverse forage base that includes Atlantic and river herrings, bay anchovy, sandeels, sardines, and many other small forage fishes. Additionally, most of the important predators on Atlantic menhaden are generalists; Atlantic menhaden may be a significant component of the diet for some predator size classes in some seasons and areas, but on a population scale, Atlantic menhaden are an important but not dominant component (i.e., do not comprise 50% or more) of predators' diets. In short, this ecosystem is not "wasp-waisted" like many of the ecosystems that formed the basis of previous literature on the subject (e.g., Smith et al. 2011, Pikitch et al. 2012); there is a diverse array of forage fishes to meet the demand of a generalist predator base.

ERP and BAM results suggest that most of the predation mortality on Atlantic menhaden occurs on the youngest and smallest age classes, typically ages 0-1. Given that the fishery harvest is dominated by ages 2+, and the highly variable nature of recruitment, it appears that the main driver for Atlantic menhaden availability to predators is recruitment success. That success is only marginally tied to adult population size; a host of environmental parameters may be more important across the population size of Atlantic menhaden so far seen, as suggested by Hilborn et al. (2017). Furthermore, Atlantic menhaden's life history lends itself to resiliency under exploitation. They are highly fecund, and larval data indicate they spawn nearly year-round across the coast, providing a buffer against unfavorable environmental conditions (Simpson et al. 2016). The majority of the population matures before peak selectivity in the fishery, allowing most individuals to spawn for at least one year before they are fully vulnerable to the fishery.

17.2 Synthesis of Management Advice

The ERP WG recommends using the BAM single-species assessment model in conjunction with the NWACS-MICE model to establish sustainable harvest levels for Atlantic menhaden that take into account their role as forage fish.

This approach combined the individual strengths of each model: BAM provided the single-species information, which incorporates the more nuanced structure and recruitment variability of the statistical catch at-age model, and the NWACS-MICE model provided an evaluation of the impact of proposed harvest scenarios on important predator species. The relative harvest strategy from the NWACS-MICE model that meets management objectives for Atlantic menhaden and the key predators can then be translated into a TAC using the single-species model. The NWACS-MICE model was chosen as the ERP model for this analysis as it included both top-down effects of predation on Atlantic menhaden biomass and bottom-up effects of Atlantic menhaden population size on predator biomass. The NWACS-FULL model was the only other model that explicitly included both types of feedback within the ecosystem, but the data demands of the NWACS-FULL model would make updating it on the frequent timeframe necessary for management difficult. The NWACS-FULL model indicated that striped bass was the most sensitive fish species, consistent with the NWACS-MICE model, suggesting that harvest strategies developed through the NWACS-MICE model that are sustainable for striped bass should also be sustainable for the major predatory fishes in the system. The NWACS-FULL model indicated that nearshore piscivorous birds, which includes species like osprey and herons, were also sensitive to Atlantic menhaden harvest. Based on the results of the scenarios examined in the NWACS-FULL model, nearshore piscivorous bird responses are expected to be similar to striped bass responses. As a result, harvest strategies developed with the NWACS-MICE model that maintain or rebuild striped bass biomass are likely to have a similar positive effect on nearshore piscivorous birds, though the model would not capture the potential effects on nearshore piscivorous birds in the full ecosystem context.

There are downsides, however, to this approach. As outlined in ASMFC (2017), translation between two models with different levels of complexity such as different age structures, recruitment assumptions, and selectivities, can increase uncertainty. Likewise, propagating error for both models through the translation process can also pose challenges and make it more difficult to assess the risk and uncertainty associated with each management strategy. More work on this topic is needed, but these issues could be resolved in consultation with managers about their preferred level of risk.

As the NWACS-MICE surface plots show (Figure 144-Figure 146), there is no one “right answer” to the ecological reference point question. Sustainable harvest for Atlantic menhaden depends on the management objectives of both the predator species and Atlantic menhaden. The approach used to develop the NWACS-MICE surface plots provide a tool for managers to evaluate the tradeoffs between levels of Atlantic menhaden harvest, levels of predator harvest, and resulting biomass for all modeled species, not just striped bass. Managers have already performed this type of evaluation in a qualitative way with the *ad hoc* buffering approach used in recent Atlantic menhaden management. The tool presented would allow for this evaluation in a quantitative, transparent way, which is the overarching goal of the ecological reference point process.

18 RESEARCH AND MODELING RECOMMENDATIONS

The ERP WG endorsed the research recommendations laid out in the single-species assessment to improve the understanding of Atlantic menhaden population dynamics, especially the recommendations to develop an Atlantic menhaden-specific coastwide fishery-independent index of adult abundance and to continue to investigate environmental covariates related to productivity and recruitment on a temporal and spatial scale.

In addition, the ERP WG identified a number of research needs to improve the multispecies modeling efforts and the development of ecological reference points for Atlantic menhaden, as well as process considerations to fully implement ecosystem-based fishery management.

18.1 Future Research and Data Collection

18.1.1 Short term

1. Expand collection of diet and condition data along the Atlantic coast to provide seasonally and regionally stratified annual, year-round monitoring of key predator diets to provide information on prey abundance and predator consumption. This could be done through existing data collection programs.

18.1.2 Long term

1. Improve monitoring of population trends and diet data in non-finfish predators (e.g., birds, marine mammals) and data-poor prey species (e.g., bay anchovies, sand eels, benthic invertebrates, zooplankton, and phytoplankton) to better characterize the importance of Atlantic menhaden and other forage species to the ecosystem dynamics.

18.2 Modeling Needs

18.2.1 Short term

1. Conduct a management-strategy evaluation (MSE) to identify harvest strategies that will maximize the likelihood of achieving the identified ecosystem management objectives.
2. Continue development of the NWACS-MICE model to incorporate recruitment deviations (from external models or primary productivity time series) to better capture the productivity dynamics of Atlantic menhaden and other species.
3. Continue development of the VADER model to include bottom-up effects of Atlantic menhaden abundance on key predator species.
4. Continue development of the NWACS-FULL model to bring other species up to date and continue exploring the impacts of fishing on higher trophic level predators like birds and mammals.

18.3 Management Process Needs

18.3.1 Short term

1. Develop a coordinated timeline of assessments and assessment updates for Commission-managed species in order to provide the most up-to-date multispecies inputs for the NWACS-MICE model during ERP assessment updates.

18.3.2 Long term

1. Develop a plan to coordinate management of Atlantic menhaden and their predator species across management Boards. This will require changes to the way the Commission has historically operated. These species are currently managed by separate Boards within the Commission, and management objectives, including *F* and *B* targets for each species, are set independently of each other. For successful ecosystem-based fishery management, consistent management objectives for individual species and the ecosystem should be set holistically with the engagement of all managers and stakeholders.

18.4 Timing of Future Assessments

The ERP WG recommended updating the NWACS-MICE model in conjunction with the next single-species assessment update (in approximately three years), and recommended a full ERP model benchmark in six years if sufficient progress has been made on the modeling research recommendations. The ERP benchmark process should include updating and reevaluating the other models considered here, as well as any other promising models that could address management objectives, to continue to improve the understanding of Atlantic menhaden's role in the ecosystem.

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20 TABLES

Table 1. ERP models explored and the fundamental management objectives they address.

	FUNDAMENTAL OBJECTIVES											
	Sustain menhaden to provide for fisheries				Sustain menhaden to provide for predators				Provide stability for all types of fisheries		Minimize risk to sustainability due to changing environment	
	PERFORMANCE MEASURES				PERFORMANCE MEASURES				PERFORMANCE MEASURES		PERFORMANCE MEASURES	
	Abundance/ biomass of menhaden	Menhaden yield objectives	Age Composition	Historical distribution (Age comp as proxy)	Abundance/ biomass of predators	Predator yield objectives	Predator nutrition	Prey availability relative to predator distribution	Stability in yield for directed menhaden fisheries	Stability in yield for non-menhaden fisheries	*Model explicitly considers uncertainty about future environment for menhaden*	*Model explicitly considers uncertainty about future environment for predators*
Single-Species Models												
BAM Statistical Catch-at-Age Model (current model)	X	X	X	X					X		*	
Multi-Species Models												
Surplus Production												
Steele-Henderson	X	X					X (proxy)		X		*	*
Time-varying r	X	X							X		*	*
Multi-species Catch-at-Age (MSSCA)	X	X	X	X	X	X	X (proxy)	*	X	X	*	*
Ecopath with Ecosim (EwE)	X	X	X	X	X	X	X (proxy)	*	X	X	*	*

*: Indicates it is possible to modify the model to meet that performance objective, but would require extensive additional work

Table 2. Annual population-level consumption and associated ranked levels of consumption for the top five predator species consuming Atlantic menhaden based on the NEFSC Food Habits Database. Consumption is shown in most multi-year averages (note: multi-year averages are for the most recent x number of years noted working back from 2012) and averages for the whole analysis time period from 1981-2012. Shaded cells indicate species that were not included in the final list of key ERP predators.

	Annual population-level consumption				Ranked levels of consumption			
	1 year	5 year	10 year	All years '81+	1 year	5 year	10 year	All years
Spiny dogfish	142,944,946	96,031,910	85,632,752	80,475,599	1	1	1	1
Striped bass	4,052,220	30,601,675	17,793,908	7,816,855	2	2	2	2
Bluefish	1,465,923	2,049,989	2,196,640	2,608,901	3	3	3	3
Weakfish	463,150	376,622	1,007,166	787,034	4	5	4	4
Smooth dogfish	446,791	588,205	900,819	757,217	5	4	5	5
Atlantic angel shark	345,432	180,740	139,435	141,519	6	7	7	6
Clearnose skate	34,830	31,340	18,387	9,554	8	8	8	10
Dusky shark	-	-	3,483	100,672	-	-	10	8
Goosefish	258,958	211,633	145,905	124,572	7	6	6	7
Sandbar shark	-	7,199	5,530	15,533	-	9	9	9
Spiny butterfly ray	4,245,350	4,639,049	6,437,951	4,737,723	2*	3*	3*	3*

*: Spiny butterfly ray consumption estimates were not included in the ranking because of the extremely small sample size of stomachs available for the analysis.

Table 3. Single-species reference points and total biomass equivalents. *: Estimates for bluefish and weakfish are based on preliminary assessment updates and may not match values used in management.

Species	SSB Target Definition	SSB Target	B Target Proxy
Atlantic herring	Projected SSB when fishing at $F_{40\%SPR}$	189,000 mt	448,000 mt
Bluefish	Projected SSB when fishing at $F_{40\%SPR}$	261,591 mt*	276,892 mt*
Spiny dogfish	Biomass of females > 80cm	159,288 mt	511,776 mt
Striped bass	125% of female SSB in 1995	114,295 mt	277,361 mt
Weakfish	Not defined	N/A	N/A

Species	SSB Threshold Definition	SSB Threshold	B Threshold Proxy
Atlantic herring	½ SSB target	94,500 mt	224,000 mt
Bluefish	½ SSB target	130,795 mt*	138,446 mt*
Spiny dogfish	½ SSB target	79,644 mt	255,888 mt
Striped bass	Female SSB in 1995	91,436 mt	221,889 mt
Weakfish	30% of unexploited SSB	8,815 mt*	11,489 mt*

Species	*F* Target Definition	*F* Target
Atlantic menhaden	Median of mean age 2-4 *F* 1960-2012	0.22
Atlantic herring	Not defined; use 90% of *F* threshold as proxy	0.46
Bluefish	Not defined; use 90% of *F* threshold as proxy	0.14
Spiny dogfish	Not defined; use 90% of *F* threshold as proxy	0.22
Striped bass	*F* rate projected to achieve SSB target	0.20
Weakfish	$Z_{SPR30\%}$ =0.98; based on M=0.43	F=0.55*

Species	*F* Threshold Definition	*F* Threshold
Atlantic menhaden	Maximum of mean age 2-4 *F* 1960-2012	0.60
Atlantic herring	$F_{40\%MSY}$	0.51
Bluefish	$F_{40\%MSY}$	0.16*
Spiny dogfish	*F* rate projected to achieve SSB target	0.24
Striped bass	*F* rate projected to achieve SSB threshold	0.24
Weakfish	$Z_{SPR20\%}$=1.36; based on M=0.43	F=0.93*

Table 4. Single-species estimates of total biomass and *F* in 2017 and percent change needed to achieve target and threshold values. *: Estimates for bluefish and weakfish are based on preliminary assessment updates and may not match values used in management.

Species	Status Quo Biomass	% Change to reach B Threshold	% Change to reach B Target
Atlantic menhaden	4,677,129 mt	N/A	N/A
Atlantic herring	239,472 mt	-6%	+87%
Bluefish	117,107 mt*	+18%	+136%
Spiny dogfish	641,132 mt	-60%	-20%
Striped bass	173,663 mt	+41%	+77%
Weakfish	3,209 mt*	+258%	N/A

Species	Status Quo F	*F* Multiplier to reach Threshold	% Change to reach *F* threshold	*F* Multiplier to reach Target	% Change to reach *F* target
Atlantic menhaden	0.11 (ages 2-4)	5.455	+445%	2.000	+100%
Atlantic herring	0.45 (ages 7-8)	1.133	+13%	1.022	+2%
Bluefish	0.34 (Full F) *	0.471	-53%	0.412	-59%
Spiny dogfish	0.15 (Full F)	1.600	+60%	1.467	+47%
Striped bass	0.31 (Full F)	0.774	-23%	0.645	-35%
Weakfish	0.23 (Full F)*	4.043	+304%	2.391	+139%

Table 5. Estimated parameters, starting values, bounds, parameter estimates, and coefficient of variation (CV) from the SPMTVr model.

Parameter	Description	Starting value	Lower/upper bounds	SPMTVr Estimate	SPMTVr CV
r_{1957}	Intrinsic growth rate (1957)	0.7	0.0001/2.1	0.74	0.96
K	Carrying capacity (1,000 t)	6,828,000	1,000/20,000	2,182,790	0.02
B_{1957}	Initial biomass (1,000 t)	2,424,000	200/10,000	2,182,820	0.02
q_{RCPUE}	Catchability (RCPUE index)	0.001	$1x10e^{-8}$/0.005	0.00076	0.03
q_{NAD}	Catchability (NAD index)	0.001	$1x10e^{-8}$/0.005	0.00057	0.04
q_{MAD}	Catchability (MAD index)	0.001	$1x10e^{-8}$/0.005	0.00057	0.03

Table 6. Summary of results for index-based fishing-only and Steele-Henderson predator-prey surplus production models with candidate predators. Shaded cells indicate parameters at constraint. Parameter *r* is the intrinsic rate of increase; *K* is maximum biomass of ages 1+ Atlantic menhaden; B_{1985} is the initial biomass in 1985 of ages 1+ Atlantic menhaden; *d* is maximum Atlantic menhaden biomass consumption per predator biomass; *A* of ages 1+ Atlantic menhaden where predator satiation begins; and MT = metric tons. Estimates with parameter *d* = 0 do not produce M_2.

Model Predator 1 Predator 2	Fishing	Fishing Bass	Fishing Bluefish	Fishing Dogfish	Fishing Bass Bluefish
AIC_c	-156	-156	-149	-154	-149
SSQ / N	0.170	0.161	0.170	0.170	0.170
Parameters					
r	0.32	2.27	0.30	0.32	3.00
K (MT)	3,430,522	1,071,224	4,998,378	3,433,550	4,456,472
B_{1985} (MT)	3,022,384	775,014	2,977,523	3,022,199	3967296.312
q	0.00000030	0.00000110	0.00000032	0.00000030	0.00000019
Predator 1 d		11.0	5.1	0	14.0
Predator 1 A (mt)		1,143,513	4,933,403	10,000,000	507,280
Predator 2 d					0
Predator 2 A (mt)					10,000,000

Table 7. Summary of stock status metrics, conditions for breaching their thresholds, estimated risk of exceeding their thresholds, and mean and 5th and 95th percentiles in 2017 from the Steele-Henderson surplus production model.

			Statistic		
Metric	Threshold	Risk (%)	Mean	5th %	95th %
B / B_{MUP}	$\leq$ 1.0	0	1.27	1.26	1.29
Z_2 / Z_{MUP}	$\geq$ 1.0	0	0.73	0.71	0.74
D_t / P_t	$\leq$ 2.0	0	2.87	2.72	2.96
F / Z_2	$\leq$ 0.4	0	0.31	0.30	0.32
F / F_{MUP}	$\geq$ 1.0	0	0.62	0.60	0.64

Table 8. Parameter estimates from the Steele-Henderson surplus production model for base and sensitivity runs. See Section 11.4 for descriptions of sensitivity runs. Parameter *r* is the intrinsic rate of increase; K is maximum biomass of ages 1+ Atlantic menhaden; B_{1985} is the initial biomass in 1985 of ages 1+ Atlantic menhaden; d is maximum Atlantic menhaden biomass consumption per predator biomass; A of ages 1+ Atlantic menhaden where predator satiation begins; MUP = maximum usable production, and mt = metric tons.

	Parameter					
Model run	*r*	*K* (mt)	B_{1985} (mt)	*d*	*A* (mt)	MUP (mt)
Base	2.27	1,071,224	775,014	11.0	1,143,513	608,517
Long	2.56	971,189	706,019	7.0	783,581	621,302
d bio	2.01	1,200,251	871,548	15.6	1,548,475	602,859
d wider	1.66	1,402,670	1,092,780	17.0	2,025,389	582,062
d constraint	2.21	1,093,466	787,412	12.6	1,263,289	605,021
minus 20%	2.05	1,175,749	910,940	9.0	1,151,076	602,375
plus 20%	2.22	1,089,415	777,844	17.0	1,477,231	604,163

Table 9. Correlations among model parameters for base and sensitivity runs of the Steele-Henderson surplus production model. *n*=7.

Parameter	Statistic	r	K	1985 biomass	Bass d
K	ρ	-0.99			
	p-value	<.0001			
1985 biomass	ρ	-0.96	0.99		
	p-value	0.0004	<.0001		
d	ρ	-0.67	0.63	0.50	
	p-value	0.1018	0.1315	0.2522	
A	ρ	-0.91	0.90	0.83	0.90
	p-value	0.0044	0.0053	0.0217	0.0064

Table 10. Summarized percentage differences between Steele-Henderson model base run and sensitivity analyses estimates of B / B_{MUP}, Z / Z_{MUP}, and D_t / P_t estimates for 1985-2017.

Sensitivity run	Base	Long run	d penalty	d bioen	d wide	minus 20%	plus 20%
				B / B_{MUP}			
Maximum	4.5%	-0.5%	4.5%	4.5%	3.6%	0.9%	2.9%
Minimum	-7.7%	-4.8%	-0.6%	-0.6%	-7.0%	-5.7%	1.7%
5th %	-4.4%	-4.6%	0.1%	0.1%	-6.2%	-5.0%	1.7%
95th %	3.3%	-1.3%	3.4%	3.4%	3.0%	-0.4%	2.8%
Median	1.2%	-3.7%	2.6%	2.6%	0.1%	-2.3%	2.4%
Average	0.0%	-3.4%	2.3%	2.3%	-0.6%	-2.6%	2.3%
				Z / Z_{MUP}			
Maximum	8.2%	7.5%	8.2%	-1.1%	3.3%	4.7%	-0.7%
Minimum	-5.7%	0.8%	4.7%	-5.0%	-5.6%	0.6%	-5.7%
5th %	-4.4%	1.4%	4.9%	-4.7%	-4.6%	0.8%	-4.9%
95th %	7.3%	7.4%	7.7%	-1.6%	2.7%	4.0%	-0.9%
Median	1.5%	4.1%	6.5%	-3.0%	-0.6%	2.9%	-2.5%
Average	1.2%	4.5%	6.5%	-3.0%	-0.5%	2.7%	-2.9%
				D_t / P_t			
Maximum	14.2%	9.5%	5.2%	5.2%	14.2%	9.1%	4.5%
Minimum	-10.8%	-6.5%	-8.5%	-8.5%	-10.8%	-5.3%	-7.8%
5th %	-6.1%	-6.1%	-6.3%	-6.3%	-1.6%	-2.8%	-6.5%
95th %	8.0%	8.1%	4.2%	4.2%	9.9%	6.5%	3.7%
Median	1.2%	-2.6%	1.0%	1.0%	6.3%	2.3%	0.8%
Average	1.0%	1.0%	1.3%	1.5%	2.7%	1.2%	0.2%

Table 11. Parameter estimates for base (1985-2017) and retrospective runs of the Steele-Henderson surplus production model. Parameter *r* is the intrinsic rate of increase; K is maximum biomass of ages 1+ Atlantic menhaden; d is maximum Atlantic menhaden biomass consumption per predator biomass; A of ages 1+ Atlantic menhaden where predator satiation begins; MUP = maximum usable production, and mt = metric tons.

	Parameters					
	r	K	B_{1985} (mt)	d	A (mt)	SSQ / N
Base run	2.27	1,071,224	775,014	11.0	1,143,513	0.16
End year			Retrospective			
2013	1.98	1,267,432	944,767	16.9	1,692,663	0.15
2014	2.40	1,042,689	767,328	12.8	1,178,648	0.15
2015	2.13	1,128,231	820,564	10.4	1,169,501	0.16
2016	2.33	1,042,940	742,861	11.6	1,132,136	0.16
Index pairs			Index removal			
RCPUE-MAD	2.01	1,140,899	824,306	17.0	1,688,439	0.21
RCPUE-NAD	1.26	1,360,902	767,694	7.7	2,138,622	0.22
NAD-MAD	2.12	1,769,138	1,459,544	17.0	1,363,111	0.31

Table 12. Correlations of Steele-Henderson model parameters used in projections. Parameter *r* is the intrinsic rate of increase; *K* is maximum biomass of ages 1+ Atlantic menhaden; *d* is maximum Atlantic menhaden biomass consumption per predator biomass; A of ages 1+ Atlantic menhaden where predator satiation begins.

Parameter	r	K	d
K	-0.50		
d	0.11	0.39	
A	-0.59	0.44	0.55

Table 13. Summary of parameters, their distribution, and shape, scale, and location values for their probability density functions used in Monte Carlo simulations of four management scenarios for the Steele-Henderson surplus production model.

Parameter	Species	Distribution	Shape	Scale	Location
r	Menhaden	Log logistic	α = 0.10	β = 7.75	γ = 2.19
K	Menhaden	Laplace		σ = 15,713	u = 1,064,665
d	Bass	Laplace		σ = 0.30	u = 11.0
A	Bass	Laplace		σ = 17,155	u = 1,140,035
2018 biomass	Menhaden	Laplace		σ = 8,903	u = 676,885
2018 biomass	Bass	Normal		CV = 0.06	u = 134,796

Table 14. Summary of Steele-Henderson surplus production model projection results. Bass = Ages 3+ striped bass. Atlantic menhaden = Ages 1+ Atlantic menhaden. All parameters reported are for terminal year estimates that were considered "equilibrium" estimates for a strategy. 5% and 95% = bounds of the 90% percentile interval for simulated results. mt = metric tons.

Species	Parameter	Mean	5%	95%	Threshold breach risk
Strategy		Status Quo			
Bass	Biomass (mt)	134,000			
Menhaden	Harvest (mt)	175,000			
Menhaden	B / B_{MUP}	1.28	1.22	1.33	0%
Menhaden	Z / Z_{MUP}	0.72	0.67	0.78	0%
Bass	D_t / P_t	2.89	2.7	3.09	0%
Strategy		Status quo F for menhaden, bass recover			
Bass	Biomass (mt)	260,000			
Menhaden	Harvest (mt)	132,000			
Menhaden	B / B_{MUP}	0.95	0.85	1.05	< 80%
Menhaden	Z / Z_{MUP}	1.05	0.81	1.34	< 55%
Bass	D_t / P_t	1.83	1.49	2.13	< 85%
Strategy		Menhaden target F, bass recover			
Bass	Biomass (mt)	260,000			
Menhaden	Harvest (mt)	226,000			
Menhaden	B / B_{MUP}	0.82	0.74	0.89	100%
Menhaden	Z / Z_{MUP}	1.19	1.00	1.42	< 95%
Bass	D_t / P_t	1.40	1.18	1.61	100%
Strategy		Menhaden at current harvest, bass D_t / P_t at threshold			
Bass	Biomass (mt)	215,000			
Menhaden	Harvest (mt)	175,000			
Menhaden	B / B_{MUP}	1.01	0.92	1.10	< 45%
Menhaden	Z / Z_{MUP}	0.99	1.10	1.21	< 60%
Bass	D_t / P_t	2.01	2.23	2.23	< 50%

Table 15. Symbols and terms used in the VADER model formulation.

Symbol	Definition
i	Species (used to designate prey species)
a	Age class (used to designate prey species age)
j	Predator species
b	Predator species age
t	Year
k	Fishery independent index
n	Number of indices
l	Vector of species-specific surveys
m	Month
$N_{i,a,t}$	January 1 abundance-at-age (10^6 fish)
$Z_{i,a,t}$	Instantaneous total mortality-at-age per year
$C_{i,a,t}$	Fishery catch-at-age (commercial and recreational harvest and dead discards, 10^6 fish)
$F_{i,a,t}$	Instantaneous fishing mortality-at-age per year
$s_{i,a}$	Fishery selectivity-at-age
$FIC_{i,a,t}$	Fishery independent catch (CPUE)
q_i	Fishery independent catchability
$r_{i,a}$	Fishery independent survey selectivity-at-age
$Sel_{x,i}$	Selectivity generated by logistic or double logistic functions
α_1, α_2, β_1, β_2	Logistic and double logistic ascending or descending limb parameters
$M_{i,a,t}$	Instantaneous natural mortality
$M0_{i,a}$	Residual natural mortality (time invariant)
$M2_{i,a,t}$	Instantaneous natural mortality due to predation
$W_{i,a,t}$	Average annual species-specific weight-at-age
$CB_{j,b}$	Consumption to biomass ratio (time invariant)
$B_{i,b,t}$	Biomass-at-age (10^6 kg)
$\phi_{i,a,j,b,t}$	Available prey biomass (10^6 kg)
$\tilde{\nu}_{i,a,j,b,t}$	Scaled prey suitability
$\nu_{i,a,j,b}$	Prey suitability
B_{eco}	Total ecosystem biomass (10^6 kg)
$\rho_{i,j}$	Prey species preference

Table 16 (Continued). Symbols and terms used in the VADER model formulation

Symbol	Definition
$g_{i,a,j,b}$	Predator size preference
η_j	Preferred predator to prey weight ratio
$B_{other,t}$	Total biomass of other food
$P_{i,a,t}$	Proportion-at-age
I	Dataset
LL_I	Log likelihood of dataset I
D_I	Objective function weighting for dataset I
TC	Total fishery catch (10^3 mt)
TS	Total survey catch (CPUE)
CP	Fishery catch age proportions
SP	Survey catch age proportions
FH	Food habits proportions
Pen_i	Total likelihood penalty for each species
Pwt_p	Objective function weighting for penalty p
Yr1pen	Year 1 abundance penalty
Rpen	Recruitment penalty
Bpen	Biomass penalty
Yr1	Year 1 abundance-at-age
Rthresh	Threshold value for the CV of log recruitment variability
Bthresh	Threshold value for age-specific biomass
Age1	Recruitment

Table 16. Components of the VADER model likelihood function by assumed distributions and including penalty functions for the VADER model. Small constants (10^{-3}) are added to the lognormal and multinomial calculations to keep the calculations from terminating if they reach zero.

Equation	Definition
$\mathrm{LL}_{\mathrm{Total}} = \mathrm{LL}_{\mathrm{TC}} + \mathrm{LL}_{\mathrm{TS}} + \mathrm{LL}_{\mathrm{CP}} + \mathrm{LL}_{\mathrm{SP}} + \mathrm{LL}_{\mathrm{FH}} + \sum_i Pen_i$	Total log likelihood
$LL_I = \sum_{t,i} \frac{1}{2} * ln\left(\frac{\hat{I} + 10^{-4}}{I + 10^{-4}}\right)^{ln(cv^2)}$	Lognormal distribution component
$LL_I = -\Gamma(nsamp * e^{dpar}) - \sum_{t,i,a} \Gamma\left((nsamp * I) + \left((nsamp * e^{dpar}) * \hat{I}\right)\right) + \sum_{t,i,a} \Gamma\left((nsamp * e^{dpar}) * \hat{I}\right)$	Dirichlet multinomial distribution component
$Pen_i = Pwt_{Yr1_i} * Yr1pen_i + Pwt_{Age1_i} * Rpen_i + Pwt_{B_i} * Bpen_i$	Total penalty
$Yr1pen_i = \sum_a (N_{i,a,t=1} - Yr1_{i,a})^2$	Year 1 penalty
$Rpen_i = 0.01 * (CV(N_{i,a=1,t}) - Rthresh_i)^2$	Recruitment penalty. Applied when the CV > *Rthresh*
$Bpen_i = \sum_{a,t} 0.01 * (B_{i,a,t} - Bthresh_i)^2$	Biomass penalty. Applied when *B* < *Bthresh*

Table 17. Indices used for each species for the Base and Alternate runs of the VADER model.

Bluefish		
	Used in Base run	Used in sensitivity run
NEFSC Albatross		X
MRIP CPUE	X	X
NC PSIGNS/P915	X	
Composite YOY	X	
Weakfish		
	Used in Base run	Used in sensitivity run
MRIP CPUE	X	
DE 30' Trawl	X	
NJ Ocean Trawl		X (offshore)
Composite YOY	X	
NC PSIGNS/P915		X (inshore)
Atlantic Herring		
	Used in Base run	Used in sensitivity run
Shrimp Survey		X
NEFSC Fall Albatross (1985-2008)	X	
NEFSC Fall Bigelow (2009-2017)	X	
Striped bass		
	Used in Base run	Used in sensitivity run
Composite YOY	X	
MD Spawning Stock		X
MRIP CPUE	X	
CT LISTS	X	
Atlantic menhaden		
	Used in Base run	Used in sensitivity run
SAD	X	X
MAD	X	X
NAD	X	X
Composite YOY	X	X
Spiny Dogfish		
	Used in Base run	Used in sensitivity run
NMFS Trawl (converted to Albatross units)	X	X

Table 18. Effective sample size and CVs for Atlantic menhaden catch and indices used in the VADER model.

	Atlantic menhaden									
	Catch		YOY		NAD		MAD		SAD	
Year	CV	Sample Size	CV	Sample Size	CV	Sample Size	CV	Sample Size	CV	Sample Size
1985	0.05	213	0.4	0	0	0	1.15	89	0	0
1986	0.05	146	0.4	0	0	0	1.14	89	0	0
1987	0.05	191	0.4	0	0	0	1.18	89	0	0
1988	0.05	185	0.4	0	0	0	1.14	0	0	0
1989	0.05	173	0.4	0	0	0	1.18	89	0	0
1990	0.05	171	0.4	0	0.4	18	1.15	0	0.4	22
1991	0.05	194	0.4	0	0.4	12	1.16	0	0.4	29
1992	0.05	142	0.4	0	0.4	24	1.16	0	0.4	23
1993	0.05	137	0.4	0	0.4	24	1.19	0	0.4	26
1994	0.05	132	0.4	0	0.4	18	1.14	0	0.4	10
1995	0.05	125	0.4	0	0.4	14	1.13	0	0.4	15
1996	0.05	116	0.4	0	0.4	34	1.2	0	0.4	21
1997	0.05	114	0.4	0	0.4	30	1.22	0	0.4	25
1998	0.05	115	0.4	0	0.4	18	0.4	89	0.4	29
1999	0.05	107	0.4	0	0.4	43	0.43	0	0.4	13
2000	0.05	92	0.4	0	0.4	30	0.36	89	0.4	12
2001	0.05	125	0.4	0	0.4	36	0.38	89	0.4	30
2002	0.05	110	0.4	0	0.4	51	0.43	89	0.4	32
2003	0.05	101	0.4	0	0.4	25	0.36	0	0.4	108
2004	0.05	115	0.4	0	0.4	48	0.38	0	0.4	47
2005	0.05	99	0.4	0	0.4	62	0.39	89	0.4	112
2006	0.05	105	0.4	0	0.4	33	0.42	89	0.4	134
2007	0.05	133	0.4	0	0.4	63	0.42	89	0.4	51
2008	0.05	111	0.4	0	0.4	52	0.45	89	0.4	527
2009	0.05	101	0.4	0	0.4	40	0.41	89	0.4	565
2010	0.05	111	0.4	0	0.4	25	0.39	89	0.4	554
2011	0.05	109	0.4	0	0.4	58	0.38	89	0.4	613
2012	0.05	93	0.4	0	0.4	60	0.44	89	0.4	610
2013	0.05	72	0.4	0	0.4	40	0.4	96	0.4	590
2014	0.05	89	0.4	0	0.4	58	0.38	96	0.4	621
2015	0.05	111	0.4	0	0.4	70	0.45	74	0.4	645
2016	0.05	108	0.4	0	0.4	50	0.43	86	0.4	527
2017	0.05	107	0.4	0	0.4	46	0.41	95	0.4	619

Table 19. Effective sample size and CVs for striped bass catch and indices used in the VADER model.

	Striped Bass							
	Catch		YOY		MRIP CPUE		CT LISTS	
Year	CV	Sample Size	CV	Sample Size	CV	Sample Size	CV	Sample Size
1985	0.1	70	0.4	0	0.4	36	0	0
1986	0.1	70	0.4	0	0.4	36	0	0
1987	0.1	70	0.4	0	0.4	36	0.4	12
1988	0.1	70	0.4	0	0.4	36	0.4	12
1989	0.1	70	0.4	0	0.4	36	0.4	12
1990	0.1	70	0.4	0	0.4	36	0.4	12
1991	0.1	70	0.4	0	0.4	36	0.4	12
1992	0.1	70	0.4	0	0.4	36	0.4	12
1993	0.1	70	0.4	0	0.4	36	0.4	12
1994	0.1	70	0.4	0	0.4	36	0.4	12
1995	0.1	70	0.4	0	0.4	36	0.4	12
1996	0.1	70	0.4	0	0.4	36	0.4	12
1997	0.1	70	0.4	0	0.4	36	0.4	12
1998	0.1	70	0.4	0	0.4	36	0.4	12
1999	0.1	70	0.4	0	0.4	36	0.4	12
2000	0.1	70	0.4	0	0.4	36	0.4	12
2001	0.1	70	0.4	0	0.4	36	0.4	12
2002	0.1	70	0.4	0	0.4	36	0.4	12
2003	0.1	70	0.4	0	0.4	36	0.4	12
2004	0.1	70	0.4	0	0.4	36	0.4	12
2005	0.1	70	0.4	0	0.4	36	0.4	12
2006	0.1	70	0.4	0	0.4	36	0.4	12
2007	0.1	70	0.4	0	0.4	36	0.4	12
2008	0.1	70	0.4	0	0.4	36	0.4	12
2009	0.1	70	0.4	0	0.4	36	0.4	12
2010	0.1	70	0.4	0	0.4	36	0.4	12
2011	0.1	70	0.4	0	0.4	36	0.4	12
2012	0.1	70	0.4	0	0.4	36	0.4	12
2013	0.1	70	0.4	0	0.4	36	0.4	12
2014	0.1	70	0.4	0	0.4	36	0.4	12
2015	0.1	70	0.4	0	0.4	36	0.4	12
2016	0.1	70	0.4	0	0.4	36	0.4	12
2017	0.1	70	0.4	0	0.4	36	0.4	12

Table 20. Effective sample size and CVs for bluefish catch and indices used in the VADER model.

	Bluefish							
	Catch		YOY		MRIP CPUE		NC PSIGNS	
Year	CV	Sample Size	CV	Sample Size	CV	Sample Size	CV	Sample Size
1985	0.15	24	0.71	0	0.11	30	0	0
1986	0.15	24	0.75	0	0.11	30	0	0
1987	0.15	24	0.57	0	0.1	30	0	0
1988	0.15	24	0.44	0	0.11	30	0	0
1989	0.15	24	0.43	0	0.1	30	0	0
1990	0.15	24	0.43	0	0.1	30	0	0
1991	0.15	24	0.43	0	0.1	30	0	0
1992	0.15	24	0.5	0	0.1	30	0	0
1993	0.15	24	0.48	0	0.09	30	0	0
1994	0.15	24	0.49	0	0.09	30	0	0
1995	0.15	24	0.44	0	0.09	30	0	0
1996	0.15	24	0.48	0	0.09	30	0	0
1997	0.15	12	0.47	0	0.1	10	0	0
1998	0.15	12	0.48	0	0.1	10	0	0
1999	0.15	12	0.53	0	0.1	10	0	0
2000	0.15	12	0.43	0	0.1	10	0	0
2001	0.15	12	0.42	0	0.1	10	0.13	10
2002	0.15	12	0.4	0	0.1	10	0.17	10
2003	0.15	12	0.43	0	0.1	10	0.15	10
2004	0.15	12	0.41	0	0.11	10	0.18	10
2005	0.15	48	0.38	0	0.1	45	0.17	20
2006	0.15	48	0.4	0	0.1	45	0.2	20
2007	0.15	48	0.41	0	0.1	45	0.15	20
2008	0.15	48	0.39	0	0.1	45	0.14	20
2009	0.15	48	0.44	0	0.1	45	0.14	20
2010	0.15	48	0.4	0	0.1	45	0.13	20
2011	0.15	48	0.42	0	0.1	45	0.19	20
2012	0.15	48	0.41	0	0.1	45	0.16	20
2013	0.15	48	0.4	0	0.11	45	0.19	20
2014	0.15	48	0.33	0	0.1	45	0.14	20
2015	0.15	48	0.42	0	0.1	45	0.15	20
2016	0.15	48	0.41	0	0.1	45	0.22	20
2017	0.15	48	0.45	0	0.1	45	0.17	20

Table 21. Effective sample size and CVs for weakfish catch and indices used in the VADER model.

	Weakfish							
	Catch		YOY		MRIP CPUE		DE30 Trawl	
Year	CV	Sample Size	CV	Sample Size	CV	Sample Size	CV	Sample Size
1985	0.12	50	0.45	0	0.3	25	0	0
1986	0.12	50	0.43	0	0.26	25	0	0
1987	0.18	50	0.37	0	0.29	25	0	0
1988	0.2	50	0.38	0	0.28	25	0	0
1989	0.12	50	0.33	0	0.33	25	0	0
1990	0.14	50	0.3	0	0.33	25	0	0
1991	0.12	50	0.3	0	0.32	25	0.62	25
1992	0.13	50	0.3	0	0.35	25	0.62	25
1993	0.12	50	0.34	0	0.31	25	0.62	25
1994	0.12	50	0.32	0	0.29	25	0.64	25
1995	0.11	50	0.34	0	0.27	25	0.62	25
1996	0.13	50	0.3	0	0.27	25	0.62	25
1997	0.11	50	0.32	0	0.27	25	0.63	25
1998	0.11	50	0.31	0	0.27	25	0.63	25
1999	0.11	50	0.3	0	0.28	25	0.62	25
2000	0.1	50	0.29	0	0.28	25	0.64	25
2001	0.12	50	0.3	0	0.3	25	0.63	25
2002	0.11	50	0.3	0	0.32	25	0.63	25
2003	0.15	50	0.31	0	0.36	25	0.63	25
2004	0.17	50	0.29	0	0.31	25	0.62	25
2005	0.12	50	0.3	0	0.33	25	0.62	25
2006	0.13	50	0.3	0	0.36	25	0.62	25
2007	0.13	50	0.3	0	0.45	25	0.62	25
2008	0.21	50	0.32	0	0.44	25	0.63	25
2009	0.14	50	0.29	0	0.57	25	0.62	25
2010	0.14	50	0.31	0	0.43	25	0.62	25
2011	0.15	50	0.3	0	0.43	25	0.62	25
2012	0.14	50	0.3	0	0.39	25	0.62	25
2013	0.13	50	0.3	0	0.46	25	0.62	25
2014	0.15	50	0.3	0	0.44	25	0.62	25
2015	0.16	50	0.31	0	0.4	25	0.62	25
2016	0.2	50	0.31	0	0.34	25	0.62	25
2017	0.13	50	0.3	0	0.48	25	0.62	25

Table 22. Effective sample size and CVs for Atlantic herring catch and indices used in the VADER model.

	Atlantic Herring					
	Catch		NEFSC Albatross		NEFSC Bigelow	
Year	CV	Sample Size	CV	Sample Size	CV	Sample Size
1985	0.1	15	0.95	0	0	0
1986	0.1	6	0.95	0	0	0
1987	0.1	8	0.79	16	0	0
1988	0.1	9	0.95	14	0	0
1989	0.1	9	0.91	15	0	0
1990	0.1	13	0.95	9	0	0
1991	0.1	10	0.95	14	0	0
1992	0.1	14	0.51	18	0	0
1993	0.1	13	0.88	17	0	0
1994	0.1	7	0.45	18	0	0
1995	0.1	8	0.73	22	0	0
1996	0.1	9	0.64	25	0	0
1997	0.1	10	0.72	20	0	0
1998	0.1	11	0.23	27	0	0
1999	0.1	10	0.41	22	0	0
2000	0.1	10	0.52	16	0	0
2001	0.1	10	0.56	16	0	0
2002	0.1	10	0.82	20	0	0
2003	0.1	10	0.95	20	0	0
2004	0.1	9	0.5	20	0	0
2005	0.1	10	0.54	16	0	0
2006	0.1	10	0.95	20	0	0
2007	0.1	10	0.41	21	0	0
2008	0.1	9	0.73	19	0	0
2009	0.1	8	0	0	0.95	9
2010	0.1	8	0	0	0.37	9
2011	0.1	8	0	0	0.63	9
2012	0.1	8	0	0	0.27	9
2013	0.1	9	0	0	0.4	8
2014	0.1	9	0	0	0.44	7
2015	0.1	8	0	0	0.3	9
2016	0.1	6	0	0	0.82	9
2017	0.1	7	0	0	0.71	6

Table 23. Effective sample size and CVs for spiny dogfish catch and indices used in the VADER model.

	Spiny Dogfish			
	Catch		NEFSC Albatross	
Year	CV	Sample Size	CV	Sample Size
1985	0.05	70	0.95	0
1986	0.05	70	0.95	0
1987	0.05	70	0.79	16
1988	0.05	70	0.95	14
1989	0.05	70	0.91	15
1990	0.05	70	0.95	9
1991	0.05	70	0.95	14
1992	0.05	70	0.51	18
1993	0.05	70	0.88	17
1994	0.05	70	0.45	18
1995	0.05	70	0.73	22
1996	0.05	70	0.64	25
1997	0.05	70	0.72	20
1998	0.05	70	0.23	27
1999	0.05	70	0.41	22
2000	0.05	70	0.52	16
2001	0.05	70	0.56	16
2002	0.05	70	0.82	20
2003	0.05	70	0.95	20
2004	0.05	70	0.5	20
2005	0.05	70	0.54	16
2006	0.05	70	0.95	20
2007	0.05	70	0.41	21
2008	0.05	70	0.73	19
2009	0.05	70	0.95	9
2010	0.05	70	0.37	9
2011	0.05	70	0.63	9
2012	0.05	70	0.27	9
2013	0.05	70	0.4	8
2014	0.05	70	0.44	7
2015	0.05	70	0.3	9
2016	0.05	70	0.82	9
2017	0.05	70	0.71	6

Table 24. Parameter estimates and standard deviations from the VADER model for predation interactions, average recruitment and average fishing mortality.

Parameter	Description	Estimate	Standard Deviation
iRho[1]	Predation Interaction Parameter – striped bass on menhaden	2.19	0.11
iRho[2]	Predation Interaction Parameter – bluefish on menhaden	0.37	0.43
iRho[3]	Predation Interaction Parameter – weakfish on menhaden	-0.05	0.54
iRho[4]	Predation Interaction Parameter – spiny dogfish on menhaden	0.93	0.17
iRho[5]	Predation Interaction Parameter – striped bass on weakfish	3.27	0.43
iRho[6]	Predation Interaction Parameter – bluefish on weakfish	-1.00	0.00
iRho[7]	Predation Interaction Parameter – spiny dogfish on menhaden	2.81	1.23
iRho[8]	Predation Interaction Parameter – striped bass on Atl herring	2.17	0.23
iRho[9]	Predation Interaction Parameter – bluefish on Atl herring	2.03	0.32
iRho[10]	Predation Interaction Parameter – weakfish on Atl herring	-0.94	2.48
iRho[11]	Predation Interaction Parameter – spiny dogfish on Atl herring	2.87	0.22
aAge1[1]	Average Recruitment Menhaden	10.61	0.05
aAge1[2]	Average Recruitment Striped Bass	4.60	0.04
aAge1[3]	Average Recruitment Bluefish	4.06	0.04
aAge1[4]	Average Recruitment Weakfish	4.53	0.12
aAge1[5]	Average Recruitment Herring	8.20	0.09
aAge1[6]	Average Recruitment Spiny Dogfish	3.62	0.05
aFt[1]	Average F Menhaden	-1.05	0.09
aFt[2]	Average F Striped Bass	-1.55	0.05
aFt[3]	Average F Bluefish	-0.97	0.06
aFt[4]	Average F Weakfish	-0.72	0.08
aFt[5]	Average F Herring	-0.65	0.14
aFt[6]	Average F Spiny Dogfish	-2.27	0.05

Table 25. Parameter estimates and standard deviations from the VADER model for initial abundance at age for Atlantic menhaden, striped bass, bluefish, and weakfish.

Parameter	Description	Estimate	Standard Deviation
iYr1[1]	Year 1 Menhaden Age 1	11.98	0.00
iYr1[1]	Year 1 Menhaden Age 2	10.38	0.00
iYr1[1]	Year 1 Menhaden Age 3	8.54	0.00
iYr1[1]	Year 1 Menhaden Age 4	6.31	0.00
iYr1[1]	Year 1 Menhaden Age 5	5.80	0.00
iYr1[1]	Year 1 Menhaden Age 6	4.35	0.01
iYr1[1]	Year 1 Menhaden Age 7	4.40	0.01
iYr1[2]	Year 1 Striped Bass Age 1	3.65	0.01
iYr1[2]	Year 1 Striped Bass Age 2	2.50	0.03
iYr1[2]	Year 1 Striped Bass Age 3	1.79	0.05
iYr1[2]	Year 1 Striped Bass Age 4	0.75	0.11
iYr1[2]	Year 1 Striped Bass Age 5	0.41	0.12
iYr1[2]	Year 1 Striped Bass Age 6	-0.06	0.14
iYr1[2]	Year 1 Striped Bass Age 7	-0.34	0.16
iYr1[2]	Year 1 Striped Bass Age 8	-1.79	0.34
iYr1[2]	Year 1 Striped Bass Age 9	-2.50	0.49
iYr1[2]	Year 1 Striped Bass Age 10	-2.32	0.45
iYr1[2]	Year 1 Striped Bass Age 11	-2.64	0.56
iYr1[2]	Year 1 Striped Bass Age 12	-2.59	0.56
iYr1[2]	Year 1 Striped Bass Age 13	-2.87	0.83
iYr1[2]	Year 1 Striped Bass Age 14	-2.48	0.92
iYr1[2]	Year 1 Striped Bass Age 15	-1.78	0.52
iYr1[3]	Year 1 Bluefish Age 1	4.19	0.01
iYr1[3]	Year 1 Bluefish Age 2	4.20	0.01
iYr1[3]	Year 1 Bluefish Age 3	3.31	0.01
iYr1[3]	Year 1 Bluefish Age 4	2.54	0.03
iYr1[3]	Year 1 Bluefish Age 5	2.26	0.04
iYr1[3]	Year 1 Bluefish Age 6	1.66	0.07
iYr1[3]	Year 1 Bluefish Age 7	3.05	0.02
iYr1[4]	Year 1 Weakfish Age 1	4.50	0.00
iYr1[4]	Year 1 Weakfish Age 2	3.88	0.01
iYr1[4]	Year 1 Weakfish Age 3	2.49	0.03
iYr1[4]	Year 1 Weakfish Age 4	1.87	0.05
iYr1[4]	Year 1 Weakfish Age 5	1.27	0.10
iYr1[4]	Year 1 Weakfish Age 6	0.65	0.18
iYr1[4]	Year 1 Weakfish Age 7	0.99	0.12

Table 26. Parameter estimates and standard deviations from the VADER model for initial abundance at age for Atlantic herring and spiny dogfish and Dirichlet parameters.

Parameter	Description	Estimate	Standard Deviation
iYr1[5]	Year 1 Herring Age 1	8.11	0.00
iYr1[5]	Year 1 Herring Age 2	8.05	0.00
iYr1[5]	Year 1 Herring Age 3	6.26	0.00
iYr1[5]	Year 1 Herring Age 4	5.79	0.00
iYr1[5]	Year 1 Herring Age 5	4.70	0.00
iYr1[5]	Year 1 Herring Age 6	3.97	0.01
iYr1[5]	Year 1 Herring Age 7	1.27	0.14
iYr1[5]	Year 1 Herring Age 8	2.91	0.03
iYr1[6]	Year 1 Spiny Dogfish Age 1	5.01	0.00
iYr1[6]	Year 1 Spiny Dogfish Age 2	4.52	0.01
iYr1[6]	Year 1 Spiny Dogfish Age 3	3.39	0.02
iYr1[6]	Year 1 Spiny Dogfish Age 4	2.31	0.05
iYr1[6]	Year 1 Spiny Dogfish Age 5	1.81	0.08
iYr1[6]	Year 1 Spiny Dogfish Age 6	1.71	0.08
iYr1[6]	Year 1 Spiny Dogfish Age 7	1.58	0.09
iYr1[6]	Year 1 Spiny Dogfish Age 8	1.51	0.10
iYr1[6]	Year 1 Spiny Dogfish Age 9	1.46	0.10
iYr1[6]	Year 1 Spiny Dogfish Age 10	1.42	0.11
iYr1[6]	Year 1 Spiny Dogfish Age 11	1.37	0.11
iYr1[6]	Year 1 Spiny Dogfish Age 12	1.33	0.12
iYr1[6]	Year 1 Spiny Dogfish Age 13	1.30	0.12
iYr1[6]	Year 1 Spiny Dogfish Age 14	1.27	0.13
iYr1[6]	Year 1 Spiny Dogfish Age 15	1.24	0.13
iYr1[6]	Year 1 Spiny Dogfish Age 16	1.22	0.13
iYr1[6]	Year 1 Spiny Dogfish Age 17	1.21	0.14
iYr1[6]	Year 1 Spiny Dogfish Age 18	1.21	0.14
iYr1[6]	Year 1 Spiny Dogfish Age 19	1.23	0.14
iYr1[6]	Year 1 Spiny Dogfish Age 20	1.25	0.14
iYr1[6]	Year 1 Spiny Dogfish Age 21	1.72	0.09
log_dm_Cac[1]	Dirichlet Parameter Catch at age Menhaden	-1.71	0.13
log_dm_Cac[2]	Dirichlet Parameter Catch at age Striped Bass	5.00	0.00
log_dm_Cac[3]	Dirichlet Parameter Catch at age Bluefish	5.00	0.00
log_dm_Cac[4]	Dirichlet Parameter Catch at age Weakfish	-1.08	0.15
log_dm_Cac[5]	Dirichlet Parameter Catch at age Herring	5.00	0.00
log_dm_Cac[6]	Dirichlet Parameter Catch at age Spiny Dogfish	-0.11	0.09
log_dm_Sac[1]	Dirichlet Parameter Survey at age Menhaden	-2.07	0.09
log_dm_Sac[2]	Dirichlet Parameter Survey at age Striped Bass	5.00	0.00
log_dm_Sac[3]	Dirichlet Parameter Survey at age Bluefish	-0.05	0.13
log_dm_Sac[4]	Dirichlet Parameter Survey at age Weakfish	-0.69	0.14
log_dm_Sac[5]	Dirichlet Parameter Survey at age Herring	5.00	0.00
log_dm_Sac[6]	Dirichlet Parameter Survey at age Spiny Dogfish	5.00	0.00

Table 27. Parameter estimates and standard deviations from the VADER model for fishery selectivity parameters.

Parameter	Description	Estimate	Standard Deviation
sel_params[1]	Fishery Selectivity Parameter – menhaden α1	2.50	0.00
sel_params[2]	Fishery Selectivity Parameter – menhaden β1	3.30	0.11
sel_params[3]	Fishery Selectivity Parameter – menhaden α2	5.00	0.00
sel_params[4]	Fishery Selectivity Parameter – menhaden β2	1.00	0.00
sel_params[5]	Fishery Selectivity Parameter – striped bass α1	5.15	0.06
sel_params[6]	Fishery Selectivity Parameter – striped bass β1	1.00	0.00
sel_params[7]	Fishery Selectivity Parameter – striped bass α2	14.72	326.94
sel_params[8]	Fishery Selectivity Parameter – striped bass β2	2.83	188.49
sel_params[9]	Fishery Selectivity Parameter – bluefish α1	1.17	0.08
sel_params[10]	Fishery Selectivity Parameter – bluefish β1	2.00	0.00
sel_params[11]	Fishery Selectivity Parameter – bluefish α2	6.08	0.38
sel_params[12]	Fishery Selectivity Parameter – bluefish β2	1.00	0.00
sel_params[13]	Fishery Selectivity Parameter – weakfish α1	3.32	0.09
sel_params[14]	Fishery Selectivity Parameter – weakfish β1	1.50	0.00
sel_params[15]	Fishery Selectivity Parameter – herring α1	3.31	0.20
sel_params[16]	Fishery Selectivity Parameter – herring β1	1.00	0.00
sel_params[17]	Fishery Selectivity Parameter – herring α2	7.00	0.00
sel_params[18]	Fishery Selectivity Parameter – herring β2	1.00	0.00
sel_params[19]	Fishery Selectivity Parameter – spiny dogfish α1	6.00	0.00
sel_params[20]	Fishery Selectivity Parameter – spiny dogfish β1	0.43	0.03

Table 28. Parameter estimates and standard deviations from the VADER model for survey selectivity parameters.

Parameter	Description	Estimate	Standard Deviation
FICsel_params[8]	Survey Selectivity Parameter – NAD α1	3.80	0.15
FICsel_params[9]	Survey Selectivity Parameter- NAD β1	2.18	0.13
FICsel_params[10]	Survey Selectivity Parameter – MAD α1	4.00	0.00
FICsel_params[11]	Survey Selectivity Parameter – MAD β1	2.56	0.09
FICsel_params[12]	Survey Selectivity Parameter – MAD α2	3.67	0.85
FICsel_params[13]	Survey Selectivity Parameter – MAD β2	3.50	0.01
FICsel_params[14]	Survey Selectivity Parameter – SAD α1	1.32	0.01
FICsel_params[15]	Survey Selectivity Parameter – SAD β1	6.50	0.00
FICsel_params[16]	Survey Selectivity Parameter – SAD α2	2.00	0.00
FICsel_params[17]	Survey Selectivity Parameter – SAD β2	1.86	0.23
FICsel_params[33]	Survey Selectivity Parameter – MRIP α1	4.18	0.08
FICsel_params[34]	Survey Selectivity Parameter - MRIP β1	1.00	0.00
FICsel_params[35]	Survey Selectivity Parameter – CTLIST α1	4.00	0.00
FICsel_params[36]	Survey Selectivity Parameter – CTLIST β1	0.77	0.05
FICsel_params[37]	Survey Selectivity Parameter – CTLIST α2	13.00	0.00
FICsel_params[38]	Survey Selectivity Parameter – CTLIST β2	1.00	0.00
FICsel_params[46]	Survey Selectivity Parameter – MRIP α1	2.00	0.00
FICsel_params[47]	Survey Selectivity Parameter - MRIP β1	5.00	0.00
FICsel_params[48]	Survey Selectivity Parameter – PSIGNS α1	2.00	0.00
FICsel_params[49]	Survey Selectivity Parameter – PSIGNS β1	5.00	0.00
FICsel_params[50]	Survey Selectivity Parameter – PSIGNS α2	3.21	0.31
FICsel_params[51]	Survey Selectivity Parameter – PSIGNS β2	2.00	0.00
FICsel_params[59]	Survey Selectivity Parameter – MRIP α1	3.58	0.13
FICsel_params[60]	Survey Selectivity Parameter - MRIP β1	1.50	0.00
FICsel_params[61]	Survey Selectivity Parameter – DE30 α1	1.76	0.15
FICsel_params[62]	Survey Selectivity Parameter – DE30 β1	1.50	0.00
FICsel_params[63]	Survey Selectivity Parameter – Alb α1	5.10	0.29
FICsel_params[64]	Survey Selectivity Parameter – Alb β1	1.00	0.00
FICsel_params[65]	Survey Selectivity Parameter – Bigelow α1	4.56	0.54
FICsel_params[66]	Survey Selectivity Parameter – Bigelow β1	1.00	0.00
FICsel_params[67]	Survey Selectivity Parameter – Alb α1	6.00	0.00
FICsel_params[68]	Survey Selectivity Parameter – Alb β1	0.43	0.04

Table 29. Contributions of the various components by species to the VADER model objective function value

Likelihood component	Menhaden	Striped Bass	Bluefish	Weakfish	Atlantic Herring	Spiny Dogfish
Total fishery catch	1.5	1.2	7.6	111.3	0.4	43.2
Total survey catch	34.2 30.8 252.9 54.3	12.3 27.5 29.6	13.1 20.9 11.3	97.2 47.0 25.5	17.3 3.1	23.2
Fishery catch age proportions	5,004.3	5,533.0	1,716.8	2,554.2	546.5	7,162.1
Survey catch age proportions	1,681.2 2,796.7 5,345.2	2,667.3 760.0	2,058.2 453.3	1,249.5 879.2	708.3 115.7	1,587.2
Food habits	0	487.0	98.8	75.8	0	326.0
Year 1 penalty	3.0e-005	18.6	0.1	6.4	6.9e-005	147.7
Recruitment penalty	0	0	0	5.9	0	0.01
Biomass penalty	0	0	0	0	0	0
Total Likelihood Value	15,201	9,518.23	4,380.2	5,039.9	1,391.4	9,141.8

Table 30. Ecopath inputs representing the base year of 1985 for the NWACS-MICE model. BA rate is biomass accumulation rate and *Q/B* is consumption per biomass. For multistanza groups, the adult age stanza is considered to be age 1+ unless otherwise noted.

n	Group name	Biomass (1e6 mt)	BA rate (/year)	Z or PB (/year)	Q/B (/year)	Total Landings (1e6 mt)
1	striped bass 0-1	0.008	0.113	1.132	7.152	
2	striped bass 2-5	0.036	0.113	0.582	3.004	0.001
3	striped bass 6+	0.018	0.113	0.335	1.820	0.003
4	menhaden juv	0.282	0.114	1.764	9.402	0.005
5	menhaden adult	1.704	0.114	1.454	3.804	0.329
6	spiny dogfish	0.272	0.000	0.321	1.810	0.005
7	bluefish juv	0.004	-0.064	2.069	12.331	0.001
8	bluefish adult	0.220	-0.064	0.656	3.139	0.032
9	weakfish juv	0.001	0.000	1.453	9.977	
10	weakfish adult	0.013	0.000	1.310	3.770	0.003
11	Atlantic herring 0-1	0.008	0.137	1.371	10.829	0.002
12	Atlantic herring 2+	0.150	0.137	0.823	3.700	0.059
13	anchovies	0.271	0.000	2.200	7.333	
14	benthos	14.546	0.000	2.432	12.469	
15	zooplankton	13.559	0.000	45.850	154.600	
16	phytoplankton	8.596	0.000	186.436		
17	Detritus	12.974	0.000			

Table 31. Diet matrix for the NWACS-MICE model with columns as predators and rows as prey. The numbers in the column and row headings correspond to the groups listed in Table 30. Imp = diet import.

Pred/ Prey	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
1		0.001	0.002			0.001	0.001	0.000							
2			0.001												
3															
4	0.020	0.078	0.076			0.003	0.009	0.027	0.013	0.025					
5	0.020	0.079	0.228			0.002		0.049		0.018					
6						0.001									
7		0.001	0.001					0.010							
8			0.001					0.001							
9	0.002	0.004					0.001	0.001	0.001	0.006					
10								0.001		0.001					
11		0.002	0.011			0.008		0.003			0.003	0.002			
12		0.022	0.051			0.062		0.029			0.003	0.004			
13	0.112	0.232	0.254			0.007	0.444	0.212	0.436	0.445	0.027	0.027			
14	0.514	0.353	0.101			0.218	0.024	0.032	0.204	0.169	0.177	0.177	0.101	0.090	0.001
15	0.146	0.016	0.002	0.420	0.570	0.234	0.011	0.009	0.238	0.164	0.784	0.784	0.684	0.021	0.261
16		0.001	0.003	0.301	0.223			0.002	0.001	0.000			0.155	0.229	0.490
17				0.278	0.206								0.060	0.413	0.199
Imp.	0.186	0.213	0.269	0.000		0.465	0.510	0.624	0.107	0.172	0.006	0.006		0.246	0.048

Table 32. Ecopath estimates of trophic level, ecotrophic efficiency, and mortality rates from the NWACS-MICE model.

n	Group name	Trophic level	Ecotrophic Efficiency	Fishing mortality	Predation Mortality	Other Mortality
1	striped bass 0	3.307	0.089	0.000	0.101	1.031
2	striped bass 2-5	3.540	0.037	0.020	0.001	0.561
3	striped bass 6+	3.787	0.511	0.171	0.000	0.164
4	menhaden juv	2.562	0.080	0.019	0.121	1.623
5	menhaden adult	2.762	0.154	0.193	0.031	1.230
6	spiny dogfish	3.385	0.063	0.019	0.001	0.301
7	bluefish juv	3.959	0.855	0.173	1.596	0.300
8	bluefish adult	3.906	0.229	0.148	0.002	0.506
9	weakfish juv	3.624	0.919	0.022	1.313	0.118
10	weakfish adult	3.686	0.231	0.222	0.080	1.008
11	Atlantic herring 0-1	3.320	0.834	0.248	0.895	0.228
12	Atlantic herring 2+	3.320	0.938	0.395	0.377	0.051
13	anchovies	3.027	0.433	0.000	0.952	1.248
14	benthos	2.108	0.521	0.000	1.266	1.165
15	zooplankton	2.337	0.899	0.000	41.207	4.643
16	phytoplankton	1.000	0.669	0.000	124.635	61.802
17	Detritus	1.000	0.370		0.000	0.000

Table 33. Predation mortality matrix from the NWACS-MICE model with columns as predators and rows as prey. The numbers in the column and row headings correspond to the groups listed in Table 30.

Pred/ Prey	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
1		0.0126	0.0079			0.0392	0.0064	0.0347							
2			0.0009												
3															
4	0.0042	0.0301	0.0091			0.0046	0.0017	0.0669	0.0006	0.0043					
5	0.0007	0.0050	0.0045			0.0006		0.0200		0.0005					
6						0.0012									
7		0.0246	0.0077					1.5634							
8			0.0002					0.0022							
9	0.0979	0.3474					0.0437	0.5916	0.0101	0.2220					
10								0.0761		0.0041					
11		0.0225	0.0435			0.4606		0.2356			0.0279	0.1047			
12		0.0158	0.0114			0.2029		0.1320			0.0015	0.0132			
13	0.0249	0.0927	0.0315			0.0134	0.0872	0.5395	0.0196	0.0785	0.0090	0.0556			
14	0.0021	0.0026	0.0002			0.0074	0.0001	0.0015	0.0002	0.0006	0.0011	0.0067	0.0138	1.1258	0.1041
15	0.0006	0.0001	0.0000	0.0821	0.2725	0.0085	0.0000	0.0004	0.0002	0.0006	0.0052	0.0320	0.1004	0.2847	40.4197
16		0.0000	0.0000	0.0928	0.1685			0.0001	0.0000	0.0000			0.0359	4.8343	119.5028

Table 34. Time series of abundance and catch used in the Ecosim component of the NWACS-MICE model.

Group Name	Index Type	Index Name	Years	Weight
striped bass 0	rel. biomass	composite YOY index	1985-2017	3.476
striped bass 2-5	rel. biomass	recreational cpue	1985-2017	2.015
striped bass 2-5	rel. biomass	Connecticut Long Island Sound trawl survey	1987-2017	2.686
striped bass 6+	rel. biomass	recreational cpue	1985-2017	2.015
striped bass 6+	rel. biomass	Connecticut Long Island Sound trawl survey	1987-2017	2.686
menhaden juv	rel. biomass	composite YOY index	1985-2017	1.981
menhaden adult	rel. biomass	composite northern adult index	1990-2017	1.476
menhaden adult	rel. biomass	composite mid Atlantic adult index	1985-2017	1.419
menhaden adult	rel. biomass	composite southern adult index	1990-2017	1.787
spiny dogfish	rel. biomass	NEFSC trawl survey	1985-2017	2.500
bluefish juv	rel. biomass	composite YOY index	1985-2017	2.202
bluefish adult	rel. biomass	recreational cpue	1985-2017	9.925
bluefish adult	rel. biomass	NC Pamlico Sound inshore gillnet survey	2001-2017	6.082
weakfish juv	rel. biomass	composite YOY index	1985-2017	3.137
weakfish adult	rel. biomass	recreational cpue	1985-2017	2.867
weakfish adult	rel. biomass	Delaware 30' trawl survey	1991-2017	1.603
Atlantic herring 2+	rel. biomass	NEFSC Fall survey Albatross	1985-2008	1.412
Atlantic herring 2+	rel. biomass	NEFSC Fall survey Bigelow	2009-2017	1.842
striped bass 2-5	catch	total landings from stock assessment	1985-2017	12.655
striped bass 6+	catch	total landings from stock assessment	1985-2017	12.655
menhaden adult	catch	total landings from stock assessment	1985-2017	21.666
bluefish adult	catch	total landings from stock assessment	1985-2017	6.938
weakfish juv	catch	total landings from stock assessment	1985-2017	7.458
weakfish adult	catch	total landings from stock assessment	1985-2017	7.458
Atlantic herring 2+	catch	total landings from stock assessment	1985-2017	10.000
menhaden juv	rel. catch	total landings from stock assessment	1985-2017	21.666
bluefish juv	rel. catch	total landings from stock assessment	1985-2017	6.938
Atlantic herring 0-1	rel. catch	total landings from stock assessment	1985-2017	10.000
spiny dogfish	forced catch	total landings from stock assessment	1985-2017	1.000

Table 35. Ecosim scenarios for the NWACS-MICE model.

Scenario	Description	Start SS	End SS
sim1	Default configuration with foraging time adjustment set to 0.5 for all juvenile stanzas and prey switching = 0.	2582	1200
sim1.1	sim 1 with vulnerability caps, M_0 changing with foraging time, and foraging time changing with predator abundance for juvenile striped bass	2582	1269
sim3	Aimed at evaluating the sensitivity to prey switching (parameter set at 1, when default value is 0).	2590	1088
sim3.5	Sim3 with vulnerability caps applied and juvenile striped bass with risk sensitive foraging and M_0 constant relative to foraging time.	2590	1186
sim9	fit 17 vulnerabilities and 11 PP splines	2582	1031
sim9.1	fit 28 vulnerabilities and 33 PP splines	2582	1096
sim 12.3	Fit to time series with recruitment deviation, prey switching 1.5 for menhaden predators	2461	1062

Table 36. Equilibrium F_{MSY} values from the NWACS-MICE model.

Group	sim1	sim1.1	sim3	sim3.5
striped bass ages 6+	0.171	0.154	0.305	0.171
menhaden adult	0.657	0.637	0.954	0.837
bluefish adult	0.856	0.723	NA	0.729
weakfish adult	NA	NA	1.097	0.794
Atlantic herring 2+	0.237	0.237	0.395	0.335

Table 37. Biomass and fishing mortality reference points from single species stock assessments with conversions for sim 3.5 of the NWACS-MICE model.

Biomass Reference Points								
Species	**Single Species B_{2017}**	**Single Species B_{TARGET}**	**Single Species $B_{THRESHOLD}$**	**B_{TARGET}/ B_{2017}**	**$B_{THRESHOLD}$/ B_{2017}**	**Ecosim B_{2017}**	**Ecosim B_{TARGET}**	**Ecosim $B_{THRESHOLD}$**
Striped bass (age 6+)	97,046	153,244	119,722	1.58	1.23	0.110	0.174	0.136
Menhaden (adult)	3,581,000	NA	NA	NA	NA	2.344	NA	NA
Spiny dogfish	641,132	511,776	255,888	0.80	0.40	0.314	0.251	0.125
Bluefish (adult)	92,794	198,717	99,359	2.14	1.07	0.104	0.223	0.112
Weakfish (adult)	3,209	NA	11,489	NA	3.58	0.007	NA	0.026
Atlantic herring (age 2+)	239,472	448,000	224,000	1.87	0.94	0.453	0.848	0.424
Fishing Mortality Reference Points								
Species	**Single Species F_{2017}**	**Single Species F_{TARGET}**	**Single Species $F_{THRESHOLD}$**	**F_{TARGET}/ F_{2017}**	**$F_{THRESHOLD}$/ F_{2017}**	**Ecosim F_{2017}**	**Ecosim F_{TARGET}**	**Ecosim $F_{THRESHOLD}$**
Striped bass (age 6+)	0.310	0.197	0.240	0.635	0.774	0.294	0.187	0.228
Menhaden (adult)	0.110	0.220	0.600	2.000	5.455	0.049	0.098	0.267
Spiny dogfish	0.150	0.220	0.240	1.467	1.600	0.035	0.052	0.056
Bluefish (adult)	0.340	0.160	0.320	0.471	0.941	0.384	0.181	0.361
Weakfish (adult)	0.230	0.550	0.930	2.391	4.043	0.069	0.165	0.278
Atlantic herring (age 2+)	0.450	0.460	0.510	1.022	1.133	0.283	0.290	0.321

Table 38. Proportion of trials with change in biomass (*ΔB*REL) at or below a given percentage and median *ΔB*REL from 500 Ecosim projections for each *F* scenario from the NWACS-MICE model.

	Years out	-20%	-15%	-10%	-5%	0%	5%	10%	15%	20%	Median *ΔB*REL
Atlantic menhaden *F=Current TAC*											
striped.bass.adult	4 yr	0.00	0.00	0.00	0.00	0.68	1.00	1.00	1.00	1.00	-0.01
	40 yr	0.00	0.00	0.00	0.16	0.70	0.98	1.00	1.00	1.00	-0.02
spiny.dogfish	4 yr	0.00	0.00	0.00	0.00	0.61	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.30	1.00	1.00	1.00	1.00	0.00
bluefish.adult	4 yr	0.00	0.00	0.00	0.00	0.67	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.70	1.00	1.00	1.00	1.00	-0.01
weakfish.adult	4 yr	0.00	0.00	0.00	0.00	0.71	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.86	1.00	1.00	1.00	1.00	0.00
Atl.herring.adult	4 yr	0.00	0.00	0.00	0.00	0.70	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.30	0.99	1.00	1.00	1.00	0.01
Atlantic menhaden *F=F*TARGET											
striped.bass.adult	4 yr	0.00	0.00	0.00	0.02	0.86	1.00	1.00	1.00	1.00	-0.03
	40 yr	0.00	0.00	0.10	0.57	0.88	1.00	1.00	1.00	1.00	-0.06
spiny.dogfish	4 yr	0.00	0.00	0.00	0.00	0.76	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.12	1.00	1.00	1.00	1.00	0.01
bluefish.adult	4 yr	0.00	0.00	0.00	0.00	0.84	1.00	1.00	1.00	1.00	-0.01
	40 yr	0.00	0.00	0.00	0.00	0.88	1.00	1.00	1.00	1.00	-0.01
weakfish.adult	4 yr	0.00	0.00	0.00	0.00	0.88	1.00	1.00	1.00	1.00	-0.01
	40 yr	0.00	0.00	0.00	0.00	0.89	1.00	1.00	1.00	1.00	0.00
Atl.herring.adult	4 yr	0.00	0.00	0.00	0.00	0.88	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.12	0.73	1.00	1.00	1.00	0.04
Atlantic menhaden *F=F*THRESHOLD											
striped.bass.adult	4 yr	0.00	0.12	0.58	0.86	1.00	1.00	1.00	1.00	1.00	-0.11
	40 yr	0.56	0.75	0.90	0.97	1.00	1.00	1.00	1.00	1.00	-0.21
spiny.dogfish	4 yr	0.00	0.00	0.00	0.00	0.99	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.00	0.66	1.00	1.00	1.00	0.04
bluefish.adult	4 yr	0.00	0.00	0.00	0.56	1.00	1.00	1.00	1.00	1.00	-0.05
	40 yr	0.00	0.00	0.00	0.39	1.00	1.00	1.00	1.00	1.00	-0.05
weakfish.adult	4 yr	0.00	0.00	0.00	0.05	1.00	1.00	1.00	1.00	1.00	-0.03
	40 yr	0.00	0.00	0.00	0.00	0.99	1.00	1.00	1.00	1.00	-0.01
Atl.herring.adult	4 yr	0.00	0.00	0.00	0.00	1.00	1.00	1.00	1.00	1.00	0.00
	40 yr	0.00	0.00	0.00	0.00	0.00	0.07	0.27	0.54	0.90	0.14

Table 39. Ecosystem model trophic groups used in the NWACS-FULL model. Trophic groups are arranged by node number and arranged into broader categories. Eight species were modeled using multi-stanza groups that were identified as small (S), medium (M), or large (L), and the defining ages (in years) and fork lengths (cm) for each of the eight species is listed.

		Small		Medium		Large	
Node	Node Name	age	size	age	size	age	size
Primary Producers							
1	Phytoplankton						
2	Other primary producers						
Bacteria							
3	Bacteria						
Zooplankton							
4	Microzooplankton						
5	Small copepods						
6	Large copepods						
7	Gelatinous zooplankton						
8	Micronekton						
Benthic Invertebrates							
9	Macrobenthos - polychaetes						
10	Macrobenthos - crustaceans						
11	Macrobenthos - molluscs						
12	Macrobenthos - other						
13	Megabenthos - Filterers						
14	Megabenthos - other						
15	Shrimp and Similar Species						
Forage Fishes							
16	Mesopelagics						
17	Atlantic herring						
18	Alosines						
19-21	Atlantic menhaden	0	<14	1-2	14-24	3+	>24
22	Anchovies						
23	Atlantic mackerel						
24	Squid						
25	Butterfish						
26	small pelagic - other						
Fishes							
27-29	Bluefish	0	<30	1-3	30-60	4+	>60
30-32	Striped bass	0-1	<25	2-6	25-70	7+	>70
33-35	Weakfish	0	<20	1-2	20-40	3+	>40
36-37	Spiny dogfish	0-5	<60			6+	>60
38-40	Cod	0-1	<20	2-3	20-50	4+	>50
41	Haddock						
42	Hakes						
43	Atlantic croaker						
44-45	Yellowtail flounder	0	<20			1+	>20
46-47	Summer flounder	0	<25			1+	>25
48	Skates						
49	Demersal benthivores - other						
50	Demersal piscivores - other						
51	Demersal omnivores - other						
52	Medium pelagic - other						
Apex Predators							
53	Sharks - coastal						
54	Sharks - pelagic						
55	Large pelagics (HMS)						
56	Pinnipeds						
57	Baleen whales						
58	Odontocetes						
59	Seabirds						
60	Shorebirds - piscivorous						
Detritus							
61	Detritus						

Table 40. Basic inputs and outputs for Sim2 of the NWACS-FULL model. Values include biomass accumulation (BA) rates, total instantaneous mortality (*Z*) or production to biomass (*P/B*), consumption to biomass (*Q/B*), trophic level (*TL*), ecotrophic efficiency (*EE*), fishing mortality rate (*F*), predation mortality rate (M_2), and other mortality (M_0).

N	Group	Biomass (mt/km2)	BA rate (/yr)	Z or P/B (/year)	Q/B (/yr)	Total Landings (mt/km2)	TL	EE	F	M_2	M_0
1	Phytoplankton	30.000	0.000	180.700	0.000	0.000	1.000	0.943		170.357	10.343
2	Other primary producers	1.621	0.000	55.570	0.000	0.009	1.000	0.900	0.006	50.007	5.557
3	Bacteria	7.700	0.000	91.250	380.208	0.000	2.000	0.941		85.867	5.383
4	Microzooplankton	7.000	0.000	85.000	283.400	0.000	2.264	0.934		79.351	5.649
5	Small copepods	16.000	0.000	46.000	140.000	0.000	2.152	0.992		45.624	0.376
6	Large copepods	17.966	0.000	46.000	150.000	0.000	2.388	0.978		44.994	1.006
7	Gelatinous zooplankton	6.349	0.000	40.000	145.326	0.000	3.085	0.590		23.612	16.388
8	Micronekton	7.654	0.000	14.250	85.497	0.000	2.723	0.675		9.612	4.638
9	Macrobenthos - polychaetes	17.452	0.000	2.500	17.500	0.002	2.377	0.800	0.000	2.001	0.499
10	Macrobenthos - crustaceans	7.000	0.000	3.600	21.000	0.000	2.535	0.763		2.747	0.853
11	Macrobenthos - molluscs	8.340	0.000	2.200	13.949	0.275	2.246	0.835	0.033	1.805	0.362
12	Macrobenthos - other	21.000	0.000	2.000	16.059	0.000	2.349	0.865	0.000	1.731	0.269
13	Megabenthos - filterers	5.500	0.000	1.200	6.660	0.041	2.120	0.868	0.007	1.034	0.158
14	Megabenthos - other	4.498	0.000	2.300	15.533	0.350	2.895	0.739	0.078	1.622	0.600
15	Shrimp and Similar Species	0.470	0.144	2.000	6.660	0.021	2.751	0.833	0.046	1.477	0.333
16	Mesopelagics	0.090	0.000	1.100	3.700	0.000	3.238	0.961		1.057	0.043
17	Atlantic herring	0.800	0.000	1.700	5.300	0.285	3.495	0.952	0.357	1.262	0.081
18	Alosines	0.200	0.000	1.300	4.400	0.025	3.367	0.876	0.123	1.016	0.161
19	Atlantic menhaden (S)	1.340	0.234	1.766	15.860	0.018	2.533	0.151	0.013	0.252	1.500
20	Atlantic menhaden (M)	5.562	0.234	1.498	6.993	1.194	2.685	0.200	0.215	0.085	1.198
21	Atlantic menhaden (L)	1.135	0.234	1.229	4.160	0.418	2.837	0.453	0.368	0.189	0.672
22	Anchovies	1.100	0.000	2.200	7.333	0.000	3.060	0.973		2.141	0.059
23	Atlantic mackerel	1.740	0.000	0.550	2.170	0.052	3.546	0.786	0.030	0.402	0.118
24	Squid	1.267	0.407	5.720	19.000	0.042	3.859	0.958	0.033	5.041	0.239
25	Butterfish	1.488	0.020	1.312	4.230	0.024	3.833	0.888	0.079	1.066	0.147
26	Small pelagic - other	1.400	0.000	1.200	4.000	0.004	3.397	0.910	0.003	1.089	0.108
27	Bluefish (S)	0.015	-0.173	2.500	17.977	0.008	4.435	0.907	0.537	1.730	0.233
28	Bluefish (M)	0.257	-0.173	0.893	5.786	0.102	4.406	0.585	0.397	0.125	0.371
29	Bluefish (L)	0.618	-0.173	0.461	3.139	0.150	4.498	0.606	0.243	0.036	0.182

Table 40 continued. Basic inputs and outputs for Sim2 of the NWACS-FULL model.

N	Group	Biomass (mt/km2)	BA rate (/yr)	Z or P/B (/year)	Q/B (/yr)	Total Landings (mt/km2)	TL	EE	F	M_2	M_0
30	Striped bass (S)	0.018	0.011	1.130	7.543	0.000	3.795	0.307		0.347	0.783
31	Striped bass (M)	0.126	0.011	0.536	2.973	0.005	3.934	0.100	0.037	0.017	0.483
32	Striped bass (L)	0.071	0.011	0.325	1.820	0.012	4.072	0.558	0.176	0.005	0.144
33	Weakfish (S)	0.006	-0.083	2.800	13.520	0.000	3.923	0.849		2.378	0.422
34	Weakfish (M)	0.055	-0.083	0.913	4.869	0.001	3.977	0.398	0.022	0.341	0.550
35	Weakfish (L)	0.082	-0.083	0.845	2.813	0.037	4.063	0.533	0.449	0.001	0.395
36	Spiny dogfish (S)	0.303	0.050	0.200	3.600	0.000	4.188	0.719		0.144	0.056
37	Spiny dogfish (L)	1.200	0.050	0.200	1.810	0.025	4.263	0.296	0.038	0.022	0.141
38	Atlantic cod (S)	0.055	-0.228	1.087	5.059	0.010	3.630	0.910	0.174	0.815	0.098
39	Atlantic cod (M)	0.144	-0.228	1.125	2.603	0.132	3.948	0.936	0.920	0.133	0.073
40	Atlantic cod (L)	0.277	-0.228	0.700	1.500	0.122	4.318	0.658	0.441	0.020	0.239
41	Haddock	0.254	0.000	0.700	3.000	0.082	3.634	0.924	0.323	0.325	0.053
42	Hake	1.000	0.000	1.296	3.850	0.109	4.164	0.869	0.109	1.017	0.169
43	Atlantic croaker	0.350	0.000	0.994	3.550	0.027	3.569	0.298	0.079	0.218	0.698
44	Yellowtail flounder (S)	0.007	0.000	2.700	12.168	0.000	3.569	0.975		2.633	0.067
45	Yellowtail flounder (L)	0.187	0.000	0.850	2.900	0.085	3.536	0.658	0.457	0.102	0.291
46	Summer flounder (S)	0.011	0.119	2.400	10.379	0.009	4.206	0.873	0.837	1.258	0.304
47	Summer flounder (L)	0.159	0.119	1.050	2.900	0.084	4.516	0.543	0.525	0.044	0.480
48	Skates	1.000	0.000	0.250	0.900	0.011	3.805	0.807	0.011	0.191	0.048
49	Demersal benthivores - other	2.300	0.000	0.600	2.000	0.119	3.555	0.977	0.052	0.535	0.014
50	Demersal piscivores - other	1.300	0.000	0.450	1.500	0.089	4.079	0.747	0.068	0.268	0.114
51	Demersal omnivores - other	1.100	0.000	0.550	1.833	0.101	3.885	0.991	0.092	0.453	0.005
52	Medium pelagic - other	0.021	0.000	0.450	1.838	0.001	4.707	0.658	0.056	0.240	0.154
53	Sharks - coastal	0.008	0.000	0.200	1.247	0.001	4.601	0.564	0.099	0.014	0.087
54	Sharks - pelagic	0.016	0.000	0.113	0.690	0.000	4.644	0.194	0.003	0.019	0.091
55	Large pelagics (HMS)	0.070	0.000	0.579	6.794	0.027	4.494	0.671	0.386	0.003	0.191
56	Pinnipeds	0.035	0.000	0.075	5.581	0.000	4.530	0.118		0.009	0.066
57	Baleen whales	0.464	0.000	0.040	3.217	0.000	3.541	0.012		0.000	0.040
58	Odontocetes	0.060	0.000	0.040	14.301	0.000	4.611	0.922		0.037	0.003
59	Seabirds	0.007	0.000	0.279	80.000	0.000	4.264	0.373		0.104	0.175
60	Shorebirds - piscivorous	0.007	0.000	0.279	80.000	0.000	3.997	0.005		0.001	0.278
61	Detritus	52.600	0.000	0.000	0.000	0.000	1.000	0.871			

Table 41. Diet composition matrix for Sim2 of the NWACS-FULL model. Columns indicate the predators (labeled by node number) and rows are prey. (Page 1 of 3)

Node	Prey \ predator	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	Phytoplankton	0.243	0.225	0.744	0.668	0.107	0.233	0.131	0.177	0.424	0.236	0.692		0.067	0.026			0.278	0.221
2	Other primary producers	0.023						0.015	0.012	0.010	0.013	0.006						0.023	0.018
3	Bacteria		0.196			0.025		0.313	0.166	0.201	0.244	0.120	0.270	0.397	0.015				
4	Microzooplankton		0.054	0.111	0.060	0.031									0.075			0.140	0.180
5	Small copepods			0.011	0.114	0.303	0.149		0.015						0.439	0.399	0.601	0.140	0.180
6	Large copepods				0.065	0.432	0.323		0.033						0.429			0.140	0.180
7	Gelatinous zooplankton				0.042	0.035										0.021	0.002		
8	Micronekton						0.059	0.015	0.014	0.009	0.019			0.134	0.013	0.365	0.227		
9	Macrobenthos - polychaetes							0.005	0.099		0.021		0.133			0.001	0.002		
10	Macrobenthos - crustaceans				0.001			0.003	0.001		0.001		0.046	0.026		0.139	0.134		
11	Macrobenthos - molluscs							0.001	0.010	0.001	0.011		0.110			0.002			
12	Macrobenthos - other				0.001			0.014	0.084	0.011	0.011		0.146	0.061		0.001	0.005		
13	Megabenthos - filterers							0.003	0.014	0.010	0.001		0.012						
14	Megabenthos - other							0.001	0.002	0.001	0.007		0.012			0.003	0.002		
15	Shrimp and Similar Species													0.001		0.030	0.005		
16	Mesopelagics																		
17	Atlantic herring															0.005			
18	Alosines																		
19	Atlantic menhaden (S)																		
20	Atlantic menhaden (M)																		
21	Atlantic menhaden (L)																		
22	Anchovies															0.027	0.003		
23	Atlantic mackerel																		
24	Squid					0.000													
25	Butterfish																		
26	Small pelagic - other															0.004	0.019		
27	Bluefish (S)																		
28	Bluefish (M)																		
29	Bluefish (L)																		
30	Striped bass (S)																		
31	Striped bass (M)																		
32	Striped bass (L)																		
33	Weakfish (S)																		
34	Weakfish (M)																		
35	Weakfish (L)																		
36	Spiny dogfish (S)																		
37	Spiny dogfish (L)																		
38	Atlantic cod (S)																		
39	Atlantic cod (M)																		
40	Atlantic cod (L)																		
41	Haddock																		
42	Hake															0.001			
43	Atlantic croaker																		
44	Yellowtail flounder (S)																		
45	Yellowtail flounder (L)																		
46	Summer flounder (S)																		
47	Summer flounder (L)																		
48	Skates																		
49	Demersal benthivores - other																		
50	Demersal piscivores - other															0.001			
51	Demersal omnivores - other																		
52	Medium pelagic - other																		
53	Sharks - coastal																		
54	Sharks - pelagic																		
55	Large pelagics (HMS)																		
56	Pinnipeds																		
57	Baleen whales																		
58	Odontocetes																		
59	Seabirds														0.001				
60	Shorebirds - piscivorous																		
61	Detritus	0.734	0.526	0.135	0.049	0.068	0.235	0.498	0.374	0.333	0.436	0.182	0.270	0.314	0.001			0.278	0.221
	Import																		

Table 41. (Continued) Diet composition for Sim2 of the NWACS-FULL model. Columns indicate the predators (labeled by node number) and rows are prey. (Page 2 of 3)

Node	Prey \ predator	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
1	Phytoplankton	0.163	0.130																		
2	Other primary producers	0.014	0.025						0.002	0.001		0.001	0.003	0.001		0.001					
3	Bacteria																				
4	Microzooplankton	0.220																			
5	Small copepods	0.220	0.300	0.400		0.028	0.454				0.027			0.005	0.002	0.001			0.002		
6	Large copepods	0.220	0.300		0.152		0.172				0.027			0.005	0.002	0.001			0.002		
7	Gelatinous zooplankton			0.023	0.008	0.507	0.018	0.002	0.004	0.006					0.001	0.002	0.220	0.151			
8	Micronekton		0.084	0.365	0.509	0.112	0.186	0.009	0.009		0.092	0.016	0.002	0.229	0.174	0.131	0.096	0.020	0.150	0.024	0.003
9	Macrobenthos - polychaetes		0.010	0.002		0.019	0.001	0.001	0.002	0.001	0.019	0.034	0.028	0.009	0.010	0.005	0.019	0.005	0.075	0.024	0.016
10	Macrobenthos - crustaceans		0.091	0.128	0.118	0.314	0.135	0.016	0.017	0.013	0.461	0.229	0.033	0.139	0.124	0.107	0.043	0.019	0.473	0.147	0.029
11	Macrobenthos - molluscs					0.002				0.001	0.005	0.041	0.016	0.001	0.002		0.112	0.074	0.016	0.061	0.055
12	Macrobenthos - other				0.021	0.006	0.004	0.001			0.005	0.002	0.003			0.001	0.004	0.004	0.087	0.096	0.014
13	Megabenthos - filterers											0.008	0.001				0.025	0.089	0.015	0.044	0.036
14	Megabenthos - other				0.004			0.005	0.015	0.015	0.019	0.038	0.021	0.015	0.017	0.028	0.014	0.021	0.030	0.184	0.144
15	Shrimp and Similar Species			0.011	0.001	0.001	0.001	0.001	0.001	0.001	0.005	0.001		0.039	0.027	0.009	0.012	0.007	0.053	0.061	0.019
16	Mesopelagics				0.001			0.001	0.003	0.003							0.004	0.001			
17	Atlantic herring			0.003					0.031	0.032		0.023	0.062				0.048	0.043		0.067	0.222
18	Alosines								0.019	0.016	0.010	0.008	0.012		0.001	0.002		0.008		0.002	0.009
19	Atlantic menhaden (S)							0.009	0.040	0.020	0.020	0.078	0.076	0.013	0.024	0.027	0.001	0.004			
20	Atlantic menhaden (M)								0.050	0.040	0.020	0.078	0.129		0.010	0.030		0.003			
21	Atlantic menhaden (L)									0.010		0.001	0.099			0.001		0.001			
22	Anchovies			0.014			0.024	0.444	0.253	0.190	0.112	0.232	0.254	0.436	0.463	0.408		0.015			
23	Atlantic mackerel							0.002	0.024	0.020		0.001	0.012				0.086	0.119		0.005	0.019
24	Squid			0.007	0.158	0.010	0.003	0.116	0.177	0.220	0.001	0.016	0.045	0.009	0.020	0.029	0.142	0.122		0.007	0.006
25	Butterfish				0.025			0.266	0.115	0.119		0.021	0.028	0.001	0.009	0.018	0.008	0.020			0.004
26	Small pelagic - other			0.017			0.001	0.035	0.067	0.073	0.113	0.060	0.073	0.066	0.054	0.090	0.028	0.078	0.097	0.157	0.152
27	Bluefish (S)								0.002	0.002		0.001	0.001								0.011
28	Bluefish (M)									0.001			0.001								0.001
29	Bluefish (L)																				
30	Striped bass (S)							0.001	0.001			0.001	0.002					0.001			
31	Striped bass (M)												0.001								
32	Striped bass (L)																				
33	Weakfish (S)							0.001	0.001	0.001	0.002	0.004		0.001	0.003	0.002					
34	Weakfish (M)									0.002						0.023					
35	Weakfish (L)																				
36	Spiny dogfish (S)																	0.001			0.011
37	Spiny dogfish (L)																				0.011
38	Atlantic cod (S)																0.001	0.001			0.008
39	Atlantic cod (M)																				0.006
40	Atlantic cod (L)																				
41	Haddock								0.001	0.006								0.001		0.002	0.016
42	Hake			0.021	0.002			0.056	0.031	0.042	0.025	0.007	0.014	0.006	0.011	0.009	0.062	0.065	0.001	0.068	0.094
43	Atlantic croaker							0.001	0.010	0.007	0.003	0.003	0.010	0.002	0.004	0.002		0.001			
44	Yellowtail flounder (S)								0.001	0.001								0.001			0.003
45	Yellowtail flounder (L)									0.005											0.006
46	Summer flounder (S)								0.002	0.001								0.001			
47	Summer flounder (L)																				
48	Skates				0.001					0.002							0.006	0.013		0.001	0.004
49	Demersal benthivores - other			0.001				0.026	0.081	0.101	0.020	0.046	0.064	0.019	0.035	0.061	0.028	0.075		0.036	0.069
50	Demersal piscivores - other			0.004	0.001				0.009	0.009		0.009		0.001	0.001	0.004	0.001	0.013		0.004	0.011
51	Demersal omnivores - other			0.003				0.006	0.030	0.039	0.014	0.043	0.012	0.002	0.005	0.010	0.040	0.021		0.010	0.023
52	Medium pelagic - other																				
53	Sharks - coastal																				
54	Sharks - pelagic																				
55	Large pelagics (HMS)																				
56	Pinnipeds																				
57	Baleen whales																				
58	Odontocetes																				
59	Seabirds																				
60	Shorebirds - piscivorous																				
61	Detritus	0.163	0.060																		
	Import																				

Table 41. (Continued) Diet composition for Sim2 of the NWACS-FULL model. Columns indicate the predators (labeled by node number) and rows are prey. (Page 3 of 3)

Node	Prey \ predator	41	42	43	44	45	46	47	48	49	50	51	52	53	54	55	56	57	58	59	60
1	Phytoplankton																				
2	Other primary producers						0.001		0.001	0.001	0.002	0.002									
3	Bacteria																				
4	Microzooplankton																				
5	Small copepods	0.001	0.002	0.001		0.004	0.001		0.001	0.003								0.052			
6	Large copepods														0.025			0.475		0.039	
7	Gelatinous zooplankton	0.002				0.001						0.012	0.001	0.001	0.003	0.020		0.007	0.017		
8	Micronekton	0.138	0.184	0.079	0.134	0.032	0.256	0.032	0.014	0.055	0.035	0.006	0.032	0.036			0.073	0.303	0.031	0.160	
9	Macrobenthos - polychaetes	0.082	0.009	0.308	0.253	0.464	0.001		0.155	0.254	0.022	0.041	0.001								
10	Macrobenthos - crustaceans	0.242	0.222	0.159	0.457	0.333	0.110	0.041	0.176	0.290	0.062	0.063	0.025	0.012	0.010			0.056			0.028
11	Macrobenthos - molluscs	0.048	0.019	0.237	0.005	0.018	0.008	0.003	0.153	0.136	0.059	0.084		0.003				0.011			
12	Macrobenthos - other	0.284		0.006	0.023	0.041			0.005	0.048	0.001	0.013		0.019	0.010			0.021			
13	Megabenthos - filterers	0.006	0.005			0.001			0.003		0.001	0.006		0.003							
14	Megabenthos - other	0.045	0.042	0.083	0.081	0.015	0.059	0.056	0.234	0.129	0.393	0.559		0.019				0.005			0.028
15	Shrimp and Similar Species	0.017	0.044	0.006	0.025	0.003	0.025	0.006	0.016	0.015	0.036	0.013								0.022	
16	Mesopelagics		0.009				0.001				0.001	0.002	0.055		0.005			0.002	0.008		
17	Atlantic herring	0.069	0.057			0.001		0.039	0.019		0.011	0.010	0.009	0.062	0.069	0.152	0.147	0.015	0.073	0.093	
18	Alosines		0.004					0.014	0.004		0.007	0.001	0.027	0.062	0.011		0.051		0.007	0.017	0.088
19	Atlantic menhaden (S)								0.001		0.010	0.022		0.005	0.001	0.030	0.010	0.001	0.010	0.034	0.110
20	Atlantic menhaden (M)								0.001		0.020	0.030		0.020	0.028	0.030	0.029		0.030	0.034	0.165
21	Atlantic menhaden (L)										0.030			0.039	0.009	0.030	0.039		0.051	0.041	0.055
22	Anchovies		0.011	0.064			0.145	0.108	0.018	0.045	0.004	0.004	0.006	0.062	0.021	0.101	0.049	0.001	0.079	0.103	0.276
23	Atlantic mackerel	0.010	0.012					0.060	0.004		0.011			0.062	0.069	0.040	0.146	0.015	0.022	0.117	
24	Squid	0.001	0.130	0.011			0.123	0.210	0.038	0.013	0.040	0.037	0.283	0.126	0.155	0.061		0.006	0.307	0.063	
25	Butterfish	0.001	0.026	0.006			0.003	0.050	0.016		0.017	0.007	0.257	0.062	0.069	0.040	0.146	0.015	0.119	0.117	
26	Small pelagic - other	0.032	0.041	0.006	0.023	0.087	0.008	0.073	0.051		0.020	0.013	0.083	0.062	0.021	0.313	0.049	0.001	0.079	0.103	0.110
27	Bluefish (S)							0.001			0.002	0.002			0.001	0.010					
28	Bluefish (M)													0.020	0.020	0.020	0.010		0.020		
29	Bluefish (L)													0.020	0.020	0.020	0.020		0.010		
30	Striped bass (S)														0.007						0.001
31	Striped bass (M)													0.009	0.009		0.009				
32	Striped bass (L)													0.009	0.009		0.001				
33	Weakfish (S)						0.002	0.002	0.003		0.001	0.001			0.007						0.001
34	Weakfish (M)							0.020							0.007						
35	Weakfish (L)														0.007						
36	Spiny dogfish (S)		0.001					0.001			0.005			0.006	0.007	0.020	0.014		0.007	0.006	
37	Spiny dogfish (L)													0.028	0.028		0.019		0.020		
38	Atlantic cod (S)										0.011			0.004	0.005	0.020	0.010		0.004	0.002	
39	Atlantic cod (M)													0.004	0.005	0.020	0.009		0.004	0.002	
40	Atlantic cod (L)													0.004	0.005		0.009		0.004		
41	Haddock	0.001	0.004				0.002	0.026	0.008		0.008	0.001		0.013	0.010	0.010	0.012				
42	Hake	0.011	0.083				0.121	0.056	0.001		0.024	0.002	0.116	0.013	0.014	0.040	0.014		0.014	0.006	
43	Atlantic croaker						0.013	0.005	0.006	0.001	0.006	0.004		0.013	0.010		0.012				0.006
44	Yellowtail flounder (S)		0.001						0.001		0.001			0.006	0.005		0.012				
45	Yellowtail flounder (L)		0.001								0.001			0.006	0.005						
46	Summer flounder (S)											0.001		0.006	0.007		0.005		0.001	0.002	
47	Summer flounder (L)													0.006	0.007				0.007	0.002	
48	Skates								0.001		0.053	0.005	0.059	0.020	0.019	0.010	0.019				
49	Demersal benthivores - other	0.007	0.060	0.002			0.086	0.139	0.053	0.009	0.053	0.048	0.042	0.013	0.010		0.012				0.011
50	Demersal piscivores - other	0.001	0.026	0.018				0.010	0.004		0.025	0.012		0.013	0.014		0.014		0.014	0.006	0.011
51	Demersal omnivores - other		0.011	0.011			0.035	0.049	0.015		0.028	0.001	0.005	0.062	0.080		0.062		0.060	0.031	0.011
52	Medium pelagic - other														0.021	0.010					
53	Sharks - coastal													0.002	0.008						
54	Sharks - pelagic													0.010	0.019						
55	Large pelagics (HMS)													0.009	0.009						
56	Pinnipeds													0.015	0.015						
57	Baleen whales													0.012	0.010						
58	Odontocetes													0.012	0.020				0.002		
59	Seabirds													0.017	0.020						
60	Shorebirds - piscivorous													0.001							
61	Detritus														0.050			0.013			
	Import																				0.100

Table 42. Summary of the eight NWACS-FULL models fit. Models differed in the diet matrix used, the vulnerability caps (v.cap) that were employed, and the manual adjustments (Man. Adjust) that were made to improve the Atlantic menhaden stock-recruit relationship and the *F_{MSY}* dynamics for the ERP focal species. Model fits are represented by sum of squares (SS) and Akaike's Information Criteria (AIC).

Sim	Diet	v.cap	Man. Adjust	Fitting Iter.	SS	AIC
1	Base			9	1399	-778
2	**Base**		**X**	**--**	**1764**	**-281**
3	Base	X		--	2218	210
4	Base	X	X	--	2249	240
5	+Menh			12	1448	-704
6	**+Menh**		**X**	**--**	**2013**	**2**
7	+Menh	X		--	2037	28
8	+Menh	X	X	--	2271	261

Table 43. Estimates of Atlantic menhaden F_{MSY} for their three age stanzas based on projections using the base (1982) fishing mortality rates from the NWACS-FULL model. Estimates were made using Sim2 and Sim6 of the NWACS-FULL model by finding an effort multiplier (E_{MULT}) that generates the maximum Atlantic menhaden catch. Atlantic menhaden *F* rates for 2017 are included for comparison.

		Sim 2		Sim 6		
Stanza	F_{1982}	Emult	Fmsy	Emult	Fmsy	F_{2017}
age-0	0.013	2.818	0.038	1.606	0.022	0.000
age-1-2	0.215	3.424	0.735	1.909	0.410	0.038
age-3+	0.368	2.515	0.926	1.303	0.480	0.112

Table 44. Effect of fishing Atlantic menhaden at F_{TARGET} on other species from the NWACS-FULL model. Numerical values (as percentages) are the biomass differences (B_{DIFF}) and catch differences (C_{DIFF}) for species when Atlantic menhaden are fished at their target *F* levels ("TARG Menh – SQ Others" scenario) relative to the status quo scenario ("SQ Menh – SQ Others"). Calculations were made based on two different model formulations (Sim 2 and Sim 6); see text for calculation of B_{DIFF} and C_{DIFF}. Dashed line separates the ERP focal species from the other groups that were most sensitive (with at least one number greater than 3%).

	Sim 2				Sim 6			
	B_{diff}	B_{diff}	C_{diff}	C_{diff}	B_{diff}	B_{diff}	C_{diff}	C_{diff}
Group	4 yr	40 yr	4 yr	40 yr	4 yr	40 yr	4 yr	40 yr
Menhaden	-5	-5	82	82	-8	-7	77	79
Bluefish	0	1	0	1	-2	-11	-3	-11
Striped Bass	-2	-7	-1	-7	-4	-10	-3	-10
Weakfish	0	-2	0	-2	-6	-5	-6	-6
Spiny Dogfish	0	0	0	0	-4	-4	-4	-4
Atlantic Herring	0	2	0	2	0	-2	0	-2
Alosines	0	0	0	0	-5	14	-5	14
Atlantic Cod	0	5	0	5	-1	8	-1	7
Large pelagics (HMS)	-2	-1	-2	-1	-4	-4	-4	-4
Pinnipeds	0	1			0	-4		
Seabirds	-2	-1			-3	-4		
Sharks-coastal	-1	0	-1	0	-1	-4	-1	-4
Demersal Piscivores	-2	-2	-2	-2	-3	-6	-3	-6
Nearshore Pisc. Birds	-5	-9			-9	-14		

Table 45. Effects of different Atlantic menhaden fishing mortality reference points on the equilibrium biomass and catch of different trophic groups from the NWACS-FULL model. Biomass is expressed relative to the equilibrium biomass under the status quo Atlantic menhaden fishing scenario (*B/B_{2017}= B/B_{SQ}*) while catch is relative to the maximum equilibrium catch across all Atlantic menhaden fishing scenarios (*C/C_{MAX}*). Non-menhaden species were kept at their target *F* for these projections. Fishing reference points were: no Atlantic menhaden fishing (*F*=0), status quo fishing (*F_{2017}=F_{SQ}*), single-species *F_{TARGET}*, single-species *$F_{THRESHOLD}$*, EwE *F_{MSY}* based on Figure 165, and *F* for Atlantic menhaden extinction (*$F_{EXTINCTION}$*) is included for comparison. The dashed line separates the ERP focal species from other groups experiencing at least a 15% change in *B/B_{2017}* or *C/C_{MAX}*. Values differing from 1 by more than 10% are in bold, and groups with biomasses increase at higher Atlantic menhaden *F* indicated with (+).

	F=0		F_{SQ}		F_{Target}		$F_{threshold}$		F_{MSY}		$F_{Extinction}$	
Group	B/B_{SQ}	C/C_{max}	B/B_{SQ}	C/C_{max}	B/B_{SQ}	C/C_{max}	B/B_{SQ}	C/C_{max}	B/B_{SQ}	C/C_{max}	B/B_{SQ}	C/C_{max}
Menhaden	1.06	**0.00**	1.00	**0.17**	0.95	**0.31**	**0.78**	**0.64**	**0.47**	1.00	**0.00**	**0.00**
Striped Bass	1.09	1.00	1.00	0.91	0.92	**0.84**	**0.72**	**0.64**	**0.46**	**0.40**	**0.31**	**0.27**
Bluefish	1.01	1.00	1.00	0.99	0.99	0.98	0.96	0.95	**0.89**	**0.89**	**0.85**	**0.85**
Weakfish	1.01	1.00	1.00	0.99	0.99	0.98	0.98	0.96	0.95	0.93	0.92	0.90
Dogfish	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.99	0.99	0.98	1.00	0.99
Herring	1.00	0.96	1.00	0.96	1.00	0.96	0.98	0.96	0.97	0.93	1.04	1.00
Nearshore Birds	1.10		1.00		0.91		**0.68**		**0.39**		**0.23**	
Dem. Pisc.	1.03	1.00	1.00	0.97	0.98	0.95	0.94	0.90	**0.87**	**0.85**	**0.83**	**0.81**
Seabirds	1.03		1.00		0.98		0.92		**0.85**		**0.84**	
Haddock	1.02	1.00	1.00	0.98	0.98	0.96	0.93	0.91	**0.85**	**0.83**	**0.85**	**0.83**
Large Pelagics (HMS)	1.02	1.00	1.00	0.98	0.98	0.96	0.93	0.91	**0.86**	**0.84**	**0.85**	**0.83**
Shark-coastal	1.02	1.00	1.00	0.98	0.98	0.96	0.94	0.92	**0.89**	**0.87**	**0.87**	**0.85**
Med. Pelagics (+)	0.99	**0.84**	1.00	**0.85**	1.01	**0.86**	1.04	**0.88**	1.10	0.93	**1.18**	1.00
Atlantic Cod (+)	0.94	**0.63**	1.00	**0.67**	1.05	**0.71**	**1.19**	**0.80**	**1.39**	0.92	**1.51**	1.00

Table 46. ERP model strengths and weaknesses comparison

	Simulation Tested	Age Structure	Species Complexity	Top-down Predators	Bottom-up Feedback	Fits to Observed Data	Sensitivity	Ability to Incorporate Error	Projections	# of EMOs Addressed	Other Assessment Inputs Needed	Other Assessment Outputs Needed	Type of ERPs	Ability to Assess Stock Status	Other
SPMTVr	Yes	None	Single spp.	No	No	Fits trend, not finer scale variability	Sensitive to start year, mildly sensitive to Binit	CVs on input data; error propagated to output parameters	No	Low	1	0	Umsy (time varying)	Yes	Minimal assumptions /mechanisms
SH	Yes	None	Low	Yes	No	Fits trend, not finer scale variability	Sensitive to start year	No CVs on input data; stochastic projections	Yes	Med	1	1	MUP, consumption	Yes	Proprietary software
MSSCAA	Yes	Full	Med	Yes	No*	Fits indices comparably to single spp	Sensitive to M0 assumptions & diet data	CVs on input data; error propagated to output parameters and stochastic projections	Yes	Med High	5	0	Multispp. MSY, SPR, consumption	Yes	Non-age structured species are problematic
NWACS MICE	No	Stanzas	Med	Yes	Yes	Fits trends for most ERP spp well	Sensitive to pred-prey interaction strengths	CVs incorporated through weighting input data; error not propagated to output parameters; stochastic projections	Yes	High		6	Multispp. Fmsy or Bmsy, consumption	No	
Full NWACS	No	Stanzas	High	Yes	Yes	Fits trends for most ERP spp well, not some other spp	Sensitive to pred-prey interaction strengths & diet data		Yes	High		15	Multispp. Fmsy or Bmsy, consumption	No	Complex data streams (for benchmark); uncertainty from data-poor groups

*Possible with further development

21 FIGURES

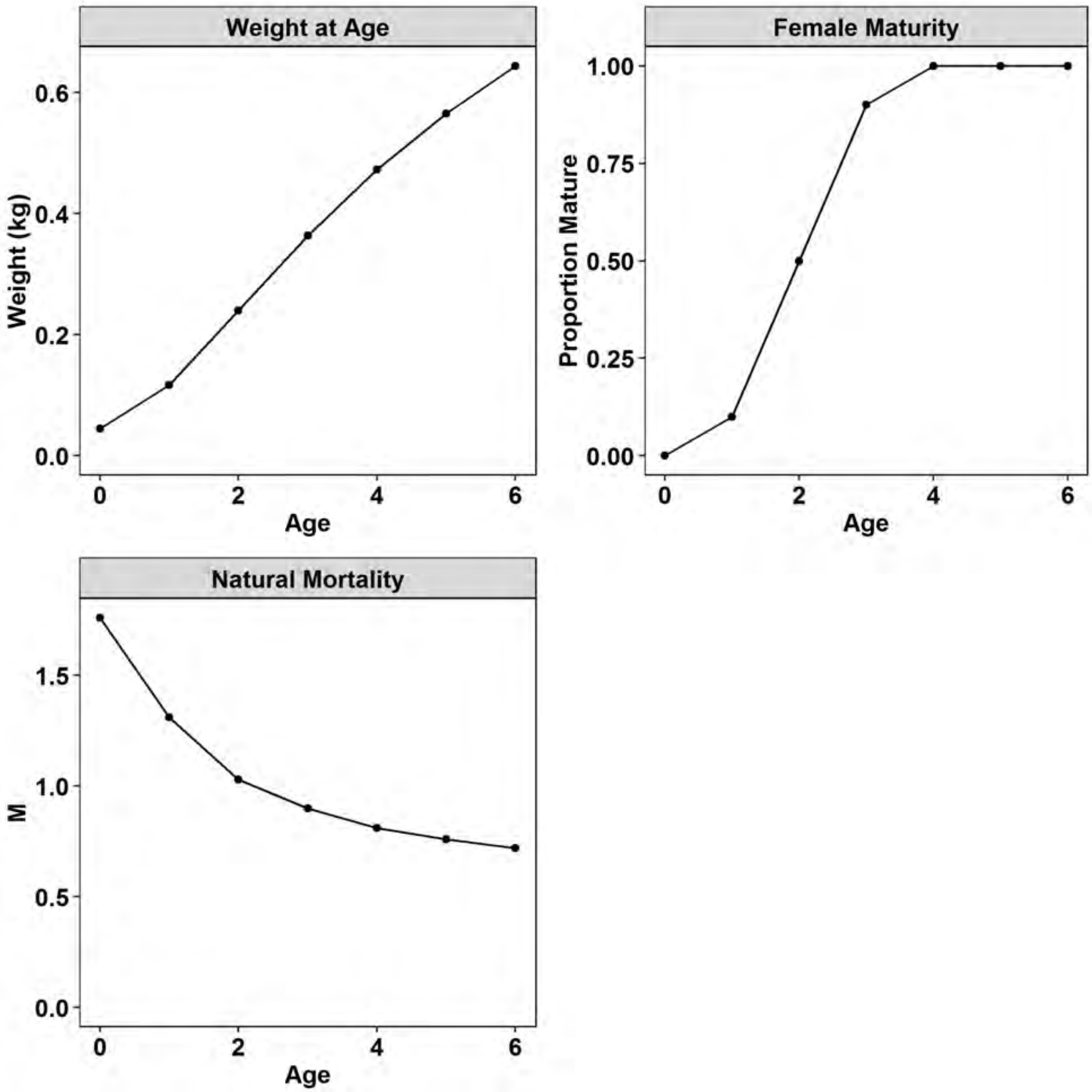


Figure 1. Time-invariant life history parameters for Atlantic menhaden.

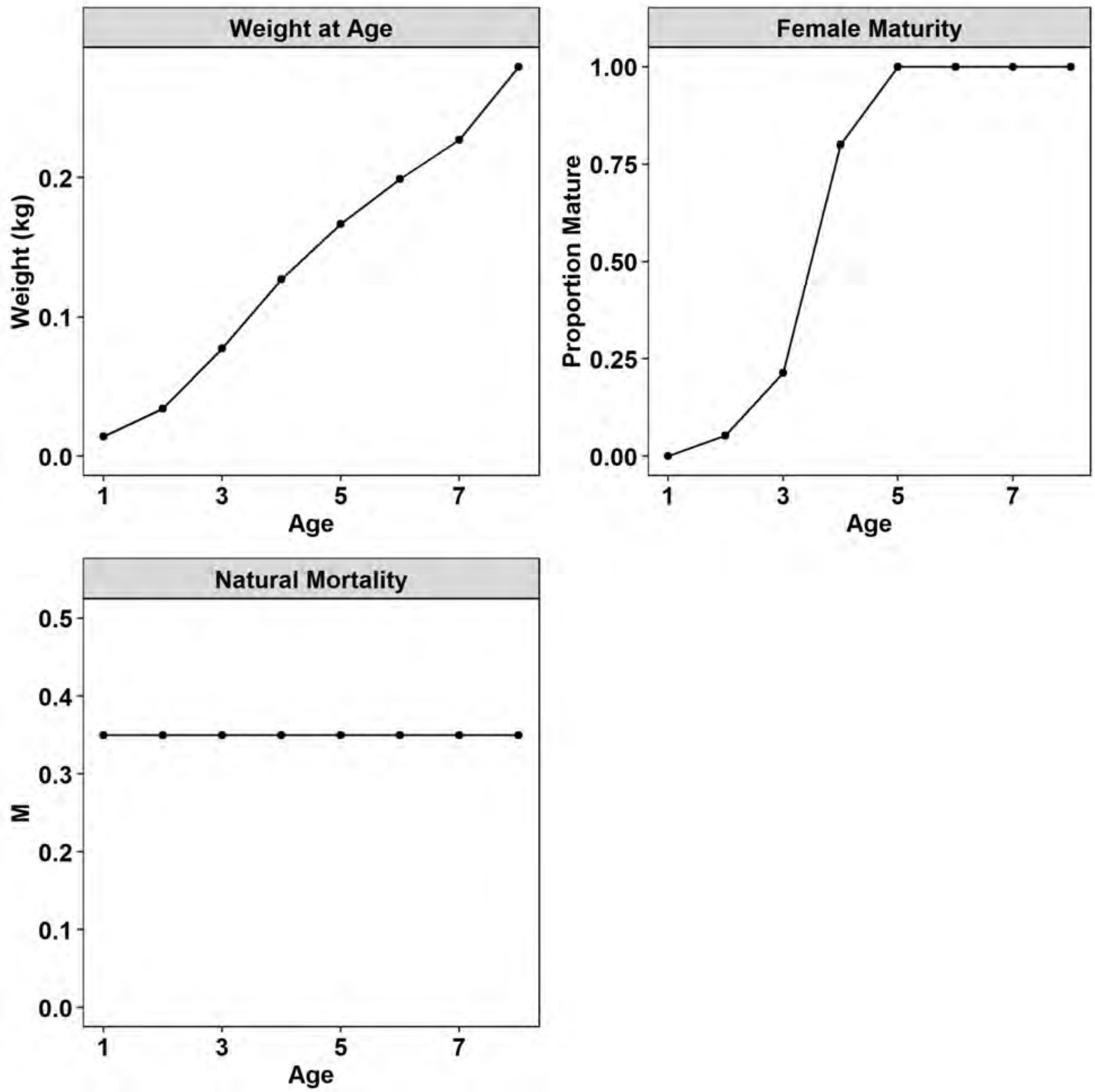


Figure 2. **Time-invariant life history parameters for Atlantic herring.**

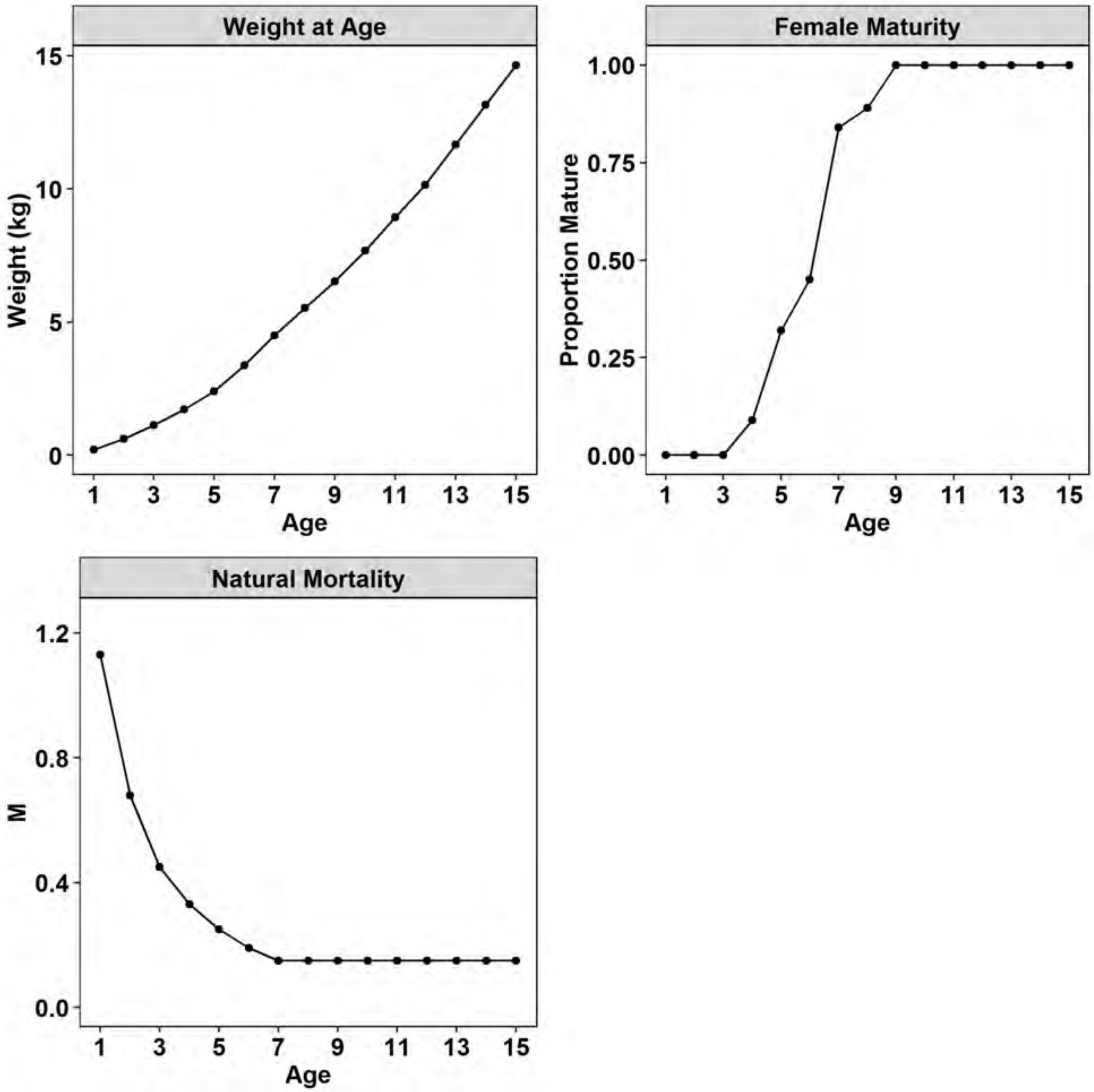


Figure 3. Time-invariant life history parameters for Atlantic striped bass.

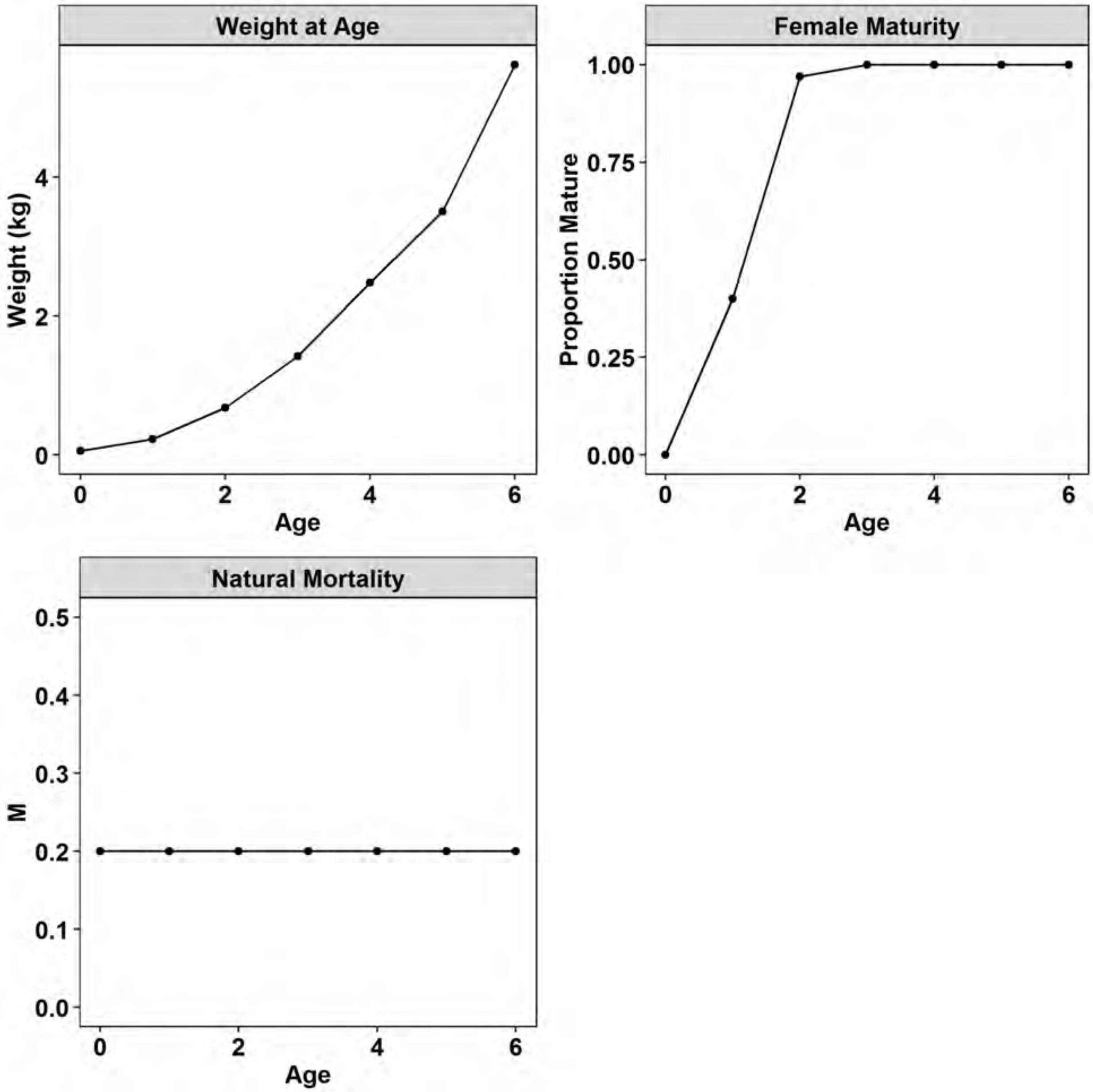


Figure 4. **Time-invariant life history parameters for bluefish.**

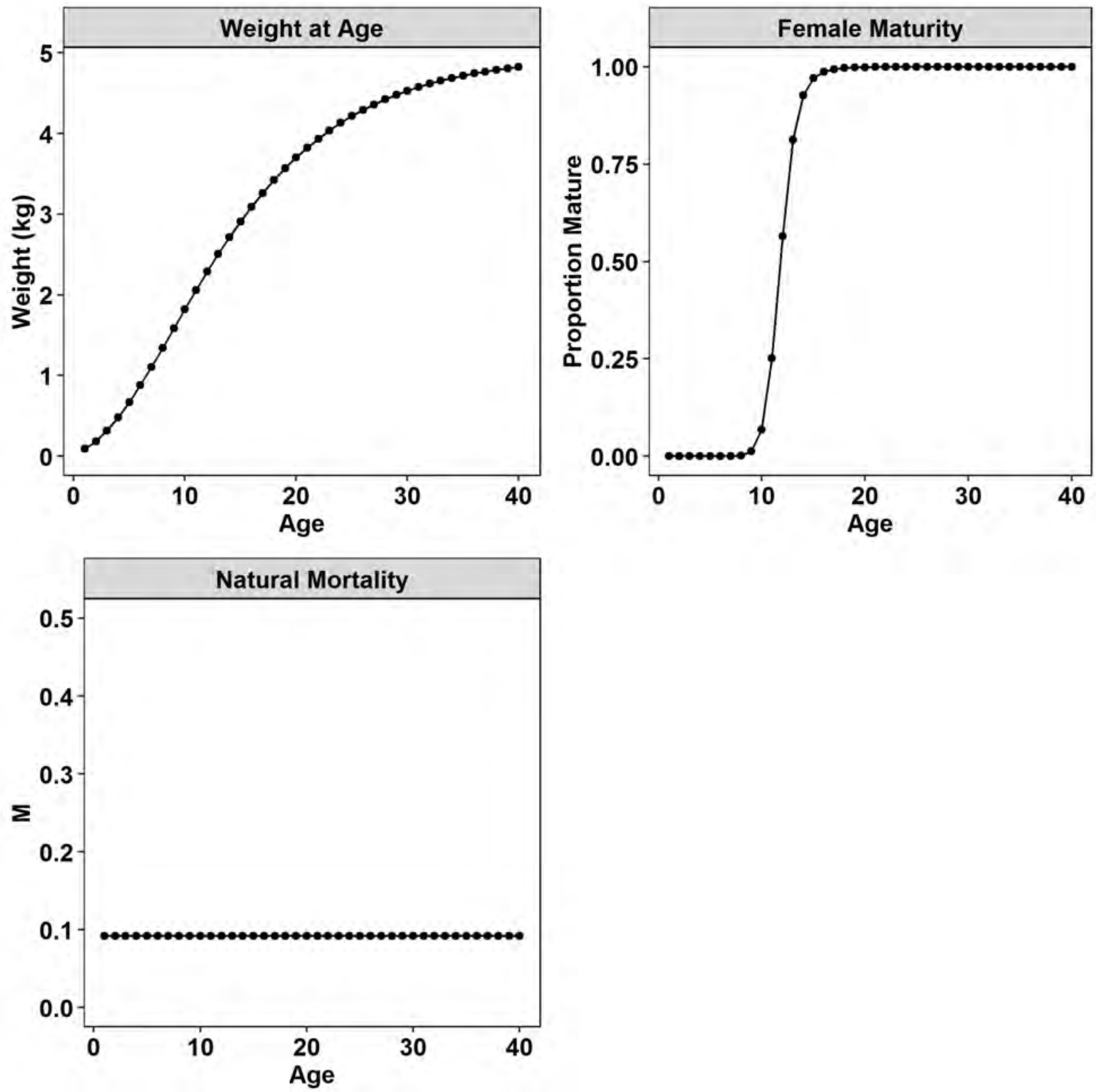


Figure 5. Time-invariant life history parameters for spiny dogfish.

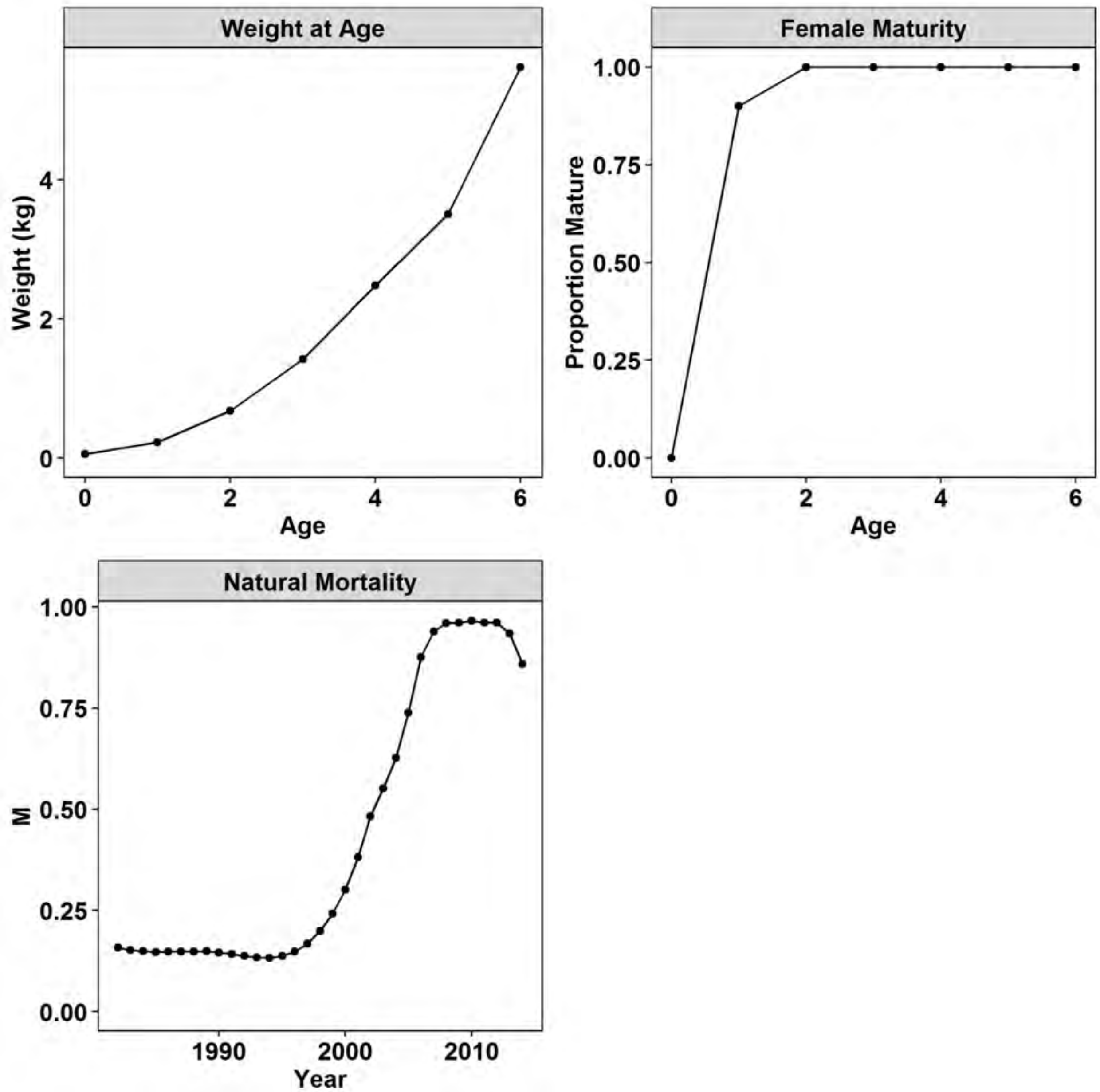


Figure 6. **Time-invariant weight at age and maturity at age parameters, and time-varying natural mortality estimates for weakfish.**

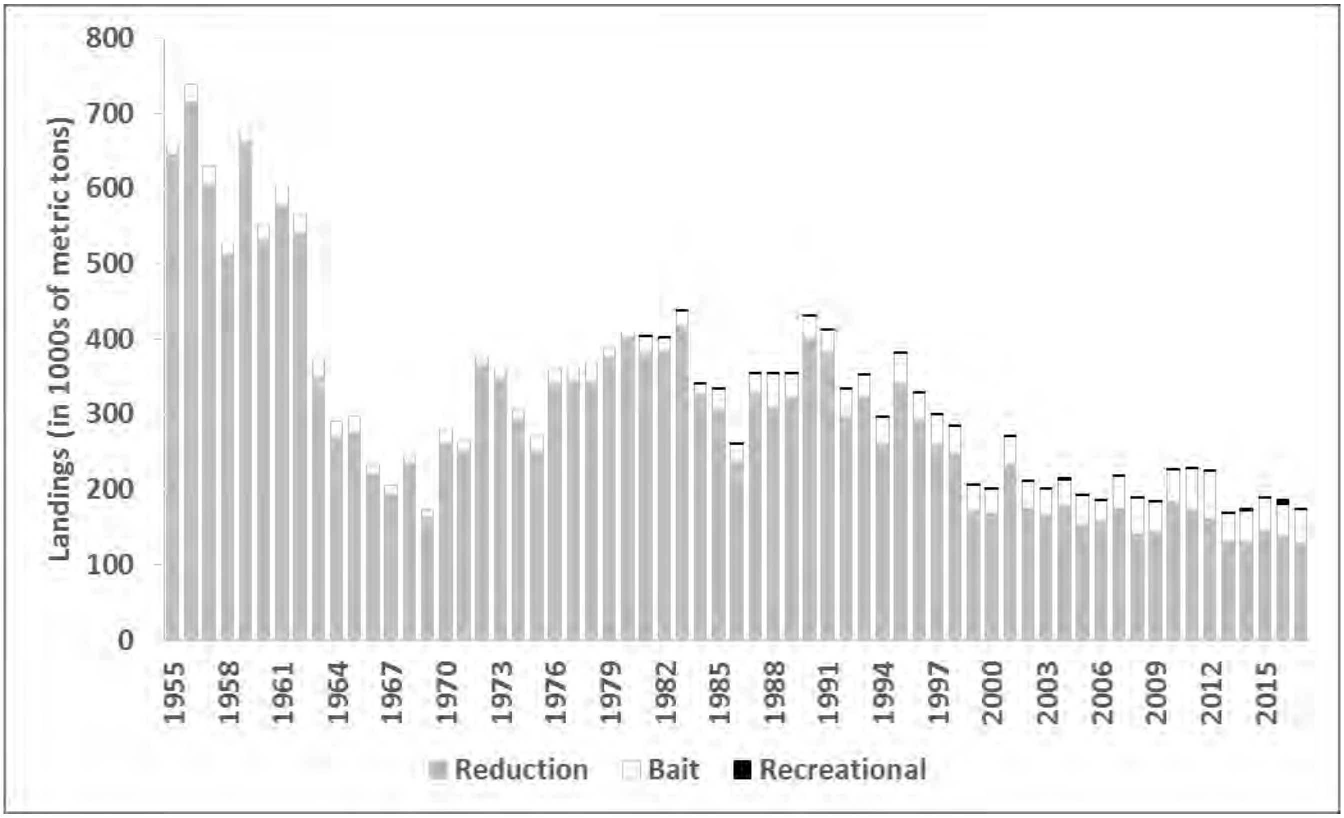


Figure 7. **Total removals of Atlantic menhaden by sector.**

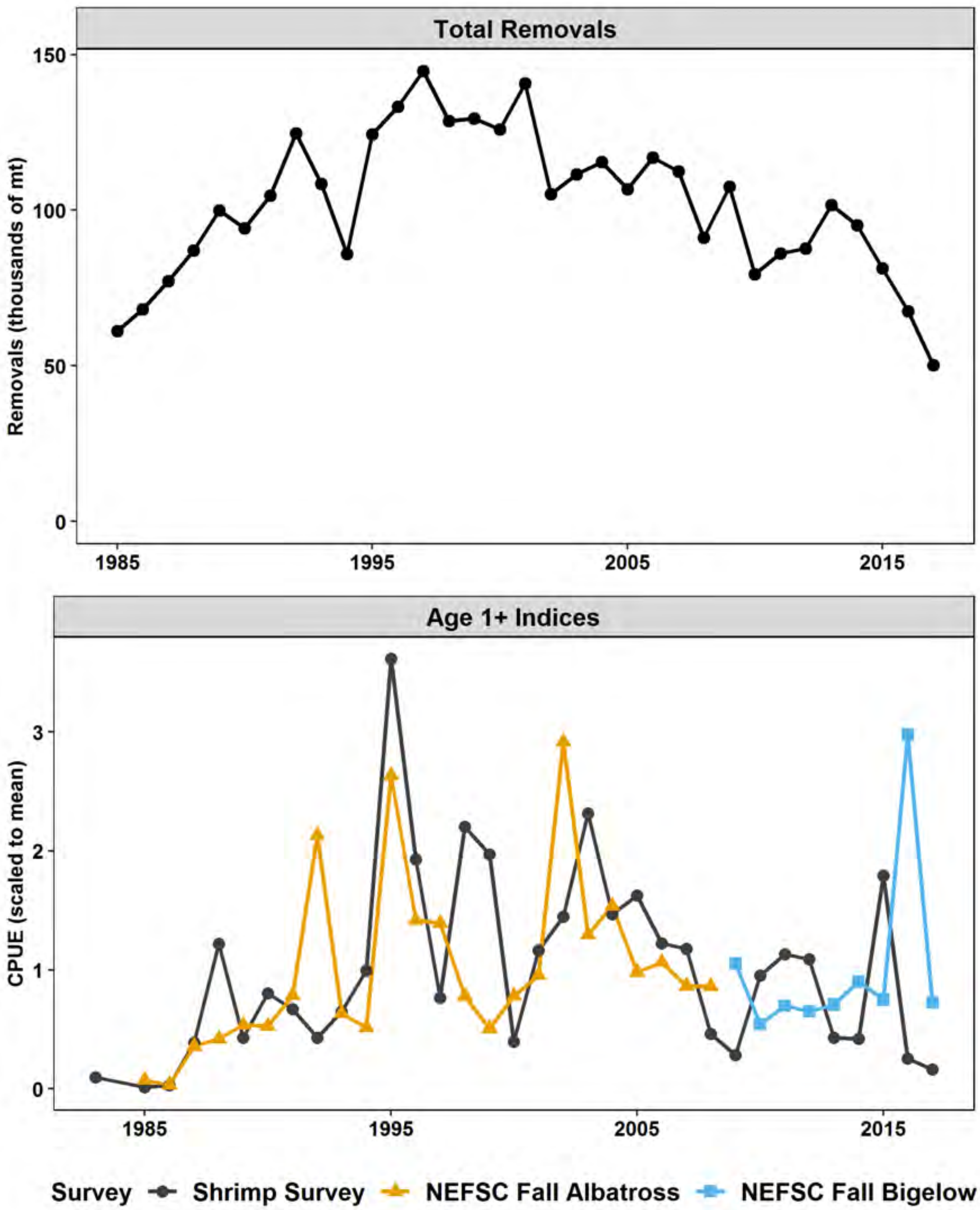


Figure 8. **Total removals (top) and indices of abundance (bottom) for Atlantic herring.**

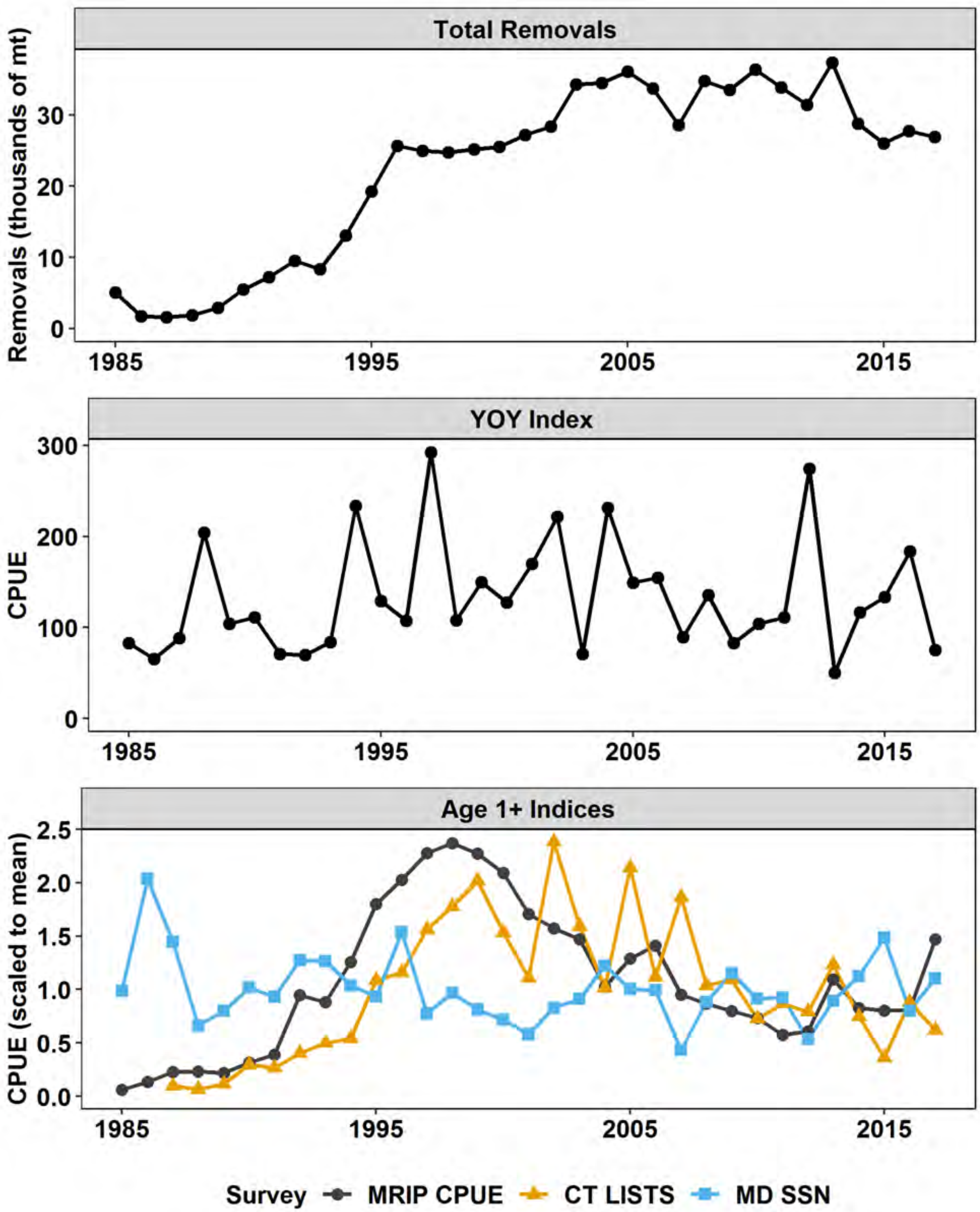


Figure 9. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for Atlantic striped bass.

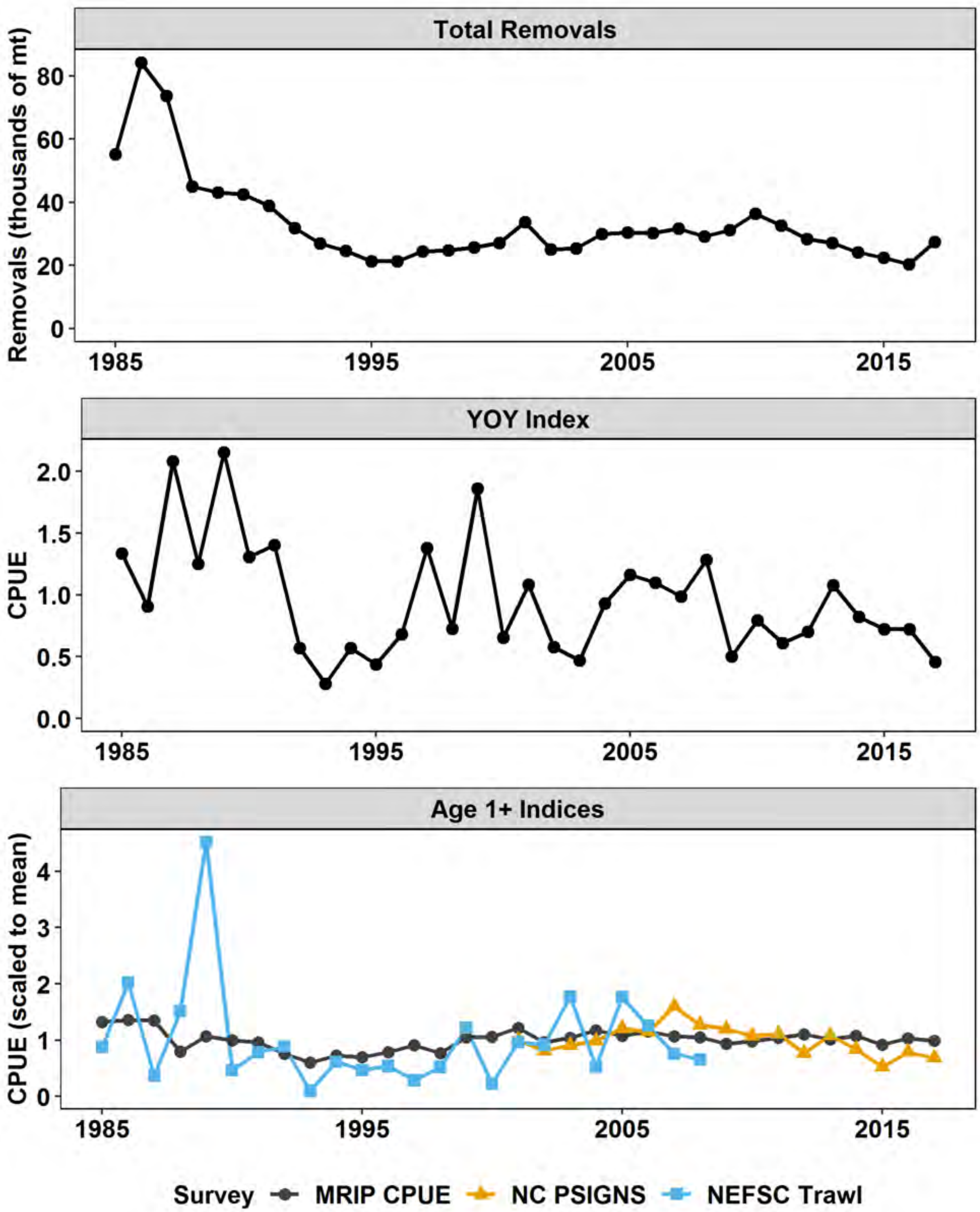


Figure 10. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for bluefish.

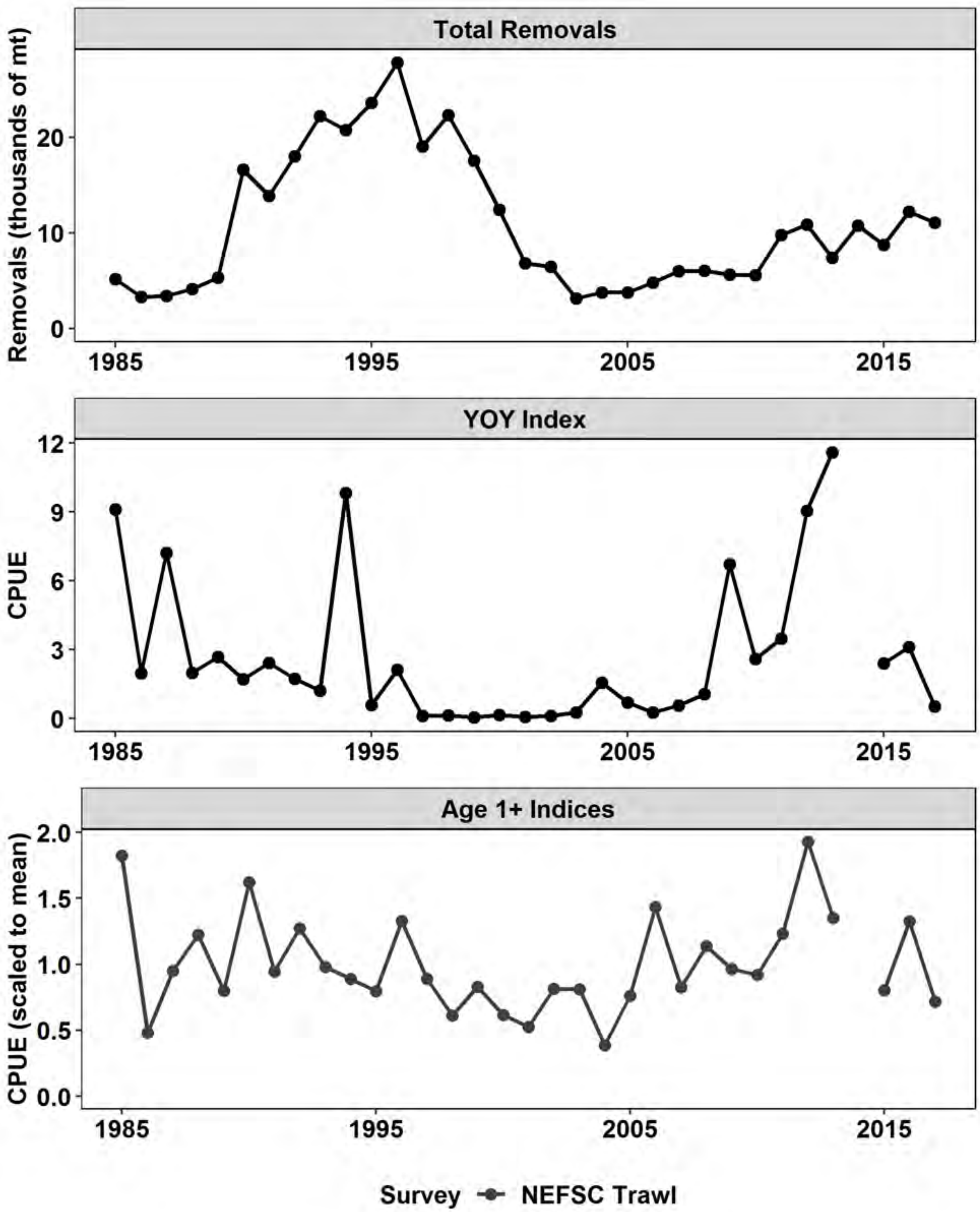


Figure 11. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for spiny dogfish.

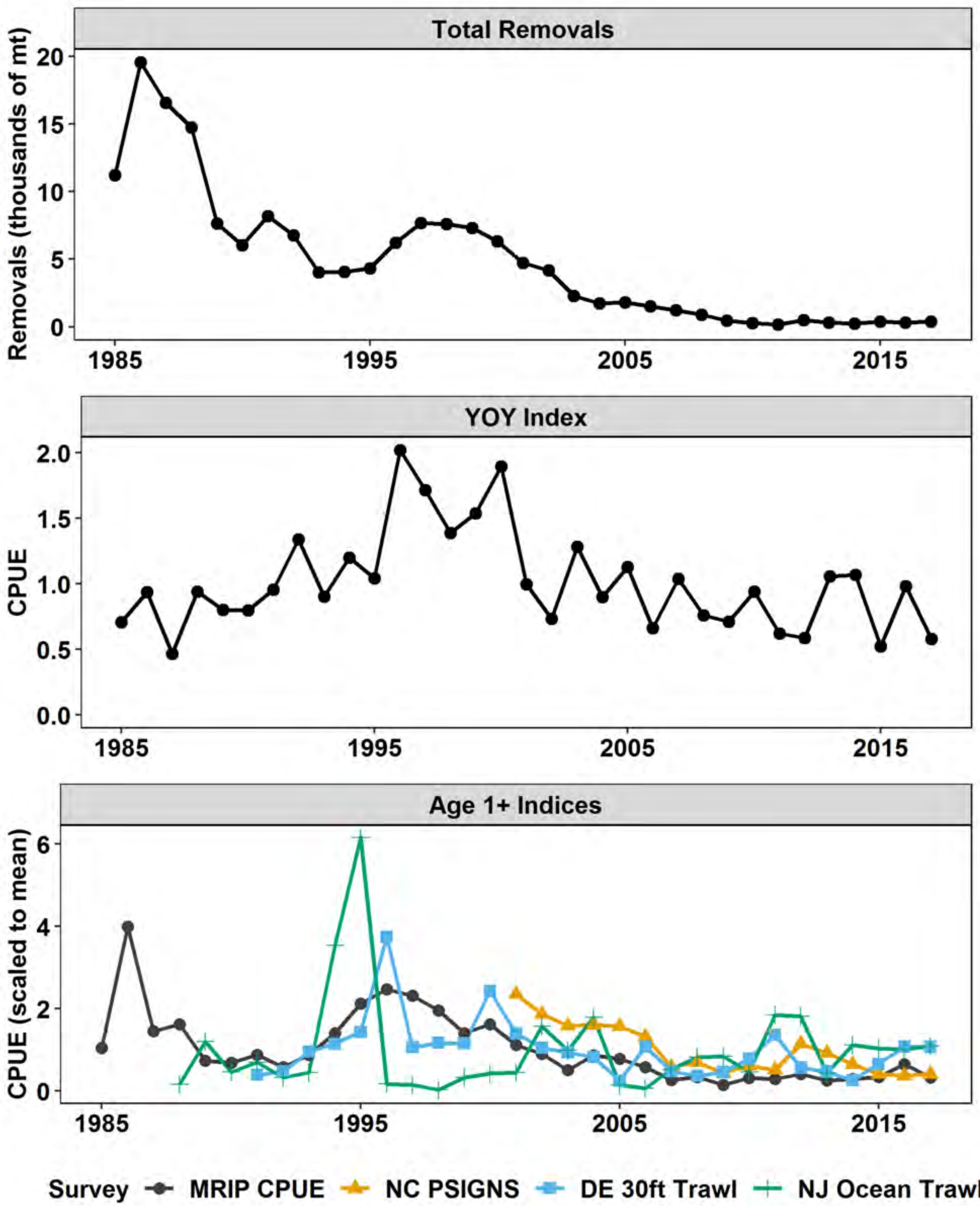


Figure 12. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for weakfish.

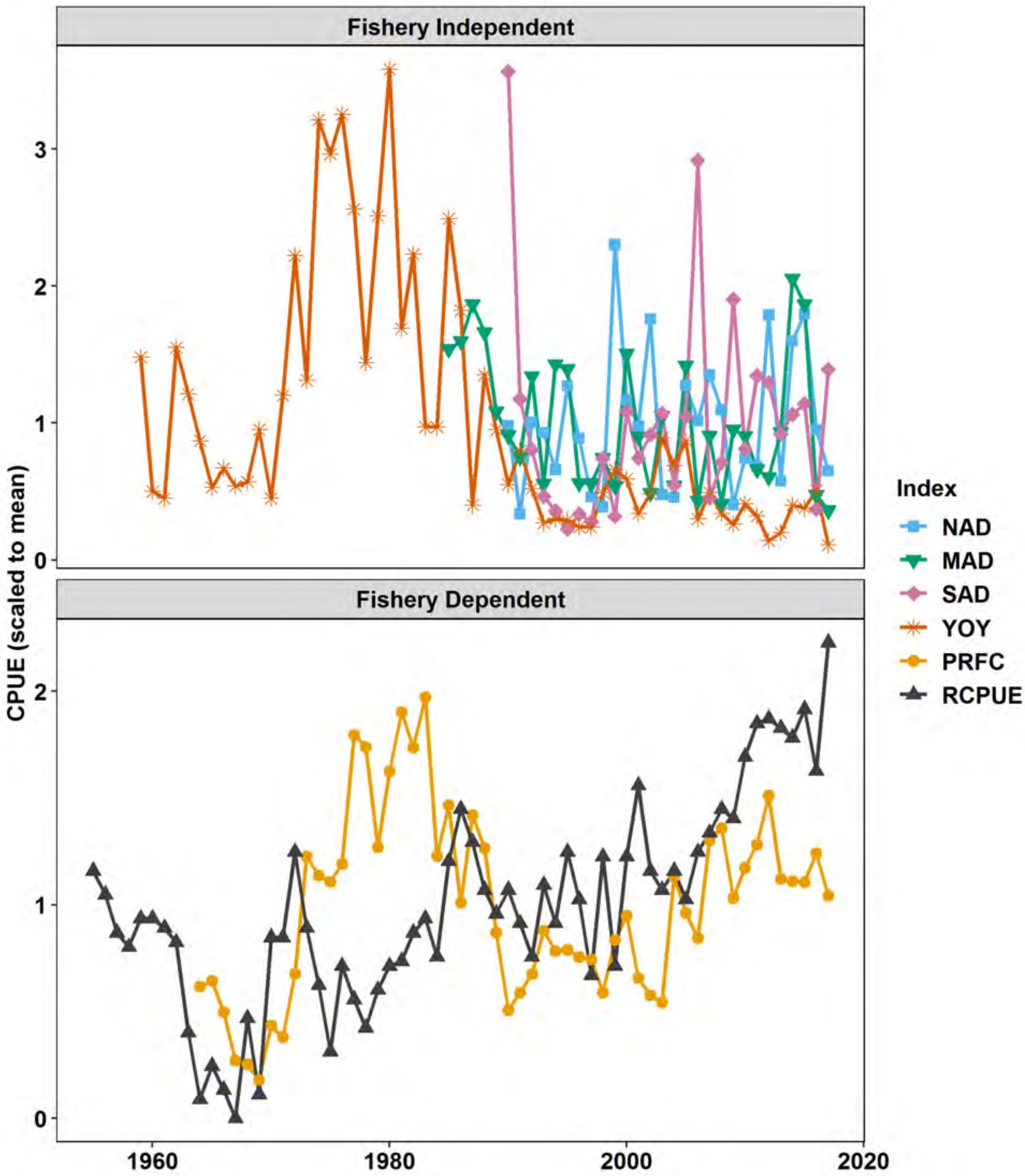


Figure 13. Fishery independent (top) and fishery dependent (bottom) indices of abundance for Atlantic menhaden.

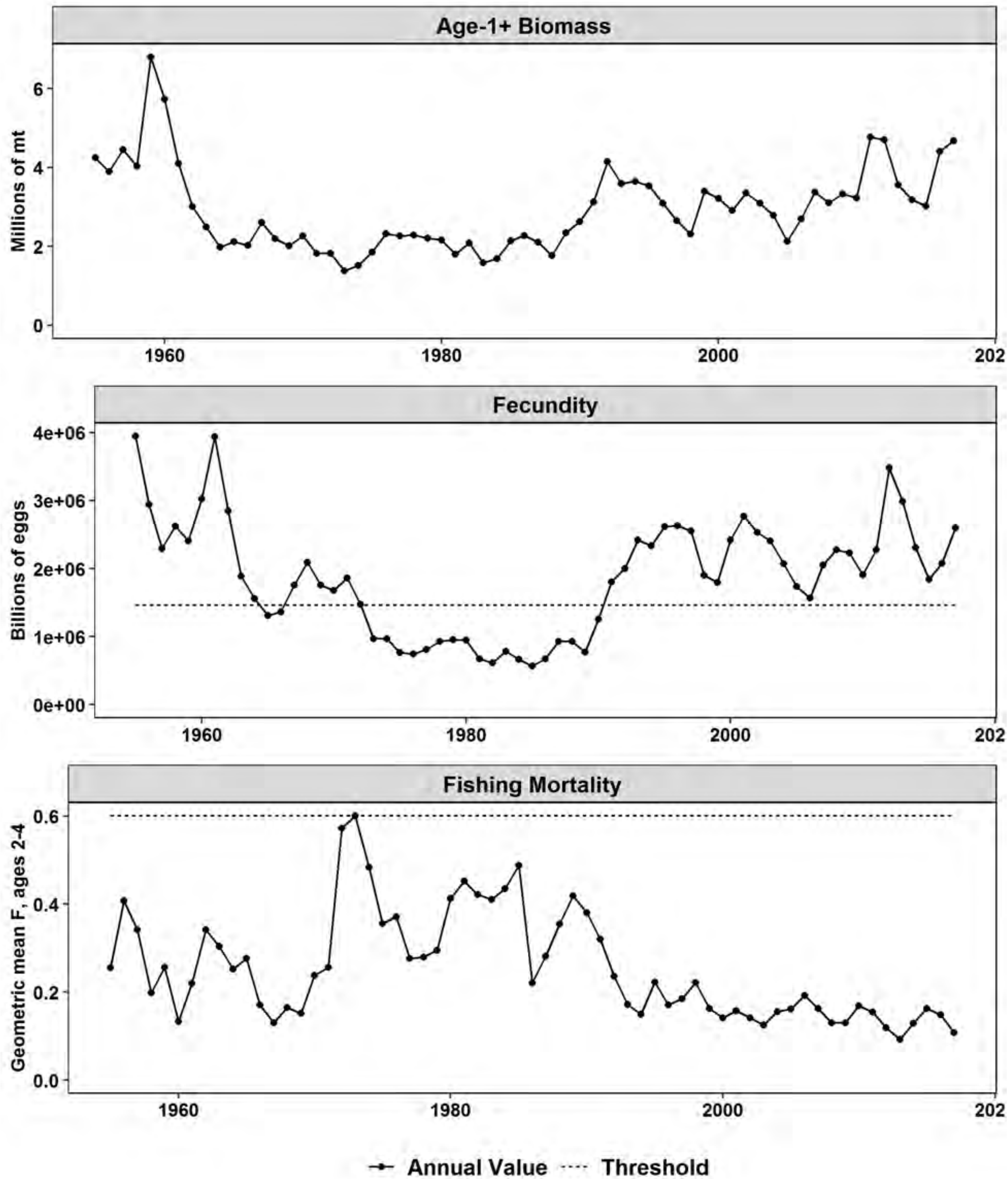


Figure 14. Age-1+ biomass, fecundity, and average *F* for Atlantic menhaden, plotted with their respective thresholds, where defined.

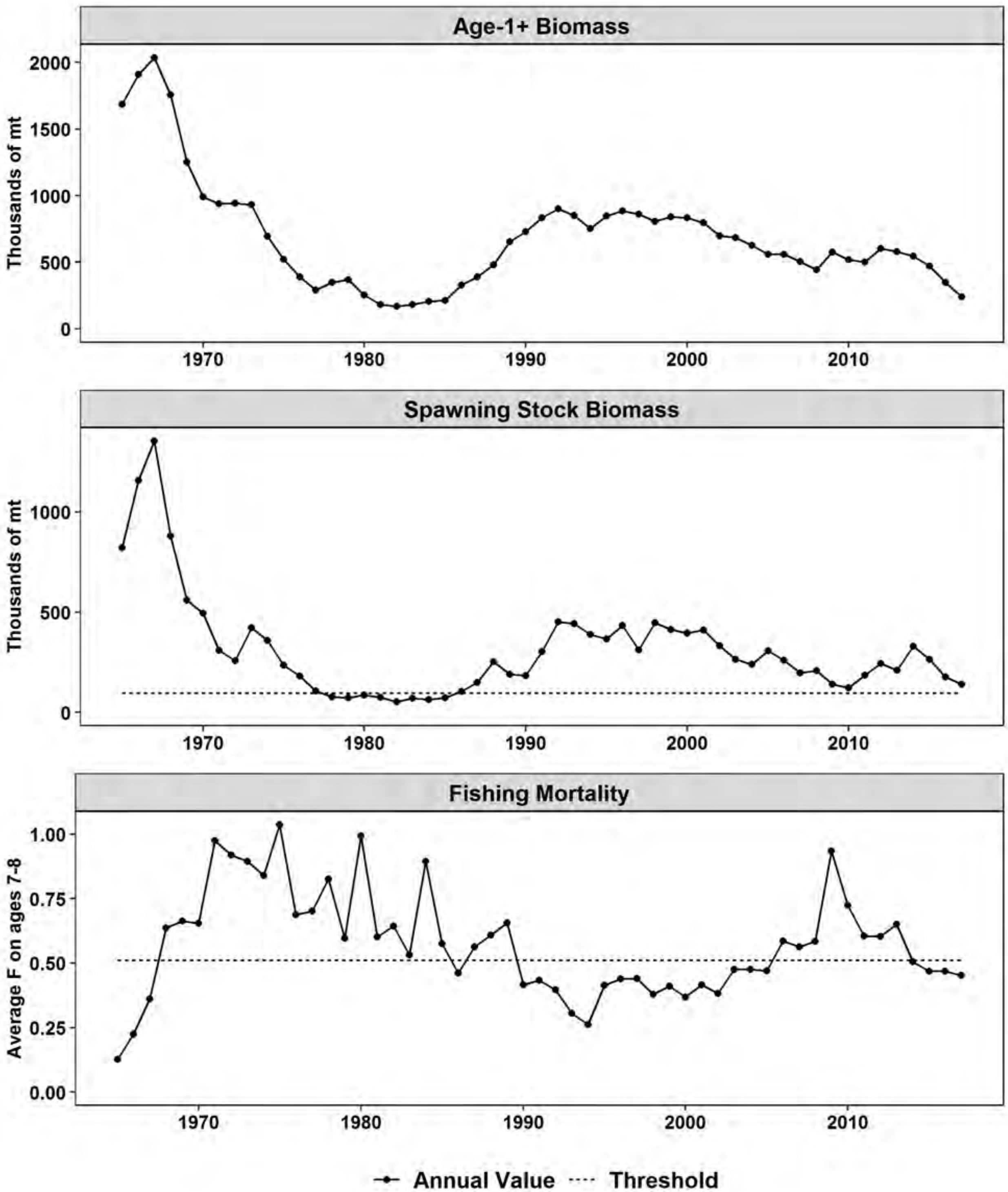


Figure 15. **Age-1+ biomass, spawning stock biomass, and average *F* for Atlantic herring, plotted with their respective thresholds, where defined.**

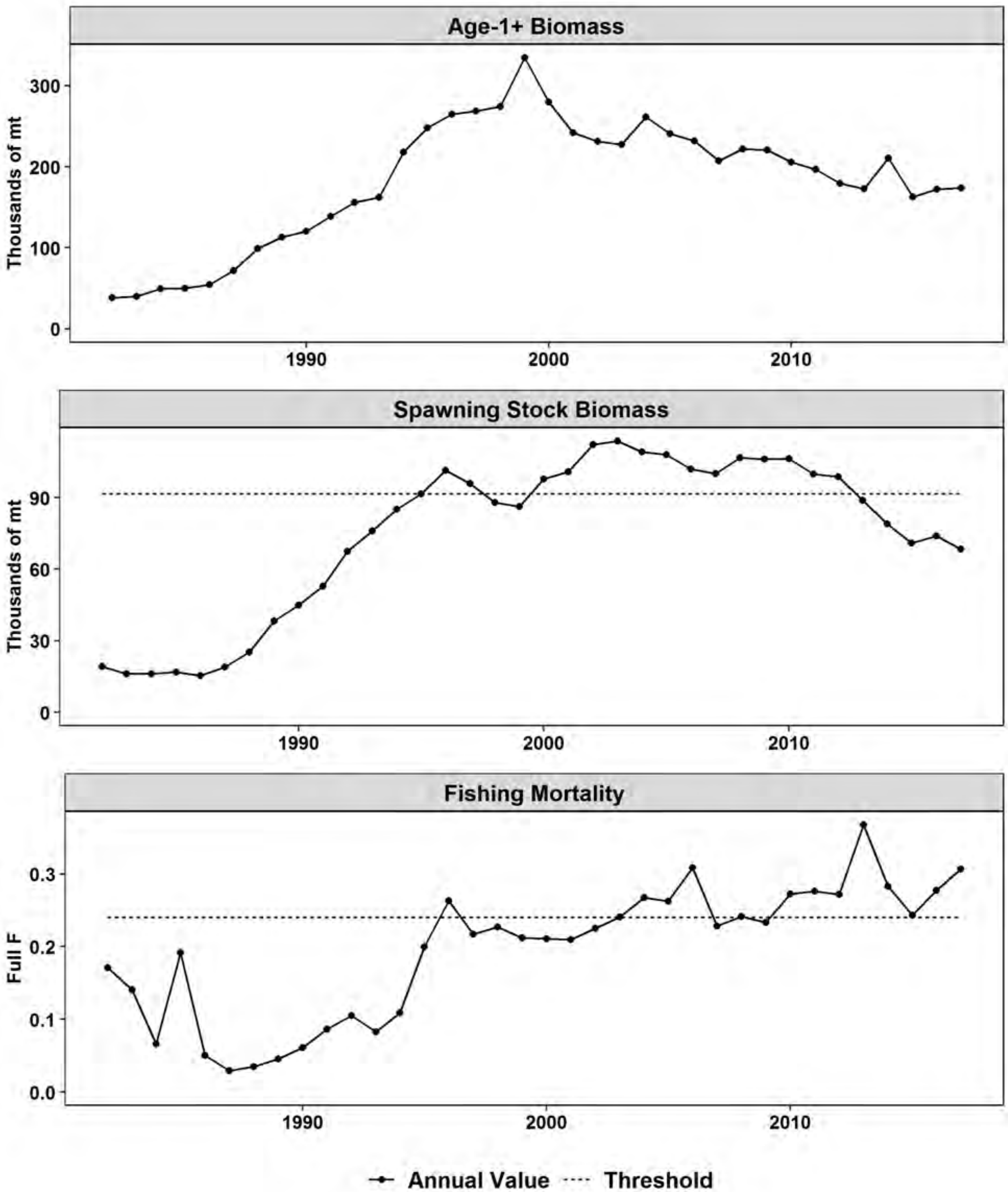


Figure 16. Age-1+ biomass, female spawning stock biomass, and average *F* for Atlantic striped bass, plotted with their respective thresholds, where defined.

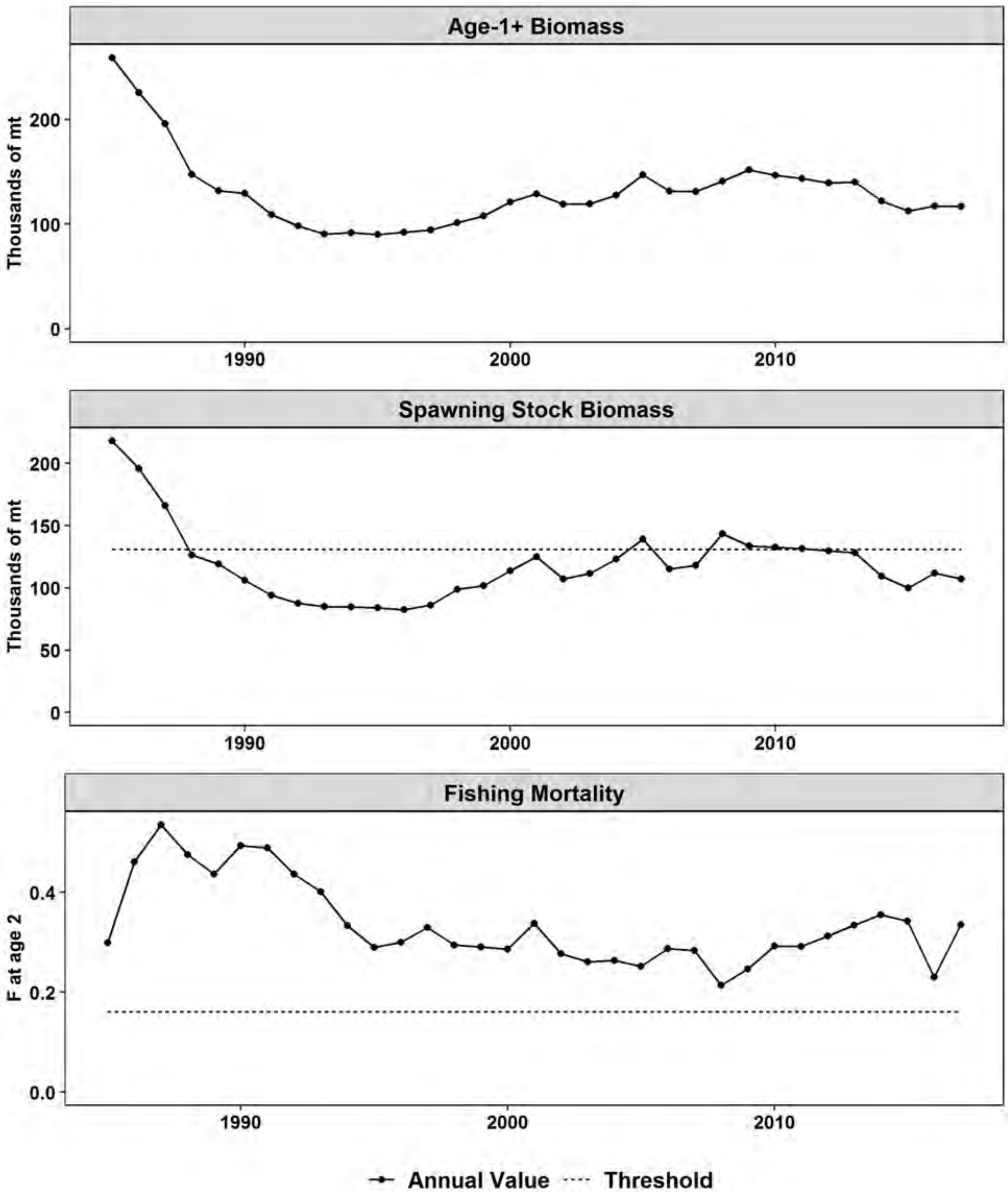


Figure 17. Age-1+ biomass, spawning stock biomass, and full *F* for bluefish, plotted with their respective thresholds, where defined. Estimates are from a preliminary assessment update with data through 2017 and may not match values used for management.

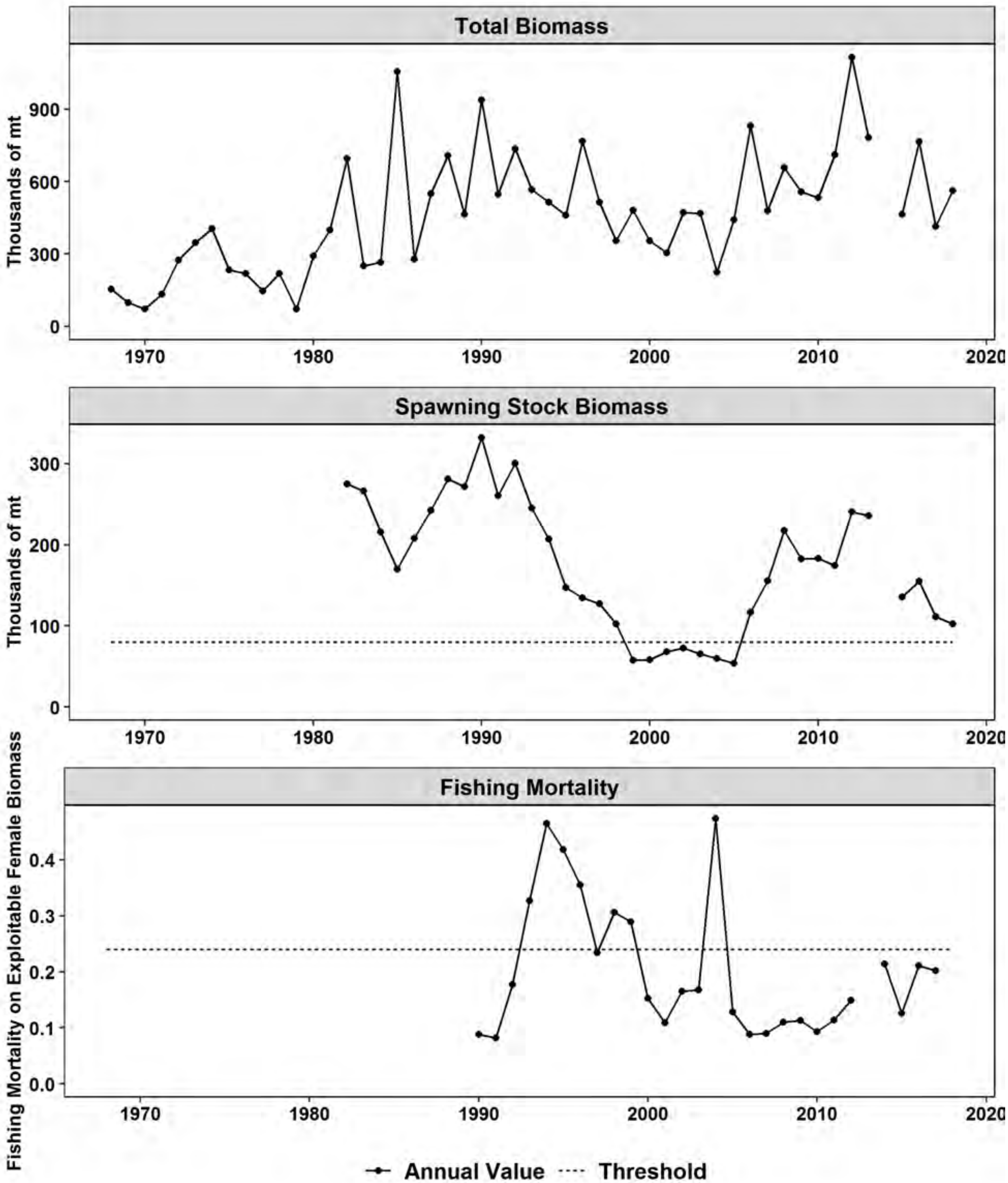


Figure 18. Total biomass, female spawning stock biomass, and *F* for spiny dogfish, plotted with their respective thresholds, where defined.

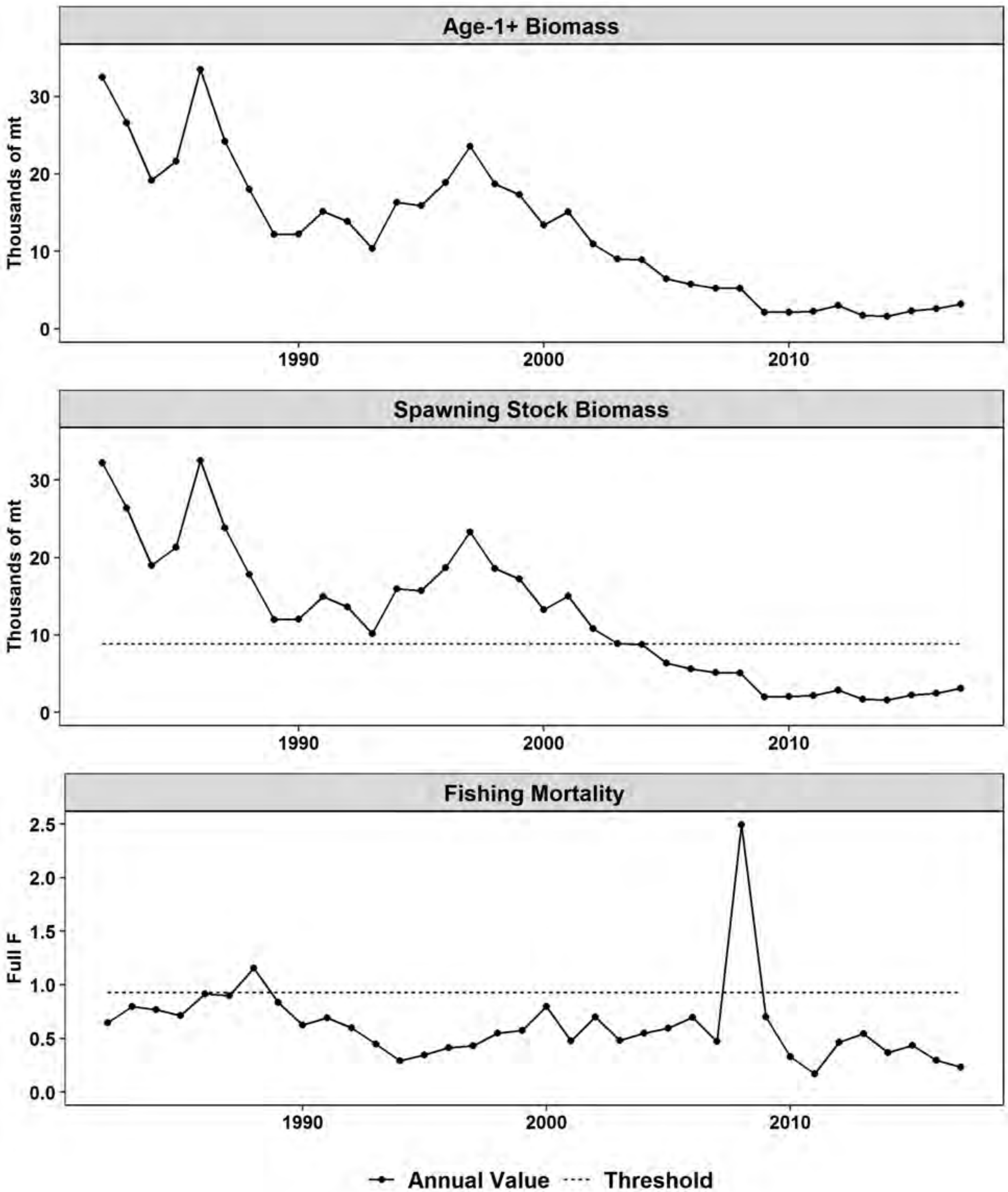


Figure 19. Age-1+ biomass, spawning stock biomass, and full *F* for weakfish, plotted with their respective thresholds, where defined. Estimates are from a preliminary assessment update with data through 2017 and may not match values used for management.

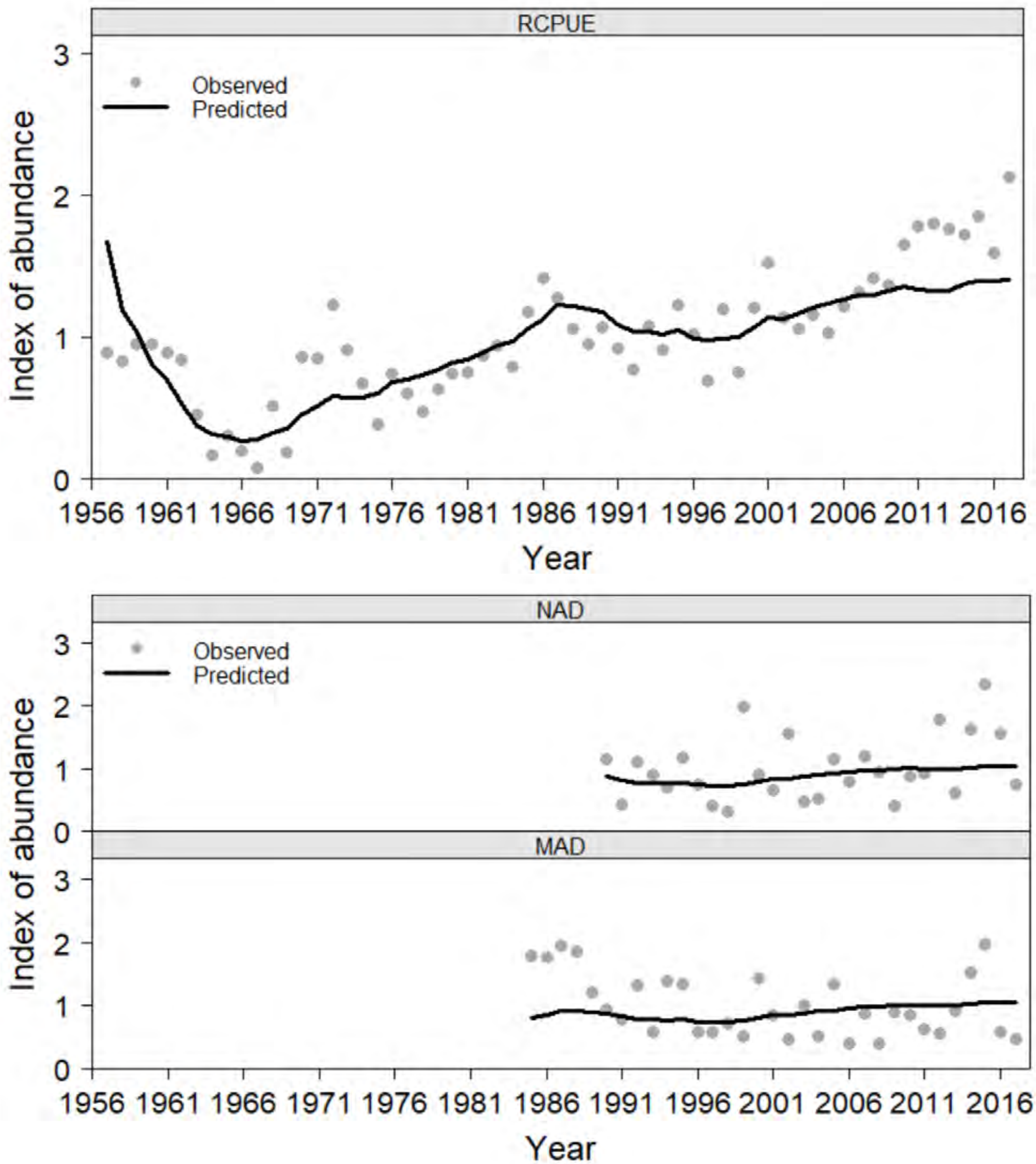


Figure 20. **Observed indices of Atlantic menhaden abundance and estimated values predicted by the SPMTVr.**

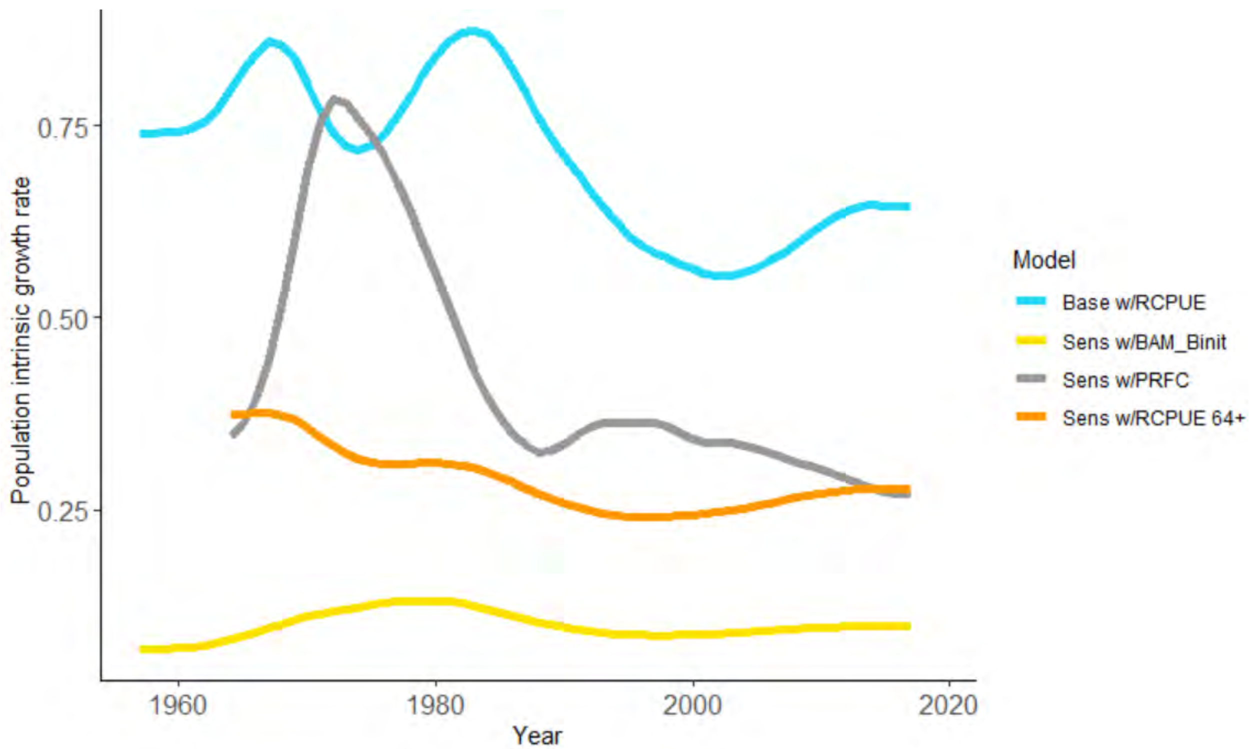


Figure 21. Comparison of estimated trend in population intrinsic growth rate (*r*) for Atlantic menhaden generated by the SPMTVr base model ("Base with RCPUE") with that of sensitivity runs examining alternate model starting year ("Sens w/RCPUE 1964+"), an alternate fishery-dependent abundance index ("Sens w/PRFC"), and an alternate starting value for initial biomass ("Sens w/BAM Binit").

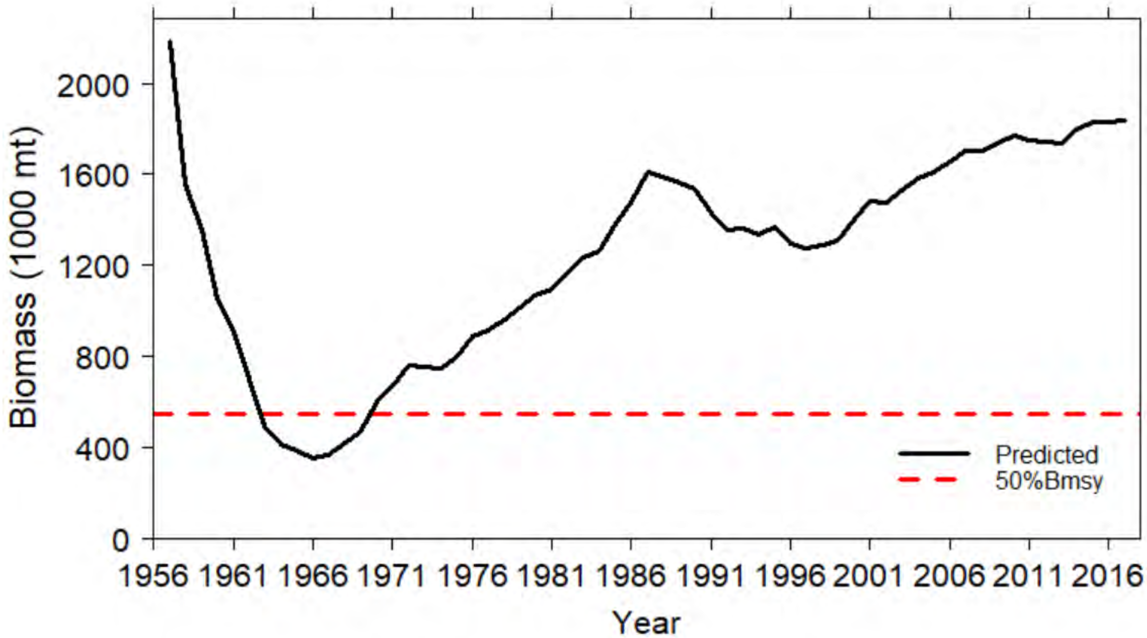


Figure 22. **Trend in total biomass estimated by the SPMTVr relative to an overfished threshold of 50% B_{MSY}.**

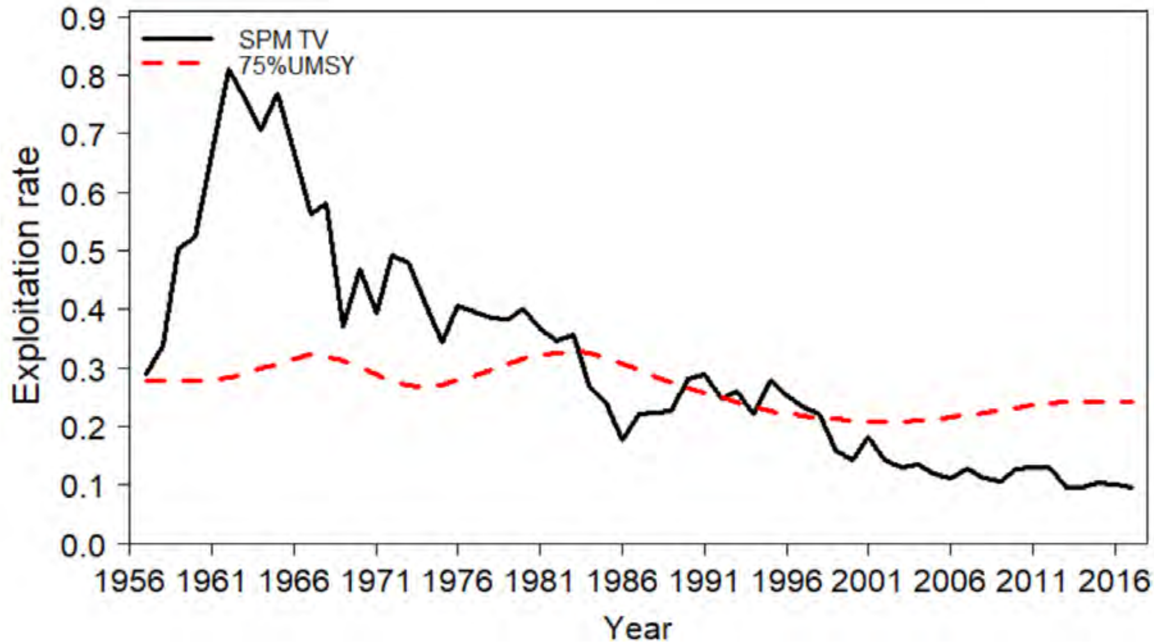


Figure 23. Exploitation rate estimated by the SPMTVr plotted with an overfishing threshold of 75% of the exploitation rate for maximum sustainable yield (U_{MSY}) which varies annually with trends in *r*.

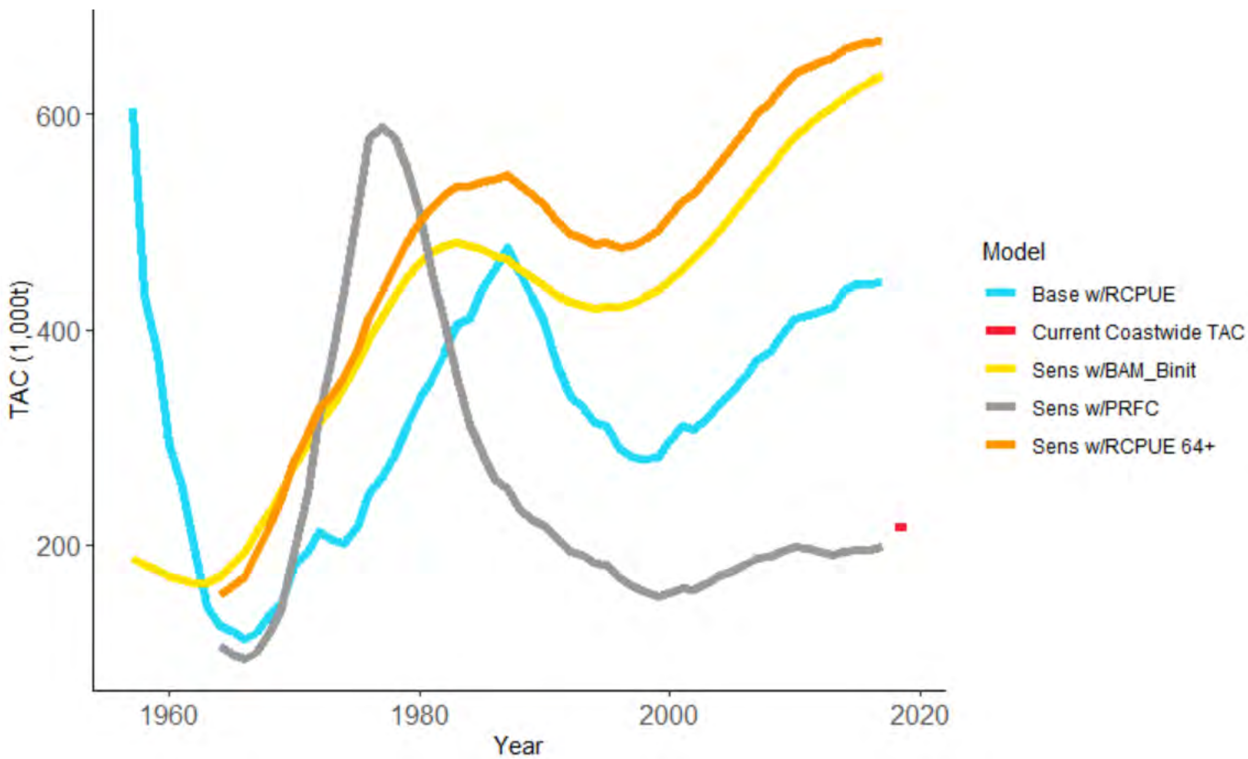


Figure 24. **A comparison of annual TAC estimates produced by the SPMTVr model's base run ("Base with RCPUE") with that of sensitivity runs examining alternate model starting year ("Sens w/RCPUE 1964+"), an alternate fishery-dependent abundance index ("Sens w/PRFC"), and an alternate starting value for initial biomass ("Sens w/BAM Binit").**

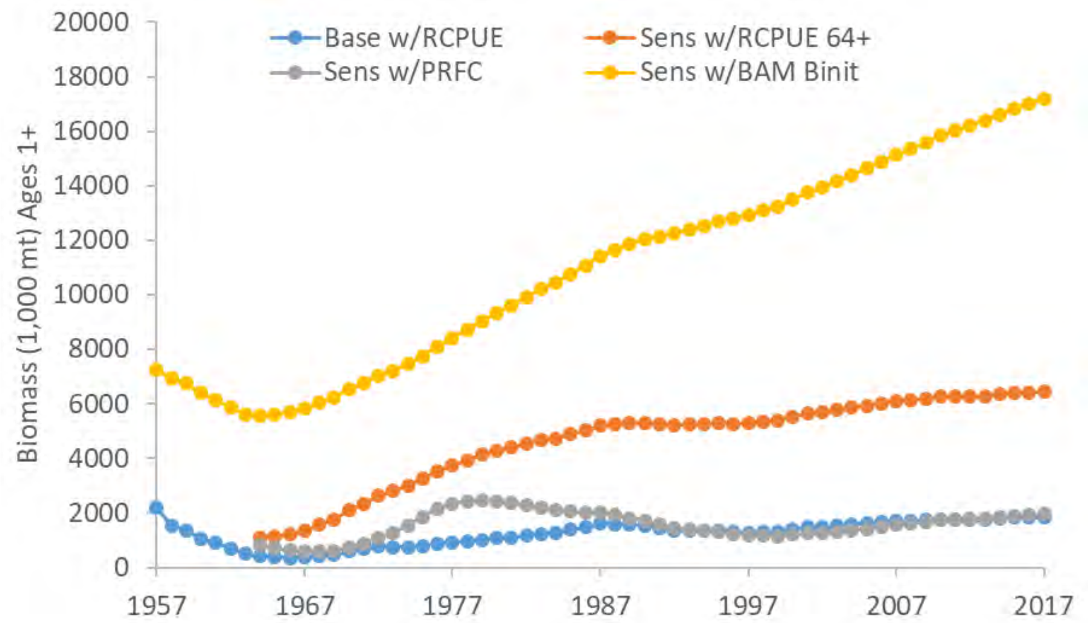


Figure 25. Comparison of base model ("Base with RCPUE") biomass estimates from the SPMTVr model for ages 1+ with that of sensitivity runs examining alternate model starting year ("Sens w/RCPUE 1964+"), an alternate fishery-dependent abundance index ("Sens w/PRFC"), and an alternate starting value for initial biomass ("Sens w/BAM Binit").

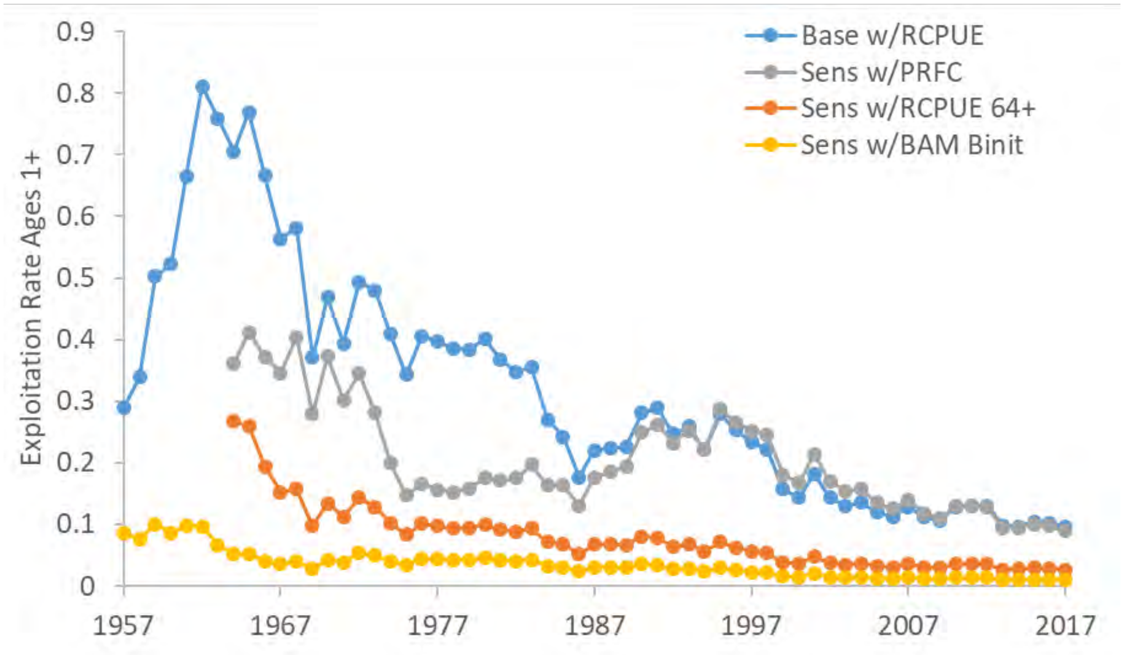


Figure 26. **Comparison of base model ("Base with RCPUE") exploitation rate estimates from the SPMTVr model for ages 1+ with that of sensitivity runs examining alternate model starting year ("Sens w/RCPUE 1964+"), an alternate fishery-dependent abundance index ("Sens w/PRFC"), and an alternate starting value for initial biomass ("Sens w/BAM Binit").**

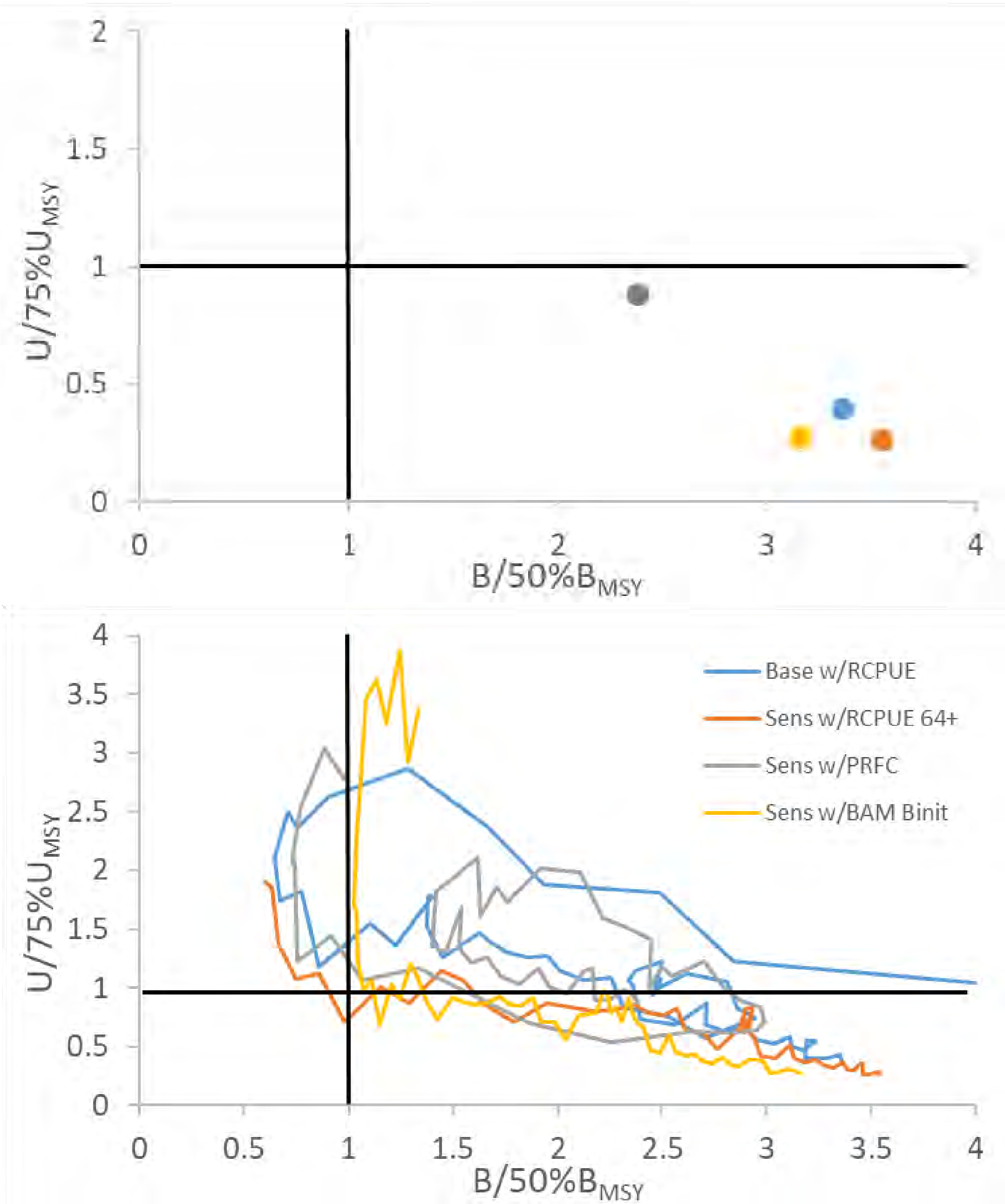


Figure 27. **Kobe plots of stock status diagram for the SPMTVr model comparing base model ("Base with RCPUE") stock status estimates with that of sensitivity runs examining alternate model starting year ("Sens w/RCPUE 1964+"), an alternate fishery-dependent abundance index ("Sens w/PRFC"), and an alternate starting value for initial biomass ("Sens w/BAM Binit"). Top panel displays stock status in the terminal year for each model and the bottom panel displays annual stock status relative to time-varying reference points.**

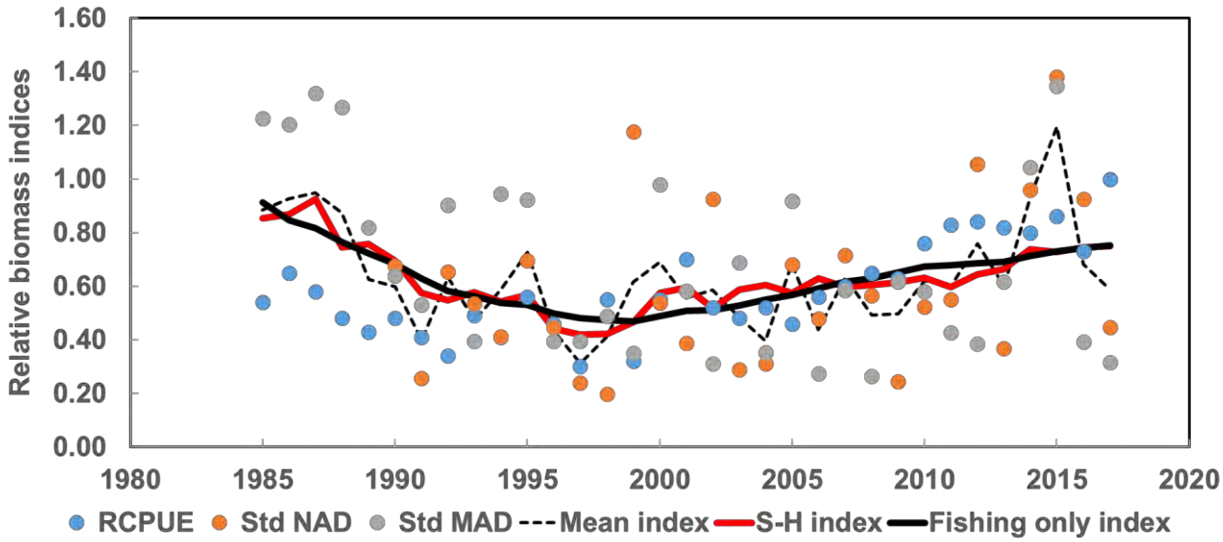


Figure 28. **Time-series of observed age-1+ Atlantic menhaden relative biomass indices, their average, and the values predicted by the fishing-only surplus production model (Fishing only index) and base Steele-Henderson model (S-H index; fishing and striped bass predation). The NAD and MAD indices are standardized into RCPUE units. Mean = average of each year's available indices.**

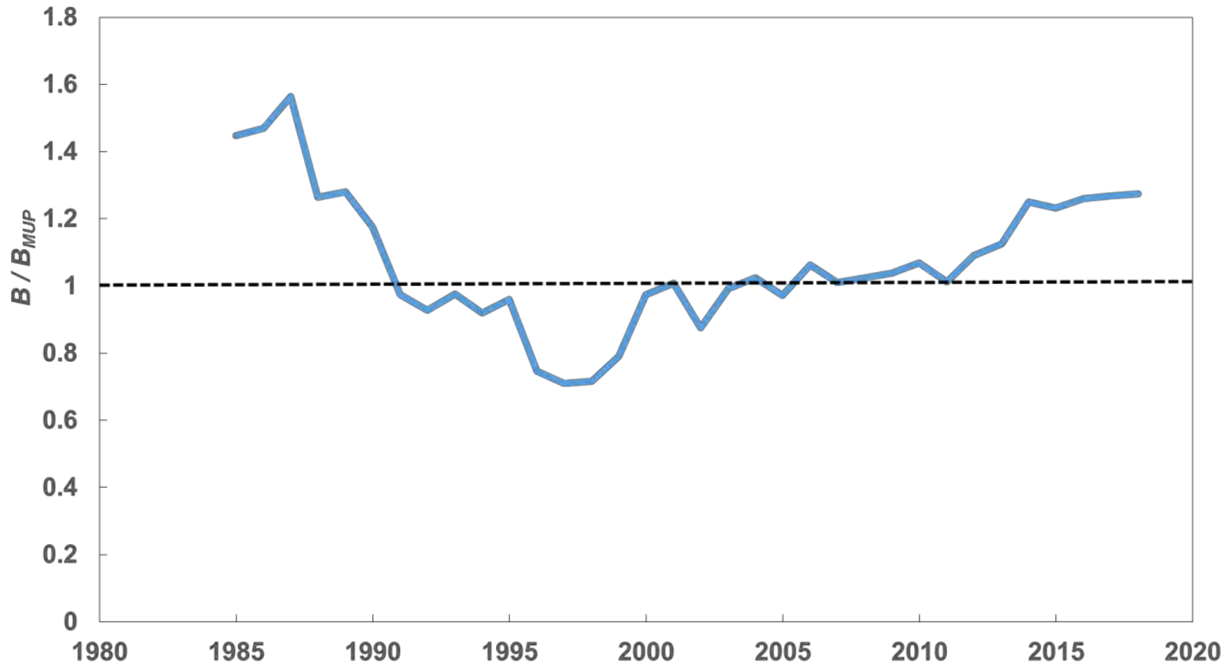


Figure 29. **Relative biomass estimates (*B/B_{MUP}*) from base Steele-Henderson (fishing plus striped bass predation) model. Values less than 1.0 breached the threshold.**

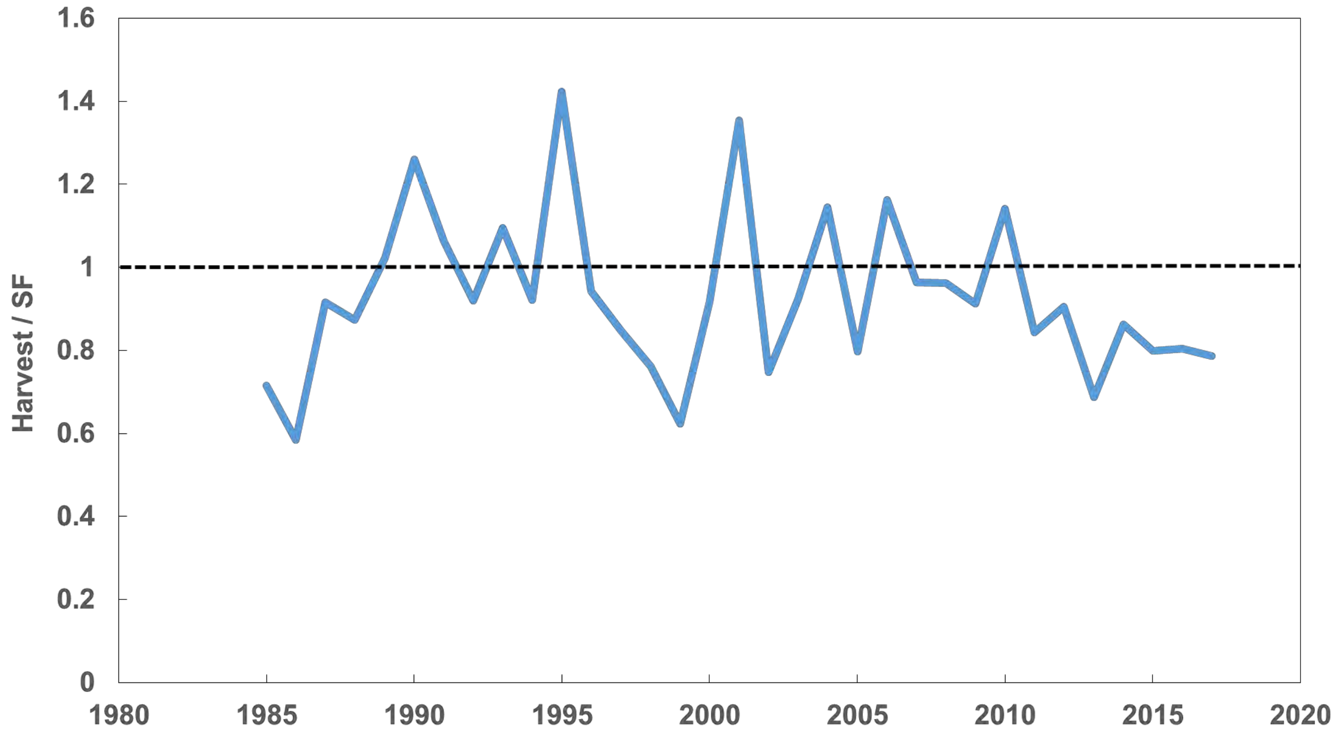


Figure 30. Harvest divided by surplus production available to the fishery after predation losses (SF) from base Steele-Henderson model (fishing plus striped bass predation). Values at 1.0 or more exceeded the threshold.

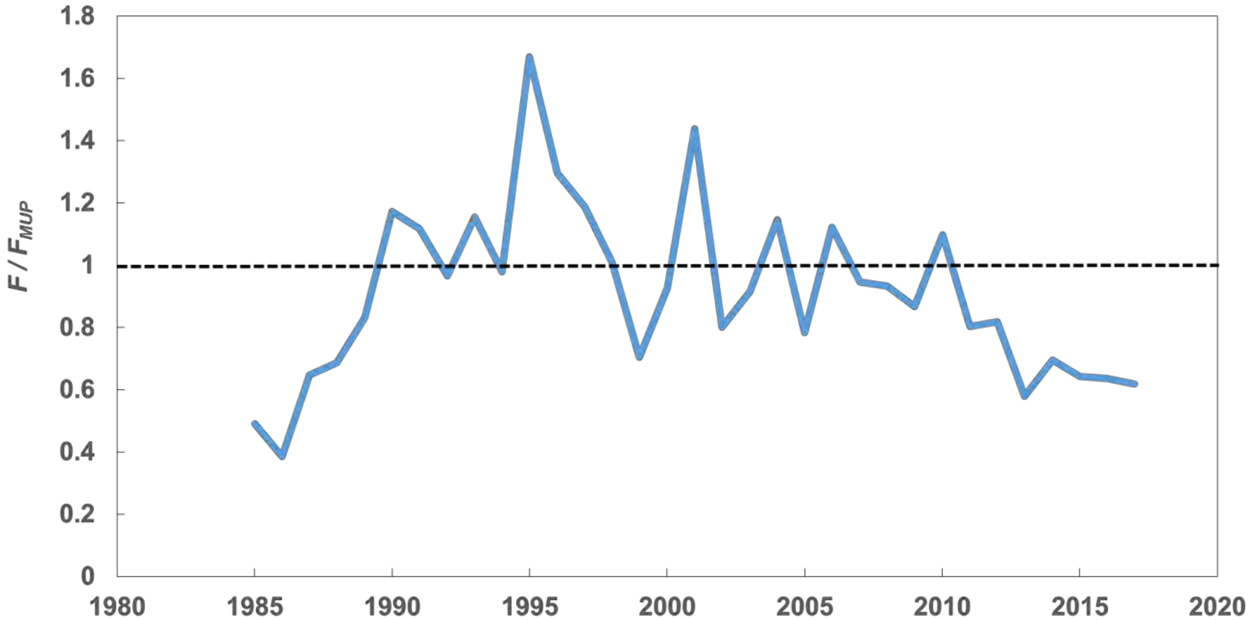


Figure 31. Relative F (F = F/F_{MUP}) estimates from base Steele-Henderson model (fishing and Striped Bass predation). Values at 1.0 or more breached the threshold.

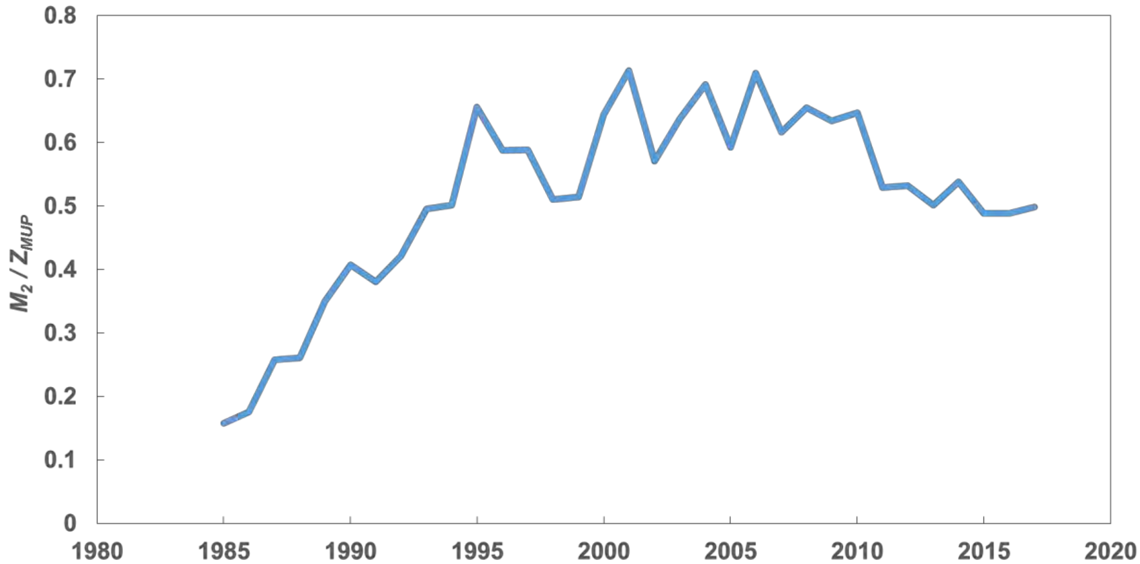


Figure 32. **Relative M_2 (M_2 / Z_{MUP}) estimates from base Steele-Henderson models (fishing and striped bass predation).**

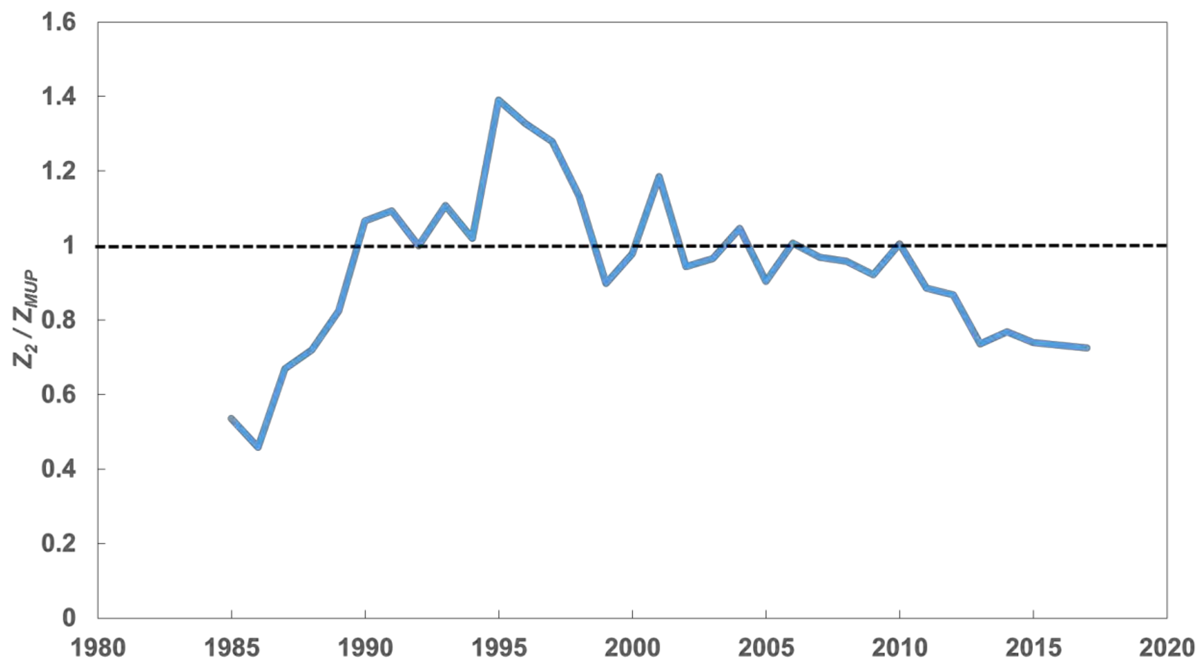


Figure 33. **Relative Z_2 estimates from base Steele-Henderson models (fishing and striped bass predation). Relative $Z_2 = Z_2 / Z_{MUP}$ for Steele-Henderson models. Values at 1.0 or more exceeded the threshold.**

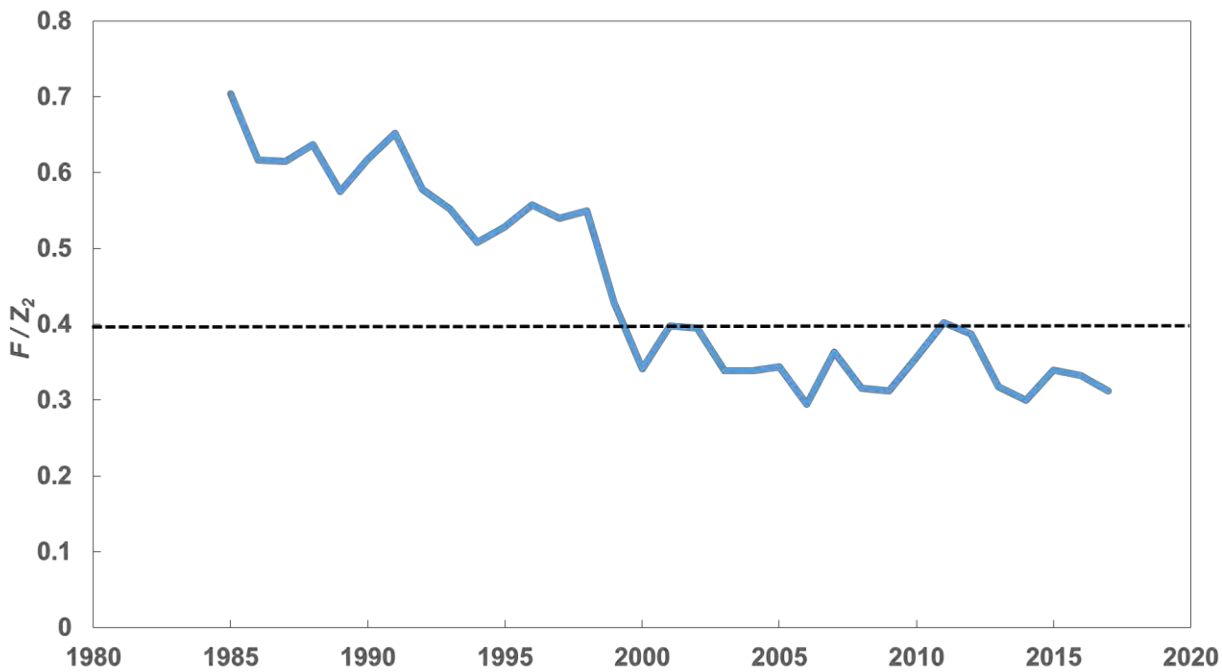


Figure 34. **Estimates of *F / Z_2* from base Steele-Henderson models (fishing and striped bass predation). Values at 0.4 (dashed line) or more exceeded the threshold.**

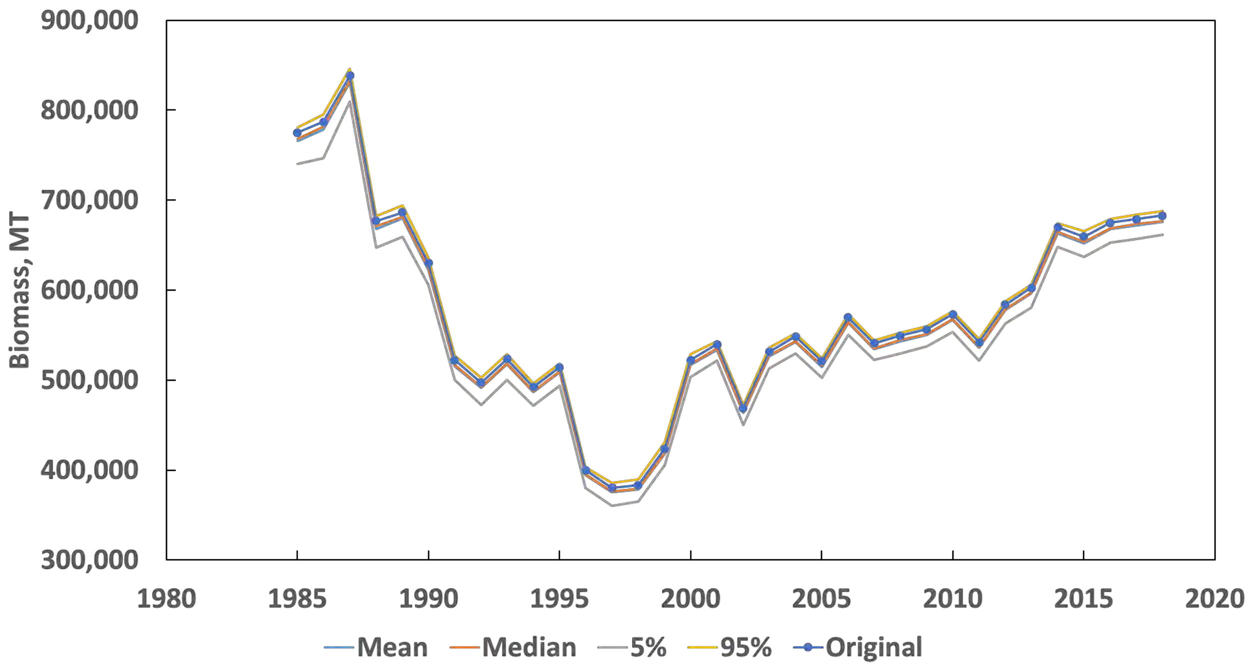


Figure 35. Time-series of age-1+ Atlantic menhaden biomass estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile). MT = metric tons.

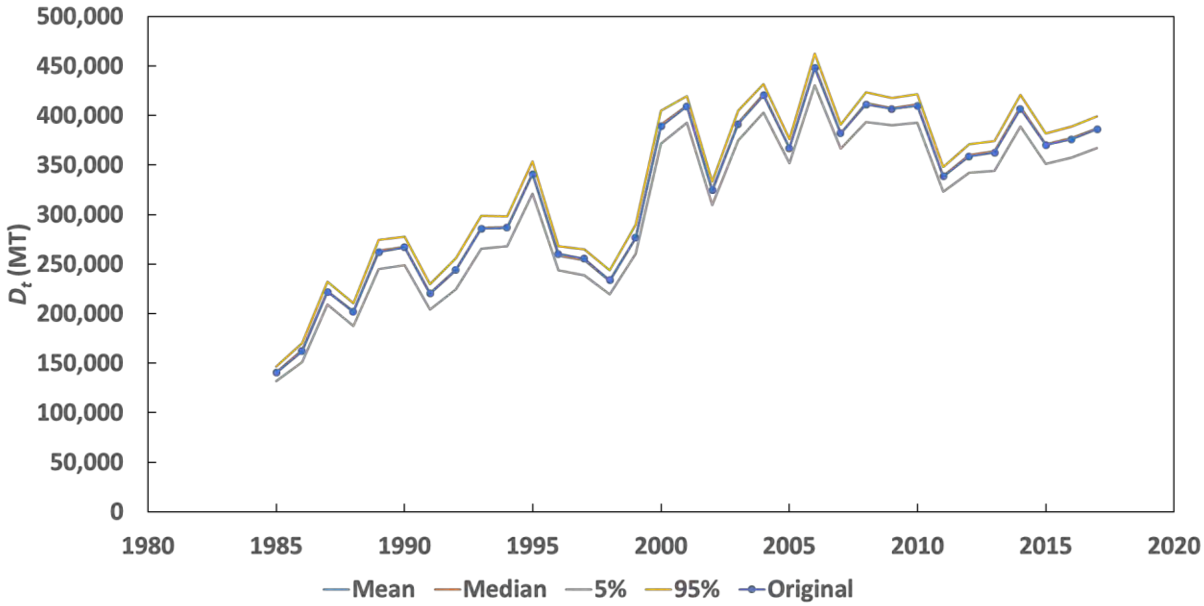


Figure 36. **Time-series of age-1+ Atlantic menhaden biomass consumed by striped bass (D_t) estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile). MT = metric tons.**

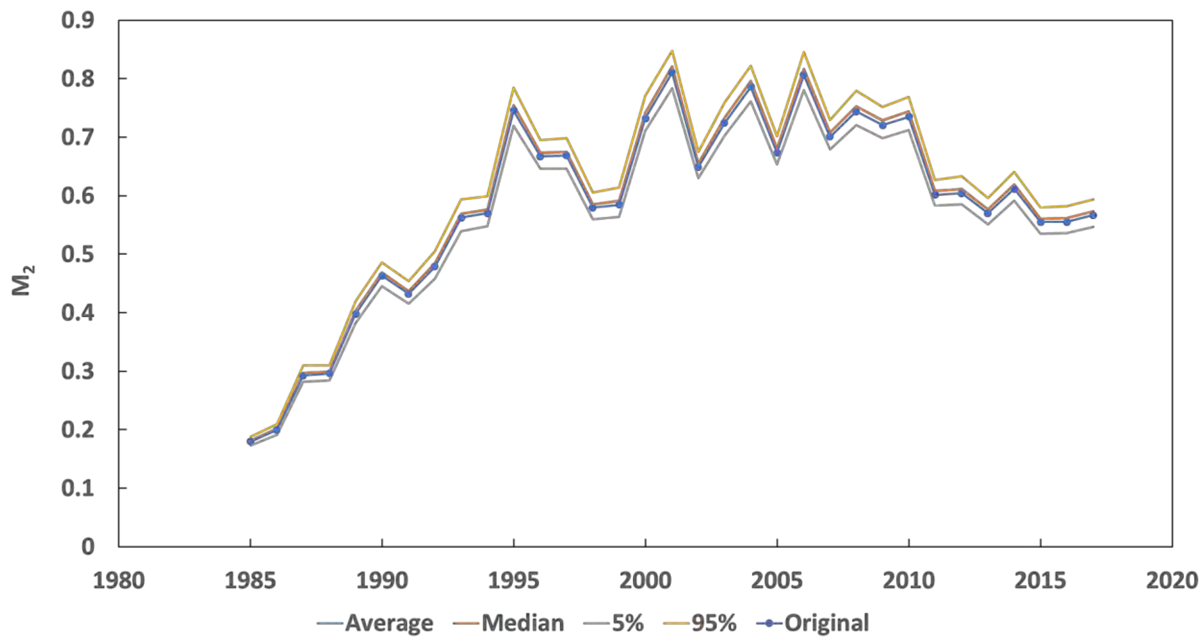


Figure 37. Time-series of age-1+ Atlantic menhaden biomass M_2 estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile).

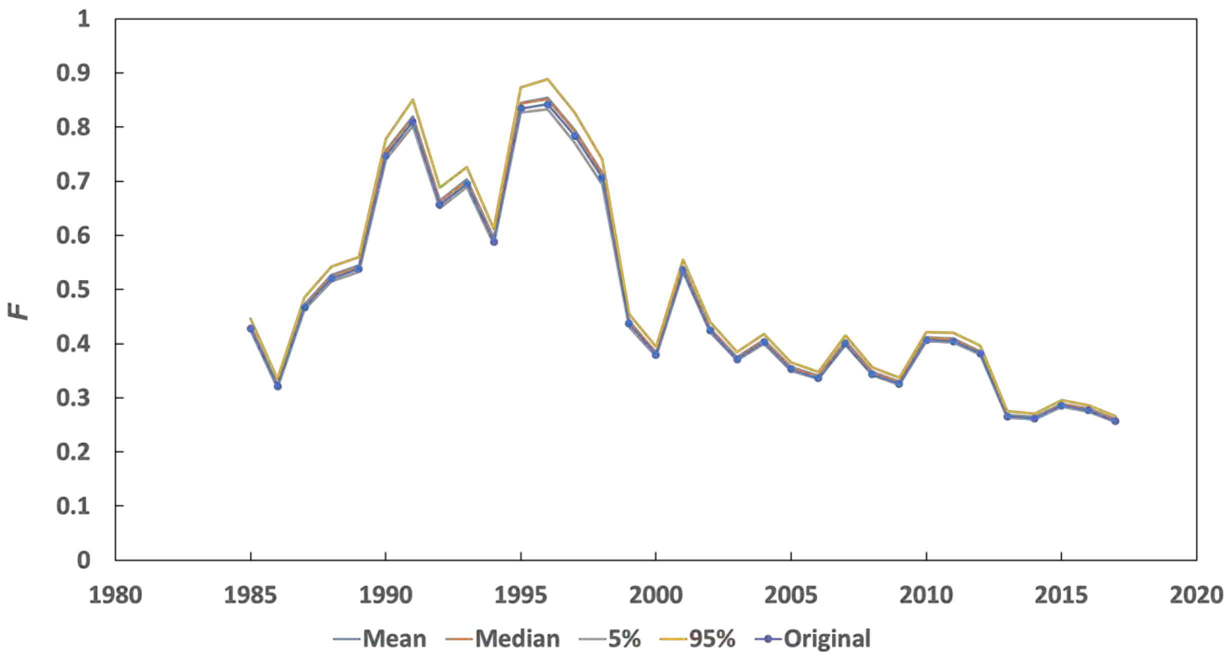


Figure 38. Time-series of ages 1+ Atlantic menhaden biomass *F* estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile).

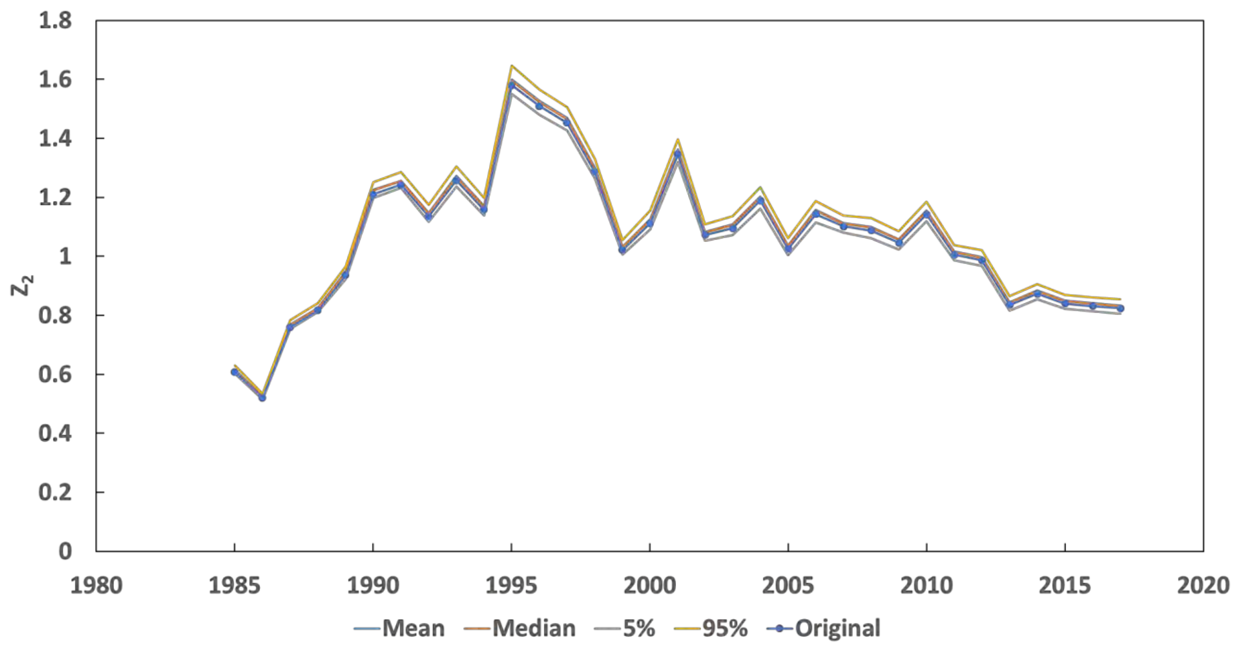


Figure 39. Time-series of age-1+ Atlantic menhaden biomass Z_2 (F + M_2) estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile).

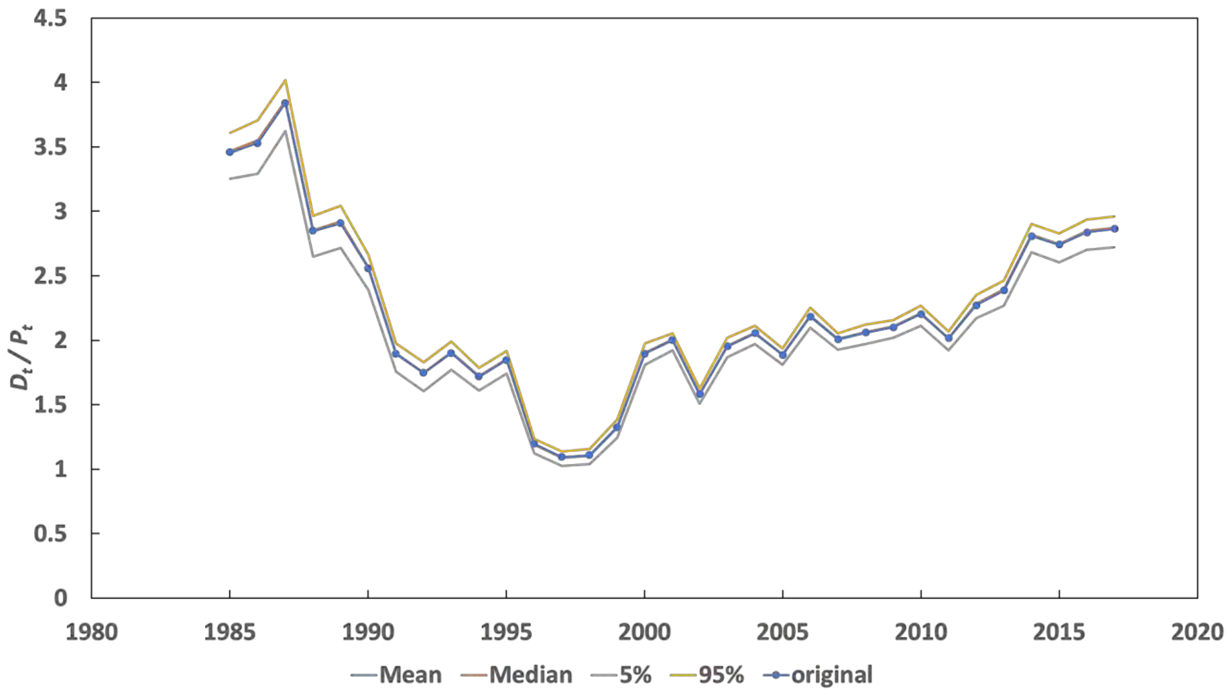


Figure 40. **Time-series of annual age-1+ Atlantic menhaden biomass consumed per striped bass biomass (D_t / P_t as MT consumed / MT striped bass) estimated by the base Steele-Henderson model (fishing and striped bass predation), and distribution of its jackknifed estimates (mean, median, 5th percentile, and 95th percentile). MT = metric tons.**

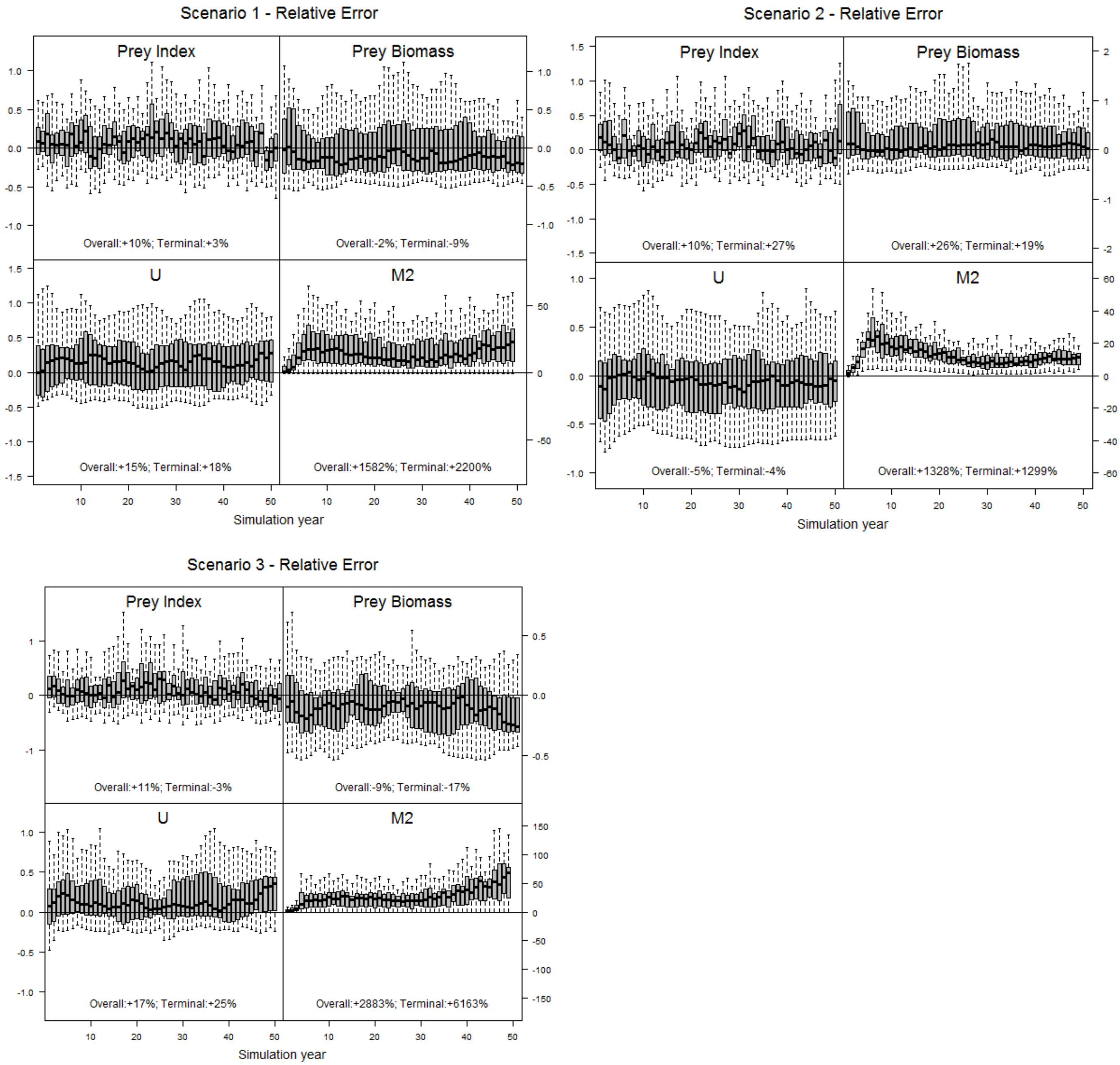


Figure 41. **Relative error from Steele-Henderson model results using simulated data. Scenario 1: predator *F* is constant; Scenario 2: predator *F* increasing; Scenario 3: predator *F* decreasing. In all scenarios prey *F* increases and then decreases.**

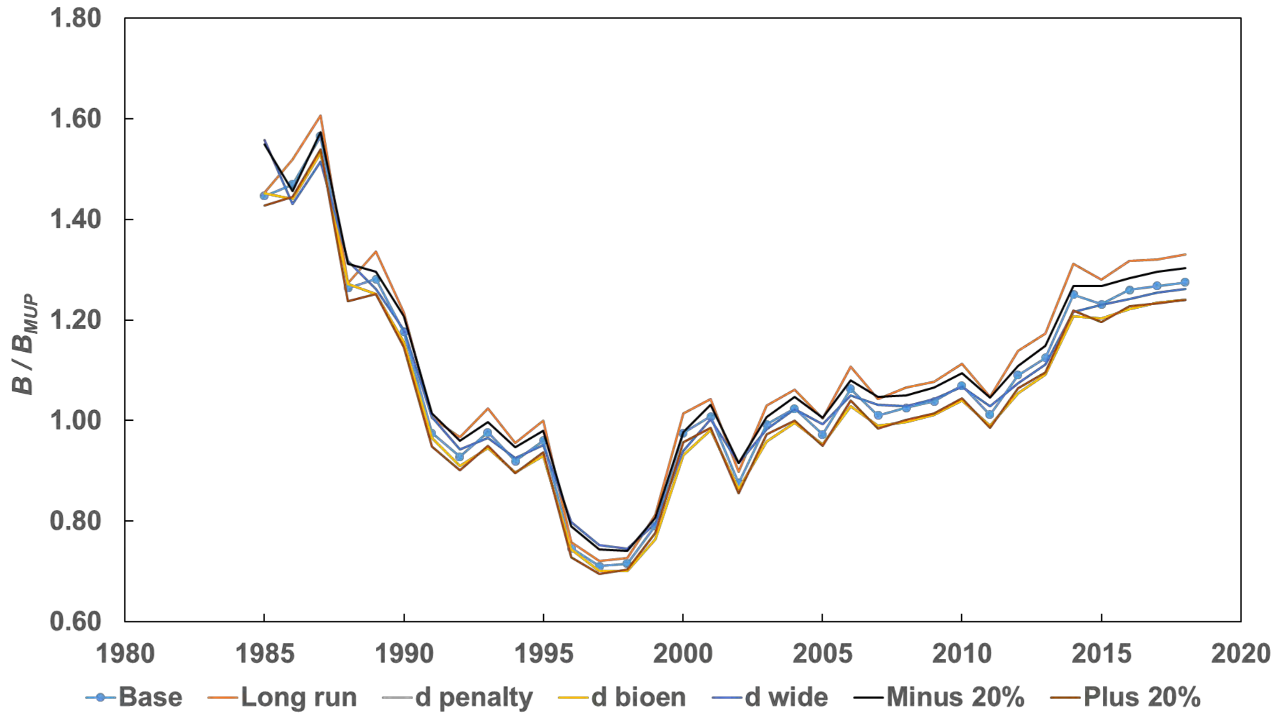


Figure 42. **Relative biomass (B / B_{MUP}) estimates from the base Steele-Henderson model (fishing and striped bass predation) and its sensitivity runs. Values at 1.0 or less exceeded the threshold. See Section 11.4 for descriptions of sensitivity runs.**

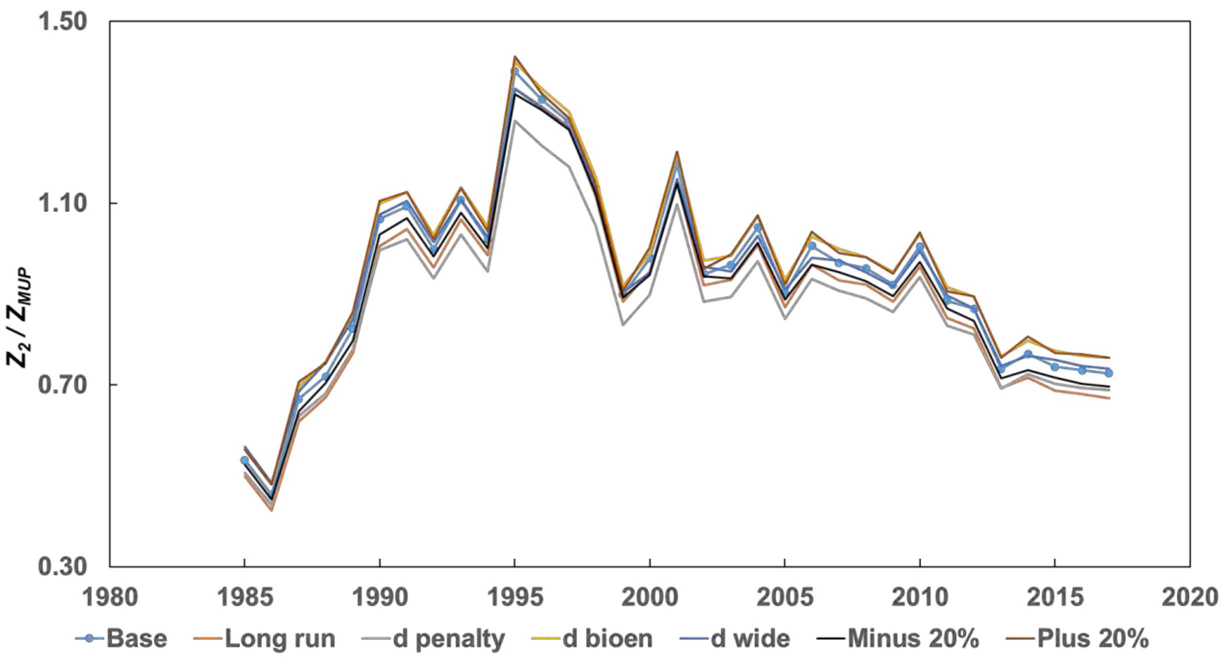


Figure 43. Relative Z_2 (Z_2 / Z_{MUP}) estimates from base Steele-Henderson model (fishing and striped bass predation) sensitivity runs. Values at 1.0 or more exceeded the threshold. See Methods for descriptions of sensitivity runs.

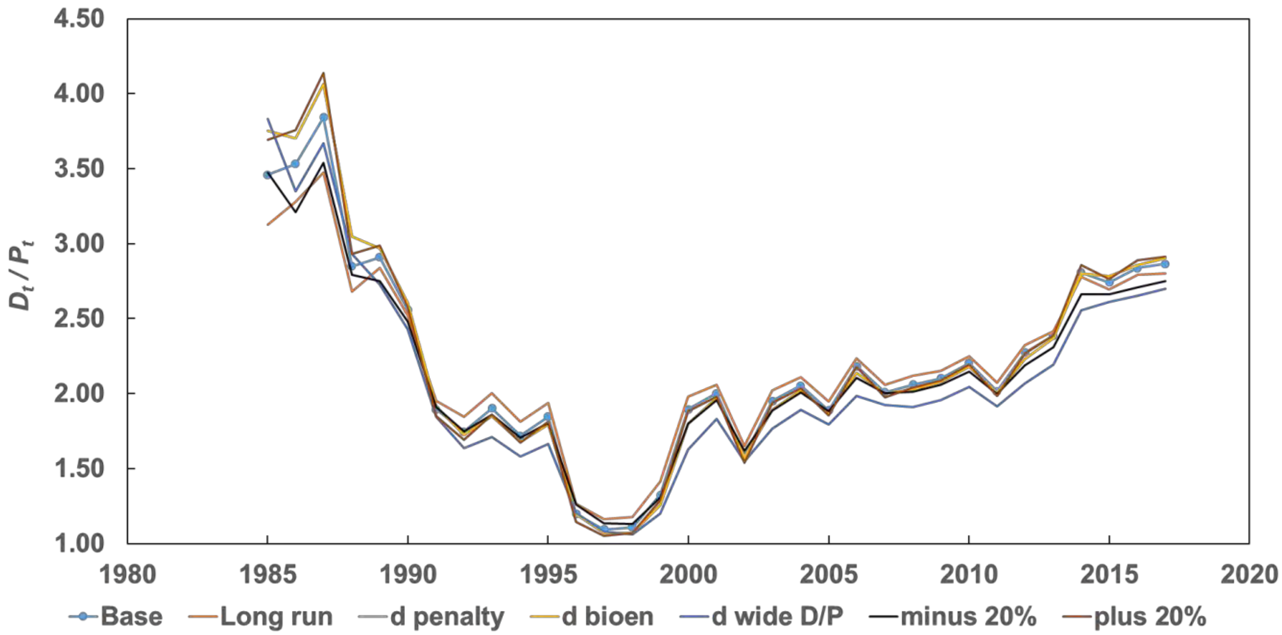


Figure 44. **Time-series of annual age-1+ Atlantic menhaden biomass consumed per striped bass biomass from the base Steele-Henderson model and its sensitivity runs (D_t / P_t as mt consumed / mt striped bass).**

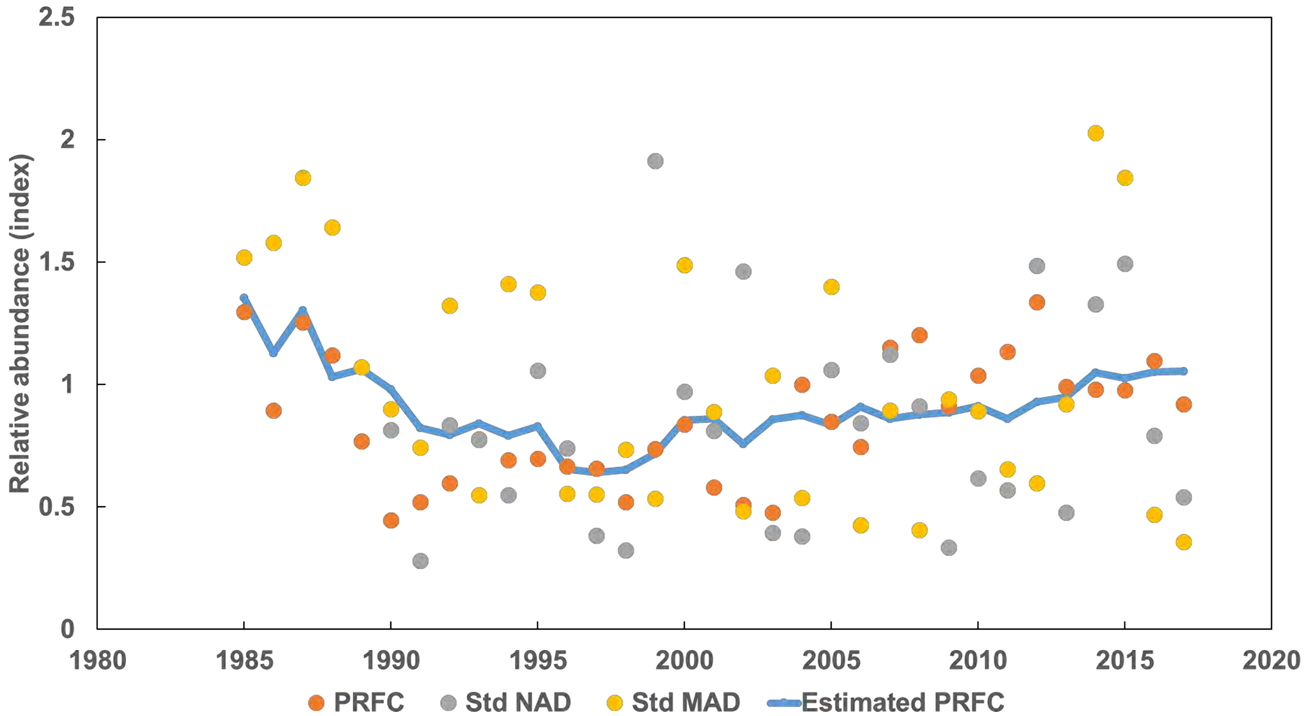


Figure 45. **Time-series of observed and predicted age-1+ Atlantic menhaden relative biomass indices from the Steele-Henderson model (fishing and striped bass predation) fit using the PRFC index. NAD and MAD indices are standardized into PRFC units.**

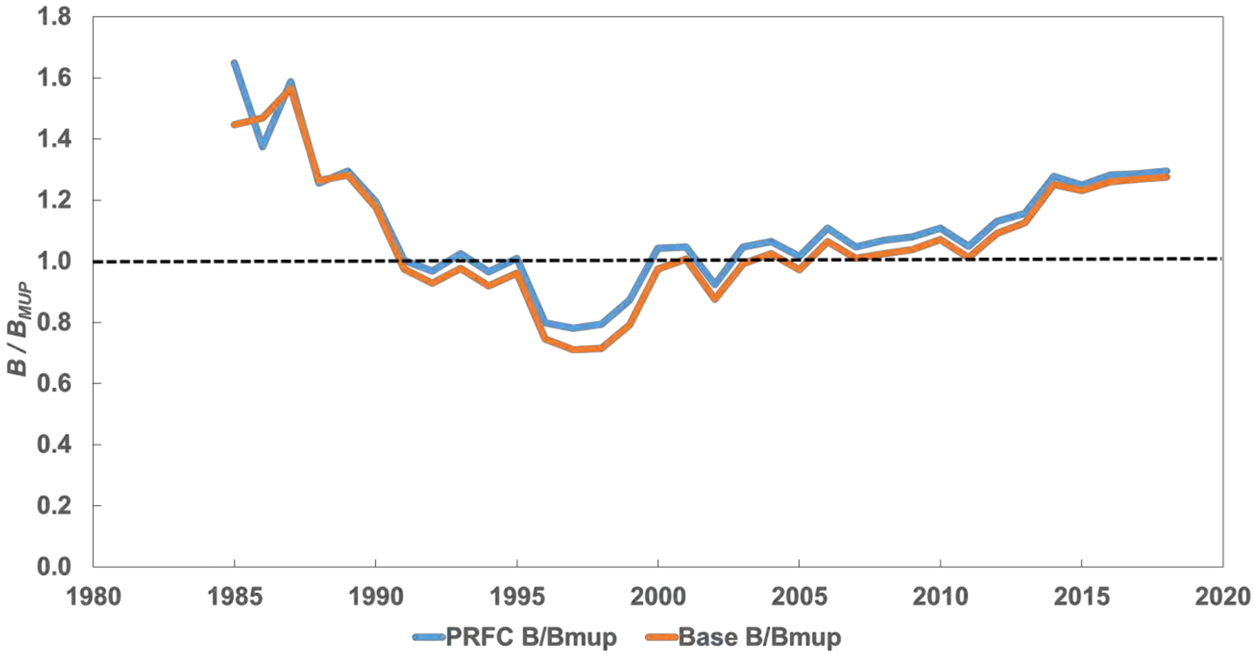


Figure 46. Relative biomass estimates from base and PRFC Steele-Henderson models (fishing and striped bass predation). Relative biomass = B / B_{MUP}. Values at 1.0 or less exceeded the threshold.

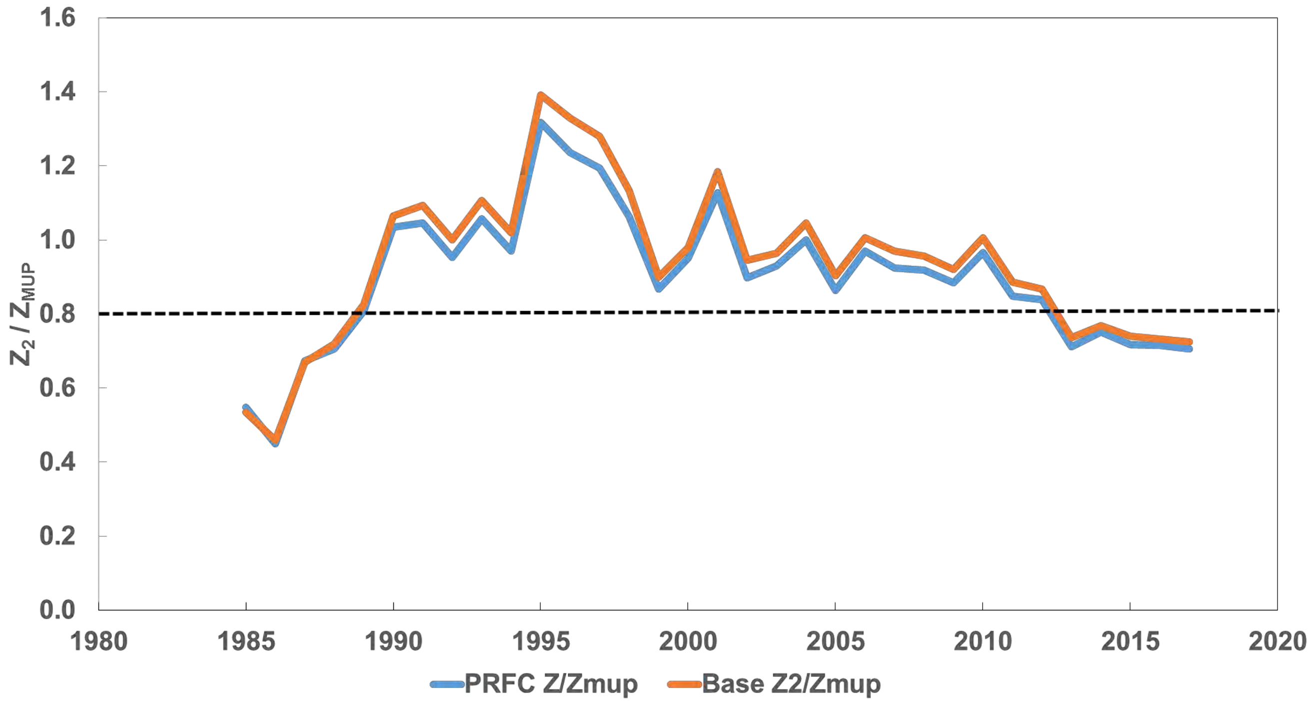


Figure 47. Relative Z_2 estimates from base and PRFC Steele-Henderson models (fishing and striped bass predation). Relative Z_2 = Z_2 / Z_{MUP}. Values at 1.0 or more exceeded the threshold.

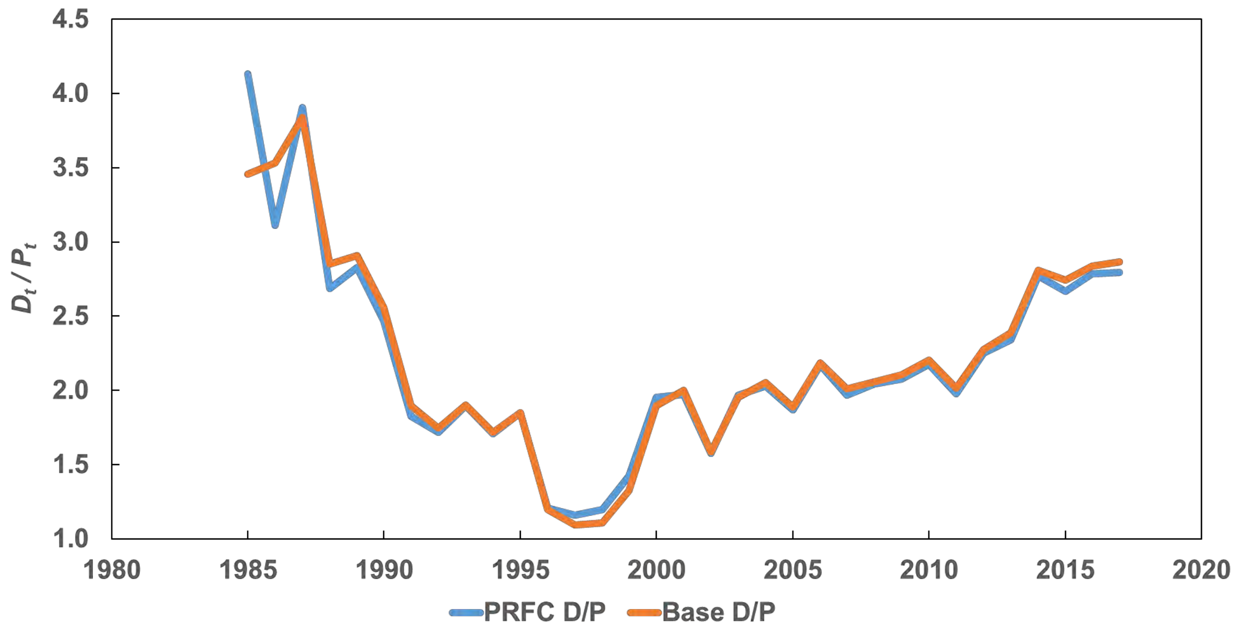


Figure 48. **Time-series of annual age-1+ Atlantic menhaden biomass consumed per striped bass biomass estimated by the base and PRFC Steele-Henderson models (D_t / P_t as mt consumed / mt striped bass).**

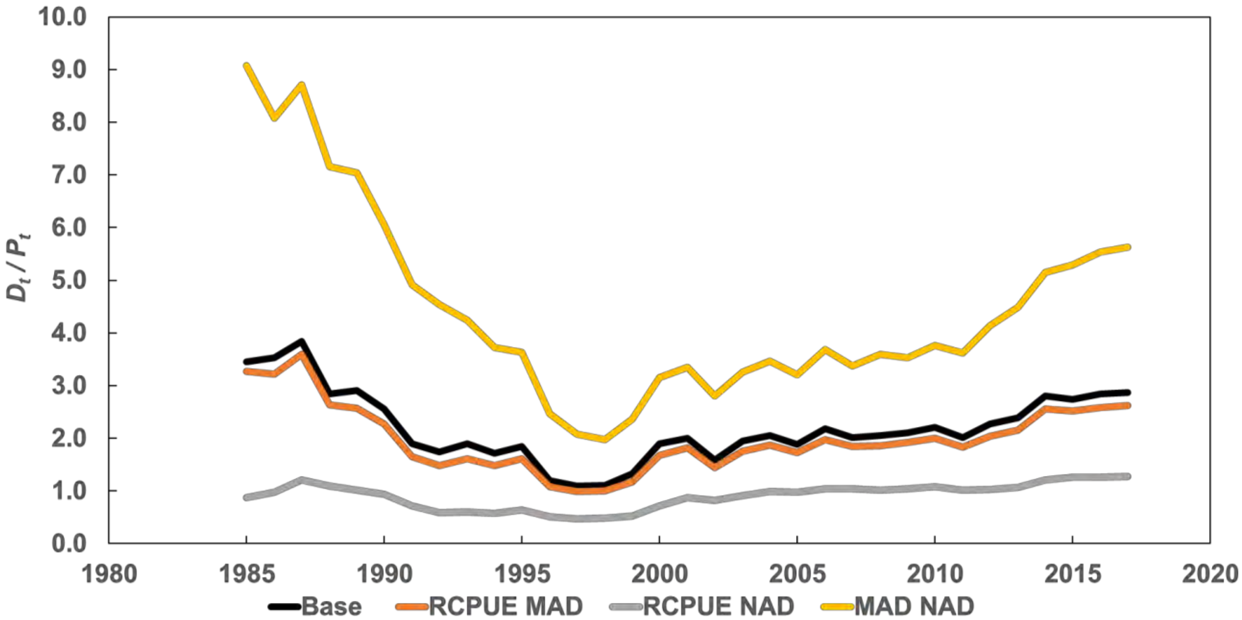


Figure 49. **Time-series of annual age-1+ Atlantic menhaden biomass consumed per striped bass biomass estimated by the base Steele-Henderson model and its index removal runs (D_t / P_t as mt consumed / mt striped bass).**

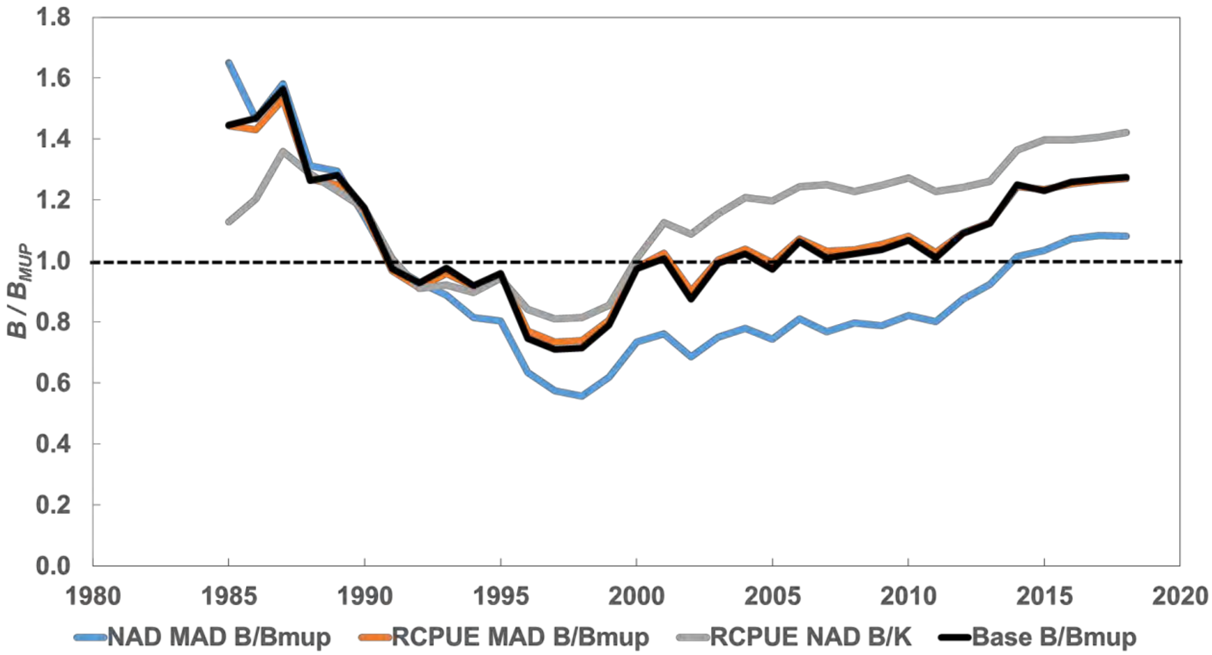


Figure 50. **Relative biomass (B / B_{MUP}) estimates from the base Steele-Henderson model (fishing and striped bass predation) and its index removal runs. Values at 1.0 or less were below the threshold.**

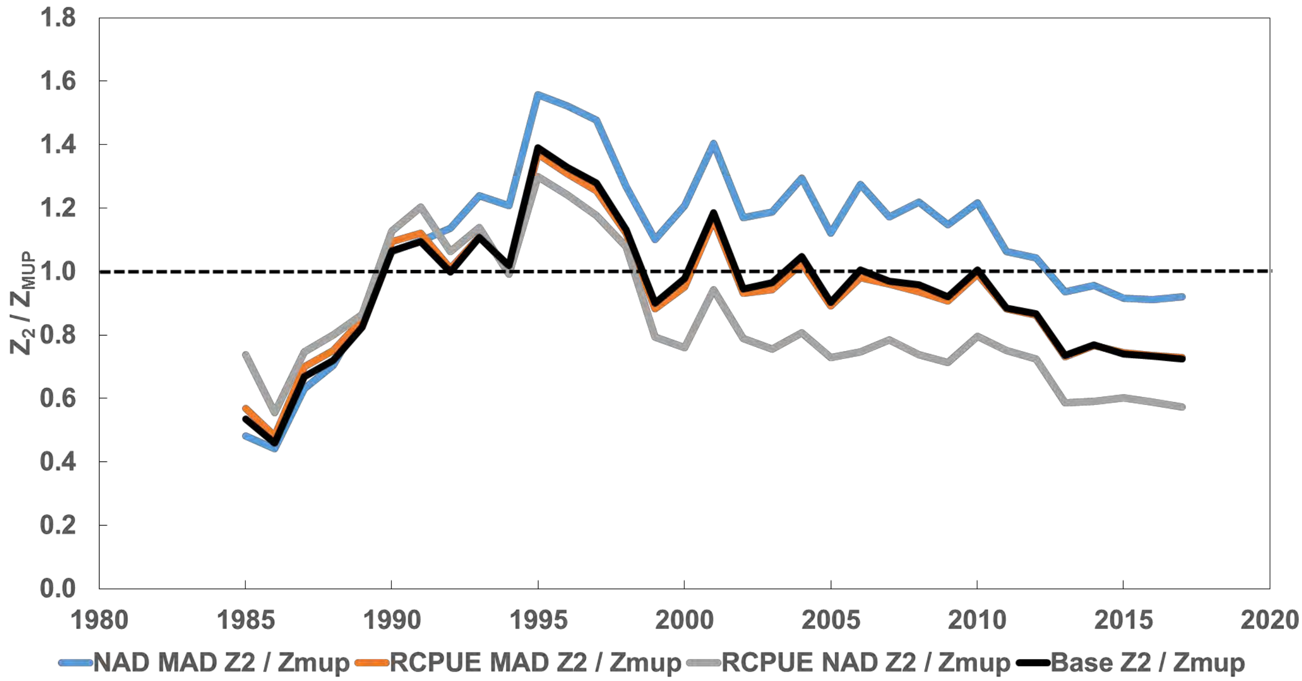


Figure 51. **Relative Z_2 (Z_2 / Z_{MUP}) estimates from base Steele-Henderson model (fishing and striped bass predation) index removal runs. Values at 1.0 or more exceeded the threshold.**

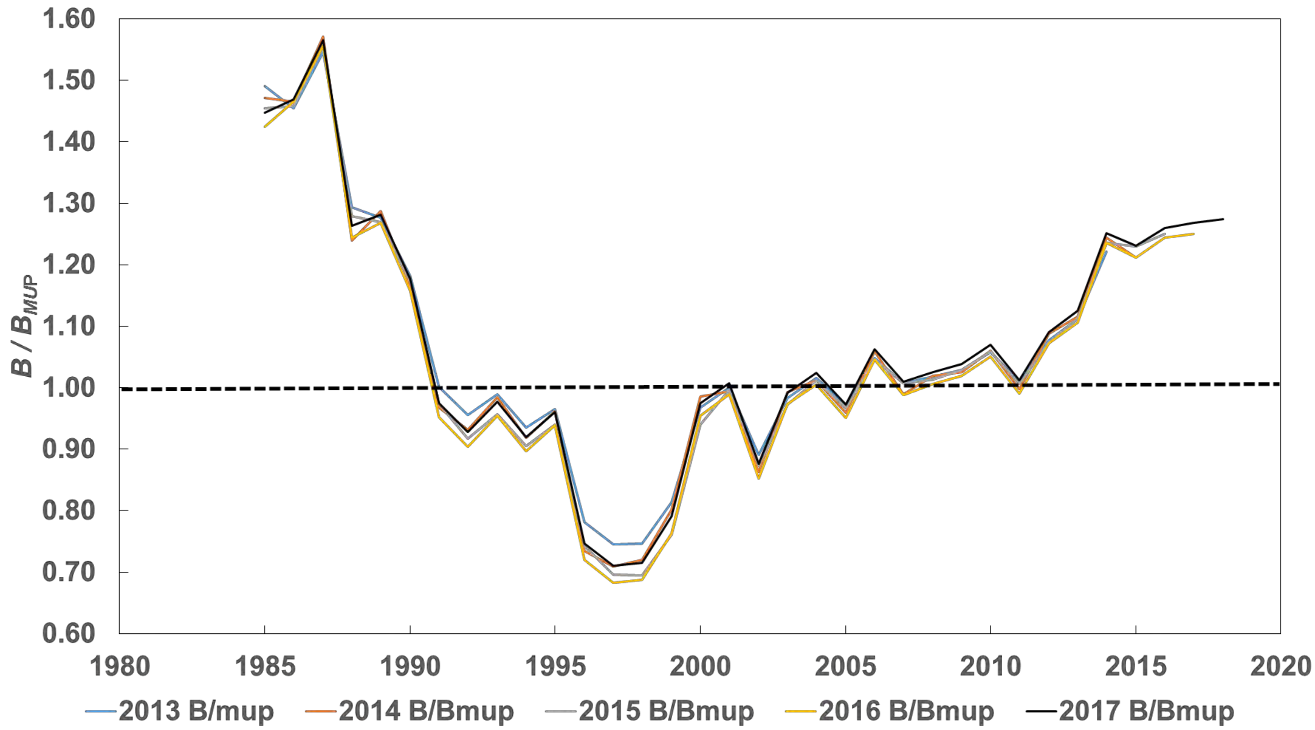


Figure 52. **Relative biomass (B / B_{MUP}) estimates from the base Steele-Henderson model (fishing and striped bass predation) and its retrospective runs. Values at 1.0 or less were below the threshold.**

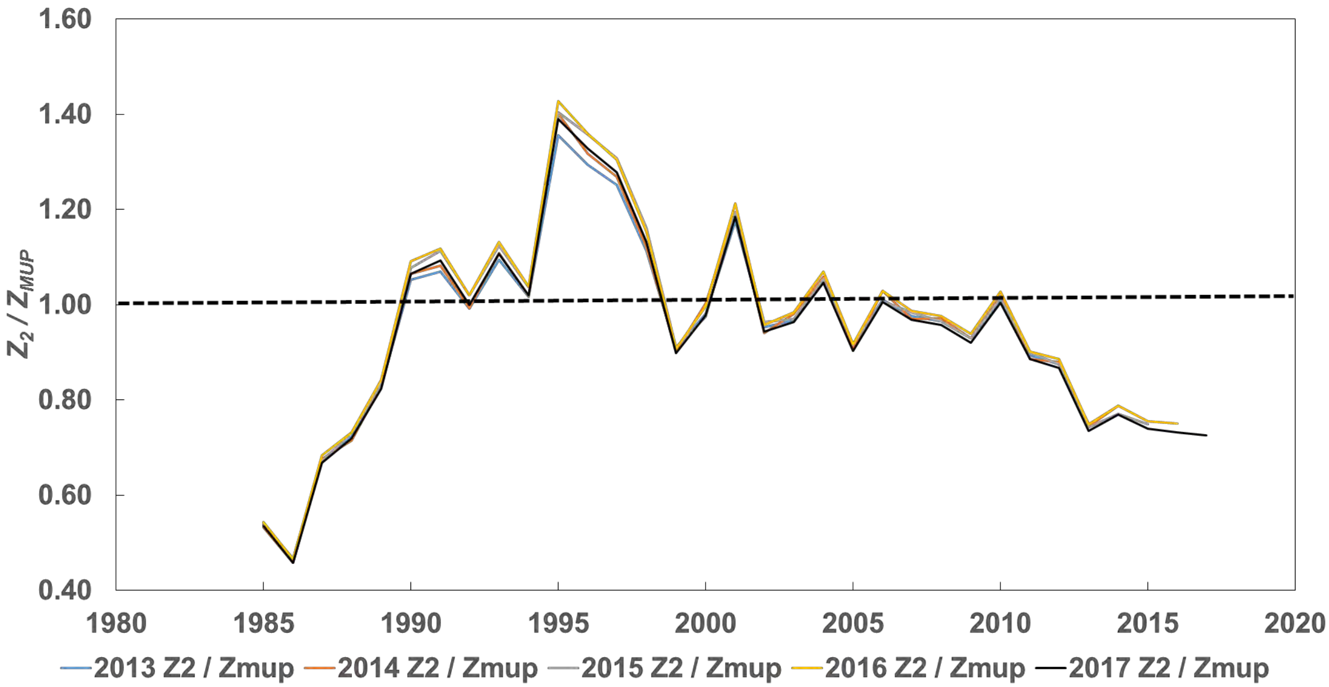


Figure 53. **Relative Z_2 (Z_2 / Z_{MUP}) estimates from base Steele-Henderson model (fishing and striped bass predation) retrospective runs. Values at 1.0 or more exceeded the threshold.**

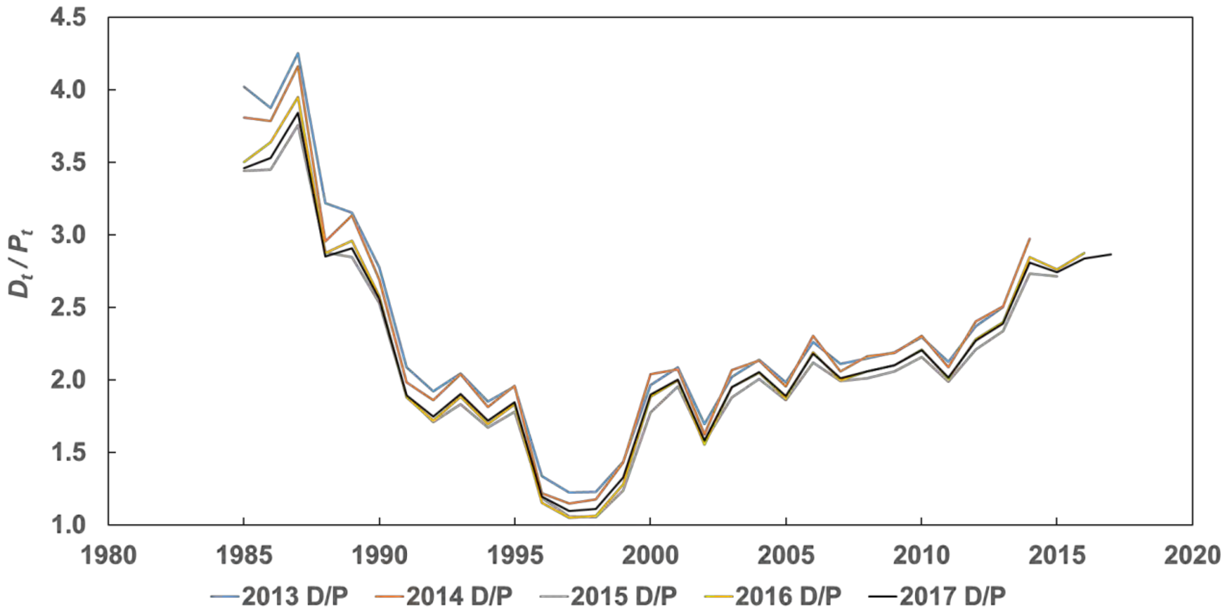


Figure 54. Time-series of annual ages 1+ Atlantic menhaden biomass consumed per striped bass biomass estimated by the base Steele-Henderson model and its retrospective runs (D_t / P_t as mt consumed / mt striped bass).

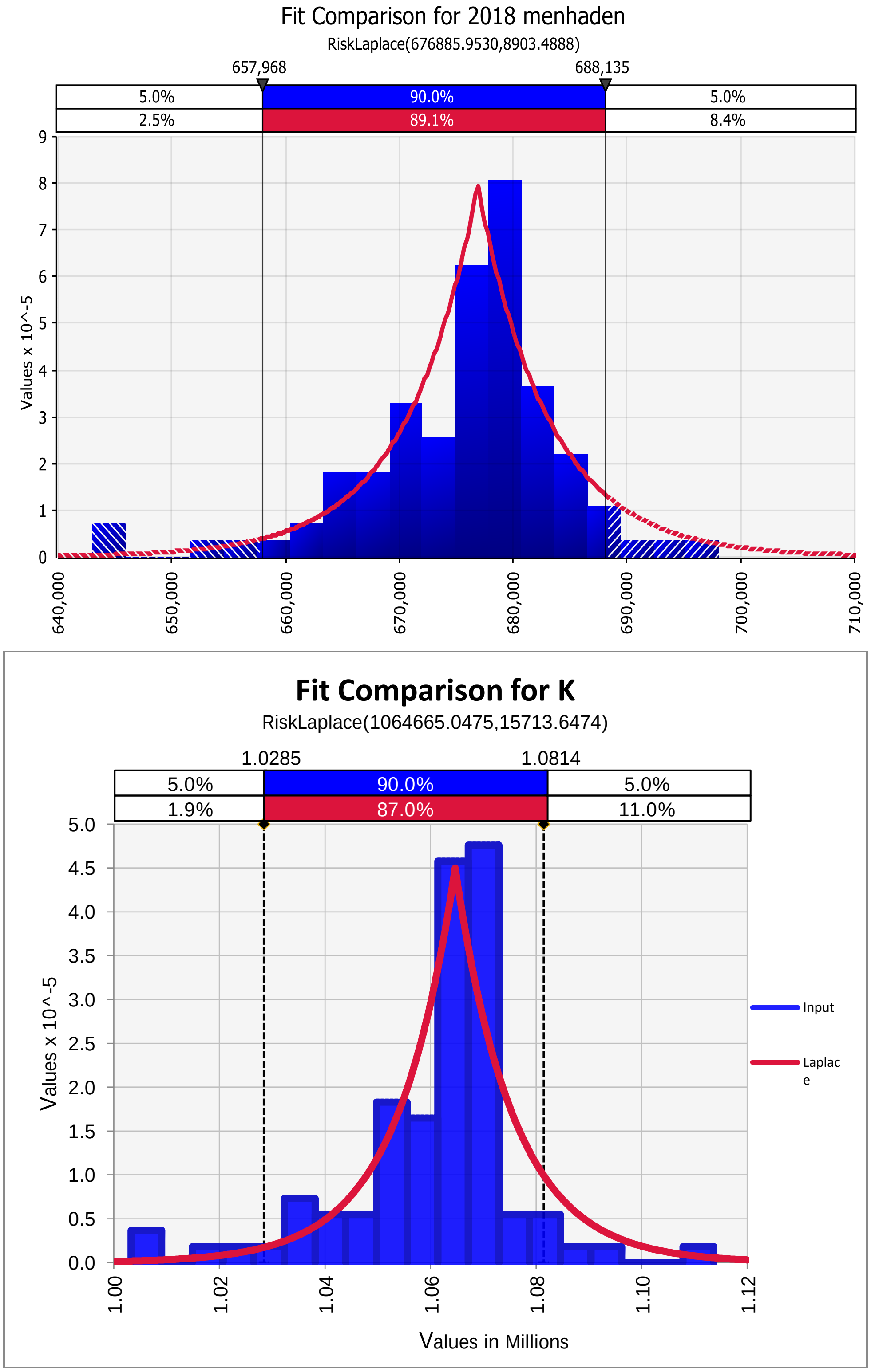
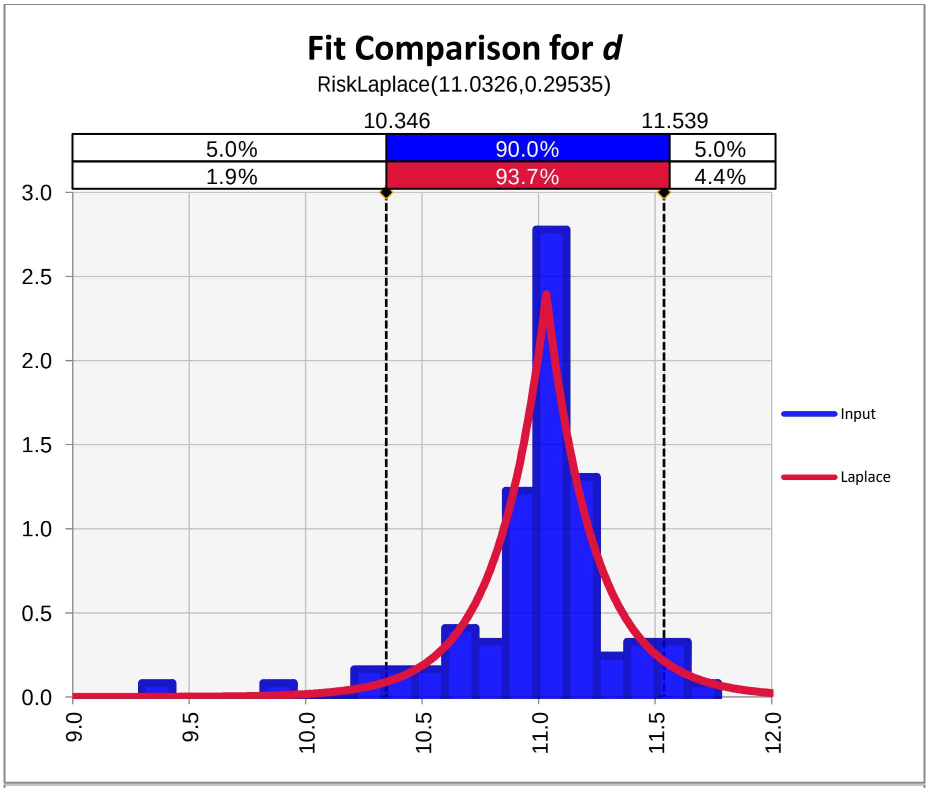


Figure 55. **Base Steele-Henderson model jackknifed distributions of January 1, 2018 Atlantic menhaden ages 1+ biomass (MT) and unfished biomass (*K*, MT) and Laplace distributions providing best fit using @Risk's distribution fitting module.**



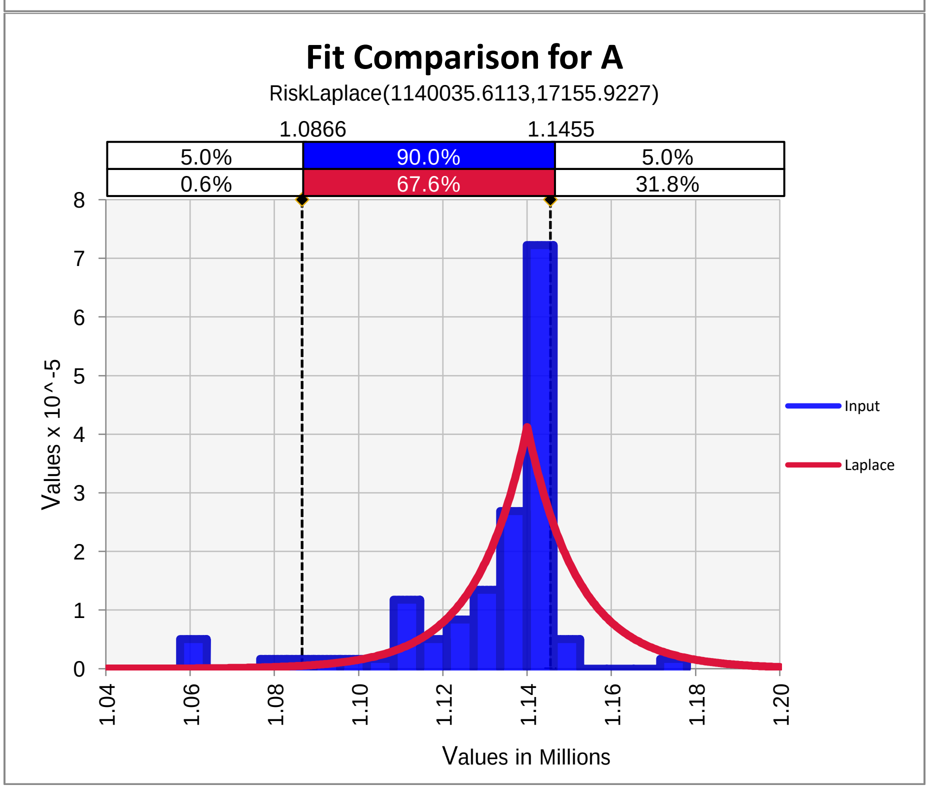


Figure 56. **Base Steele-Henderson model jackknifed distributions of parameters *d* and *A* (Atlantic menhaden ages 1+ biomass at striped bass satiation, MT) and Laplace distributions providing best fit using @Risk's distribution fitting module.**

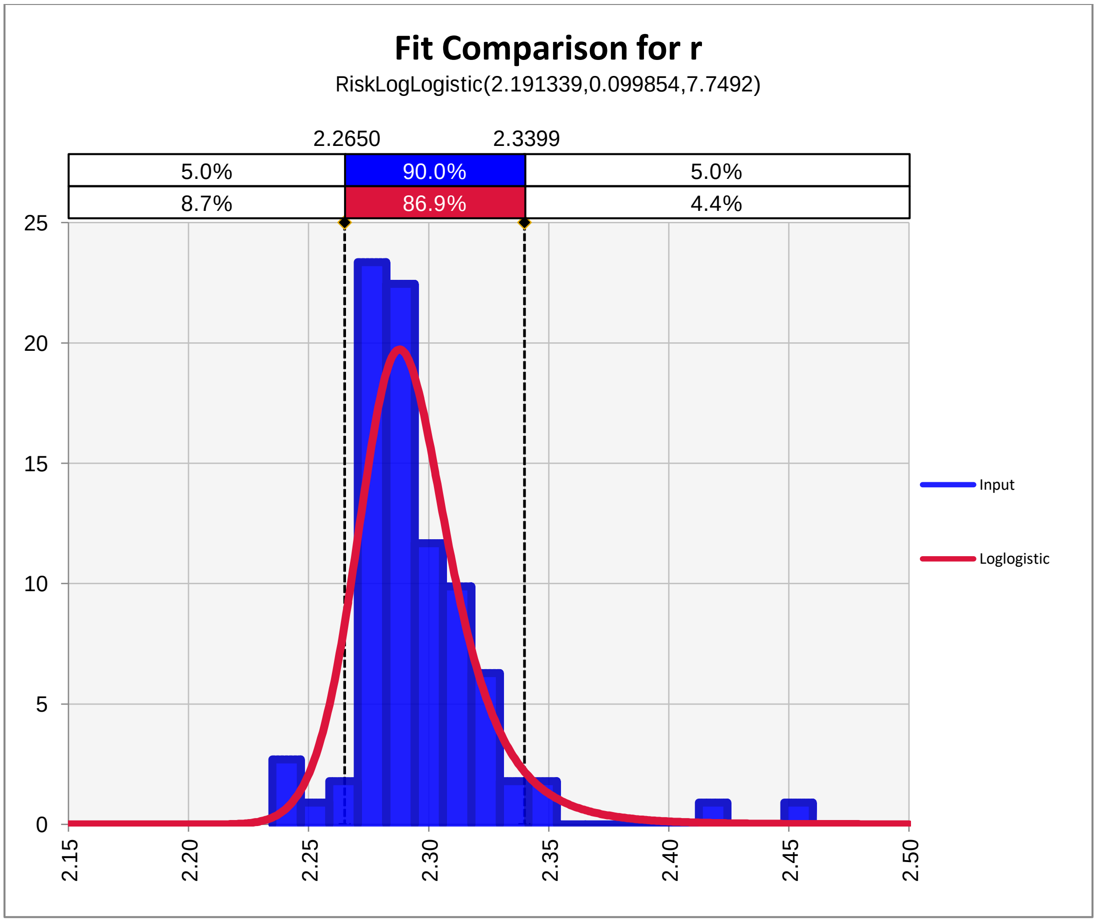


Figure 57. Base Steele-Henderson model jackknifed distribution of intrinsic growth rate, *r*, and the log logistic distribution providing best fit using @Risk's distribution fitting module

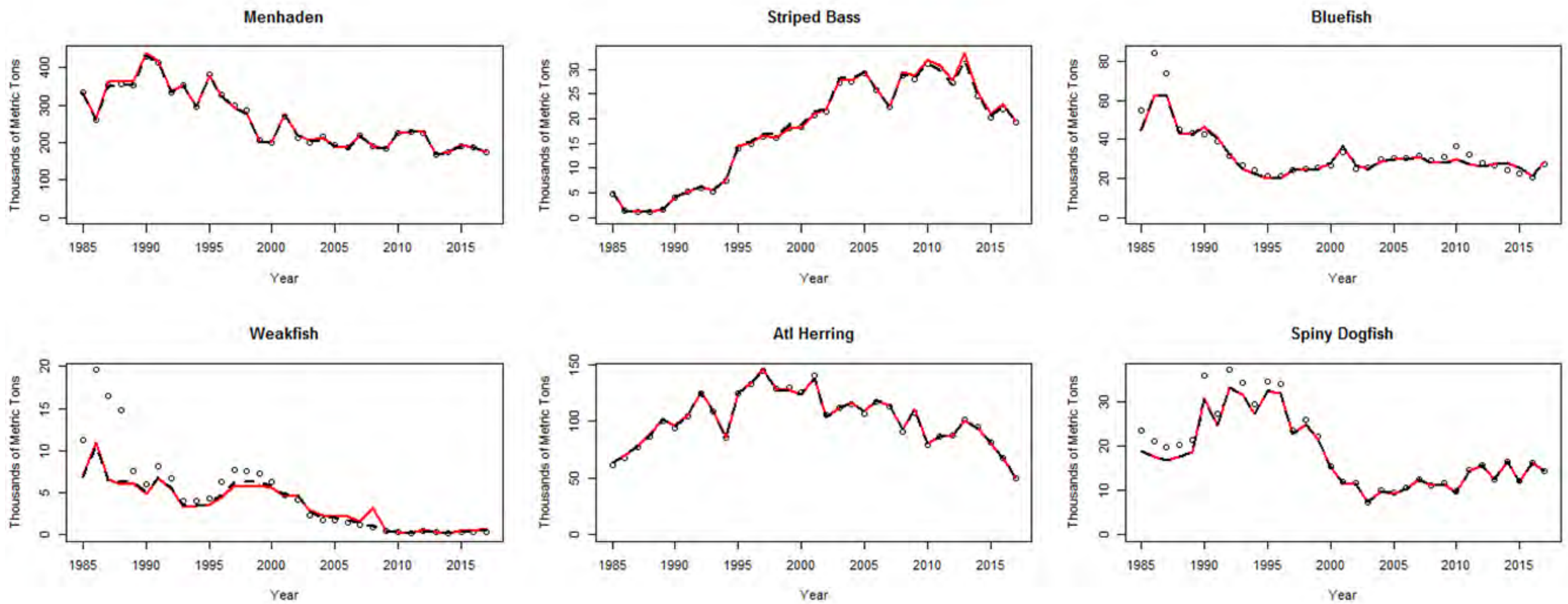


Figure 58. **Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) total annual catch from the VADER model.**

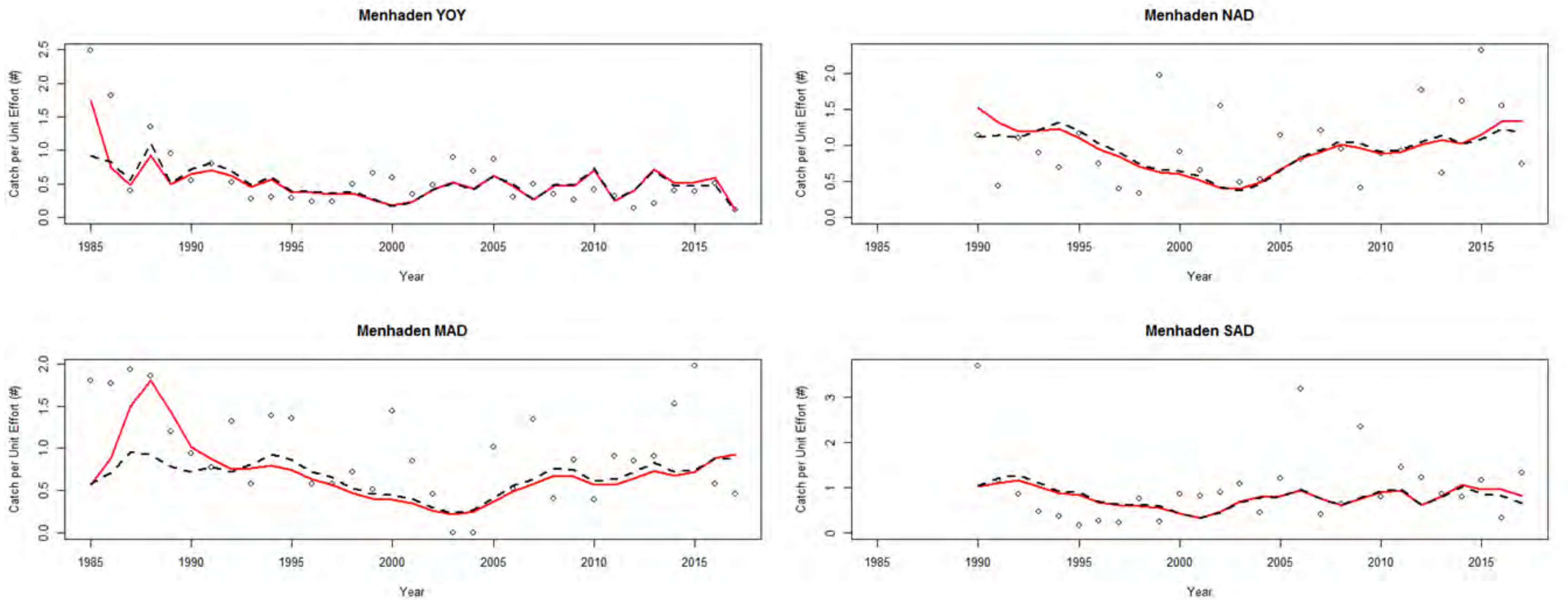


Figure 59. Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for Atlantic menhaden from the VADER model.

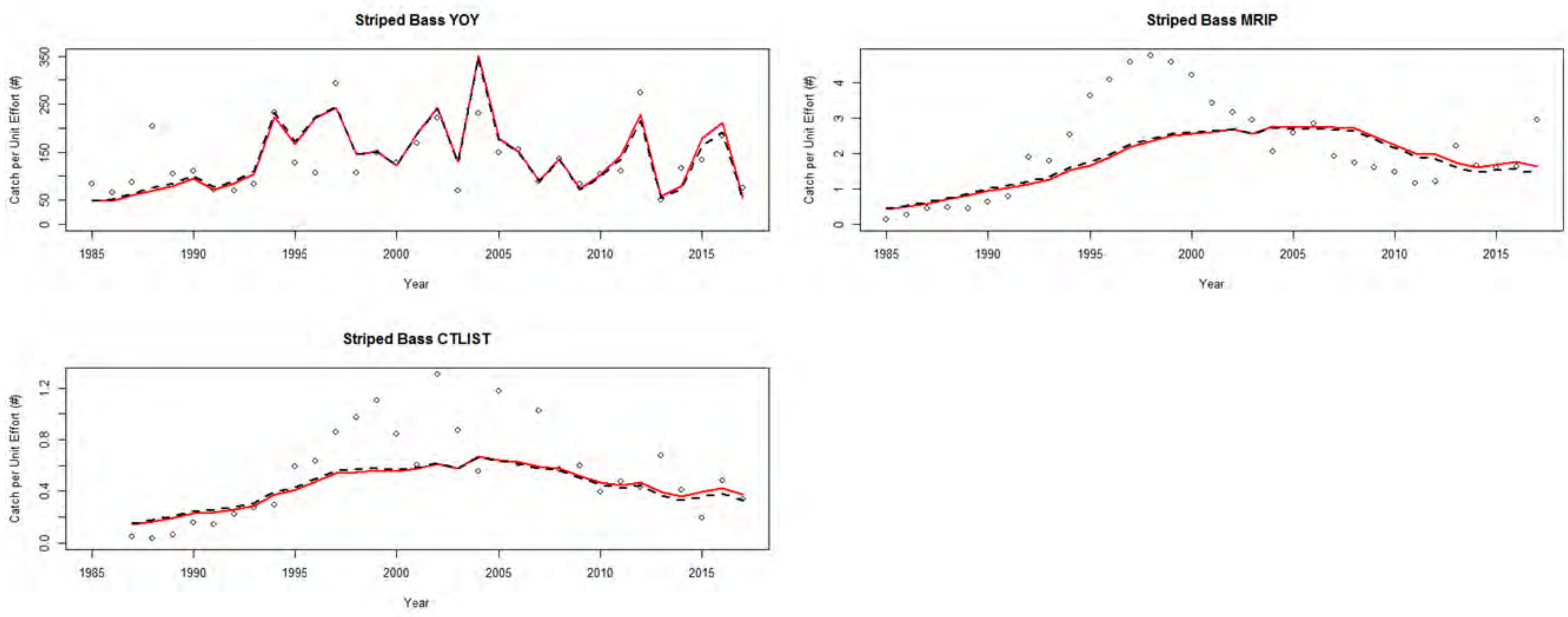


Figure 60. Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for striped bass from the VADER model.

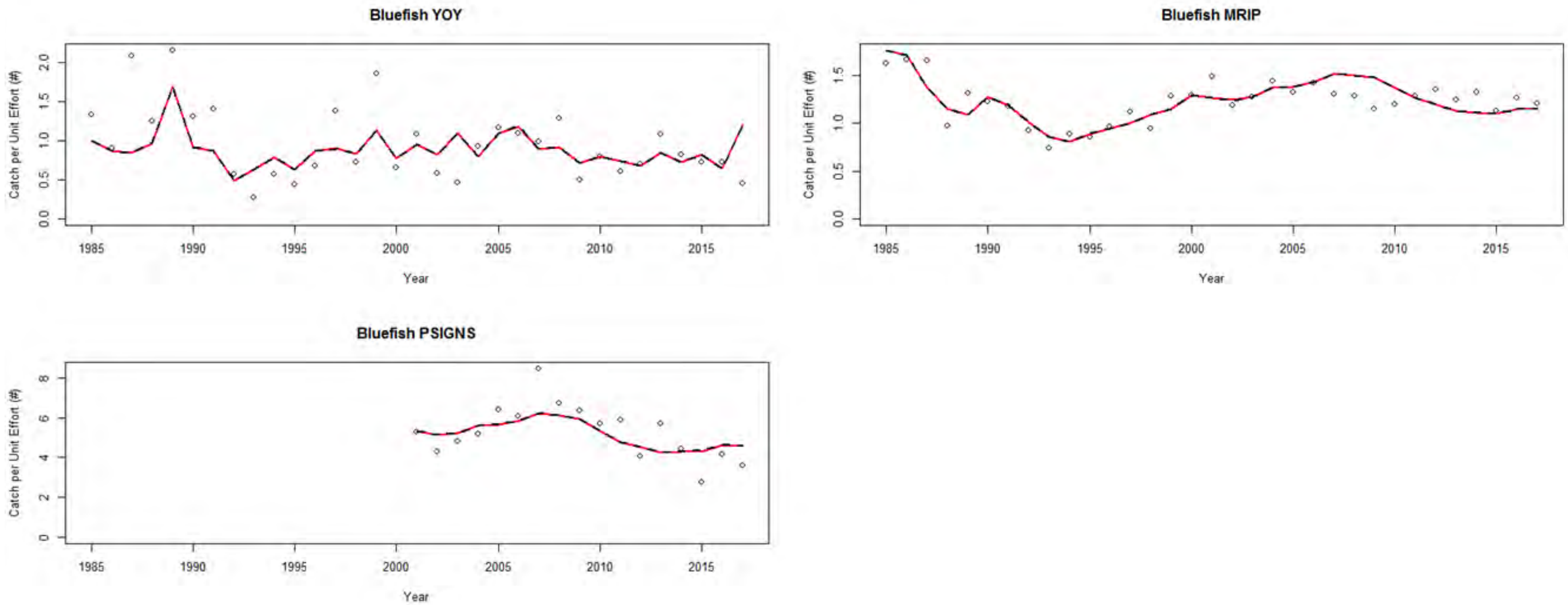


Figure 61. **Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for bluefish from the VADER model.**

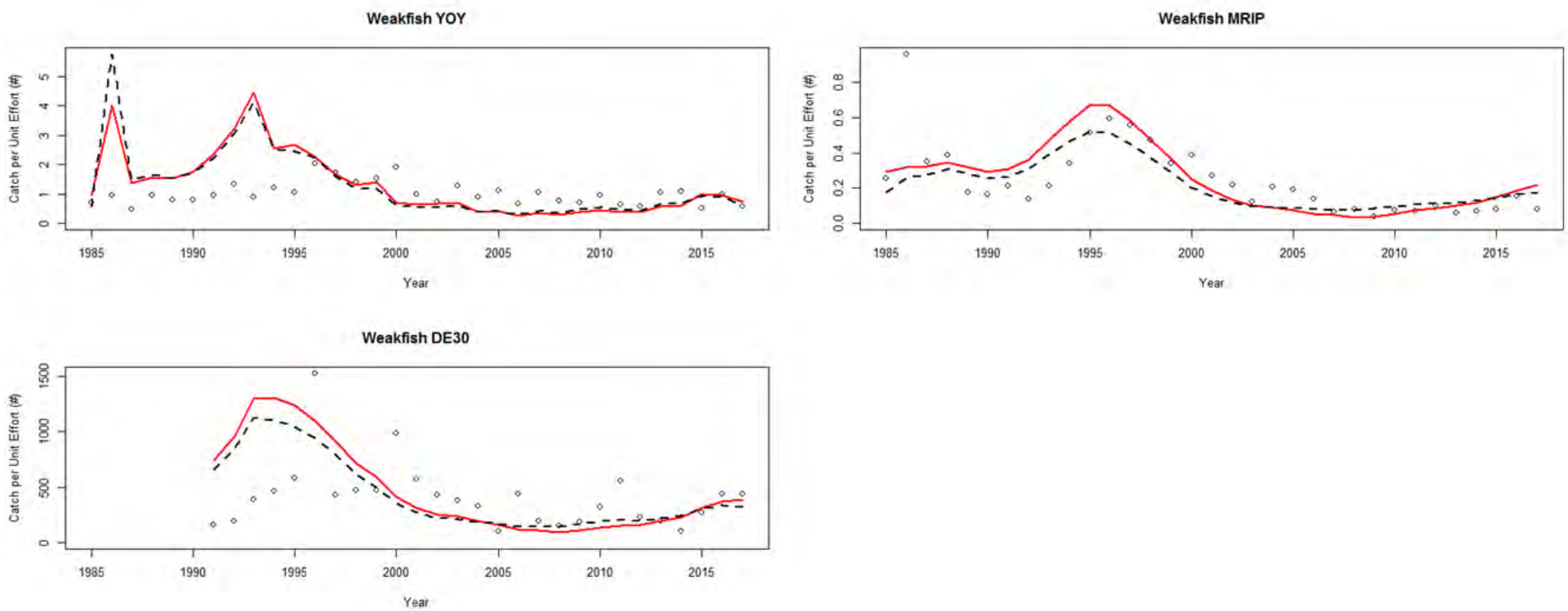


Figure 62. Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for weakfish from the VADER model.

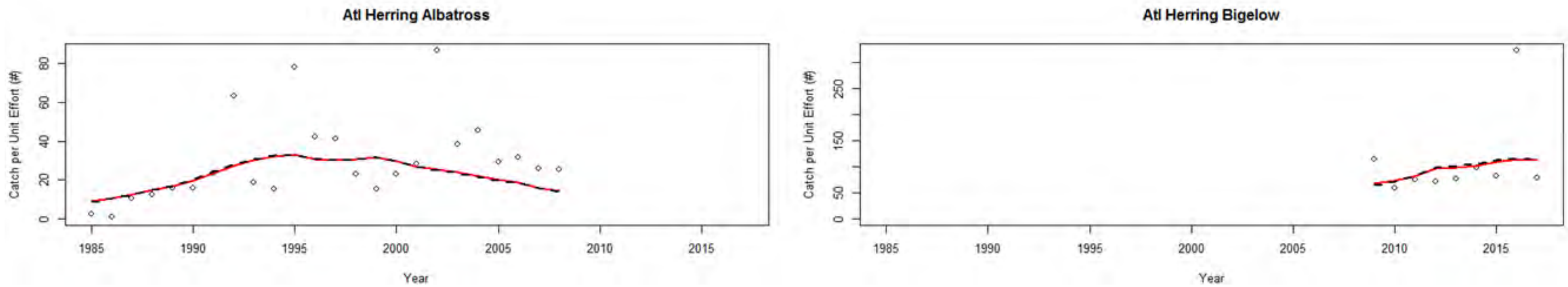


Figure 63. Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for Atlantic herring from the VADER model

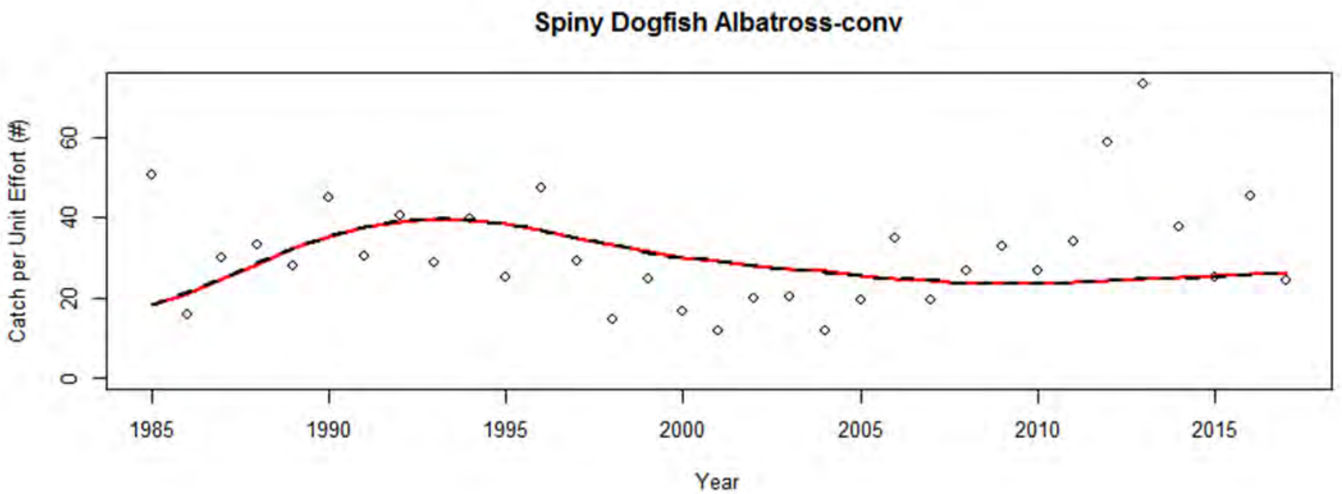


Figure 64. **Observed (open circles), predicted with no trophic interactions (dashed black line), and predicted multispecies (solid red line) indices of abundance for spiny dogfish from the VADER model**

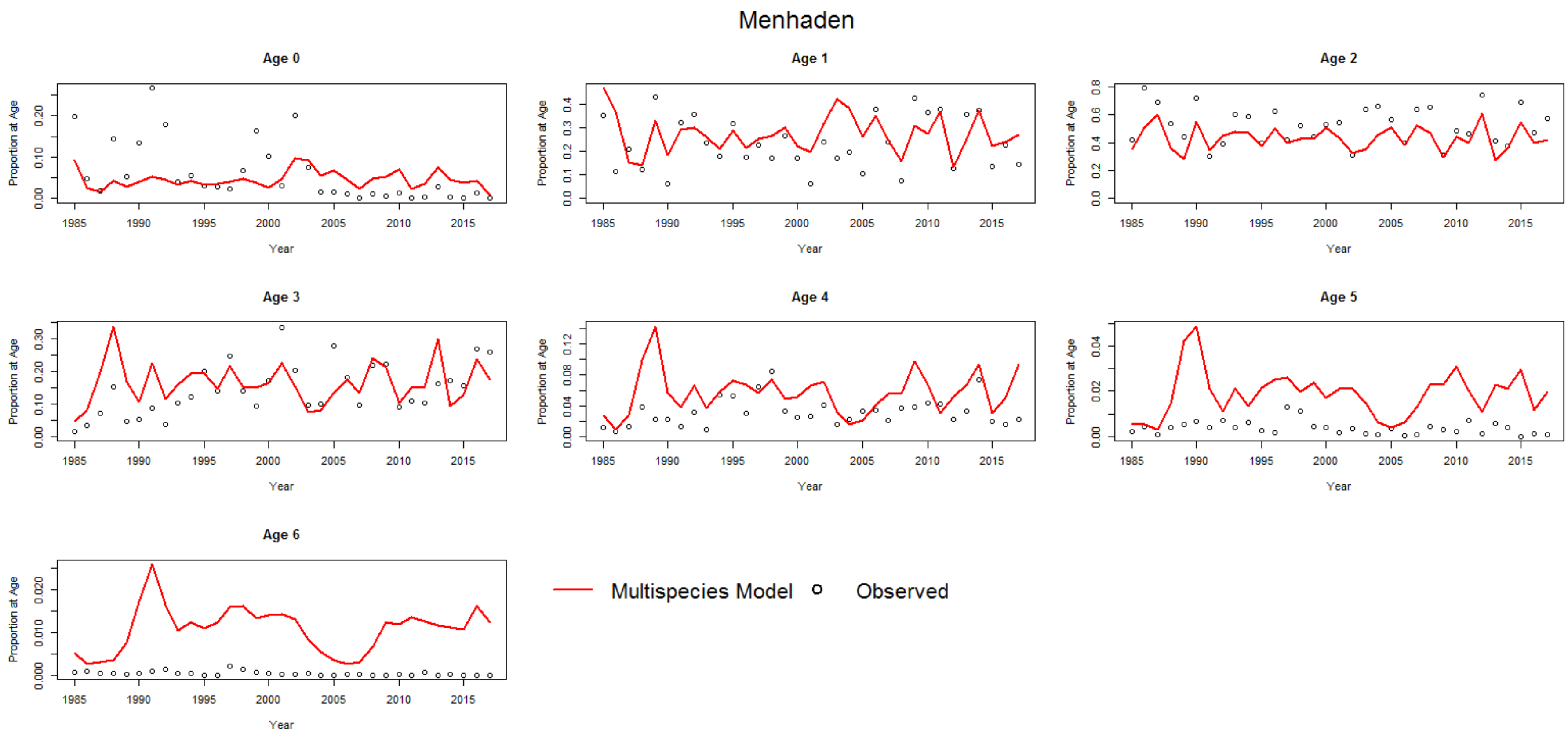


Figure 65. Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for Atlantic menhaden from the VADER model.

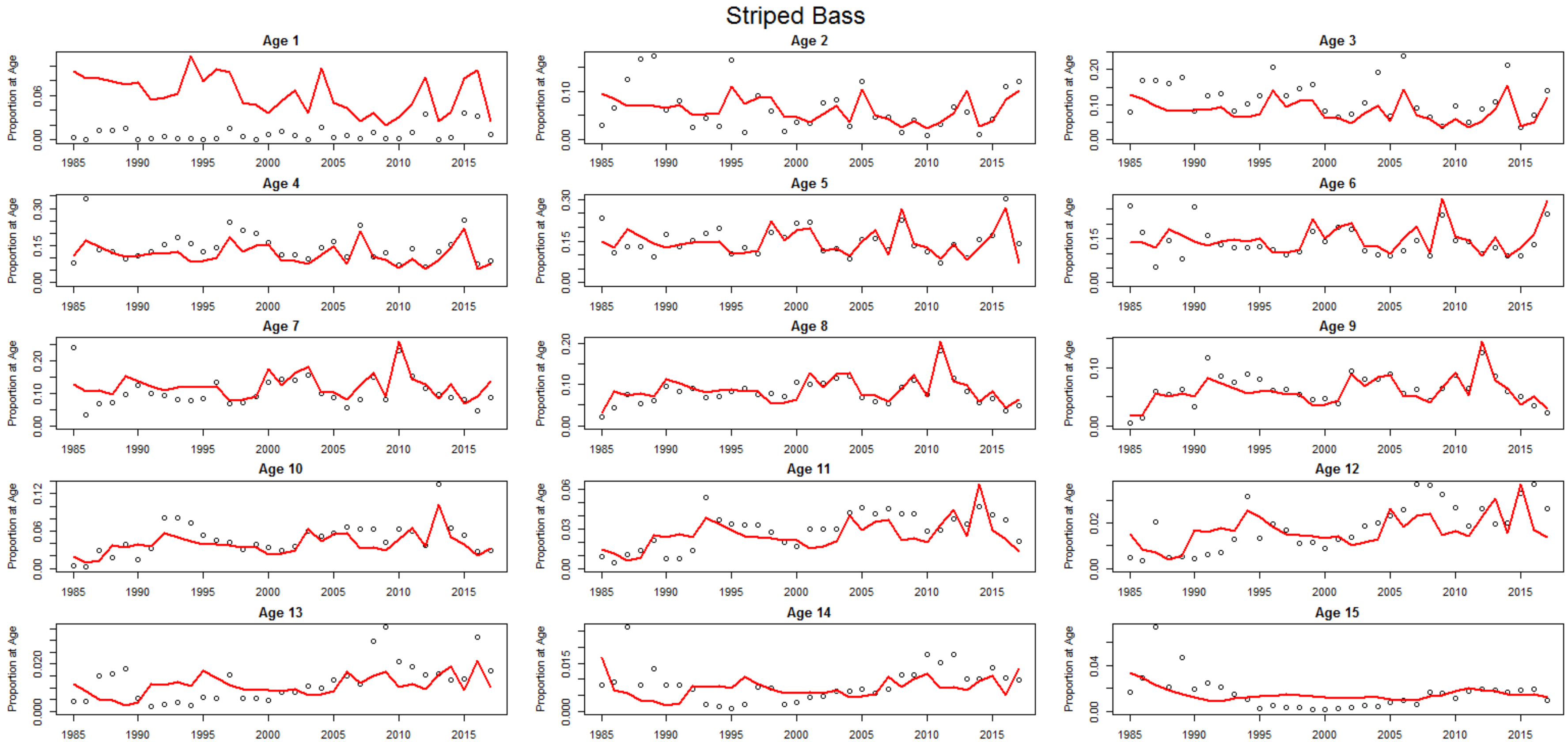


Figure 66. Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for striped bass from the VADER model.

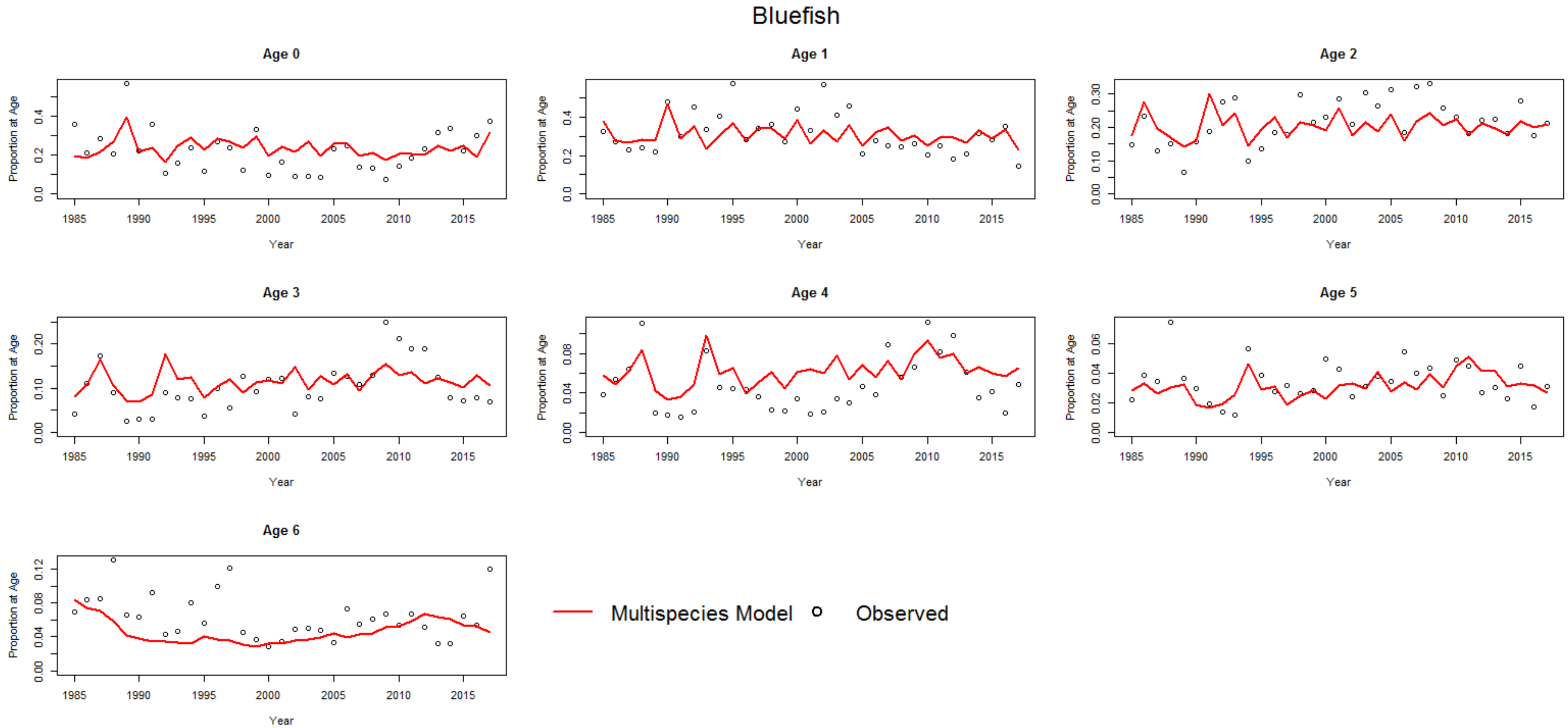


Figure 67. Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for bluefish from the VADER model.

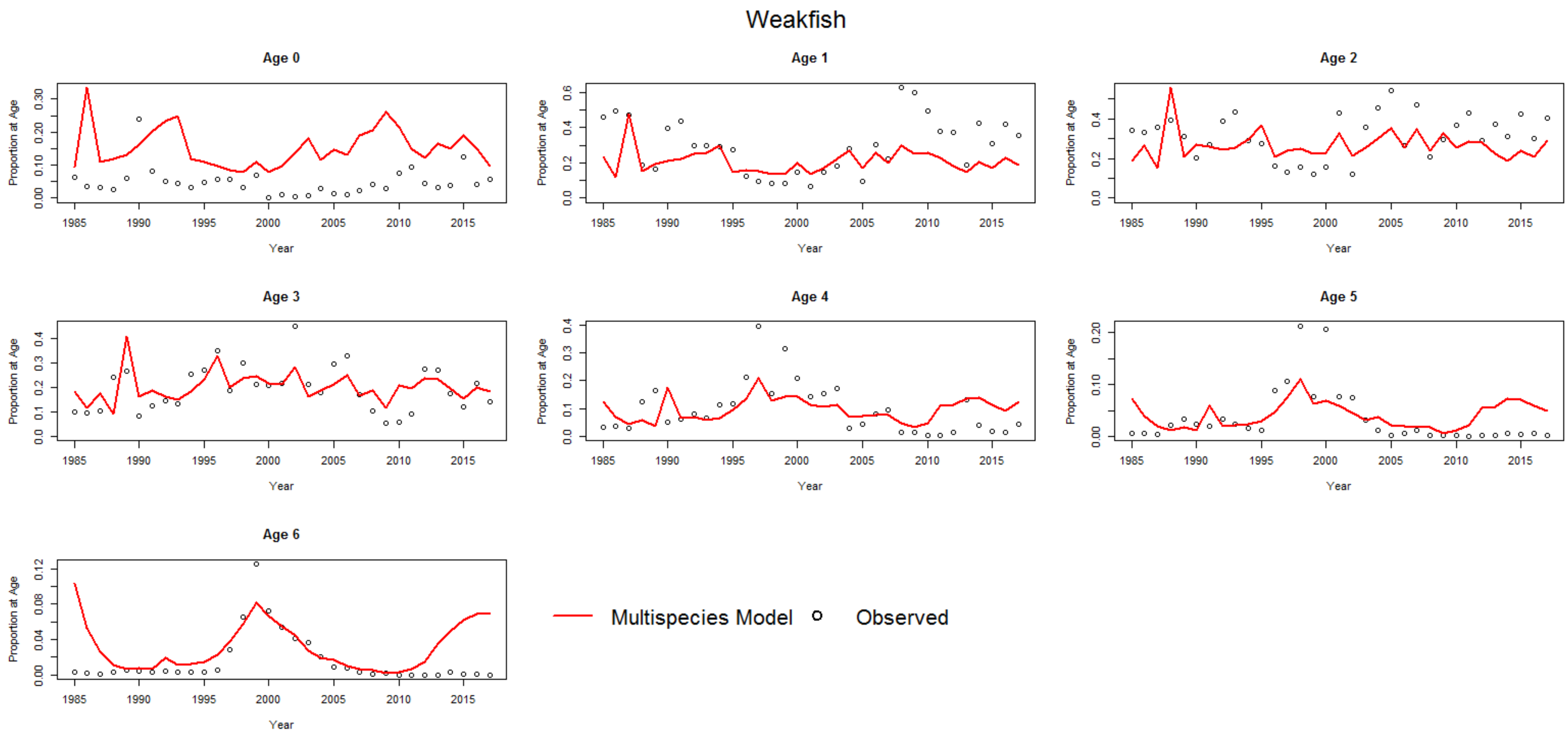


Figure 68. Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for weakfish from the VADER model.

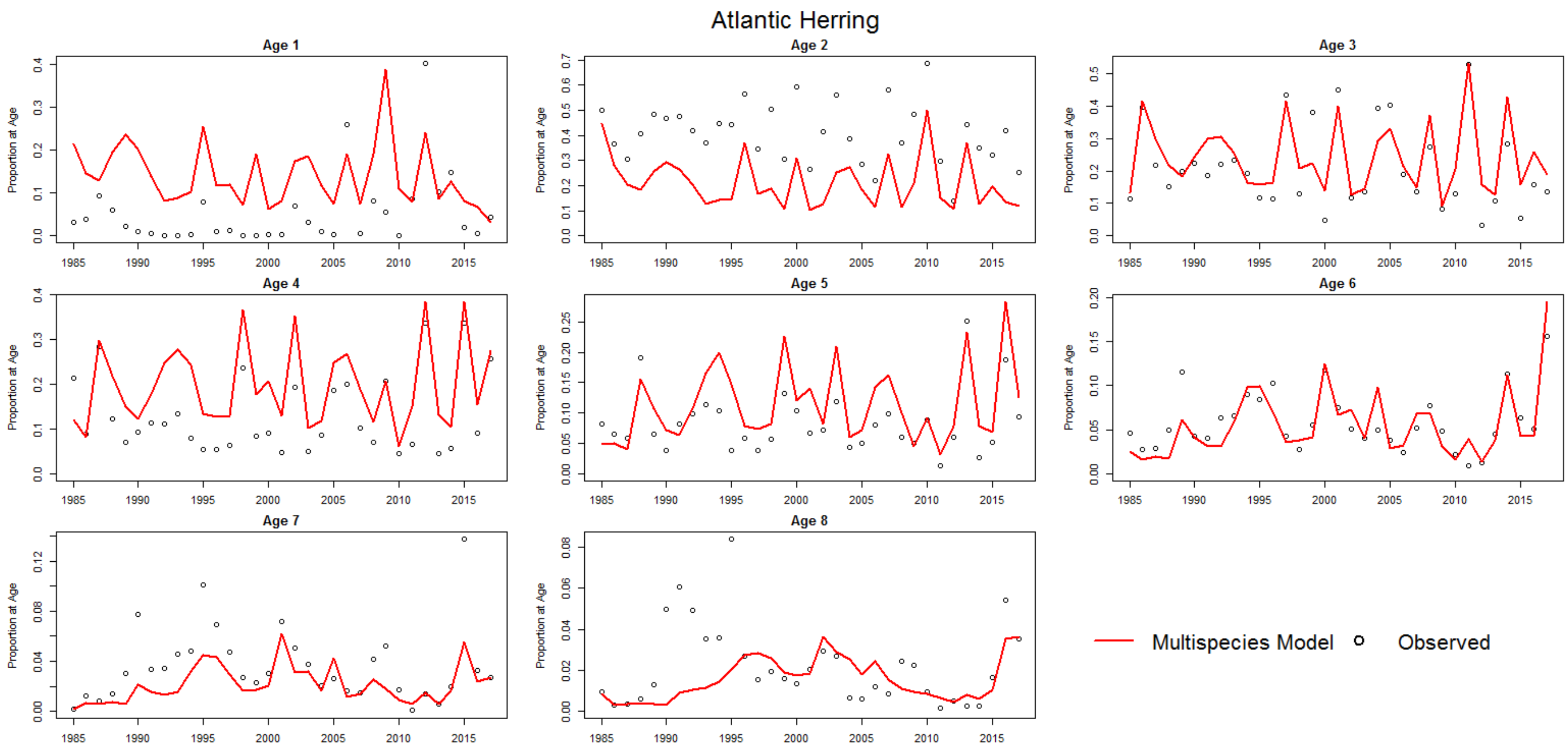


Figure 69. Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for Atlantic herring from the VADER model.

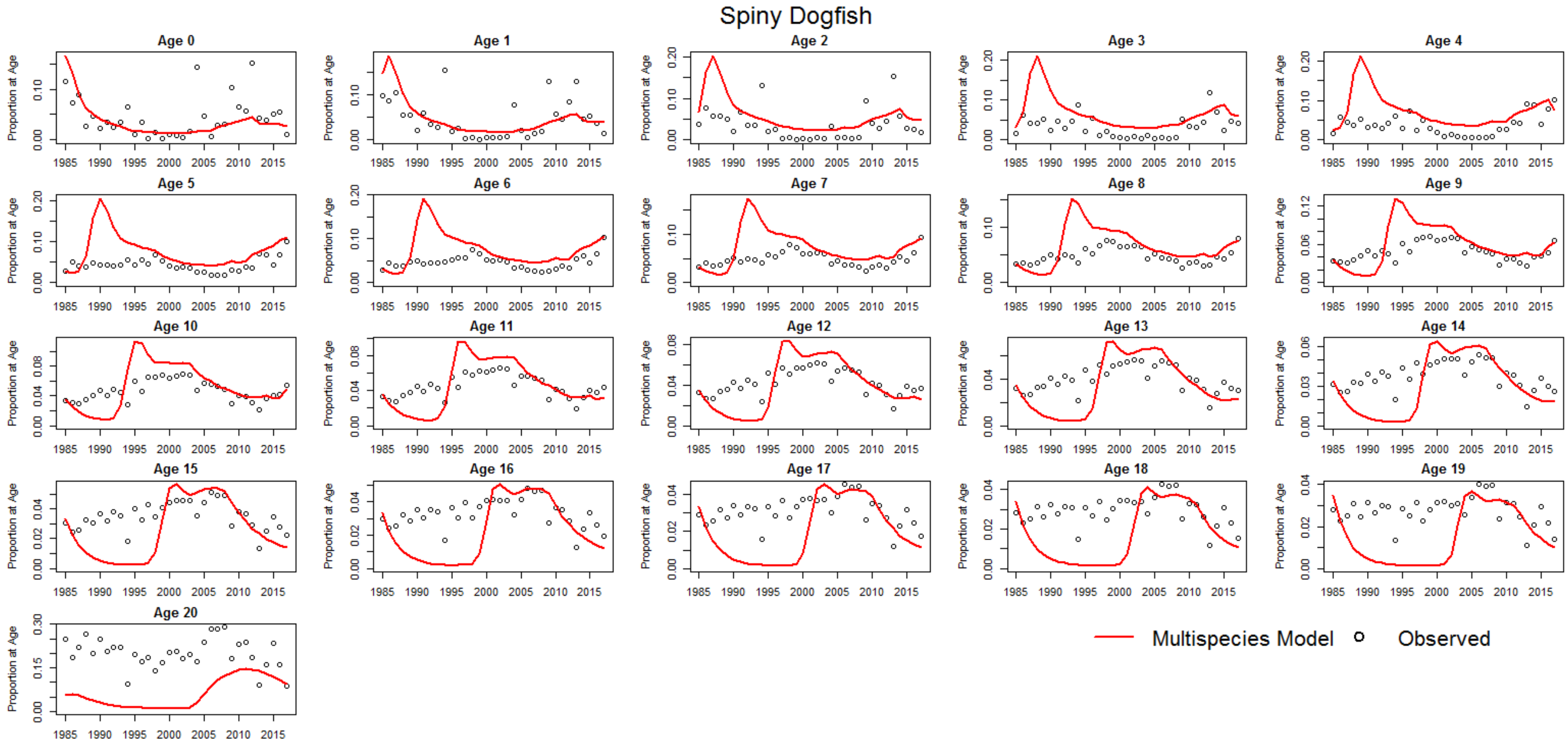


Figure 70. **Observed (open circles) and predicted multispecies (solid red line) total catch age proportions for spiny dogfish from the VADER model.**

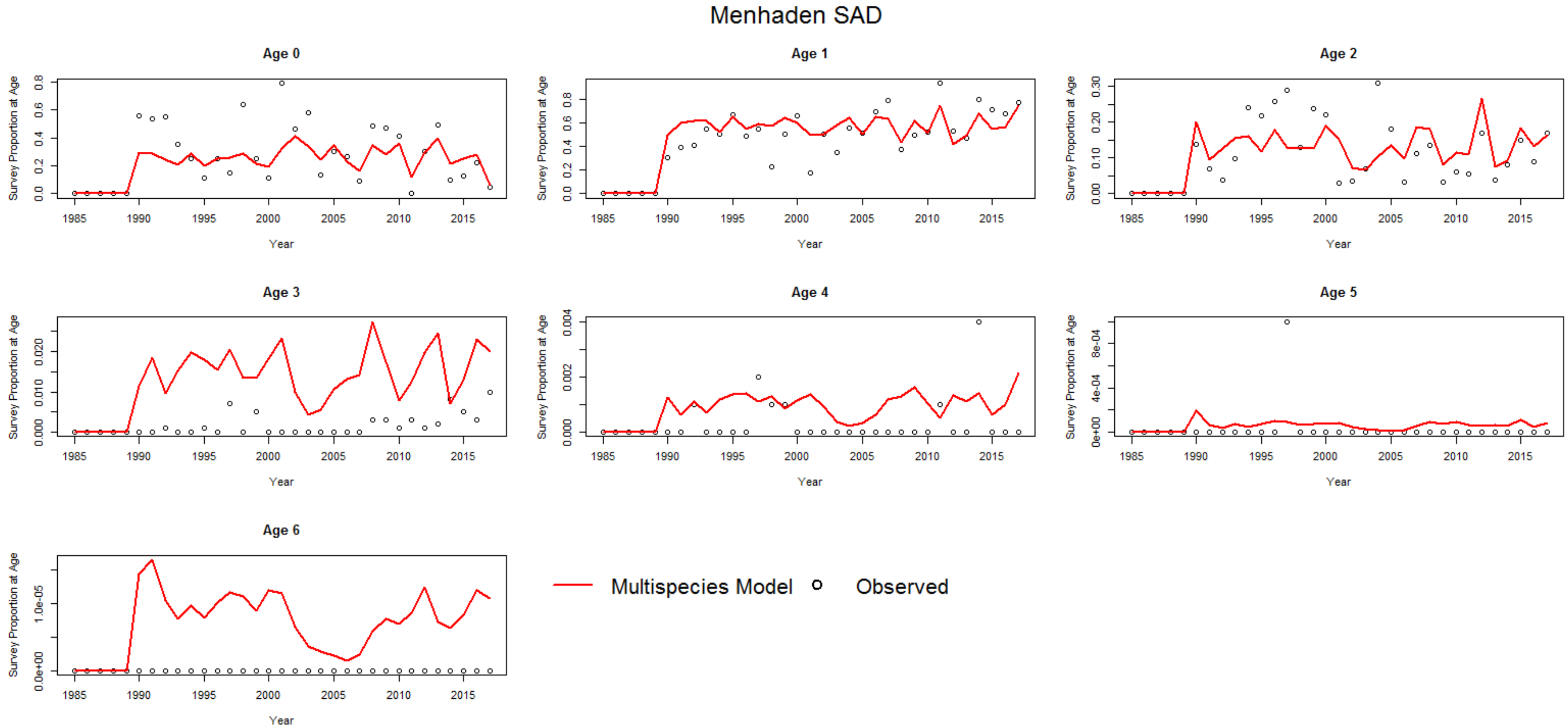


Figure 71. Observed (open circles) and predicted multispecies (solid red line) age proportions for Atlantic menhaden SAD survey from the VADER model.

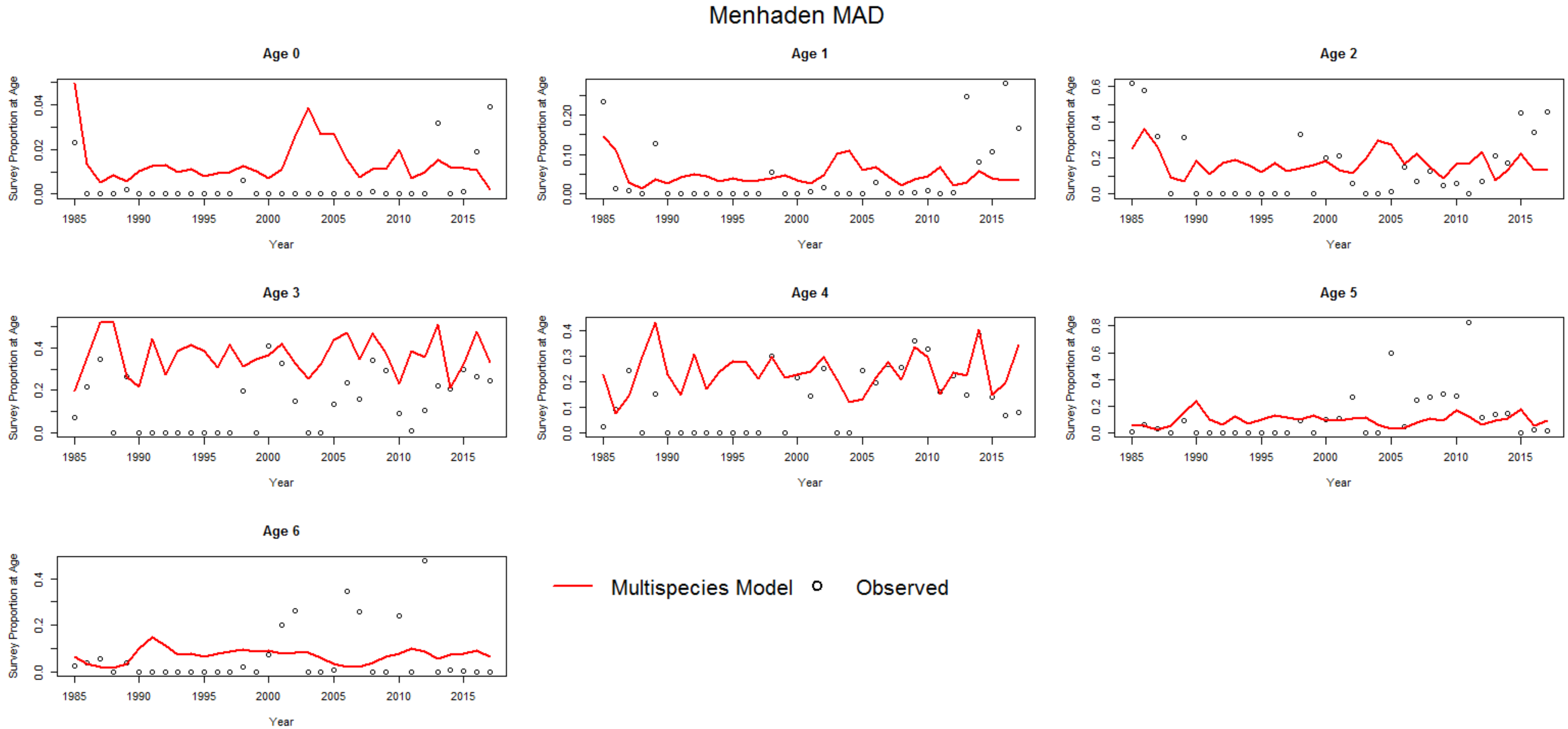


Figure 72. **Observed (open circles) and predicted multispecies (solid red line) age proportions for Atlantic menhaden MAD survey from the VADER model.**

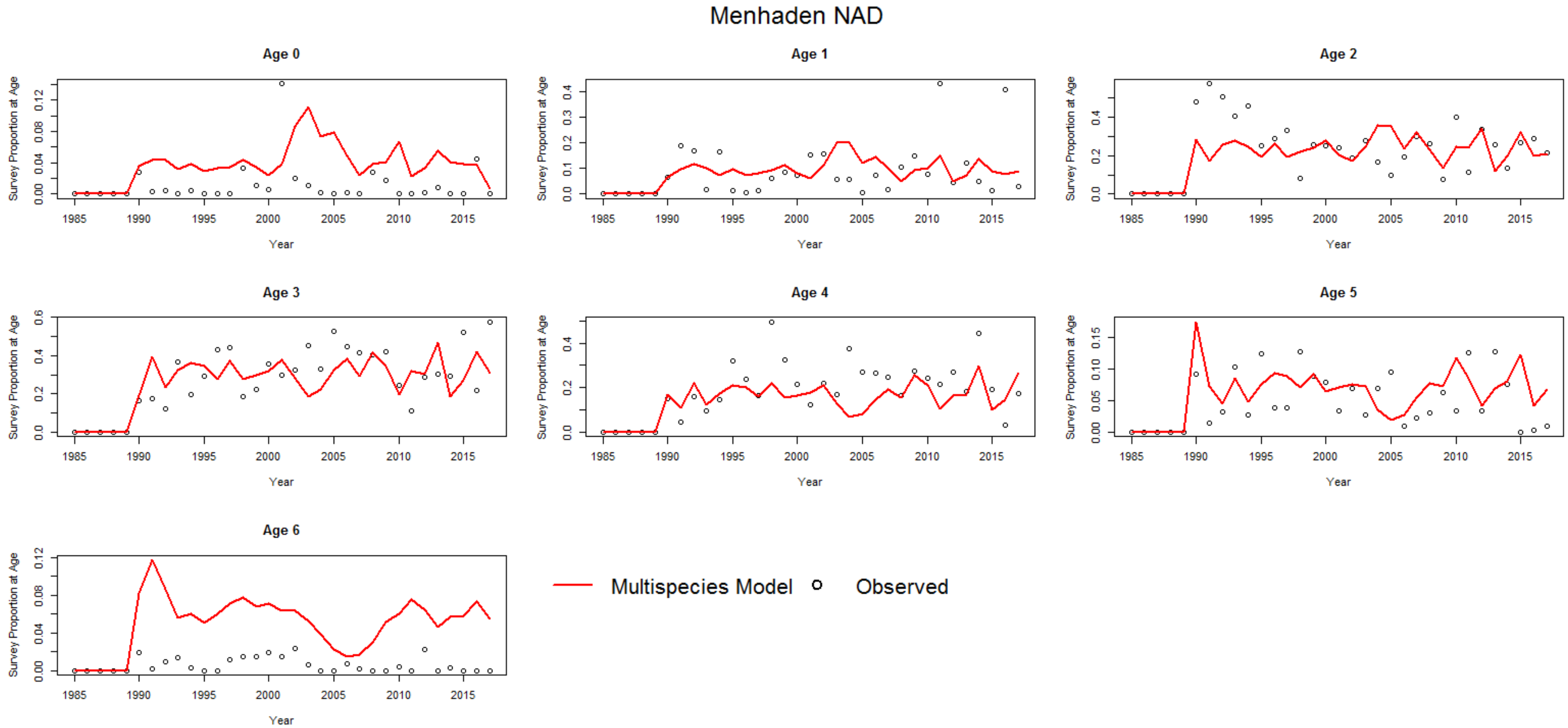


Figure 73. Observed (open circles) and predicted multispecies (solid red line) age proportions for Atlantic menhaden NAD survey from the VADER model.

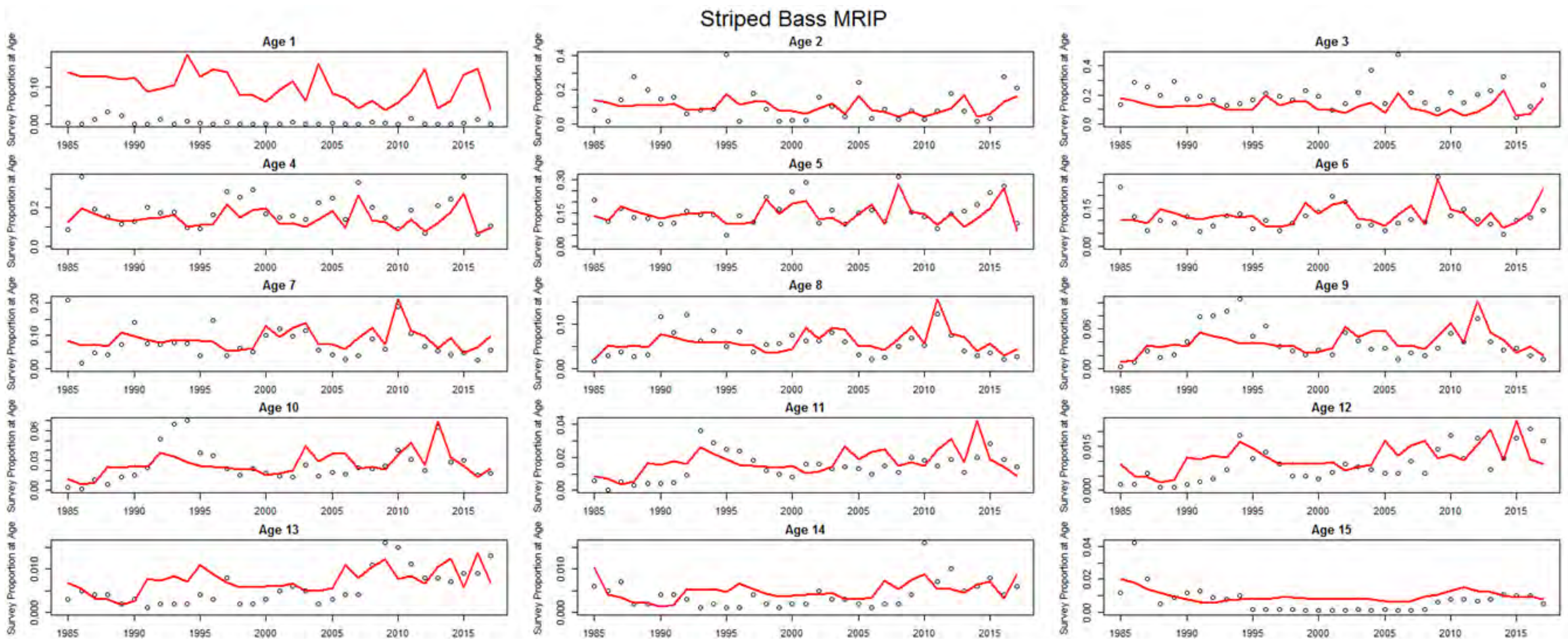


Figure 74. Observed (open circles) and predicted multispecies (solid red line) age proportions for striped bass MRIP CPUE survey from the VADER model.

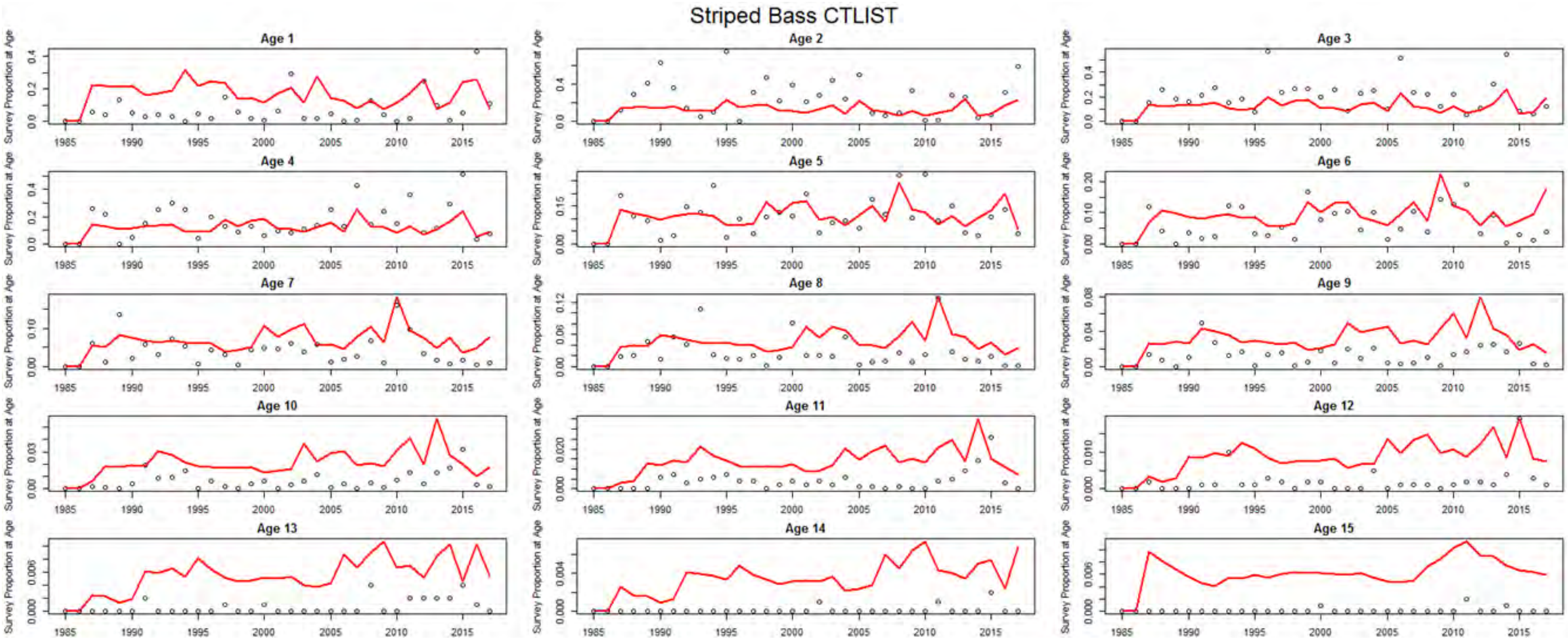


Figure 75. **Observed (open circles) and predicted multispecies (solid red line) age proportions for striped bass CT LIST survey from the VADER model.**

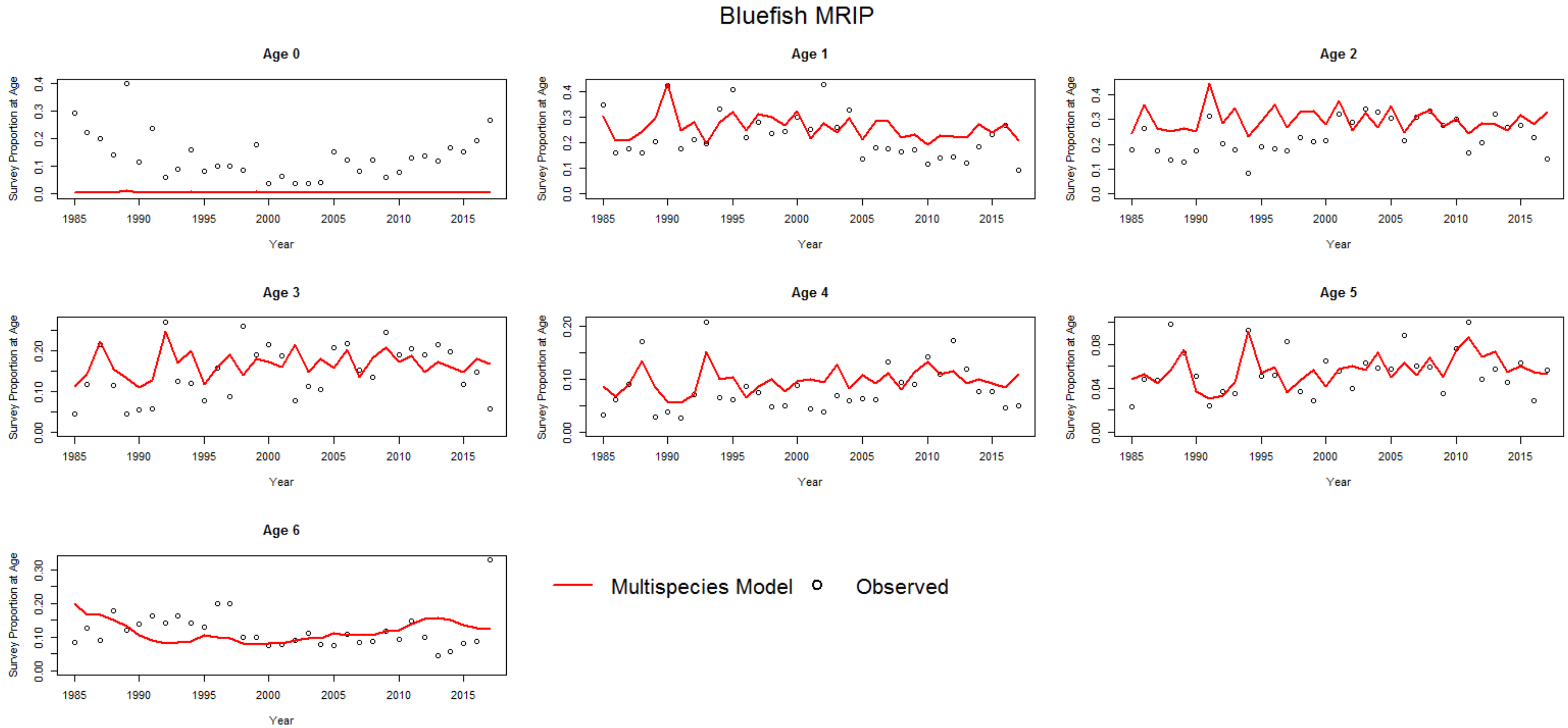


Figure 76. **Observed (open circles) and predicted multispecies (solid red line) age proportions for bluefish MRIP CPUE survey from the VADER model.**

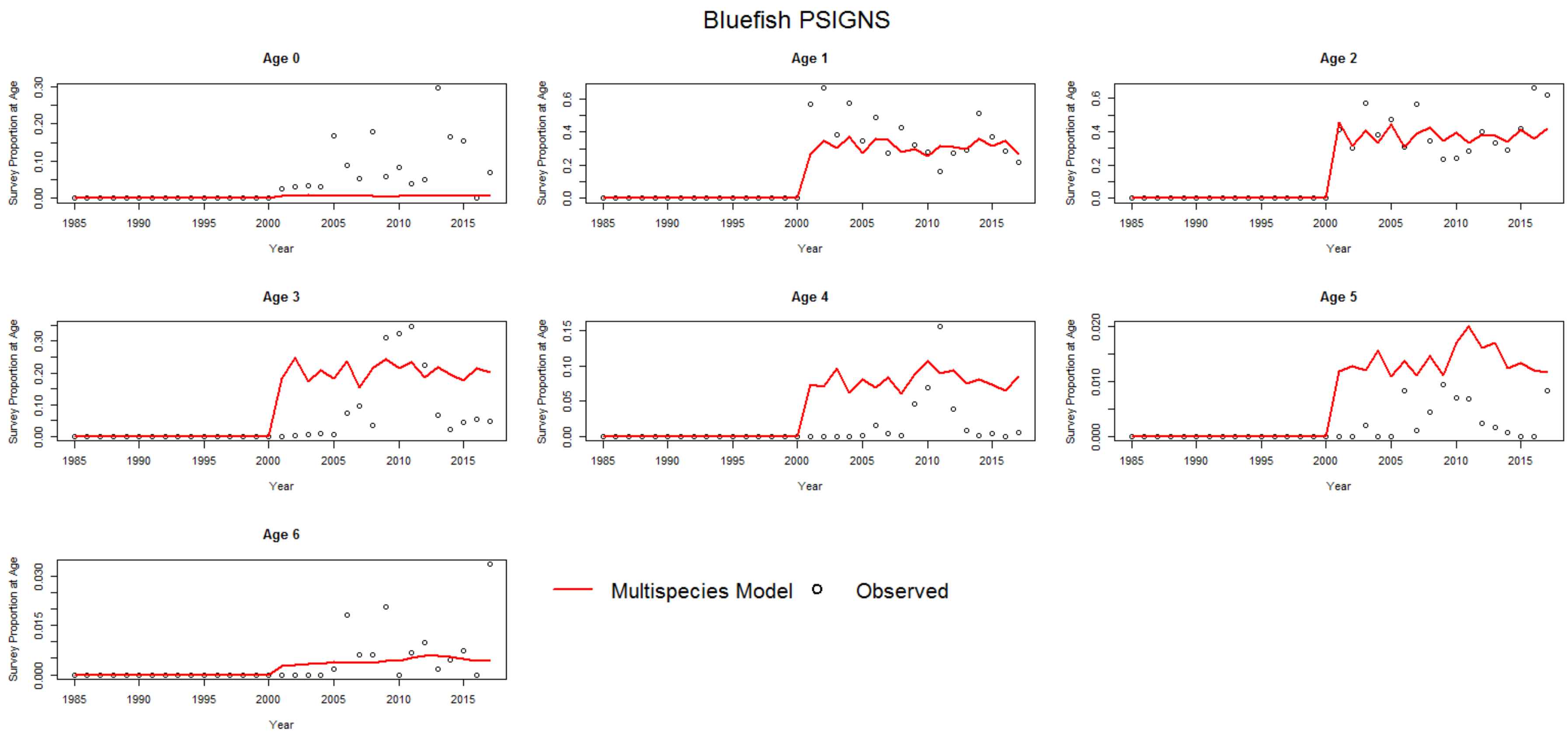


Figure 77. **Observed (open circles) and predicted multispecies (solid red line) age proportions for bluefish NC PSIGNS survey from the VADER model.**

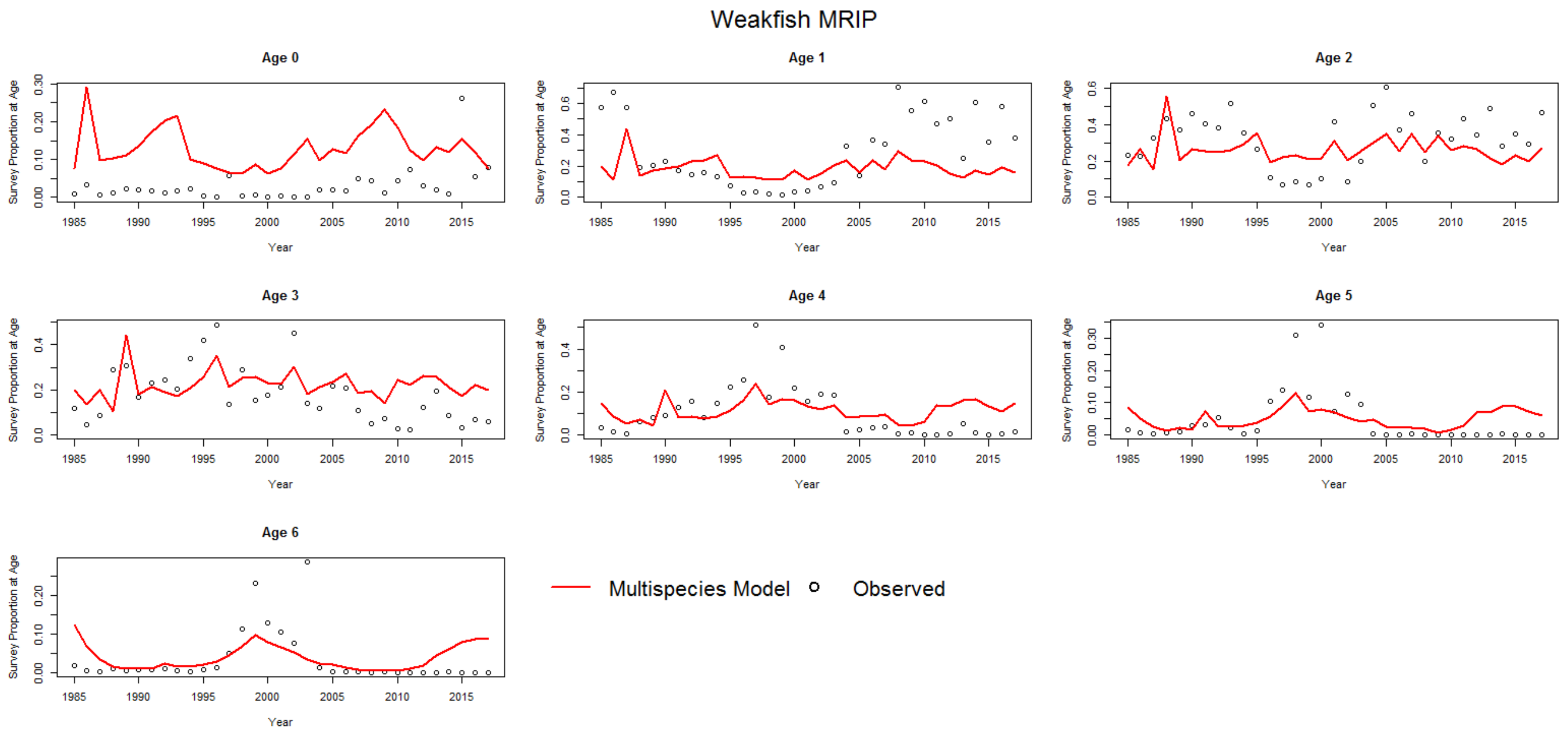


Figure 78. **Observed (open circles) and predicted multispecies (solid red line) age proportions for weakfish MRIP CPUE survey from the VADER model.**

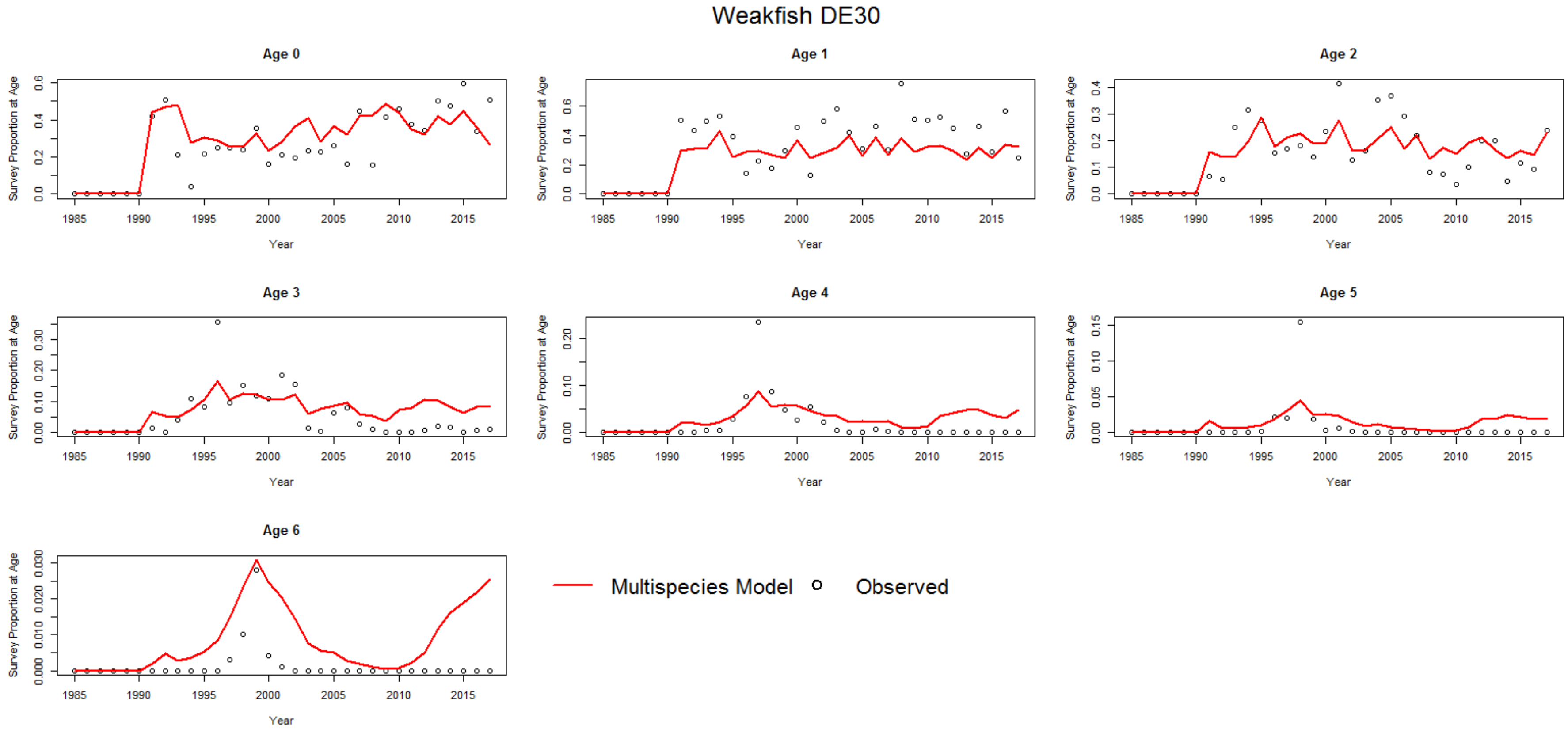


Figure 79. Observed (open circles) and predicted multispecies (solid red line) age proportions for weakfish DE 30' Trawl survey from the VADER model.

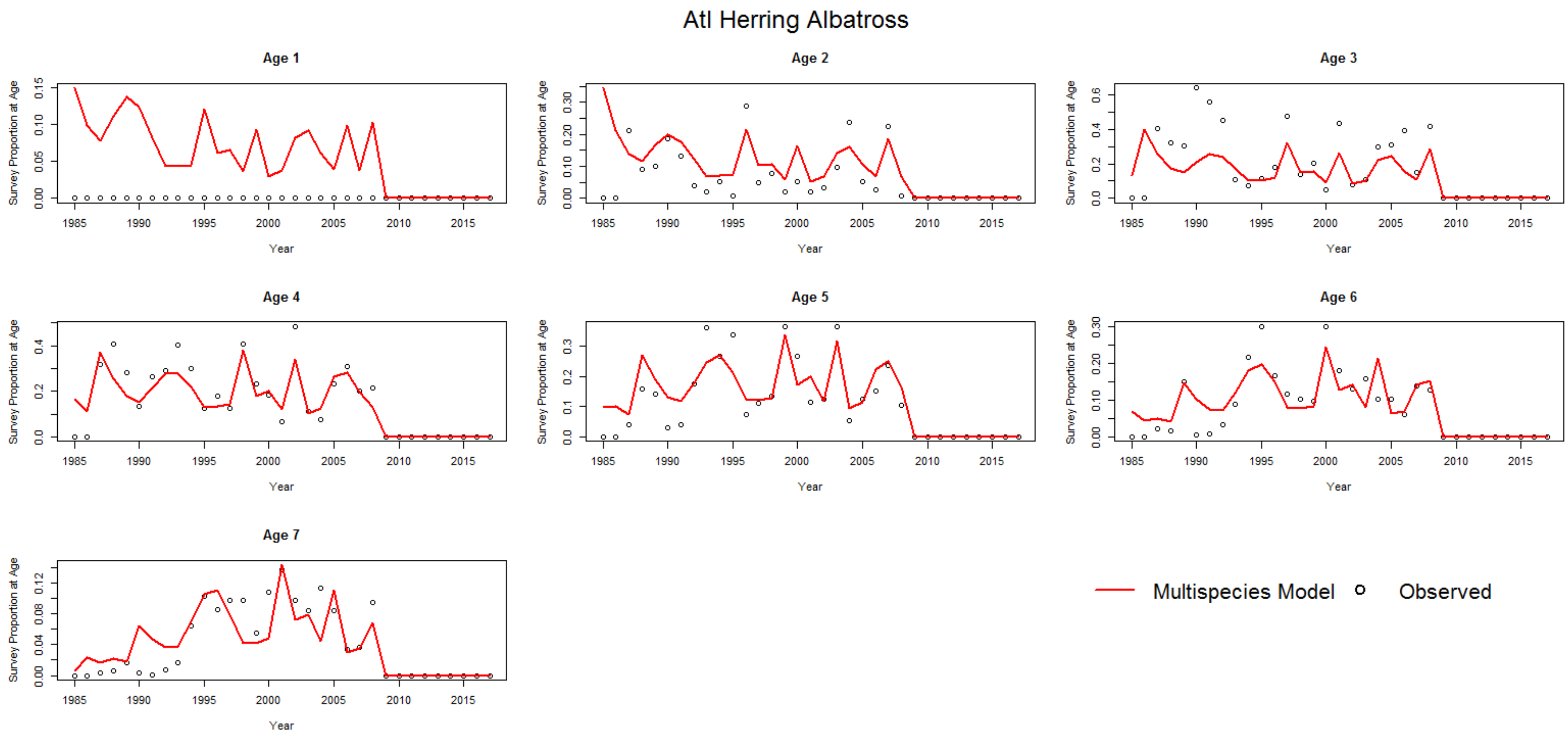


Figure 80. Observed (open circles) and predicted multispecies (solid red line) age proportions for Atlantic herring NEFSC Fall Albatross survey from the VADER model.

Figure 81. **Observed (open circles) and predicted multispecies (solid red line) age proportions for Atlantic herring NEFSC Fall Bigelow survey from the VADER model.**

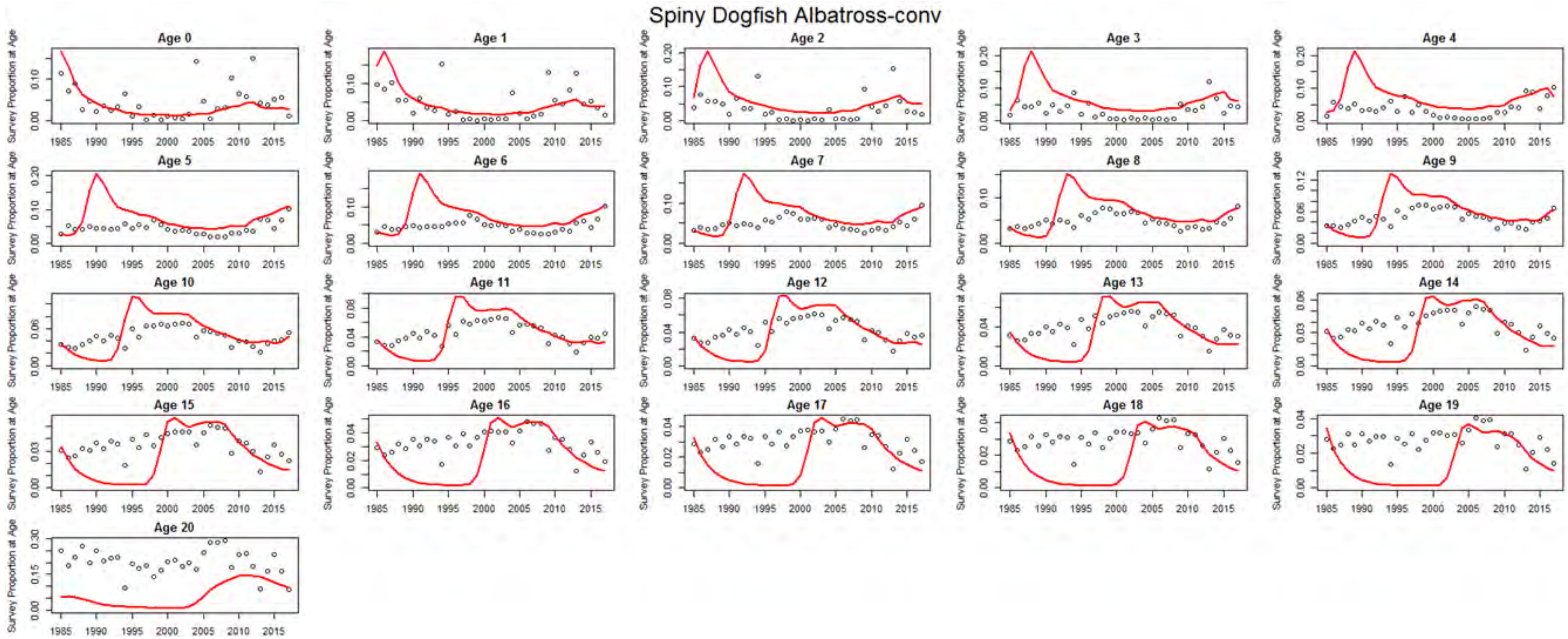


Figure 82. Observed (open circles) and predicted multispecies (solid red line) age proportions for spiny dogfish Albatross survey from the VADER model.

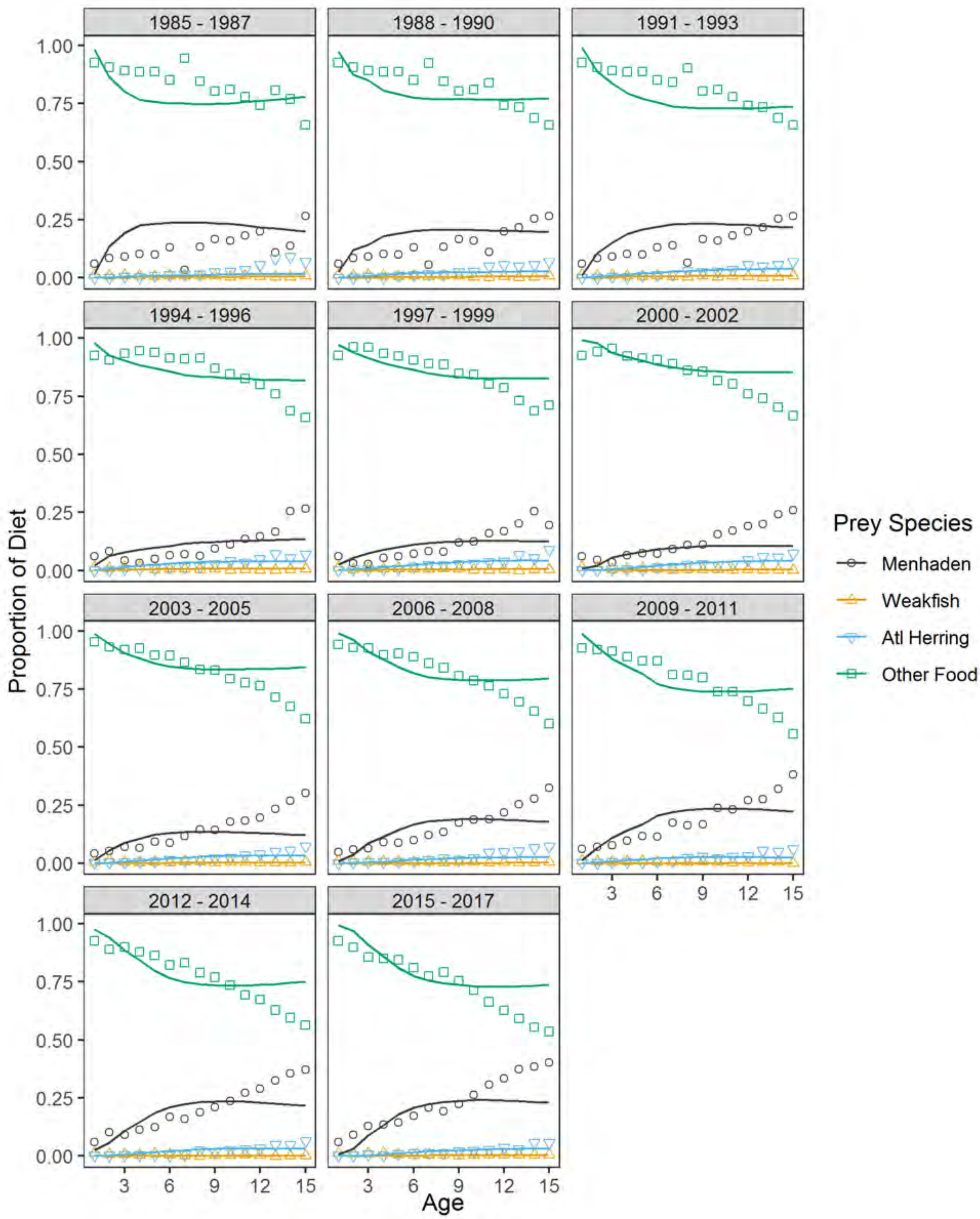


Figure 83. Observed (open points) and predicted (solid lines) diet composition data for striped bass from the VADER model.

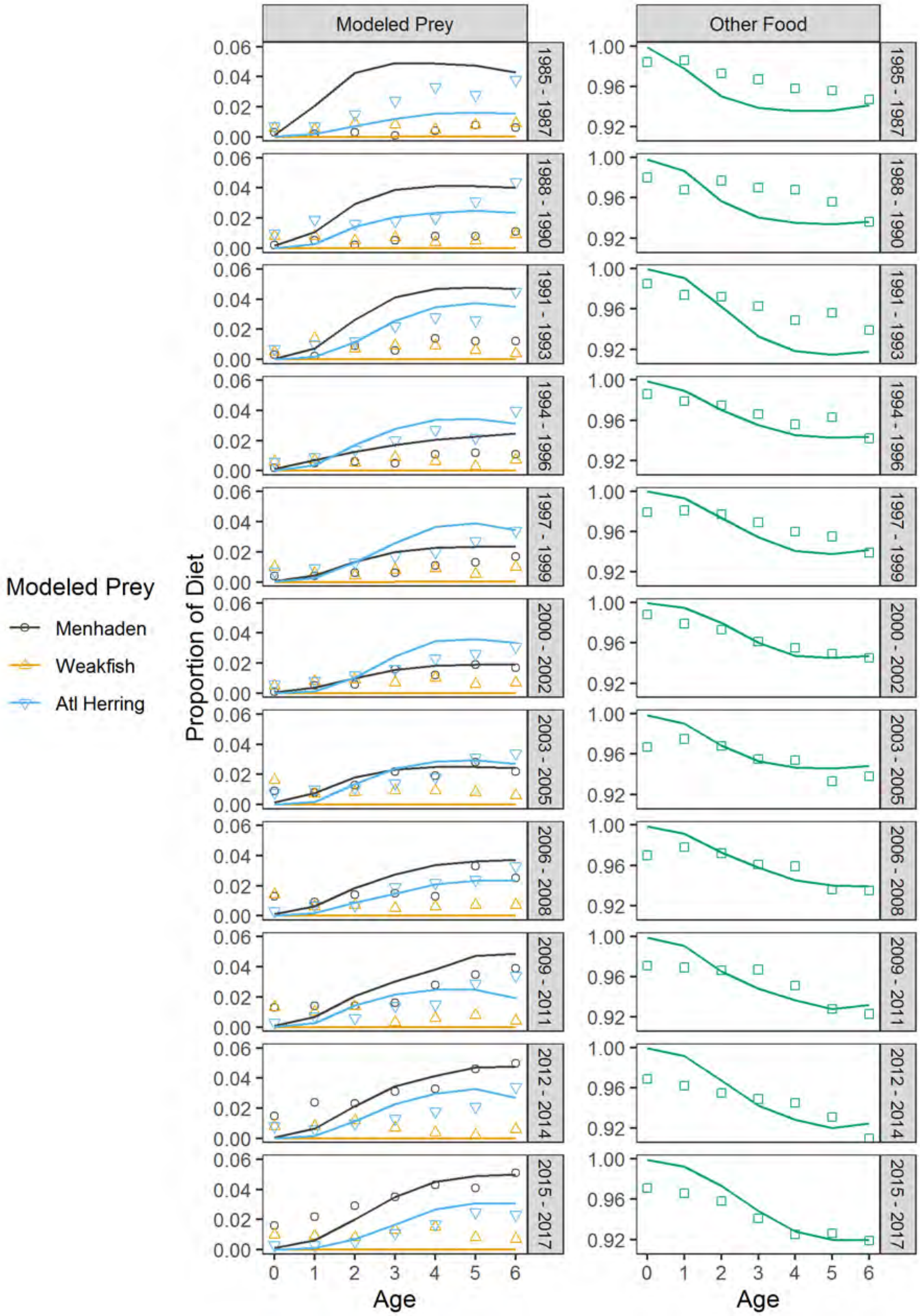


Figure 84. Observed (open points) and predicted (solid lines) diet composition data for bluefish from the VADER model.

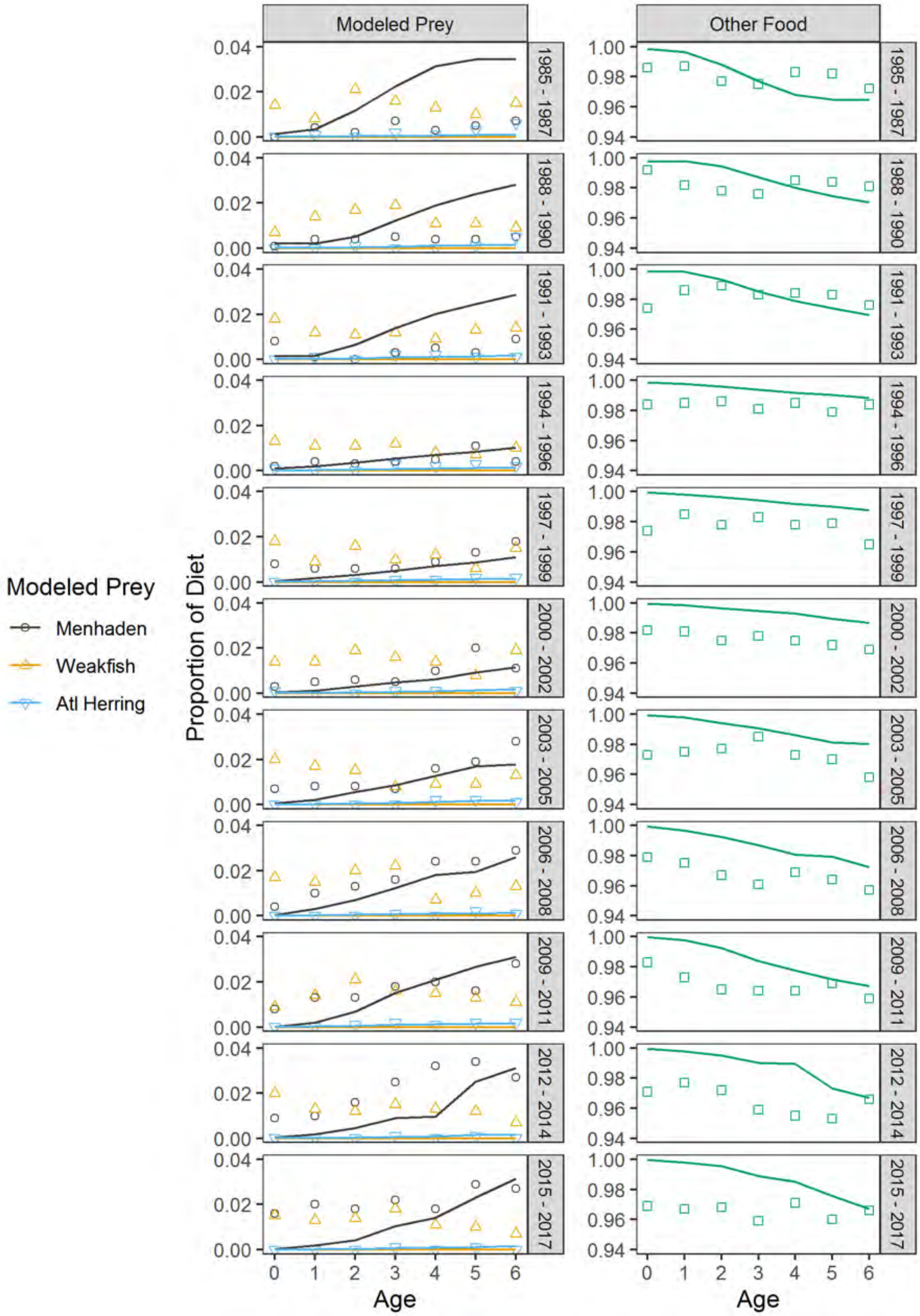


Figure 85. Observed (open points) and predicted (solid lines) diet composition data for weakfish from the VADER model.

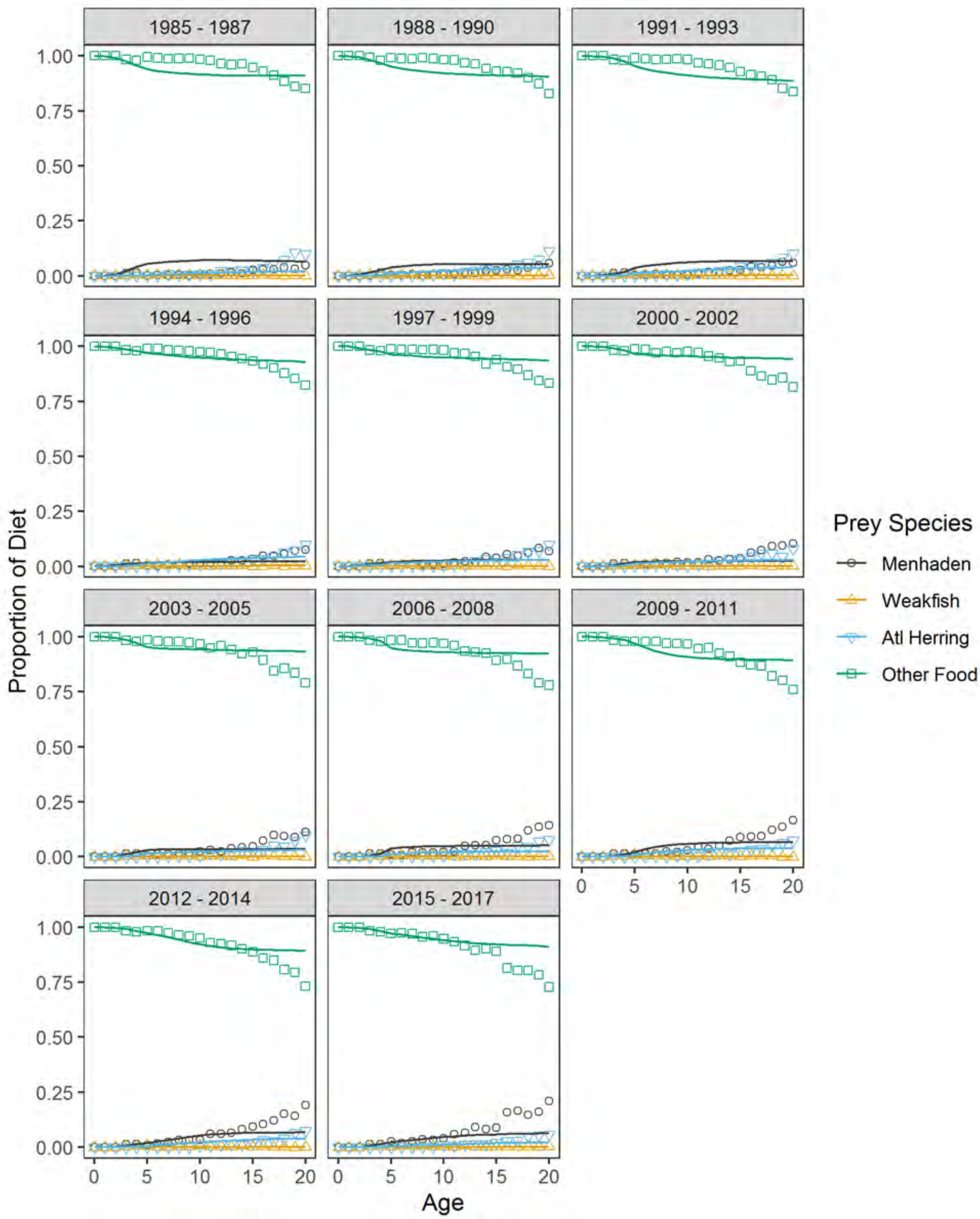


Figure 86. Observed (open points) and predicted (solid lines) diet composition data for spiny dogfish from the VADER model.

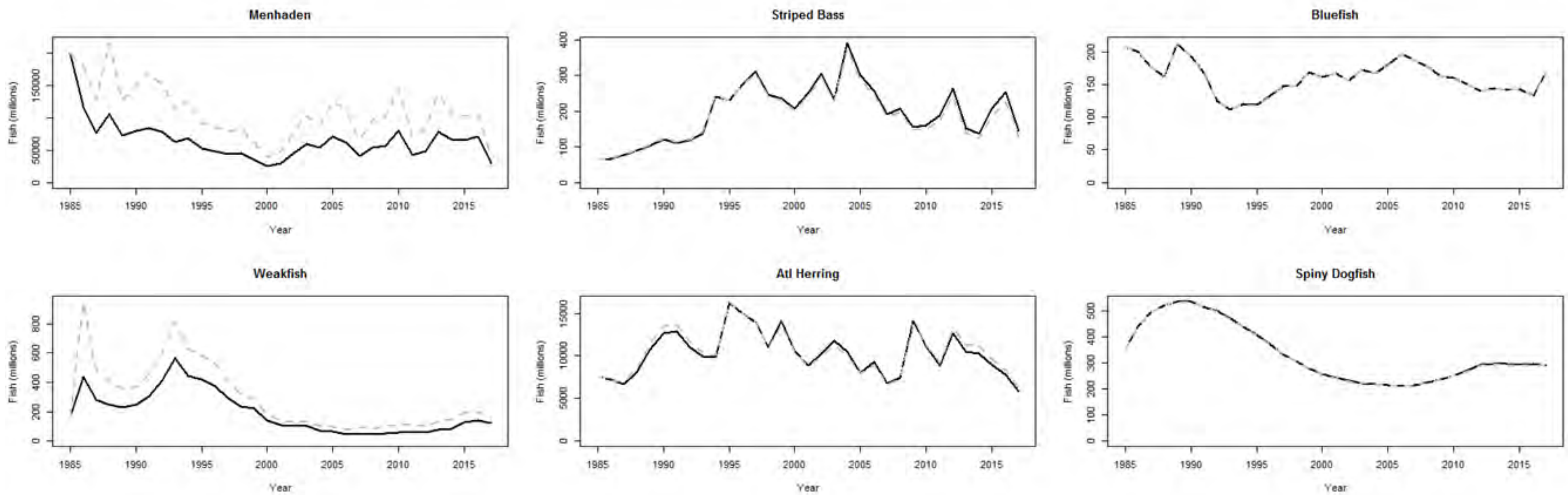


Figure 87. Predicted annual total abundance by species predicted with no trophic interactions (dashed gray line) and multispecies (solid black line) models from the VADER model.

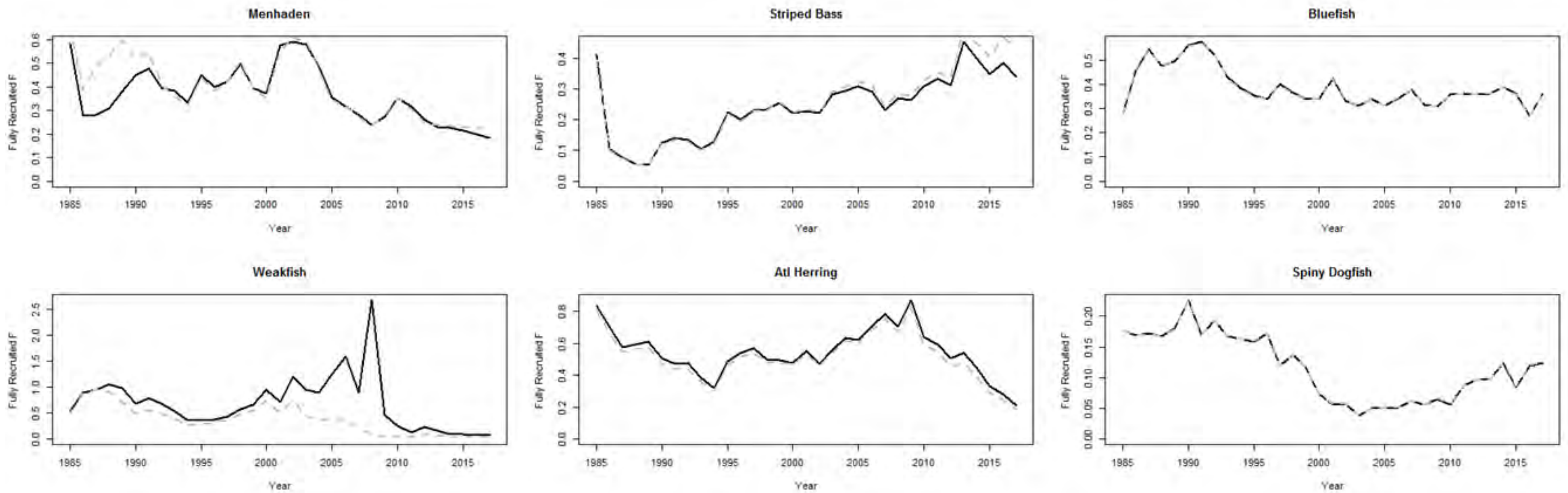


Figure 88. **Predicted annual fully recruited fishing mortality (*F*) by species from predicted with no trophic interactions (dashed black line) and multispecies (solid black line) models from the VADER model.**

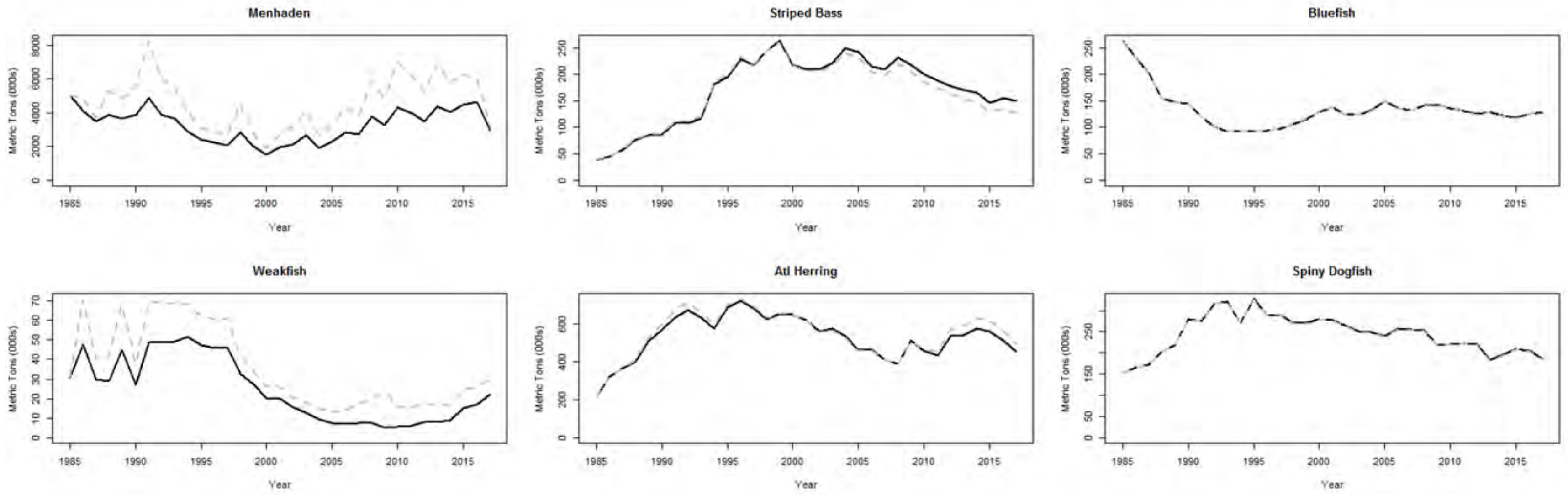


Figure 89. **Predicted annual total biomass by species from predicted with no trophic interactions (dashed gray line) and multispecies (solid black line) models from the VADER model.**

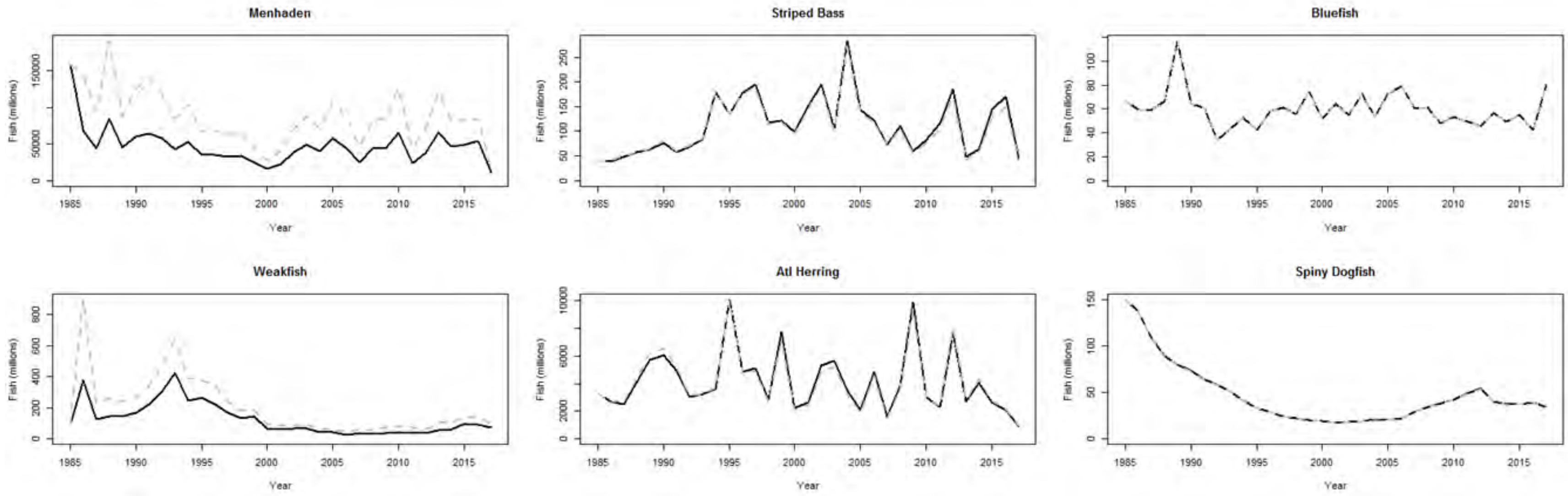


Figure 90. **Predicted annual recruitment (first age in the model, species dependent) by species from predicted with no trophic interactions (dashed grey line) and multispecies (solid black line) models from the VADER model**

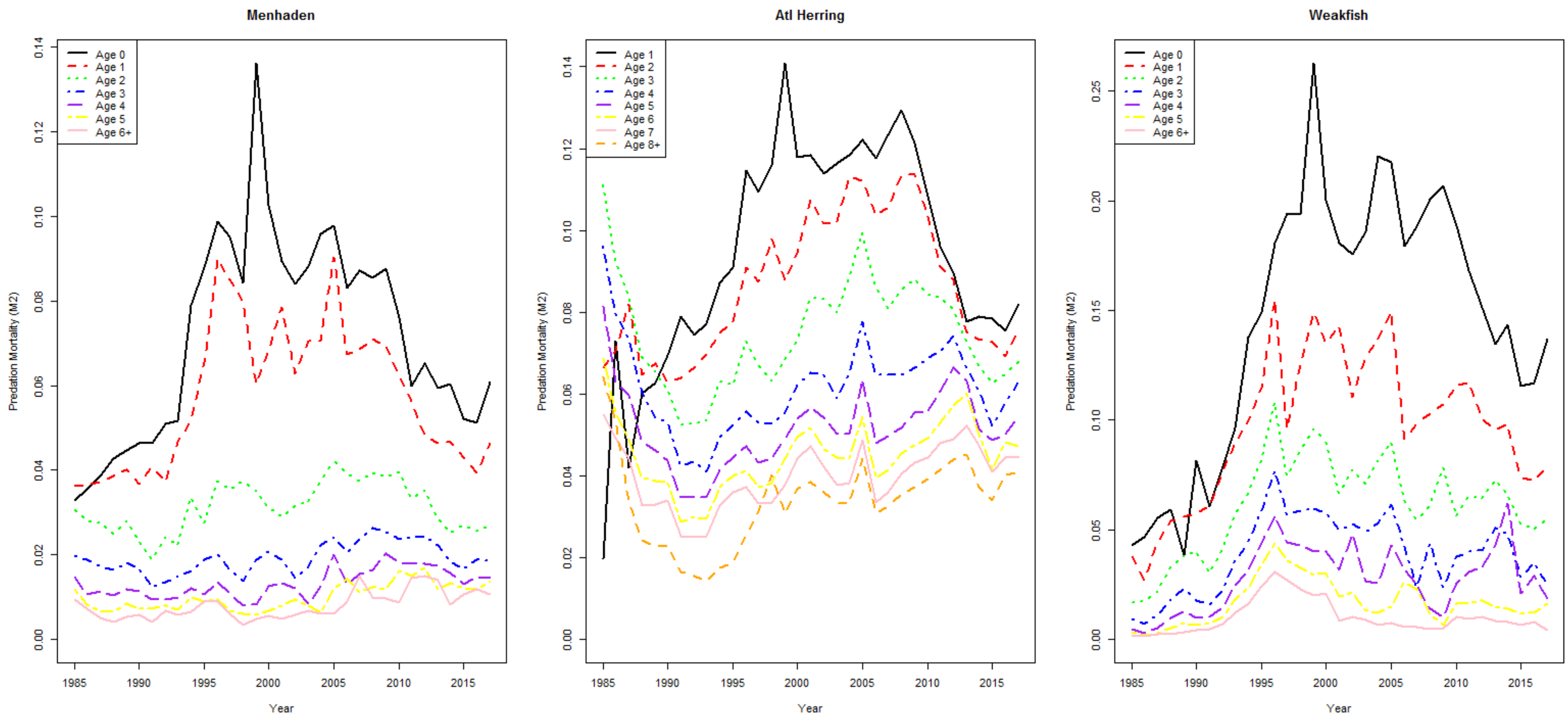


Figure 91. **Predicted annual predation mortality-at-age (M_2) for Atlantic menhaden, weakfish, and Atlantic herring from the VADER model.**

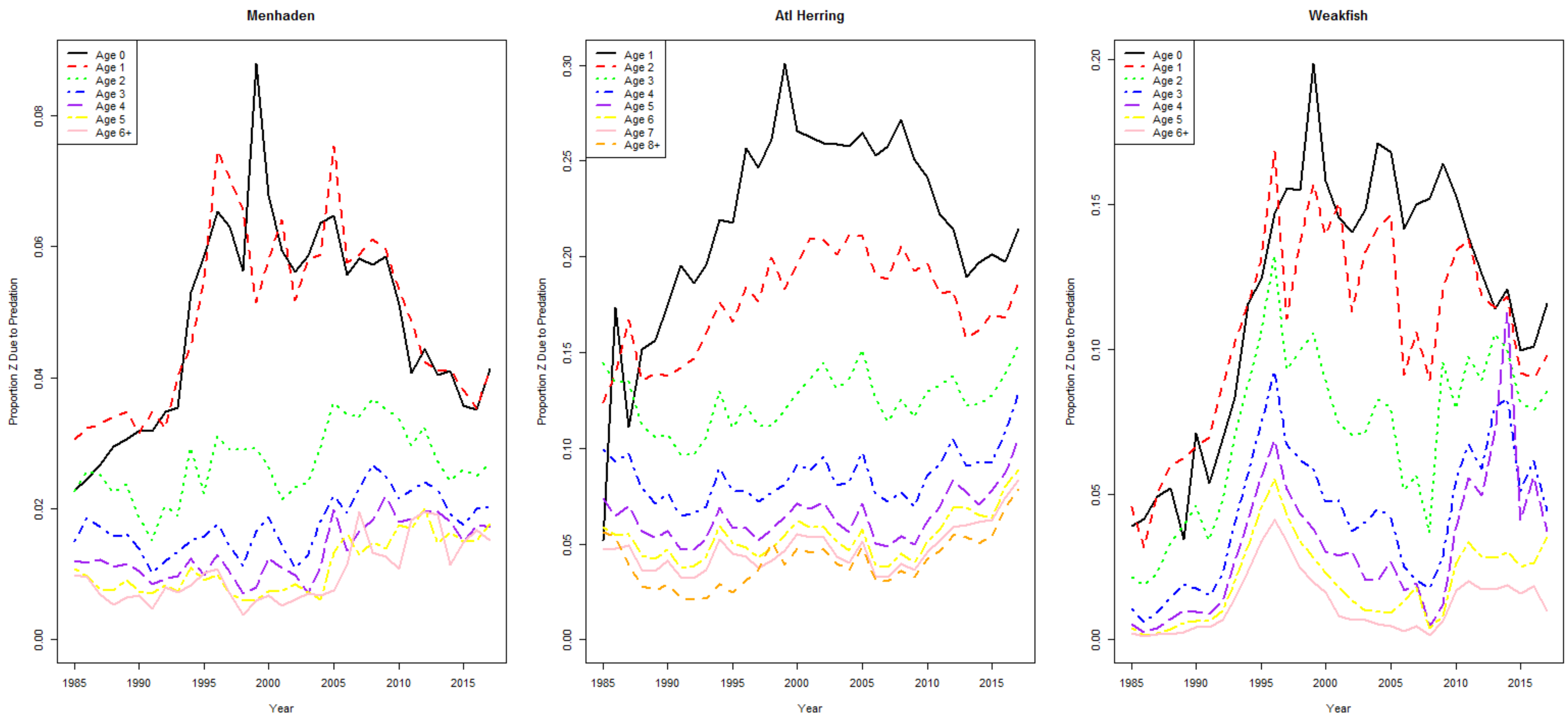


Figure 92. **Predicted proportion total mortality (*Z*) at age from predation by species from multispecies models from the VADER model.**

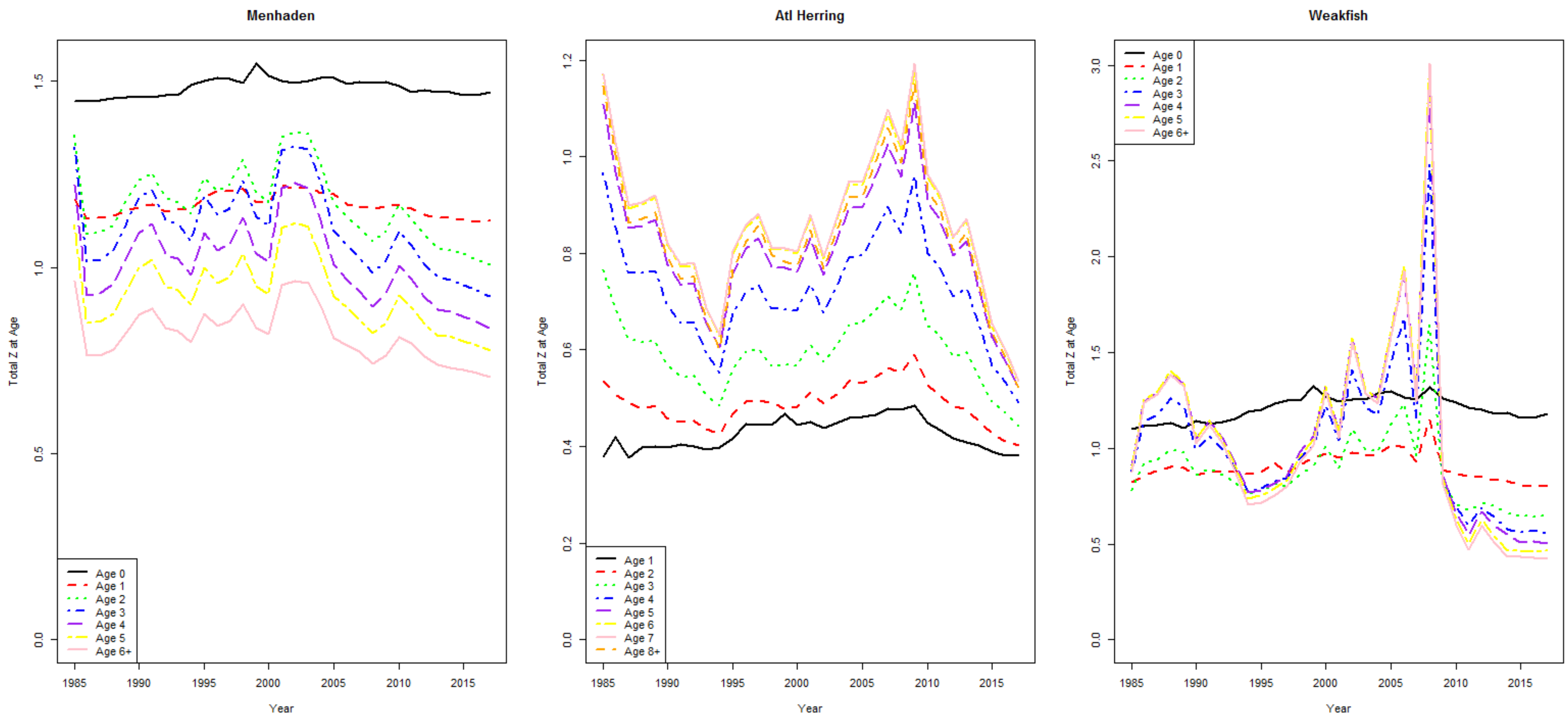


Figure 93. **Predicted annual total mortality (*Z*) at age by species from the multispecies run of the VADER model.**

Figure 94. Predicted annual consumption in thousands of metric tons by prey for striped bass, bluefish, weakfish, and spiny dogfish from the VADER model.

Figure 95. Predicted annual consumption in thousands of metric tons by predator for Atlantic menhaden, weakfish, and Atlantic herring from the VADER model.

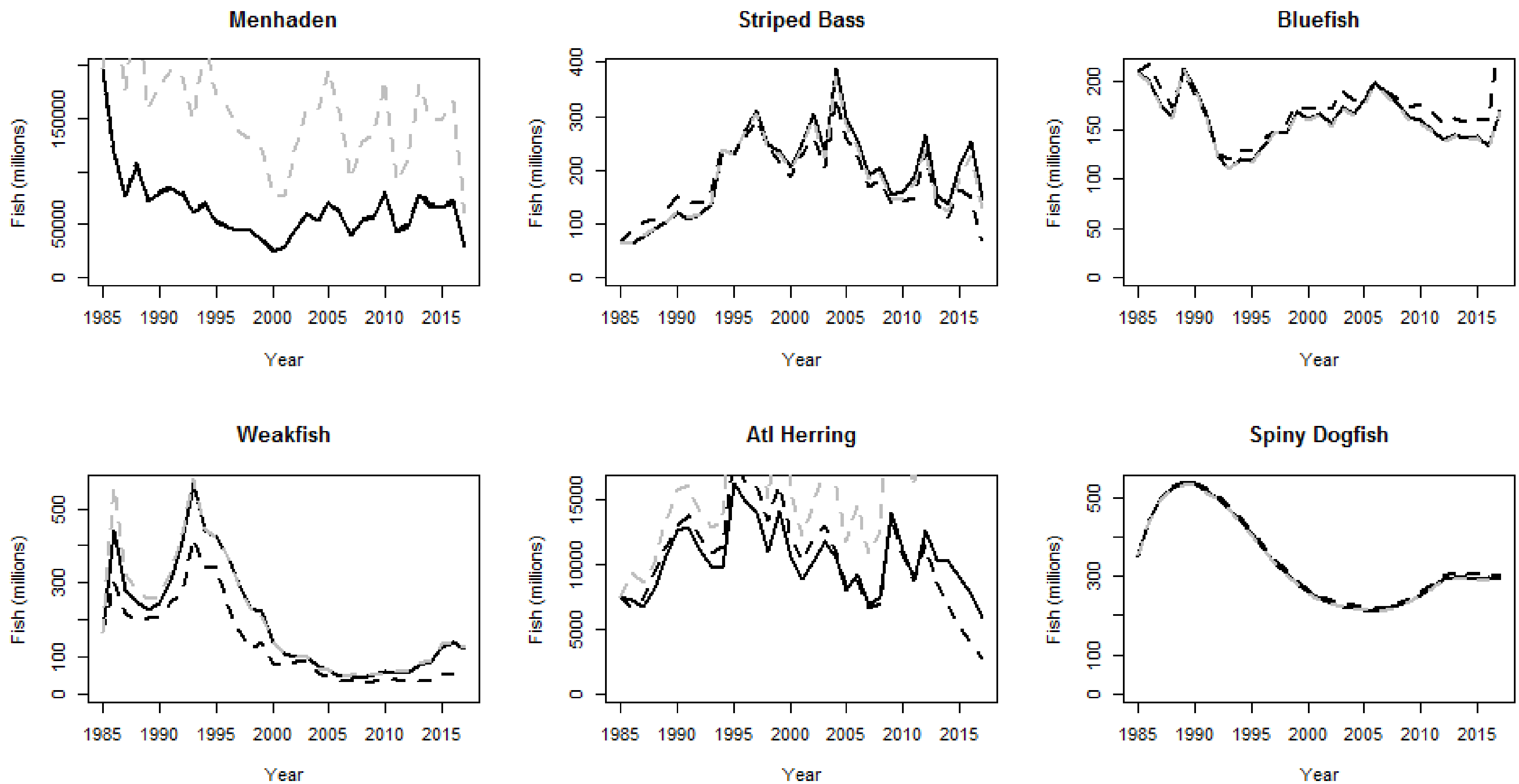


Figure 96. **Predicted annual total abundance by species predicted with alternate indices (dashed black line), alternate diet composition (dashed gray line), and multispecies (solid black line) runs from the VADER model.**

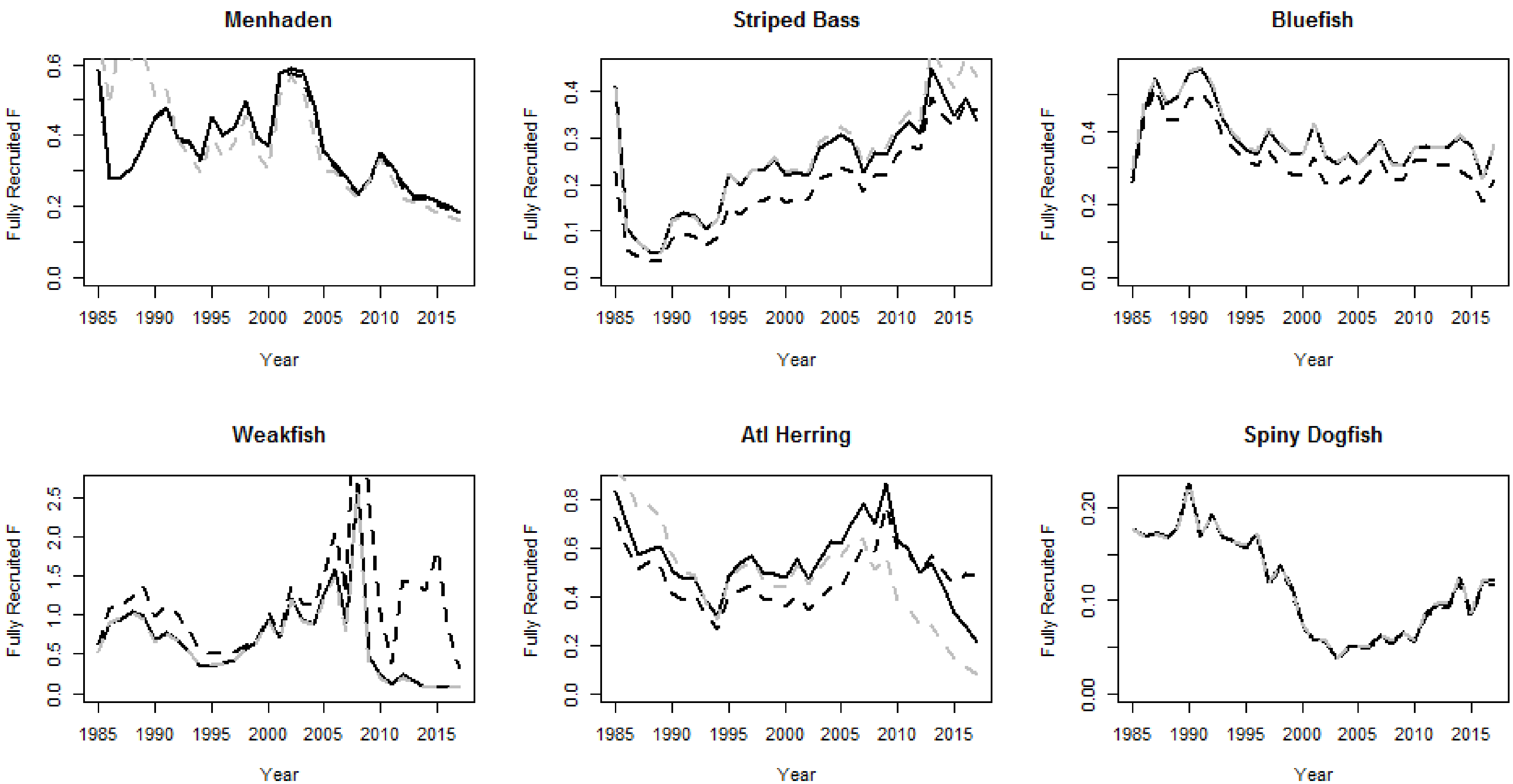


Figure 97. Predicted annual fully recruited fishing mortality (*F*) by species from predicted with alternate indices (dashed black line), alternate diet composition (dashed gray line), and multispecies (solid black line) runs from the VADER model.

Figure 98. **Predicted annual total biomass by species from predicted with alternate indices (dashed black line), alternate diet composition (dashed gray line), and multispecies (solid black line) runs from the VADER model.**

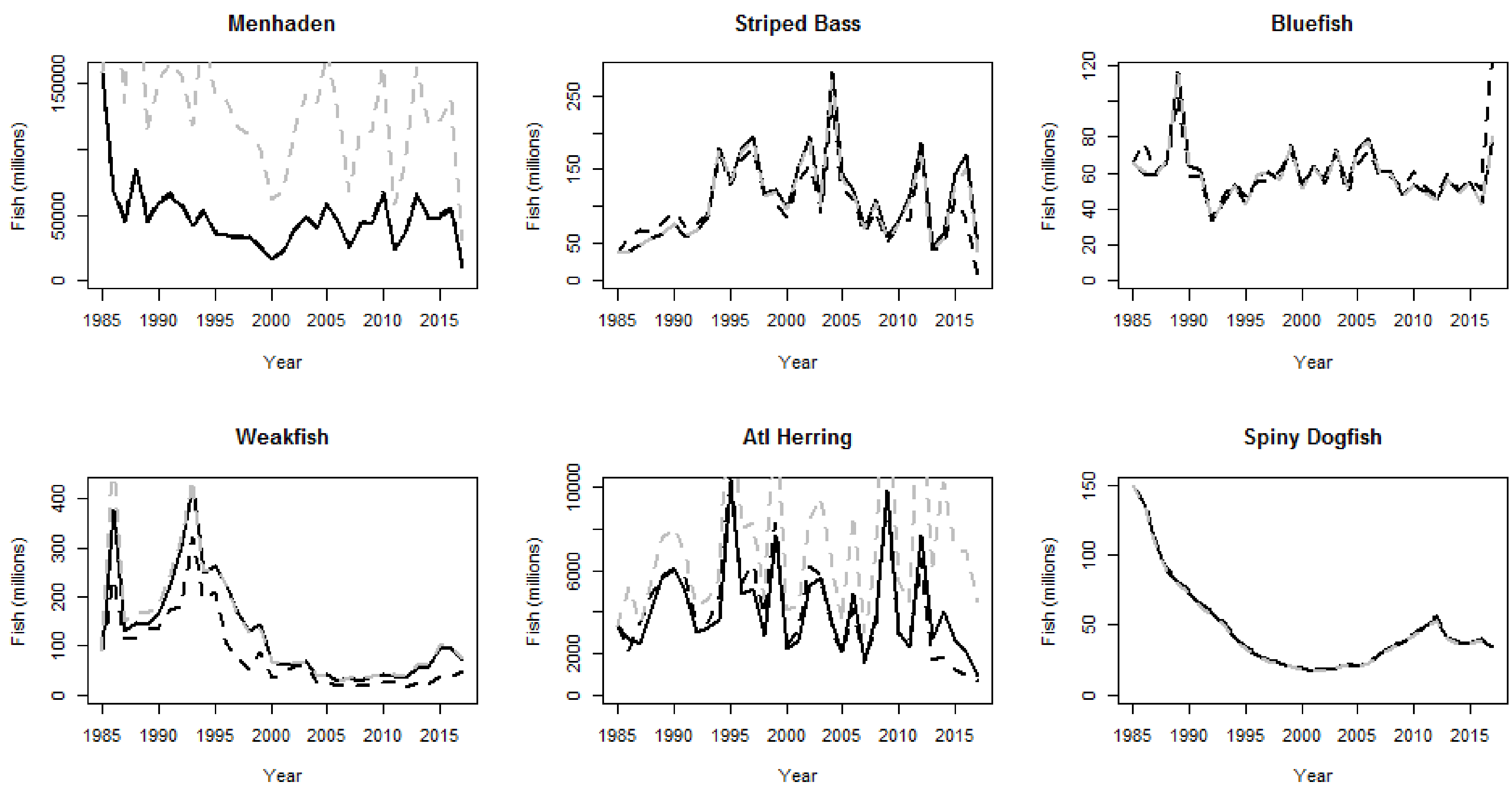


Figure 99. **Predicted annual recruitment (first age in the model, species dependent) by species from predicted with alternate indices (dashed black line), alternate diet composition (dashed gray line), and multispecies (solid black line) runs from the VADER model.**

Figure 100. Predicted average predation mortality (*M_2*) for Atlantic menhaden, weakfish, and Atlantic herring from the alternate diet run (dashed gray line) and the base run (solid black line) from the VADER model.

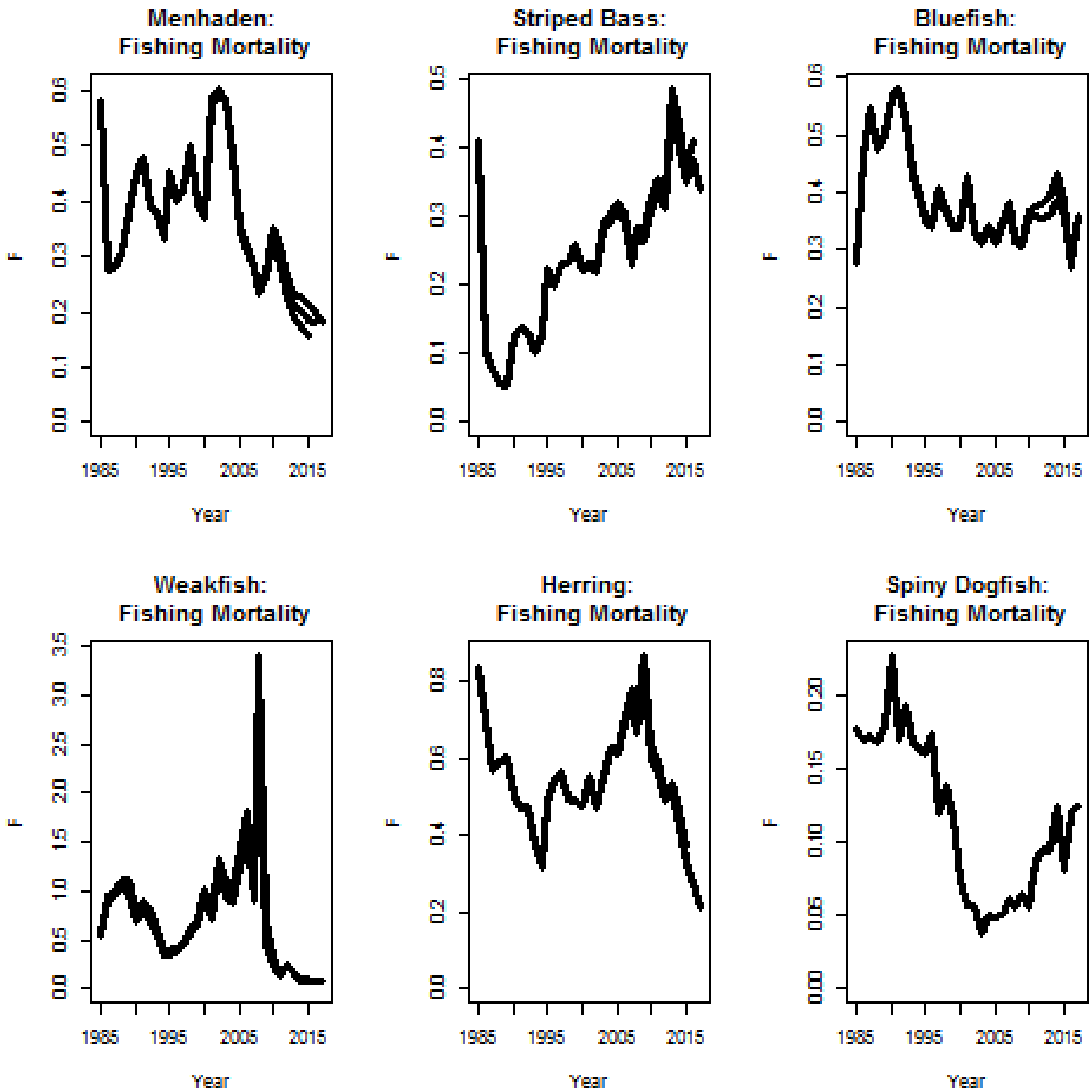


Figure 101. **Retrospective analysis for full fishing mortality for all six species from the VADER model.**

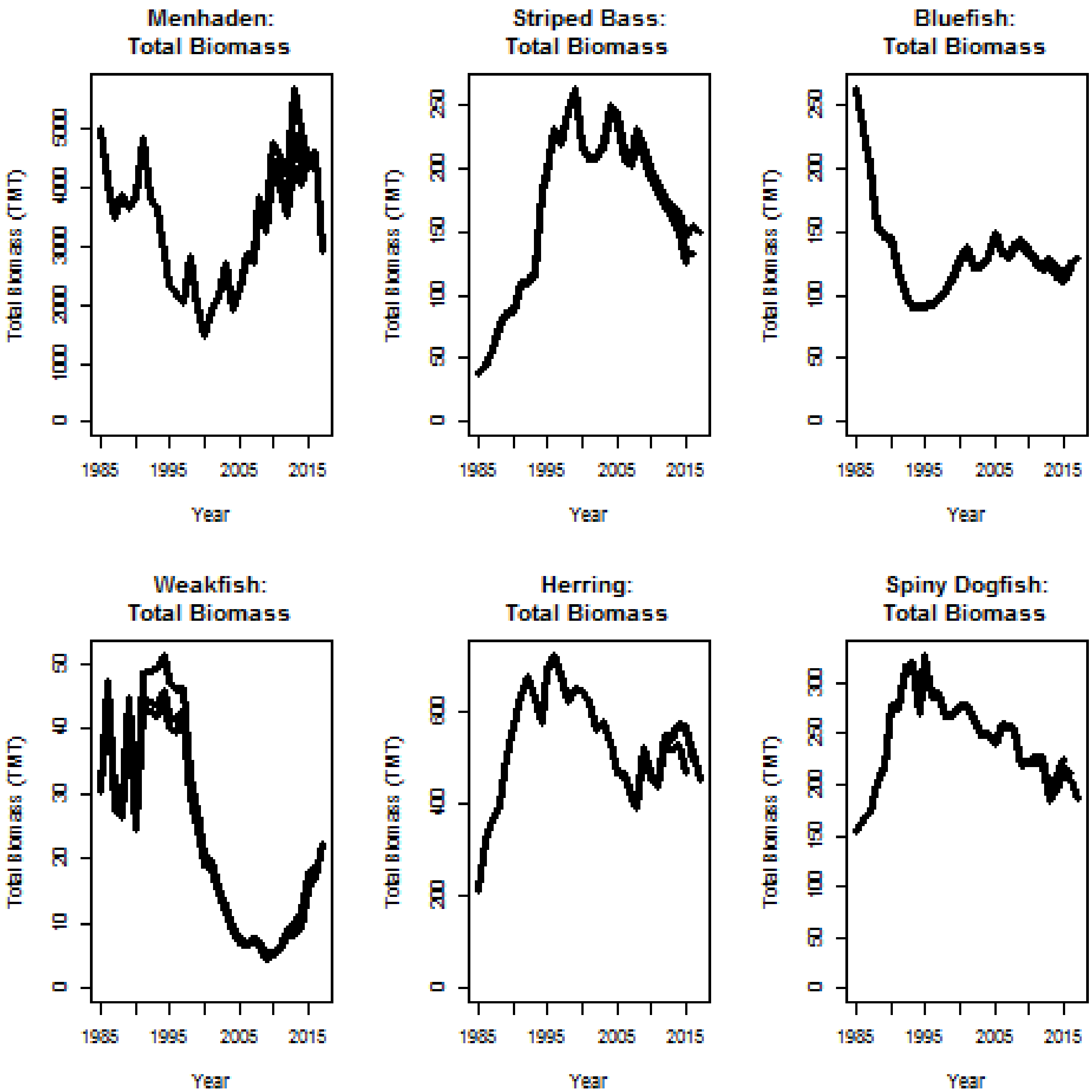


Figure 102. Retrospective analysis for total biomass for all six species from the VADER model.

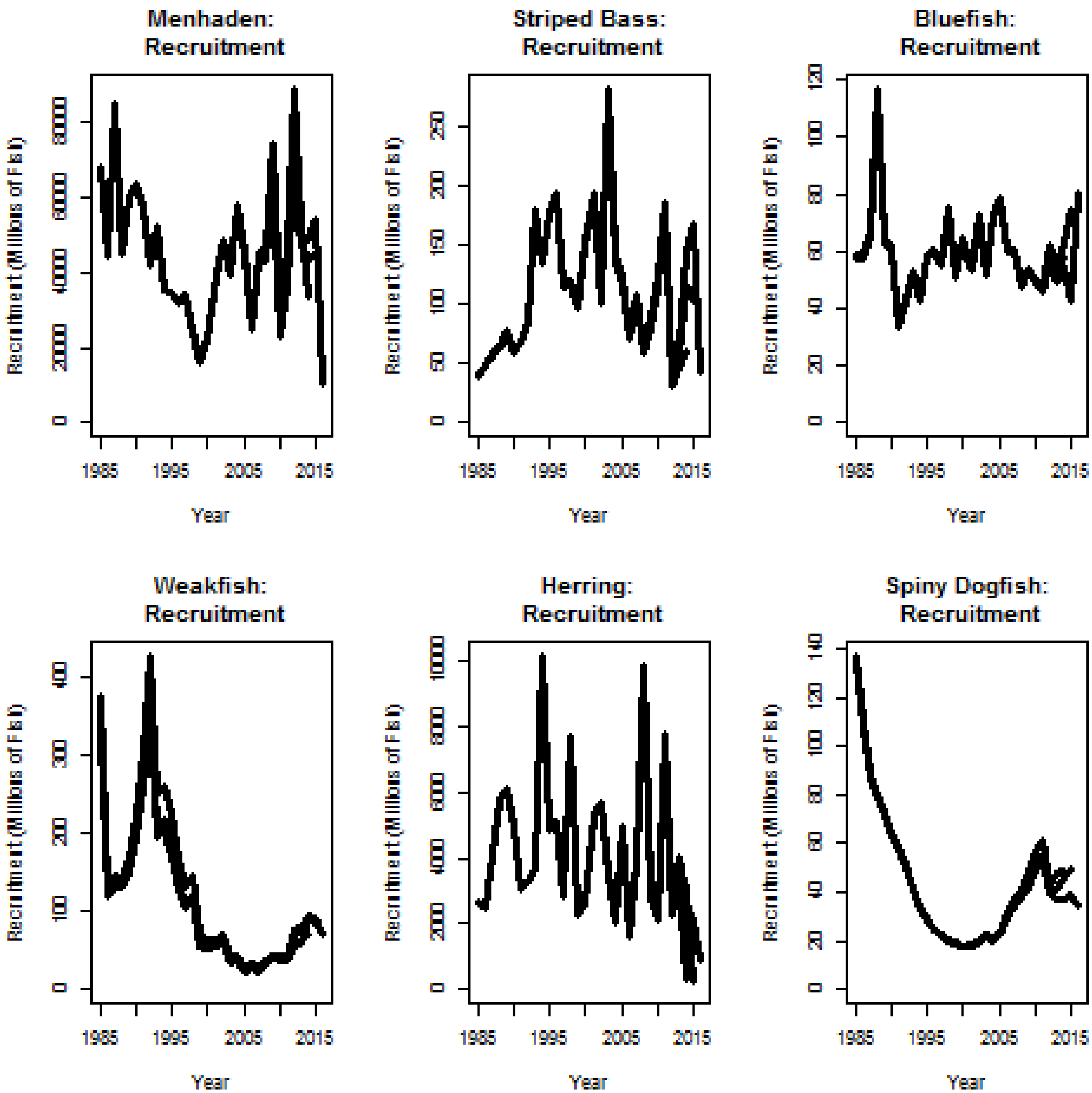


Figure 103. Retrospective analysis for recruitment for all six species from the VADER model.

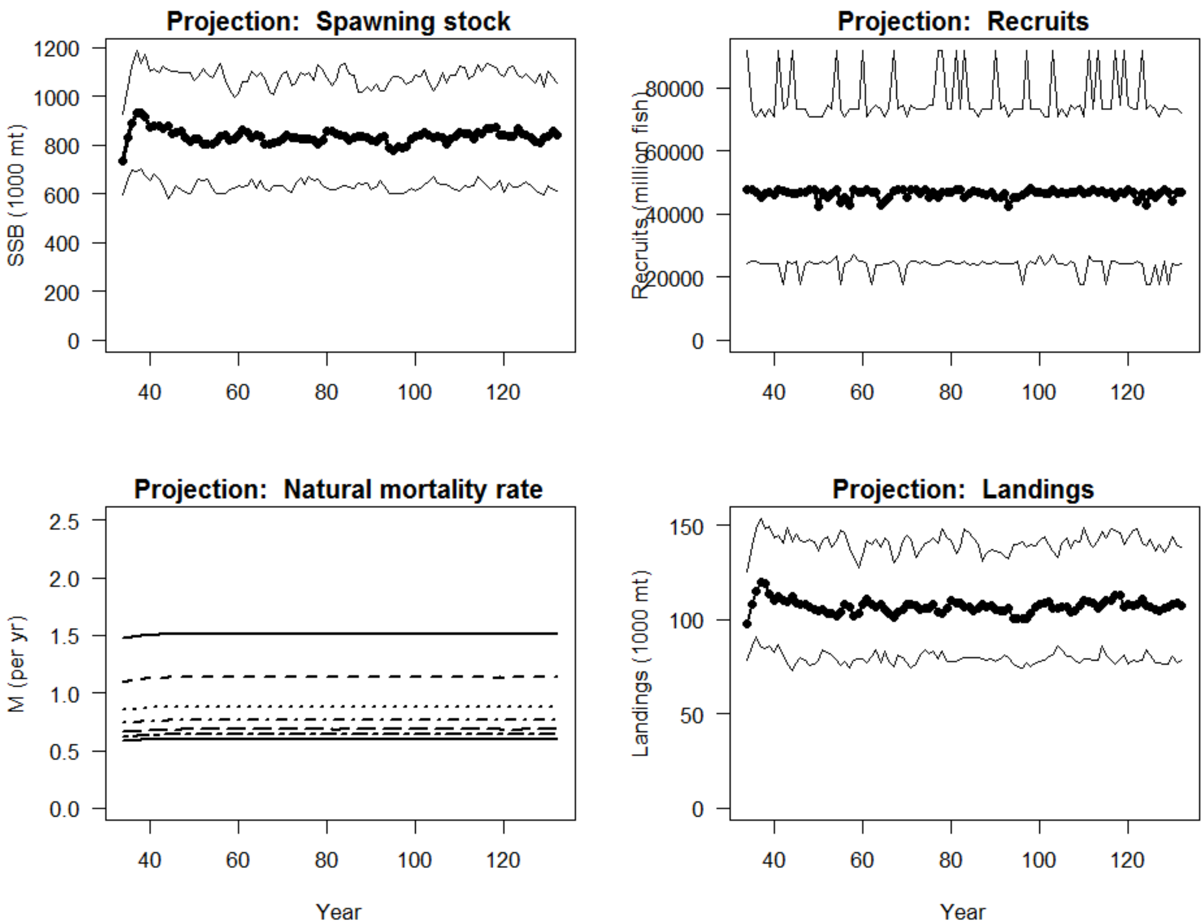


Figure 104. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic menhaden under scenario 1 from the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the M plot, the different lines represent *M*-at-age.**

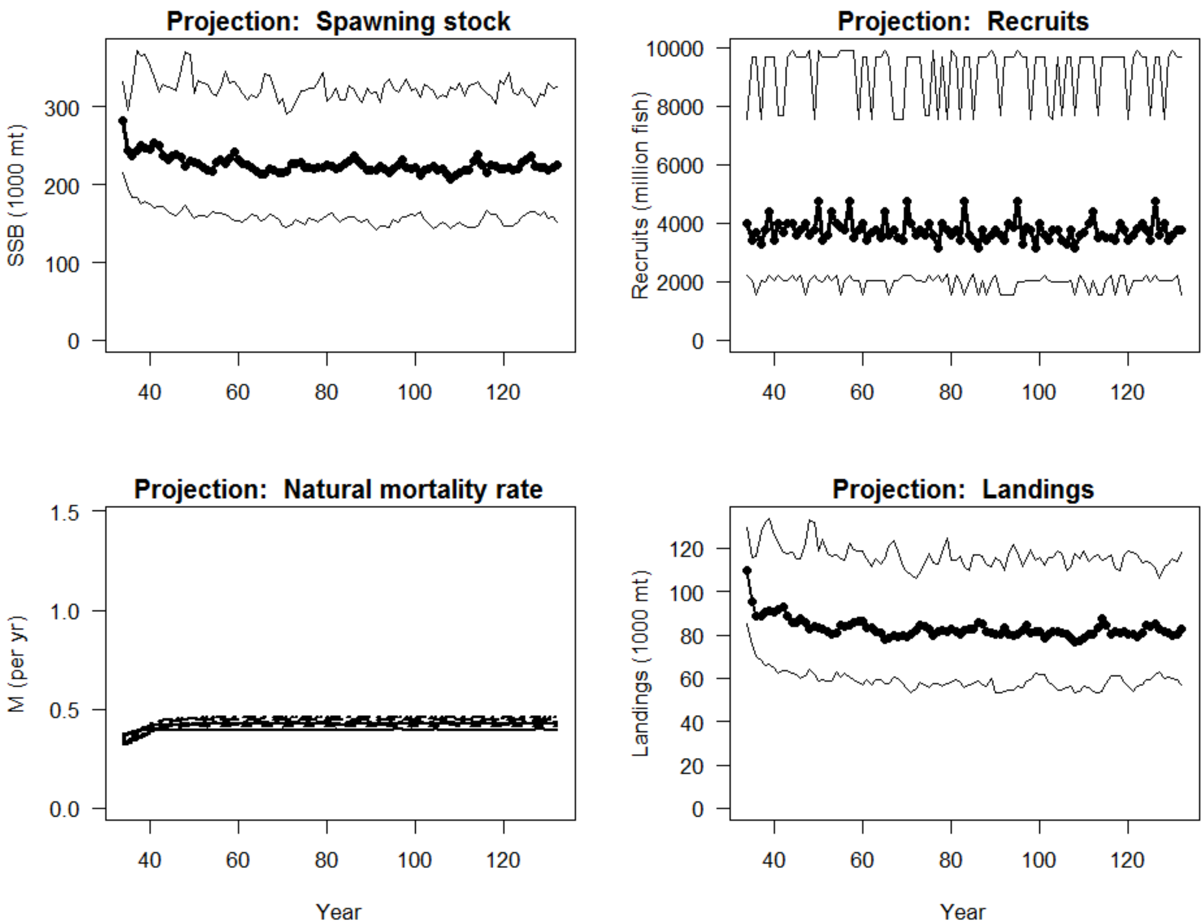


Figure 105. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic herring under scenario 1 for the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*) For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

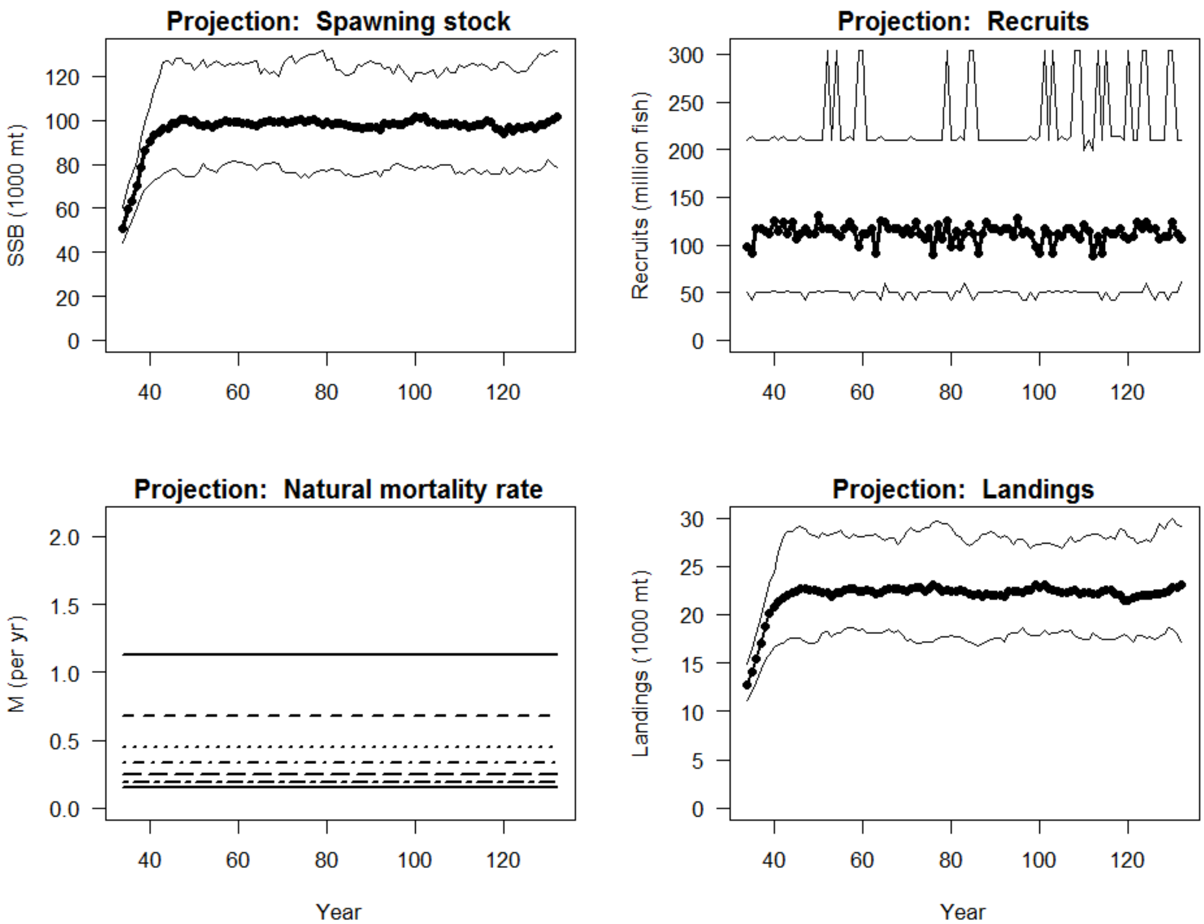


Figure 106. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for striped bass under scenario 1 for the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*).For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

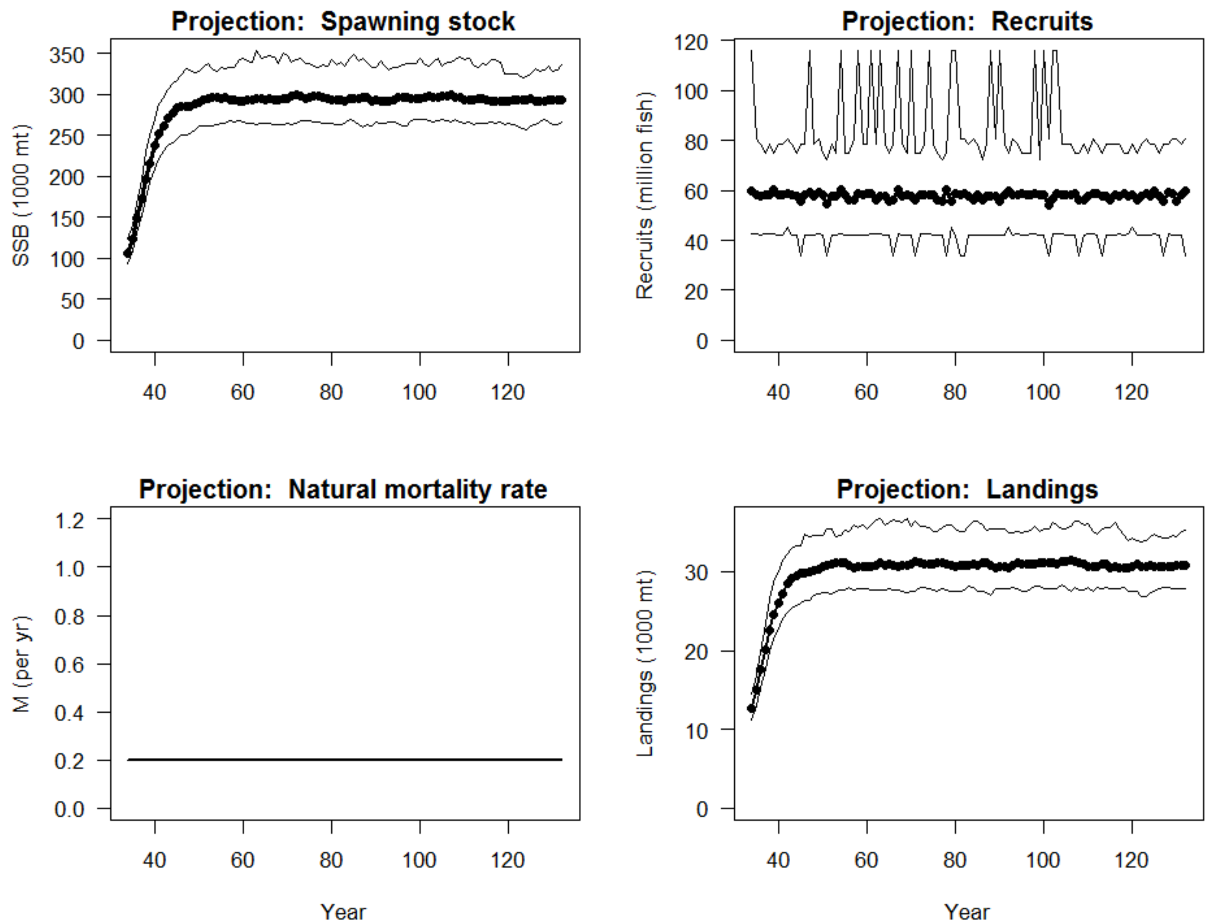


Figure 107. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for bluefish under scenario 1 for the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

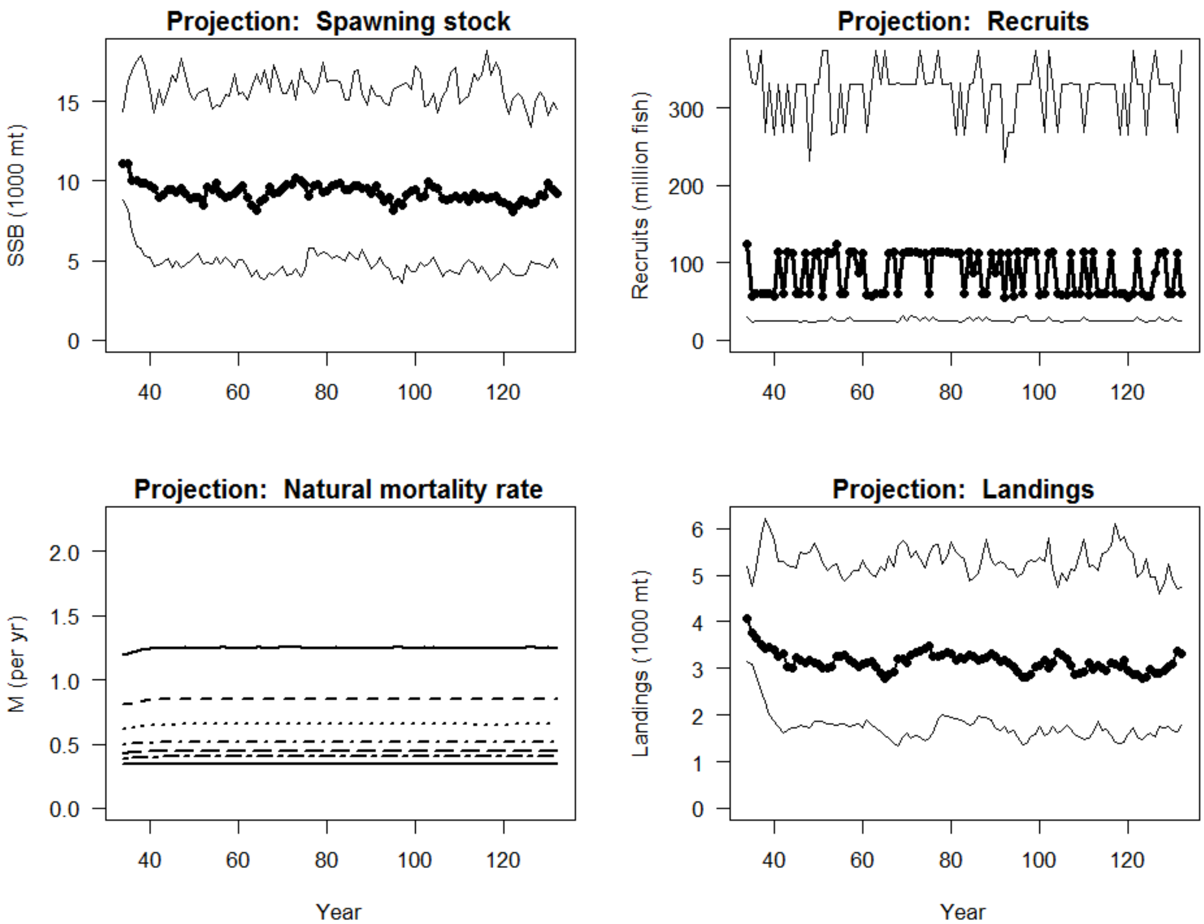


Figure 108. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for weakfish under scenario 1 for the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

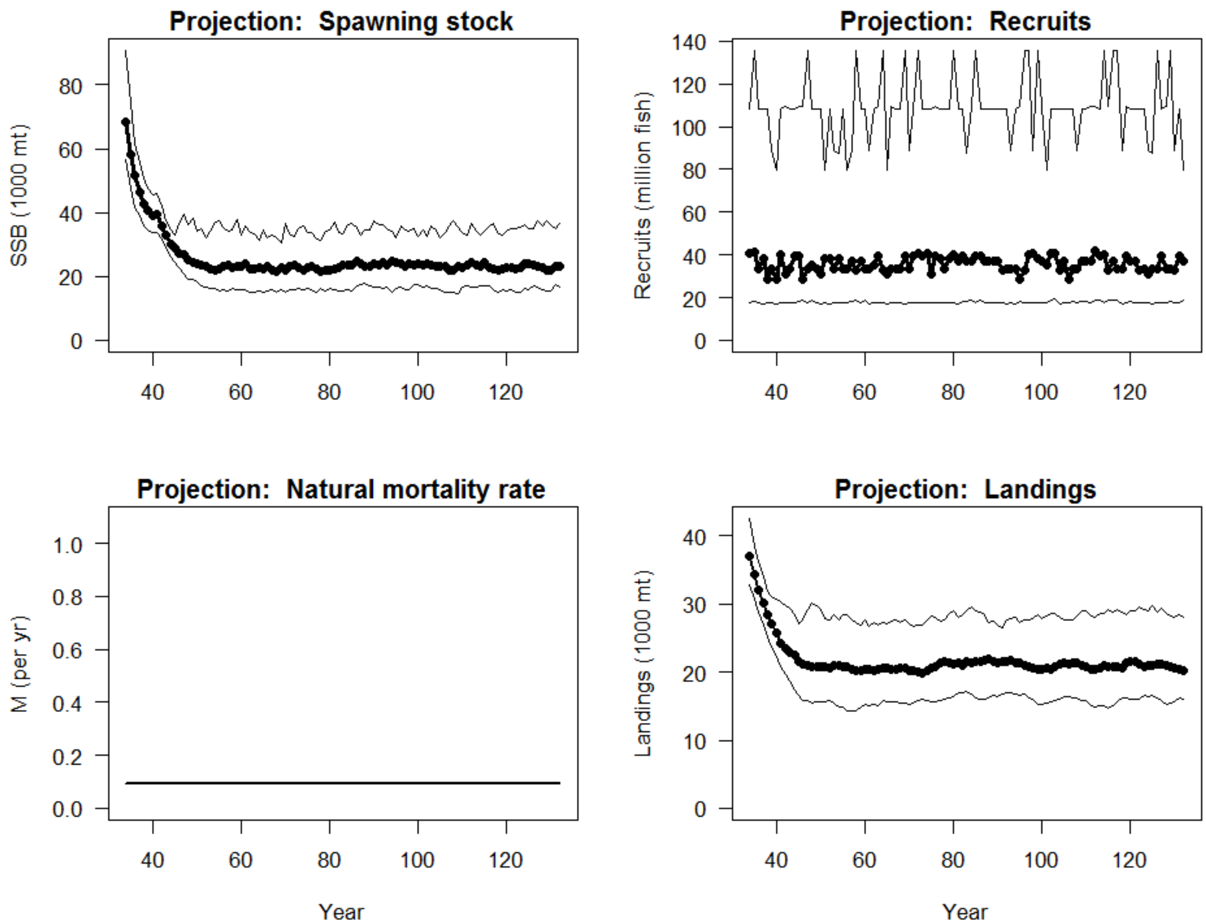


Figure 109. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for spiny dogfish under scenario 1 for the VADER model (Atlantic menhaden at status quo *F*, all other species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

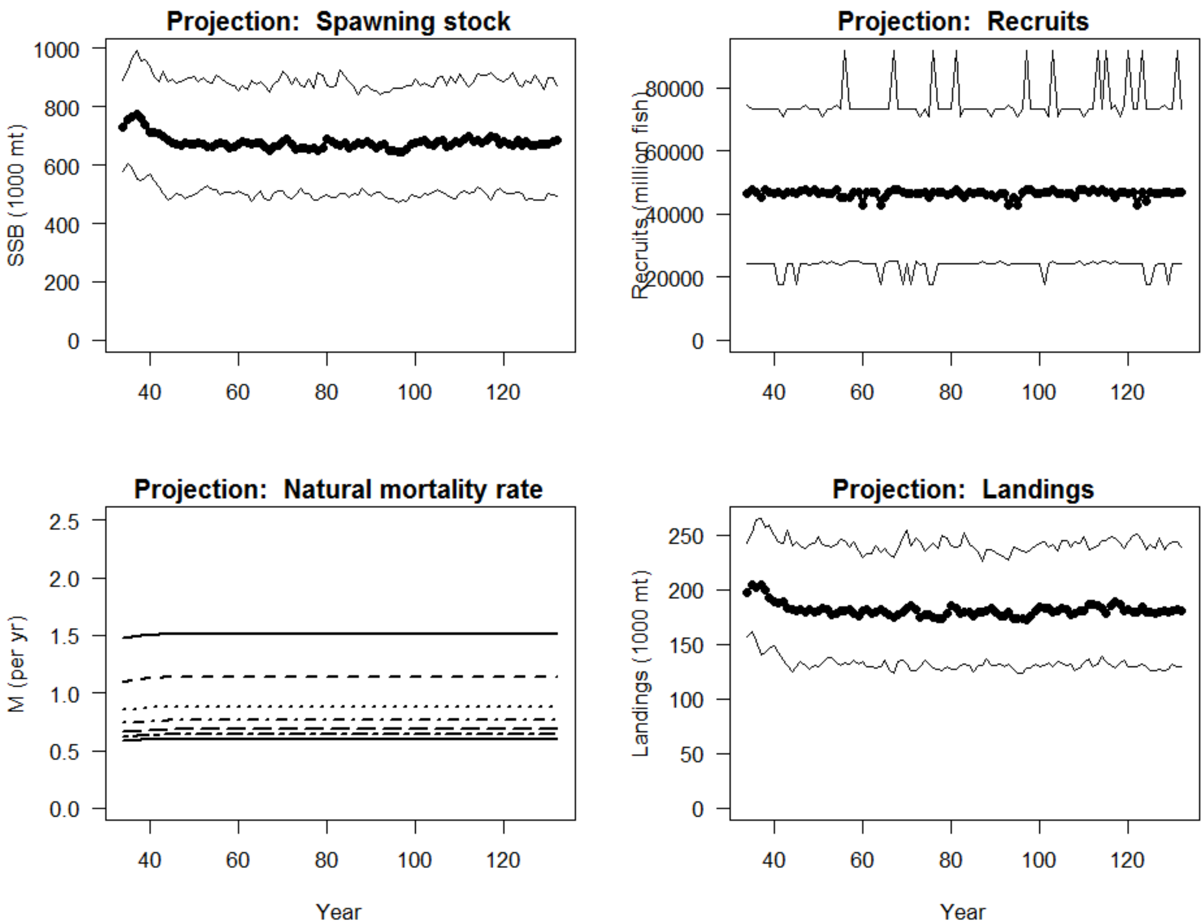


Figure 110. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic menhaden under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

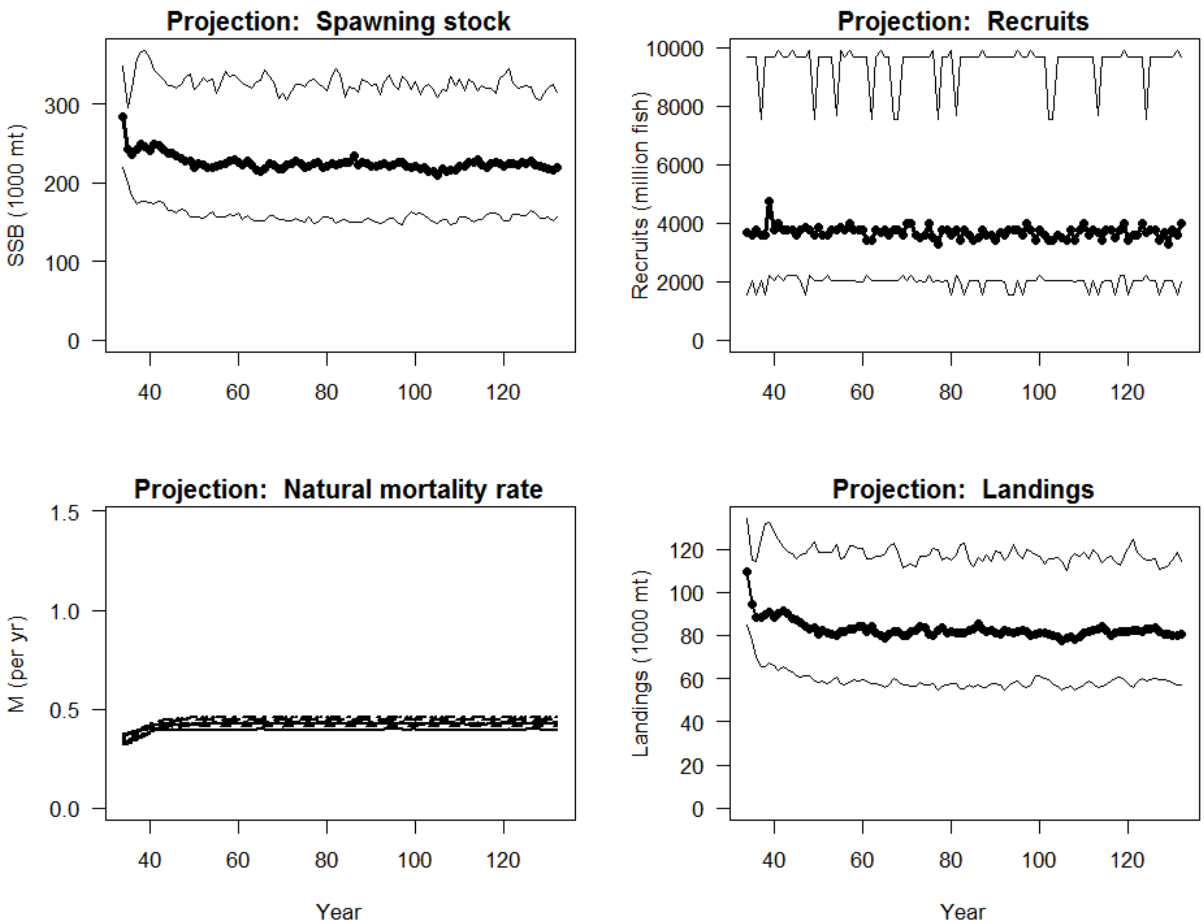


Figure 111. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic herring under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

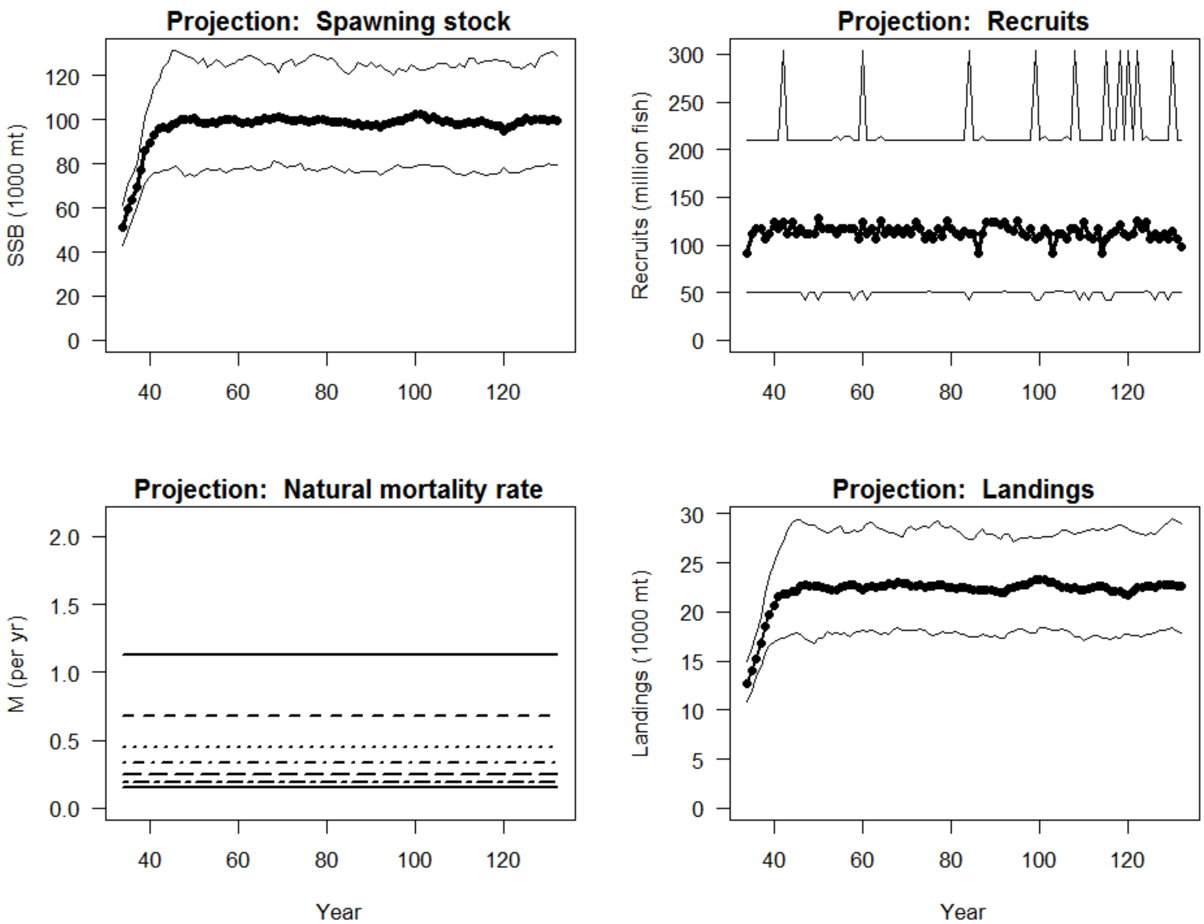


Figure 112. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for striped bass under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

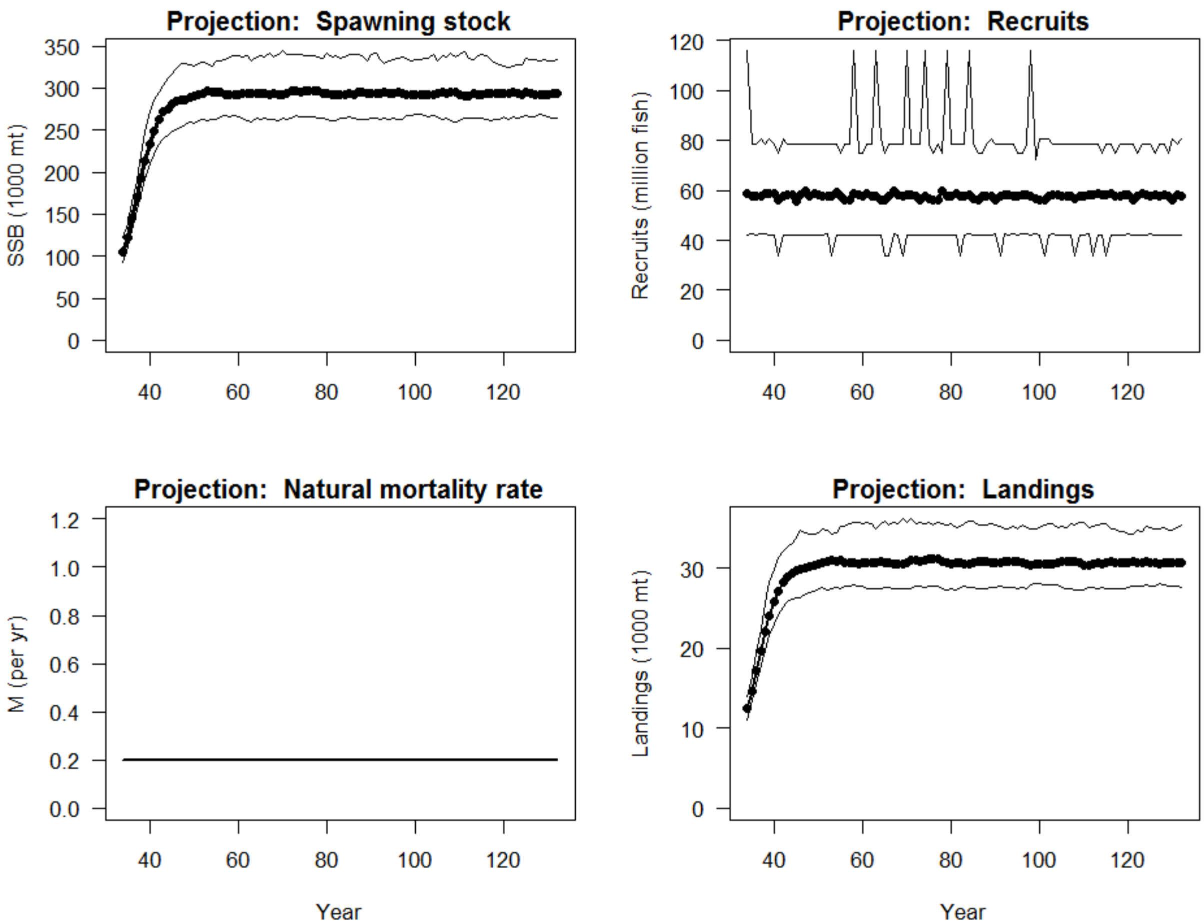


Figure 113. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for bluefish under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

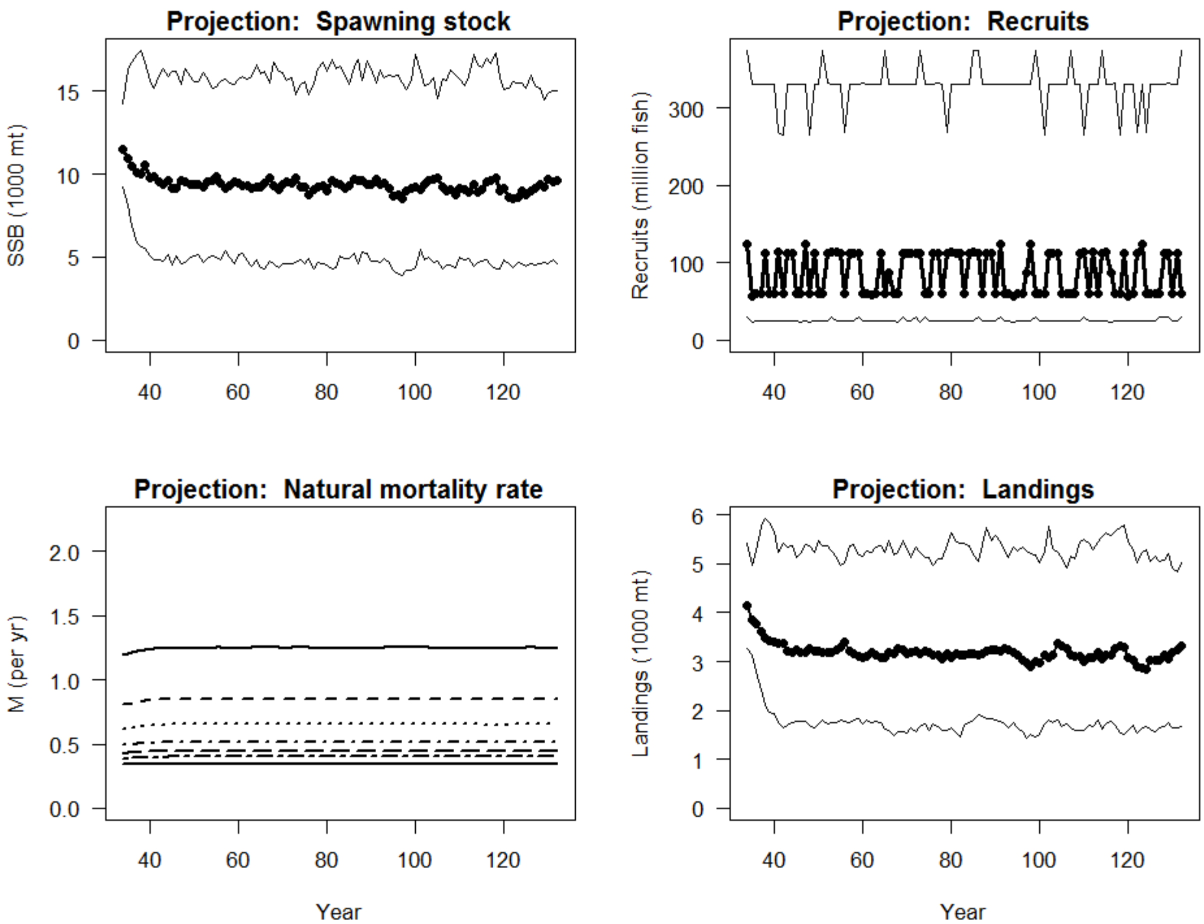


Figure 114. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for weakfish under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

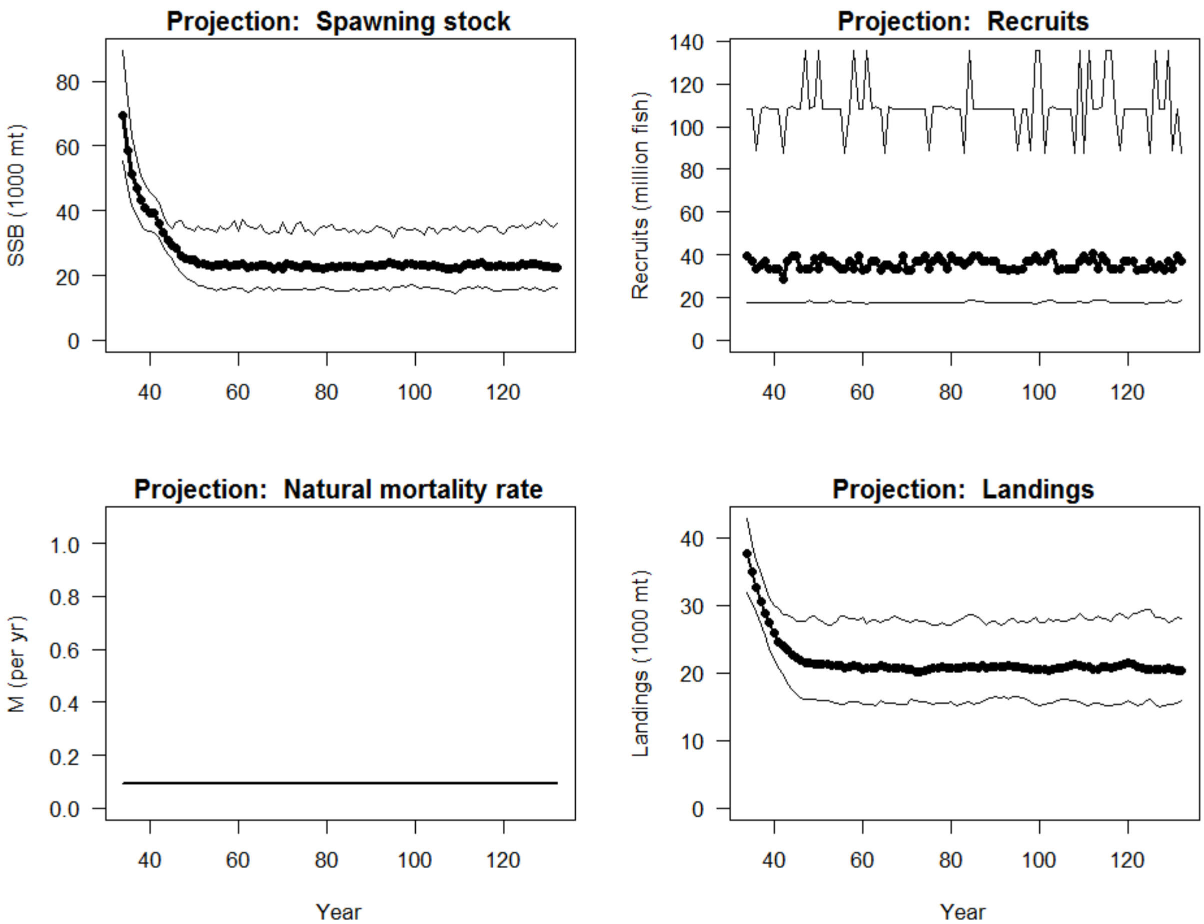


Figure 115. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for spiny dogfish under scenario 2 for the VADER model (all species at target *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

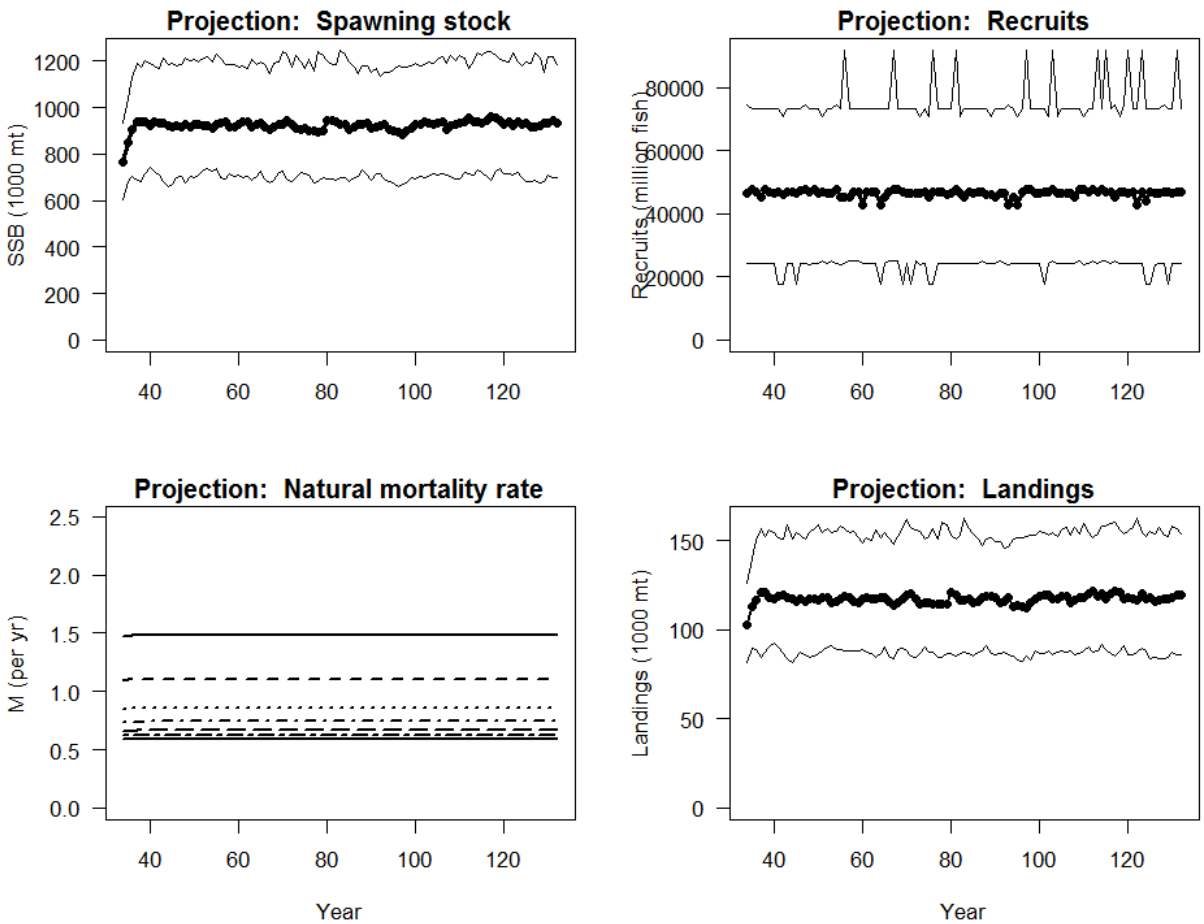


Figure 116. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic menhaden under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

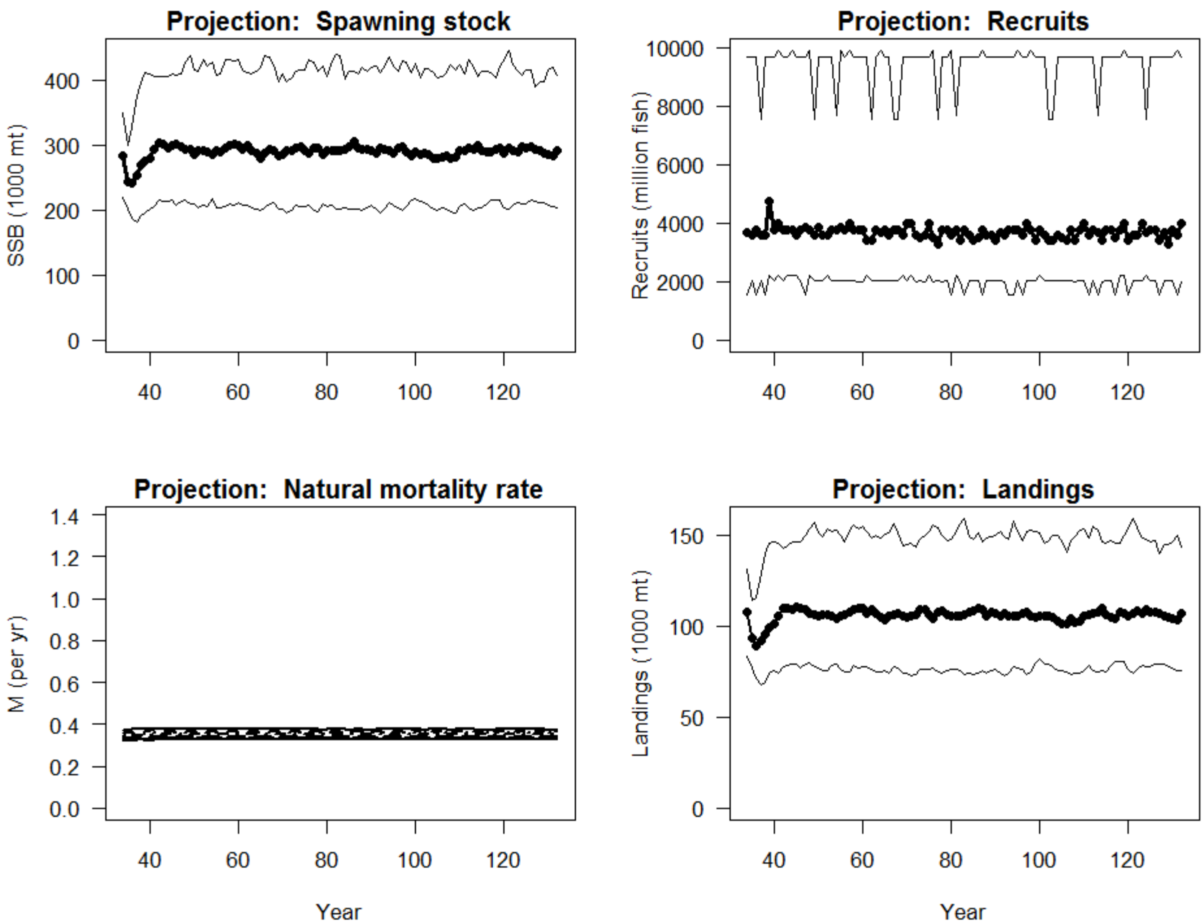


Figure 117. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic herring under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

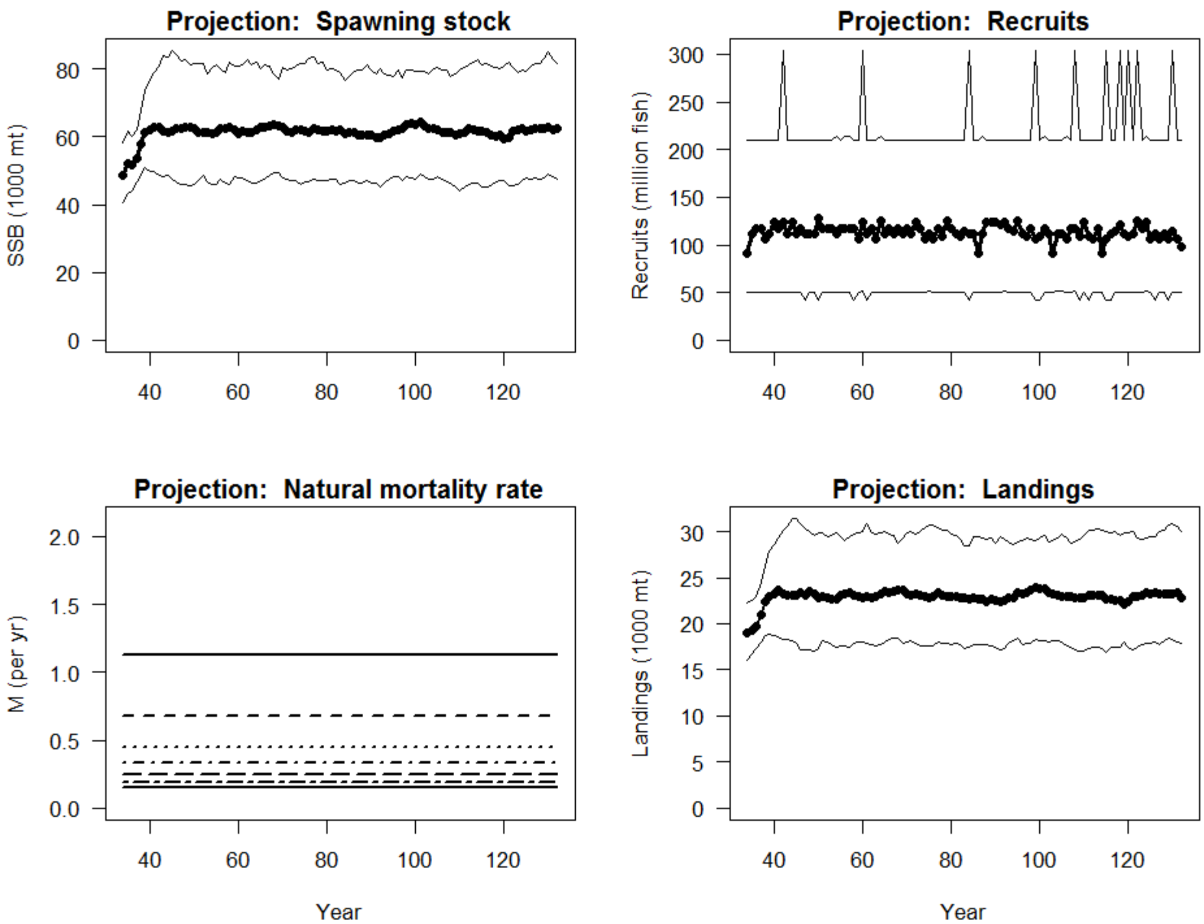


Figure 118. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for striped bass under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

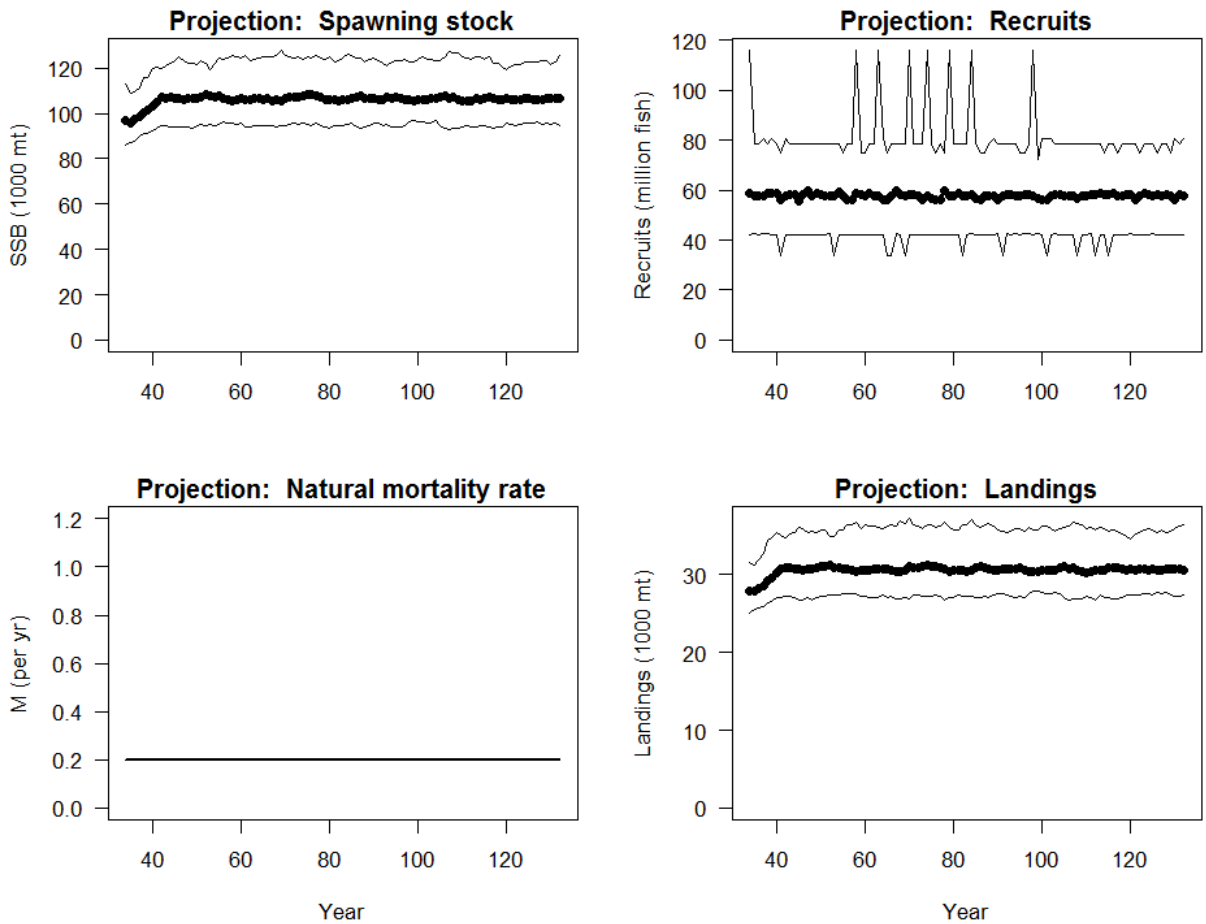


Figure 119. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for bluefish under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

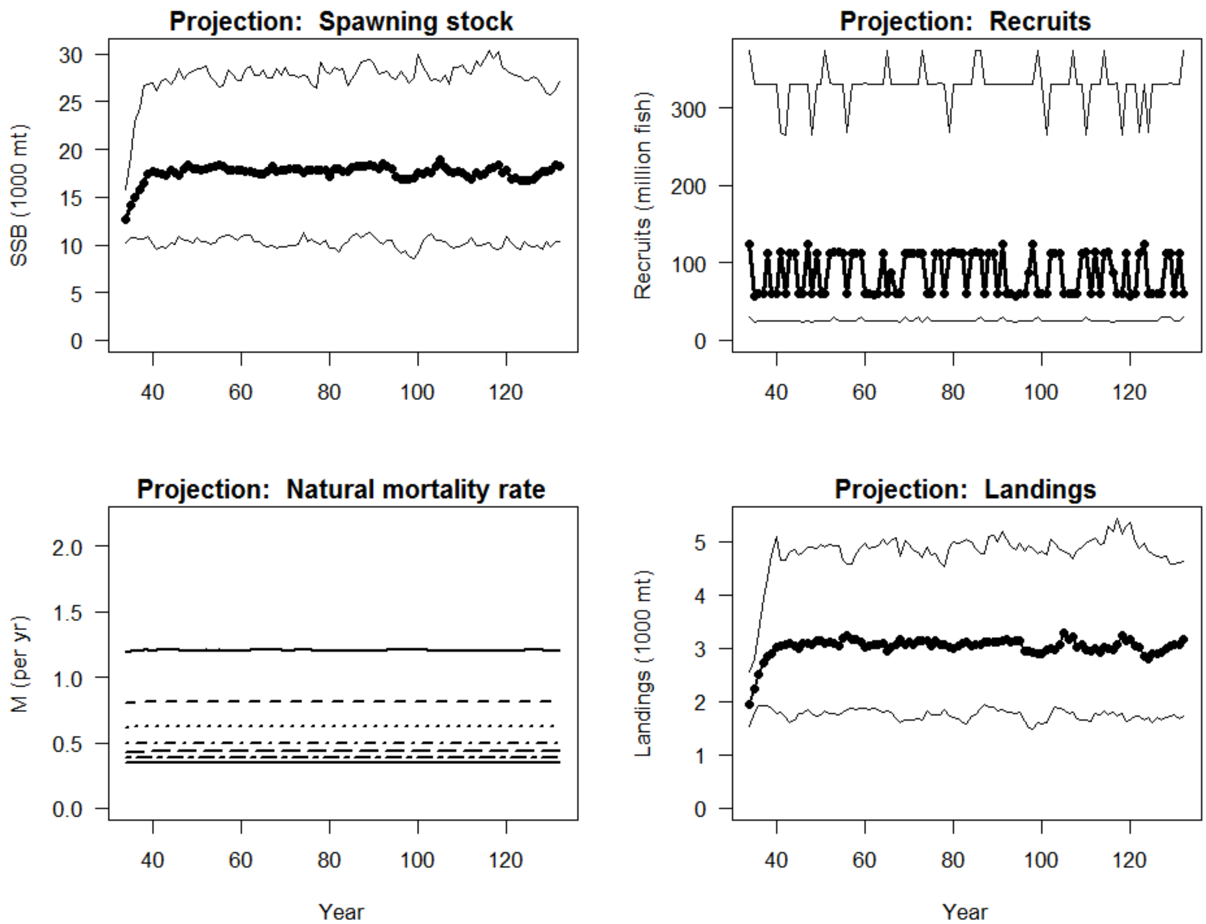


Figure 120. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for weakfish under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

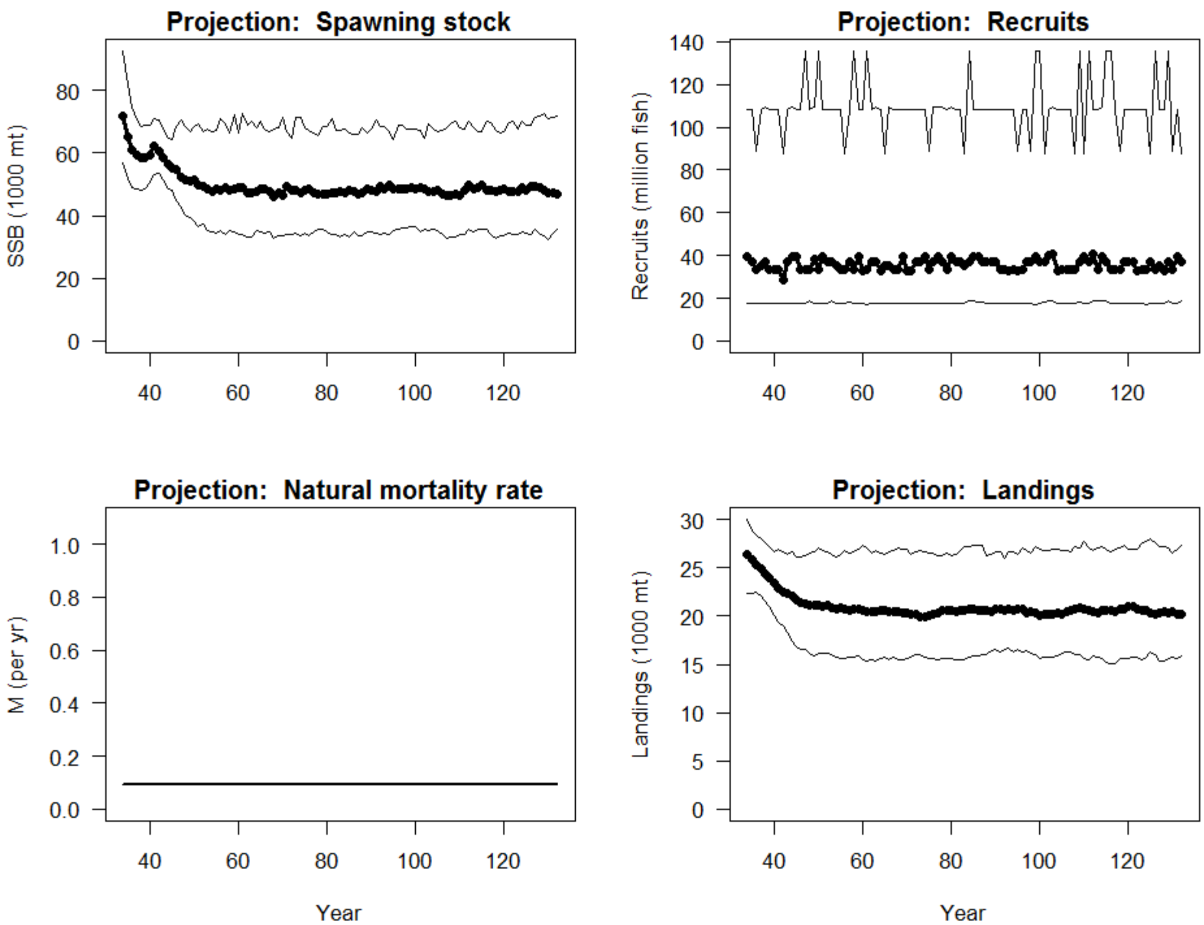


Figure 121. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for spiny dogfish under scenario 3 for the VADER model (all species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

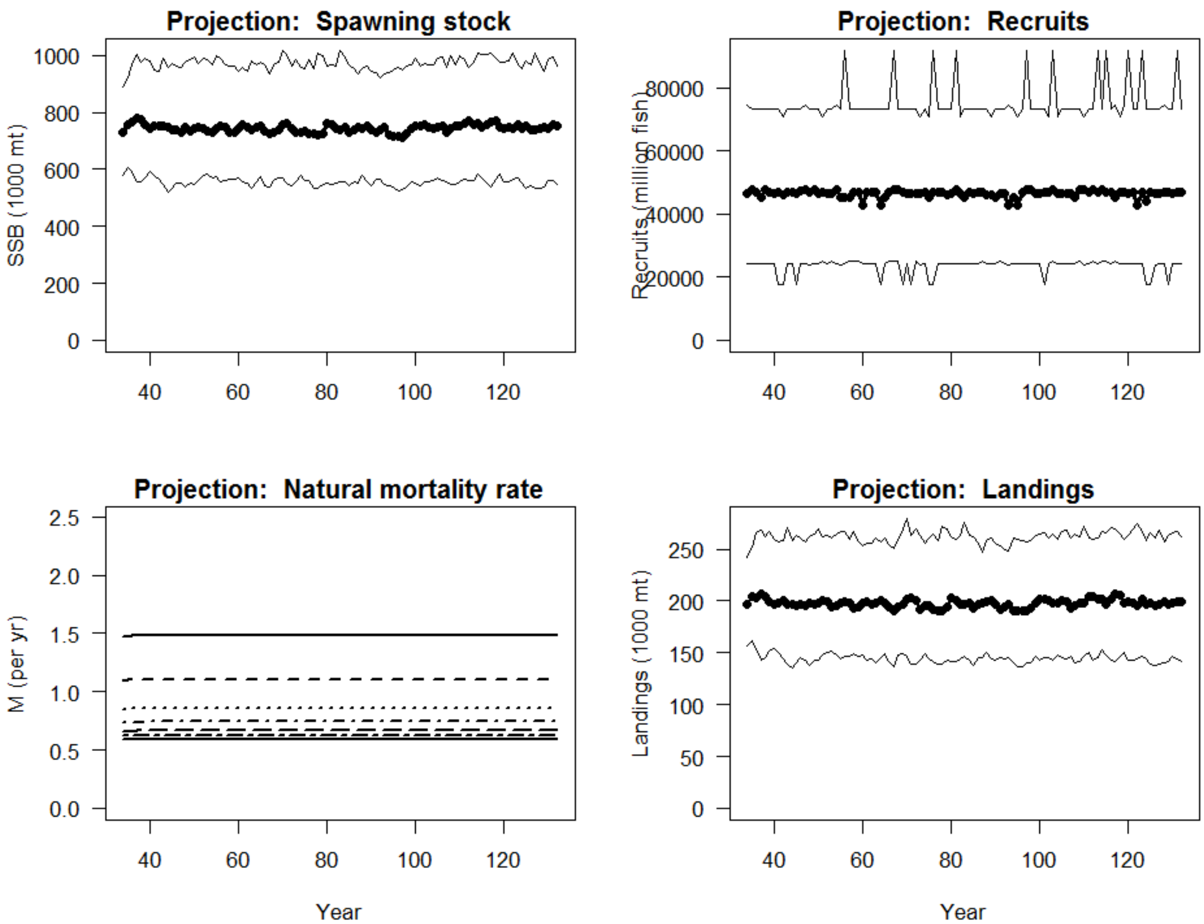


Figure 122. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic menhaden under scenario 4 for the VADER model (Atlantic menhaden at target *F*, all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

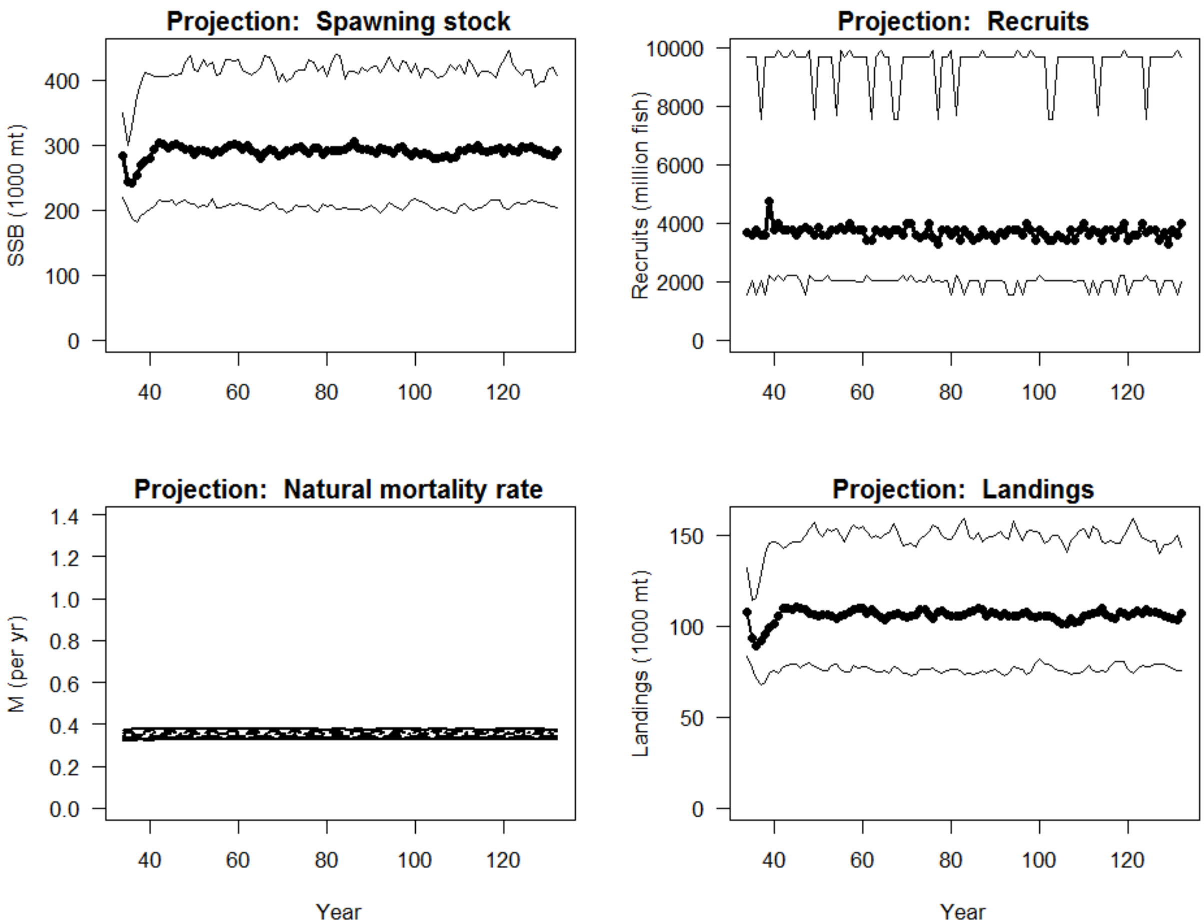


Figure 123. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for Atlantic herring under scenario 4 for the VADER model (Atlantic menhaden at target *F*, all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

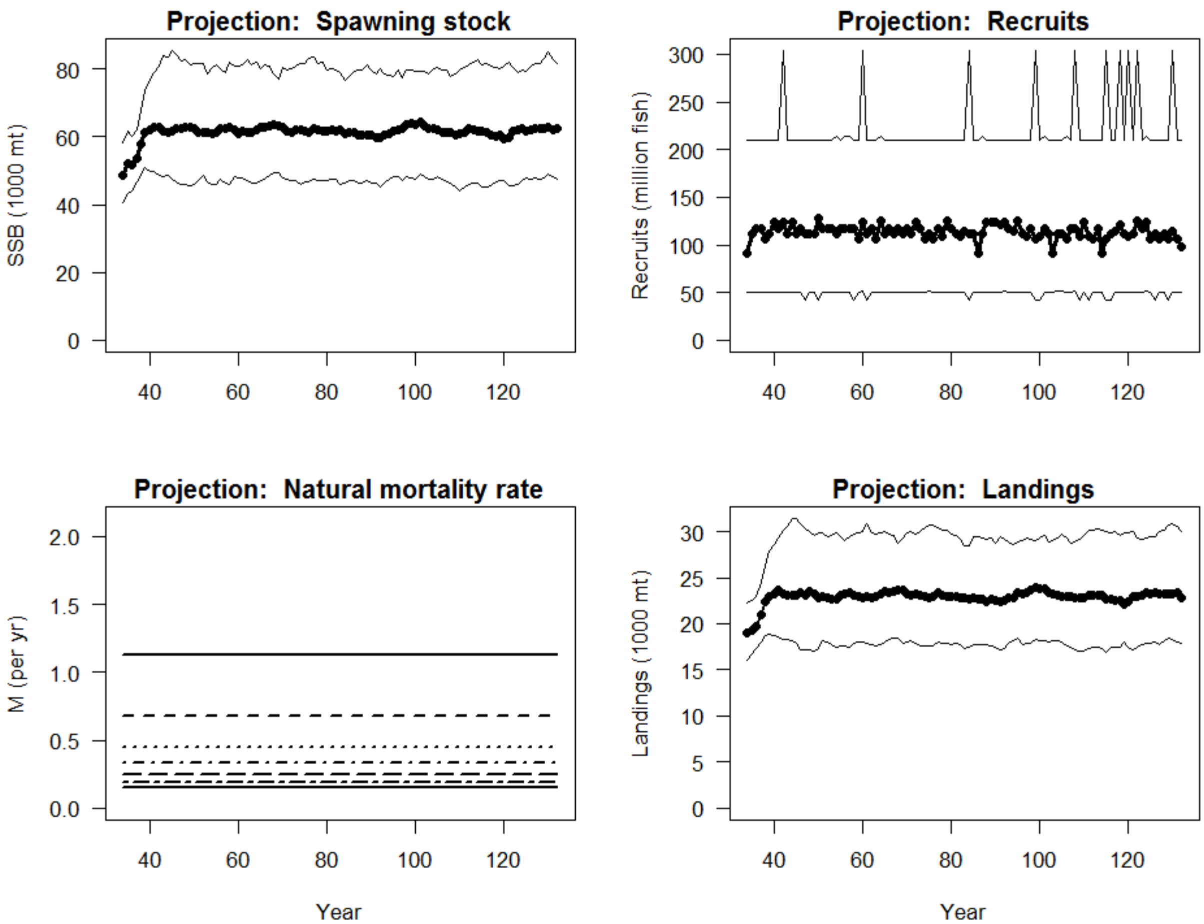


Figure 124. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for striped bass under scenario 4 for the VADER model (Atlantic menhaden at target *F*, all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

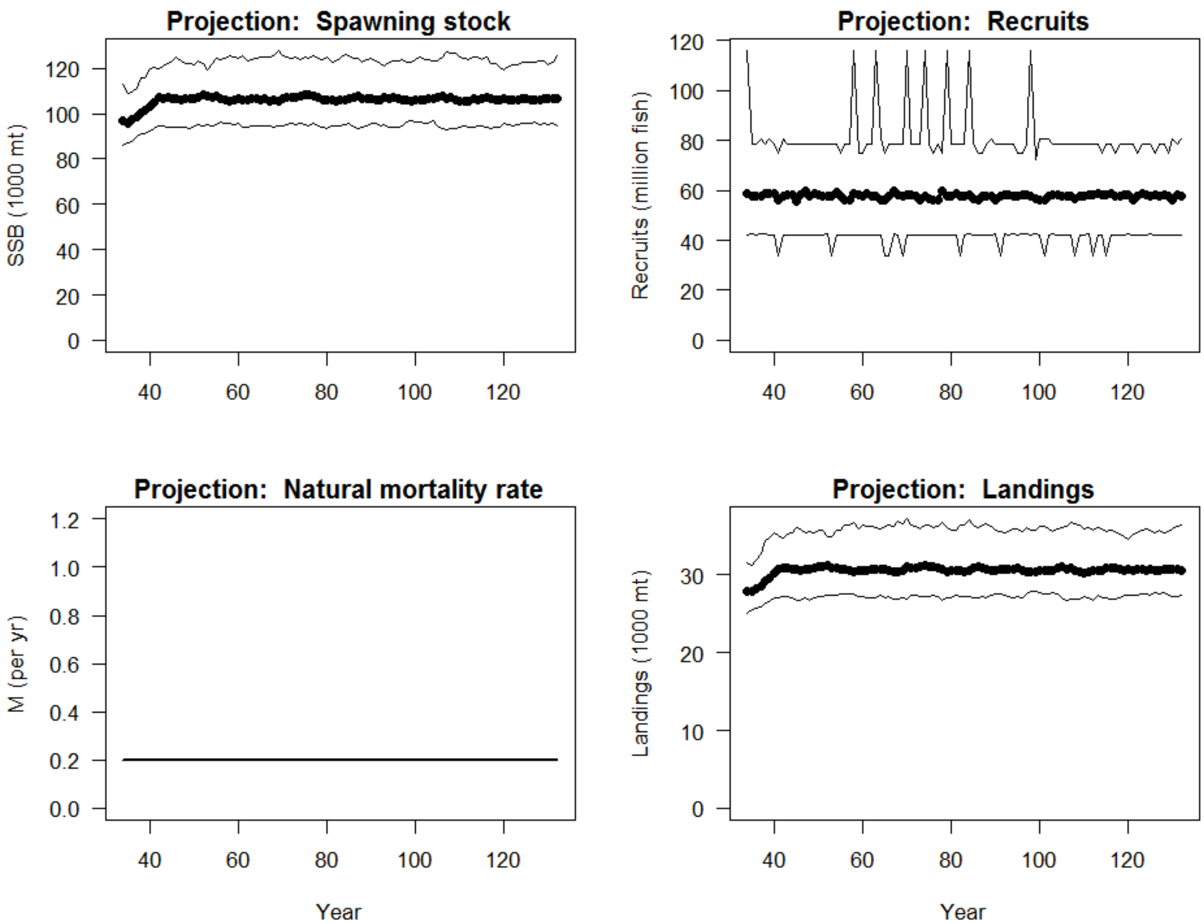


Figure 125. **Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for bluefish under scenario 4 for the VADER model (Atlantic menhaden at target *F*, all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.**

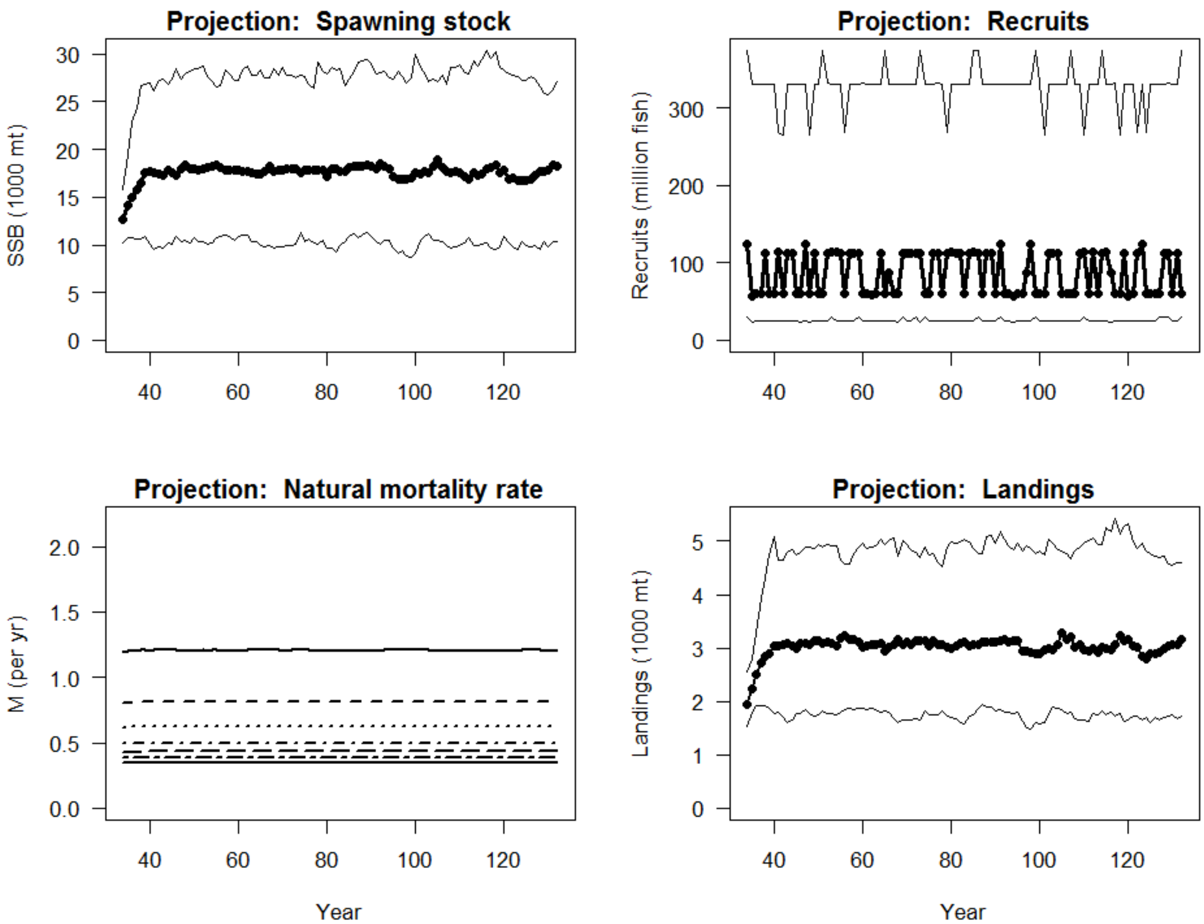


Figure 126. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for weakfish under scenario 4 for the VADER model (Atlantic menhaden at target *F,* all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

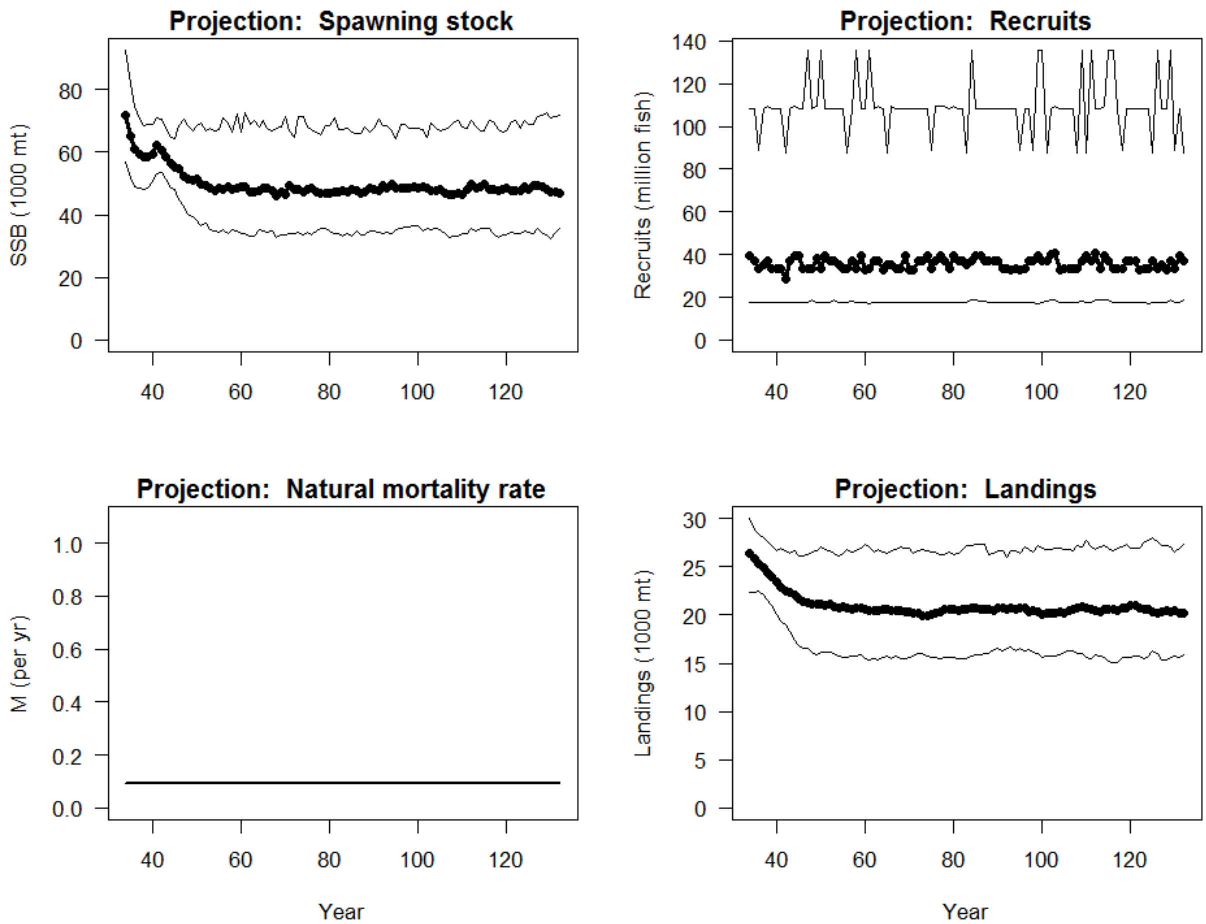


Figure 127. Projected spawning stock biomass, recruitment, natural mortality at-age, and landings for spiny dogfish under scenario 4 for the VADER model (Atlantic menhaden at target *F*, all other species at status quo *F*). For the SSB, recruitment, and landings plots the thin solid lines are the 5th and 95th percentiles, the solid line with circles is the median. For the *M* plot, the different lines represent *M*-at-age.

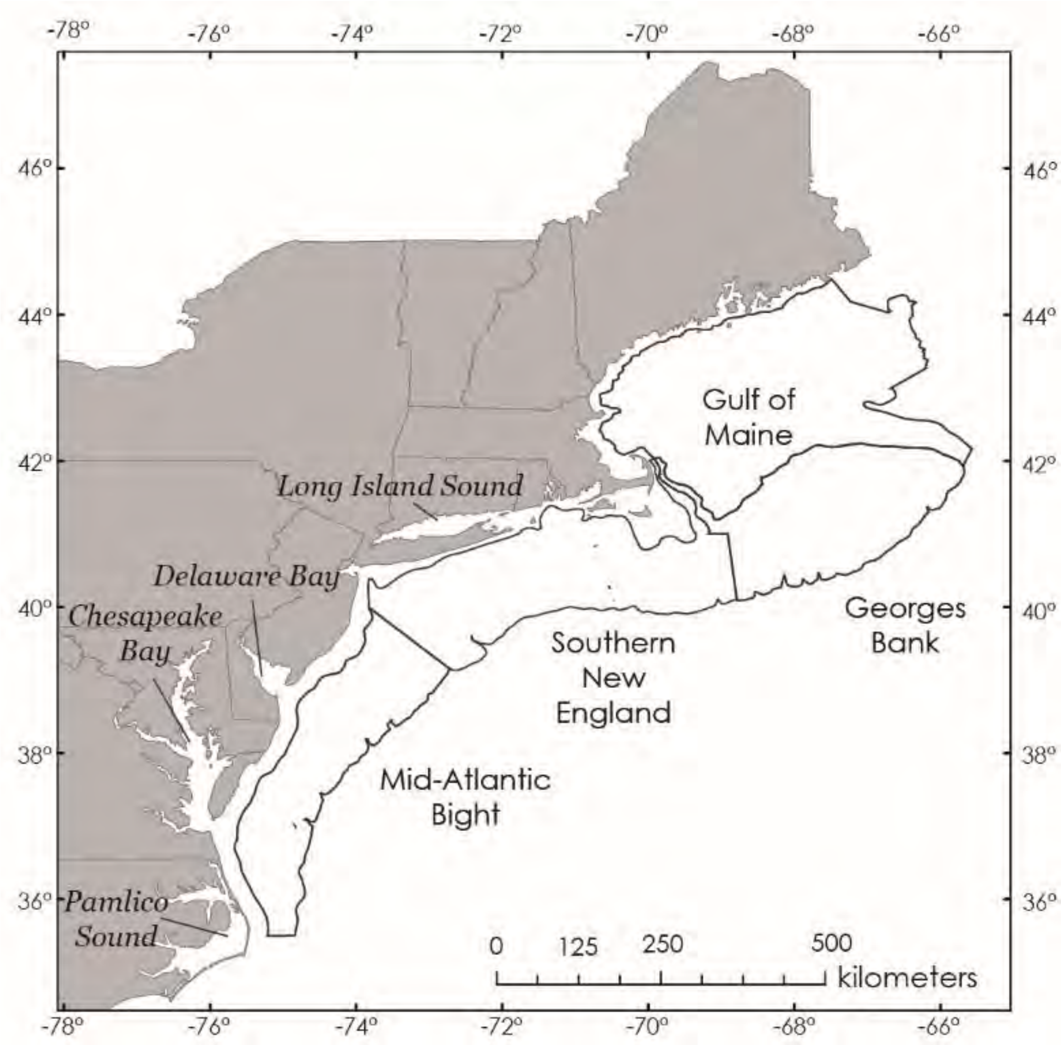


Figure 128. Map of the Northwest Atlantic Continental Shelf (NWACS) system, with major subregions and estuaries labeled. Figure modified from Link et al. (2006).

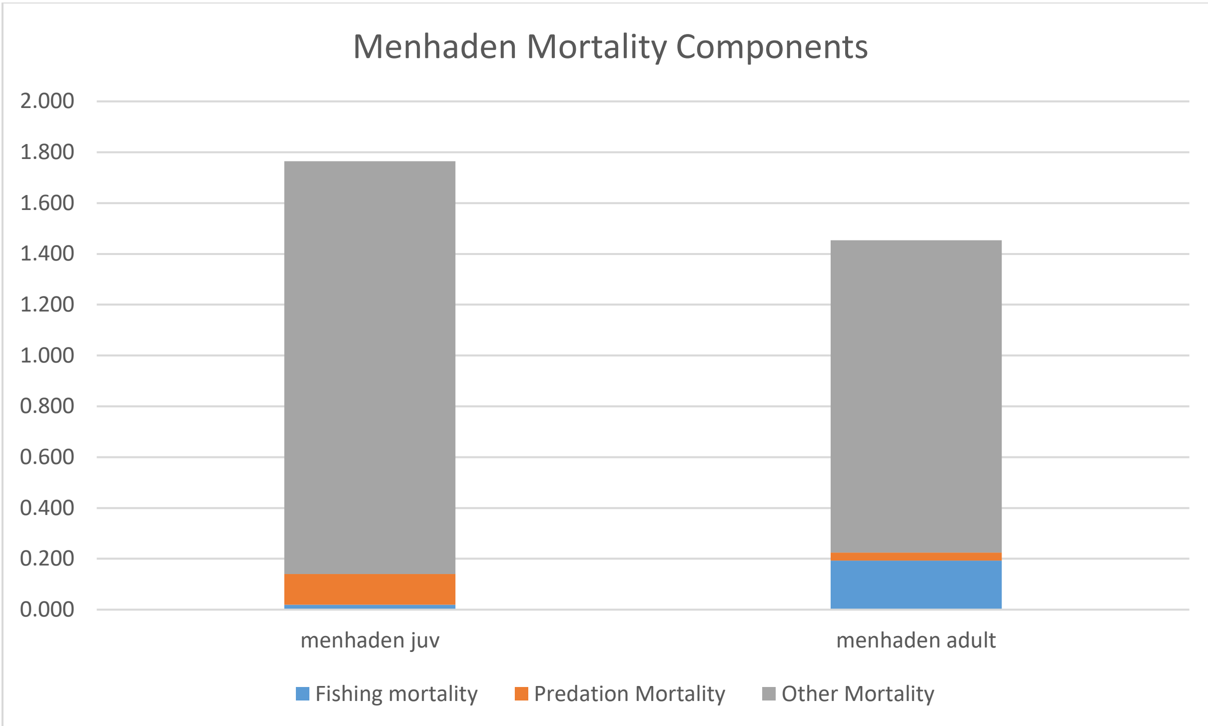


Figure 129. Ecopath Atlantic menhaden mortality components from the NWACS-MICE model.

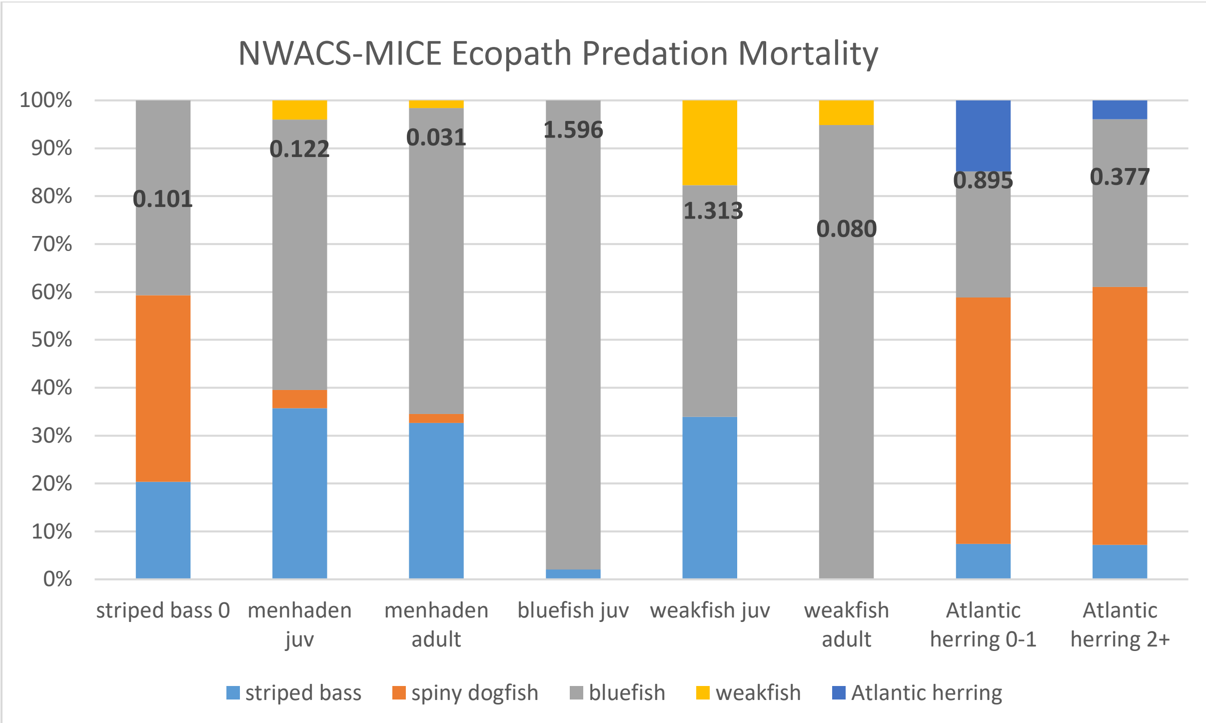


Figure 130. Predation mortality rates by species for the NWACS-MICE model. Each bar represents a prey item and the colors are the contributions by each predator. The values for each bar are the total predation mortality rates.

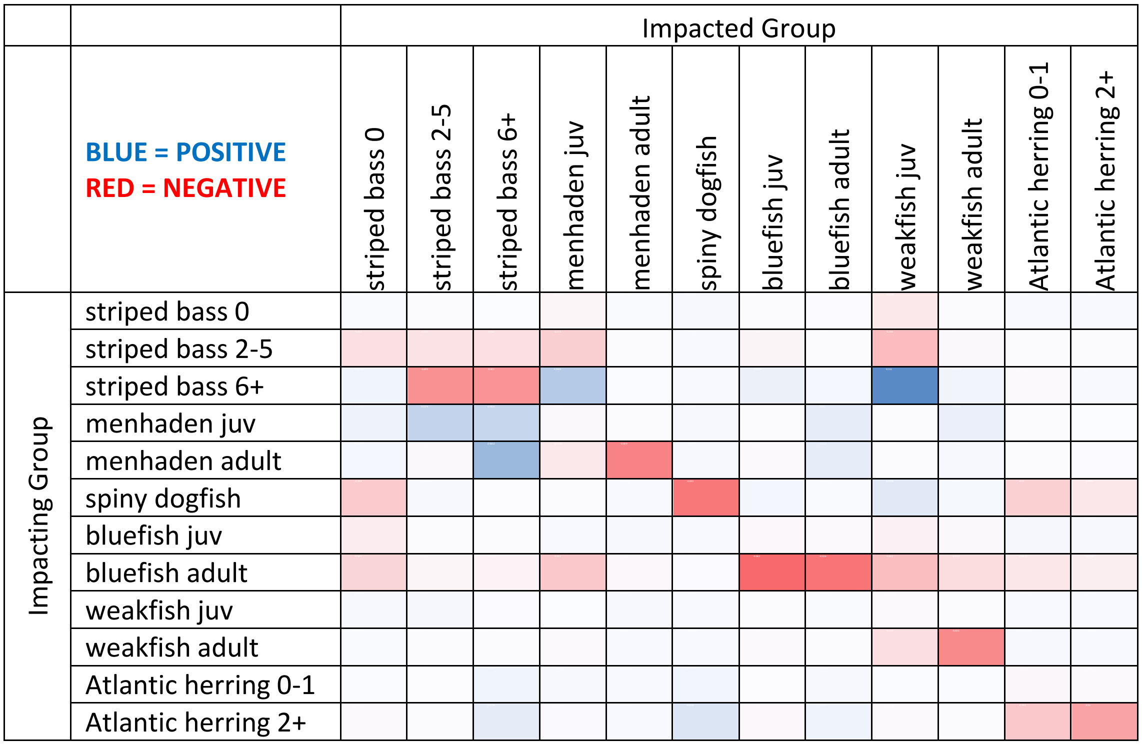


Figure 131. Mixed trophic impacts from the NWACS-MICE model.

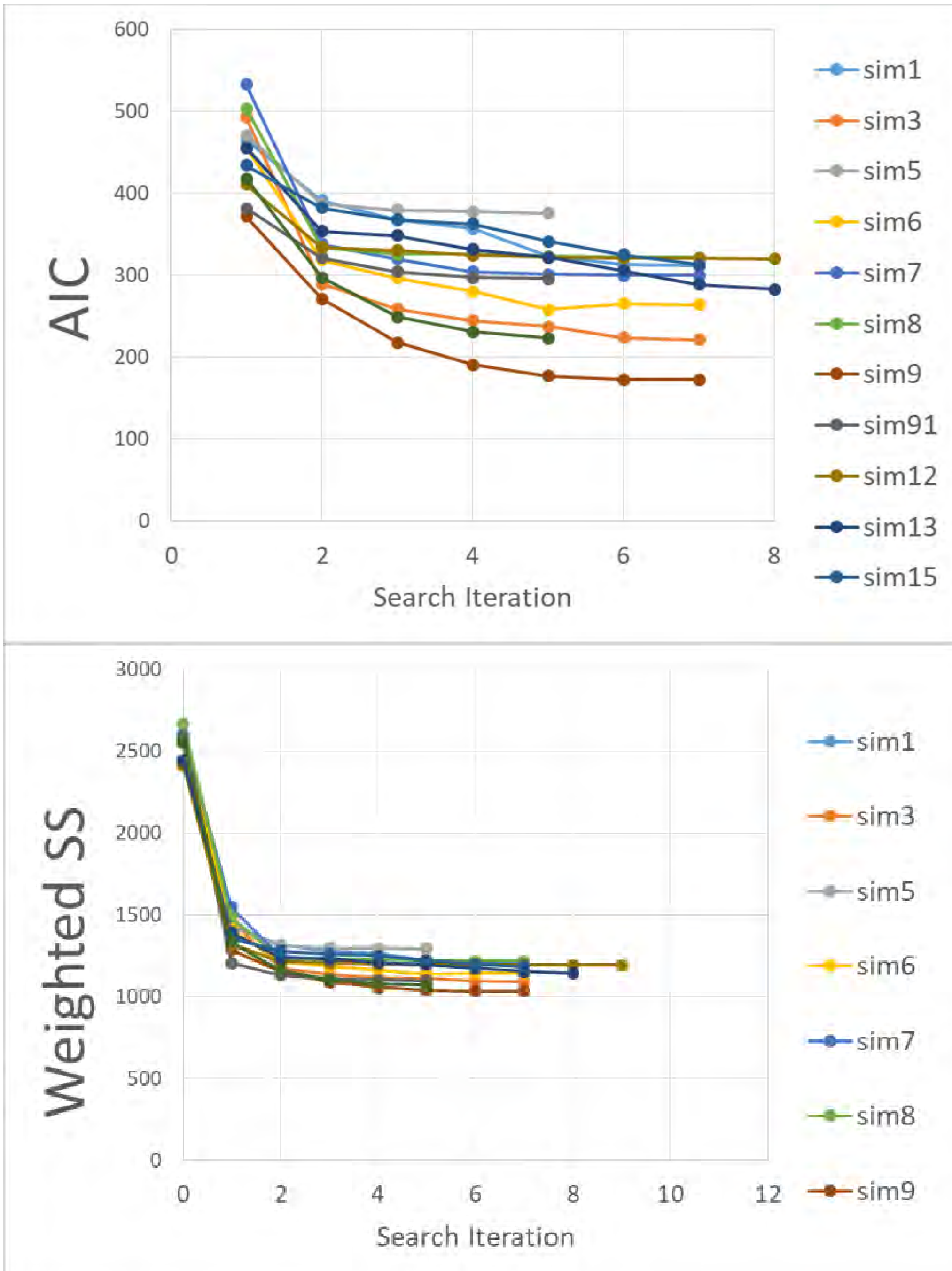


Figure 132. AIC (top) and weighted sums of squares (bottom) by simulation for repeated search iterations from the NWACS-MICE model.

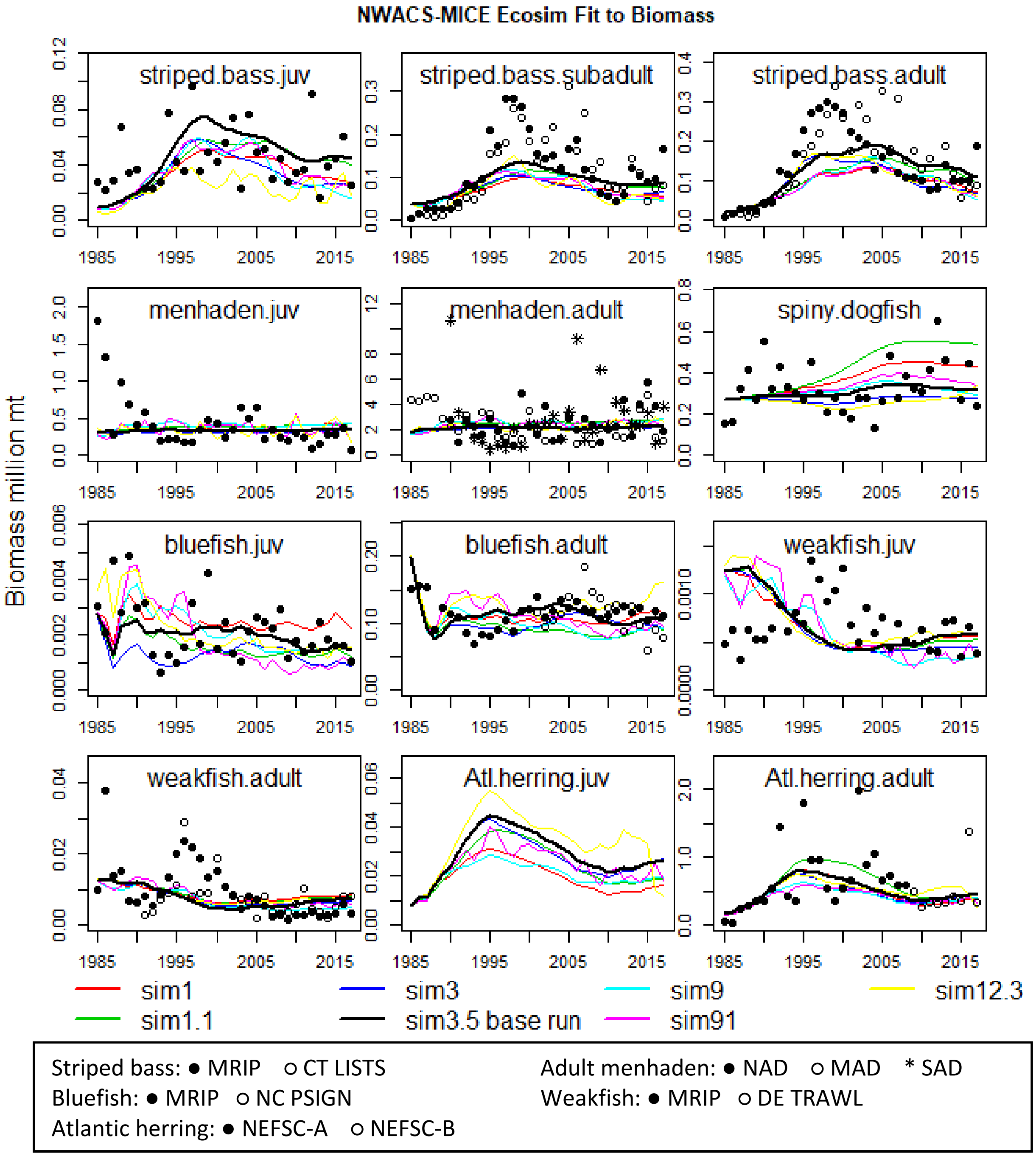


Figure 133. Ecosim fits to biomass from seven alternative runs of the NWACS-MICE model. Observed indices are listed in Table 34.

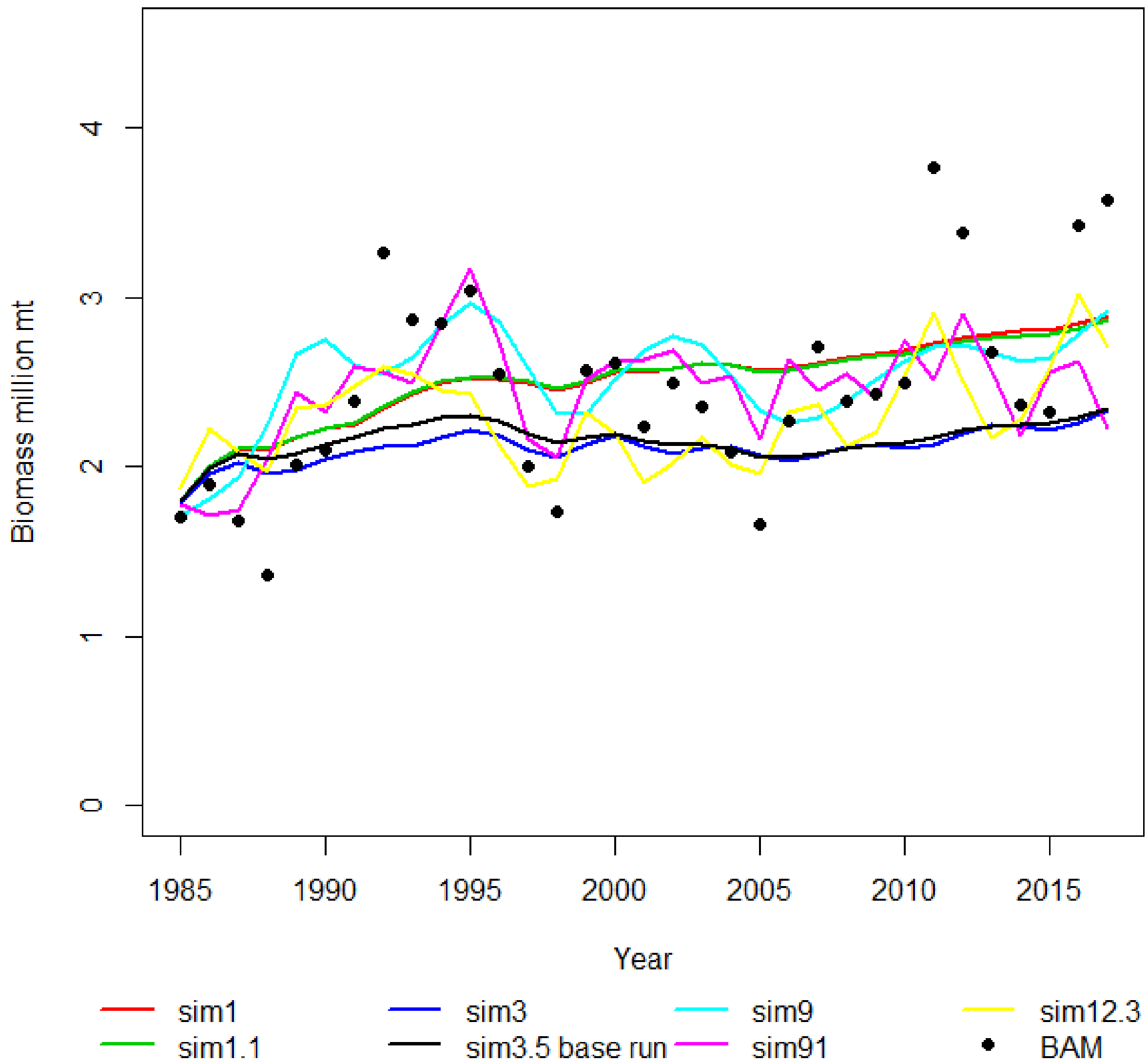


Figure 134. Atlantic menhaden age 1+ biomass predicted by Ecosim from the NWACS-MICE model plotted with age 1+ biomass from the single-species model (BAM).

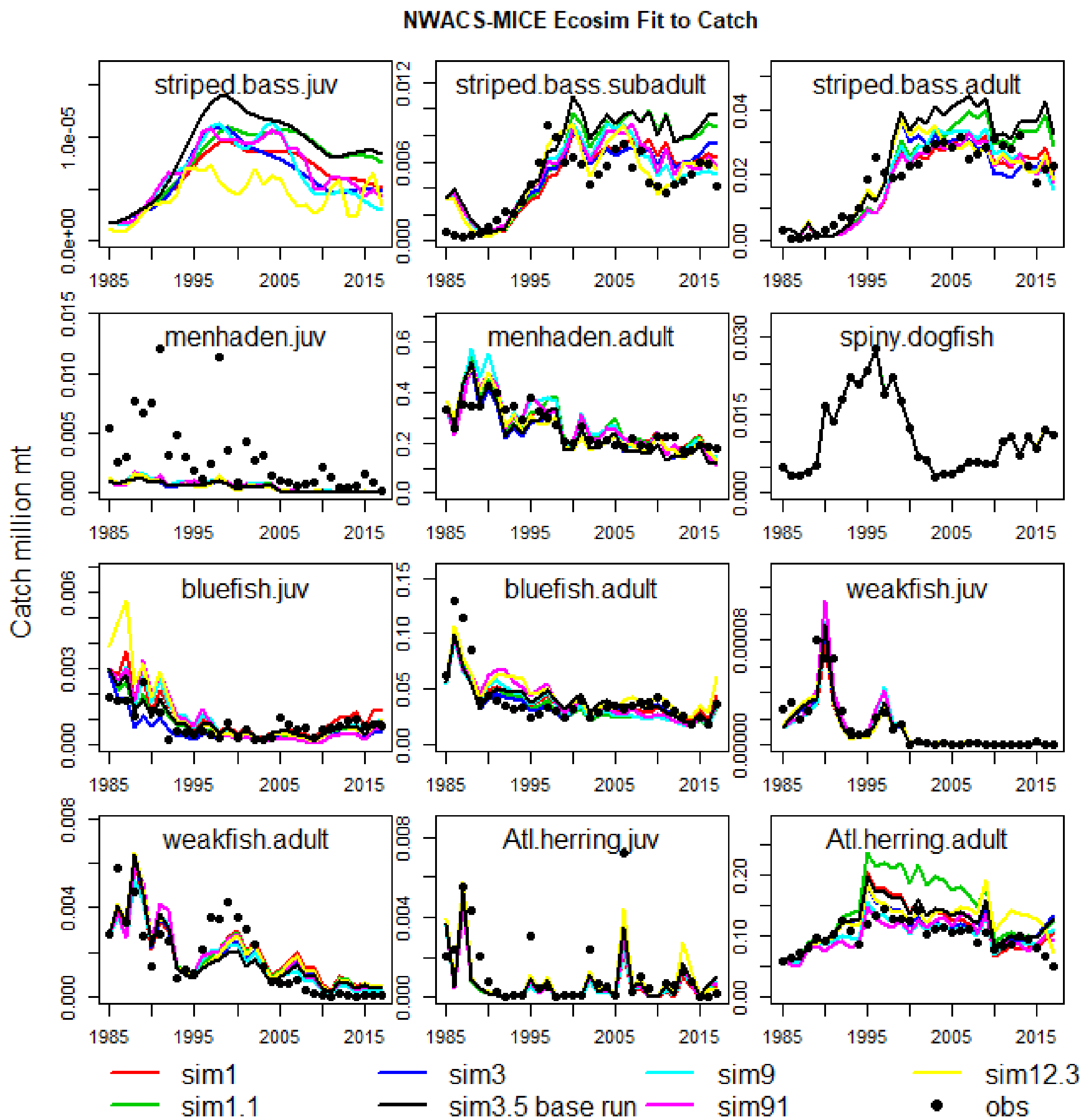


Figure 135. Ecosim fits to observed catch from seven alternative runs of the NWACS-MICE model. Observed catch series are listed in Table 34.

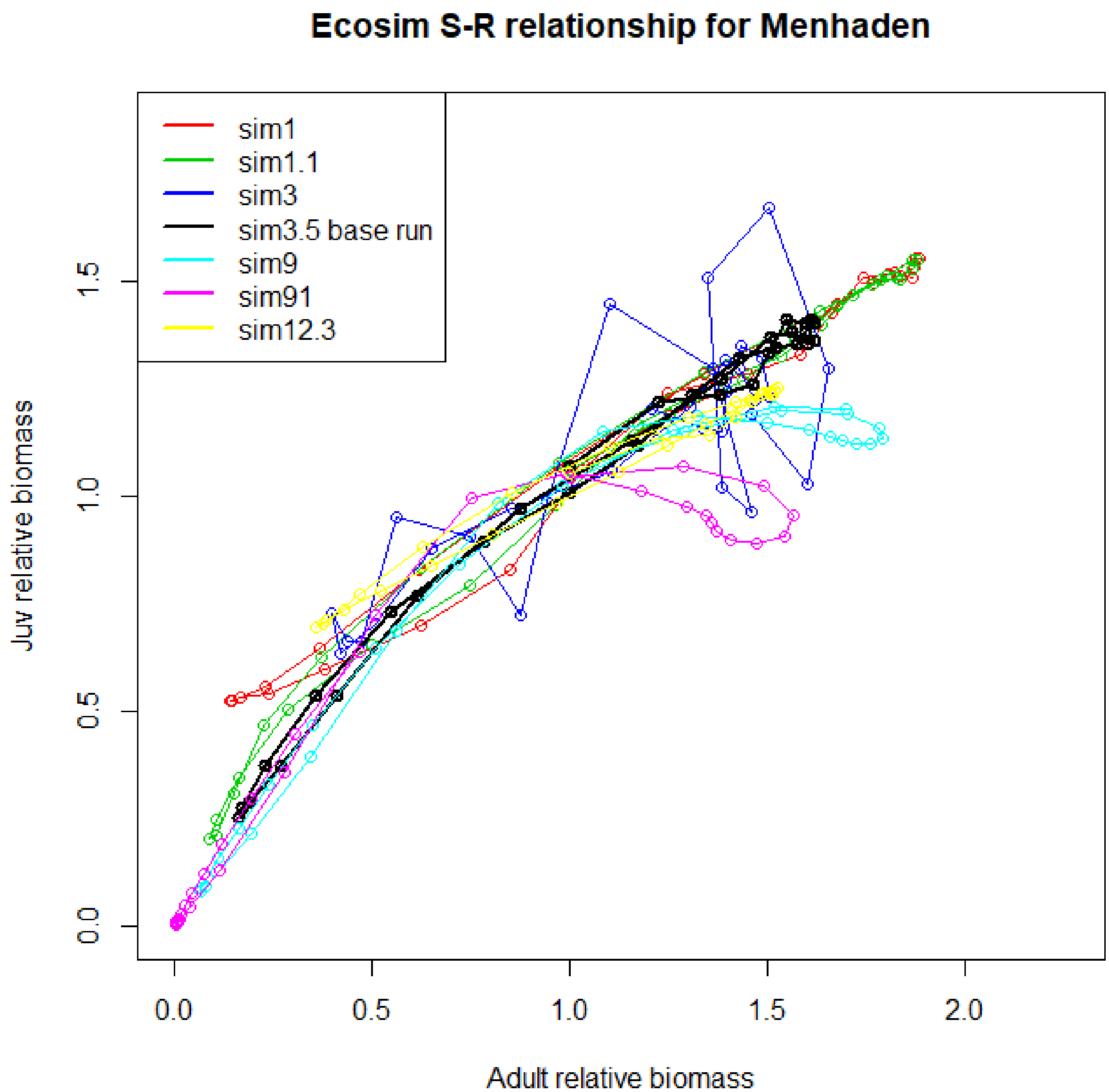


Figure 136. Atlantic menhaden stock-recruit plot from alternative runs of the NWACS-MICE model.

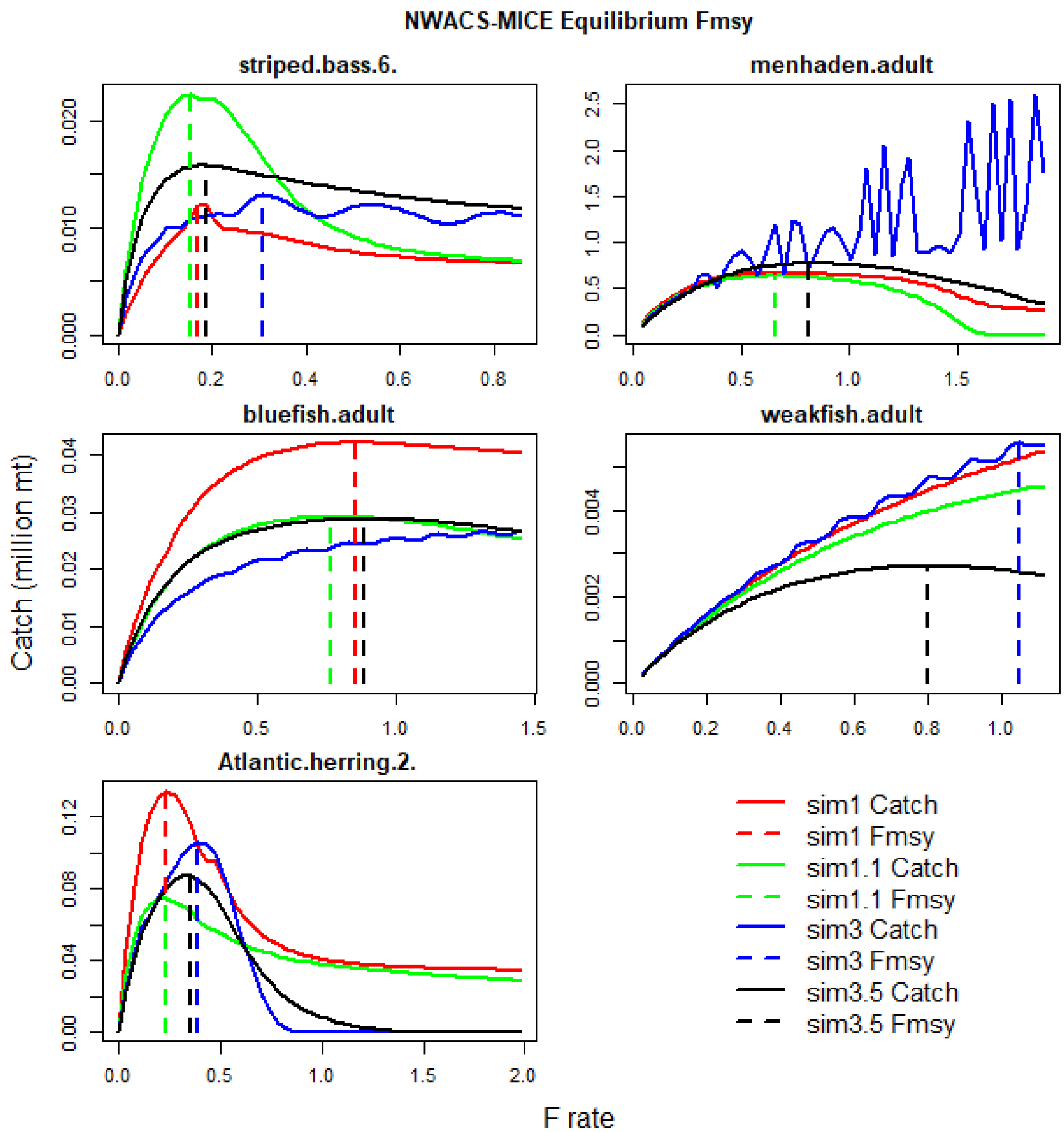


Figure 137. Equilibrium MSY curves from four alternative Ecosim runs (non-stationary system) from the NWACS-MICE model. The solid lines show long term yield over a range of *F*. The dotted lines indicate the *F* at maximum yield, F_{MSY}. Absence of a dotted line indicates that a F_{MSY} value was not found.

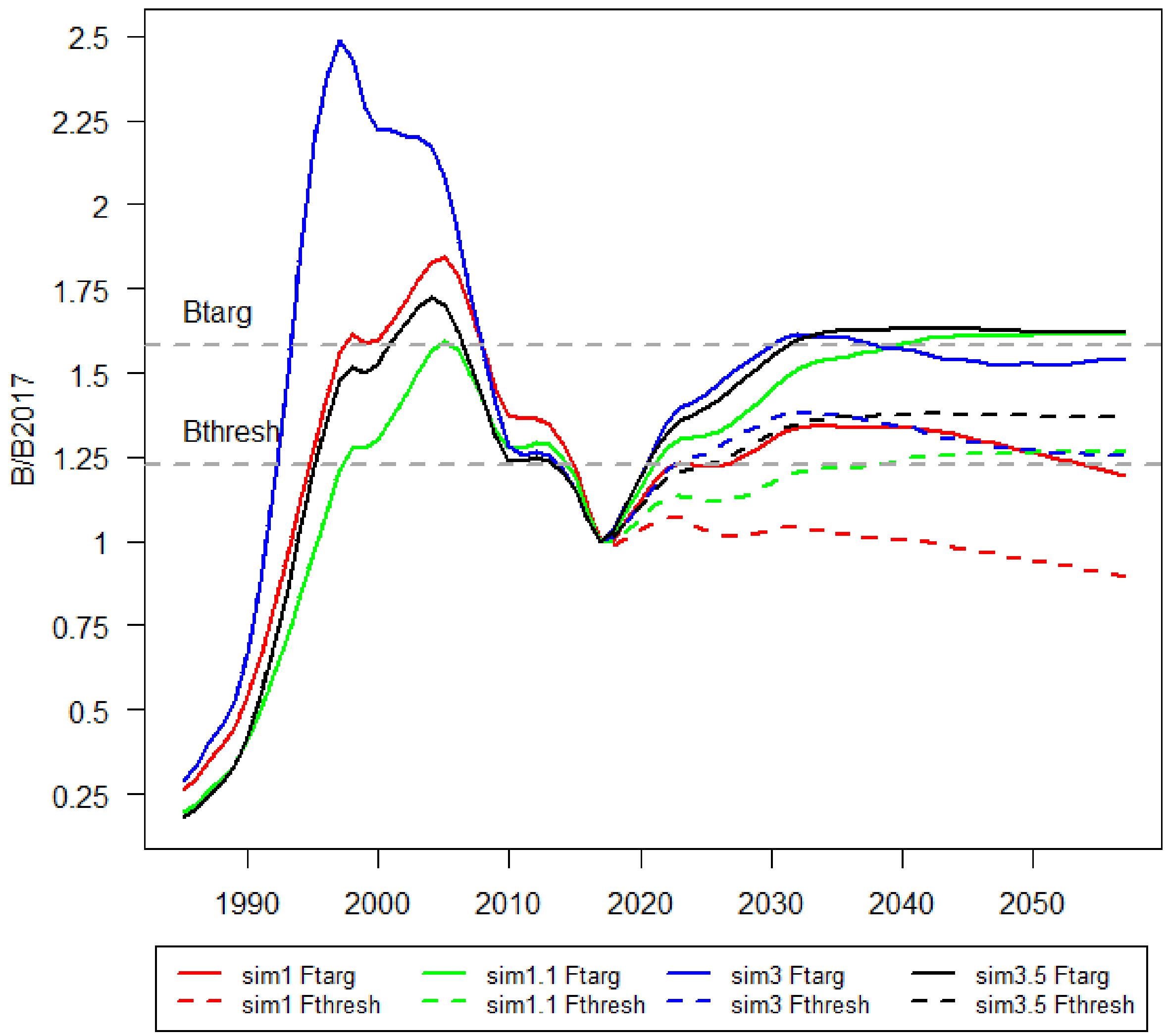


Figure 138. Striped bass age 6+ biomass (scaled to 2017) projected under target and threshold fishing mortality rates from the NWACS-MICE model. All other species were held constant at status quo *F* (*F*$_{2017}$).

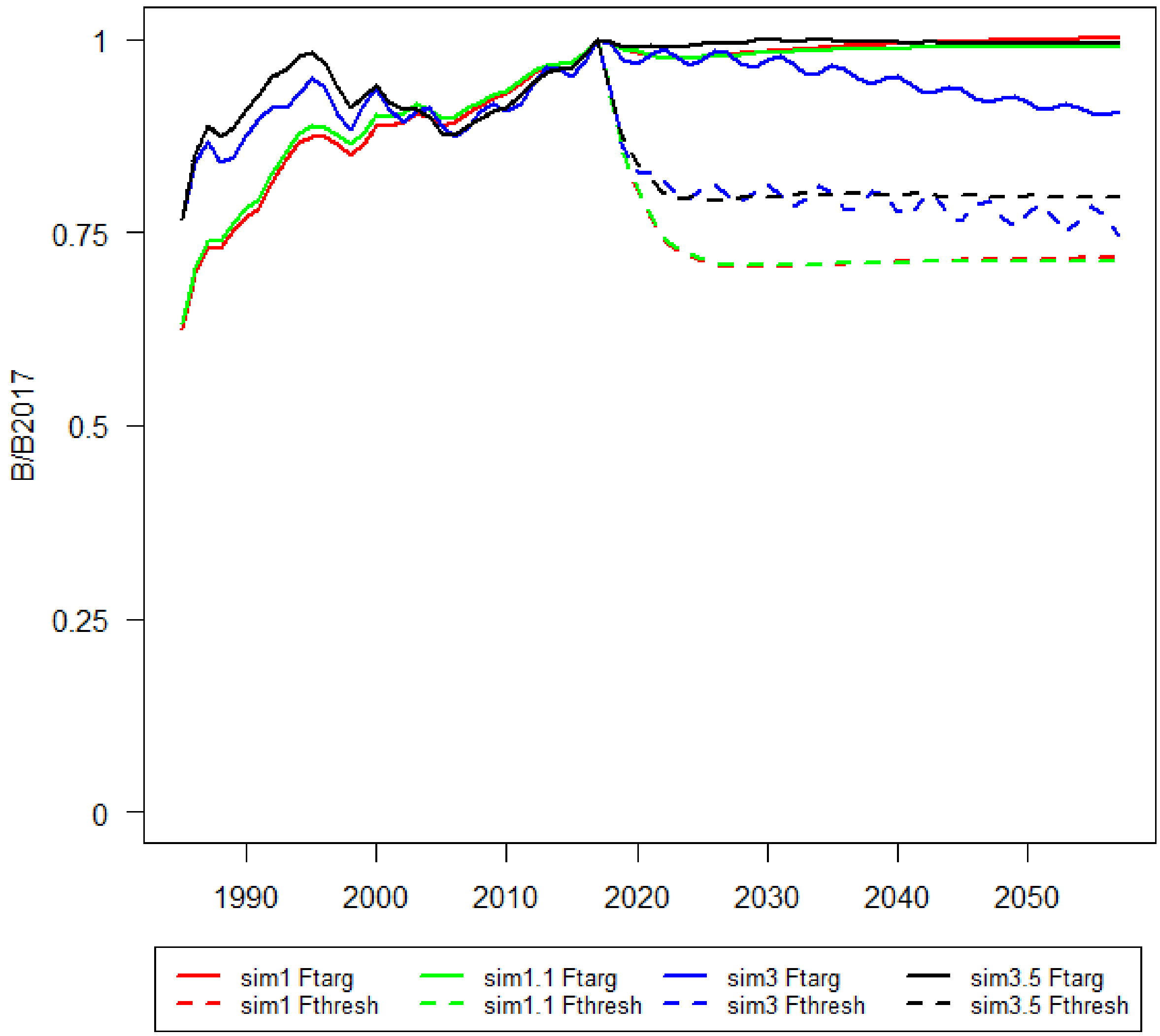


Figure 139. Atlantic menhaden age 1+ biomass projected under target and threshold fishing mortality rates from the NWACS-MICE model. All other species were held constant at status quo *F* (F_{2017}).

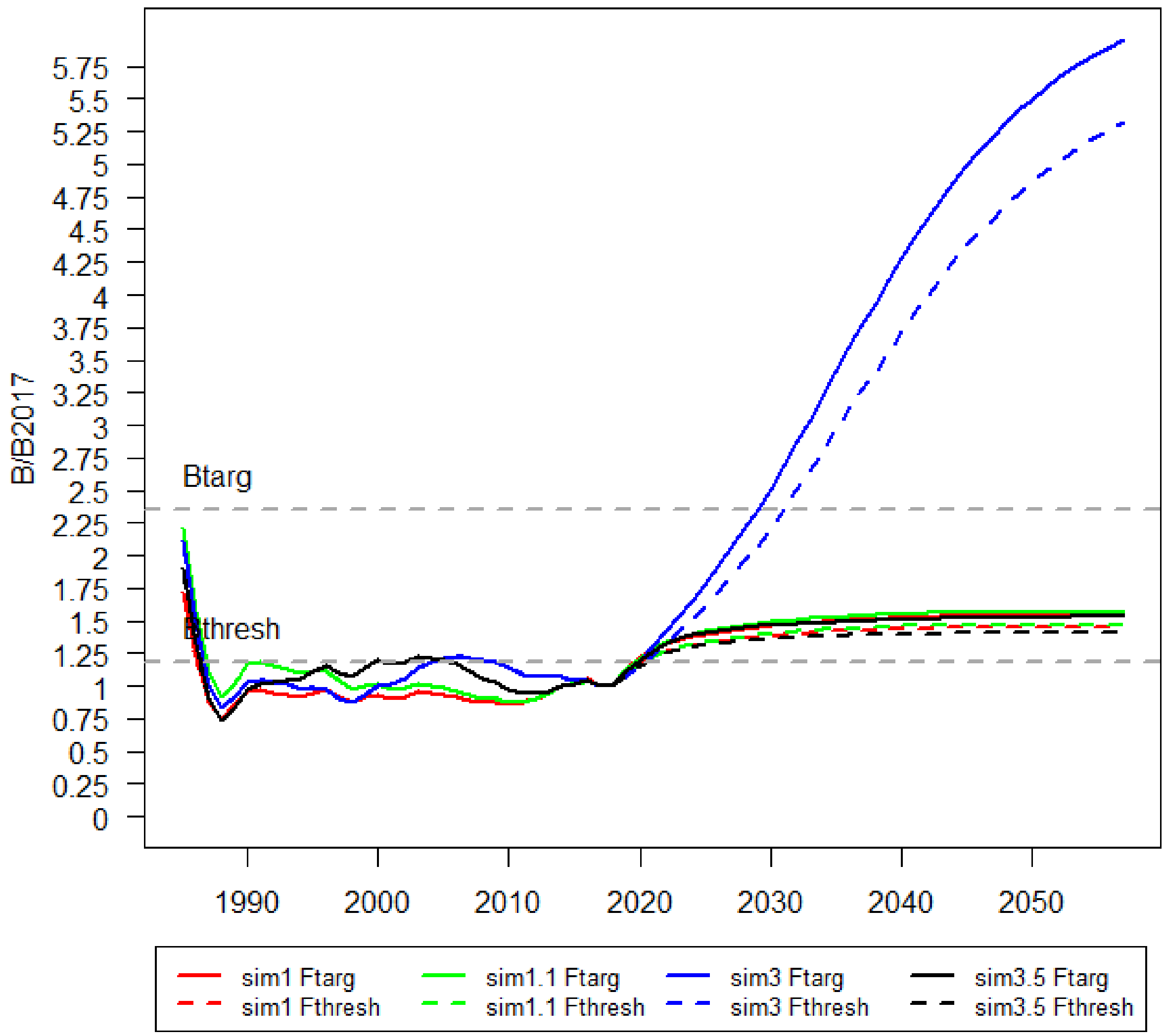


Figure 140. Bluefish age 1+ biomass projected under target and threshold fishing mortality rates from the NWACS-MICE model. All other species were held constant at status quo *F* (F_{2017}).

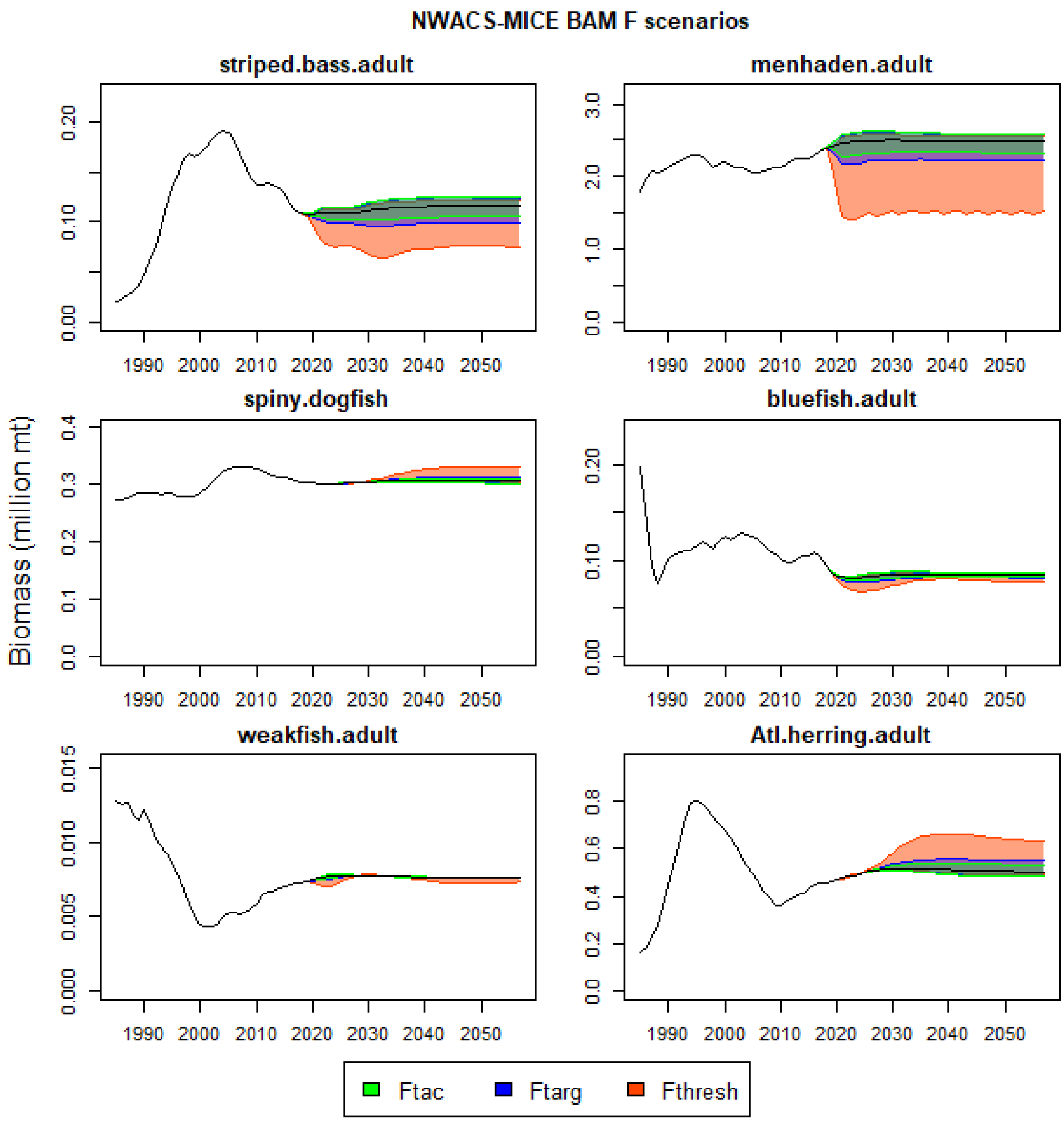


Figure 141. Biomass trajectories from the NWACS-MICE model under the BAM *F* scenarios: Ftac= *F* associated with current TAC; Ftarg = single-species *F* target; Fthresh= single-species *F* threshold. Shaded regions show the full range of biomass predicted under each scenario. In the BAM *F* scenarios, all other species were held constant at their status quo *F* (F_{2017}).

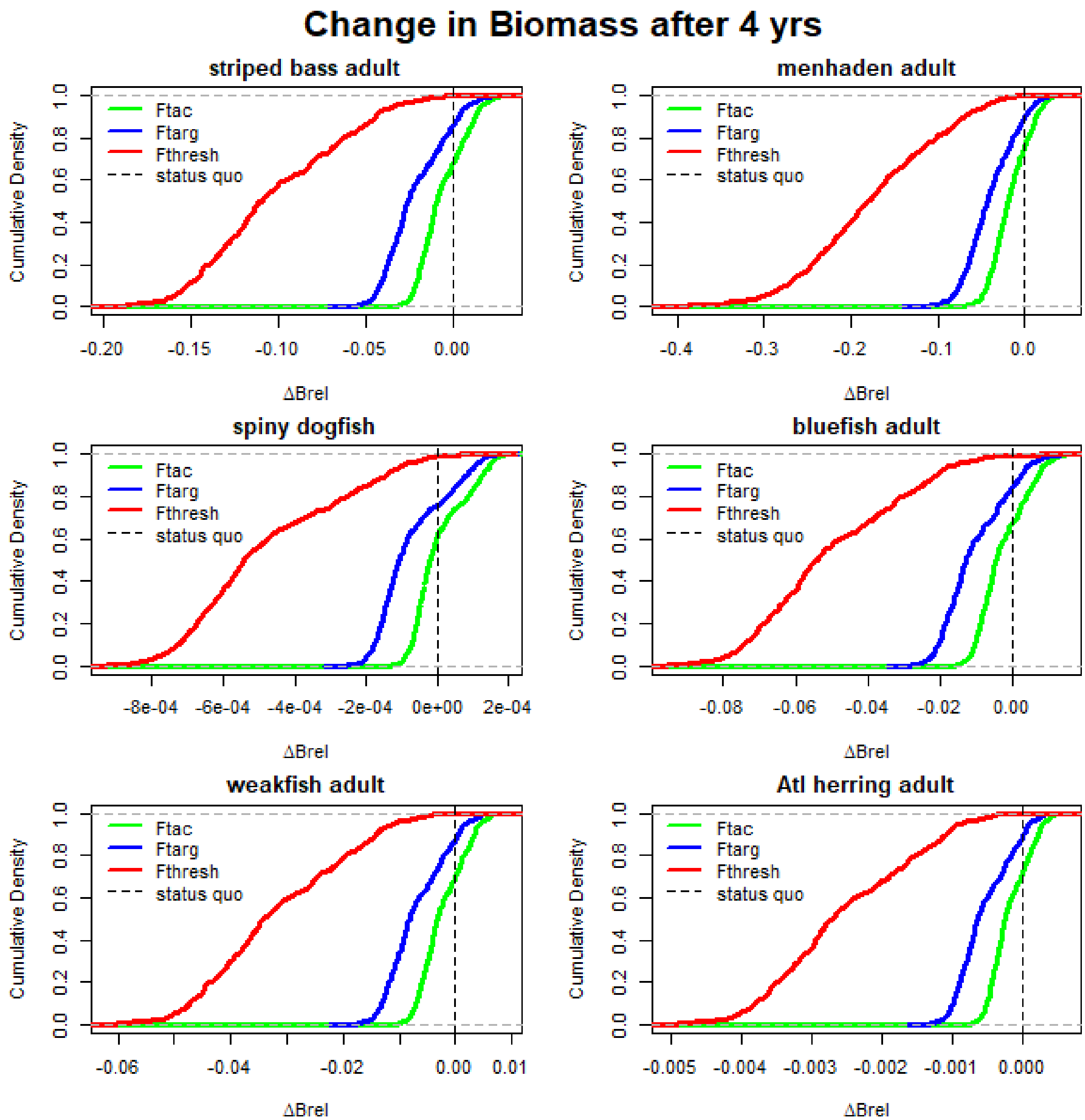


Figure 142. Cumulative density plots of the change in biomass relative to 2017 biomass (ΔB_{REL}) from the NWACS-MICE model for each species after four years of fishing menhaden at current TAC, target, and threshold fishing mortalities (from BAM). Each line shows the proportion of trials (out of 500) that cause biomass to decline below the value on the x-axis. Analysis was conducted using sim3.5.

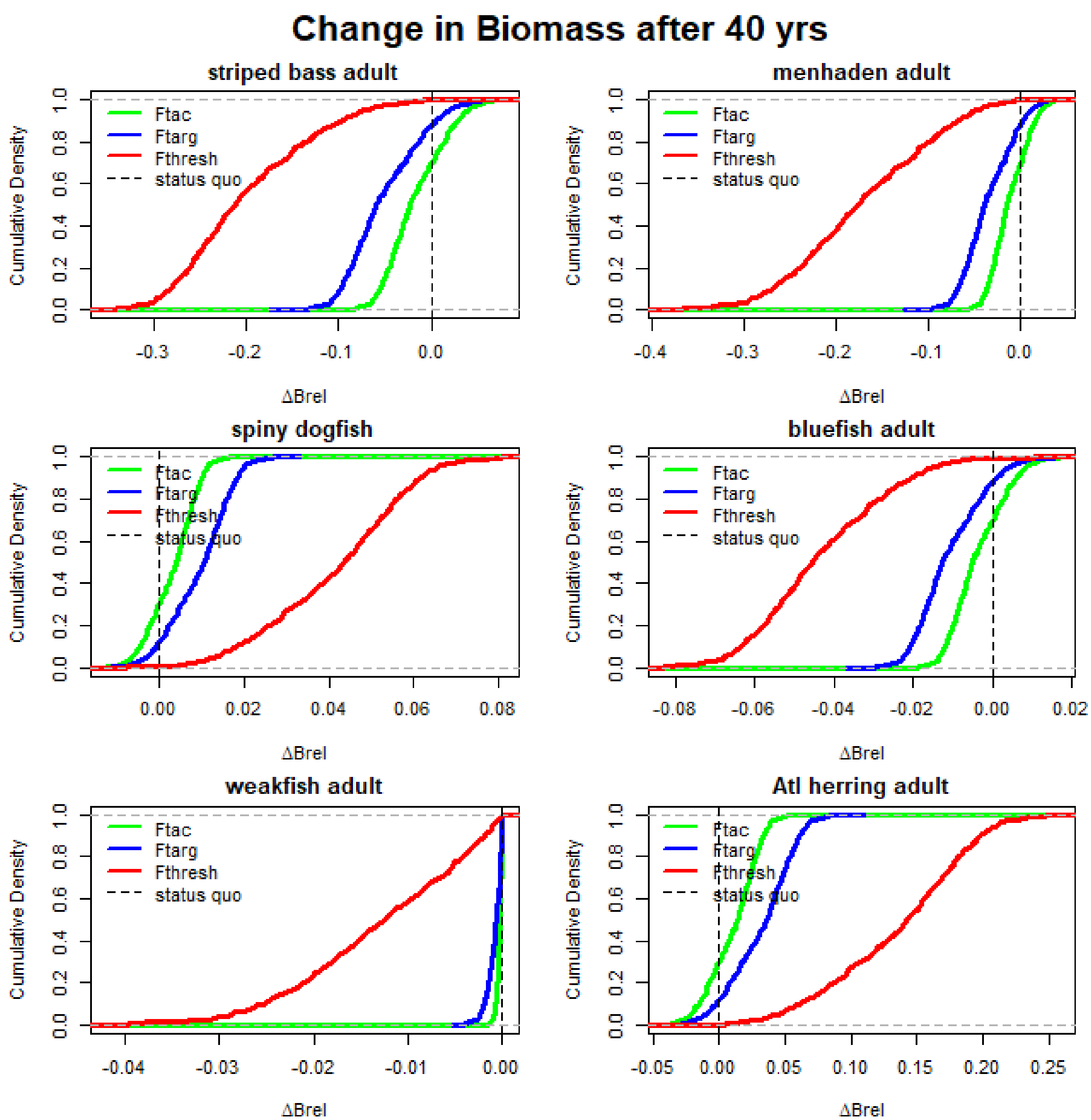


Figure 143. **Cumulative density plots of the change in biomass relative to 2017 biomass (*ΔB*REL) from the NWACS-MICE model for each species after forty years of fishing Atlantic menhaden at current TAC, target, and threshold fishing mortalities (from BAM). Each line shows the proportion of trials (out of 500) that cause biomass to decline below the value on the x-axis. Analysis was conducted using sim3.5.**

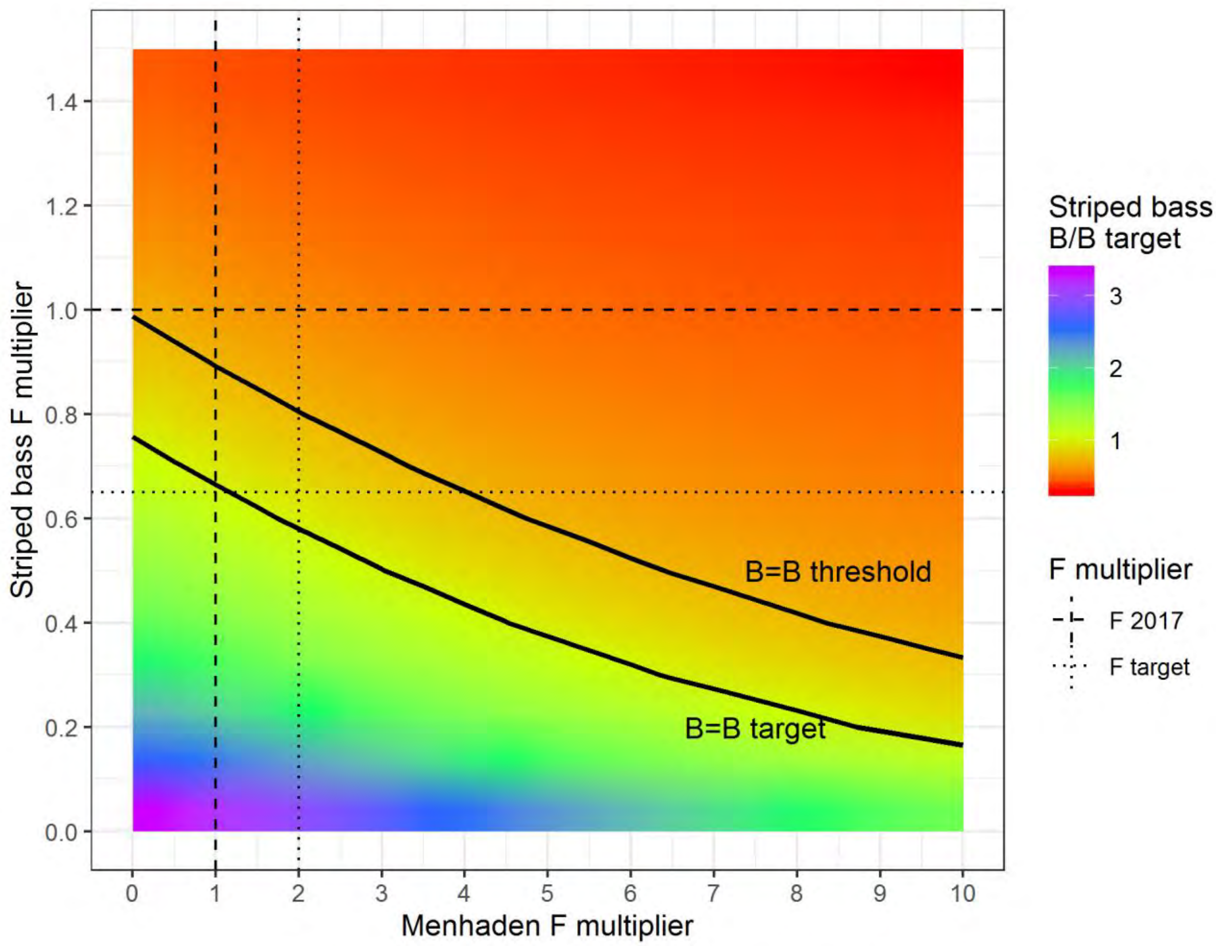


Figure 144. Striped bass age 6+ biomass ratio (*B/B*TARGET) in the terminal year of the NWACS-MICE projections as a function of fishing mortality on both Atlantic menhaden and striped bass. The solid black lines represent the contours where striped bass *B*=*B*THRESHOLD and *B*=*B*TARGET. The dashed lines highlight specific *F* scenarios where *F* is equivalent to the *F* in 2017 or the single-species *F* target for each species.

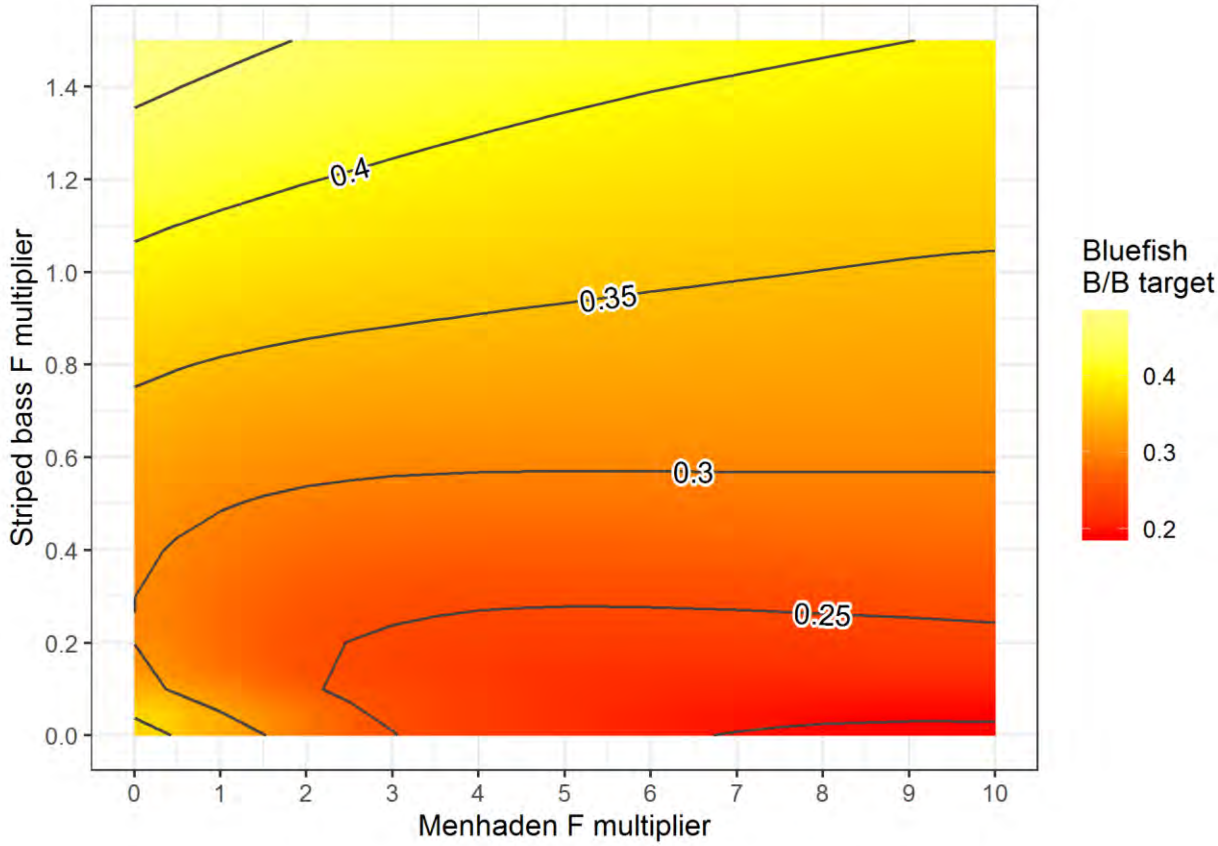


Figure 145. **Bluefish age 1+ biomass ratio (*B/B*$_{TARGET}$) in the terminal year of the NWACS-MICE projections as a function of fishing mortality on both Atlantic menhaden and striped bass. In these projections, bluefish *F* was held constant at 2017 value.**

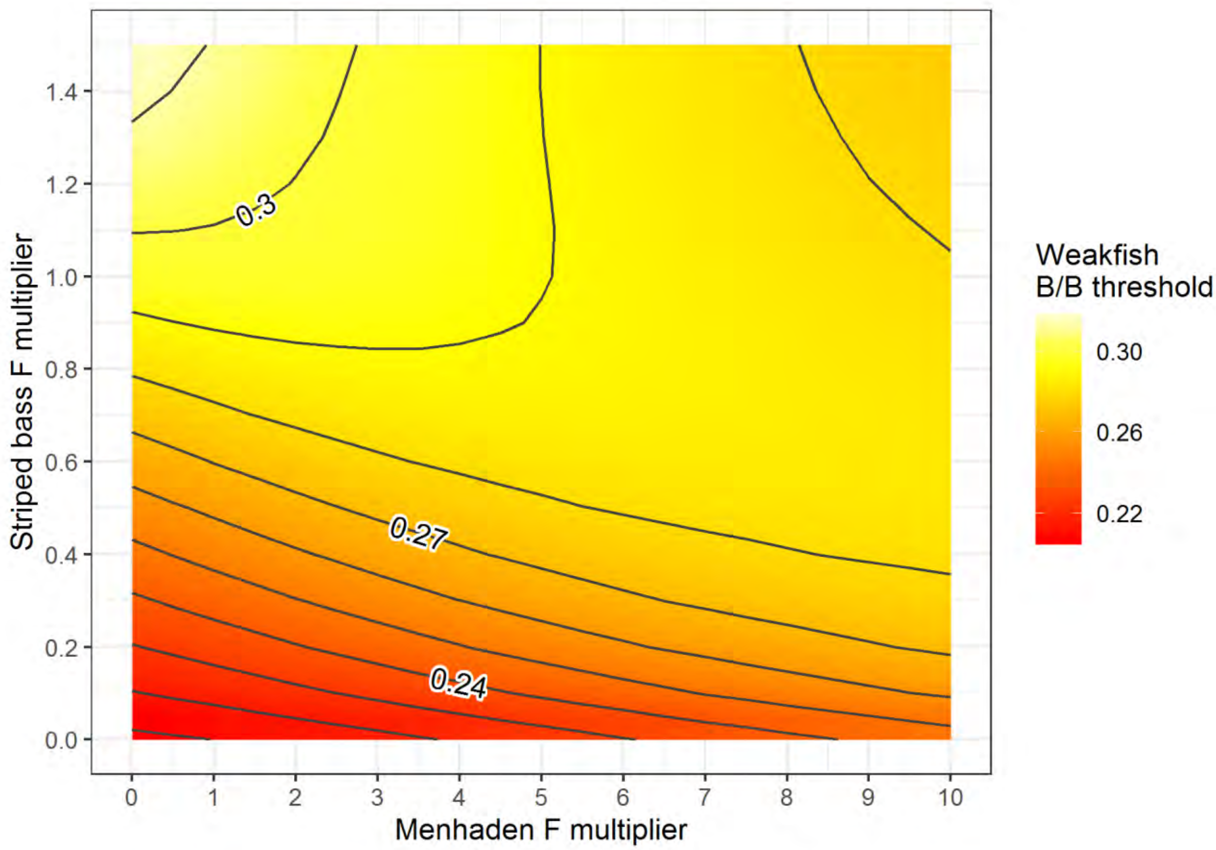


Figure 146. Weakfish age 1+ biomass ratio (*B/B*$_{THRESHOLD}$) in the terminal year of the NWACS-MICE projections as a function of fishing mortality on both Atlantic menhaden and striped bass. In these projections, weakfish *F* was held constant at the 2017 value.

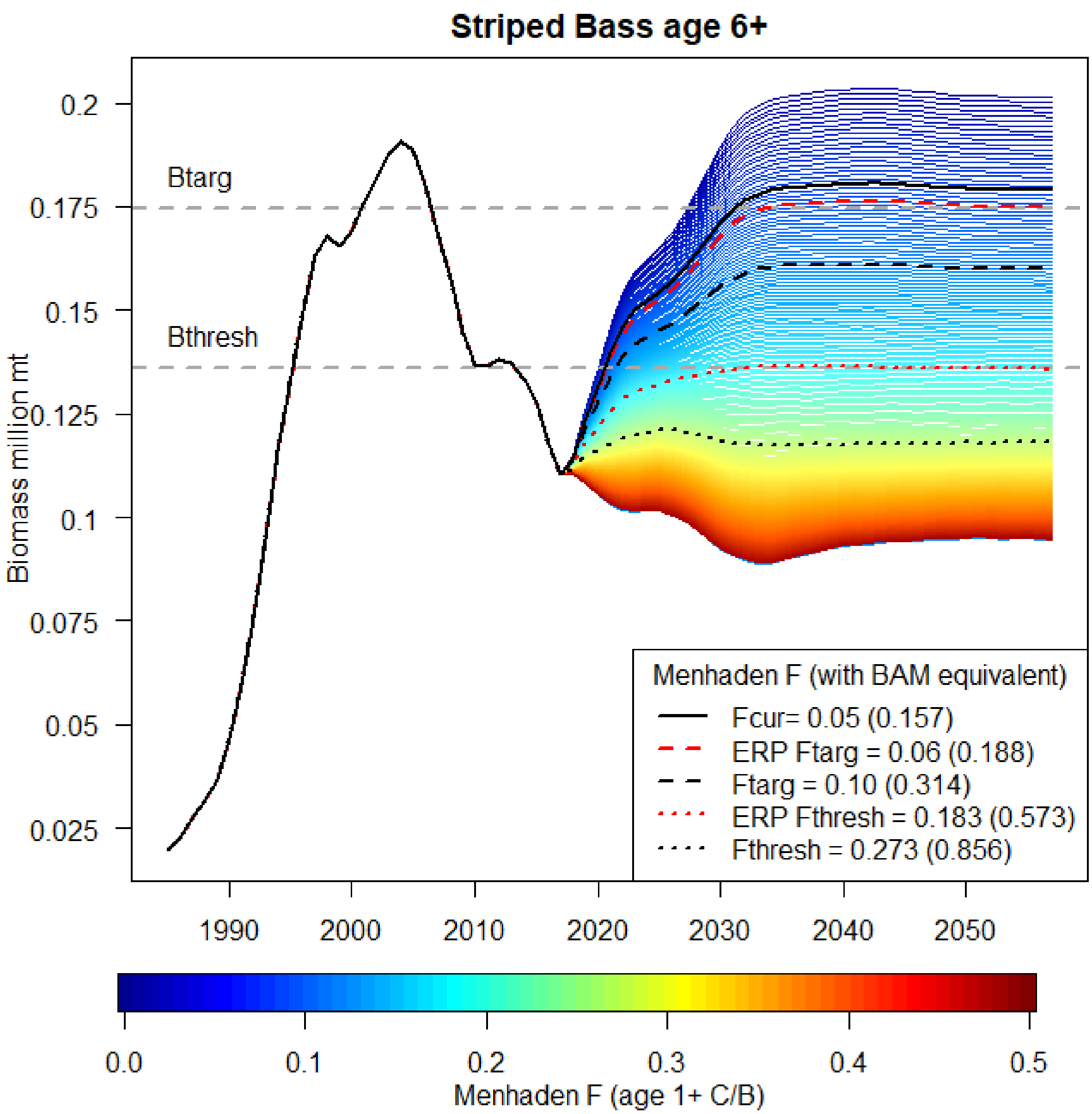


Figure 147. Striped bass age 6+ biomass from the NWACS-MICE model, projected under striped bass *F* = F_{TARGET} from 2018-2057 over a range of Atlantic menhaden *F*. All other species were held constant at their status quo *F*.

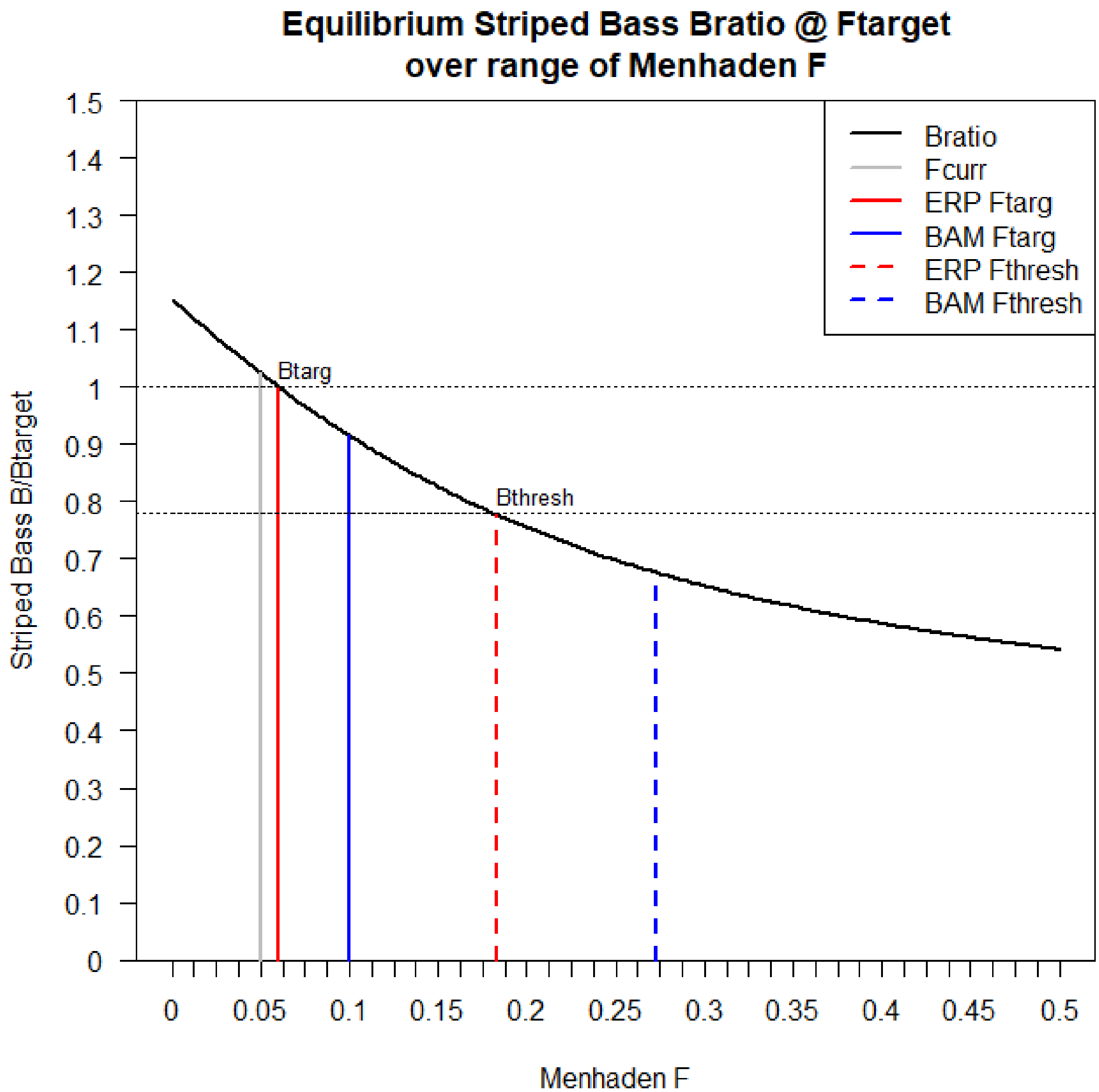


Figure 148. Terminal year biomass ratio (*B/B*$_{TARGET}$) from the NWACS-MICE model for age 6+ striped bass over a range of Atlantic menhaden *F* with striped bass fished at their *F* target. Vertical solid and dotted lines indicate the BAM single-species target and threshold *F* as well as the current *F* and the proposed ERP target and threshold *F* for Atlantic menhaden.

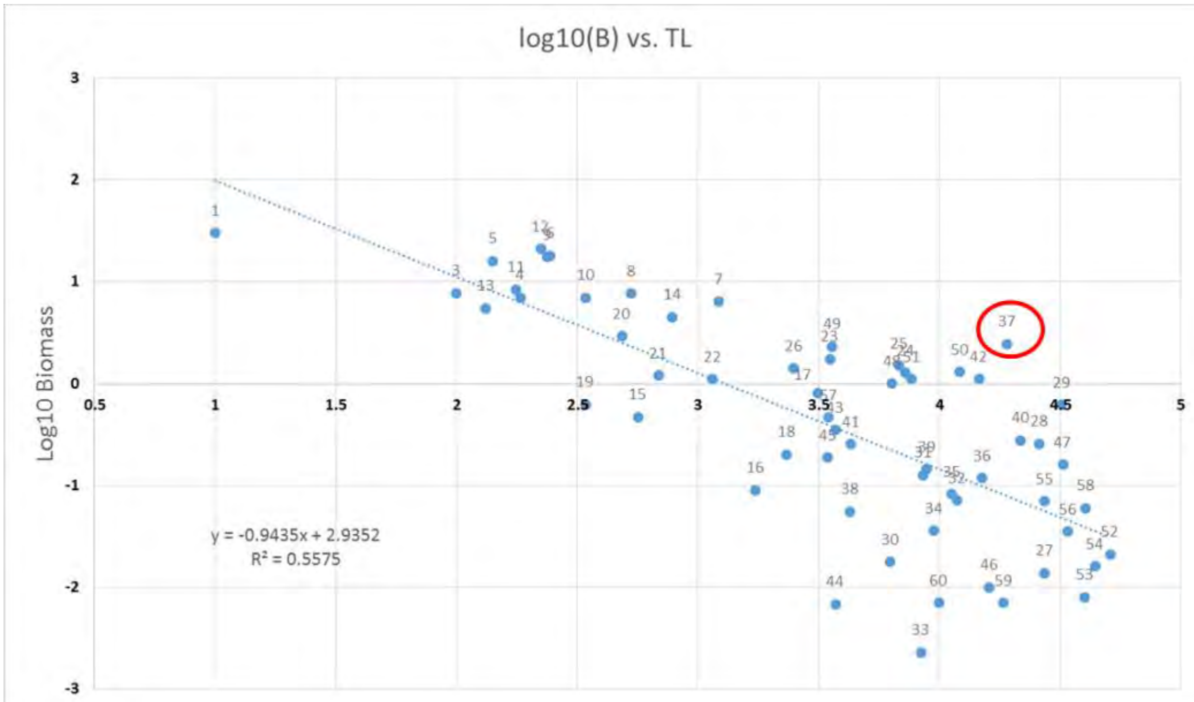


Figure 149. Relationship between log (base 10) biomass and trophic level (TL) for all trophic groups in the NWACS-FULL model before balancing. The decline in biomass is expected (Link 2010), but the red circle highlights large spiny dogfish as an outlier which was used as justification for reducing their biomass in the model.

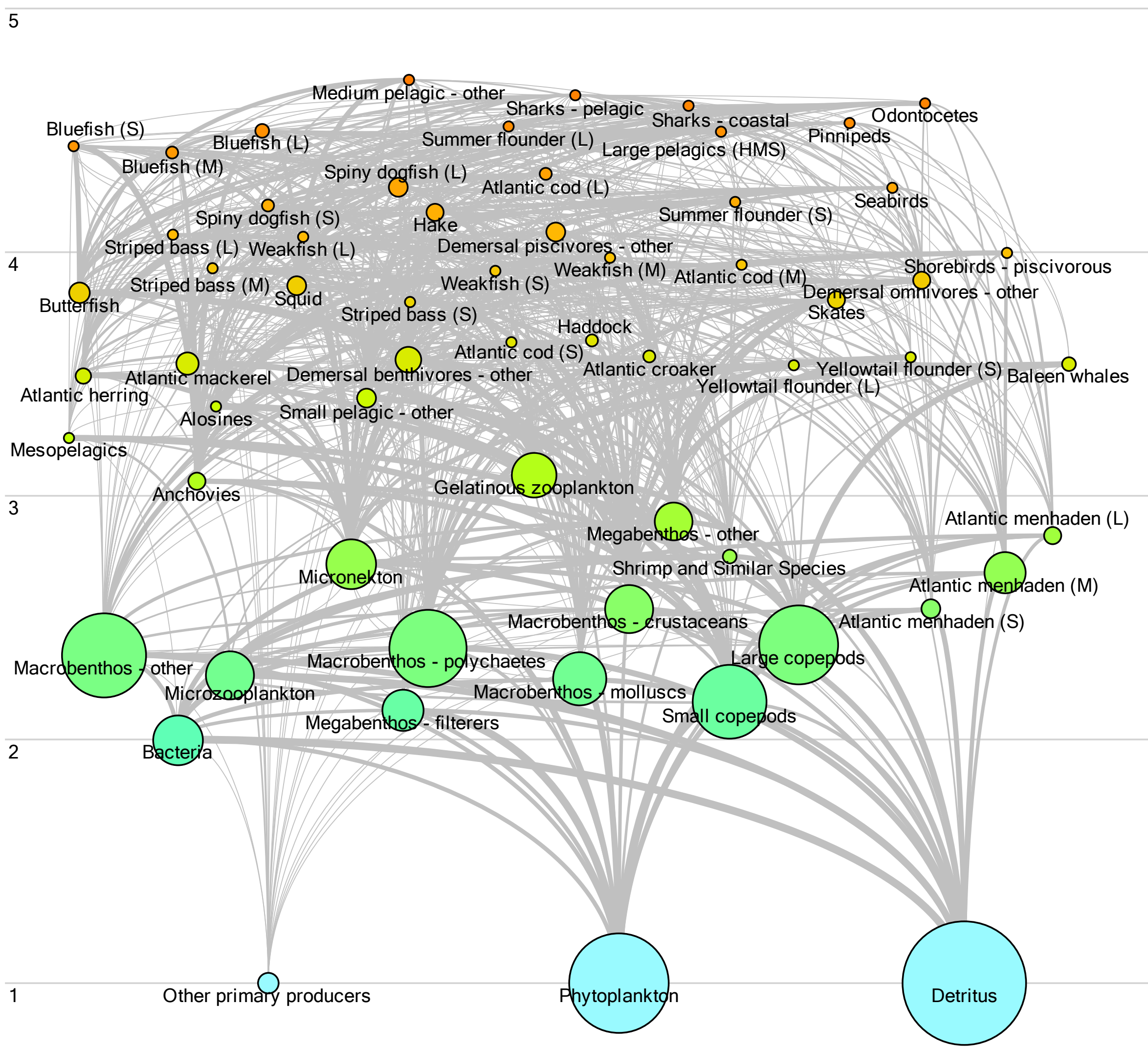


Figure 150. Flow diagram for the NWACS-FULL Model. Nodes represent biomass of modeled trophic groups (scaled to the logarithm of the group's biomass). Lines represent trophic linkages (with a scaled thickness). Colors and horizontal lines denote trophic levels.

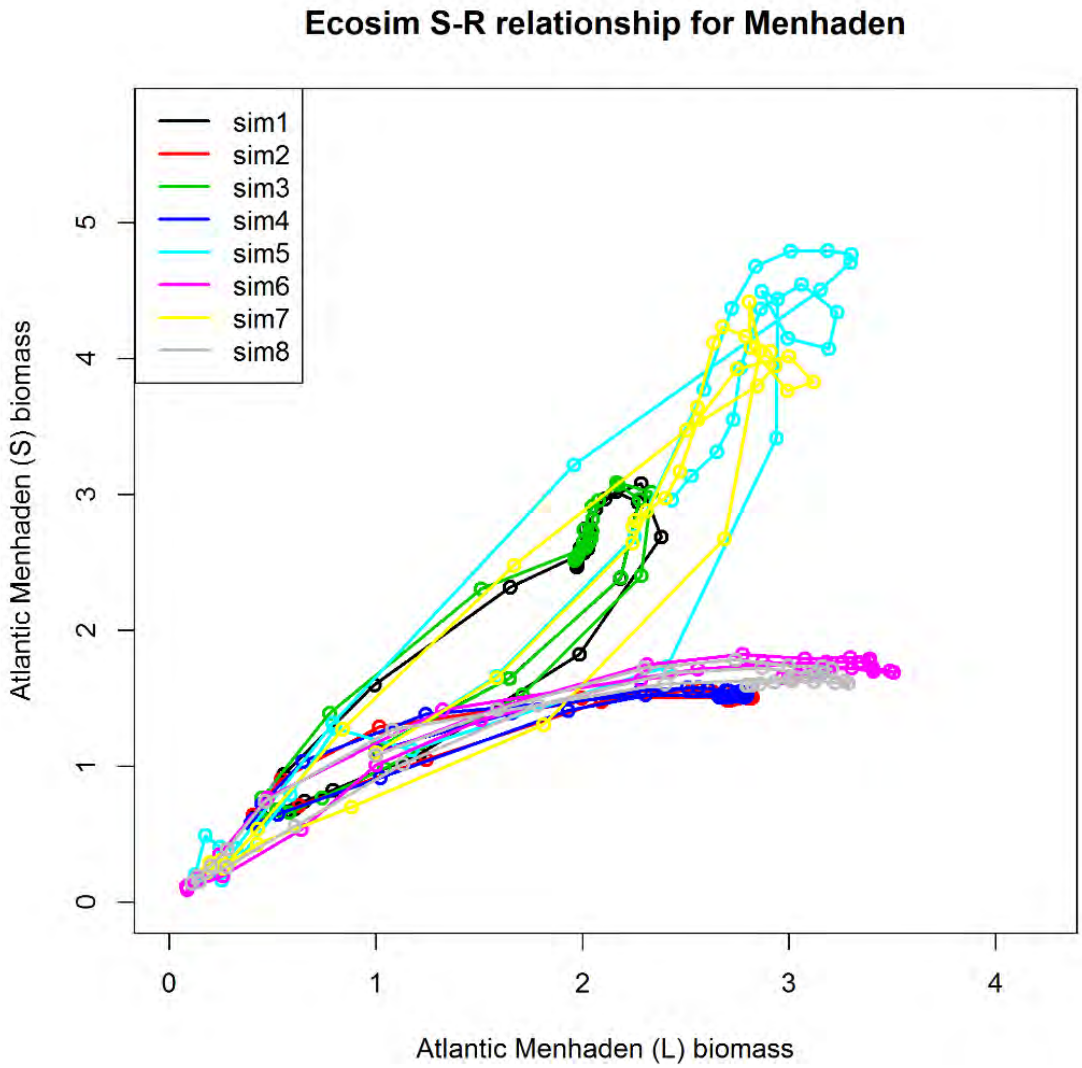


Figure 151. Emergent Atlantic menhaden stock-recruitment relationship from the NWACS-FULL model. Lines depict the relationship between age-3+ (i.e., Large, L) Atlantic menhaden biomass and age-0 (i.e., small, S) biomass for eight different simulations. Some simulations (Sims 2, 4, 6, and 8) had vulnerability parameters manually adjusted to obtain a Beverton-Holt stock-recruitment relationship.

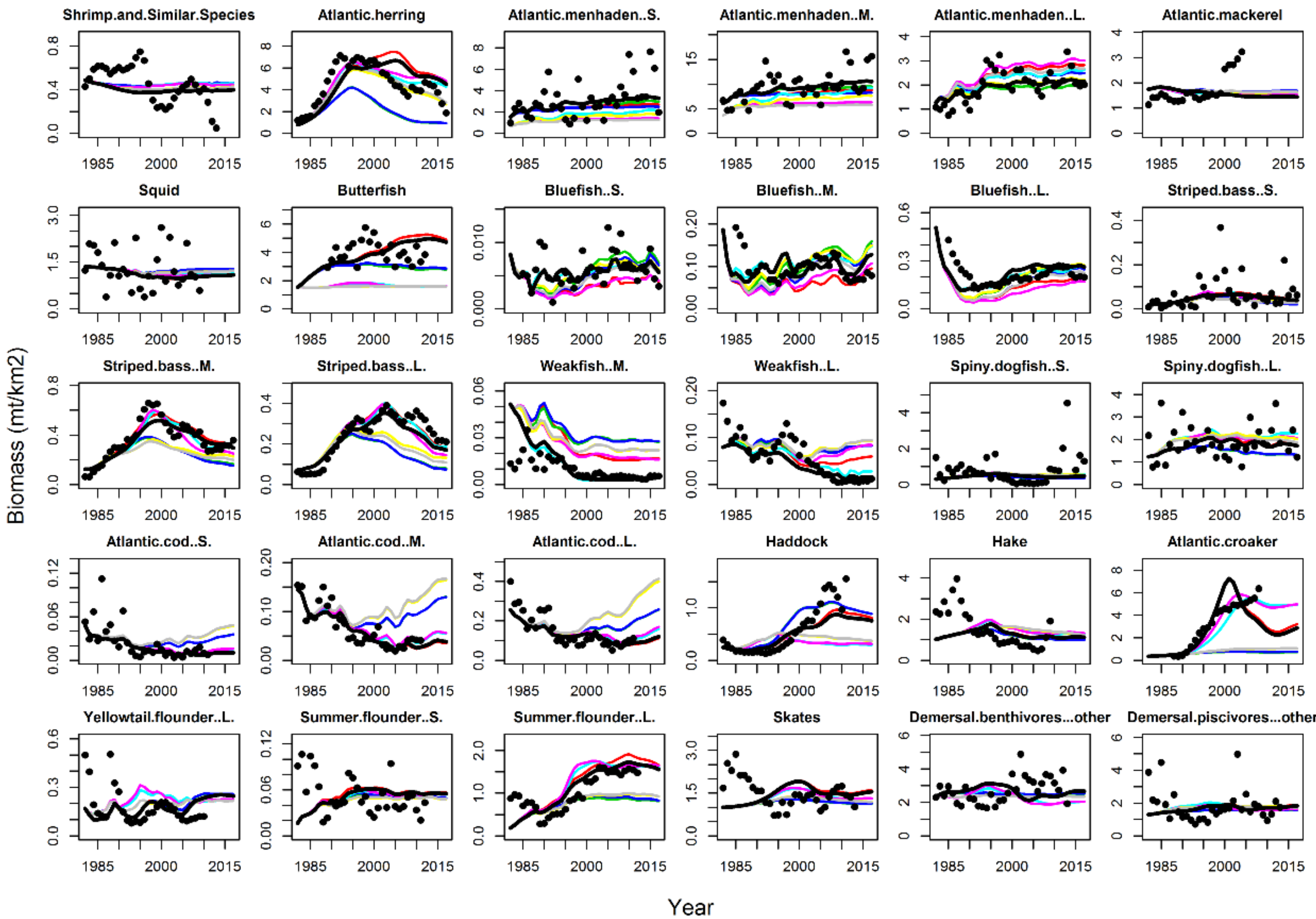


Figure 152. **Biomass fits from the NWACS-FULL model. Lines depict predicted biomass estimates by year for different model simulations (sim 1-8: 1-black, 2-red, 3-green, 4-blue, 5-cyan, 6-pink, 7-yellow, 8-gray). Points depict time series of relative biomass from stock assessments and fisheries surveys (magnitude of points is scaled based on sim 1). Panels are labeled by trophic group name and stanza if applicable (S-small, M-medium, L-large). Trophic groups without observed, empirical data are excluded.**

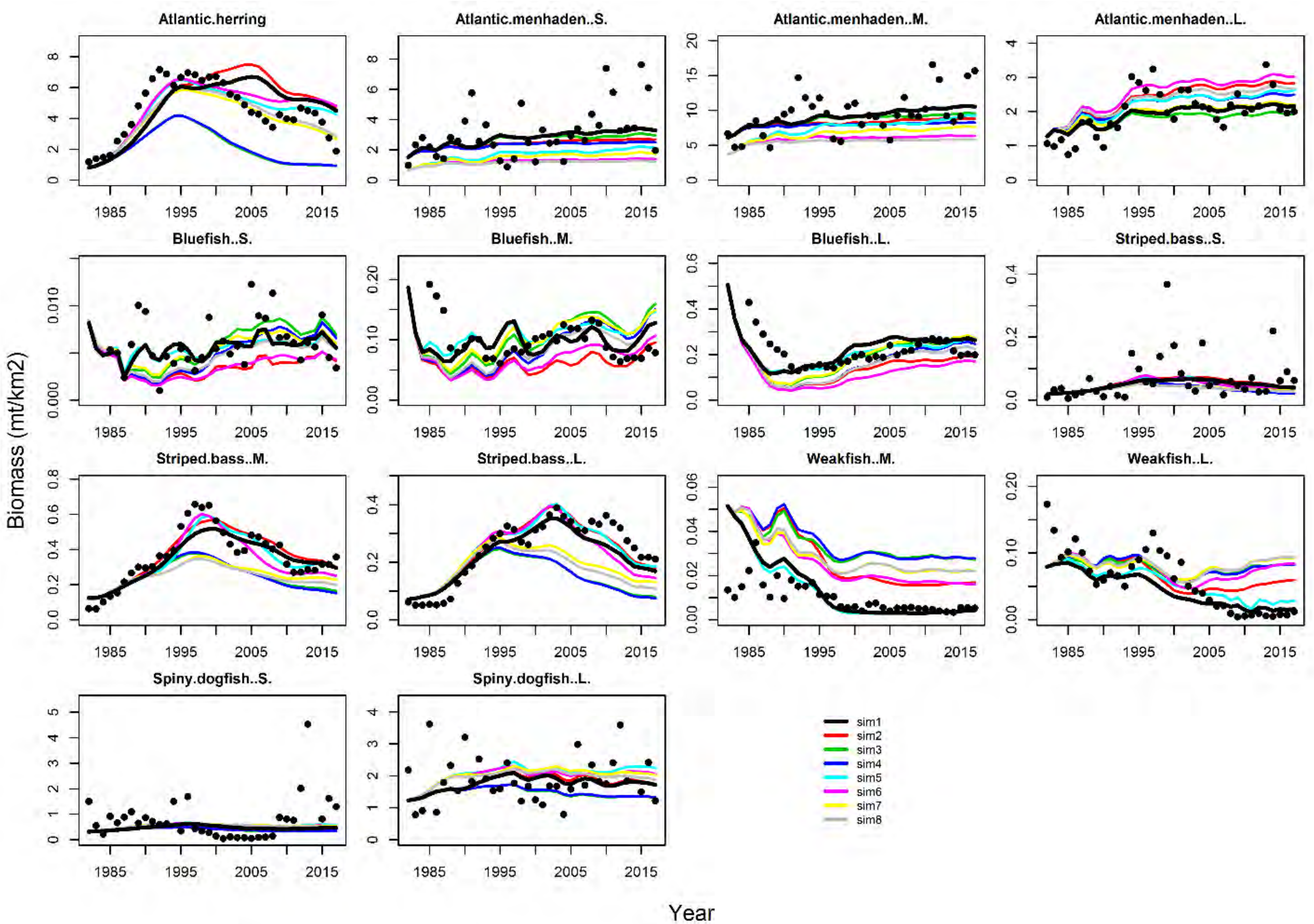


Figure 153. Biomass fits from the NWACS-FULL model for select ERP focal species (Atlantic herring, Atlantic menhaden, bluefish, striped bass, weakfish, and spiny dogfish). See Figure 152 for full description of symbols and lines.

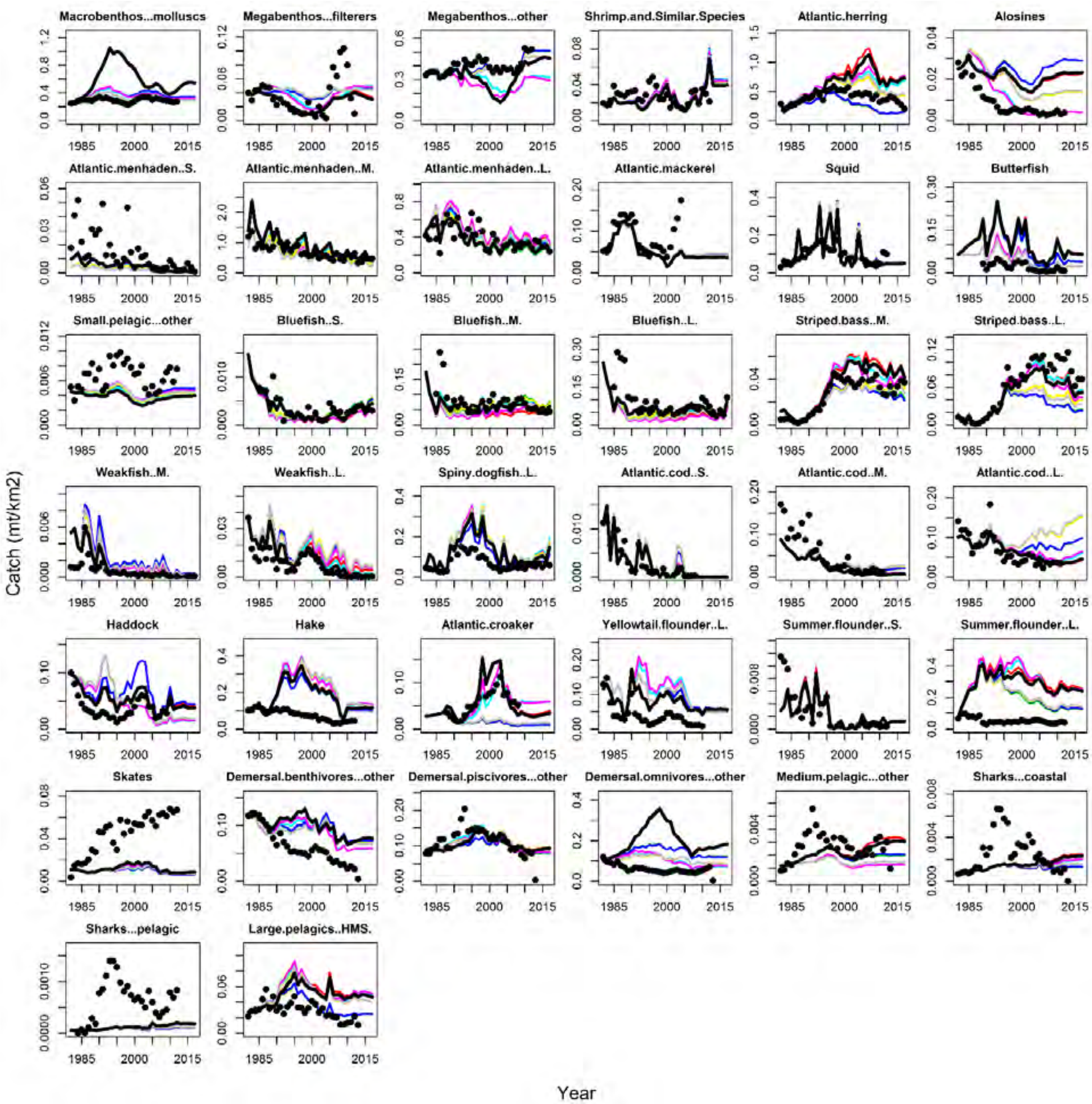


Figure 154. Catch fits from the NWACS-FULL model. Points are the observed catches and lines depict predicted catch by year for different model simulations (sim 1-8: 1-black, 2-red, 3-green, 4-blue, 5-cyan, 6-pink, 7-yellow, 8-gray). Panels are labeled by trophic group and stanza if applicable (S-small, M-medium, L-large). Trophic groups without observed, empirical data are excluded.

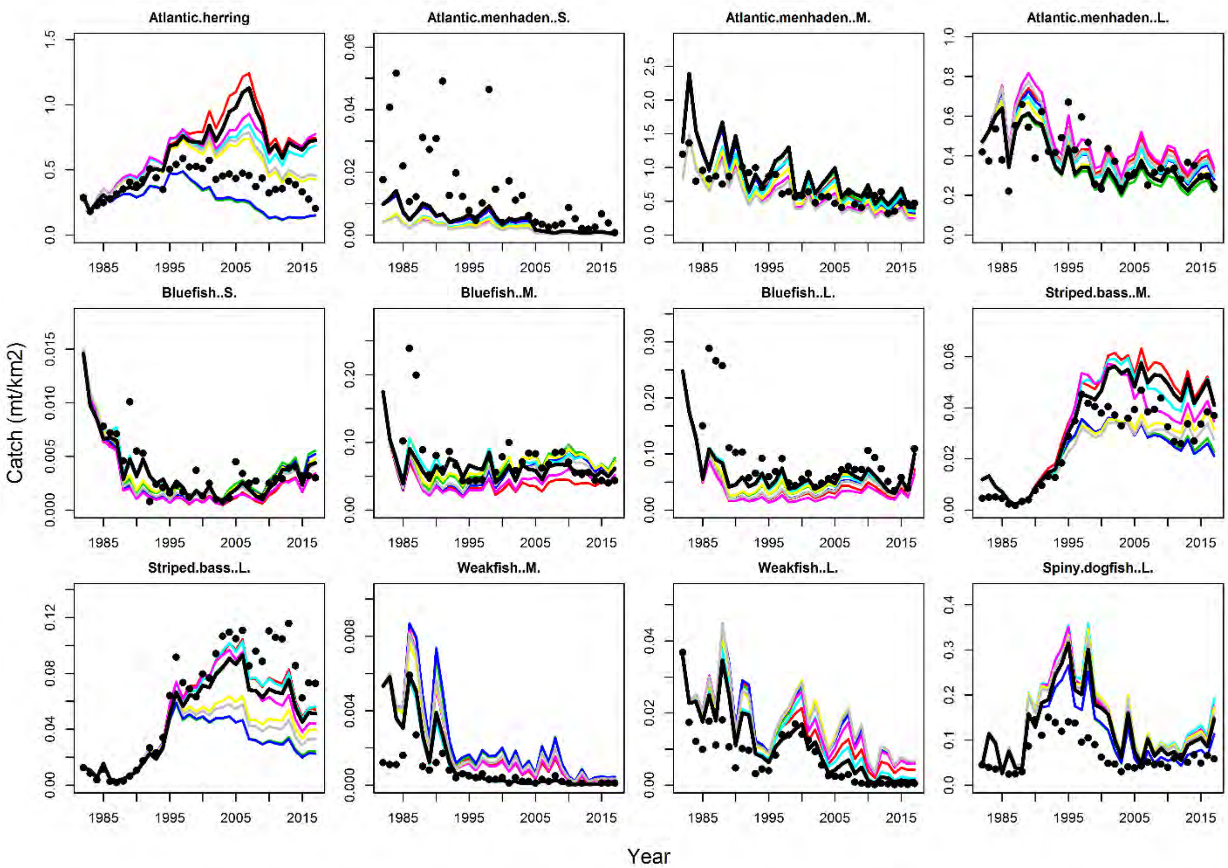


Figure 155. Catch fits for ERP focal species from the NWACS-FULL model. (Atlantic herring, Atlantic menhaden, bluefish, striped bass, weakfish, and spiny dogfish). See Figure 154 for full description.

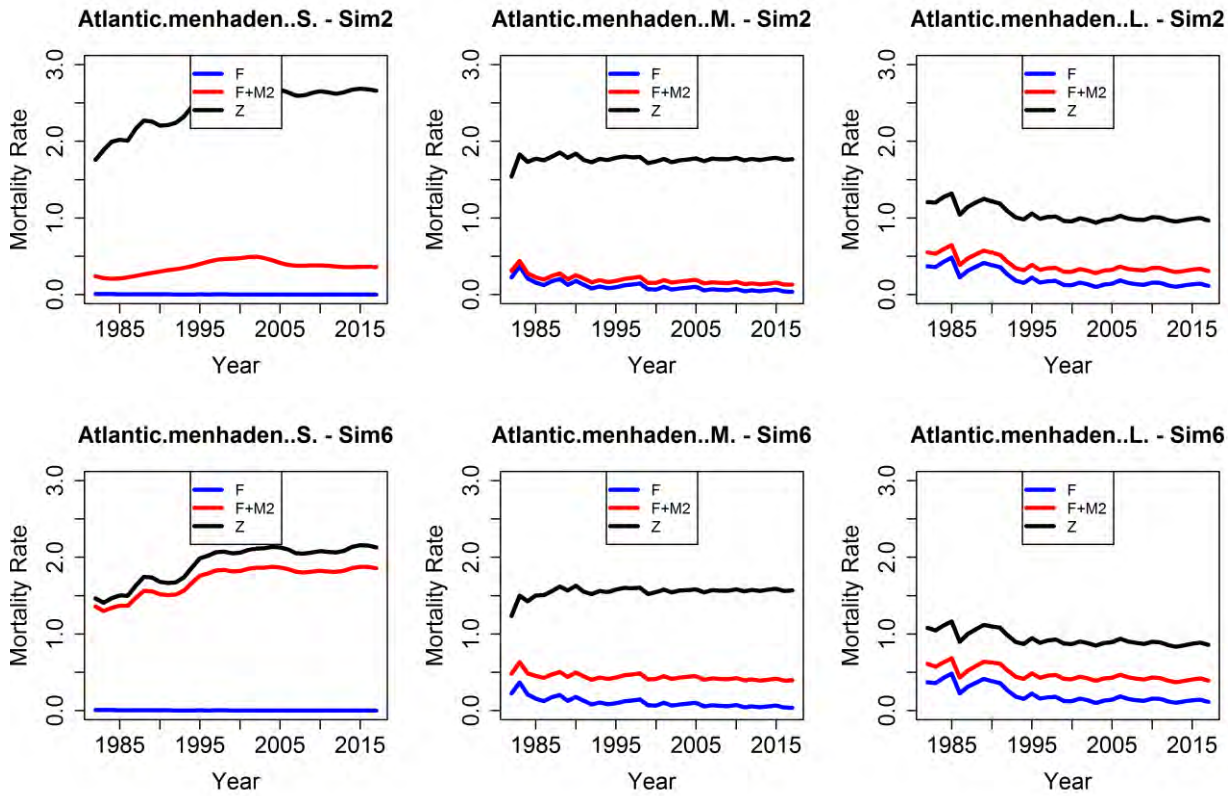


Figure 156. Instantaneous mortality rates for three age-classes of Atlantic menhaden from the NWACS-FULL model (S – age-0, M – age-1-2, L – age3+) based on sim 2 (upper panels) and sim 6 (lower panels)

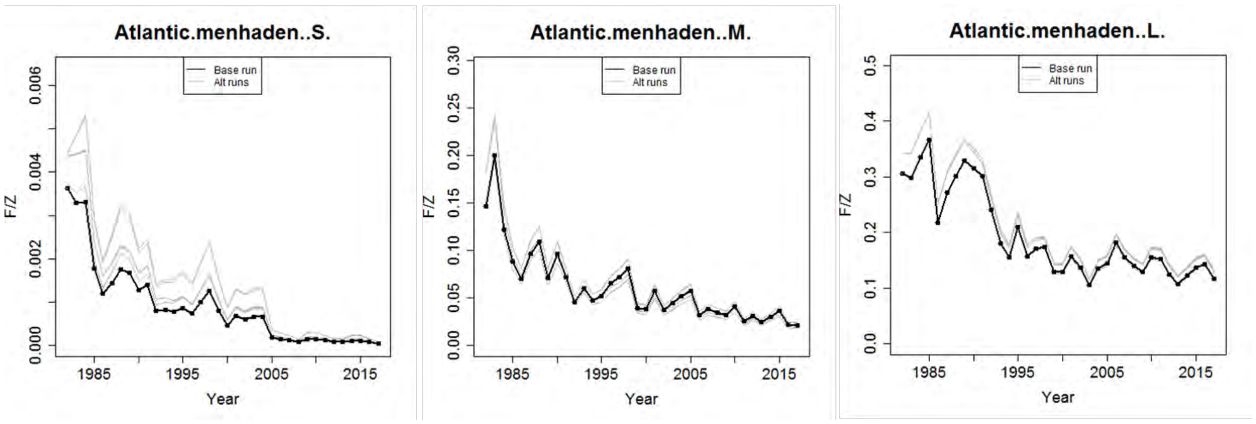


Figure 157. Fishing mortality (*F*) as a proportion of total instantaneous mortality (*Z*) for eight simulations of the NWACS-FULL model

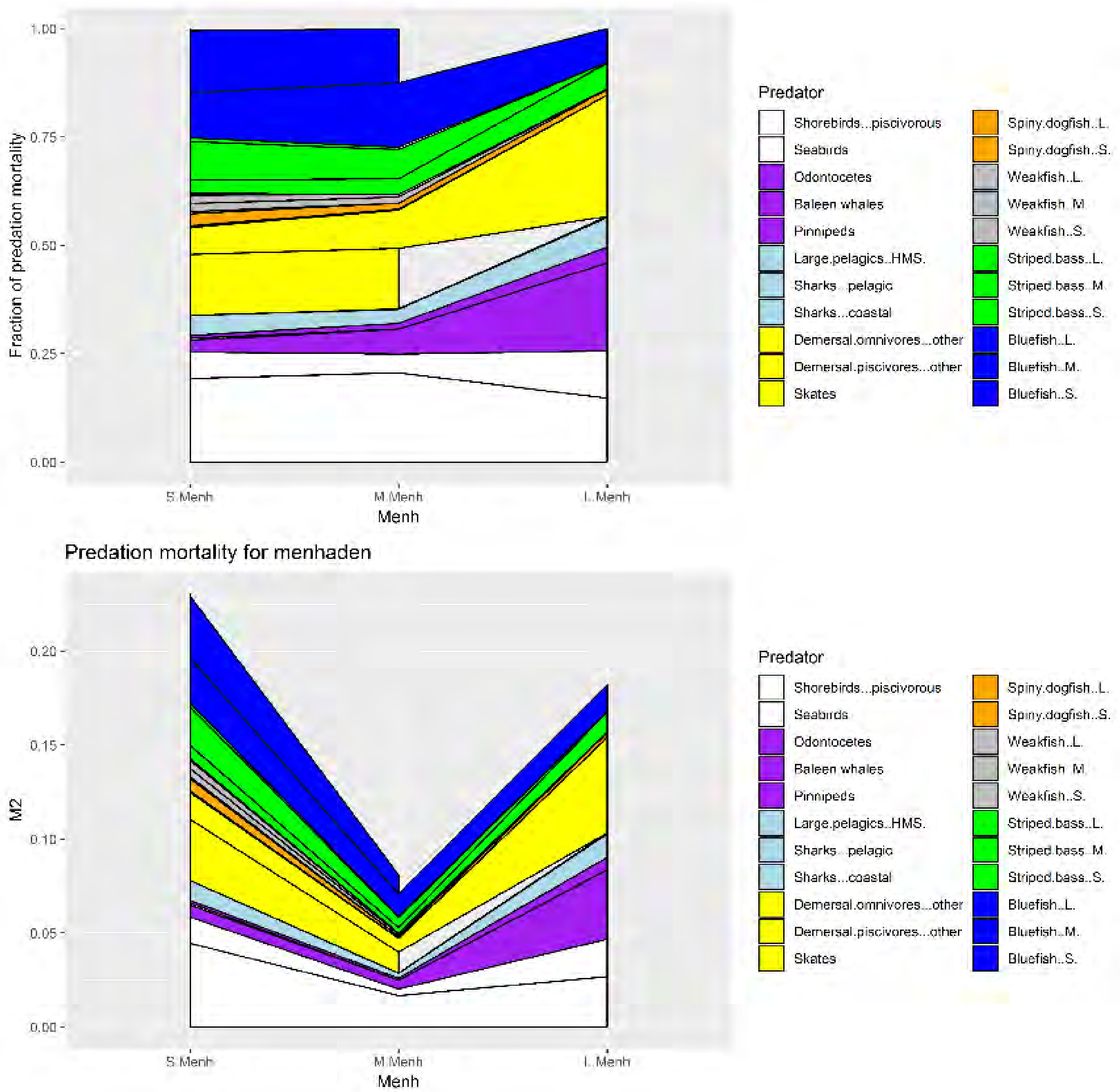


Figure 158. Predator contributions to Atlantic menhaden M_2 (bottom panel) and as fraction of total M_2 (upper panel), based on sim2 of the NWACS-FULL model. Predators and size classes are grouped by color.

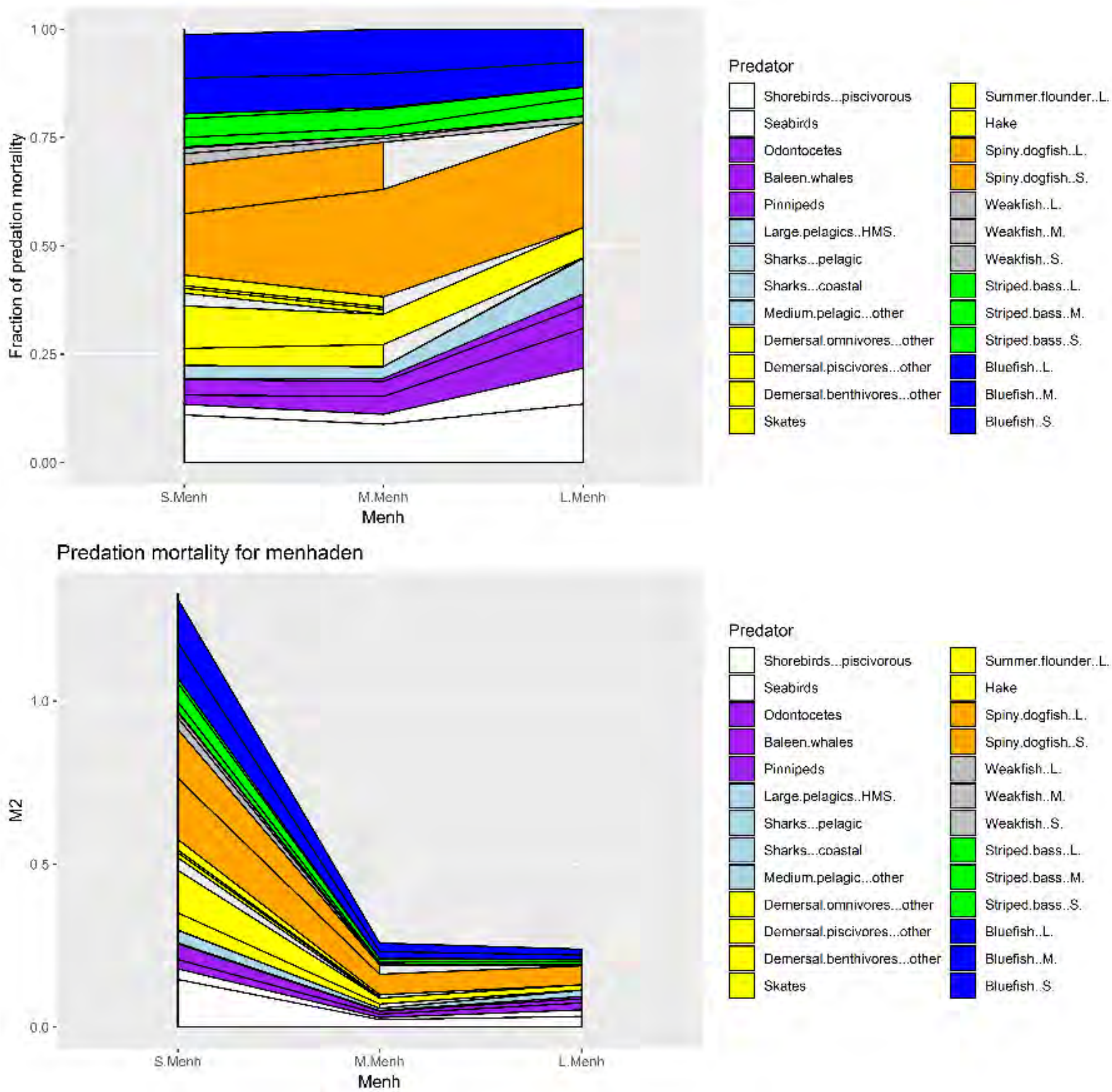


Figure 159. Predator contributions to Atlantic menhaden *M_2* (bottom panel) and as fraction of total *M_2* (upper panel), based on sim6 of the NWACS-FULL model. Predators and size classes are grouped by color.

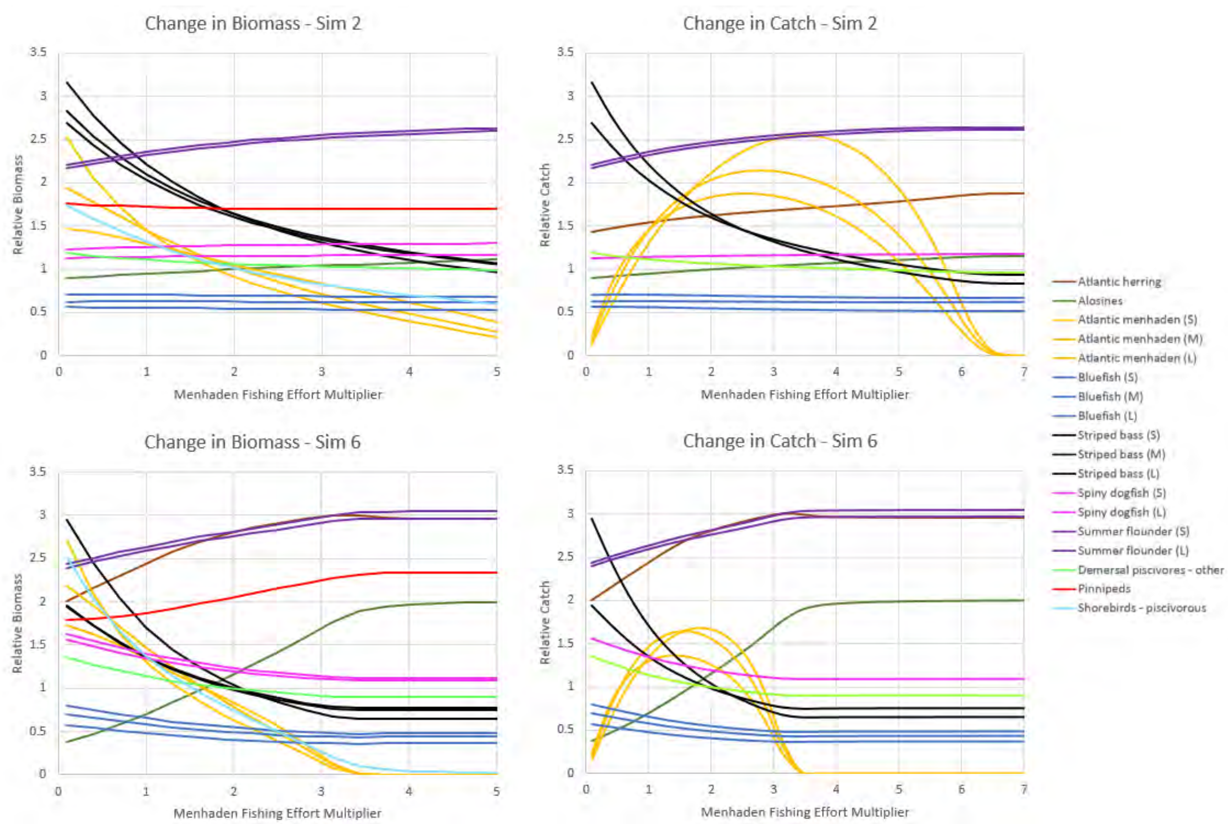


Figure 160. Effect of Atlantic menhaden fishing effort on the relative equilibrium biomass and catch of select trophic groups from the NWACS-FULL model. Non-menhaden species were kept at their Ecopath base (1982) *F* rates while Atlantic menhaden fishing effort was scaled from 0 to 5 times the 1982 values. Results are presented for Sim 2 (upper panels) and Sim 6 (lower panels).

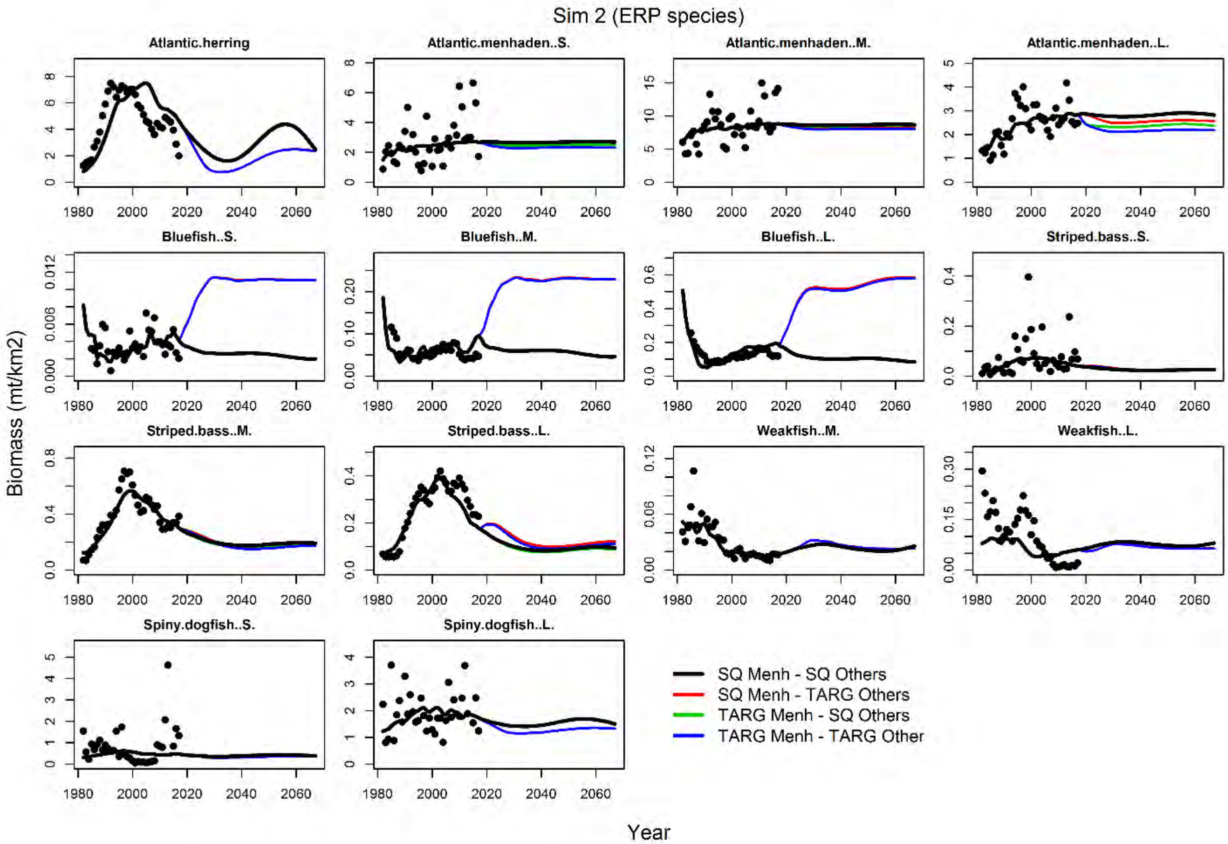


Figure 161. Projected biomass of select species based on Sim 2 of the NWACS-FULL model under four different fishing scenarios. Points are relative observed biomass values and lines are the model predictions when fishing rates are held at 2017 status quo (SQ) levels or target levels (TARG) for Atlantic menhaden (Menh) or focal ERP focal species (Others).

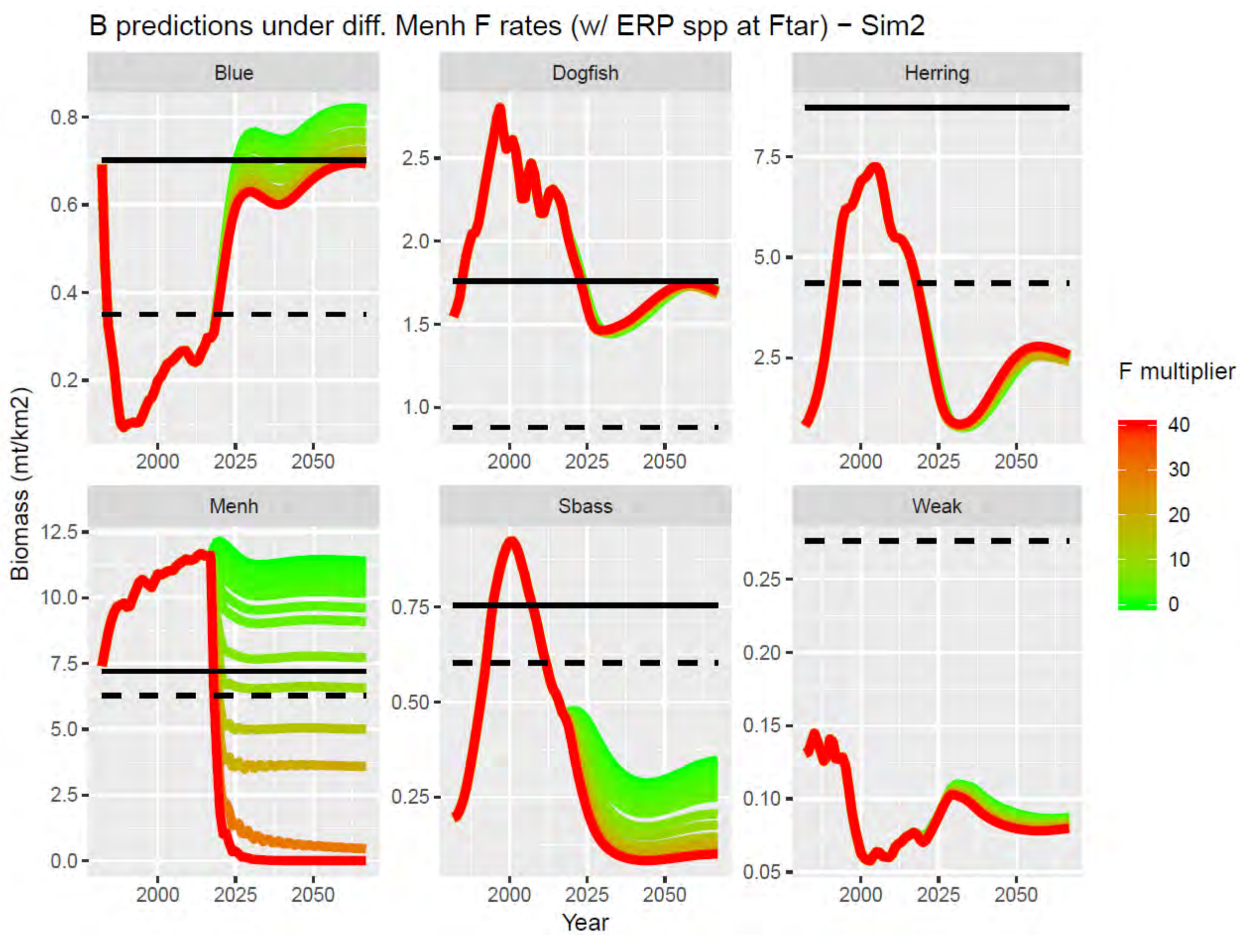


Figure 162. Biomass predictions from the NWACS-FULL model for select species under different Atlantic menhaden *F* rates while fishing the ERP focal species at their respective *F* targets. Atlantic menhaden *F* rates were scaled from F_{2017} using an *F*-multiplier for each simulation. Black horizontal lines denote the Biomass thresholds (dashed) and targets (solid) for each species (as available).

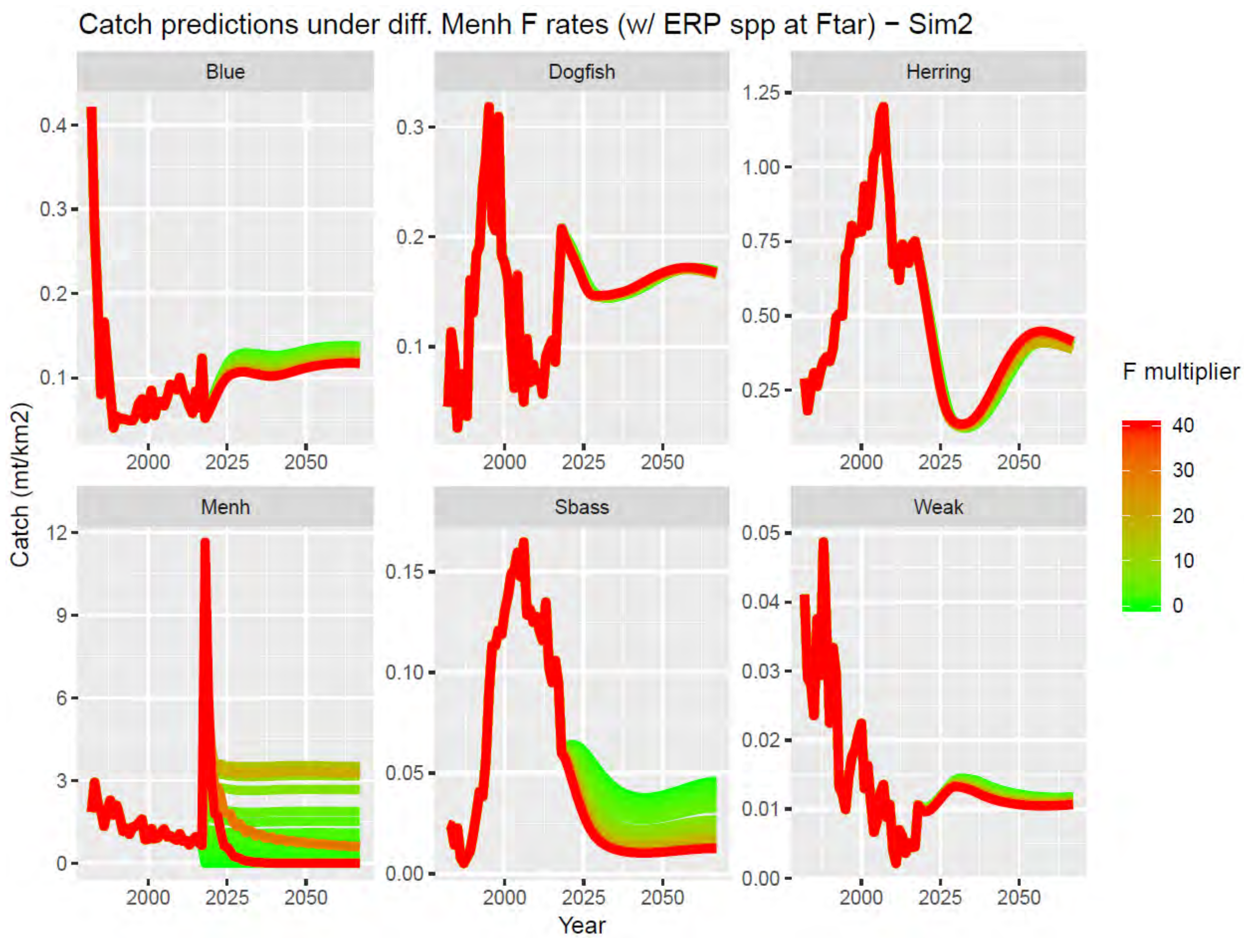


Figure 163. Catch predictions from the NWACS-FULL model for select species under different Atlantic menhaden *F* rates while fishing the ERP focal species at their respective *F* targets. Atlantic menhaden *F* rates were scaled from F_{2017} using an *F*-multiplier for each simulation. Black horizontal lines denote the Biomass thresholds (dashed) and targets (solid) for each species (as available).

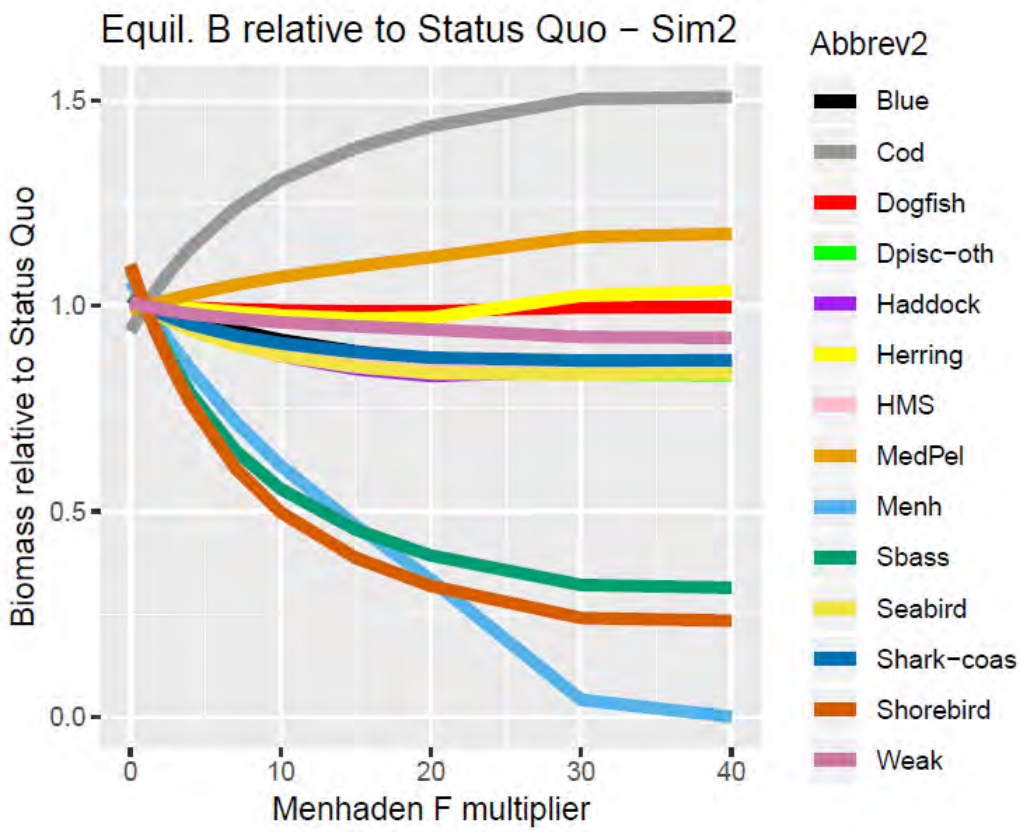


Figure 164. Effect of Atlantic menhaden fishing on equilibrium biomass of select trophic groups (projected for 50 years) relative to their equilibrium biomass under status quo Atlantic menhaden fishing rates from the NWACS-FULL model. ERP focal species were fished at their target *F* while Atlantic menhaden *F* rates were scaled from 0 to 40 times the 2017 values using an F-multiplier. Biomasses for all species were summed across age-stanzas (if applicable). Lines are plotted for all ERP focal species and other trophic groups with non-negligible (>15%) responses.

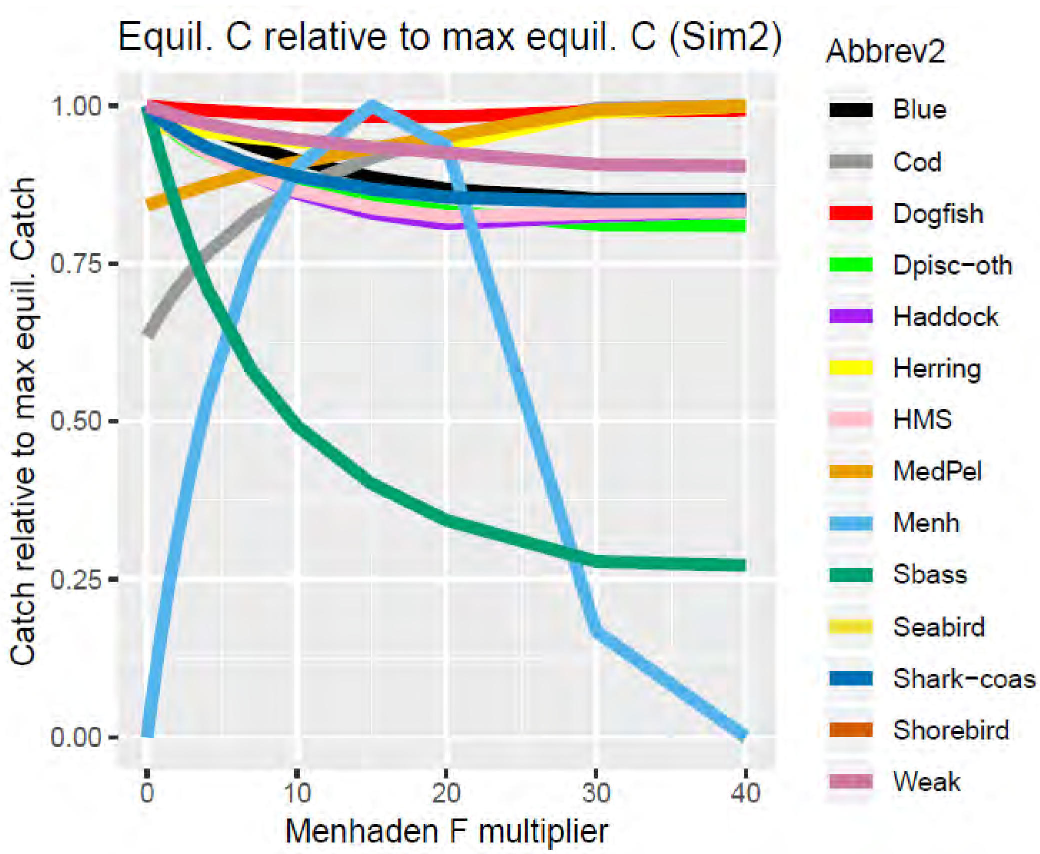


Figure 165. Effect of different Atlantic menhaden fishing mortality projections on the equilibrium (50-year) catch of selected trophic groups relative to the maximum equilibrium catch observed across all fishing scenarios from the NWACS-FULL model. Non-menhaden species were kept at their target *F* while Atlantic menhaden *F* rates were scaled from 0 to 40 times the 2017 values using an F-multiplier. Catches for all species were summed across age-stanzas (if applicable). Lines are plotted for all ERP focal species and other trophic groups with non-negligible (>15%) responses.

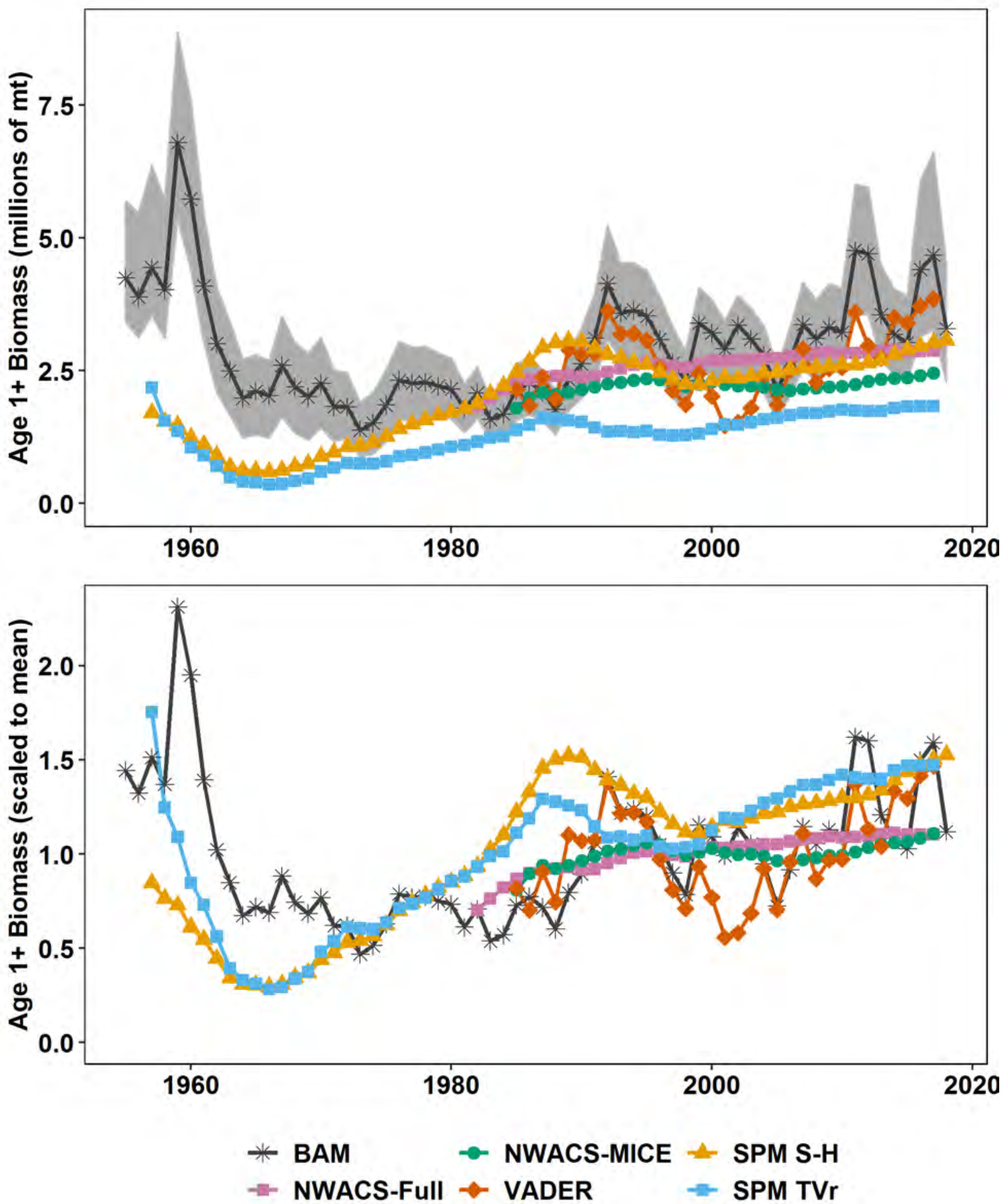


Figure 166. Estimates of age-1+ biomass from the base runs of the ERP models (top) and scaled to their respective time series means (bottom). Shaded area on top plot indicates the MCMC confidence intervals from the single species assessment model.

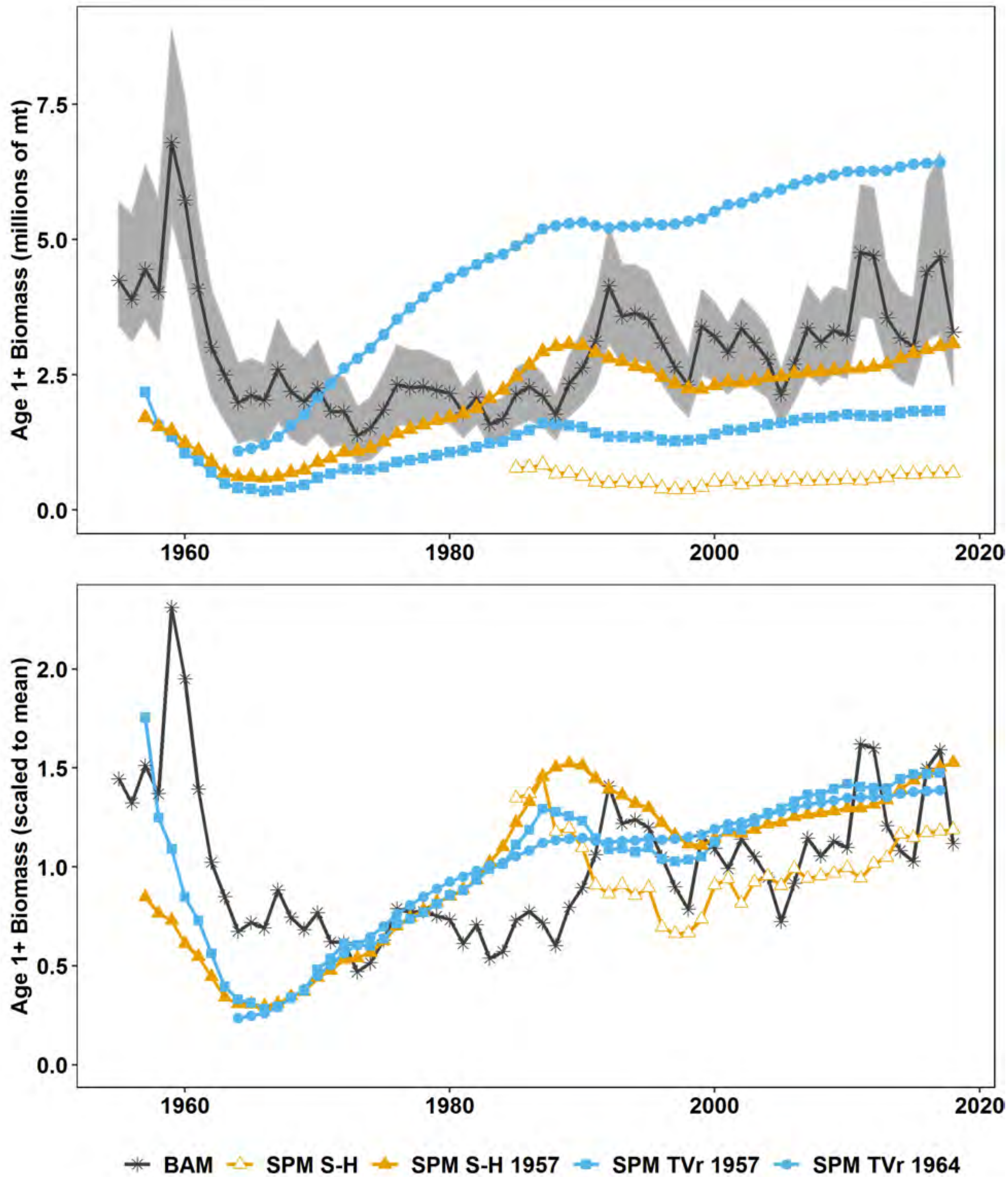


Figure 167. Estimates of age-1+ biomass from the single species (BAM) assessment model plotted with the Steele-Henderson and time-varying *r* surplus production models with different starting years (top) and scaled to their respective time-series means (bottom).Shaded area on top plot indicates the MCMC confidence intervals from the single species assessment model.

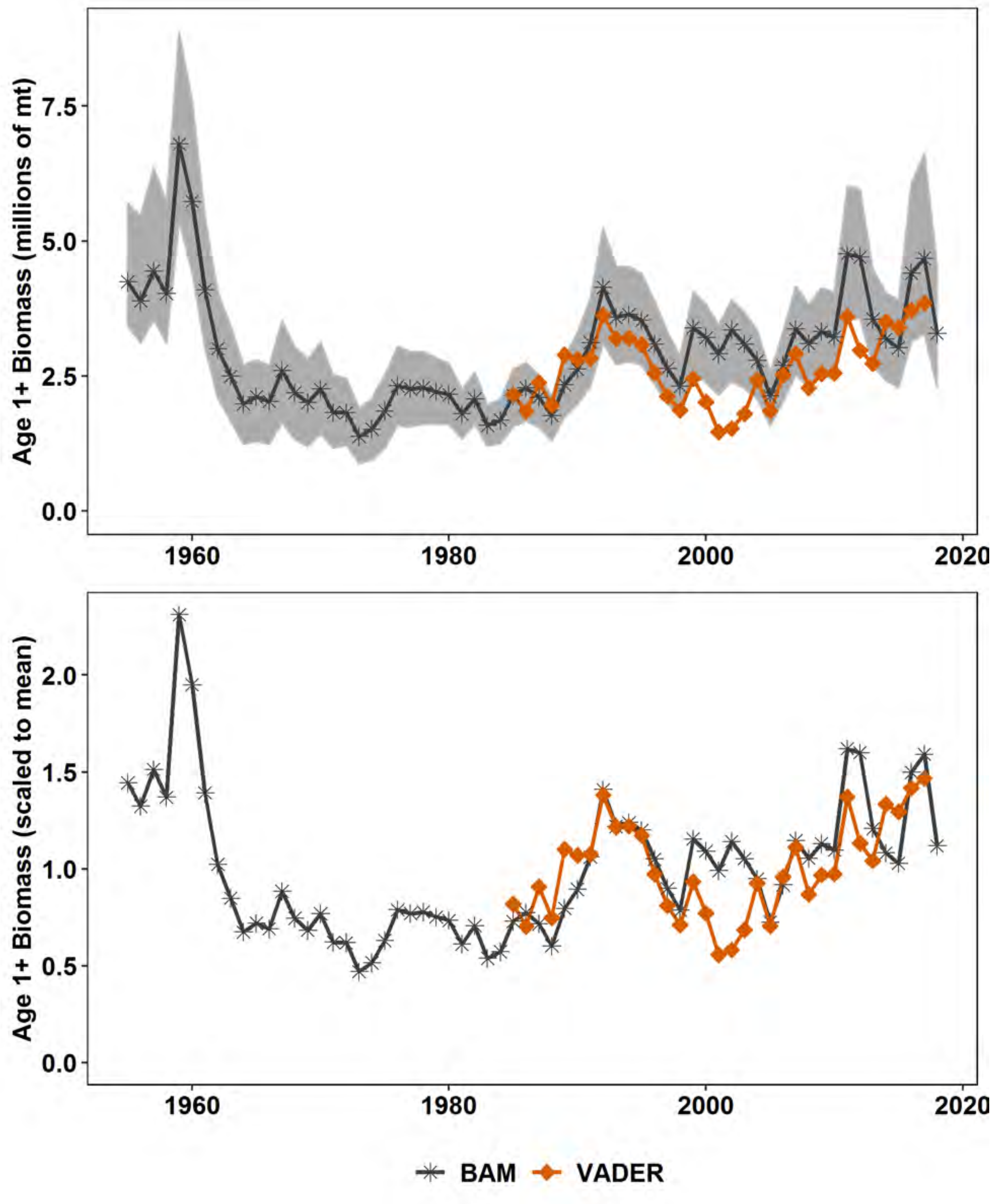


Figure 168. Estimates of age-1+ biomass from the single species (BAM) assessment model plotted with the multispecies statistical catch-at-age (VADER) model (top) and scaled to their respective time-series means (bottom).Shaded area on top plot indicates the MCMC confidence intervals from the single species assessment model.

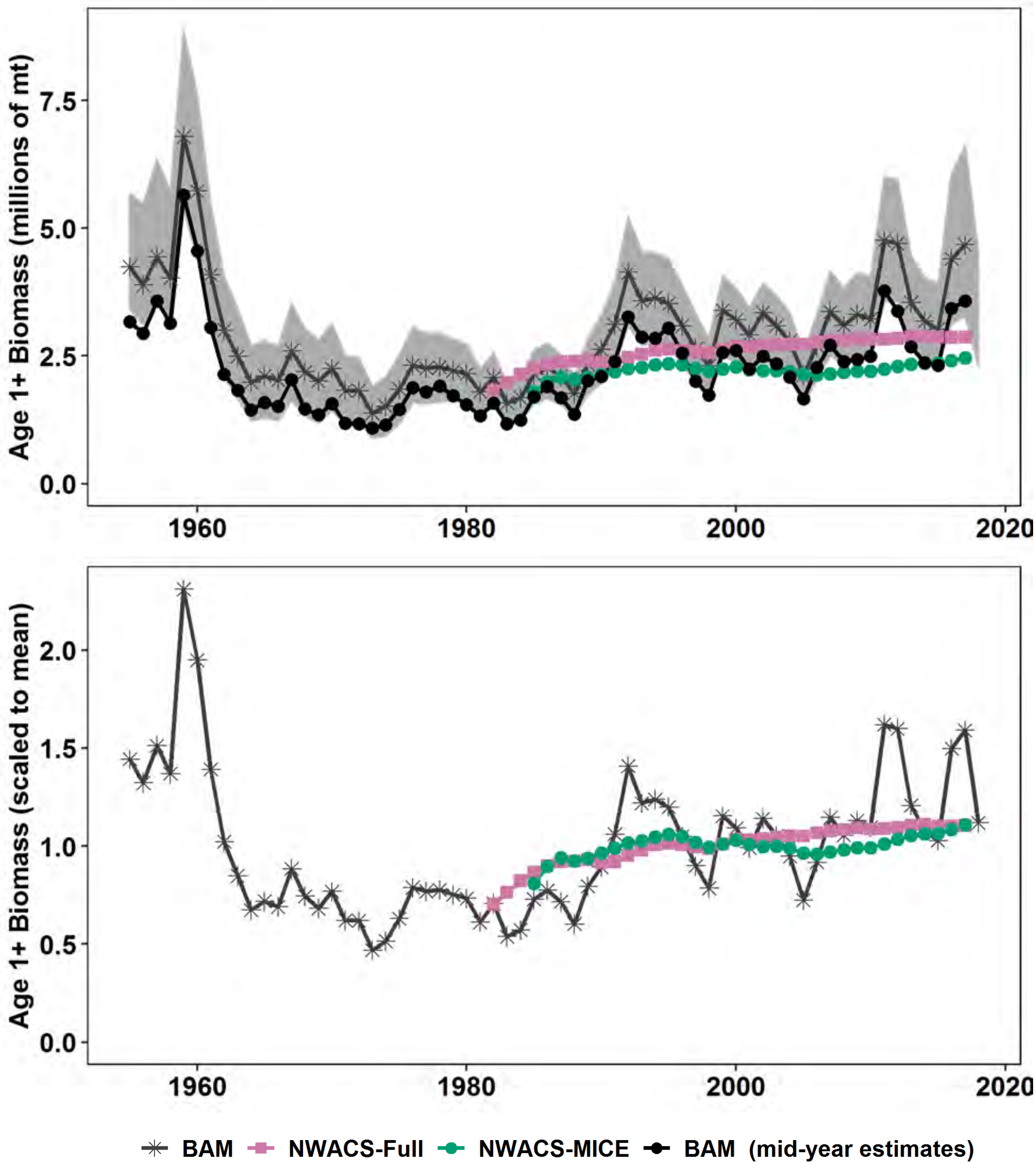


Figure 169. **Estimates of age-1+ biomass from the single species assessment model at the start of the year (BAM) and at the middle of the year (BAM mid-year estimates) plotted with the NWACS model estimates (top) and scaled to their respective time series means (bottom). Shaded area on top plot indicates the start of year biomass MCMC confidence intervals from the single species assessment model.**

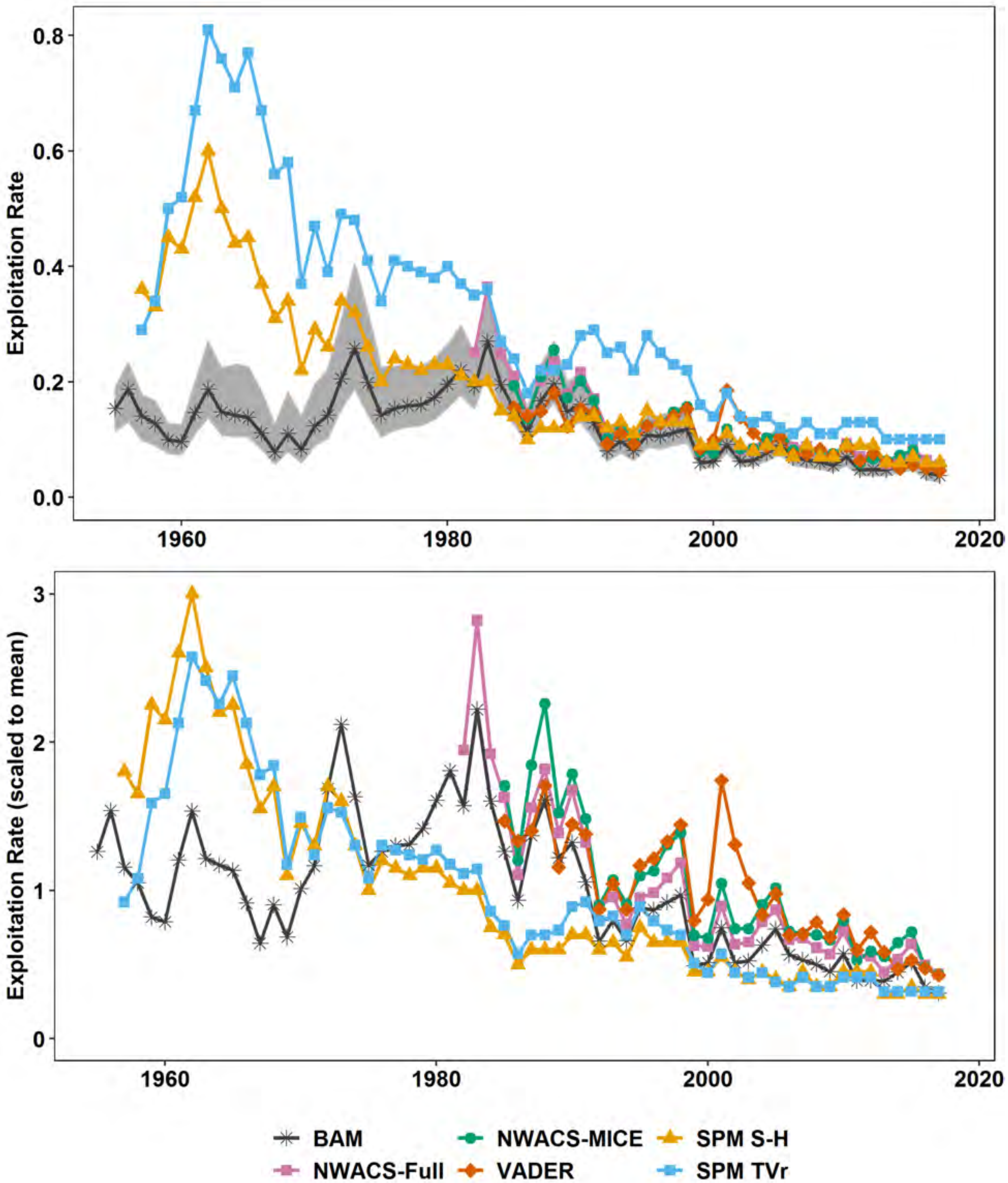


Figure 170. Exploitation rates from the single species assessment model plotted with the exploitation rates from the ERP models (top) and scaled to their respective time series means (bottom).Shaded area on top plot indicates MCMC confidence intervals from the single species assessment model.

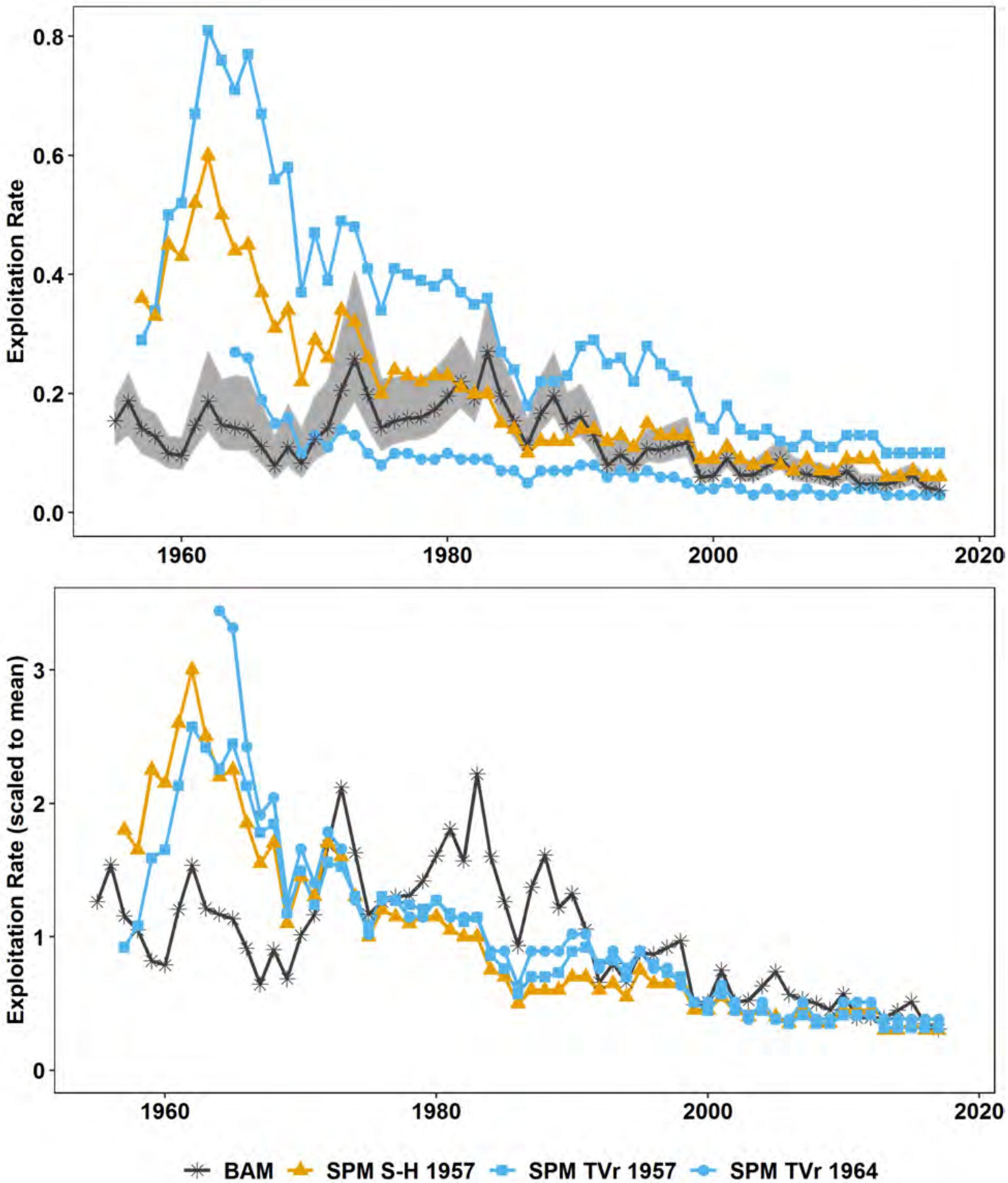


Figure 171. Exploitation rates from the single species model plotted with the exploitation rate estimates from the surplus production models with differing start years. Shaded area on top plot indicates MCMC confidence intervals from the single species assessment model.

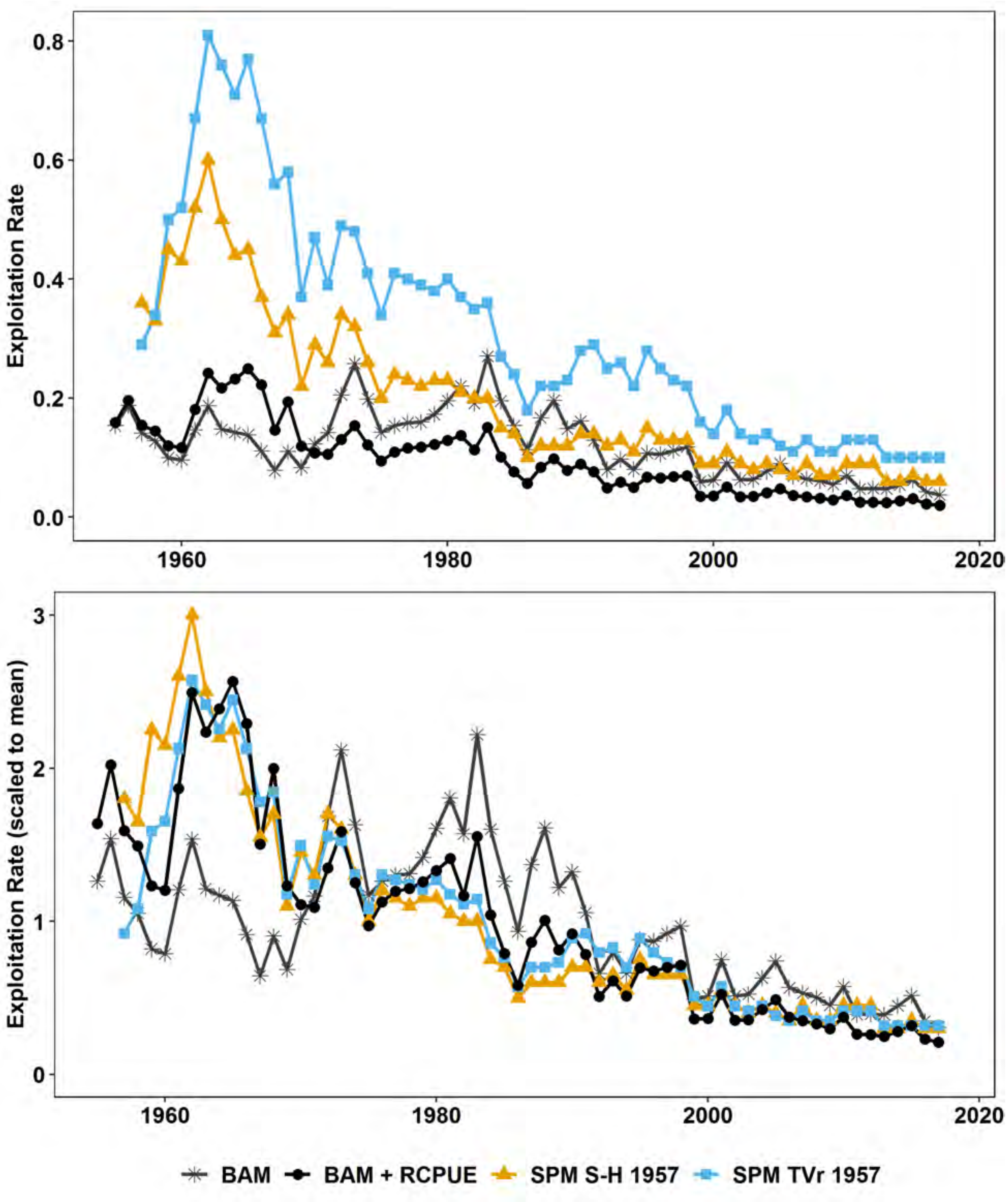


Figure 172. Estimates of exploitation rate from the surplus production models, the base run of the BAM, and a sensitivity run of the BAM that included the RCPUE index.

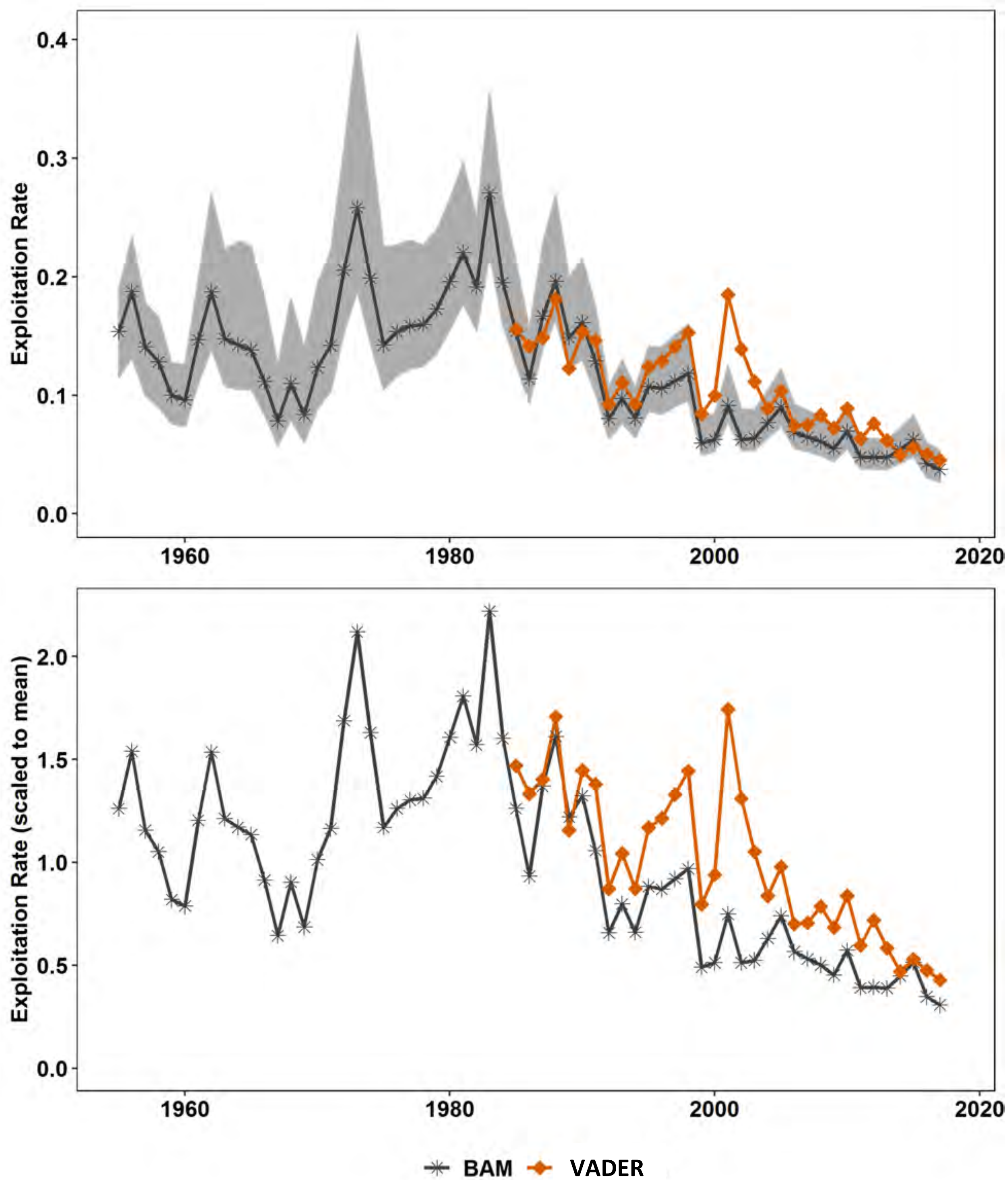


Figure 173. **Exploitation rate estimates from the single species assessment model (BAM) plotted with the exploitation rate estimates from the multispecies statistical catch-at-age (VADER) model (top) and scaled to their respective time series means (bottom). Shaded area on top plot indicates MCMC confidence intervals from the single species assessment model.**

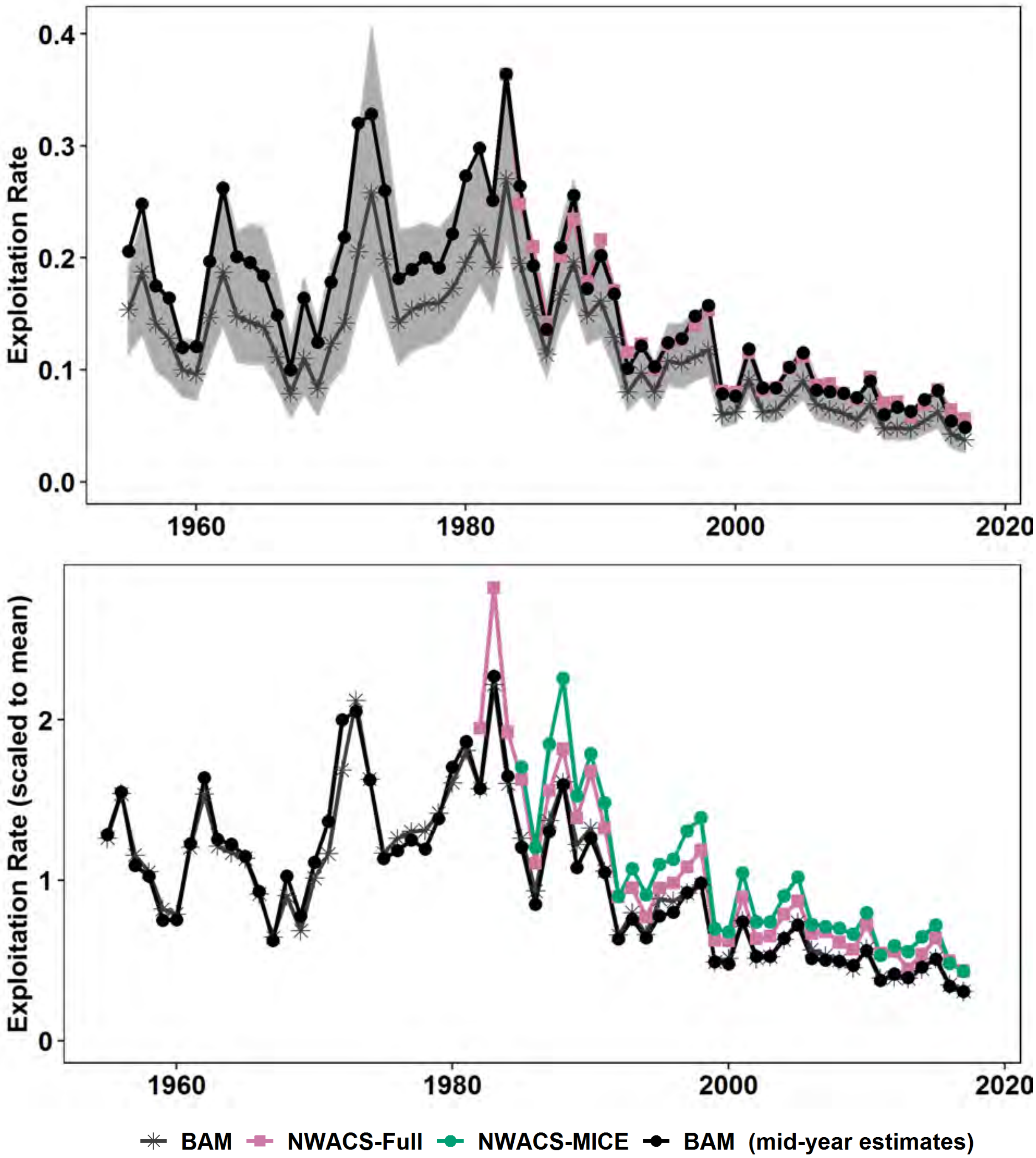


Figure 174. **Estimates of exploitation rate from the single species assessment model at the start of the year (BAM) and at the middle of the year (BAM mid-year estimates) plotted with the NWACS model estimates (top) and scaled to their respective time series means (bottom). Shaded area on top plot indicates the start of year biomass MCMC confidence intervals from the single species assessment model.**

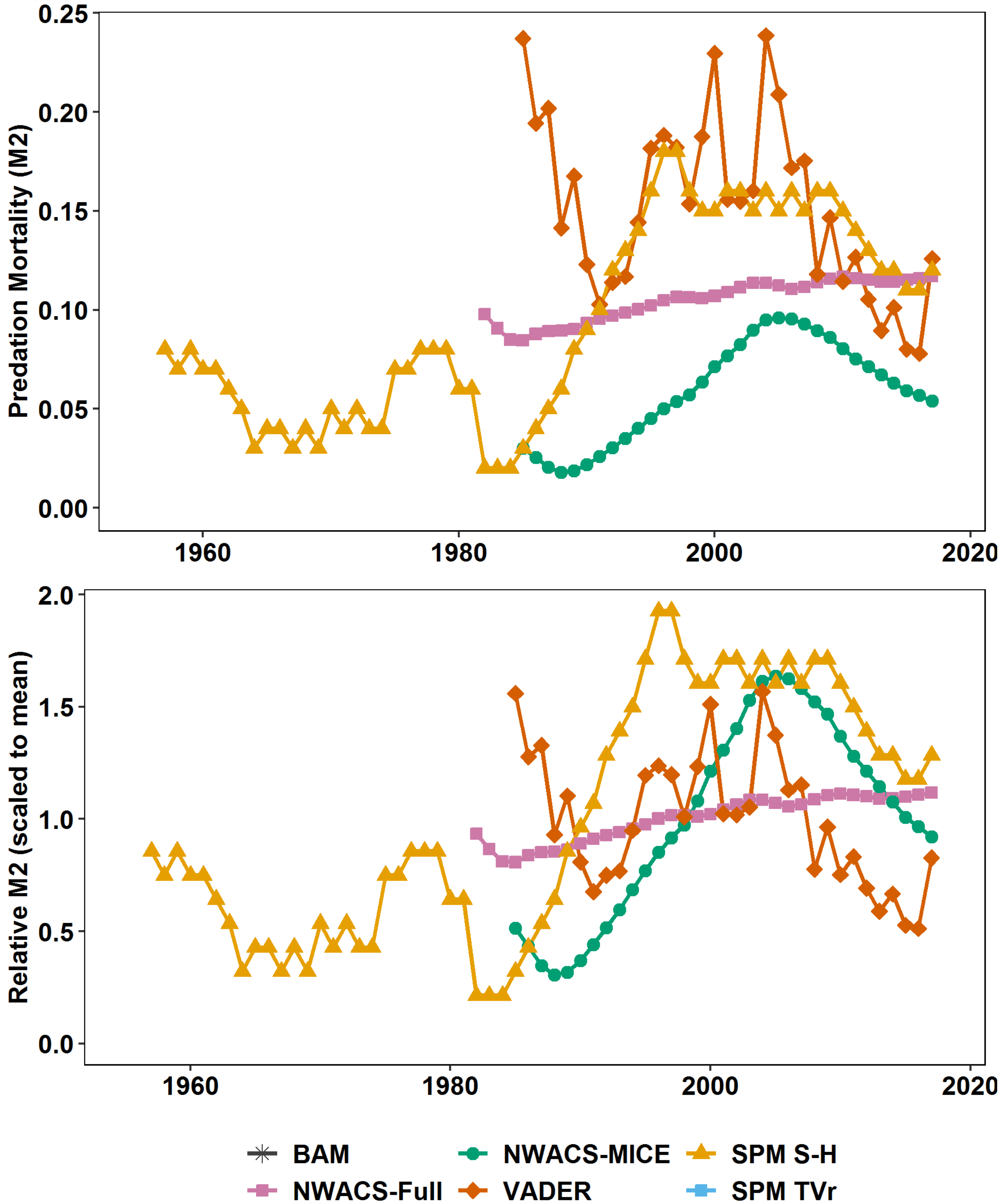


Figure 175. **Estimates of modeled predation mortality (*M_2*) from the ERP models (top) and scaled to their respective time-series means (bottom).**

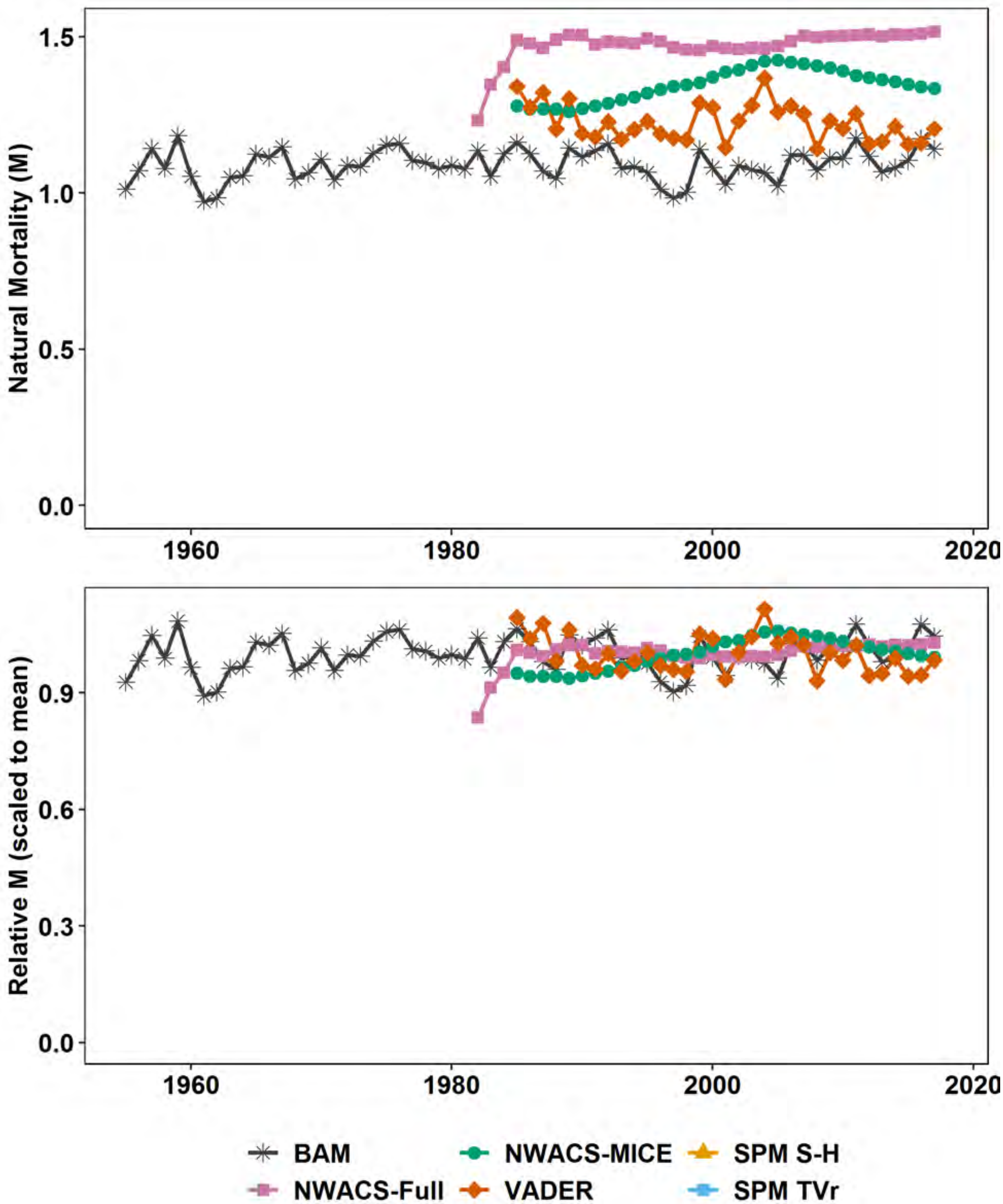


Figure 176. Estimates of total natural mortality (*M*) from the ERP models plotted with the natural mortality estimate from the single-species assessment model.

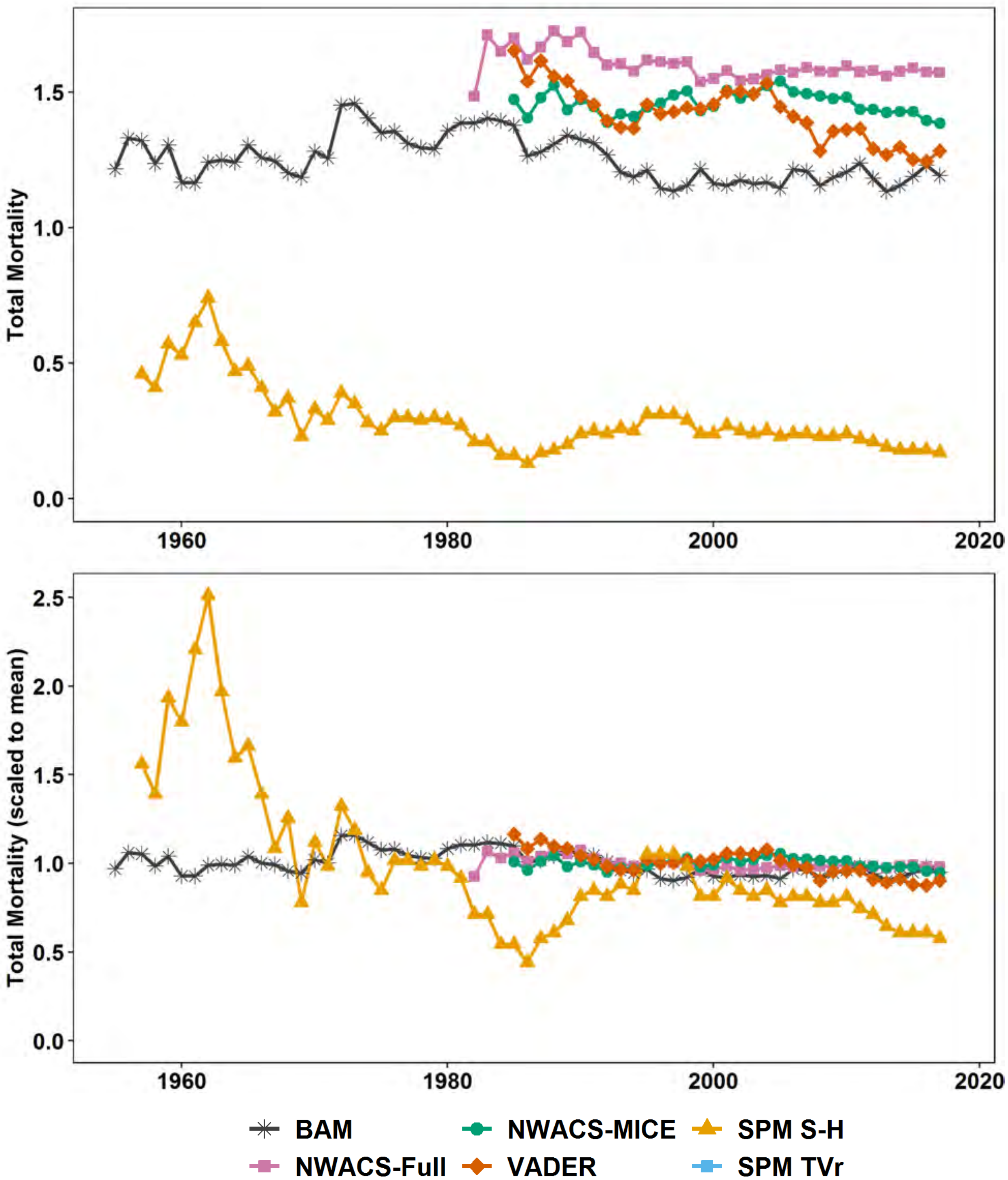


Figure 177. Estimates of total mortality from the single species assessment model (BAM) plotted with the total mortality estimates from the EPR models (top) and scaled to their respective time series means (bottom).