SEDAR

Southeast Data, Assessment, and Review

SEDAR 102
Stock Assessment Report

ASMFC Atlantic Menhaden and Ecological Reference Points

October 2025

SEDAR
4055 Faber Place Drive, Suite 201
North Charleston, SC 29405

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SEDAR 102 Atlantic Menhaden and Ecological Reference Points Stock Assessment Introduction

The 2025 Stock Assessment of Atlantic Menhaden and Ecological Reference Points occurred through a joint Atlantic States Marine Fisheries Commission (ASMFC) and Southeast Data, Assessment and Review (SEDAR) process. The ASMFC conducted a Data Workshop in October 2023, two Methods Workshops in October 2023 and November 2024 and an Assessment Workshop in March 2025. Participants of the Data and Assessment Workshops included the ASMFC Atlantic Menhaden Stock Assessment Subcommittee and the Ecological Reference Points Work Group (ERP WG). SEDAR coordinated a Review Workshop from August 12-15, 2025. Participants included members of the Atlantic Menhaden ERP Work Group, the Stock Assessment Subcommittee and a Review Panel consisting of a chair appointed by ASMFC, and two reviewers appointed by the Center for Independent Experts. This report is the culmination of a three-year effort to gather and analyze available data for Atlantic Menhaden Ecological Reference Points.

Updating the Beaufort AssessmentModel (BAM) for the Atlantic menhaden single species stock assessment and further developing an Ecopath with Ecosim (EwE) model for generating Ecological Reference Points, in combination provides the best scientific advice for managing the menhaden fishery. In the current assessment, the methods used in the EwE model were reviewed and deemed sound by the Review Panel. The Review Panel recommended that stock status for this species be evaluated by comparing the single species assessment to the Ecological Reference Points arising from the EwE model. The Review Report and Stock Assessment Report will be provided to the ASMFC Atlantic Menhaden Fishery Management Board in October 2025.

The ASMFC and its committees thank the SEDAR 102 reviewers for their time and expertise in providing a thorough evaluation of the Atlantic Menhaden and Ecological Reference Points stock assessments. Additionally, ASMFC would like to recognize the SEDAR staff for their dedicated work in coordinating the review. Finally, ASMFC thanks the Stock Assessment Subcommittee, Technical Committee, ERP WG, and all of the individuals who contributed to completion of the stock assessments.

SEDAR

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SEDAR 102

ASMFC Atlantic Menhaden and Ecological Reference Points

SECTION II: Assessment Process Report

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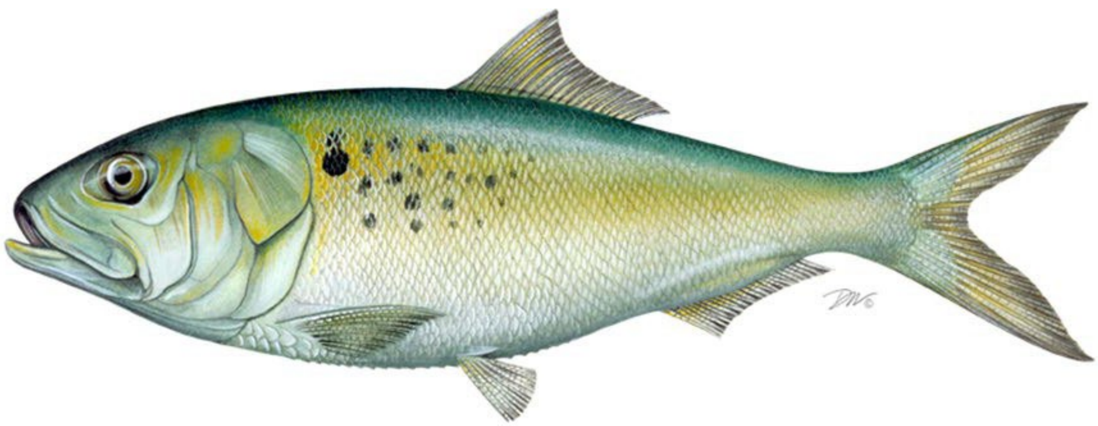
2025 Atlantic Menhaden Ecological Reference Point Stock Assessment Report



Vision: Sustainably Managing Atlantic Coastal Fisheries

Atlantic States Marine Fisheries Commission

2025 Atlantic Menhaden Ecological Reference Point Benchmark Stock Assessment

Prepared by the
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The development of the NWACS-MICE and NWACS-FULL ecosystem models for this assessment would not have been possible without a Saltonstall-Kennedy grant (NA23NMF4270272) and the Lenfest Ocean Program.

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 Northwest Atlantic Continental Shelf ecosystem. The Northwest 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 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 Northwest Atlantic Continental Shelf ecosystem, including larger fish such as striped bass and bluefin tuna, birds such as bald eagles and ospreys, and marine mammals like humpback whales and bottlenose dolphins. 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 developed an Ecopath with Ecosim (EwE) model of intermediate complexity with a limited predator/prey field that could be used in conjunction with the single-species statistical catch-at-age model to provide catch advice. This model was successfully peer-reviewed as part of the 2020 ERP benchmark assessment (SEDAR, 2020b). For this assessment, the moderate complexity EwE model was refined to account for seasonal and spatial dynamics and updated with new data as the preferred model. A full-complexity EwE model and a multispecies statistical catch-at-age model were developed further as complementary models to provide context to the intermediate complexity EwE model results.

A suite of five key predator and prey species (referred to as the ERP species) were identified from diet data and other considerations as part of the 2020 benchmark. The ERP species decisions were reviewed during this assessment, and the ERP WG elected to carry all five species forward. 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 many of the other predators identified. The multispecies statistical catch-at-age model included all of the ERP 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 WG considered other candidate ERP species based on manager and stakeholder feedback and elected not to include additional species in the intermediate complexity models, but updated and refined species groups within the full EwE model to better address these concerns. In particular, the "highly migratory species" group was replaced with an explicit bluefin tuna group, and osprey were split out from "nearshore piscivorous birds" to take advantage of species-specific data for osprey and provide information on a high-profile species.

The ERP models were parameterized with the best available data for Atlantic menhaden and the ERP species. For Atlantic menhaden, data from the single species update assessment conducted in parallel with this assessment were used. The Menhaden single species assessment update used the same statistical catch-at-age model developed for the 2020 benchmark assessment (SEDAR, 2020a). However, the update used a new, lower estimate of natural mortality based on a revised analysis of the historical tagging data, resulting in lower estimates of biomass and higher estimates of *F* compared to the 2020 benchmark and 2022 update. This change in *M* was carried through to the ERP models.

All ERP species had recently undergone benchmark assessments or assessment updates. Meaning life history, landings, and index data were available through 2022 or 2023, as well as estimates of fishing mortality and population size. Species and species groups in the full EwE were updated with the most recent stock assessment, catch, and fishery independent index data where available.

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 the top-down effects of predation on Atlantic menhaden and the bottom-up effects of Atlantic menhaden biomass levels on predators to quantify tradeoffs between management objectives. The EwE models were the only models that were able to evaluate both factors. The current iteration of the multispecies statistical catch-at-age model lacked the bottom-up feedback necessary to explore trade-offs between Atlantic menhaden harvest and predator biomass. Adding bottom-up feedback to this model through an empirically derived relationship between striped bass growth and menhaden abundance was explored for this assessment, but model performance was not satisfactory, and the ERP WG did not consider it ready to provide management advice.

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 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.

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, as was recommended in the previous ERP benchmark.

This approach combined the individual strengths of each model. The single-species model provided the best information on current 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, nearshore piscivorous birds, and osprey were among the most sensitive species in the full EwE models given that they showed larger changes in biomass than other species did in response to increases or decreases in fishing pressure on Atlantic menhaden, but the striped bass results were deemed the most robust as the data on striped bass diet, life history parameters, and abundance trends were more extensive and reliable than those available for osprey and nearshore piscivorous birds. The Atlantic menhaden harvest scenarios that sustain the biomass of predators included in the intermediate complexity EwE were thus expected not to cause large declines for other predators that were only included in the full EwE model. In addition, it would be more 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 with higher degrees of uncertainty for many groups. As a result, updating the full EwE model would be a significant effort with greater amount of uncertainty.

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 ERP approach was developed to assist with the decision-making process. To illustrate the potential use of the combined single-species assessment and intermediate complexity EwE model, the ERP WG put forward proof-of-concept values of an ERP target and an ERP threshold based on the current definition of ERPs adopted by the Atlantic Menhaden Management Board in 2020. 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 fished at the status quo F rates, in this case, the F rate each species experienced in 2023. The ERP fecundity target and threshold were developed from the single-species menhaden model and defined as the long-term equilibrium fecundity that results when the population is fished at the ERP F_{TARGET} and threshold, respectively.

Striped bass was the focal species for this analysis because it was the most sensitive of the ERP species to Atlantic menhaden F, and focusing on one key predator provided a more tractable example for evaluating tradeoffs among management strategies. ERPs based on striped bass biomass should not cause significant declines for other species that were less sensitive to levels of Atlantic menhaden removals.

The proof-of-concept ERP target and threshold from this assessment were similar to the ERP values currently used in management. The ERP target from this assessment was estimated at F=0.189, compared to the current ERP target of F=0.19, while the ERP threshold from this assessment was estimated at F=0.49, compared to the current ERP threshold of F=0.57. The associated fecundity (*FEC)* target and threshold for this assessment were also lower, with the FEC_{TARGET} equal to 1.67×10^{15} eggs, compared to the current FEC_{TARGET} of 2.00×10^{15} eggs. The $FEC_{THRESHOLD}$ was estimated at 1.14×10^{15} eggs, compared to the current $FEC_{THRESHOLD}$ of 1.49×10^{15} eggs.

Under these ERPs, Atlantic menhaden were not overfished and not experiencing overfishing. The 2023 estimate of F was below the ERP $F_{THRESHOLD}$ but above the F_{TARGET}, and the 2023 estimate of fecundity was above the $FEC_{THRESHOLD}$ but below the FEC_{TARGET}.

These proof-of-concept ERPs were based on the current definitions of ERPs used in management and 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 to set the final reference point values and total allowable catch.

Next Steps

This approach represents continued progress in the practical application of an ecosystem approach to fishery management. The ERP WG identified several research recommendations dealing with data collection, modeling, and the management process to improve the ERP assessment and move the ecosystem approach to management forward.

The ERP models developed for this assessment did not explicitly include spatial dynamics, although seasonal dynamics were included to implicitly represent spatial overlap of predators and prey. Incorporating finer scale dynamics would be possible for some of the models but would require both additional work on model development and higher resolution data. Spatially and seasonally resolved data were lacking or insufficient, making it difficult to parameterize and calibrate the models on that scale. The ERP WG recommended expanding the collection of diet 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, the 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, and 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 two 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 noted that there are several factors that will influence the timing of the next update and benchmark assessments for ERPs, making it difficult to recommend a specific timeline. The ERP WG recommends updating the NWACS-MICE model in conjunction with the next menhaden single-species update in 2028, if the 2027 striped bass benchmark results in a change to striped bass reference points or management objectives that could be accommodated within the current NWACS-MICE trade-off analysis framework. Otherwise, the ERP WG recommends only updating the single-species menhaden assessment until the next benchmark.

Managers and stakeholders have expressed strong interest in spatial ERPs for menhaden in the past. The ERP WG recommends convening a workshop with Board members and stakeholders similar to the 2015 Ecosystems Management Objectives workshop to identify the goals and objectives that spatial ERPs should address. This information is needed to develop a timeline for data collection and model development for spatial ERPs, including the next benchmark assessments for ERPs and menhaden.

The ERP WG also recommends that work continues on the ERP models outside the benchmark timeline in order to allow for full development and exploration of these models.

ASMFC managed species (menhaden, striped bass, weakfish) are generally coordinated, and the needs of the ERP assessment are considered when scheduling benchmarks and updates for these species. The assessment schedule for the jointly managed species (spiny dogfish, Atlantic herring, bluefish) is set by the federal Northeast Region Coordinating Council, and while ASMFC input on the timeline is considered, other priorities may rank higher. Despite this, for the last two ERP benchmark assessments, the schedules for the jointly managed species have aligned well.

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 the 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 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 birds. 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 ecological reference points for Atlantic menhaden. However, these reference point methods for Atlantic menhaden are a valuable intermediate step that benefits both managed species and stakeholders. 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 transparently and quantitatively.

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TERMS OF REFERENCE REPORT SUMMARY

TOR 1. Review and evaluate the fishery-dependent and fishery-independent data used in the Atlantic menhaden single-species assessment and the single-species assessments of the other major predator and prey species included in the ERP models, 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). 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 sampled thoroughly, 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 25% of total landings, they did not significantly increase the uncertainty of the overall fishery-dependent data used in the assessment.

Several fishery-independent survey data sets for both young-of-year (YOY) and age-1+ Atlantic menhaden were used in the Atlantic menhaden assessment. The longest-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. The additional YOY time series cover a broad range of states along the coast, and many are two to three decades in length. Similarly, most age-1+ 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. Length information was available for the indices, but because several of the surveys captured Atlantic menhaden outside the range of sizes seen in the fisheries (both larger and smaller), age-length keys developed from the fishery data could not be applied to the survey length frequencies.

The 2019 benchmark single-species assessment for Atlantic menhaden used an estimate of *M* based on the work of Liljestrand et al. (2019b). Liljestrand et al. (2019a) used a Bayesian mark-recovery model to estimate *M* and migration rates from an extensive tagging study conducted from 1966 to 1969 by the National Marine Fisheries Service. During the 2025 benchmark assessment process, Ault et al. (2023) submitted a working paper to the Atlantic menhaden Stock Assessment Subcommittee (SAS) and the Ecological Reference Points Work Group (ERP WG) that re-analyzed the historical tagging data and produced an estimate of *M* that was significantly lower than Liljestrand et al. (2019a). However, Ault et al. (2023) had used a different subset of the data and a different approach to handling key parameters. The SAS formed a working group to consult with the authors and review the data and methods of both papers to understand the differences in the results and determine the best estimate of *M*. The SAS determined that Liljestrand et al. (2019b) had overestimated *M* due to overestimating the reporting rate but did not agree with Ault et al.'s (2023) alternative, stepwise approach to estimating the reporting rate, which caused the significantly lower estimate. The SAS recommended the use of the revised estimate of *M* from the Liljestrand et al. (2019a) model as the base run and the lower estimate of *M* from Ault et al. (2023) be used as a sensitivity run in the single-species assessment update and this ERP benchmark assessment. Full details of this analysis can be found in Working Paper SEDAR 102-WP-01.

The ERP WG also recommended a change to the estimate of *M* used for the weakfish assessment for the EwE models. Weakfish are assessed using a model that estimates a time-varying *M*, which indicates *M* has increased over the time series. The EwE models struggled to balance the inputs for weakfish using the low value of *M* from the assessment in 1985, the start year of the Ecopath models. The ERP WG noted that the weakfish assessment model imposed an upper bound of 1.0 on the estimates of *M,* but Krause et al. (2020) estimated an *M* of 2.33 from tagging data from 2013-2017, the peak of the *M* estimates from the assessment model (ASMFC, 2016). The ERP WG recommended scaling the maximum estimates of *M* from the assessment model to the tag-based *M* from Krause et al. (2020) for the EwE models, which resulted in a higher *M* at the beginning of the time series and better performance from the EwE models.

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, including the new *M* estimates as recommended.

TOR 2. Characterize the 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. Of the key predator and prey species identified for the intermediate complexity models (Atlantic herring, Atlantic striped bass, bluefish, spiny dogfish, and weakfish) most data available through 2023 had been prepared by the TC or SAS responsible for the single-species assessments. The only exceptions were spiny dogfish and bluefish, which had assessments with a terminal year of 2022; preliminary landings and index data, as well as projected biomass and *F* were used for 2023 for these species so that all ERP species had information through 2023. The full ecosystem model included the most recent data for the key predator and prey species but used data outputs from federal stock assessments, time series for abundance or biomass, or data from the older 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. All of the ERP species had peer-reviewed statistical catch-at-age or catch-at-length models that include fishery-dependent and fishery-independent indices of abundance and reliable estimates of total removals. Important predators like marine mammals and prey species such as sand eels 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 species, not birds, marine mammals, or invertebrates. 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 continued the development of three models from the 2020 benchmark to estimate population parameters for menhaden in an ecosystem context. These included 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.

The 2020 iteration of the multispecies statistical catch-at-age model lacked the bottom-up feedback necessary to explore trade-offs between Atlantic menhaden harvest and predator biomass. The ERP WG explored incorporating bottom-up feedback into this model through an empirically derived relationship between striped bass growth and menhaden abundance, but model performance was not satisfactory, and the ERP WG did not consider it ready to provide management advice.

The two EwE models were updated and refined by using a more rigorous model fitting and selections process and exploring the inclusion of primary productivity forcing functions. In addition, the intermediate complexity EwE model incorporated seasonal dynamics in the predator-prey interactions and egg production and used an external vector of recruitment deviations to drive Atlantic herring egg production, to better capture recent recruitment trends for that species.

The performance of the two EwE models was improved over the 2020 benchmark with some of these changes, although the core challenges of calibrating, diagnosing, and identifying the best fit EwE models remained. Overall, both models provided a similar understanding of menhaden's role in the ecosystem, with striped bass, osprey, and nearshore piscivorous birds being the most sensitive species to fishing pressure on menhaden and other species showing weaker negative responses or even positive responses to increasing menhaden *F.*

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 model explored here, the statistical catch at age approach, 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 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 ERP approach was developed to assist with the decision-making process. To illustrate the potential use of the combined single-species assessment and intermediate complexity EwE model, the ERP WG put forward proof-of-concept values of an ERP target and an ERP threshold based the current definition of ERPs adopted by the Atlantic Menhaden Management Board in 2020. 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 fished at the status quo F rates, in this case, the F rate each species experienced in 2023. The ERP fecundity target and threshold were developed from the single-species menhaden model and defined as the long-term equilibrium fecundity that results when the population is fished at the ERP F_{TARGET} and threshold, respectively.

Striped bass was the focal species for this analysis because it was the most sensitive ERP fish species to Atlantic menhaden F, and focusing on one key predator provided a more tractable example for evaluating tradeoffs among management strategies. ERPs based on striped bass biomass are not expected to cause significant changes (relative to status quo levels) for other species that were less sensitive to levels of Atlantic menhaden removals.

The proof-of-concept ERP target and threshold from this assessment were similar to the ERP values currently used in management. The ERP target from this assessment was estimated at F=0.189, compared to the current ERP target of F=0.19, while the ERP threshold from this assessment was estimated at F=0.49, compared to the current ERP threshold of F=0.57. The associated fecundity (*FEC)* target and threshold for this assessment were also lower, with the FEC_{TARGET} equal to 1.67×10^{15} eggs, compared to the current FEC_{TARGET} of 2.0×10^{15} eggs. The $FEC_{\text{THRESHOLD}}$ was estimated at 1.14×10^{15} eggs, compared to the current $FEC_{\text{THRESHOLD}}$ of 1.49×10^{15} eggs.

Short-term projections using the status quo TAC indicated that F will be between the F_{TARGET} and the $F_{THRESHOLD}$ in 2028, with a 0.5% probability that F will be above the ERP $F_{THRESHOLD}$ and a 99.5% probability that it will be above the F_{TARGET}. Further projections based on TACs can be analyzed at the Board's request to determine which achieves the Board's preferred risk level.

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 EwE models are comprised of two modeling 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 59 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. Although the EwE models do not assume an explicit stock-recruitment relationship for multistanza groups, Beverton-Holt style relationships typically emerge from user-defined parameters regulating density dependence and compensation; as a result, the models may overstate the impact of adult mortality on future population abundance for species where recruitment is more environmentally driven.

The multispecies statistical-catch-at-age model assumed that changes in M over time were due to changes in predation mortality from modeled predators (M_2); M_2 was 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. 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 (though an implementation with this capability was explored, the ERP-WG determined it was not ready for management at this time).

Modeling of environmental factors was limited by the poor understanding of the relationship between specific environmental drivers and recruitment and mortality. Although some environmental factors like primary production forcing were explored, none of the models included explicit environmental drivers in their base runs.

None of the models included spatial dynamics; all data were pooled across the model domain area, i.e., the Northwest Atlantic Coast shelf and estuaries. 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 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 productivity of striped bass in particular was very sensitive to the vulnerability parameters of striped bass as prey for spiny dogfish in the intermediate complexity EwE model. The diet data used as input also affected model results, as with the multispecies statistical catch-at-age, especially in identifying the major predators on Atlantic menhaden. Sensitivity runs were used to characterize the uncertainty in these models, but future research is needed to explore propagating parameter uncertainty (especially vulnerabilities, biomasses, production, and diets) through to estimates of population sizes and reference points.

TOR 7. Evaluate stock status for Atlantic menhaden from recommended model(s) as related to the respective reference points (if available).
Under the proof-of-concept ERPs, Atlantic menhaden were not overfished and not experiencing overfishing. The 2023 estimate of F was below the ERP $F_{\text{THRESHOLD}}$ but above the F_{TARGET}, and the 2023 estimate of fecundity was above the $FEC_{\text{THRESHOLD}}$ but below the FEC_{TARGET}.

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.
The EwE models estimated a similar scale and trend in Atlantic menhaden biomass as the single-species assessment, although the single-species model showed more interannual variability than the EwE models, due to the differences in how recruitment was handled between the two models. The EwE models use the estimate of fishing mortality from the single-species assessment directly and so exploitation rates of age-1+ menhaden match across all models. The full complexity EwE model estimated a higher predation mortality (M_2) on both age-0 and age-1+ menhaden than the intermediate complexity model, which is expected, as the intermediate complexity model had fewer predators. Both EwE models estimated an increasing trend in predation mortality for both age-0 and age-1+ menhaden over the time-series, with the age-0 stanza experiencing a stronger increase. The age-0 stanza experienced higher predation mortality overall than the age-1+ stanza for both models.

The single-species model estimates of total mortality for age-0 menhaden were virtually constant over the time-series, as *M* is constant in that model and age-0 menhaden experience negligible fishing pressure. Age-0 *Z* for the EwE models increased from the start of the time-series but stabilized around the early 2000s. For the EwE models and the single-species models, total mortality on age-1+ menhaden decreased somewhat from the start of the series and then was generally stable from the early 2000s onwards, with a slight increase in the most recent years. The *Z* estimates from the EwE models were higher than the *Z* estimates from the single-species model for both stanzas. These differences in total mortality reflect the influence of predation on menhaden in the EwE models.

The proof-of-concept ERP target and threshold developed based on management objectives for striped bass were lower than the single-species F_{TARGET} and $F_{\mathrm{THRESHOLD}}$ previously used in management, reflecting the utility of menhaden as a forage species. The proof-of-concept ERPs were slightly lower than the current ERPs used in management, due to a combination of refinements in the ERP models and the lower estimate of *M* in the single-species menhaden assessment.

TOR 9. If a minority report has been filed, explain the majority reasoning against adopting approach suggested in that report. The minority report should explain the 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. Seasonally and spatially resolved diet and abundance data will be critical to moving towards spatially explicit ERPs. Successfully 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 that 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 the timing of the next benchmark assessment and intermediate updates, if necessary, relative to biology and current management of the species.
The ERP WG noted that there are several factors that will influence the timing of the next update and benchmark assessments for ERPs, making it difficult to recommend a specific timeline. The ERP WG recommends updating the NWACS-MICE model in conjunction with the next menhaden single-species update in 2028, if the 2027 striped bass benchmark results in a change to striped bass reference points or management objectives that could be accommodated within the current NWACS-MICE trade-off analysis framework. Otherwise, the ERP WG recommends only updating the single-species menhaden assessment until the next benchmark.

Managers and stakeholders have expressed strong interest in spatial ERPs for menhaden in the past. The ERP WG recommends convening a workshop with Board members and stakeholders similar to the 2015 Ecosystems Management Objectives workshop to identify the goals and objectives that spatial ERPs should address. This information is needed to develop a timeline for data collection and model development for spatial ERPs, including the next benchmark assessments for ERPs and menhaden.

The ERP WG also recommends that work progress on the ERP models outside the benchmark timeline to allow for full development and exploration of these models.

TERMS OF REFERENCE

For the 2025 ASMFC Atlantic Menhaden

Ecological Reference Point Benchmark Stock Assessment and Peer-Review

Board Approved January 2023

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 the single-species assessments of the other major predator and prey species included in the ERP models, 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 assessment and the single-species assessments of the other major predator and prey species included in the ERP models.

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:
   a. Sensitivity analyses to determine model stability and potential consequences of major model assumptions
   b. 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 & Brandt, 1995) and bluefin tuna (Butler et al., 2010), birds such as bald eagles (Mersmann, 1989) and osprey (Glass & Watts, 2009), and marine mammals like bottlenose dolphin (Gannon & 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 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. During the 2020 benchmark assessment (SEDAR, 2020a, 2020b), 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 to determine the best tool for ecosystem-based management of Atlantic menhaden (Table 1). The ERP WG recommended a hybrid approach combining the current single-species assessment model with an ecosystem model of intermediate complexity to quantitatively evaluate trade-offs between Atlantic menhaden harvest and biomass levels of key managed predators.

The Atlantic Menhaden Management Board accepted the results of SEDAR (SEDAR, 2020b) and began setting the coastwide total allowable catch for Atlantic menhaden in 2021 using the F_{TARGET} values derived from the recommended ERP model – the first time that an ecosystem model provided quantitative advice used in management.

This assessment builds on the results of the 2020 assessment to further develop and refine the models used to provide ecological reference points that support the goals of Atlantic menhaden management. The preferred intermediate complexity ecosystem model was refined to include seasonal dynamics as a proxy for spatial dynamics for a number of the key migratory species. Two supporting models of varying complexity were developed further to provide additional context to the preferred model results. New datasets were considered, and existing datasets were updated and revised. The ERP WG continues to recommend a hybrid approach combining an intermediate complexity ecosystem model to provide long-term strategic ecosystem advice with the single-species menhaden catch-at-age model to provide short-term tactical advice. 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.

There is more work to be done, particularly on the spatial modeling front, to address high priority management concerns, but this assessment represents an important second step on the path of quantitative, ecosystem-based fishery management.

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 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 the abundance of important predator species. 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

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 declared an 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 the first total allowable catch (TAC) limit at 170,800 mt for the commercial fishery beginning in 2013 (ASMFC, 2012) as a result of the finding that the stock was experiencing overfishing.

Amendment 3 (ASMFC, 2017) completely replaced Amendment 2 and, along with Addendum I to Amendment 3 (ASMFC, 2023), currently sets the management program for Atlantic menhaden. While the Amendment continued to manage the stock via single-species biological reference points until the quantitative ERPs were available, the Atlantic Menhaden Management Board has taken ecosystem considerations into its decisions well before the adoption of the ERPs. The Board established a cap on the removals from the reduction fishery within the Chesapeake Bay starting in 2006, before a coastwide TAC was ever implemented, due to concerns about localized depletion of Atlantic menhaden within a key predator nursery ground. The cap was based on historical landings in the Bay with the intent of preventing the expansion of the reduction fishery within the Bay. The cap has remained in place ever since and has been adjusted downwards several times as the coastwide TAC has changed. Additionally, the Board used an ad hoc approach to set the TAC for 2017-2020 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, to provide a qualitative buffer for ecosystem services.

In 2020, the Board accepted the results of the Atlantic Menhaden Single-Species and Ecological Reference Point (ERP) Assessments and Peer Review Reports for management use and approved the following ERPs in the management of Atlantic menhaden:

- ERP target: the maximum fishing mortality rate (*F*) on Atlantic menhaden that sustains Atlantic striped bass at their biomass target when striped bass are fished at their F_{TARGET}
- ERP threshold: the maximum *F* on Atlantic menhaden that keeps Atlantic striped bass at their biomass threshold when striped bass are fished at their F_{TARGET}.
- ERP fecundity target and threshold: the long-term equilibrium fecundity that results when the population is fished at the ERP F_{TARGET} and $F_{THRESHOLD}$, respectively

Under ecological reference point management, the single-species and ERP assessments are paired where the single-species metrics are evaluated against the ERPs to establish the status of the stock in an ecosystem context. The current TAC for the 2023 through 2025 fishing seasons is 233,550 mt, which is an approximate 20% increase from the 2021-2022 TAC based on the positive results of the 2022 Stock Assessment Update (ASMFC, 2022). According to Technical Committee analysis, this increase had less than a 40% probability of exceeding the target set by the ERPs. The Board approved this modest increase to provide additional fishing opportunities while maintaining a conservative risk level of exceeding the ERP target.

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 2020 Benchmark Stock Assessment for Atlantic Menhaden, the ERP WG presented a suite of preliminary ERP models and ecosystem monitoring approaches for feedback (SEDAR, 2020b). The ERP WG used the peer review recommendations from that assessment and the outcomes of the 2015 EMO Workshop (ASMFC, 2015) 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 (EwE) model with limited predator and prey components) in developing ecological reference points. A more complex EwE model was also developed, 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 EwE model was best met 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, 2017b; SEDAR, 2015, 2020a). A statistical catch-at-age model has been used since the 2004 benchmark assessment to provide management advice, the most recent iteration of which is the Beaufort Assessment Model (BAM). 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 (Garrison et al., 2010; NEFSC, 2006). 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 the statistical catch-at-age models 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.

In the 2020 ERP assessment (SEDAR, 2020b), a suite of models was explored, including a time-varying intrinsic growth rate (*r*) surplus production model, a Steele-Henderson surplus production model, a multispecies statistical catch-at-age model (hereafter referred to as Virtual Assessment for the Description of Ecosystem Responses, or VADER), a moderate complexity EwE model based on the Northwest Atlantic Coastal Shelf with a limited predator/prey field (NWACS-MICE), and a full ecosystem EwE model (NWACS-FULL). While all models captured the effect of predation on the Atlantic menhaden population, only the EwE models captured the bottom-up effects of Atlantic menhaden fishing mortality on predator species, which was a key management objective. Ultimately, NWACS-MICE was chosen over the NWACS-FULL model because the NWACS-FULL results suggested that the reduced predator set the NWACS-MICE captured the dynamics of the more responsive predators from the full ecosystem model and was able to best quantify the management trade-offs of different levels of Atlantic menhaden harvest.

The NWACS-MICE model was used to develop ERPs that would allow the most sensitive ERP species, striped bass, to reach its biomass target under long-term equilibrium conditions. For management use, they were used in conjunction with the single-species assessment model, the BAM. The BAM was used to assess whether current *F* rates were at or below the ERPs to determine overfishing status, as well as evaluate the risk of overfishing under different harvest levels in the near term to allow managers to set a TAC.

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.

2.2 Biological Reference Points

Currently, the NWACS-MICE model is used to set the ERP F_{TARGET} and $F_{THRESHOLD}$ values. 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}. The ERP target estimated from the primary model for the 2020 assessment had 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 single-species BAM was used to calculate a fecundity target and threshold based on the F_{TARGET} and $F_{THRESHOLD}$, such that fishing at the ERP F_{TARGET} or $F_{THRESHOLD}$ would result in egg production at the fecundity target or threshold in the long-term.

3 PREDATOR AND PREY SPECIES

3.1 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 Northeast Area Monitoring and Assessment Program (NEAMAP), and the Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP). The NEFSC program has systematically sampled predator food habits since 1973 (Link & 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 are 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 time series 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 from the literature 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. In addition, new state diet data collection programs in ME, RI, and NJ provided important information on prey length and seasonality as well as increased sample size for a number of predators.

3.2 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 an 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 2015 assessment (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; Link & Almeida, 2000) 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 removed 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 remaining predators were considered by the group for inclusion in the models.

These same predators were used for this assessment, but the literature was reviewed for other potential candidates, including marine mammals, specific nearshore piscivorous birds, bluefin tuna, and blue catfish.

The annual Atlantic menhaden consumption (*C*) of each predator was estimated using the methodology from Butler et al. (2010) defined as:

$$CC = BB \times PP \times DDDD \times WW \times TT \qquad (3.1)$$

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 for when predators were in the model domain. Only 2% of the stations fell outside the 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 and 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 an 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.

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 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.

3.3 Changes to Predator and Prey Species from the 2020 Benchmark Assessment

Bluefin Tuna

In previous assessments, the highly migratory species (HMS) predator group was considered a catch-all for all highly migratory pelagics. However, a review of HMS diet studies suggested that bluefin tuna could be a significant predator of Atlantic Menhaden, so the HMS group was replaced with this species. Average diet composition, biomass, and mortality time series, and spatial-seasonal distribution were then explored for this new group.

Diet composition was assessed using six studies comprising 1,800 stomach samples from bluefin tuna across the Gulf of Maine, Virginia, and North Carolina. Prey items were categorized according to NWACS prey categories, and a weighted average diet was calculated using factors based on the proportion of stomachs examined, study area size, and study duration. For biomass and mortality, estimates were derived from a stock synthesis catch-at-age model used in the 2021 ICCAT assessment. Spatial-seasonal distribution was analyzed using pop-up satellite tag (PSAT) data from 395 tagged tuna (2002–2016), and utilization distributions were calculated and mapped by sub-region, excluding land areas. For further details on these methods and their results, see Working Paper SEDAR 102-WP-03.

Blue Catfish

The Work Group assessed the impact of the blue catfish (*Ictalurus furcatus*), an introduced species in Chesapeake Bay, as a potential predator group. Originally native to the central and southern U.S., blue catfish were introduced to Virginia's rivers in the 1970s and 1980s, where they expanded rapidly into more saline waters due to their nature as opportunistic feeders. They consume a wide range of prey, including fish, crabs, and plants, with a notable increase in piscivory during colder months. A literature review of the region showed that larger catfish mainly preyed on species like gizzard shad and white perch, though their impact on menhaden was minimal and seasonal. Despite initial concerns of competing with natural predators for menhaden, the blue catfish's contribution to the predation of menhaden in Chesapeake Bay appears lower than previously thought. Given the species' varied feeding habits and distribution, it was concluded that blue catfish should not be treated as a distinct predator group for this assessment. For more details on the methods used to derive this conclusion, please see Working Paper SEDAR 102-WP-04.

Osprey

In previous assessments, ospreys were considered part of the “nearshore piscivorous bird” functional predator group. For this model update, higher-resolution data from the USGS Breeding Bird Survey (BBS) and the Partners In Flight (PIF) database was used to create an explicit osprey group, breaking them out from the “nearshore piscivorous bird” group in the models that include this group. Note that although the BBS had abundance indices for osprey, the data from southeastern Virginia south of the Chesapeake Bay to North Carolina (within BBS region “S27”) was not included because the data were aggregated with data from the Gulf of Mexico without any way to separate them.

For diet composition, a literature review in Google Scholar showed that most papers from 1970 to 2024 focused on diet data for osprey outside of the model area. However, 12 papers provided diet estimates for the eastern U.S. that fell in or slightly outside (Nova Scotia and Florida) the model area.

For a full outline of the steps used to construct the osprey abundance and biomass time series and the processed used to determine diet composition, please see Working Paper SEDAR 102-WP-07.

Marine Mammals

Similar to the previous assessment, overall lack of data and taxonomic resolution in marine mammal diet data limits the incorporation of marine mammals as predators for multispecies/food web/ecosystem models of Atlantic menhaden. However, in this assessment, two papers were used to gain rough diet composition estimates for 3 predator groups: Mysticetes, Odontocetes, and Pinnipeds.

Kenney et al. (1997) uses CETAP 1982 aerial survey data to estimate seasonal and total abundance, and seasonal and total prey consumption by 18 whale and dolphin groups within the Gulf of Maine, Georges Bank, Southern New England, and the Mid-Atlantic Bight down to Cape Hatteras. A time series could not be developed because the paper only reports the estimated abundance per species for 1982. Consumption rates were only reported in 3 prey groups (fish, squid, and zooplankton).

A paper by Smith et al. (2015) is the only broad, systematic review of marine mammal diets (and consumption rates) for the US Atlantic Coast; note that it also includes some studies outside of the area. Smith et al. (2015) 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 Google Scholar showed no additional research papers (from 2008 to 2024) 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.

Anchovies
For this iteration, a species distribution modeling (SDM) approach was used to estimate the population biomass and spatial-seasonal distribution of Atlantic coast anchovies (*Anchoa spp*.). Using data from eight trawl surveys (1985–2023) and environmental variables from the GLORYS oceanographic model anchovy catch per unit effort (CPUE) was modeled using the R package sdmTMB. Predicted CPUE was converted to biomass density using trawl sweep area and corrected for underestimation using catchability scalars derived from literature estimates in Chesapeake Bay. Final biomass estimates were calculated coastwide at high spatial resolution to find seasonal and interannual trends in anchovy abundance. The best-performing model included distinct survey effects, a seasonally varying non-linear depth effect, a non-linear bottom temperature effect, a circular seasonal effect, and spatial variation by both season and year. For further details on these methods and their results, see Working Paper SEDAR 102-WP-05.

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 the stock structure of Atlantic menhaden on the US East Coast, with a northern and southern stock hypothesized based on meristics and morphometrics (June, 1965; Sutherland, 1963). Based on size-frequency information and tagging studies (Dryfoos et al., 1972; Nicholson, 1971, 1978), the Atlantic menhaden resource is believed to consist of a single unit stock or population. Genetic studies (Anderson, 2007; Lynch et al., 2009) support the single stock hypothesis.

Migration Patterns

There have been several studies examining Atlantic menhaden migration patterns (ASMFC, 2004; Dryfoos et al., 1972; Nicholson, 1978; Roithmayr, 1963). 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 et al., 1991; Berrien & 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 (Nicholson, 1971; Roithmayr, 1963) stated that the majority of Atlantic menhaden in the north migrate south to overwinter in North Carolina, whereas Liljestrand et al. (2019a, 2019b) suggested that about 55% of Atlantic menhaden in the northern region migrate 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 & 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 & Roithmayr, 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 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 (Figure 1).

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 1955-2023 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 (Latour et al., 2023) to address a single-species research recommendation and update fecundity values for use in BAM. Based on the analysis of the study, Latour et al. (2023) 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 (SEDAR, 2020a). 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:

$$AAAA_{aaaa} = DDBBAA * WWTT_{aaaa} * SSAA * PPPP_{aaaa} \tag{4.1}$$

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 et al. (2023) resulted in higher estimated fecundity than in SEDAR 40 (2015). Refer to the single-species assessment Section 2.6 and Appendix 14.1 for more details.

Natural Mortality

For the 2020 single-species benchmark assessment (SEDAR, 2020a), 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 was used, where the Lorenzen (1996) curve of *M*-at-age as a function of weight-at-age was scaled to the tagging estimate of *M* from Liljestrand et al. (2019a, 2019b). For the 2025 single-species assessment update and the ERP benchmark, the estimate of *M* from the tagging model was revised, resulting in a lower estimate of *M*-at-age compared to the 2020 benchmark. The analysis and justification are summarized below; see Working Paper SEDAR 102-WP-01 for more details.

Liljestrand et al. (2019a) used a Bayesian mark-recovery model to estimate *M* and migration rates from an extensive tagging study conducted from 1966 to 1969 by the National Marine Fisheries Service. This study tagged over a million menhaden with internal ferromagnetic tags that were recovered by magnets installed in the reduction plants that processed harvested menhaden. Studies on tag shedding rates, tagging mortality rates, and magnet recovery rates were conducted as part of the tagging study. Liljestrand et al. (2019a) estimated an *M* of 1.16 for the fish tagged in that study. During the 2025 benchmark assessment process, Ault et al. (2023) submitted a working paper to the Atlantic menhaden SAS and ERP WG that re-analyzed the historical tagging data and produced an estimate of *M* (0.56) that was significantly lower than Liljestrand et al. (2019a) but had used a different subset of the data and a different approach to handling key parameters. Ault et al. (2023) is provided as Working Paper SEDAR 102-WP-02. The SAS formed a working group to consult with the authors and review the data and methods of both papers to understand the differences in the results and determine the best estimate of *M*.

The *M* WG determined that Liljestrand et al. (2019a) had overestimated the magnet efficiency rates for the Coston (1971) dataset. The *M* WG developed a revised time series of magnet efficiency for use in the tagging model to correct this issue. In addition, the *M* WG found that the original time series of confidential effort provided to Liljestrand et al. (2019a) could not be recreated by the Southeast Fisheries Science Center, and that the landings data for 1967 had been replaced with 1970 data in the original request. A reproducible time series of confidential effort was developed, and the landings data were corrected.

The *M* WG concluded the stepwise approach to modifying the magnet efficiencies used by Ault et al. (2023) was not appropriate, given how important this parameter (equivalent to the reporting rate in other tagging models) is to the tagging model estimates of *M*. While Ault et al. (2023) were able to fit the observed tag recoveries more closely using the adjusted magnet efficiencies, they did it at the expense of the observed magnet efficiency data. The adjusted magnet efficiency estimates varied widely from month to month but overall were lower than what was observed in the plant test data. This lower efficiency rate is what drove the lower estimate of *M* from the Ault et al. (2023) analysis.

The *M* WG recommended using the estimate of *M* from the Bayesian mark-recovery tagging model using the Coston (1971) data with the revised magnet efficiency estimates and updated confidential effort and landings data as the base run. The *M* WG recommended that the lower estimate of *M* from the Ault et al. (2023) stepwise magnet efficiencies and the confidential effort data be used as a sensitivity run. For both runs, a vector of *M*-at-age for the assessment model was developed by scaling the Lorenzen (1996) curve so that *M* at age-1.5 was equal to the point estimate of *M* from the tagging model, based on the size of tagged fish. This resulted in a vector of *M*-at-age that ranged from 1.39 for age-0 fish to 0.57 for age-6+ fish for the base run, lower than the 2020 benchmark assessment but higher than the values put forth by Ault et al. (2023) (Figure 2).

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 that ranges from North Carolina to Labrador in the Western Atlantic. In US waters, the Georges Bank-Gulf of Maine stock (GOM-GB) are fall spawners that range from NC through the Gulf of Maine and out to Georges Bank. There are two main spawning components for this meta-stock, one centered on Georges Bank

(GB), and the other in coastal portions of the Gulf of Maine (GOM; NEFSC, 2012, 2018; Shepherd et al., 2009).

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 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 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 3).

Maturity and Fecundity

Atlantic herring are 65% mature at age-3, 90% by age-4 and 100% mature by age-5 (Figure 3; NEFSC, 2018). 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, 2018).

Natural Mortality

Atlantic herring are important prey items for a variety of fish, birds, mammals, and other predators (NEFSC, 2018). Some of these predators, such as striped bass and bluefish, are also important predators of menhaden. Despite this, the benchmark assessment for Atlantic herring assumed a 0.35 natural mortality static across age and year (Figure 3) based largely on model diagnostics and a lack of change in consumption by important predators (NEFSC, 2018).

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 at 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, 1989). 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 (Shepard, 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 & Miller, 1967). The extent of the migration that individual striped bass undertake varies depending on the sex, size, and stock of the fish (Callihan et al., 2014; Hill et al., 1989; Secor & Piccoli, 2007).

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 can attain 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 4). 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 & 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 4; 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 4). *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 & 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 & Grothues, 2007; Manderson et al., 2014; O'Connor et al., 2012; Wingate & Secor, 2007). Striped bass are not usually found more than 6 to 8 km offshore (Bain & 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 & Graves, 1996; Juanes et al., 1996). Bluefish in the western North Atlantic are managed as a single stock (NEFSC, 1997; Shepard & Packer, 2006). Genetic data support a unit stock hypothesis (Davidson, 2002; Goodbred & Graves, 1996; Graves, McDowell, Beardsley, et al., 1992). 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 (Able et al., 2003; Kendall & Naplin, 1981; Marks & Conover, 1993). 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 (Klein-MacPhee, 2002; Shepherd et al., 2006; Wilk, 1977).

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 (Barger, 1990; Lassiter, 1962; Robillard et al., 2009; Salerno et al., 2001; Terceiro & Ross, 1993). Bluefish average weight is 5-6 kg at ages 6+ (Figure 5). 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 5; 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

Lorenzen's (1996) method of estimating *M*-at-age as a function of weight-at-age was used in the 2022 benchmark assessment for bluefish, with *M* ranging from 0.85 for age-0 fish to 0.27 for age 6+ fish (Figure 5; NEFSC, 2022b).

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 (Shepard & Packer, 2006). Spring-spawned larvae are subject to advection to northern waters by the Gulf Stream (G. R. Shepard & Packer, 2006). Adult and juvenile bluefish are found primarily in waters less than 20 m deep along the Atlantic coast (Shepard & 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 & Biagi, 1991; G. R. Shepard & 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 (Klein-MacPhee, 2002; Olla & Studholme, 1971). Temperature and photoperiod are the principal factors directing activity, migrations, and distribution of adult bluefish (Olla & 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). Nammack et al. (1985) estimated L_∞ from age data at 100.5 cm for females. However, preliminary analyses of mark-recapture data from fish tagged between 2011 and 2012 estimated a lower L_∞ and *k* for females, but not males (NEFSC, 2022).

Maturity and Fecundity

Spiny dogfish mature late and have low fecundity. 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). Sosebee (2022) found estimated length at 50% maturity for females from NEFSC survey data had declined in recent years, with $L_{50\%}$=79.6 cm in the 1998-2011 period and $L_{50\%}$=73.6 cm for 2012-2021. This is consistent with the lower estimates of female L_∞ from more recent tagging data, suggesting a change in the growth and maturity rates of females in recent years, potentially due to environmental changes or fishing pressure (NEFSC, 2022).

Natural Mortality

The 2022 benchmark assessment for spiny dogfish (NEFSC, 2022) used the Lorenzen (1996) relationship to calculate an age-varying *M*, with the average scaled to 0.102, the Then et al. (2015) maximum age estimator based on a maximum age of 50 years (Figure 6).

Habitat

Spiny dogfish are predominately epibenthic species, with no known associations to any particular substrate, submerged aquatic vegetation, or any other structural habitat (McMillan & 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°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 (Cordes & Graves, 2003; Graves et. al, 1992), 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 (1984) observed that migrations occur in conjunction with movements of the 16-24° 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., 2015).

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; Lowerre-Barbieri et al., 1995; Villoso, 1990). Weakfish in the catch averaged 5-6 kg at the oldest ages (Figure 7).

Maturity and Fecundity

Weakfish mature early, with 90-97% of age-1 fish estimated to be mature (Figure 7; 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 (Lowerre-Barbieri et al., 1996; Shepherd & Grimes, 1984).

Natural Mortality

Recent assessments of weakfish indicate natural mortality has increased over time (ASMFC, 2016; NEFSC, 2009). 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 0.90-0.95 from 2007-2019 (Figure 7). Krause et al. (2020) used an integrated tagging model to estimate M and F from acoustically tagged weakfish from 2013-2017 and estimated M at 2.33. Their estimate of Z was very similar to the Z from ASMFC (2019) during this time period, but they noted that the Bayesian model used in the assessment imposed a maximum upper bound on M of 1.0 during the estimation process, which may be causing the discrepancy in M results. There are several hypotheses about what caused the increase in M, including increasing predation or competition from increasing striped bass, spiny dogfish, and bottlenose dolphin populations and large-scale environmental drivers like Atlantic Multidecadal Oscillation, but no definitive conclusions can be made (Krause et al., 2020; NEFSC, 2009).

Based on the results of Krause et al. (2020) and performance issues noted during the initial ERP model configuration process, the ERP WG elected to scale the annual M time-series from the assessment model so that the estimate of M for 2013-2017 was equal to the tag-based M from Krause et al. (2020) for the EwE models. This resulted in a higher M at the beginning of the time series and better performance from the EwE models.

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 since then have averaged approximately 130,000 mt per year, the lowest period 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 2023, coastwide landings were comprised of 73% from the reduction fishery and 27% from the bait fishery. Recreational removals comprised less than 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. The average weight of released alive Atlantic menhaden was assumed equal to the average weight of the recreationally harvested fish to convert numbers of fish released alive to weight. 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 738,000 mt in 1956 to a time series low of 169,000 mt in 2013. Total removals rebounded slightly after that, with total removals in 2023 at 182,000 metric tons (Figure 8).

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, 2018). 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 485,830 mt in 1968, before declining less than 10% of that in 1983. Total removals were mostly stable from 1990 – 2010, averaging 115,000 mt, but have been steadily decreasing since 2013. With the finding that the stock was overfished and the implementation of a rebuilding plan in 2019, landings were reduced from 58,563 metric tons in 2018 to 18,613 mt in 2019 and were 10,441 mt in 2023 (Figure 9).

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 10). 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, due to a combination of stock declines and management action in response to those declines. Total removals were 12,758 mt in 2020 but rebounded in 2022 to 27,366 mt as the strong 2015 year-class became available to the ocean fishery. Removals have begun to decline again as that year-class moves out of the ocean slot, with a series of weak year-classes following it in recent years.

5.5 Bluefish

Bluefish is a predominately recreational species, with recreational removals making up about 88% of the total removals in recent years. It is assumed that 9.4% 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 9.4% of the recreational live releases. The proportion of bluefish released alive has increased over the time series from 19% in 1985 to over 80% in 2021. 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 MRIP observers on head boat trips as well as citizen science programs like the ALS and SC DNR volunteer tagging programs and volunteer angler logbook programs in Rhode Island, Connecticut, and New Jersey. Sample size was significantly lower for the southern range of bluefish compared to the northern end of the range, which is a source of uncertainty as the size range of bluefish varies along the coast.

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 83,738 mt in 1987; by the mid-1990s landings had declined to just under 20,000 mt, and remained relatively stable after that, averaging 22,254 mt from 1995 – 2017 (Figure 11). Removals from 2018-2023 have been lower, averaging 8,372 mt, due to a combination of management action and fishery performance.

5.6 Spiny Dogfish

Commercial fishermen catch spiny dogfish using longlines, trawls, and purse seines. Commercial landings are comprised of about 98% female spiny dogfish (Sosebee & Rago, 2017), as 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 & Rago, 2017). Canadian and distant water landings were obtained from the NAFO catch statistics database (Sosebee & 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, total removals were high in the 1990s, peaking at 52,380 mt in 1992 (Figure 12). 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 7,248 mt in 2003. As the stock began to improve, landings began to increase again, reaching 18,242 mt in 2014, but have declined in recent years due to a combination of management action and market factors (Figure 12), with total removals averaging 8,344 mt for 2021-2023.

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 by using lengths from the MRIP sampling of released fish on head boat 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 removals have declined significantly over the assessment time series; total removals in 2011 were 180 mt, less than 1% of their 1986 value of 19,515 mt (Figure 13). However, commercial landings and recreational catch has been increasing somewhat in recent years. Total removals in 2023 were 484 mt, still well below the height of the fishery.

6 ATLANTIC MENHADEN FISHERY-INDEPENDENT INDICES OF ABUNDANCE

6.1 Background of Analysis and Model Description

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. SEDAR (SEDAR, 2020a) conducted a principal components analysis (PCA) to evaluate which surveys had similar length compositions and used that to determine which age-1+ surveys should be combined. The Conn method was also used to combine individual abundance indices into regional indices in past menhaden single-species stock assessments in SEDAR (2015), ASMFC (2017b), and SEDAR (2020a).

6.2 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 (SEDAR, 2020a) for full methods for the indices of relative abundance in numbers to support the BAM and MSSCAA models.

The coastwide YOY index of relative abundance for Atlantic menhaden indicated high abundance in the 1970s and 1980s, with declines through the 1990s (Figure 14). YOY abundance remained low, but there was a slight increase in the terminal years of 2021-2023. The NAD index predicted variable abundance throughout the time series with high abundance occurring in the recent years of 2019-2022 before declining again in the terminal year of 2023 (Figure 14). The MAD index predicted high abundance in the beginning of the time series followed by a lower but variable abundance through the late 1990s-early 2010s (Figure 14). Abundance in the Mid-Atlantic region began to increase in the mid-2010s but then decreased and was variable through the terminal years, with 2020 representing a time series low. Abundance increased in 2021-2022 but declined in 2023. The SAD index predicted high abundance in 1990 followed by low abundance through the mid-2000s (Figure 14). The index peaked again in 2006 but then decreased and was variable through the terminal year. The terminal years of 2022-2023 both indicated relatively low abundance in the region.

Menhaden caught in the fishery-independent surveys are not currently aged, and because the surveys catch larger menhaden than the fisheries do and the large variability in size at age, it has not been possible to apply an age-length key based on the fishery-dependent samples to develop age composition for the surveys. Therefore, the BAM fit to index length composition data instead of age composition data. Length compositions were developed by combining data from each of the surveys and weighting the data by the inverse of the squared sigma values outputted from the Conn method.

7 NON-MENHADEN INDICES OF ABUNDANCE

The single-species assessments for all these species use multiple (often 5 or more) indices of relative abundance. 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, 2018). 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, 2018).

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 9). Both the NEFSC Fall Bigelow and the Summer Shrimp Survey showed a sharp decline after 2015-2016 and have stayed at low levels since then (Figure 16).

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, 2015, and 2018 (Figure 10).

For age-1+ indices, the Striped Bass TC recommended the Connecticut Long Island Sound Trawl Survey (CT LISTS) and the MRIP CPUE index. Both 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 indices showed similar trends, starting out low at the beginning of the time series and increasing through the 1990s (Figure 10). They peaked around the early 2000s and have been gradually declining since, remaining variable around a lower level since the mid-2010s. The MD SSN has varied without trend over that time period (Figure 10); 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 11).

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 (2022b).

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 11).

7.4 Spiny Dogfish

The NEFSC calculates a biomass estimate for spiny dogfish based on area swept from their spring bottom trawl survey (Figure 12) which was used as an index of abundance in the most recent assessment (NEFSC, 2022a). The vessel change from the R/V Albatross to the R/V Bigelow in 2009 has been accounted for with the calibration factors from Miller et al. (2010) to create a single time-series from 1982-2022; the survey was not successfully completed in 2014, 2020, or 2023. The index has been variable over time, with an increasing trend since the mid-2000s.

In the past, the NEFSC has calculated a recruitment index from this survey as well, based on length cut-offs, but that index was not updated for the most recent assessment (NEFSC, 2022a), as the full-length composition was input into the model instead.

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 13).

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 30 ft 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 head boat 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 generally showed similar trends, increasing from the late 1980s through the mid-1990s before declining to low levels (Figure 13). However, the MRIP CPUE has shown an increase in recent years, while the DE 30 ft Trawl has not. The NC PSIGNS index and NJ OT were variable without a strong trend over its time-series, but both have shown a few strong peaks in recent years (Figure 13).

8 NON-MENHADEN 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. Atlantic menhaden, striped bass, and weakfish had assessment updates with a terminal year of 2023, although the weakfish assessment update results were preliminary at the time the data were made available to the ERP WG. Bluefish and spiny dogfish had assessment updates with a terminal year of 2022; projections were used to provide estimates of biomass and *F* in 2023, based on preliminary removals for 2023.

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 as described below, 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 Section 10.2 and Section 11.2.2.3 below for why this was necessary and how these values were used). Note that there are differences in terminology between ASMFC-assessed species (Atlantic menhaden, striped bass, weakfish) and federally assessed species (bluefish, spiny dogfish) in how targets and thresholds are defined. For ASMFC-assessed species, *F* targets and SSB targets are related so that fishing at the F_{TARGET} in the long term will cause the stock to stabilize at the SSB_{TARGET}, and the same with the *F* thresholds and SSB thresholds. For federally managed species, no F_{TARGET} is defined, and the F_{MSY} proxy is used as the $F_{THRESHOLD}$, so fishing at the $F_{THRESHOLD}$ in the long-term will result in the population stabilizing at the SSB_{TARGET} (the SSB_{MSY} proxy). The $SSB_{THRESHOLD}$ is 50% of the SSB_{TARGET} value. To drive the long term EwE projections appropriately, the F_{TARGET} for each species was defined as the *F* rate that would allow the population to stabilize at the SSB target in the long-term. Reference points, *B* equivalents, and *B* and *F* scalars are shown in Table 3 and Table 4.

8.1 Atlantic Herring

Atlantic herring are assessed with a statistical catch-at-age model, the ASAP program from the NEFSC Toolbox. According to the 2024 update assessment (NEFSC, 2024), Atlantic herring were overfished, but overfishing was not occurring in 2023, 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 2,242,869 mt in 1967 to a low of 107,229 mt in 2021 (Figure 15). Total biomass in 2023 was 136,256 mt. SSB showed a similar pattern, ranging from a high of 1,338,318 mt in 1967 to a low of 46,825 mt in 2019 (Figure 15). It was determined that a retrospective adjustment was needed for the terminal year of the assessment; this scaled the model-estimated 2023 value of 74,977 mt down to 47,955 mt, which is below the $SSB_{THRESHOLD}$ of 93,184 mt. The decline in total biomass and SSB in recent years has been attributed to a prolonged period of low recruitment (NEFSC, 2024).

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 0.07 in 1965 to a high of 0.90 in 1975 (Figure 15). A retrospective adjustment was also applied to *F* in the terminal year, which scaled the model-estimate average *F* in 2023 from 0.19 to 0.26, below the $F_{THRESHOLD}$ of 0.45.

The Atlantic herring assessment used for the 2025 ERP assessment is an update of the same model used in the 2020 ERP assessment; as a result, the trend and scale of total biomass and average *F* are similar between the two assessments (Figure 16). A Research Track assessment for Atlantic herring was completed in 2025, too late to be incorporated into this assessment, which used a state-space model (the Woods Hole Assessment Model, WHAM; NEFSC, 2025). The results were similar in terms of scale, but WHAM attributed some of the recent declines in Atlantic herring to deviations in survival from one age-class instead of solely low recruitment in recent years.

8.2 Striped Bass

Striped bass are assessed with a statistical catch-at-age (SCA) model. According to the 2024 assessment update (ASMFC, 2024), Atlantic striped bass were overfished, but overfishing was not occurring in 2023, 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 under the current low recruitment regime. The estimate of age-1+ 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 36,012 mt in 1982 to a peak of 341,690 mt in 1999 before beginning to decline (Figure 17). Total biomass was 151,582 mt in 2015 but has been increasing in recent years due to a combination of management action to reduce *F* and the recruitment of strong year-classes in 2011, 2015, and 2018. Total biomass was 201,445 in 2023. Female SSB showed a similar trend, starting out at low levels and increasing steadily through the late-1980s and 1990s, but peaking later than total biomass at 118,927 mt in 2003 before beginning to gradually decline; the decline became sharper in 2012 (Figure 17). Female SSB has been increasing since 2018, as the strong year-classes that added to total biomass began maturing. Female SSB was estimated at 86,535 mt in 2023, below both the $SSB_{THRESHOLD}$ of 89,513 mt and the SSB_{TARGET} of 111,892 mt.

Total *F* increased for both the ocean fleet and the Chesapeake Bay fleet beginning 1990 and has been above the threshold for much of the time series. Total *F* declined after 2017 due to a combination of management action and declines in exploitable biomass. Total *F* in 2023 was 0.18, above the F_{TARGET} of 0.17 but below the $F_{THRESHOLD}$ of 0.21 (Figure 17).

The Atlantic striped bass assessment used for the 2025 ERP assessment is an update of the same model used in the 2020 ERP assessment; as a result, the trend and scale of total biomass and average *F* are very similar between the two assessments (Figure 18).

8.3 Bluefish

Bluefish are assessed with a state-space catch-at-age model, the WHAM program from the NEFSC Toolbox. In the 2023 update, the assessment indicated bluefish were not overfished and overfishing was not occurring. The SSB_{TARGET} (the B_{MSY} proxy) is calculated by using AgePro to project the population forward under $F=F_{35\%SPR}=F_{THRESHOLD}$ 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 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 declined from a high of 323,807 mt at the beginning of the time series until the mid-1990s before beginning to increase for a brief period of time; total biomass trended downward again after the mid-2000s, but has been increasing somewhat in recent years (Figure 19). The estimate of total biomass in 2023 was 78,751 mt. SSB has shown a similar trend, with the estimate of SSB in 2023 at 52,747 mt, above the $SSB_{THRESHOLD}$ but below the SSB_{TARGET} (Figure 19).

F is reported as *F* at age 2, the age of full selectivity for bluefish. *F* has been above the $F_{THRESHOLD}$ for almost the entire time series, until it dropped from a time-series high of 0.50 in 2017 to below the threshold of 0.24 in 2018, after which it continued to decline. The estimate of *F* in 2023 was 0.15, below the $F_{THRESHOLD}$.

The bluefish assessment underwent a benchmark assessment in 2022, between the 2020 and 2025 ERP assessments (NEFSC, 2022b). Major changes included switching from ASAP to WHAM and using an age-varying estimate of *M* instead of the previous age-constant value, as well as changes to how information on recreational releases were handled. However, the trend and scale of total biomass and full *F* are generally similar between the two assessments (Figure 20).

8.4 Spiny Dogfish

Spiny dogfish are assessed using a length-structured Stock Synthesis model. Based on the 2023 assessment update, spiny dogfish were not overfished and overfishing was not occurring in 2018 (NEFSC, 2023). The $F_{THRESHOLD}$ is defined as $F_{60\%SPR}$, reflecting spiny dogfish's low productivity. The SSB_{TARGET} is the expected spawning output at $F_{60\%SPR}$ and the $SSB_{THRESHOLD}$ is one-half the SSB_{TARGET}. The ratio of total biomass at $F_{60\%SPR}$ to initial biomass was applied to the Stock Synthesis estimates of initial biomass to convert the spawning output reference points into a total biomass target and threshold for the ERP models that use total biomass.

Estimates of total biomass have been mostly flat over the ERP assessment time-series, showing a decline through the late 1980s before stabilizing (Figure 21). Spawning output showed a similar trend, with a more pronounced decline through the late 1990s followed by a more pronounced increase from 2000-2010; spawning output has been declining since 2011 (Figure 21). Spawning output in 2022 was estimated at 191 million pups, just above the SSB_{TARGET} of 188 million pups.

F is reported as the *F* on ages 12+, the fully selected age-classes. *F* peaked in the mid-1990s and has generally been declining since then, with the last 20 years of *F* hovering around the $F_{THRESHOLD}$ (Figure 21). *F* was 0.02 in 2023, just below the $F_{THRESHOLD}$ of 0.025.

The spiny dogfish assessment underwent a benchmark assessment in 2022, between the 2020 and 2025 ERP assessments (NEFSC, 2022a). The major change was switching from an area-swept biomass approach to a length-structured Stock Synthesis model. As a result, there were significant changes in both the scale and the trend of estimates of total biomass and fishing mortality between the spiny dogfish assessment used for the 2020 ERP assessment and the one used in the 2025 ERP assessment. Estimates of total biomass were higher and estimates of *F* were lower for the new assessment approach, and the variability in both metrics was reduced (Figure 22).

8.5 Weakfish

Weakfish are assessed using a Bayesian statistical catch-at-age model that estimates a time-varying natural mortality rate. Although fishing mortality rates have been low in recent years, the increasing trend in natural mortality has prevented stock recovery, with weakfish being considered depleted (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.

As noted in the life history section above, the Bayesian model used an upper bound of 1.0 on the time-varying estimates of *M,* but empirical tagging data estimated a much higher estimate of *M* for 2013-2017 (Krause et al., 2020). The ERP WG elected to scale the time-series of *M* estimated by the model so that the estimates for 2013-2017 were equal to the estimate of *M* from the tagging study. An ASAP model was configured using the same input data as the Bayesian model, but with the time-varying *M* fixed as input from the Bayesian model. The model was configured so that it produced similar results in *F* and biomass to the Bayesian model when using the *M* estimated by that model, with the upper bound of 1.0, and then rerun with the input vector scaled to the *M* estimated by Krause et al. (2020), which was converted to an age-and time-varying matrix by scaling the Lorenzen (1996) curve such that the *M* on age-2.5 was equal to the annual *M* each year, based on the age of fish reported tagged by Krause et al. (2020), in order to provide more accurate estimates of *M* by stanza for the EwE models. Reference points were not recalculated for the revised ASAP run, and the ratio of biomass to the biomass target and *F* to the F_{TARGET} from the run with the original scale of *M* were used in the EwE projections.

The preliminary update with the capped *M* indicated that total age-1+ biomass of weakfish has declined since the beginning of the time series, with a brief rebound in the mid-1990s (Figure 23). Total biomass has been increasing somewhat in recent years. Spawning stock biomass showed very similar trends to age-1+ biomass, since weakfish are 90% mature at age-1 (Figure 23). The preliminary estimate of SSB in 2023 was below the $SSB_{THRESHOLD}$.

Full *F* for weakfish declined through the early 1990s before increasing again; *F* spiked in 2008 but has been below average since then (Figure 23). *F* in 2023 was 0.22, below the $F_{THRESHOLD}$ of 0.76.

The weakfish assessment with the modified *M* estimated higher biomass and recruitment and lower *F* values over the time-series compared to the assessment used in SEDAR (SEDAR, 2020b; Figure 24).

9 BEAUFORT ASSESSMENT MODEL (BAM)

9.1 Description and Configuration

The Beaufort Assessment Model (BAM) has been used to assess Atlantic menhaden since 2010 (ASMFC, 2010; SEDAR, 2015, 2020a). 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 in both fishing effort and the size structure of the population available in each area. The base run of the model included the revised vector of *M*-at-age.

Uncertainty is characterized using a Monte Carlo bootstrap (MCB) approach, where realizations of *M* and fecundity-at-age are drawn from distributions based on the uncertainty in those parameters and the model is refit. The output of these MCB runs is used in the projections to evaluate the impact of different levels of total allowable catch (TAC) in the short term. Instead of assuming a static median value for recruitment, as is done for many assessment projection methodologies, recruitment was projected using nonlinear time series analysis methods (Deyle et al., 2018). Specifically, projections were based on the MCB runs, which allows recruitment to change from year to year in the projections based on how recruitment has changed in the past under similar conditions.

9.2 Results

BAM estimates of age-1+ biomass have fluctuated over time from an estimated high of over 3,308,000 mt in 1959 to a low of 863,000 mt in 1973. From 1980 to the present, biomass has been increasing or stable 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 or relatively stable since approximately 1990.

Stock status for Atlantic menhaden is determined by the ERP reference points adopted by the Board in 2020. The ERP F_{TARGET} and threshold were developed through the 2020 ERP benchmark assessment. The single-species BAM was used to calculate a fecundity target and threshold based on the ERP F_{TARGET} and $F_{\mathrm{THRESHOLD}}$, such that fishing at the ERP F_{TARGET} or $F_{\mathrm{THRESHOLD}}$ would result in egg production at the fecundity target or threshold in the long-term.

With the change in *M* in the base run, the estimates of *F* and fecundity from the BAM are no longer comparable to these reference points, as the ERP F_{TARGET} and $F_{\mathrm{THRESHOLD}}$ were developed using the BAM run with the higher estimate of *M.* For stock status and projection results using the proof-of-concept ERPs developed for this assessment, see Section 13.3 below.

For more detailed information on the BAM configuration and results, see the single-species assessment report (Appendix I) and SEDAR (2020a).

10 INTERMEDIATE COMPLEXITY ECOPATH WITH ECOSIM MODEL (NWACS-MICE)

10.1 Model Overview

The Northwest Atlantic Continental Shelf Model of Intermediate Complexity for Ecosystem assessment (NWACS-MICE) was previously used to develop ecological reference points (ERPs) for Atlantic menhaden (Chagaris et al., 2020; SEDAR, 2020b). No additional species were added to the existing model during this updated assessment, but the inputs were revised using recent stock assessments, additional data sources, and new analytical procedures.

The NWACS-MICE model was developed using the Ecopath with Ecosim (EwE) software. EwE is a trophic modeling framework designed to integrate and manage biomass and food web data across entire ecosystems. It has been extensively applied in aquatic ecosystem research and fisheries management (Christensen & Walters, 2004; Colléter et al., 2015; Pauly et al., 2000). The Ecopath module within EwE provides a mass-balanced, static snapshot of the ecosystem, representing energy flow among functional groups and species, and incorporates age structure when needed. It establishes the baseline conditions from which dynamic simulations in Ecosim are initiated. Within Ecopath, each group's production is distributed among mortality sources such as fishing, predation, natural causes, and migration, ensuring mass balance throughout the food web. Ecosim, the dynamic simulation component of EwE, operates on a monthly time step and simulates biomass trajectories over time using a system of differential equations. These equations capture gains through consumption and losses through predation, fishing, natural mortality, and emigration (Walters et al., 1997). More information on the foundational methods and assumptions of EwE can be found in Walters et al. (1997, 2000, 2005) and Christensen and Walters (2004).

In selecting an appropriate modeling framework for menhaden ERPs, it was essential to use a modeling system capable of capturing both the top-down control exerted by predators on their prey and the bottom-up influence menhaden exert on predator populations. Ecosim models these interactions using foraging arena theory, which posits that prey become available to predators only within specific spatial and temporal contexts. The transfer of prey biomass from an inaccessible to an accessible state is governed by the vulnerability exchange rate parameters V_{ij} which determine the rate at which prey i becomes susceptible to consumption by predator j (Ahrens et al., 2012; Bentley et al., 2024). These V_{ij} parameters play a key role in regulating predation mortality. Low values for a predator imply limited accessibility to their prey, which dampens predation pressure and constrains predator growth. Conversely, higher V_{ij} result in more pronounced top-down effects by facilitating a rapid flow of prey into the vulnerable pool, allowing predators to exert greater control over prey populations. Additional parameters in Ecosim refine these dynamics further. The foraging time adjustment (FTA) mechanism accounts for changes in feeding behavior based on density-dependence and predator risk, effectively moderating exposure to predation and balancing growth against survival. The prey switching parameter is also important here because it enables predators to adjust their search rates disproportionately in response to fluctuations in prey availability. This is modeled through a power function that modifies the search rate a_{ij} over time t, depending on changes in prey biomass B_i and a switching exponent P_j for predator j. Values of P_j closer to 2 reflect a highly responsive switching behavior, whereas values near 0 imply fixed preferences. In our case, incorporating prey switching allows the model to account for the possibility that predators might compensate for changes in menhaden availability by targeting alternative prey.

10.2 Initial conditions (Ecopath)

The NWACS-MICE model consists of 17 functional groups, species, or age stanzas and includes striped bass (ages 0, 2-5, and 6+), Atlantic menhaden (ages 0 and 1+), spiny dogfish, bluefish (ages 0 and 1+), weakfish (ages 0 and 1+), Atlantic herring (ages 0 and 2+), anchovies, benthic invertebrates, zooplankton, phytoplankton and detritus. Striped bass, menhaden, dogfish, bluefish, weakfish, and herring make up the six "ERP species." A single fishing fleet was created for each of the ERP species so that fishing mortality rates could be specified separately. As such, all landings and dead discards are combined to produce a single F, and each fleet only catches its own target species. Biomass, mortality, landings, and diet inputs were updated or revised for all model groups, while the consumption rates remained unchanged from the previous version of the model. The Ecopath model base year is 1985 and the biomass and catch units are in mt/km^2, with a model area of 441,000 km^2. The model area was calculated from the ¼ degree depth raster downloaded from the ETOPO 2022 database on the NOAA server, excluding all land and grid cells with depth > 2000m, and using the area() function in the R raster package.

10.2.1 ERP Species Inputs

Biomass, mortality, landings, growth, and maturity parameters for the ERP species were obtained directly from the single species stock assessment model output files. All six ERP species were recently assessed with a terminal year of 2023 for herring, menhaden, striped bass, and weakfish, and 2022 for spiny dogfish and bluefish (Table 5). A separate R-script was written for each species that reads in the stock assessment model report file, derives Ecopath inputs, and extracts timeseries information for Ecosim. The scripts are intended to reduce errors, maintain consistency across species, enhance reproducibility, and make future updates easier. They are stored and maintained on a GitHub repository that can be made available as needed.

In going from age-structured stock assessment output to Ecopath inputs, the appropriate biomass value to use is the mean, or mid-year estimate, as this allows for the Ecopath fishing mortality to be instantaneous when annual landings are included. Atlantic menhaden and spiny dogfish were the only species with assessment models that included estimates of mid-year biomass at age. Striped bass reported January-1 biomass-at-age while all others contained number- and weight-at-age matrices to calculate biomass-at-age. January-1 biomass was adjusted to mean (or mid-year) biomass as $\bar{B}_{a,y} = B_{a,y}(1 - \exp(-Z_{a,y}))/Z_{a,y}$. Mean biomass at age was then summed into Ecopath age stanzas and converted to units of mt/km^2 for inputs to NWACS-MICE, both as Ecopath base year inputs (for 1985) and as timeseries for Ecosim.

Catch estimates from the stock assessment models were aggregated across fleets, partitioned across ages, and summed for landings and dead discards prior to input to NWACS-MICE. Atlantic menhaden was the only species with landings-at-age in units of weight (mt) that could easily be summed into age stanzas. Atlantic herring and weakfish contained total catch in weight that was converted to age using composition matrices; bluefish and striped bass reported total catch in numbers that was proportioned across ages using composition matrices and converted to weight; and spiny dogfish reported catch-at-age in numbers that was converted to weight. The catch-at-age matrices included all fishery removals (landings plus dead discards) and these were summed into age stanzas for each year as inputs to Ecopath (for base year 1985) and Ecosim timeseries.

Natural mortality was assumed to be age-varying in all ERP species' stock assessments except Atlantic herring. For those with age-varying *M* (usually based on the Lorenzen equations), the NWACS-MICE age stanza *M* was calculated as the average across ages in the stanza, weighted by the numbers at age. Fishing mortality for each stanza was calculated by dividing catch by biomass. The sum of *F* and *M* was then input as *Z* for the NWACS-MICE ecosystem model.

10.2.2 Anchovies

SEDAR 102-WP-05 describes the estimation of coastwide biomass using a species distribution model (SDM). Briefly, the SDM utilized 59,119 observations of anchovy CPUE from 8 different trawl surveys. The R package sdmTMB was used to construct species distribution models that accounted for spatial patterns via Gaussian Markov random fields. The model form that achieved the lowest BIC included a survey effect (with NEFSC split into separate Albatross and Bigelow series), a non-linear depth effect that varied seasonally, a non-linear bottom temperature effect, and a circular seasonal effect. In addition, the best fitting model also had a spatial pattern that varied seasonally (by 2-month block), as well as overtime (according to an AR1 process).

Continuous rasters of depth (GEBCO digital elevation model) and monthly average bottom temperature (GLORYS oceanographic model) were used to inform SDM predictions of anchovy CPUE across the entire model domain at 5 km x 5 km resolution in the units of the NEFSC-Albatross survey, for each month and year, 1985-2023. CPUE was converted to biomass density using a catchability scalar to obtain coastwide estimates of anchovy biomass. Likewise, the time series of annual biomass estimates was estimated by averaging the coastwide estimates across all months, by year. Average annual coastwide anchovy biomass predicted by the SDM generally varied between 200,000 and 800,000 mt over the period 1985-2023 (Figure 25). The 1985 estimate of coastwide anchovy biomass was 280,189 mt and was converted to 0.6353 mt/km2 for entry to NWACS-MICE. Production (P/B) and consumption (Q/B) rates were 2.2 and 7.3, respectively, the same values used in the previous model.

10.2.3 Benthic Invertebrates

Benthic invertebrate biomass was estimated from data archived on the NOAA National Benthic Inventory (NBI) repository (https://products.coastalscience.noaa.gov/nbi/). Ten benthic invertebrate surveys from North Carolina to Massachusetts were used to estimate benthic invertebrate biomass (Table 6, Figure 26). A total of 3,504 benthic grab samples containing abundance and standardized density (n/m^2) from 1,951 different invertebrate taxa. A list of invertebrate genera from the NBI datasets was submitted to the Microsoft Copilot AI tool (using OpenAI's GPT models) to provide mean body mass estimates. Genus body mass was averaged for higher level taxonomic classifications and used to convert numerical densities to biomass densities (g/m^2). For each collection, the biomass density was summed across taxa and the total invertebrate biomass was averaged across all collections. The average biomass density for benthic invertebrates from the NBI datasets was 51.91 mt/km^2. Production (P/B) and consumption (Q/B) rates were unchanged from the previous model at 2.432 and 12.47, respectively (SEDAR, 2020b).

10.2.4 Zooplankton

SEDAR 102-WP-06 describes the calculations used to estimate zooplankton biomass and timeseries for the NWACS model domain. Seasonal zooplankton data were available from 1985-2023 for the entire model domain from the NOAA NEFSC plankton cruises. Zooplankton were first grouped into microzooplankton, small copepods, large copepods, gelatinous zooplankton, and micronekton. Numerical densities were converted to total weights based on sizes of representative species within each zooplankton category. Biomass was extrapolated across four large geographic regions (Mid-Atlantic Bight, Georges, Gulf of Maine, and Scotian Shelf), summed, and then divided by the total model area. The 1985 estimate of total zooplankton biomass was 10.8 million mt and was input to NWACS-MICE as 24.47 mt/km^2. Production (P/B) and consumption (Q/B) rates were unchanged from the previous model at 45.85 and 154.6, respectively.

10.2.5 Phytoplankton

Phytoplankton biomass was estimated from the long-term (1993-2023) average of monthly phytoplankton carbon (phC) estimates from the GLORYS oceanographic dataset. For each month and grid cell, phC was summed across depth layers and then averaged over modeled grid cells to generate a monthly timeseries. The average monthly phC was 50.95 mmolC m^{-2}, which was converted to g C m^{-2} using the carbon atomic mass of 12.011, converted to dry mass (x5.5), and the wet weight (x7.7). The estimated phytoplankton biomass input to the NWACS Ecopath model was 25.45 mt km^{-2}. The production rate for phytoplankton was unchanged from the previous NWACS-MICE model at 186.4 mt/km^2/yr.

10.2.6 Detritus

Detritus biomass was increased from 12.97 mt/km^2 in the previous NWACS-MICE model to 29.42 in the current version.

10.2.7 Diet Matrix

Diet data were obtained from the NOAA Northeast Fisheries Science Center (NEFSC), the New Jersey Ocean Trawl Survey (NJOT), the Northeast Area Monitoring and Assessment Program (NEAMAP), the Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP), the Rhode Island Department of Environmental Management (RIDEM), and a literature database assembled during earlier MSVPA efforts (Table 7). The NJOT, NEAMAP, ChesMMAP, and RIDEM diet datasets contained data on individual stomachs, while the NEFSC data were aggregated by region and decade, and the MSVPA diet data were available as summarized in the literature.

These data were standardized, combined into a single dataset, the prey taxonomy was queried using the 'taxize' package in R, and each predator and prey were assigned to model groups.

We generated a probabilistic representation of fish diet compositions from multiple data sources following the approach in Ainsworth et al. (2010) and Masi et al. (2014). Looping through each model predator, bootstrapping was done to sample 30% of the data (with replacement) 10,000 times to build a distribution. Because a record in the diet dataset could represent ≥1 fish, the bootstrap samples were weighted by the square root of the number of stomachs represented by the sample. The square root was used so as not to overemphasize aggregated data from the NEFSC and MSVPA datasets. For each bootstrap, the Dirichlet multinomial was fitted using the fitDirichlet function in the R compositions package, alpha and beta parameters were extracted, and the posterior beta distribution function was determined for each prey item (Figure 27). Lastly, the maximum likelihood estimate of the diet proportion was taken from the posterior distribution. In addition, lower and upper 95% confidence intervals were estimated for each model predator and prey in the diet dataset. The Dirichlet was unable to fit the data for anchovies and menhaden, likely because of the highly aggregated nature of the prey items, so their diets remained unchanged from the previous version of the NWACS-MICE model (SEDAR, 2020b). Similarly, the diet of benthic invertebrates and zooplankton also were unchanged from the previous version of NWACS-MICE.

When estimating the Dirichlet, we included unidentified prey items as separate prey categories (e.g., unidentified fish, crustacean, clupeid) so that their proportions could be distributed to the MLE estimates of the identified prey items. The distribution of UI prey was conducted hierarchically, going from low to high unidentified taxonomies.

Lastly, the diet data provided did not include prey size, either because the prey were too digested or because it was not included with the initial data request. Therefore, the prey proportions for multistanza groups had to be distributed across age stanzas. The RIDEM dataset was the only one with predator-prey size data to inform this. We attempted to use predator-specific empirical cumulative distribution functions (ECDF) of all fish prey with size information to partition the diet composition to age classes. However, this resulted in too much diet assigned to youngest age stanzas and mass imbalance in the Ecopath model. During the balancing process, we resorted to partitioning the diet across age stanzas based on stanza biomass. The final diet matrix for the NWACS-MICE model is shown in Table 8.

10.2.8 Mass Balance

Four groups were out of balance in the initial Ecopath model, meaning that removals due to predation or fishing exceeded the production of that group, and the ecotrophic efficiency (*EE*) was >1 (Table 9). The unbalanced groups were striped bass age-0, juvenile bluefish, juvenile weakfish, and juvenile Atlantic herring. It is common for the youngest age stanzas to have high *EE* because biomass is estimated to be small, and seemingly trivial amounts of diet can result in high predation mortality rates (M_2). In these cases, the diet matrix was adjusted to lower M_2. For weakfish, the stock assessment assumes an increase in M overtime (presumably due to predation) that has caused Z to remain high, preventing any stock recovery. Therefore, it was desirable to have a high *EE* such that increases in predation over the Ecosim simulation would increase *Z* and depress stock size. Therefore, we set the *EE* to 0.99 for weakfish and allowed the model to solver for biomass accumulation.

10.3 Simulation model development and calibration (Ecosim)

10.3.1 Time series data

10.3.1.1 Observed time series

The NWACS-MICE Ecosim model was calibrated to annual time-series of observed relative abundance and fishery removals from 1985-2023 (Table 10). A total of 23 relative abundance indices from various fishery independent and dependent surveys were taken directly from the stock assessment report files. A catch timeseries was included for each ERP species and age stanza (n=12). Catch time series were obtained from the stock assessments and included landings and dead discards, converted to units mt/km^2. Catch timeseries for non-leading age stanzas were treated as relative, because the stable age assumption in Ecopath creates discrepancies in scale of biomass and therefore landings, when forced by fishing mortality. Biomass (mt/km^2) trends estimated by the stock assessment models were included in the time series file only to allow for visual comparison between NWACS-MICE and stock assessment predictions. They were given a weight of 0 and had no influence on model calibration.

10.3.1.2 Forcing time series

In Ecosim, forcing time series can be included to drive fishing mortality, primary production, spawning seasonality, and predator-prey interactions. We developed a total of eight forcing time series to consider when calibrating the model.

1. Annual species- and stanza-specific fishing mortality rates (*F*) from 1985-2023 were calculated from stock assessment model output as total landings in mt/km^2 (harvest and dead discards) divided by mean (or mid-year) annual biomass in mt/km^2.
2. A separate set of monthly fishing mortality rates were developed by proportioning annual *F* to months using directed effort estimates from MRIP (striped bass, bluefish, weakfish, and spiny dogfish), monthly effort data for Atlantic menhaden from CDFRs, and monthly landings data for Atlantic herring. However, the models with monthly fishing effort forcing were exploratory only and deemed unsuitable for use without accompanying monthly catch data.
3. Annual and primary production (PP) forcing time series were obtained from the Global Ocean Reanalysis products (GLORYS) hosted by the Copernicus Marine Service. The GLORYS physics and biogeochemical data were extracted from the Copernicus Marine server (using an R script) for all available years (1993-2023) and the region bounded by 62.5°- 80° W and 33°- 46° N. The variables included in the queried GLORYS dataset are three-dimensional estimates of chlorophyll-a, net primary production, dissolved oxygen, phytoplankton carbon, temperature, and salinity. Surface, bottom, and depth integrated monthly spatial rasters were developed from the netcdf files and averaged over grid cells to produce annual timeseries. To drive primary production in NWACS-MICE, we use the depth-integrated chlorophyll-a product from GLORYS. In the annual PP forcing time series, we assume a value of 1 (i.e., no forcing) for the years before the GLORYS data (1985-1992).

4. A monthly PP forcing time series was developed from the monthly chlorophyll-a rasters from GLORYS (Figure 28). We used the average monthly chlorophyll-a from the first five years of data for 1985-1992 to retain the seasonal component in early years of the simulation.
5. Ecosim allows for linking seasonal or long-term forcing functions to the production of eggs/larvae in multistanza groups, driving the youngest stanza biomass independent of their density. Egg production forcing can be used to represent match/mismatch dynamics when primary production forcing is also included or density-independent mortality resulting from abiotic factors. The purpose of including monthly egg production in NWACS-MICE was to moderate the availability of juvenile stanzas to predators. Monthly egg production vectors were obtained from published reproductive histology studies and scaled to the maximum value and input to Ecosim as seasonal forcing functions (Table 11).
6. Seasonal forcing functions on predator-prey vulnerability parameters were included to account for seasonal-spatial overlap patterns for key predator-prey interactions. Predator-prey overlap indices (OI) by predator age and season were developed as part of the earlier MSVPA work. The OIs were averaged across ages within a stanza and a spline function was fit to extrapolate the seasonal OI estimates to months and then scaled to the mean. Monthly vulnerability forcing was only applied to the non-juvenile age stanzas of striped bass, bluefish, weakfish, and spiny dogfish and only for adult menhaden and Atlantic herring prey k_{ij} (Figure 29).
7. After a number of exploratory runs were conducted, it became apparent that a) the model was unable to fit the recent low abundances of Atlantic herring, and b) the model was attempting (unsuccessfully) to use predation mortality by spiny dogfish and striped bass to drive down Atlantic herring biomass. This created 'tension' in the model such that Atlantic herring, spiny dogfish, and striped bass could not all be fit simultaneously. Therefore, an additional long-term egg forcing function was developed for Atlantic herring using the estimated recruitment deviations from the 2024 Age Structured Assessment Program (ASAP) stock assessment model. Inclusion of this forcing function implies some density independent process is affecting early life history stages as estimated by the ASAP model. The annual log recruitment deviations from ASAP were exponentiated and rescaled to a mean of 1 and then multiplied by the seasonal component to get a monthly, long-term forcing function on egg production for Atlantic herring (Figure 30).
8. Lastly, to represent an alternative hypothesis that survival of other ages is the cause of persistent low stock biomass in Atlantic herring, we included a time series forcing function for the adult stanza on the 'other mortality', M_0, term. The Atlantic herring M_0 forcing was derived from the numbers-at-age deviations (NAAdevs) estimated by the 2024 Woods Hole Assessment Model (WHAM). The NAAdevs were inverted and rescaled to a mean of 1, and then averaged across age for each year, weighted by the numbers at age (Figure 30). However, it is not recommended to use these M_0 forcing functions due to uncertainties about their scale, interpretation, and integration across ages.

10.3.2 Ecosim Model Configurations

Various model configurations were developed and tested, including annual 'continuity' models and those with combinations of the eight forcing functions intended to represent some seasonal (monthly) aspect of the system. The continuity model (run1) is the simplest, and has the same configuration used in the previous ERP assessment, where ERP species are driven by annual fishing mortality and no other forcing functions are included. Combined, the models account for dynamics in fishing mortality, primary production, seasonal and long-term spawning (i.e., egg production), and predator-prey spatial overlaps. The forcing functions were included individually and in combinations. The model configurations are summarized in Table 12.

10.3.3 Model Calibration Procedure

The NWACS-MICE was calibrated to time series data of observed abundance and catch by adjusting the predator-prey vulnerability parameters. In Ecosim, the vulnerability exchange rates are modified via input multipliers (k_{ij}), which can be more easily interpreted as the maximum amount of predation mortality (relative to Ecopath base predation mortality) that a predator can exert on its prey if the predator were at its carrying capacity. Because the predator-prey vulnerabilities are conditioned on the baseline Ecopath model, the time dynamic forcing functions included, and other foraging arena parameters (e.g. prey switching, feeding time adjustments) the model should ideally be recalibrated when any of those inputs are changed. Therefore, we developed a systematic approach to calibrating the various NWACS-MICE Ecosim models in Table 12.

First, consider there is one vulnerability multiplier parameter for each predator prey interaction, and they can be referred to as a single value shared across all predator prey interactions (k), equal for all prey of predator j (k_j), or unique for each predator-prey interaction (k_{ij}).

Estimated vulnerability multipliers may be sensitive to their initial starting values. Therefore, models were calibrated using 5 different initial starting values for the k, k_j, or k_{ij} parameters.

- k=2: The default, all values in the matrix are set equal to 2.
- $k_j = B_{unf}/B_o$: vulnerabilities scaled based on proportion of Ecopath biomass to carrying capacity using the 'estimate vulnerabilities' form in Ecosim.
- k_{ij} = Vmax: Vulnerability cap, assuming max predation mortality (M_{2ij}) is 75% of prey natural mortality M_i.
- k_j = TL: vulnerabilities scaled based on trophic level, ranging from 1.2 to 100.
- k_{ij} = random: each k_{ij} chosen at random, repeated 5 times.

Prey switching occurs when a predator disproportionately takes more of the prey as it becomes more abundant. This is modeled in Ecosim, where the relative search rate (a_{ij}) is proportional to prey biomass raised to the power P_j, $a_{ij} = B_j^{P_j}$. If P_j=0, no prey switching occurs and the relative search rate is 1. If $P_j < 1$, the a_{ij} varies slower than the prey biomass, such that the predator will continue searching for prey even if it becomes less abundant. If $P_j > 1$, then a_{ij} varies faster than prey abundance, meaning that the predator will quickly switch when prey increase or decrease. The prey switching parameter was assumed to be 1.0 for all menhaden predators in the previous version of the MICE model. In this assessment, we evaluated P_j values of 0.0, 0.5, 1.0, 1.5, and 2 during the calibration process.

The maximum relative P/B parameter in Ecosim can be used to constrain primary producers' response to changes in nutrient or primary production forcing. In the primary production models, the maximum relative production rate for phytoplankton was tested at values of 1, 2 (default), and 3.

There are several ways to calibrate Ecosim models, with regards to the number and sequence of k_j or k_{ij} to be estimated. Ecosim calibration procedures are best reviewed and described by Bentley et al. (2024). Here, we tested 3 different calibration procedures.

- All k_j – A single k_j is estimated for each consumer in the model, and that k_j is applied to all prey items of predator j.
- k_j with timeseries – A single k_j is estimated for each consumer in the model that has time series data, and that k_j is applied to all prey items of predator j.
- k_{ij} repeated search – Identify and estimate the most sensitive k_{ij} parameters. Repeat the process five times, each time estimating a new set of sensitive k_{ij}. This approach was used in the previous ERP assessment.

A total of 150 NWACS-MICE Ecosim runs were developed and calibrated across the different models, initial vulnerabilities, prey switching settings, max rel PB, and calibration procedure. Not all parameter and model combinations were evaluated, rather we began with a more thorough evaluation of models 1-3 and based on those results we eliminated several parameter configurations from further evaluation (Table 12). For example, it was determined early on that initializing k_j based on proportion of carrying capacity (B_{unf}/B_o) led to better fitting models. Therefore, these initial starting value configurations were explored in additional model runs leading up to the base run. As we moved closer to a base run, fewer configurations were explored because many were determined to perform poorly in earlier runs.

10.3.4 Summary of Calibrated Models

A total of 150 Ecosim runs were calibrated across a range of model configurations for forcing functions, initial vulnerabilities, calibration procedures, prey switching parameters, and maximum relative primary production rates (Figure 31). After the first 72 runs were calibrated for models 1-3, we evaluated the sum-of-squared residuals (SS) to determine which initial starting values and calibration procedures to carry forward. While there was a lot of spread in the estimated SS by initial k_{ij}, the general pattern suggested that initializing the vulnerability matrix with $k_j= B_{unf}/B_o$ resulted in better fitting models (Figure 31). The calibration method resulting in the lowest SS was the repeated k_{ij} search method, followed by all predators k_j, and lastly predator k_j with timeseries. Based on the set of initial runs, we continued calibrations with the repeated k_{ij} search procedure and the B_{unf}/B_o initial vulnerability setting, while testing additional values for prey switching power P_j, and max rel PB (for PP forcing models only).

Across all 150 calibrated runs, equally good fitting models were obtained in Ecosim models 1-5, and especially for those with monthly egg production (model 4) and seasonal vulnerabilities (model 5; Figure 31). The Ecosim models that included monthly fishing effort (models 6 and 7) were unable to produce models with lower SS than those in models 1-5. Model runs with P_j<1 tended to produce slightly better fits than those with P_j>1. Lastly, the total number of k_{ij} estimated in each model that used the repeated search methodology varied from around 60 to 110.

Including the Atlantic herring forcing functions resulted in lower SS than previously achieved in other model types (Table 13). The lowest SS was achieved in run 145 (SS=1791), which included the Atlantic herring M_0 forcing from the NAA deviations (model type 13). However, run 145 did not appreciably improve the fit to the recent low biomass of Atlantic herring, nor did it resolve the tension between spiny dogfish and striped bass within the model. In addition, the conversion of NAA deviations to an 'other mortality' multiplier might not be valid.

The next best fitting run was 144 (SS=1800), which included seasonal egg production and k_{ij} multipliers along with the long-term Atlantic herring forcing function derived from estimated recruitment deviations (model type 12; Table 13). After that, the continuity models and those with seasonal k_{ij} and egg forcing were represented. There were 15 models with SS falling below the 10th percentile (SS=1,895), and they included 2 runs each from models 1-2, 1 run each from models 4-5, and 3 runs each for models 11-13. None of the runs that included monthly PP (models 3, 8, 10) or monthly fishing mortality forcing (models 6-7) were in the top 10% of SS.

For the top 10 percentile of runs, the total number of estimated k_{ij} after 5 iterations of the repeated search procedure ranged from 63 in run 95 to 93 in run 150 (mean 81.33), which is about 2-4 parameters for each timeseries that combined have 1,720 data points. Therefore, we do not consider these models to be overparameterized. The number of k_{ij} estimated on lower (<1.01) or upper (≥1e10) bounds ranged from 13 in run 142 to 43 in run 105 (mean=23). On average the model estimated more parameters on the lower bound (17.2) than the upper bound (6.0).

10.3.5 Estimated Vulnerabilities for ERP Species

The tradeoff relationships and trophic impacts of harvest policies are expected to be sensitive to the vulnerability parameters. In comparing the top 10 percent of model runs, only run 95 estimated no k_{ij}'s on bounds for adult menhaden (as prey i) and runs 95, 136, 137, 144, 149, and 150 estimated no k_{ij} on bounds for juvenile menhaden. Most model runs estimated between 0-4 k_{ij} on bounds combined for juvenile and adult menhaden as prey (out of 13 possible k_{ij} where i is menhaden). Weakfish adults were most likely to have k_{ij} estimated on a bound for adult menhaden (10/15 runs in Table 13).

Overall, the k_{ij} for Atlantic menhaden and striped bass interactions were generally estimable by the model, but the values ranged widely. The model estimated moderate to high k_{ij} for most striped bass and Atlantic menhaden (as prey i) interactions (Table 14). This occurs because striped bass biomass was severely depleted during the Ecopath base year of 1985, requiring larger k_{ij} to increase consumption rates and therefore biomass gains. For adult menhaden (Table 14) the estimated k_{ij} by striped bass age 2-5 stanza ranged from 2.075-2,341 and from 12.53-6.43e5 (excluding bounds) for striped bass age 6+ stanza. Only runs 105 and 137 estimated a k_{ij} for striped bass and adult menhaden at an upper bound (>=1e10) and only 3 runs estimated a striped bass adult menhaden k_{ij} at a lower bound (<=1.01). Estimated k_{ij} for juvenile menhaden ranged from 2.523-2.21e6 for striped bass age-0; 1.475-1,260 for striped bass age 2-5, and 1.870-1,636 for striped bass age 6+ (excluding lower and upper bounds). In 3 of 15 runs, striped-bass age 0 and juvenile menhaden k_{ij} was estimated at the lower bound, while one k_{ij} was estimated at bounds for striped bass age 2-5 and 6+ stanzas. High k_{ij} for menhaden as prey were most likely.

For adult and juvenile Atlantic herring as prey, k_{ij} were generally estimated to be low for adult bluefish, moderately high for striped bass, and high for spiny dogfish (Table 13), and insensitive (not estimated) to weakfish. To fit the recent low biomass of Atlantic herring, the model would estimate higher predation by striped bass and dogfish, which is reflected in their high k_{ij} estimates. Of the top 10%, only runs 144, 149, and 150 estimated no k_{ij} on bounds, either for juvenile or adult Atlantic herring. This indicates that the inclusion of forcing functions specific to Atlantic herring (models 12 and 13) improved the parameter estimation for their predators. On the other hand, run 105 estimated 5 k_{ij} on bounds, out of 10 possible k_{ij} where Atlantic herring are prey. The k_{ij} for adult bluefish and Atlantic herring was most likely to be estimated on a lower bound.

Another pattern revealed from the model calibrations is that the k_{ij} for spiny dogfish as a predator was often estimated to be very high, especially for herring, menhaden, and weakfish as prey, more so than other predators (Table 14). Conversely, the spiny dogfish and striped bass k_{ij} were often estimated to be low.

10.3.6 Model Outputs

10.3.6.1 Biomass

The biomass trajectories from the top 10% of runs in Table 13 were fairly consistent across model runs for each species (Figure 32). Striped bass age 6+ biomass was predicted to be much higher in the PP models (runs 77 and 105) and spiny dogfish biomass diverged across runs during the last half of the simulation. Menhaden biomass did not vary much over the duration of the simulation except for in the PP models. Juvenile bluefish biomass exhibited high variability across model runs, while biomass predictions for the adult group were consistent across runs, with a similar pattern for weakfish. Atlantic herring 0-1 biomass was variable across model runs, with poor fits to the data except in the model (model 12) that used recruitment deviations as a forcing function on egg production (runs 144, 142, 149, 150). In addition, only AH Rdev models were able to capture the low biomass observed in recent years for Atlantic herring. Anchovy and benthic invertebrate biomass declined in most runs, except those that included PP forcing, and zooplankton and phytoplankton biomass was flat unless bottom-up PP forcing was included.

Because Ecosim does not estimate annual parameters (e.g. recruitment deviations), it was unable to fit the high variability often observed in the individual indices (Figure 33). Only the models with primary production forcing (runs 77 and 105) and Atlantic herring recruitment deviations (run

150) exhibited high interannual variability on the order of that in the observed indices. During calibration, zero weight was given to the biomass time series estimated by the stock assessment models, yet NWACS-MICE was able to reproduce those trends quite well for at least one age stanza of each ERP species. Spiny dogfish is an exception, because the stock assessment and survey index produce conflicting trends, and NWACS-MICE fits the trawl survey better. For Atlantic menhaden, the inclusion of primary production forcing introduced variability that tracks with the assessment biomass of the adults but was not reflective of the high variability observed in the data.

10.3.6.2 Catch

Catch is estimated in Ecosim as $F*B$, where F is forced fishing mortality and biomass is estimated at each time step. In general, discrepancies in fits to catch are reflective of over or underestimation of biomass when the Ecosim model is forced by F (i.e., $C=F*B$). The NWACS-MICE Ecosim-predicted catch was consistent across model runs for each species, except striped bass and spiny dogfish, which tended to overestimate catch in several model runs, especially since about 2005 (Figure 34). Atlantic herring 2+ did not fit the catch data well in recent years, due to the model's inability to capture the recent biomass declines experienced for that stock. In the preferred base run (150), Atlantic herring under predict catch for much of the time series but do fit the lows observed since about 2019.

10.3.6.3 Mortality

Fishing mortality was forced in the NWACS-MICE model (Figure 35), while predation (M_2) and other mortality (M_0) were estimated. Total predation mortality varied considerably across model runs for most species, owing to the direct relationship between the vulnerability parameters and consumption rates (higher k_{ij} allow for higher predation mortality). For example, the vulnerability parameters for spiny dogfish on striped bass age 6+ were estimated to be high in runs 77, 85, and 105 (Table 14), leading to high biomass of spiny dogfish (Figure 32), and high predation mortality on Atlantic menhaden, Atlantic herring, and striped bass (Figure 36). Generally, predation mortality for the youngest age stanzas and forage groups varied the most over time and across model runs.

10.3.6.4 Diet Composition

Ecosim predicts diet proportions for all consumer groups in the model at a monthly timestep based on the initial preferences defined by the Ecopath diet matrix, predator-prey biomasses, and foraging arena parameters (vulnerabilities, feeding time adjustment rates, and prey switching). The predicted proportion of menhaden in the diets of its predators varied over time and across model runs for most ERP predators (Figure 37). For striped bass age 6+, the proportion of menhaden in the diet was predicted to be relatively flat at 40% across all model runs, with a gradual increase over time. Alternatively, the prediction of menhaden in the diets of striped bass age 2-5 varied across model runs, with some runs predicting a decline from 20% to less than 10%, while other runs predicted a gradual increase from 20% up to about 40%. The contribution of menhaden to the diets of spiny dogfish was relatively flat over time at around 15%, with some variability among model runs. For adult bluefish and weakfish, menhaden percent diet increased slightly over time in most model runs. Seasonal forcing functions were applied to k_{ij} for menhaden as prey to striped bass, spiny dogfish, bluefish, and weakfish (Figure 29) in models 5 and 11-13. This resulted in striped bass eating slightly less menhaden during the spring than in the fall and winter, while spiny dogfish consumed more menhaden in the early spring than in late summer/fall (Figure 38).

The main consumers of Atlantic herring in the NWACS-MICE model were striped bass age 6+, spiny dogfish, and adult bluefish. For these groups, the predicted contribution of herring to their diets were similar, and tracked with the biomass of herring, which increased from 1985-2000 and has been in a decline since (Figure 39). Run 150 tended to estimate lower diet proportions of herring than the other runs. The seasonal effect was more evident for Atlantic herring than Atlantic menhaden, with much more pronounced swings in diet percentage in model runs with these effects (Figure 40). Striped bass age 6+ percent diet of herring was about 5% in fall and winter, increasing to 20% in the spring and early summer. Spiny dogfish diet was composed of more herring during the fall (7-15%) compared to spring (4-8%). The seasonal and annual predictions for Atlantic herring showed considerable variability across model runs.

10.3.7 Equilibrium Analysis

An equilibrium analysis was conducted as a model diagnostic, and to establish single-species *F* values for use in projections. The intention of using equilibrium projections as a diagnostic is to check whether the ERP species in NWACS-MICE are responding to fishing in a similar way as would be predicted by their single-species stock assessments. That is, biomass and yield reference points should be achieved under similar values of relative fishing mortality in both the NWACS-MICE and single-species stock assessments. In the first ERP assessment, the equilibrium analysis was run using the F_{MSY} module in Ecosim, which starts the equilibrium projections from the Ecopath base year. However, the biomass accumulation we assumed in this version of the Ecopath model was causing issues with that approach. Therefore, the equilibrium analysis was run using 40-year projections that extend from the terminal year of the model (2023). This was accomplished using the Multisim plugin to iterate over *F* rates for each species individually, while holding all others constant at terminal year 'status quo' *F*. Fishing mortality was incremented from 0 to approximately 2**M* for each species. This approach to equilibrium diagnostics is advantageous because it is not affected by initial dynamics determined by biomass accumulation in the Ecopath base year, and it is consistent with the projection methodology that will be used for the tradeoff analysis.

The NWACS-MICE equilibrium yield curves exhibited several different patterns (Table 15, Figure 41). In most cases, the curve followed a dome-shaped pattern with a clear indication of equilibrium F_{MSY} (e.g., striped bass 6+; Figure 41), and this is the ideal scenario when the F_{MSY} is consistent with single-species *F* reference points. In other cases, the yield curve was broadly dome-shaped, indicating that the species can tolerate high *F* before biomass starts to decline (e.g., weakfish, Atlantic herring). Lastly, some model runs produced asymptotic yield curves (e.g., Atlantic menhaden, Atlantic herring, bluefish). For some species, all three patterns were evident and the equilibrium yield curves were highly variable across model runs. This is because the vulnerability parameters also determine how a species will respond to fishing through their own compensatory mechanisms as well as any top-down predator effects in the future projections. For example, the difference between runs 144 and 150 for striped bass is an increase in a single k_{ij} from 1.0 to 1.5, for spiny dogfish preying on striped bass age 2-5. In fact, spiny dogfish are driving much of the equilibrium results. For instance, when spiny dogfish are most productive (high MSY, run 137) almost all other species are least productive.

In run 150, an adjustment was made to a single k_{ij} parameter so that striped bass would approximate their B_{TARGET} when fished at their F_{TARGET} (Figure 42, Figure 43). The model is very sensitive to spiny dogfish as a predator on striped bass. Changing the k_{ij} for striped bass 2-5 and spiny dogfish from 1.000569 to 1.5 gets striped bass to approximate B_{TARGET} when fished at their single species F_{TARGET} (F_{MULT}=0.934). The SS increased from 1,800 to 1,874. The same k_{ij} in run 139 was estimated to be 3.47. In the equilibrium projections, Atlantic herring and weakfish did not reach their biomass target in almost all runs, even at F=0 (Figure 42).

10.4 Base Run Scenario (run 150)

A base run was selected from all the candidate model runs based on model fit to timeseries as measured by the sum-of-squared residuals, the number of parameters estimated, and on bounds, equilibrium diagnostics, and other emergent properties. In Ecosim, models may be capable of fitting historical time series but then produce unreasonable results in long-term projections, especially when vulnerabilities are estimated on bounds. Thus, it is recommended to also inspect projections under different F levels prior to using a model for management (Heymans et al., 2016). Upon evaluating the first 122 runs (for model types 1-8), it became evident that model runs including the seasonal egg production and vulnerability forcing had better diagnostics, which was important for the ERP workgroup in defining the base run. In addition, incorporating these seasonal forcing functions could resolve a key model limitation identified during the previous assessment that led to diet proportions of Atlantic herring and menhaden predicted to be high throughout the year, even though they interact only seasonally with their predators due to migrations. We began exploring additional models (model type 11) that incorporated the seasonal egg and k_{ij} forcing functions. Run 139 included the estimated k_{ij} from run 86, with vulnerability caps applied to striped bass, menhaden, and spiny dogfish. Vulnerability caps were used in the previous assessment to resolve issues in the equilibrium diagnostics. They are calculated based on an assumption for the maximum proportion of natural mortality (M_{2max}) that could be attributed to a single predator (Bentley et al., 2024; Chagaris et al., 2020), here assumed to be 75% of M. In addition, run 139 used a custom setup for prey switching, where P_j=0 for striped bass (all stanzas), menhaden (all stanzas), and Atlantic herring 0-1; P_j=0.5 for spiny dogfish, bluefish, weakfish, anchovies, inverts, and zooplankton; and P_j=1 for Atlantic herring 2+. The P_j values were selected through inspection of other fits and diagnostics in other model runs.

However, like all other model runs, run 139 was unable to capture the recent declines in herring biomass and attempted to do so with predation mortality from spiny dogfish and striped bass. This created a situation where the model could not produce satisfactory fits for striped bass, spiny dogfish, and Atlantic herring simultaneously. In addition, run 139 did not allow striped bass to get close to their biomass target when fished at their F_{TARGET}. Therefore, another set of models was developed, including the Atlantic herring forcing functions to drive long-term egg production (i.e., recruitment deviations) or adult survival (from NAA deviations). Run 144 was developed from run 139 by applying the Atlantic herring (AH) recruitment deviations from the ASAP model to force egg production over time. This implies that some density independent processes are affecting early life history stages as estimated by the ASAP model. This is not a new phenomenon. As outlined in both the assessment update (NEFSC, 2024) and in the most recent benchmark (NEFSC, 2025), Atlantic herring has been experiencing dramatically low recruitment over the past decade, independent of spawning stock biomass.

After adding the long-term egg production forcing, the model was calibrated following the iterative k_{ij} search procedure for five iterations, initialized with run 139 k_{ij}'s. The sum of squared residuals (SS) in run 144 was 1,800, which was the 2nd lowest SS out of the 149 models evaluated to that point. There were 93 k_{ij} parameters (out of 107 total) estimated in run 144, and 15 of those (16%) were on lower (n=13) or upper (n=2) bounds. This is a low ratio of parameters on bounds compared to other models. However, the equilibrium diagnostics revealed that striped bass in the NWACS-MICE model run 144 were much more productive than the single species model suggests (i.e. they exceeded B_{TARGET} when fished at F_{TARGET}; Figure 42), which is in contrast to recent data trends and estimated recruitment by the stock assessment model.

Run 150 was selected as the preferred base run because it resolved issues in runs 139 and 144. The difference in striped bass productivity between runs 139 (low productivity) and 144 (high productivity) was explained by a small difference to a single estimated vulnerability parameter. Changing the k_{ij} for striped bass 2-5 as prey to spiny dogfish from 1.000569 in run 144 (or 3.47 in run 139) to 1.5 in run 150 gets striped bass to approximate B_{TARGET} when fished at their single species F_{TARGET} (F_{MULT}=0.934). The value of 1.5 was somewhat subjective and based on iterative runs aiming for striped bass to achieve B_{TARGET} when fished at F_{TARGET}. This particular k_{ij} parameter should be low because spiny dogfish are a biomass-dominant group in the model, and they have a high baseline predation mortality rate on striped bass in Ecopath (1985, when striped bass biomass was low). Thus, there is not much scope to increase consumption and baseline M_2 without exceeding hypothesized vulnerability caps and resulting in unreasonable predation mortality rates on striped bass by spiny dogfish. Lastly, run 150 was preferred because it established that striped bass are responding to fishing in a similar way to the stock assessment under future status quo conditions, which is essential for policy feedback with the single species model and reference points, and is a prerequisite for conducting simulations manipulating menhaden fishing mortality.

10.5 Trade-off Analysis and Ecological Reference Points

As in the previous assessment, the ecological reference points (ERPs) were identified from an equilibrium tradeoff analysis using 40-year projections under different combinations of menhaden and striped bass fishing mortality rates. A total of 441 projection simulations from the NWACS-MICE model were conducted over combinations of menhaden and striped bass fishing mortality, going from 0 to approximately 2*M for each species. Fishing mortality was modified using an F multiplier on the terminal year (2023) F rate for each species and stanza, and the tradeoff analysis was conducted using the Multisim plugin to automate the simulations. Fishing mortality for all other species was held constant at their status quo (2023) F rate. Any applied seasonal forcing functions were included in the projection runs. For run 150, we assumed status quo low recruitment for Atlantic herring (average of the most recent three years) to project forward. The single-species assessment for striped bass uses empirical rather than model-based reference points, with the spawning stock biomass threshold defined as the 1995 estimate of female SSB and the SSB_{TARGET} defined as 125% of the threshold. The NWACS-MICE equivalent, equilibrium biomass of striped bass age 6+, was calculated as the mean of the last three years of projections and divided by the biomass target, which was 1.25 times the estimated biomass in 1995.

Results from the tradeoff analysis are illustrated in a surface plot to show how striped bass are predicted to respond to different levels of menhaden and striped bass fishing mortality (Figure 44). A clear negative relationship between menhaden harvest and striped bass biomass is evident. Under the current menhaden and striped bass F, striped bass biomass was estimated at their biomass threshold. When fished at their current F rate, striped bass do not reach their biomass target under any menhaden F scenarios.

When striped bass were fished at their F_{TARGET} (F_{MULT}=0.934), striped bass reached their biomass target at a menhaden fishing mortality multiplier of 0.737, a 26.3% reduction from current menhaden F rates. From this, the ERPs are established as the maximum fishing mortality rate on Atlantic menhaden that will allow striped bass to reach their biomass target (or threshold) when fished at their F_{TARGET} (Table 16). The resulting ERP F_{TARGET} and $F_{THRESHOLD}$ multipliers are 0.737 and 1.916, respectively. The current full fishing mortality (F_{FULL}) for Atlantic menhaden estimated by the BAM is 0.256, and the equivalent ERP F_{FULL} rates are 0.189 and 0.490 for the target and threshold, respectively.

10.6 Discussion

10.6.1 Summary Overview

The NWACS-MICE was updated from the 2020 assessment version (Chagaris et al., 2020; SEDAR, 2020b) and used to simulate trophic dynamics and estimate ecological reference points for Atlantic menhaden. The current version of the NWACS-MICE includes updates to ERP species inputs consistent with the most recent stock assessments, a reanalysis of the diet data using a robust and reproducible statistical procedure, an exhaustive calibration effort, and the inclusion of seasonal and long-term forcing functions. In doing so, we improved reproducibility and reduced error by developing scripts to process datasets and addressed key uncertainties that arose during the previous ERP assessment.

While parameterization of the current NWACS-MICE was much improved over the previous assessment, the core challenges of calibrating, diagnosing, and using an ecosystem model for management were just as persistent. Ecosim model predictions are conditioned on the combination of Ecopath inputs, the Ecosim foraging arena parameters, and any included forcing functions. It is impossible to evaluate all combinations of parameters, and only the k_{ij} are estimable by the Ecosim calibration routine. This prevents us from conclusively identifying a global best fit model, as would be expected from an age-structured stock assessment model. We attempted to overcome this challenge through extensive calibration of 150 different model runs but also recognize that it is only a fraction of possible configurations. Unlike stock assessment models, there are no standard procedures and metrics for diagnosing ecosystem models after they have been calibrated. Here, we used 3 metrics to diagnose the NWACS-MICE–model fit (the SS), parameters on bounds, and equilibrium results. Any models that performed poorly in one or more diagnostics were discounted as a base run, and/or adjustments were made to resolve the issue. What is deemed to be a 'good' model is somewhat subjective and based on the preference to have a model with low SS, few parameters on bounds, and acceptable equilibrium results. Nevertheless, the multitude of model runs improved our understanding of the complex model dynamics, allowed us to track the effects of specific parameters, and provided a suite of models from which to compare and develop management advice.

10.6.2 Ecological Reference Points

Making direct comparisons between the current ERPs and those established during the 2019 assessment is confounded by changes in the assessment models for the ERP species, including the change to *M* for Atlantic menhaden in this assessment, and refinements in the NWACS-MICE model, as well as the current status quo conditions (2017 vs 2023) assumed in the tradeoff projections. Regardless, the ERP F_{TARGET} estimated presently did not change much from the previous ERP assessment. In the current assessment, the ERP F_{TARGET} was estimated to be 26.3% lower than the current *F* of 0.256 for Atlantic menhaden, resulting in an ERP F_{TARGET} of 0.189 (in BAM F_{FULL} units). In the previous assessment, the 2017 *F* was estimated to be 0.157, and the ERP *F* target was estimated to be 20% higher, at an *F* rate of 0.188. It is difficult to say whether the near identical estimate of F_{TARGET} for the two models is indicative of robustness or a mere coincidence. However, the ERP F_{TARGET} may produce different estimates of total allowable catch than previously due to the lower estimate of *M* used in the BAM, which results in lower estimates of biomass.

The tradeoff analysis was repeated for a subset of model runs. While the exact reference points varied across the different model runs, the tradeoff relationship was always negative, with a fairly consistent slope and shape to the curve (Figure 45). That is, the proportional effect of menhaden harvest on striped bass is robust and essentially linear over the range of *F* rates likely to be exerted on menhaden (where $F_{MULT} < 2$). The challenge in the current NWACS-MICE was in estimating the productivity of striped bass, which shifted the tradeoff relationships up or down on the y-axis (Figure 45). Our base run was selected in part because it was more consistent with the single species stock assessment reference points for striped bass (i.e., the biomass target was achievable when fished at F_{TARGET}). To establish ERPs for other model runs (with higher or lower productivity) the associated F_{TARGET} multipliers would need to be adjusted to be compatible with the productivity regime (assuming the B_{TARGET} doesn't change).

In addition to uncertainty from model estimation and parameterization, ERPs are also affected by uncertainty in future ecosystem conditions. For example, in the base run, run 150, the low Atlantic herring recruitment deviations were assumed to persist into the future. However, if Atlantic herring recruitment returns to the long-term mean, the estimates of the ERP target and threshold are higher (Table 17), indicating that a higher *F* can be applied to menhaden and striped bass will remain at their target, as Atlantic herring are a more abundant and productive alternative prey in this scenario.

10.6.3 Consequential Changes to Ecopath Inputs

An important change was made to Atlantic menhaden Ecopath inputs in this version of NWACS-MICE due to the lower natural mortality assumed in the current single species stock assessment model. The 2019 BAM assumed a higher natural mortality rate that was size dependent, going from 1.76 for age-0 to 0.72 for age-6+ menhaden. This led to a 1985 biomass estimate of 4.985 million mt. In the current BAM, the natural mortality rate is 1.39 for age-0 and declines to 0.57 for age 6+, with a 1985 biomass of 2.360 million mt. Thus, the current version of NWACS-MICE assumes a smaller and slightly less productive Atlantic menhaden stock in the Ecopath base year than previously. In Ecopath, this resulted in a substantial increase in the ecotrophic efficiency (*EE*), from 0.08 and 0.154 in the previous model to 0.238 and 0.405 for juvenile and adult menhaden, respectively. The low *EE* in NWACS-MICE was cause for concern during the previous assessment and was explained by the combination of high menhaden biomass and total mortality, with a limited set of predators represented. The adjustments made in the current assessment bring the *EE* up to levels that are more appropriate given the role of menhaden as a forage fish and the number of predators explicitly modeled in the NWACS-MICE.

Another consequential change to the model was the increase in spiny dogfish biomass estimates based on a new statistical catch-at-age model. Previously, spiny dogfish biomass was estimated to be 271,550 mt in 1985, based on area swept calculations from the NEFSC spring bottom trawl survey conducted on the shelf within the model domain. The most recent stock assessment using Stock Synthesis estimated the 1985 biomass to be 735,389 mt. One possible explanation for the discrepancy is that the area-swept approach is representative of biomass on the shelf and not extrapolated beyond the survey area, while the stock assessment is estimating stock size by scaling to total removals, with some of those removals occurring outside the model domain in deeper water. This is supported by a satellite tagging study (Carlson et al., 2014) that estimated a total home range that extends far beyond the shelf for spiny dogfish tagged off North Carolina, and a core usage area that only halfway overlaps with the model domain. Spiny dogfish tagged in the Gulf of Maine had a core usage area that was completely inside the model domain and a total home range that only partially overlapped with the model domain (Carlson et al., 2014).

The combined effect of lower menhaden biomass and *M*, higher striped bass biomass, and a revised diet matrix is most evident in the predation mortality rates. In the previous model, spiny dogfish only exerted high predation mortality on Atlantic herring, and there were no trophic connections to striped bass age 2-5 and 6+, bluefish, or weakfish. The current NWACS-MICE has spiny dogfish preying on the older stanzas of striped bass, bluefish, and weakfish. Previously, the proportion of menhaden in the diets of spiny dogfish was estimated to be <1%; and the current estimate is now 12%. This increases the baseline M_2 rates by spiny dogfish substantially, from <0.01 to 0.14 for adult menhaden. The baseline M_2 rates by spiny dogfish on striped bass also increased from zero to 0.22 and 0.11 for ages 2-5 and 6+. Thus, the reformulation of the NWACS-MICE Ecopath resulted in a baseline 1985 condition with spiny dogfish playing a more dominant role as a predator than in the previous model. Whether or not spiny dogfish are really such a dominant predator on the other ERP species remains an open question.

10.6.4 Spiny dogfish as top-down driver in the NWACS-MICE system

The dominance of spiny dogfish in the NWACS-MICE had notable consequences in the Ecosim simulations. During calibration, the k_{ij} estimates for spiny dogfish prey were more highly variable across species and model runs than for other ERP species, and more of them were on the upper bound. Runs with high k_{ij} (77, 105, and 137) resulted in large biomass gains for spiny dogfish beginning in 2005, which overestimated catch and led to large increases in predation mortality for their prey. While the predicted biomass gains matched the trawl survey index, they were opposite to the estimated biomass trajectory from the stock assessment, which was flat with a slight decrease (but given zero weight in calibration). The tradeoff in fitting the model was to improve fits to Atlantic herring and other prey at the expense of spiny dogfish fitting to their catch data. The systematic overestimation of catch for spiny dogfish since 2005 suggests model misspecification, where Ecosim is using spiny dogfish as a lever to try and drive down Atlantic herring biomass.

The productivity of striped bass is very sensitive to the vulnerability parameters of striped bass as prey for spiny dogfish. Because spiny dogfish are such a dominant predator in the model, with high baseline M_2 rates on striped bass, small differences in the k_{ij} can lead to big differences in predation mortality and therefore biomass over the long term. In run 144, the k_{ij}=1.0 held predation mortality constant at their baseline rates for striped bass age 2-5. In run 139, the k_{ij}=3.5 (low productivity), predation mortality increased by more than 2-fold up to M_2 = 0.5. In the status quo projection, spiny dogfish biomass continues to increase into the future such that these impacts are amplified over the long term and have a large effect on the ERPs.

11 FULL ECOPATH WITH ECOSIM MODEL (NWACS-FULL)

11.1 Model Overview

An Ecopath with Ecosim (EwE) full ecosystem model was developed for the U.S. Northwest Atlantic continental shelf (NWACS) to inform fisheries management of Atlantic Menhaden (hereafter menhaden) within an ecosystem context. This NWACS model simulated 59 trophic groups and 8 fishing fleets using data from 1985 to 2023 and is built on previous versions of the model (Buchheister et al., 2017a; Buchheister et al., 2017b; SEDAR, 2020b). The major updates and improvements to the previous version of the NWACS FULL model (SEDAR, 2020b) includes the following: 1) all available biomass, catch, fishing mortality, and fishing effort time series were updated to 2023 using recent stock assessment and fishery dependent and independent survey data; 2) the diet composition matrix was updated using a more synthetic analytical approach with updated data; 3) multistanza groups for key ERP trophic groups were updated to match the NWACS-MICE model; 4) osprey (*Pandion haliaetus*) were added as a functional group and bluefin tuna were used as the representative species for the large pelagic highly migratory species (HMS) group; 5) additional time series or key parameters were developed for several trophic groups (anchovies, five zooplankton groups, osprey, baleen whales, odontocetes, and HMS); and 6) a primary production forcing function was developed and evaluated.

The full NWACS model provides a holistic, ecosystem-wide perspective on menhaden fisheries management, intended to complement the NWACS-MICE model. By incorporating predator-prey feedbacks among all groups, this model allows an exploration of the broader impacts of menhaden fishing on the ecosystem, including birds, marine mammals, and other fishes.

11.2 Methods

11.2.1 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 (Christensen & Walters, 2004; Colléter et al., 2015; Pauly et al., 2000). 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 input for each group (solving for the fourth): biomass, production/biomass ratio, consumption/biomass ratio, and ecotrophic efficiency. Typically, ecotrophic efficiency (*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 (Christensen et al., 2008; Christensen & Walters, 2004; Walters et al., 1997).

In Ecosim, biomass dynamics are modeled on a monthly time step as a series of differential equations, where the 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). 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, can be included.

Vulnerability parameters, v_{ij}, 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 vulnerability multipliers (k_{ij}) 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 (Bentley et al., 2024). The k_{ij} parameters must be greater than or equal to 1, with low values restricting flow into the vulnerable state, which thereby limits consumption and prevents any biomass gains in the predator. High k_{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 species with a low initial biomass and low M_{2ij} on their prey, the k_{ij}

parameters must be quite high 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 k_{ij} and then searching for the values that minimize the sum of squared deviations 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. The weights may be adjusted upwards to emphasize fits to species of particular interest. Examples of model fitting procedures are described in (Bentley et al., 2024; Chagaris et al., 2015; Heymans et al., 2016).

11.2.2 Ecopath with Ecosim Model Description

11.2.2.1 The NWACS Ecosystem Model

Spatial structure. The spatial domain for the model is the Northwest Atlantic Continental Shelf ecosystem, spanning the continental shelf of the Northwest Atlantic Ocean from just south of North Carolina to Maine, USA to Nova Scotia, Canada (Figure 46). The model domain includes four continental shelf subregions, following the regional strata of the NEFSC trawl survey: Mid-Atlantic Bight (MAB), Southern New England (SNE), Georges Bank (GB), and Gulf of Maine (GOM). Our model also represents the estuaries along the coastline, such as the Chesapeake Bay, Delaware Bay, and Long Island Sound (Figure 46). 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 Multispecies Virtual Population Analysis (MSVPA) developed for Atlantic menhaden (Garrison et al., 2010) and to other 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 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 441,000 km^2. The model area was calculated from a ¼ degree depth raster, excluding all land and grid cells with depth > 2000m, using the area() function in the R raster package. This is an increase from the area (246,662 km^2) used for the previous ERP assessment, extending farther south and north to be consistent with the NWACS MICE model.

Temporal structure. The model was parameterized using available data for the ecosystem from 1985 to 2023. The initial year, 1985, was chosen because this is the first year of available biomass and catch data for many of the single species-stock assessments.

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 59 different groups.

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 previous Ecopath models in the region, known as the Energy Modeling and Analysis eXercise (EMAX) project (Link et al., 2006, 2008). Given that the initial application of our NWACS model was for menhaden, important menhaden predators (e.g., striped bass, bluefish, weakfish, spiny dogfish *Squalis acanthias*) are represented as individual species, as are alternative prey for those predators (e.g., Atlantic herring *Clupea harengus*, Atlantic mackerel *Scomber scombrus*, anchovies *Anchoa spp.*). Other fish species (e.g., Atlantic cod *Gadus morhua*, summer flounder *Paralichthys dentatus*) that are of particular management concern or ecological significance were also modeled explicitly. Several fishes were partitioned into multiple age stanzas to account for documented ontogenetic differences in diets (Buchheister & Latour, 2015; Garrison & Link, 2000; Smith & Link, 2010) 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 naming of multi-stanza groups, stanzas were labeled as either small (S), medium (M), or large (L), but they represent specific ages and lengths for each species (Table 18). All groups were modeled using biomass densities (mt/km^2).

This version of the NWACS FULL ecosystem model involved some changes to the trophic structure compared to the last benchmark assessment (SEDAR, 2020a), reducing the total number of groups from 61 to 59. First, multistanza definitions were changed to be consistent with the NWACS MICE model. Specifically, the number of stanzas was reduced from 3 to 2 stanzas for Atlantic menhaden, Bluefish, and Weakfish and from 2 to 1 stanza for Spiny Dogfish (Table 18). Also, Atlantic herring were modeled with 2 multistanza groups instead of 1 for greater flexibility to capture trophic changes and food web complexities explored in more detail by the NWACS MICE model. Second, we added Osprey (*Pandion haliaetus*) as a distinct group because 1) this is a menhaden predator of conservation interest to many stakeholders (e.g., (Watts et al., 2024) and 2) we were able to obtain some basic estimates of biomass and an index of relative abundance through time (see Section 3.1.3). Lastly, we chose to use Bluefin Tuna as representative of the broader Highly Migratory Species (HMS) group because this was the HMS species for which we had the most information, including diet studies and stock assessment-derived biomass and mortality rates (see Working Paper SEDAR 102-WP-03).

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

11.2.2.2 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. Here we provide a summary of the data sources.

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. This model builds upon the previous versions of the NWACS FULL model, originally developed in 2017 (Buchheister et al., 2017a; Buchheister et al., 2017b) and updated in 2019 for the last Menhaden ERP assessment (SEDAR, 2020b). The original model, as well as the present version, continues to utilize many parameters from the Energy Modeling and Analysis eXercise (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, and we used these parameters 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. We used stock assessment and fisheries independent survey data to parameterize these groups, when possible. Notable changes from the last benchmark Menhaden ERP Assessment (SEDAR, 2020b) include the development of new input data for anchovies (Working Paper SEDAR 102-WP-05) and the five zooplankton groups (Working Paper SEDAR 102-WP-06).

Biomass

When available, biomass estimates for fished groups were obtained from the most recent stock assessments (Table 19). Data from multiple assessments were combined in cases where there were multiple stocks (e.g., Eastern GOM, Western GOM, GB, SNE Atlantic cod) or multiple species (white and red hake) within the modeled domain. In these situations, absolute biomasses (in mt) were summed across stocks, whereas P/B (i.e., *Z*) were calculated as biomass-weighted averages. All absolute biomasses were divided by the model area (441,000 km^2) to obtain the biomass density in mt/km^2.

A separate R-script was written for each species that reads in the stock assessment model report file, derives Ecopath inputs, and extracts timeseries information for Ecosim. The scripts are intended to reduce errors, maintain consistency across species, enhance reproducibility, and make future updates easier. If a mid-year biomass estimate was not provided, a mean, or mid-year biomass estimate was calculated as $\bar{B}_{a,y} = B_{a,y}(1 - \exp(-Z_{a,y}))/Z_{a,y}$ for age a and year y, as described for the NWACS MICE model (see Section 10.2.1). For the eight species modeled using multiple stanzas (Table 18), mean biomass at age was calculated and summed across Ecopath age stanzas and then converted to units of mt/km^2 for inputs into the NWACS FULL model.

Fisheries-independent trawl survey data were obtained primarily from the Northeast Fisheries Science Center (NEFSC) to parameterize the biomasses of non-assessed species groups (e.g., skates, demersal benthivores, demersal omnivores, demersal piscivores; Table 19). The NEFSC trawl survey is a longstanding fishery-independent monitoring program that has been conducted from 1963 to present, sampling depths from 27-366 m on the continental shelf, including the MAB, SNE, GB, and GOM regions (Azarovitz, 1981). Surveys are conducted in spring and fall. All species captured by the NEFSC trawl were re-classified into the NWACS group definitions, and season-specific 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. Season-specific time series were provided by Andrew Beet (NOAA NEFSC, personal communication). For groups that used the NEFSC time series, we determined which season (spring, fall, or a mean of the two) was most appropriate based on the consistency and magnitude of time series, typically opting to use the mean of the two seasons.

The Northeast Area Monitoring and Assessment Program (NEAMAP) was also used to provide shorter time series of minimum trawlable biomass (mt/km^2) for a few groups that were not well represented in the longer NEFSC survey. The NEAMAP bottom trawl survey is a collaborative, fisheries-independent monitoring program run by the Virginia Institute of Marine Science (VIMS) that has been conducted since 2007, targeting shallow nearshore waters (5–30 m) along the U.S. Atlantic coast from Cape Cod, Massachusetts to Cape Hatteras, North Carolina. Surveys are conducted seasonally in spring and fall using standardized gear and protocols designed to complement offshore NEFSC surveys (Bonzek et al., 2008). Data were provided by Dr. James Gartland and Chris Bonzek (VIMS, personal communication).

Additional group-specific biomass inputs and time series were obtained from the literature or from new analyses using available data. The methods used in these instances can be found in other sections of this report, specifically for zooplankton groups (see Working Paper SEDAR102-WP-06), anchovies (see Working Paper SEDAR102-WP-05), osprey (see Section 3.1.3), and marine mammals (see Section 3.1.3).

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, we calculated an average for each of our age stanzas, weighted by the biomass at age within each stanza. *F* rates were calculated as C/B_{MIDYEAR} using time series from stock assessment. For more details on the mortality estimates for the ERP species, see Section 10.

Consumption/biomass and unassimilated food
Q/B values were primarily obtained from the EMAX models (Link et al., 2006, 2008), and other ecosystem models (Christensen et al., 2009), or empirical relationships (Palomares & Pauly, 1998; Pauly, 1989). 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 (AE) is 1 – UA/Q. We assumed a UA/Q value of 0.2 for carnivorous fishes and higher trophic levels (Christensen et al., 2008). For lower trophic levels, we relied on estimates of UA/Q from the EMAX models, although several of these were increased during the balancing process to balance the detritus group.

Diet composition

The diet composition matrix for the NWACS-FULL model was fully updated for this benchmark assessment, following the same methods described for the NWACS-MICE model (see Section 10.2.7). Briefly, diet data were obtained from food habits programs (e.g., NEFSC, NEAMAP, CHESMMAP, NJOT, RIDEM) and the MSVPA literature database focused on the ERP species used in the ERP Assessments (Table 20; SEDAR, 2015, 2020b). Data were made available at the individual stomach level or summarized by trophic group over broader spatiotemporal scales. These data were combined into a single standardized dataset, with predator and prey taxa assigned accordingly to the model's trophic groups. For multi-stanza groups, predators were defined based on the size-cutoffs for each age class (Table 18), but prey was not classified by age or size because that information was not generally available in the databases. Any unidentified material (e.g., unidentified crustacean, unidentified fish, unidentified matter) was sequentially apportioned to any appropriate identified prey groups based on their relative proportions.

Probabilistic representations of fish diets were estimated following the methods of Masi et al. (2014), as described in (see Section 10.2.7). A bootstrapping approach resampled 30% of the diet data (weighted by the square root of the number of stomachs represented by each sample), and a Dirichlet multinomial was fitted to each sample. The maximum likelihood estimate of the diet proportion for each predator was used.

Predator diets were modified to apportion the contribution of any multi-stanza prey groups across age classes. For example, a 21% contribution of menhaden to the diet of medium (age 2-5) striped bass was allocated among the two menhaden age-classes. These allocations were based on size selectivity information for predators when available (including all ERP species) and on general guidelines when size selectivity information was not available (Buchheister et al., 2017a).

Diets for any predator groups that were not included in the updated diet data analysis were set equal to the values used in the last NWACS-FULL model (SEDAR, 2020b). This included the diet composition for Atlantic menhaden, which is based on a literature synthesis because current food habits programs do not adequately sample or describe menhaden diets.

Catch and Effort

Catch and effort data were obtained from the Atlantic Coastal Cooperative Statistics Program (ACCSP), the Marine Recreational Information Program (MRIP), and species-specific stock assessments. Annual commercial landings data by weight within the model spatial domain were obtained from the ACCSP (Katie Drew, ASMFC personal communication), aggregated by NWACS-FULL trophic groups and gear types. All unique commercial gear types were classified into dredge, trawl, trap, gill net, purse seine, longline, or other. Reported landings data for bivalves changed from using "meat" weight to whole-body "live" weight in 1994, and we had data for both from 1994-2023. To account for this change, we developed an average conversion factor (CF = live weight / meat weight) for macrobenthic mollusks (CF=7.11) and megabenthos-filterers (CF=8.25) to rescale meat weights from 1985-1993.

Commercial effort data included the number of trips, weight landed on those trips, total landings for each gear, and proportion of total landings represented by reported trips. Initially, gear-specific time series of effort were developed as: (trip counts * total landings/trip landings). However, these gear-specific effort time series were deemed to be poor representations of effort because of poor data coverage (both through time and within years). Consequently, we resorted to developing gear-specific effort time series under the simplifying assumption that effort was proportional to gear-specific landings (scaled to the 1985 value), as was done in previous versions of the NWACS FULL model. Any confidential data were excluded, but these were typically for the gear-specific information of species groups with very low commercial catch (e.g., anchovies).

Recreational catch was obtained from an online NOAA database that reports MRIP data (https://www.fisheries.noaa.gov/foss), downloading information for all species by state from NC to ME. Recreational data included estimates of catch that was brought back to the dock and could be identified by a trained interviewer (Type A) and catch that was used for bait, released dead, or filleted as identified by anglers (Type B1). Recreational effort data were downloaded from MRIP as annual angler trips for the north- and mid-Atlantic regions for all fishing modes combined, scaled to the initial 1985 value.

For assessed species, we preferentially used the landings data from the assessment reports, as these were more detailed and tended to be larger, and we presumed them to be 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 determined from the landings analysis. For all multi-stanza groups, catch-at-age matrices from the stock assessment were used to partition catch among stanzas.

Biomass accumulation rates

Biomass accumulation rates (BA_i/B) were calculated for all assessed species with a biomass time series. BA 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. For multistanza groups, negative BA results in shifts to the initial equilibrium age structure towards older individuals, while positive BA shift the age distribution towards younger aged fish. Biomass accumulation rates were calculated as the rate of change in biomass per year from 1985-1986 [$(B_{1986}- B_{1985})/ B_{1985}$], based on data availability. BA 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}).

11.2.2.3 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; otherwise, this indicates that the ecosystem predators and fisheries are utilizing more than 100% of a group's production. For the NWACS-FULL model, we parameterized the Ecopath model using all the new and updated values obtained from stock assessments, fisheries independent surveys, the new diet analysis, and group-specific evaluations (anchovy, zooplankton groups, osprey, marine mammals, HMS). Any missing parameters were taken from the last, peer-reviewed Menhaden ERP Assessment (SEDAR, 2020b), which relied heavily on the original NWACS-FULL model (Buchheister et al., 2017a; Buchheister et al., 2017b).

After the initial parameterization, there were 19 out of 59 groups with *EE*>1, and these were sequentially and systematically addressed, starting with groups that were most out of balance. Numerous and diverse changes were made during the balancing process, and we note some general principles or issues uncovered (as opposed to an exhaustive, detailed list):

1. The balancing process uncovered some data-processing errors (e.g., coding issues), which were investigated and corrected.
2. *EE*s tended to be high for many of the youngest stanzas of the multistanza groups, because the Ecopath software estimates the biomass of these groups from an older "leading" stanza. Those estimates assume body growth follows a von Bertalanffy growth curve with weight proportional to length-cubed and the species has reached a stable age-size distribution (Christensen et al., 2008), but often those estimates are lower than what is estimated from the stock assessments. This was typically solved by increasing the BA rate (as informed by rates calculated from the biomass time series), which increased the biomass for the younger stanza.
3. Changes to *B* or *Z* for any assessed species (especially the ERP spp.) were typically avoided to maintain consistency with published stock assessments. However, changing *B* or *Z* of non-assessed groups would be considered.
4. BA rates and diets would be preferentially adjusted to bring groups into balance. Diet changes tended to focus on the predators that contributed to the highest M_2 values for the out-of-balance prey group. Some predators with relatively high biomasses (e.g., Spiny Dogfish, Bluefish, Hake, Skate) were common contributors to high *EE* of their prey groups, but relatively small changes to their diets could have big impacts on prey *EE* values.

11.2.3 Ecosim Model Configurations & Calibration

As with all ecosystem models, there is no single, fully objective method for arriving at a final model that best replicates historical trends in biomass and 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 NWACS-FULL model was calibrated to the observed biomass and catch time series in three different phases using methods similar to the last ERP assessment. Phase A was an initial fitting using an iterative process to optimize the model residual sum of squares under four model configurations. Phase B involved modifications of the best fit from Phase A to improve realism and species-specific diagnostics. Lastly, Phase C involved an ad hoc adjustment to further improve model diagnostics.

11.2.3.1 Phase A (Initial fitting)

For Phase A of the calibration, we fit four different models, each using an iterative process. The four model configurations were based on a 2x2 factorial combination of: a) whether or not an annual primary production forcing function was used, and b) whether all starting k_{ij} for ERP species were set at a value of 2 (as for all other groups) or whether the starting k_{ij} for ERP species were estimated based on the unfished biomass to the starting 1985 biomass [B_{unf}/B_0] as done for the NWACS-MICE model (Section 10.2.1). The annual primary production (PP) forcing function was developed from depth-integrated chlorophyll *a* concentration from the Global Ocean Reanalysis products (GLORYS) hosted by the Copernicus Marine Service (Section 10.2.5).

The "Fit to Time Series" utility in Ecopath was used in which the most sensitive vulnerabilities for N-1 different predator-prey interactions were estimated by reducing the residual sums of squares (SS) of the model fits, where N is the number of observed time series used to fit the model (N=77 for this model). This process was done iteratively until there were no substantial reductions in SS or AIC. No more than 76 k_{ij} were fitted during each tuning iteration, but different vulnerability values could be estimated during each iteration.

When fitting models in Ecosim, biomass and catch time series were weighted based on the data source and importance of the species group. Time series for the six ERP species had a weight of 10, time series that came from stock assessment reports had a weight of 4, and all other time series (typically from surveys or databases) had a weight of 1. For multistanza groups, biomass and catch timeseries for the "leading" stanza of each species were treated as an absolute measure, whereas all non-leading stanzas were treated as a relative measure. The feeding time adjustment rate was set to 0.5 for the youngest stanza (and set to 0 otherwise) to allow for compensatory recruitment effects as recommended by Christensen et al. (2008). Also, the prey switching parameter was assumed to be 1.0 for all modeled groups, allowing groups to disproportionately consume more of a prey as it becomes more abundant at an intermediate rate (this parameter can range from 0 with no switching to 2 for rapid switching).

11.2.3.2 Phase B (Vulnerability adjustments)

For Phase B of the calibration, three different types of vulnerability adjustments were made to the best fit from Phase A. First, manual vulnerability adjustments were made to the minimum k_{ij} for the ERP species to improve their F_{MSY} and stock-recruitment dynamics. The minimum vulnerability value for a selected ERP species was evaluated iteratively using the "F_{MSY}" tool within Ecopath. For this, species-specific fleets were created manually to change *F* across all age stanzas, and then the tool estimates the equilibrium yield after 40 years when fishing effort is multiplied from 0 to a maximum, user-specified amount (typically 10-20 fold). Often, with minimum k_i =1, predator j could sustain unrealistically high levels of fishing without having a decline in relative catch, or the yield curves would not be smooth due to dynamic instabilities in the model projections. The minimum k_i values for a given ERP species j would be iteratively adjusted from 1 to 1.01, 1.05, 1.1, 1.2, and 1.5 to generate a new yield curve. The lowest value that generated a smooth, dome-shaped yield curve was selected as the optimal minimum value. This was done separately for each ERP species, yielding v_{min} values between 1.01 and 1.2. To evaluate stock-recruitment dynamics for the five ERP species with multistanzas, the model was run simulating a 10-35 fold increase in fishing effort followed by a linear decline of fishing effort to zero (over ~10 years), which then lasted 15-20 years. This was done separately for each ERP species, allowing the emergent stock recruitment relationship between the biomasses of the youngest and oldest stanzas to be plotted. No changes were deemed necessary to improve stock-recruitment relationships because all evaluated ERP groups showed some degree of compensation, as would be expected.

The second and third vulnerability adjustments explored during Phase B of the calibration were to apply a minimum k_{ij} value of 1.01 for all groups and apply a maximum k_{ij} value for all groups. After completing Phase A of the fitting process, the estimated k_{ij} could range from 1 to 10^9. We employed an arbitrary minimum vulnerability value of k_{ij}=1.01 instead of k_{ij}=1 to minimize the chance of explosive population growth and unstable dynamics. 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, similar to the MICE model (Section 10.3.3).

11.2.3.3 Phase C (Additional vulnerability adjustment)

After conducting projections using the best model from Phase B, we decided to make a minor adjustment to two vulnerability parameters to further improve model behavior for Atlantic menhaden (see results for details).

11.2.4 Projections

The NWACS-FULL model (with sim 2.9 as the base run; see results) was used to run 40-year projections under different menhaden fishing scenarios. Projections were conducted under the following conditions: 1) all ERP species were held at their target fishing mortality rates (F_{TARGET}), 2) all other F rates for non-ERP species were held at their status quo (i.e., 2023) levels, and 3) all fleet-specific fishing effort was maintained at 2023 status quo levels. Simulations were run at 50 different menhaden F rates by scaling menhaden F_{2023} rates (for each age stanza) using F-multipliers that ranged from 0 to 20 (i.e., F=F_{2023}*F-multiplier). F-multipliers were used because age-classes and abundance-based F-values from stock assessments did not correspond with biomass-based, stanza-specific F rates used within EwE (Section 8, Table 3, Table 4). F_{2023} = 0.0009 for small (age-0) menhaden and F_{2023} = 0.1274 for large (age-1+) menhaden based on the calculations from the single species stock assessment model.

Projections were run for 40 years for all menhaden F scenarios, and the relative equilibrium biomass (B_{2063} /B_{SQ}) was calculated, where B_{2063} was the equilibrium 2063 biomass for a given F scenario and B_{SQ} was the equilibrium biomass for the status quo menhaden fishing scenario (i.e., when the F-multiplier=1). Equilibrium catches are presented relative to the maximum equilibrium catch across all menhaden fishing scenarios (C_{2063}/$C_{\max}$), which would typically occur when there was no fishing on menhaden (F=0) or when fishing on menhaden was at its max ($F_{\max}$). Tabulated results focus on three specific menhaden F scenarios: no menhaden fishing (F_0), status quo (2023) menhaden fishing (F_{SQ}), and the maximum menhaden F ($F_{\max}$), which is twenty times greater than F_{SQ}.

For the ERP species (excluding menhaden), time series of projected biomasses were compared to proxies of biomass targets and thresholds developed from their respective single species stock assessments (see Section 8). These proxies were calculated using biomass multipliers (e.g., $B_{\text{TARGET}}=B_{2023}*B_{\text{TARGET multiplier}}$), based on a similar reasoning to the F-multipliers used above.

11.3 Results

11.3.1 Ecopath Model

The balanced Ecopath model output is presented in Table 21. The food web was highly interconnected and complex, with a total of 1,083 trophic links in the system and an average of 19.3 links per trophic group (Figure 47, Table 22). This was an increase of 21% compared to the NWACS FULL model in the last ERP assessment (15.6 links/group), due to the update of the diet matrix and restructuring of trophic groups.

Menhaden were consumed by a total of 24 predator groups (40.7% of the modeled trophic groups), but six of those predators had trace contributions (<0.05%) of menhaden in their diets (Figure 48). Menhaden contributed a substantial portion of the diet of some predators, notably large (age-6+) striped bass (48%), nearshore piscivorous birds (33%), large pelagics (25%), and medium (age 2-5) striped bass (22%).

Atlantic menhaden M_2 was estimated as 1.139 yr^{-1} for age-0 menhaden and 0.458 yr^{-1} for age-1+ menhaden for 1985 (Table 21). The predators that contributed the most to age-0 menhaden M_2 included nearshore piscivorous birds, bluefish, striped bass, and odontocetes, whereas spiny dogfish and nearshore birds contributed the most to age-1+ menhaden M_2 (Figure 49).

Atlantic menhaden *EE*s (or the proportion of the total production used in the system) were estimated to be 0.841 for age-0 menhaden and 0.595 for age-1+ menhaden (Table 21). Although not as close to unity as we expected for a full ecosystem model, these values were higher than obtained from the previous version of the NWACS-FULL model, with *EE*s ranging from 0.151-0.452 across three menhaden stanzas (SEDAR, 2020b).

11.3.2 Summary of Calibrated Models

Of the four models fit during Phase A of the calibration, we determined sim 2.1 to be the best (Table 23). Sim 2.1 did not use the PP forcing function and it relied on initial vulnerabilities estimated based on B_{unf}/B_0 for the ERP species. This model had the lowest SS (albeit only 2-4% less than the others), but it also had the lowest number of estimated vulnerabilities (n=230, including the 12 initial k_{ij} values estimated for the ERP stanzas) and the fewest number of values on the lower bound. Overall, sim 2.1 had ~4 k_{ij} parameters estimated per time series, and 21% of all possible vulnerabilities deviating from their initial starting values.

Phase B of the calibration indicated that model fits were very sensitive to most of the vulnerability adjustments made to sim 2.1 (Table 23). Applying the vulnerability minimums and caps increased the SS by 33-94% from sim 2.1, with the exception of the manual vulnerability adjustments made to the 6 ERP species (sim 2.5), which only had a 4.7% increase in the SS. We chose the sim 2.5 as the best compromise between maintaining good fits to the observed time series while also improving the F_{MSY} diagnostics for the ERP species, even though the model retained estimated parameters on the lower and upper bounds.

For Phase B of the NWACS-FULL model calibration, manual adjustments were made to the minimum k_{ij} for each ERP species to improve their F_{MSY} diagnostics. Prior to the adjustments (left panels in Figure 50), the yield curves for Atlantic menhaden, Atlantic herring, bluefish, and weakfish indicated varying levels of stability as well as some indication of being asymptotic. Relatively minor changes were made to the minimum k_{ij} for these four species, increasing them from 1 to 1.01-1.2 (Table 24). This led to improved yield curves that were dome-shaped with less instability and asymptotic behavior (right panels in Figure 50), although some minor issues remained. We thought the instability that remained in the Atlantic menhaden yield curve (upper right panel in Figure 50) would not be an issue when doing projections from the terminal year (instead of from 1985). In these diagnostic plots, most attention was placed on the predictions with full compensation, which allows for predator-prey feedbacks as opposed to those in a stationary system in which those feedbacks are turned off. No changes were made to the minimum k_i for striped bass or spiny dogfish because all of those parameters were greater than 2 (Figure 51). Across the six ERP species, these diagnostics suggest that increasing fishing effort by 2-10 fold from 1985 levels would generate MSY. Given the complexities of the NWACS-FULL model and sensitivities of these F_{MSY} values to small changes in vulnerabilities, we do not recommend using these values in a concrete fashion, but their plausibility is used as a model diagnostic (Heymans et al., 2016).

No changes were deemed necessary to the vulnerabilities to improve stock-recruitment relationships because all evaluated ERP groups showed some degree of compensation reflective of a Beverton-Holt relationship (Figure 52).

For Phase C of the calibration, we chose to increase the minimum k_i for Atlantic menhaden from 1.1 to 1.25 in sim 2.9, affecting only two of the 230 estimated vulnerability parameters (Table 24). This was done because some instability in the Atlantic menhaden yield curve was still evident when sim 2.5 was used for projections; originally, we thought it would not be a problem because projections were done from the terminal year (as opposed to the initial year) and would not be as affected by the biomass accumulation rates in Ecopath. This minor change from sim 2.5 to 2.9 had negligible impact (<0.05%) on the model SS (Table 23) and projection results (see next section), but it provided smoother projections across different menhaden fishing mortality rates. As a result, we chose to use sim 2.9 as the base run for the NWACS-FULL model.

For sim 2.9, a total of 218 vulnerability parameters deviated from their starting values, with 21 manually adjusted for ERP species (Table 24), 49 on lower bounds, and 57 on upper bounds (Table 23, Figure 53). Many groups did not have any k_{ij} estimated on bounds, whereas some groups had >4 k_{ij} estimated on lower bounds (e.g., bluefish (L), weakfish (L), Atlantic cod (L), summer flounder (L)) or upper bounds (demersal piscivores; Figure 53). All specific predator-prey pairs whose k_{ij} were on a lower or upper bound are listed in Table 25.

11.3.3 Fits to time series

Ecosim predictions from 1985-2023 generally corresponded well to observed historical trends in biomass, but fits varied based on the specific simulation (Figure 54). Overall, simulations 1.2, 2.1, 2.9, 3.1, and 4.1 had the best fits based on SS (Table 23), and these were comparable if not better than the biomass fits for the base run from the last Menhaden ERP Assessment (SEDAR, 2020b). For a large model with many groups, it is common for some time series to fit better than others due to tradeoffs and challenges in fitting all time series, which is one of the reasons the fits to ERP species were weighted more heavily. For some groups, all simulations tended to generate similar predictions (e.g., Atlantic mackerel, butterfish, copepods), whereas predictions could be quite variable across simulations, particularly for the ERP species (Figure 54). Biomass predictions for several groups were relatively stationary through time, particularly for groups that did not have *F* as a driver, including many invertebrates (small and large copepods, gelatinous zooplankton, micronekton) and some other groups (demersal omnivores). Some fits were poor and did not capture the overall trends in the assessment's time series (demersal benthivores, osprey, skates, small pelagics, weakfish). Some species had strong recruitment events based on stock assessments (e.g., haddock, yellowtail flounder) that were not represented in model predictions. Fits for the ERP species were good, reflecting both magnitude and trend for the best simulations, with the exceptions of spiny dogfish and weakfish, which had poor fits (Figure 55). For clarity, model fits to all biomass time-series for the base run alone (sim 2.9) are shown in Figure 56.

Model fits to the observed catch time series were also generally good, with variability by group and simulation (Figure 57). Catches for many groups were predicted well and able to capture interannual variability in many cases (e.g., Atlantic cod, butterfish, demersal benthivores, shrimp, small pelagic fishes, summer flounder, yellowtail flounder). Many of the cases of poor model fits can be attributed to the absence of detailed information on fishing mortality (e.g., alosines, Atlantic croaker, demersal omnivores and piscivores, macrobenthic crustaceans and mollusks, medium pelagic fishes, sharks, and skates); for 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 the catch data for ERP species were particularly good for the best simulations (Figure 58), although fits were more intermediate for large (age 2+) Atlantic herring, small (age 0) Atlantic menhaden, and large (age 1+) weakfish (Figure 58). For clarity, model fits to catch time-series for the base run alone (sim 2.9) are shown in Figure 59.

11.3.4 Diet Composition

The predicted proportion of menhaden in the diets of its predators varied over time and across model runs but generally showed an increasing trend in the base run for all or most of the time-series for the majority of predators, although the proportion for some groups like seabirds, other demersal piscivores, coastal sharks, and odontocetes was relatively flat (Figure 60). For large striped bass (age 6+), the proportion of menhaden in the diet was predicted to decrease slightly from the beginning of the time-series and then increase from the mid-1990s onward for most runs, stabilizing around 60% in the last several years in the base run. The prediction of menhaden in the diets of medium striped bass (age 2-5) was more variable across model runs, with the base run predicting a similar trend to large striped bass (age-6+) but at a lower scale, reaching approximately 20% by the end of the time-series, while other runs predicted an increase to around 60% with significant interannual variability or no increase at all, remaining at approximately 10% for most of the time-series (Figure 60). The contribution of menhaden to the diets of spiny dogfish increased gradually over the time-series in the base run, reaching approximately 30% by the end of the time-series, while other runs showed no trend or a declining trend (Figure 60). For adult bluefish and weakfish, menhaden percent diet increased gradually over time in most model runs, with the proportion in bluefish diets showing a slight decline from the late 2010s onward. In the base run, Atlantic menhaden was approximately 10% of the diet for adult weakfish and 40% for adult bluefish at the end of the time-series (Figure 60). Osprey and nearshore piscivorous showed an increasing trend in the base run as well, with the proportion of Atlantic menhaden in their diets reaching approximately 25% for osprey and 50% for nearshore piscivorous birds.

In contrast, the proportion of Atlantic herring in the diets of most predators increased at the beginning of the time-series and declined after the mid-1990s for most model runs (Figure 61), and the base run trends were less variable across the ERP species. The proportion of Atlantic herring in the diet of ERP predators reached 10% at the end of the time-series for large striped bass (age-6+), less than 5% for medium striped bass (age 2-5), and less than 1% for bluefish, weakfish, and spiny dogfish.

11.3.5 Menhaden mortality output from Ecosim

Atlantic menhaden mortality rates estimated using sim 2.9 show the relative contributions of fishing (*F*), predation mortality (M_2), and unexplained mortality (M_0) to total mortality (*Z*) through time (Figure 63). *F* represents a small proportion of total instantaneous mortality for age-0 and age-1+ menhaden, and M_2 was the bulk of the estimated mortality (Figure 63). M_0 (the difference between *Z* and F+M_2) was much larger for age-1+ menhaden, consistent with the lower menhaden *EE* values from Ecopath. Predator contributions to M_2 through time were relatively consistent from 1985 to 2023, but they also highlight the relative transition of striped bass being a more dominant predator of age-0 menhaden to spiny dogfish and large pelagics for age-1+ menhaden (Figure 62).

11.3.6 Projection Results

The NWACS-FULL model (with the base run parameterization of sim 2.9) was projected forward for 40 years under different menhaden *F* rates, while holding ERP species at their *F* targets and other species at status quo levels. These projections demonstrated the degree of sensitivity of different functional groups to these changes in fishing pressure. Many functional groups had negligible biomass and catch responses to the alternative menhaden *F* scenarios, whereas several had substantial changes (Figure 64, Figure 65). Of the ERP species, menhaden exhibited the strongest biomass and catch responses as expected, being driven to near extinction at the highest menhaden *F* scenario (Figure 66, Figure 67). As menhaden *F* increased, striped bass and spiny dogfish were strongly negatively affected, Atlantic herring and weakfish had negligible effects, and bluefish were strongly positively affected (Figure 66, Figure 67). The non-menhaden ERP species did not reach their respective biomass threshold or target proxies, regardless of menhaden *F* rates, except for bluefish, which only reached its biomass target proxy at an intermediate (i.e., higher than status quo) menhaden *F* rate (Figure 66).

Plotting the terminal year relative biomass and relative catch for each group, is one way to visualize the "winners" and "losers" under the different menhaden *F* scenarios (Figure 68, Figure 69, Table 26). Results for the F_{max} scenario represent predicted changes by the model under these more extreme conditions (Table 26). In the menhaden F_{max} scenario, the groups whose biomasses declined most relative to the menhaden F_{SQ} scenario (i.e., the "losers") were menhaden (-97% decline), medium pelagic fishes (-90%), nearshore piscivorous birds (-70%), osprey (-67%), haddock (-64%), and striped bass (-61%; Figure 68, Table 26). Of these groups that are harvested, the equilibrium 2063 catch relative to C_{max} was the following: Atlantic menhaden (-81%), medium pelagic fishes (-91%), haddock (-65%), and striped bass (-55%; Figure 69, Table 26). Groups that could be considered relative biomass "winners" in the menhaden F_{max} scenario were demersal benthivores (+146%), bluefish (+138%), yellowtail flounder (+43%), megabenthos other (+31%), and skates (+30%; Figure 68, Table 26), all of whom achieved their highest equilibrium catches in the Fmax scenario (Figure 69, Table 26).

Changes in relative biomass and catch in the menhaden *F*=0 scenario were much less substantial, given that this was not a large change in *F* from F_{SQ} (e.g., F_{SQ}=0.127 for menhaden age 1+). The "losers" in this scenario typically only had relative biomass declines of <2% (Figure 64, Table 26). However, C_{2063}/C_{max} in the F_0 scenario tended to be low for the groups that had the largest relative *B* increases in the F_{max} scenario (Table 26). The "winners" in the F_0 scenario typically only had modest increases (<5%), except for nearshore piscivorous birds (+34%), menhaden (+8%), osprey (+7%), striped bass (+6%), and spiny dogfish (+5%; Figure 68, Table 26).

11.4 Discussion

11.4.1 Model Assumptions, Uncertainties, and Challenges

The NWACS-FULL ecosystem model is a simplified representation of a complex ecosystem. The model assumes that the primary drivers of change in ecosystem dynamics are fishing pressure, trophic interactions, and forced changes in primary production (in the simulations with PP forcing). The present version does not account for spatially explicit dynamics nor other environmental drivers (e.g., abiotic influences on fish recruitment), but these are components that could be developed and added in future versions. The model serves as a tool for describing basic ecosystem structure and dynamics, exploring and generating hypotheses, identifying emergent patterns, and helping inform management considerations. Compared to single-species models, they incorporate more assumptions and uncertainties, and results should be interpreted with appropriate caution.

The inability of the model to fit all timeseries (n=77) equally well highlights a common tradeoff in ecosystem models. We addressed this by preferentially weighting fits to the ERP species time series, and we acknowledge there can be tension between fitting different datasets. Poor fits for many groups were influenced by a lack of reference time series or a lack of group-specific fishing pressure (as *F* or Effort), indicating the continued need for reliable information to parameterize the model.

Model projections were made assuming specific fishing rates for all modeled groups (F_{TARGET} for ERP species and status quo fishing for all other groups and fleets) and static environmental (e.g., PP) conditions. Predicted patterns are contingent on these specified conditions, and results would differ under alternative assumptions of future fishing pressure or environmental conditions.

Projection results identified several groups that were sensitive to the menhaden *F* scenarios, but it is important to note that projections, especially for the F_{max} scenario, should be examined cautiously. Many of the groups with the largest positive biomass and catch responses (i.e., winners) under the F_{max} scenario were driven by indirect trophic interactions, whereas the losers tended to be direct menhaden predators (Table 25). For winners in the F_{max} scenario, the mechanisms driving such changes in a complex ecosystem model can be hard to discern. Most of these groups (except for bluefish, yellowtail flounder, and weakfish) are not driven by a group-specific fishing mortality rate, and time series of relative biomass (e.g., from multispecies surveys) may not be representative. In the case of demersal benthivores, the large B_{2063}/B_{SQ} is misleading due to the very small B_{SQ} estimate for that group (Figure 64). Many of the losers in the F_{max} scenario (e.g., medium pelagic fish, nearshore piscivorous birds, osprey, seabirds) also lacked robust information on initial biomasses, mortality drivers, or temporal trends for the model to fit these groups well. For example, no reliable estimates of nearshore piscivorous bird biomass were developed, and we assumed their biomass was equal to the EMAX estimate for seabirds (Buchheister et al. 2017a; Buchheister et al., 2017b; SEDAR, 2020b). Lastly, in the F_{max} scenario, menhaden are nearly extirpated from the system, and using the model to extrapolate to conditions that deviate substantially from historical conditions could lead to unexpected model predictions.

Projection results can be sensitive to specific vulnerability parameters, especially when estimated near the bounds (e.g., $k_{ij} = 1$). For example, bluefish exhibited an unexpected increase in biomass as menhaden (an important prey species) were fished out of the system. Exploration of the vulnerability (k_{ij}) parameters revealed that this dynamic was largely driven by a very low k_{ij} =1.01 for large (age-2+) bluefish feeding on large (age-1+) menhaden. Thus, bluefish consumption (and consequently biomass) had negligible direct responses to menhaden declines but instead was affected indirectly by other groups in the system. Substantial increases to this parameter reduced the magnitude of the bluefish response, but at the cost of significantly degrading the fit to historical bluefish data. This illustrates the tension between historical realism and predictive behavior in EwE models and underscores the importance of vulnerability settings in shaping model outcomes (e.g., Bentley et al., 2024; Heymans et al., 2016). We acknowledge that alternative model parameterizations could lead to different model behavior (e.g., Mackinson, 2014; Figure 54, Figure 57). Comprehensive sensitivity analyses could be conducted in the future to better quantify uncertainties related to vulnerability parameters as well as other input parameters (e.g., (Chagaris et al., 2017; Heymans et al., 2016; Steenbeck et al., 2018).

As with the previous NWACS-FULL model (SEDAR, 2020b), we note that the overfished ERP species (striped bass, Atlantic herring, and weakfish) did not recover in projections when these species were fished at their target *F* rates regardless of the menhaden *F* scenario. 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. For example, striped bass productivity and degree of recovery was later found to be sensitive to the “switching power parameter” for spiny dogfish, which had not been initially evaluated or modified (unlike for the MICE model). The lack of recovery for those groups could be influenced by the following: 1) the B_{TARGET} and F_{TARGET} proxies developed for the EwE framework using relative multipliers may not be fully representative of the single species assessment models, 2) the specific parameterization of the base run of the NWACS-FULL could underestimate species recoveries, and 3) management targets from single species models may overestimate species recoveries without accounting for broader ecosystem dynamics. For example, the ability of striped bass to reach their biomass target under their *F* target in the single-species stock assessment depends on the recruitment regime assumed for future years; striped bass did not reach their spawning stock biomass target under the current F_{TARGET} if recruitment remains at the very low levels observed since 2019 (ASMFC, 2025). The NWACS-FULL model does not incorporate changes in the stock-recruitment relationship over time or environmentally-driven deviations in recruitment, so it is unclear if this agreement between the NWACS-FULL model and the single-species model is a result of the NWACS-FULL model picking up on signals in the input data of the current lower productivity of the striped bass stock, or simply an artifact of uncertainty in the vulnerability parameter estimates.

Osprey and haddock are two groups that represent some of the challenges in fitting a large ecosystem model. First, despite the explicit addition of osprey into the model with a time series of relative biomass, the model did not adequately capture the recovery of this group through time, likely because the model does not account for things like productivity increases associated with DDT banning and successful habitat restoration efforts (Watts & Paxton, 2007). The model is also not designed to address more spatially explicit osprey concerns associated with menhaden food supply in places like the lower Chesapeake Bay (Watts et al., 2024). Second, haddock biomass and catch timeseries were also not predicted well by the model, and this was one of the sensitive groups in the Fmax scenario. Haddock stocks have experienced very large, environmentally driven recruitment events which were not explicitly included in the NWACS-FULL. For both species groups, it is possible to add forcing functions to represent such productivity drivers, but it was beyond the scope of the present modeling effort to do so. Ongoing research on ecological mechanisms and additional species-specific expertise would be beneficial if such options were to be explored in the future.

Lastly, we assumed that all of the spiny dogfish biomass estimated from the stock assessment resided within the model domain, but this assumption should be revisited based on the broader-scale movement of some individuals (Carlson et al., 2014), the relatively large biomass of the group within the model, and the large estimated contribution of this group to menhaden M_2.

11.4.2 Main Findings

Fisheries management in an ecosystem context requires acknowledgment and evaluation of tradeoffs among trophic groups, fisheries yield, and system structure. The NWACS-FULL model is a tool that informs this tradeoff analysis by focusing on fishery-driven predator-prey feedbacks, complementing the NWACS-MICE and single species menhaden models. Despite the complexities, uncertainties, and challenges highlighted, it helps generate and explore ecological hypotheses, provides information on the potential directions and patterns of change for diverse species groups, and identifies important research gaps in the system (Christensen & Walters, 2011).

Striped bass was the ERP species (excluding menhaden) that was most negatively sensitive to projected increases in Menhaden fishing rates. Relative to the F_{SQ} scenario, striped bass biomass declined by 61% in the extreme F_{max} scenario where menhaden were effectively extirpated from the system, and striped bass biomass increased by 6% if menhaden fishing ceased (Table 24). These values bracket the range of striped bass effects predicted by the NWACS-FULL model under the base run, and minor changes to menhaden *F* rates would have smaller impacts on striped bass (Figure 68). The projected changes to striped bass biomass had corresponding changes to striped bass relative catch.

The sensitivity of striped bass to menhaden fishing in the NWACS-FULL model is a more robust result despite the model uncertainties highlighted (e.g., unexpected bluefish results) for a variety of reasons:

- Striped bass age stanzas did not have any vulnerability parameters estimated on the lower bound and only one k_{ij} parameter on the upper bound (for a non-menhaden prey group). For comparison, bluefish had 5 k_{ij} initially estimated on lower bounds (including for menhaden) and 3 on upper bounds.
- Striped bass historical data were well represented by NWACS-FULL predictions and those fits were not as sensitive to minor changes in vulnerability parameters (unlike bluefish).
- Menhaden are a substantially more important prey group in the diets of striped bass compared to other ERP species (Figure 60).
- Through various parameterizations and updates of the NWACS FULL model (2017, 2019, 2025), striped bass have repeatedly emerged as one of the most sensitive predator groups, unlike the other ERP species whose responses to menhaden fishing have been more muted or variable.
- The NWACS-MICE model also affirms striped bass as a more sensitive predator, and that a simplified model avoids the many additional challenges and uncertainties inherent in the NWACS-FULL model stemming from its complexity.

Consequently, the ERP WG recommends the continued focus on striped bass as a sensitive predator that should be used to inform ERP development.

The NWACS-FULL model identified other groups that were “losers” (e.g., the three different bird groups, medium pelagic fishes, Haddock) or “winners” (e.g., bluefish, demersal benthivores) as menhaden fishing mortality increased. Compared to the 2020 menhaden ERP assessment

(SEDAR, 2020b), the updated NWACS-FULL model showed a greater number of groups affected in the F_{max} scenario, likely due to the higher menhaden *EE* values and stronger dietary linkages to predators (particularly striped bass) in the present model. However, these results are less robust than for striped bass due to reduced data quality and availability, indirect ecological mechanisms that are harder to predict, and greater sensitivity to model parameterization. These groups would benefit from additional research and data (particularly the bird groups), as well as further ecosystem model development that incorporates additional productivity drivers beyond trophic, fishery, and primary production mechanisms.

Projections under the F_0 scenario indicated that biomass responses of most groups would be relatively small, between -1% to +8% relative to the F_{SQ} scenario (Table 23, Table 24), except for nearshore birds (+34%) whose input data have higher uncertainty. These relatively minor responses in the ecosystem were influenced by the relatively low terminal year (i.e., 2023) fishing mortalities estimated for small (age-0) menhaden ($F_{2023} < 0.01$) and large (age-1+) menhaden ($F_{2023} = 0.127$). This suggests that small changes to menhaden fishing rates away from F_{SQ} are not expected to lead to substantial changes in biomasses for the modeled groups.

A large proportion (~50%) of Atlantic menhaden predation mortality came from predators that are not ERP species (Figure 62), and thus not included in the NWACS-MICE model. This included nearshore piscivorous birds, seabirds, odontocetes, baleen whales, large pelagics, and demersal piscivores. Enhanced efforts were made to improve parameterization of several groups (birds, marine mammals, large pelagics), but data availability and quality for these groups is reduced compared to fish stocks that are monitored and assessed regularly. Continued research into these groups is recommended to evaluate their influence on menhaden dynamics.

The NWACS FULL model can provide information on the relative scaling of biomass estimates coming from single-species assessments. The low menhaden *EE* in the 2020 NWACS-FULL model was hypothesized to be partially caused by a mismatch in the scale of the menhaden biomass estimate from the single species model compared to other ecosystem components. The new menhaden assessment has borne out that hypothesis with the substantial reduction in total menhaden biomass, largely caused by the reduced estimate of natural mortality. Developing the present NWACS-FULL model suggested a potential biomass scaling issue for Atlantic herring, with the assessment-estimated biomass substantially lower than necessary for the consumptive demand in the model. Atlantic Herring displayed high *EE* (>1) in the initial fitting stages, and it was a challenge balancing this group within the model. The NWACS-MICE model estimates a high ecotrophic efficiency (*EE*>0.9) despite not including many other herring predators. Such findings underscore one of the benefits of the NWACS-FULL model in synthesizing available information at an ecosystem scale to look for anomalies and identifying areas of further investigation.

As ecosystem approaches to fisheries management continue to develop, tools like the NWACS-FULL model can help guide long-term thinking, inform ERP development, and assist in management decision-making. This model leverages and integrates single-species stock assessment models to provide a fuller, more thorough description of a complex ecosystem by explicitly accounting for predator-prey feedbacks and quantifying the tradeoffs among different management scenarios. However, given the challenges and uncertainties in modeling the full ecosystem, the NWACS-FULL model is intended as a supporting model to the NWACS-MICE and single species assessment models.

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

Some of the earliest multispecies work done was to connect virtual population analysis models together with predation functions (Gislason & Helgason, 1985; Helgason & Gislason, 1979; Livingston & Jurado-Molina, 2000; Sparre, 1991). This modeling approach can be helpful in a complex fisheries modeling environment because 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 (Curti et al., 2013; Lewy & Vinther, 2014; van Kirk et al., 2010). The goal of these multispecies approaches is to create more realistic information on which to base fisheries management practices (Gislason, 1999; Moustahfid et al., 2009). The multispecies statistical catch-at-age model developed for the 2020 ERP benchmark (hereafter referred to as Virtual Assessment for the Description of Ecosystem Responses, or VADER) was constructed around the six ERP species: Atlantic menhaden, striped bass, bluefish, weakfish, Atlantic herring, and spiny dogfish. The VADER model adopts the more progressive statistical approach for its modeling methodology and works from the strong foundational work of Curti et al. (2013).

The main critique from the 2020 peer review of the VADER model was that it lacked bottom-up feedback. The concept of bottom-up feedback in ecosystem models emphasizes how prey abundance can influence predator growth and mortality. To look into the empirical information to determine if this relationship could be modeled, explorations focused on identifying whether relationships exist between prey availability and predator growth (e.g., weight-at-age or length-at-age metrics). This exploration addressed critical questions:

- Does prey biomass (e.g., menhaden or aggregated prey) influence predator weight-at-age (WAA)?
- Are there significant density-dependent effects within the predator population?
- How do these interactions vary across age groups and temporal scales?

This exploratory work is recounted in detail in Working Paper SEDAR 102-WP-08. The one relationship that emerged as significant from the empirical data was in the striped bass length-at-age dataset, where striped bass length-at-age was positively related to total menhaden biomass. Schiano et al. (2024) developed a simulation model with bottom-up feedback functionality between menhaden and striped bass, which had a connection to length-at-age for striped bass where lower menhaden biomass resulted in lower striped bass length-at-age and therefore higher natural mortality. These calculations were built into VADER as an exploration of this type of effect in the VADER model explicitly for striped bass.

However, the inclusion of this relationship caused model performance to degrade significantly, and the issues could not be resolved in time for the completion of this report. The ERP WG did not recommend using the results for management. A full description of the VADER model and the bottom-up feedback mechanics are available in Working Paper SEDAR 102-WP-09.

13 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, 2017b) 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, which were peer-reviewed and adopted for management in 2020.

13.1 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) which explicitly include consideration of menhaden's role as forage. However, the value of these models is not in a single MSY-based reference point but rather in the ability to evaluate the tradeoffs between fishing pressure and stock biomass across a number of species and fisheries.

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 recommends 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.

13.2 ERP Target and Threshold

The ERP WG recommends 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 Atlantic Menhaden Management Board adopted specific definitions of the ERP targets and thresholds for management use in 2020, so for this assessment, the ERP WG put forward proof-of-concept values of an ERP target and an ERP threshold based on these definitions. However, the final values for the ERP target and threshold will be a management decision that takes into account the management objectives for both Atlantic menhaden and their predators.

The proof-of-concept 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 fished at the status quo F rates (in this case, the F rate each species experienced in 2023). The ERP fecundity target and threshold were developed from the single-species menhaden model and defined as the long-term equilibrium fecundity that results when the population is fished at the ERP F_{TARGET} and $F_{\text{THRESHOLD}}$, respectively.

Striped bass was the focal species for this analysis because it was the most sensitive ERP fish species to Atlantic menhaden F, and focusing on one key predator provided a more tractable example for evaluating tradeoffs among management strategies. ERPs based on striped bass biomass are not expected to cause significant declines for other species that were less sensitive to levels of Atlantic menhaden removals.

The proof-of-concept ERP target and threshold from this assessment were similar to the ERP values currently used in management and reviewed previously (SEDAR, 2020b). The ERP target from this assessment was estimated at F=0.189, compared to the current ERP target of F=0.19, while the ERP threshold from this assessment was estimated at F=0.49, compared to the current ERP threshold of F=0.57. The associated fecundity (*FEC)* target and threshold for this assessment were also lower, with the FEC_{TARGET} equal to $1.67\text{x}10^{15}$ eggs, compared to the current FEC_{TARGET} of $2.0\text{x}10^{15}$ eggs. The $FEC_{\text{THRESHOLD}}$ was estimated at $1.14\text{x}10^{15}$ eggs, compared to the current $FEC_{\text{THRESHOLD}}$ of $1.49\text{x}10^{15}$ eggs.

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.

13.3 Stock Status and TAC Setting

Estimates of full F and fecundity from the base run of the BAM were used to evaluate stock status. Under these proof-of-concept ERPs, Atlantic menhaden were not overfished and not experiencing overfishing. The 2023 estimate of F was below the ERP $F_{\text{THRESHOLD}}$ but above the F_{TARGET}, and the 2023 estimate of fecundity was above the $FEC_{\text{THRESHOLD}}$ but below the FEC_{TARGET} (Table 27, Figure 79).

Short-term projections at the current TAC of 233,550 mt were conducted. Under a constant, status quo TAC of 233,550 mt, in 2028, F will be between the F_{TARGET} and the $F_{\text{THRESHOLD}}$, with a 0.5% probability that F will be above the ERP $F_{\text{THRESHOLD}}$ and a 99.5% probability that it will be above the F_{TARGET} (Figure 80, Table 28).

A suite of projections based on different TACs and different probabilities of achieving the F_{TARGET} or exceeding the $F_{\text{THRESHOLD}}$ will be completed after the peer review with the proof-of-concept ERPs, so that the Board can select a TAC that aligns with their risk tolerance for this species.

14 DISCUSSION

14.1 Synthesis of Findings

The ERP WG continued the development of the ecosystem models established in the 2020 benchmark assessment, updating and refining the preferred intermediate complexity EwE model, the NWACS-MICE model, as well as the NWACS-FULL and multispecies statistical catch-at-age model, VADER. The revision of M in the Atlantic menhaden single-species assessment model was a significant change that was propagated through the ecosystem models and resulted in an improved understanding of menhaden population and ecosystem dynamics.

The EwE models estimated a similar scale and trend in Atlantic menhaden biomass as the single-species assessment, although the single-species model showed more interannual variability than the EwE models, due to the differences in how recruitment was handled between the two models (Figure 70). A similar pattern was seen for striped bass, with the EwE models following the same trends as the single-species assessment but showing less interannual variability (Figure 71). The EwE models also fell short of the single-species peak for the age 2-5 stanza in biomass in the early 2000s, which was driven by several strong year-classes. The NWACS-FULL model tracked the scale of the single-species assessment more closely for the age 2-5 stanza than the NWACS-MICE model, but the reverse was true for the age-6+ stanza. The difference between age-6+ striped bass biomass between the MICE and FULL models is due to different Ecopath inputs for biomass accumulation, where a higher value was assumed in the FULL model during the balancing process.

The EwE models use the estimate of fishing mortality from the single-species assessment directly to drive the models, so exploitation rates of age-1+ menhaden calculated from the EwE outputs should be the same across all models (Figure 72). The full complexity EwE model estimated a higher predation mortality (M_2) on both age-0 and age-1+ menhaden than the intermediate complexity model, which is expected, as the intermediate complexity model had fewer predators (Figure 73). Both EwE models estimated an increasing trend in predation mortality for both age-0 and age-1+ menhaden over the time-series, with the age-0 stanza experiencing a stronger increase. The age-0 stanza experienced higher predation mortality overall than the age-1+ stanza for both models (Figure 73).

Although the trends in predation mortality were very similar across the EwE models, there were some differences in the proportion of Atlantic menhaden in the diets of ERP predators over time. Both models show generally similar variability among simulations/runs (with some exceptions). Overall, both models predicted a generally increasing trend in the proportion of Atlantic menhaden in most predator diets over the time-series, although there were some differences (Figure 74). The diet trend for medium striped bass (age 2-5) is concave for the NWACS-FULL base model (i.e., decreasing and then increasing) and convex for MICE (i.e., increasing and then decreasing over time), with different terminal values. The NWACS-FULL predicts a higher proportion of menhaden in the diet for large (age 6+) striped bass and spiny dogfish, and a stronger increasing trend in the proportion in large bluefish diets compared to MICE. These differences likely stemmed from differences in model structure, how the vulnerability parameters were estimated, initial balancing decisions, and assumptions about prey switching. The larger prey field in the NWACS-FULL model allowed for the ERP predators to respond more dynamically as the biomass of alternative prey varied over time; whereas the MICE model included large percentages of diet 'import', which holds a portion of their consumption constant over time. Both models also predicted a declining trend in the proportion of Atlantic herring predator diets over time, but the NWACS-MICE estimates showed more interannual variability in diet proportions than the NWACS-FULL model, due to the recruitment forcing deviations in the NWACS-MICE model that results in more variable estimates of juvenile Atlantic herring biomass (Figure 75). Although the scale of the estimated proportion of Atlantic herring in the diets was generally similar across models, the NWACS-MICE model predicted a much higher proportion of Atlantic herring in spiny dogfish diets than the NWACS-FULL model did, while the NWACS-FULL model predicted a much higher proportion of Atlantic herring in large striped bass diets than the NWACS-MICE model did (Figure 75). The larger proportion of Atlantic herring in the diets of striped bass for the NWACS-FULL model is explained by the absence of the seasonal vulnerability forcing function that was only included in the NWACS-MICE model. The seasonal forcing function was included exactly for the purpose of moderating consumption of herring by striped bass within a year after issues were raised in the previous assessment.

The single-species model estimates of total mortality for age-0 menhaden (Figure 76) and age 0-1 striped bass (Figure 77) were virtually constant over the time-series, as M is constant in those models and fishing pressure is negligible on those age-classes. Age-0 *Z* for the EwE models increased from the start of the time-series but stabilized around the early 2000s for both species (Figure 76 and Figure 77). For menhaden, the NWACS-MICE and NWACS-FULL estimates *of* Z were both higher than the age-0 *Z* from the single-species model, as was the NWACS-FULL estimate of *Z* for age 0-1 striped bass, but the NWACS-MICE estimate of *Z* was slightly less than the single-species model. However, direct comparisons between mortality rates from the BAM and EwE models are confounded by the different units used in those models (biomass-based F in EwE, numerical F in BAM) and how they were aggregated within age stanzas.

Total mortality on age-1+ menhaden decreased somewhat from the start of the series and then was generally stable from the early 2000s onwards, with a slight increase in the most recent years for all three models, although the *Z* estimates from the EwE models were higher than the Z estimates from the single-species model for that stanza (Figure 76). For striped bass, total mortality increased over the time-series for ages 2-5 and age-6+, although there was a downturn in the most recent few years, most likely driven by reductions in *F* (Figure 77). Total mortality from the EwE model was higher than the single-species model for age 2-5 but lower for age-6+, reflecting the increasing importance of fishing mortality as a component of total mortality for the older ages of striped bass.

The NWACS models estimated similar responses to menhaden fishing pressure for striped bass, with both showing striped bass as the ERP predator most negatively affected by increasing *F* on menhaden (Figure 78), and the MICE model being slightly more sensitive at higher menhaden F. Both models also predicted weakfish would increase under increasing menhaden *F*, although the shapes of those curves were different. However, the models differed in the responses they predicted for bluefish, spiny dogfish, and Atlantic herring. The NWACS-FULL predicted a strong positive response for bluefish to increasing menhaden *F* across the full range of *F* multipliers explored, while the NWACS-MICE response was less linear and generally more negative in response to increasing *F* (Figure 78). The NWACS-FULL model predicted that spiny dogfish would decrease and Atlantic herring would increase under increasing menhaden *F*, while the NWACS-MICE model predicted the opposite. These differences are influenced by sensitivities to the vulnerability parameters (e.g., bluefish sensitivity to a single vulnerability parameter in the NWACS-FULL) and differences in model design. Specifically, the NWACS-MICE model incorporated seasonal egg production for the ERP species, seasonal forcing functions to address differences in predator-prey overlap, and interannual recruitment variability of herring; these design enhancements improved the NWACS-MICE model behavior and could be added to the full model in future iterations.

Although the attempts to build bottom-up feedback into the VADER model were unsuccessful for this review, the importance of the external recruitment forcing function for Atlantic herring in the NWACS-MICE model underscores the utility of the multispecies statistical catch-at-age approach, which allows the fitting of recruitment deviations from observed data internally.

The consistency of results across ERP approaches presented here suggests 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 both current Atlantic menhaden management as well as aspects of Atlantic menhaden ecology and population dynamics.

The ERP models indicated that increasing fishing mortality on Atlantic menhaden to levels approaching the single-species $F_{\text{THRESHOLD}}$ as defined for management before 2020 (i.e., *F* multiplier of 4.6) would cause declines in biomass for more sensitive predator species, particularly striped bass and potentially for nearshore piscivorous birds, including osprey. However, management has consistently been more conservative than single-species reference points would have historically prescribed and has continued with a conservative approach even under the 2020 ERPs, choosing a TAC for 2022-2025 that had a very low probability of exceeding the ERP target. As a result, even with the lower biomass and higher *F* resulting from the change in *M* for menhaden, the probability of exceeding the ERP $F_{\text{THRESHOLD}}$ under the current TAC is low.

In addition to conservative management, the impact on 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, sand eels, 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 not an overly dominant component (i.e., >50%) 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., (Pikitch et al., 2012; Smith et al., 2011)); 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) and Buchheister et al. (2016). 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.

However, it should be noted that these reference points are developed on a coastwide scale. The incorporation of seasonal dynamics captures some spatial effects, as the overlap of predators and prey by season is often a proxy for spatial overlap. But these models do not yet capture the full spatial dynamics of the northwest Atlantic shelf region or the potential for more intense, small-scale interactions between species that could be occurring, particularly in important nursery grounds like the Chesapeake Bay. Fully incorporating spatial dynamics into the ERP and single-species BAM is a high priority for the assessment, managers, and stakeholders.

14.2 Synthesis of Management Advice

The ERP WG recommends continuing to use 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 it was deemed less ideal than the NWACS-MICE for management due to the greater data demands, increased uncertainty of data inputs for many groups, and higher complexity. The NWACS-FULL model indicated that striped bass was the most sensitive ERP 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 the three bird groups, medium pelagic fishes, and haddock were also sensitive to increases in Atlantic menhaden harvest. However, those results were less robust than for striped bass due to reduced data quality and availability, indirect ecological mechanisms that are harder to predict, and greater sensitivity to model parameterization. Harvest strategies developed with the NWACS-MICE model that maintain or rebuild striped bass biomass are expected to have positive effects on the three bird groups, though the MICE model would not capture the potential effects on nearshore piscivorous birds in the full ecosystem context. However, the data quality for the bird groups is less robust than for striped bass and other ERP species, which increases the uncertainty about the results for these species complexes.

There are downsides, however, to this approach. As outlined in ASMFC (2017b), 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 44), 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 as well as 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 biomasses for all modeled species, not just striped bass. Managers have already qualitatively performed this type of evaluation with the *ad hoc* buffering approach used in Atlantic menhaden management, prior to the adoption of ERPs in 2020. The use of the ERP tool allows for this evaluation to happen in a quantitative, transparent way, which is the overarching goal of the ecological reference point process. There is still room for a qualitative assessment of risk and uncertainty in the TAC setting process, as managers must decide on the level of the risk of overfishing that they are willing to tolerate.

15 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.

15.1 Progress on 2020 Benchmark Assessment Recommendations

15.1.1 Future Research and Data Collection

15.1.1.1 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.*

   The ERP WG incorporated additional diet data in the current assessment from Rhode Island and New Jersey state run programs; additional diet data for new predators were also included (e.g., literature on bluefin tuna diet). Additional data sources were discussed and considered at a May 2023 methods workshop (including striped bass condition data from MD-DNR). Additionally, the ERP WG spent considerable time streamlining the processing and analysis of diet data to facilitate additional explorations as part of the present and future assessments.

15.1.1.2 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.*

The ERP WG explored population trends and diet data for birds and marine mammals; this topic was explored in depth at a May 2023 methods workshop. Enough information was available for ospreys to allow for inclusion in the NWACS full model as its own group. Additional information sufficient to refine current input parameter estimates (primarily diet and biomass) did materialize for marine mammals (e.g., Kenney et al., 1997; see Section 3.1.3), but this group continues to be a source of uncertainty. An anchovy time series of relative abundance was developed (see Working Paper SEDAR 102-WP-05), as were timeseries for the various zooplankton groups (see Working Paper SEDAR 102-WP-06); a benthic invertebrate timeseries was developed from data available in the National Benthic Inventory (NBI; see Section 10.2.3), and a phytoplankton timeseries was developed from the GLORYS ocean model (Section 10.2.5). These all represent advances from the previous benchmark assessment.

15.1.2 Modeling Needs

15.1.2.1.1 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.*

This item has not been addressed.

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.*

NWACS-MICE (as well as NWACS-FULL) incorporates the GLORYS chlorophyll time series at annual and sub-annual timesteps to better capture intra- and interannual changes in productivity. Recruitment deviations were explored and incorporated to drive Atlantic herring egg production to better capture recent recruitment trends for that species. The ERP WG also explored primary producer forcing functions as a means to model variability in younger age stanzas.

3. *Continue development of the VADER model to include bottom-up effects of Atlantic menhaden abundance on key predator species.*

The ERP-WG made progress on this item (see Section 12). Briefly, the WG explored a wide range of data that were hypothesized to correlate changes in striped bass natural mortality with Atlantic menhaden and/or other prey. After an extensive search, the only striped bass length-at-age was found to have a statistically significant relationship with Atlantic menhaden biomass (see Working Paper SEDAR-102-WP-09. In the ERP-WG formulation, the link between striped bass and menhaden is a deterministic calculation based on empirical weight at age (WAA). The calculation accounts for the average length at age of striped bass, thus linking it back to the statistical relationship found between menhaden biomass and length-at-age of striped bass. In this way, there isn't a dynamic link between the two species, but rather a deterministic link based on input striped bass WAA each year. Implicit in this approach is that changes in WAA relative to striped bass size are driven by availability of prey (i.e., observed striped bass weights are influenced by prey).

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.*

The major updates and improvements to the previous version of the NWACS-FULL model includes the following: 1) all available biomass, catch, fishing mortality, and fishing effort time series were updated to 2023 using recent stock assessment and fishery dependent and independent survey data; 2) the diet composition matrix was updated using a more synthetic analytical approach with updated data; 3) multistanza groups for key ERP trophic groups were updated to match the NWACS-MICE model; 4) osprey were added as a functional group and bluefin tuna were used as the representative species for the large pelagic highly migratory species (HMS) group; 5) additional time series or key parameters were developed for several trophic groups (anchovies, five zooplankton groups, osprey, baleen whales, odontocetes, and HMS); and 6) a primary production forcing function was developed and evaluated.

15.1.3 Management Process Needs

15.1.3.1 Short Term

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

Some progress has been made on this item. ASMFC-managed species (menhaden, striped bass, weakfish) are generally coordinated and aligned with the ERP schedule; NEFSC (/jointly) managed species (spiny dogfish, Atlantic herring, bluefish) are nearly aligned. Any misalignments notwithstanding, the ERP WG has engaged with managers and partner agencies about the need for synchrony and data sharing. Cooperation among partners has been great, greatly facilitating updating time series for fully assessed species (e.g., spiny dogfish), as well as for non-assessed species input as indices of abundance (e.g., demersal benthivores).

15.1.3.2 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.*

A plan was proposed to managers shortly after the 2020 benchmark. There has been little progress since then.

15.2 2025 Research Recommendations

Although progress has been made on a number of research recommendations from the 2020 benchmark assessment, more work is still needed on all recommendations, and the ERP WG continues to recommend those items as crucial for continued improvements in the ERP assessment. In addition, the ERP WG identified new or more detailed recommendations for this assessment.

15.2.1 Future Research and Data Collection

15.2.1.1 Short-Term

1. Data on prey size (or age) are needed to more accurately apportion diet across age stanzas of prey. Prey size data exist for the NEFSC but were not requested with the data package. Other diet programs should augment their procedures to measure or approximate prey size, especially for the ERP species when they are identified in a stomach.
2. Genetic techniques have shown promise in expanding our understanding of diet composition; for example, preliminary genetic analyses on squid stomach contents suggest more extensive consumption of fishes (including menhaden) than previously described based on visual identification (Brian Smith, NEFSC, personal communication), and this could have substantial impacts on our understanding of ecosystem structure and dynamics. This type of work could be deployed on existing survey platforms potentially more easily (although not more cheaply) than traditional stomach content analysis. However, more work is needed to understand how to appropriately integrate the results of genetic diet composition studies with traditional studies to make the best use of both types of datasets.
3. In the NWACS-MICE, we evaluated a suite of monthly and seasonal forcing functions. To develop a fully monthly model, monthly catch and fishing mortality time series should be developed, and monthly or seasonal indices should be developed.
4. We recommend revisiting the assumption that all of the spiny dogfish biomass estimated from the stock assessment resided within the model domain (Carlson et al., 2014).

15.2.1.2 Long-Term

1. Improved data inputs for the three bird groups (nearshore piscivorous birds, osprey, seabirds) would be beneficial, including better estimates of biomass, diets, P/B, relative biomass time series, and productivity drivers at the coastwide scale that account for residency and degree of interaction with coastal waters. Collaboration with avian researchers would aid such efforts.
2. More comprehensive seasonal and spatial sampling of spiny dogfish diets are needed to confirm model predictions about the predation impacts on Atlantic herring, Atlantic menhaden, and especially striped bass.
3. Despite all the diet data used in the NWACS-MICE update, very little diet data exist for Atlantic menhaden. This is likely because filter feeders are generally overlooked in diet studies. However, as we attempt to incorporate environmental drivers into models, how energy enters the food web through menhaden becomes more important. We

recommend augmented sampling and/or a literature review of diet data for menhaden to determine contributions by phytoplankton groups, zooplankton, and detritus.

4. Ongoing research on ecological mechanisms and additional species-specific expertise would be beneficial to incorporate more quantitative or explicit drivers for recruitment, productivity, mortality, or trophic dynamics in the EwE models in the future.

15.2.2 Modeling Needs

15.2.2.1 Short Term

1. An ongoing need for ecosystem models is to quantify and depict uncertainties associated with the various data inputs, estimated parameters (e.g., vulnerabilities and switching power parameters), and model predictions. Options for this include conducting Monte Carlo sensitivity analyses based on input parameter variability by using the Ecosampler module (e.g., Steenbeck et al., 2018), the Multisim tool, or evaluating various sensitivity runs with alternative parameterizations.
2. The ability to estimate additional parameters beyond the vulnerabilities in EwE is critically needed. As we have shown in the current ERP assessment, Ecopath inputs (e.g. spiny dogfish) and prey switching have implications for the resulting management advice. Recently, a new approach to EwE modeling, Ecostate (Thorson et al., 2025), addresses many of these issues by allowing the model to estimate both initial mass-balance parameters (Ecopath) and time-dynamic parameters (Ecosim). However, Ecostate does not yet include the full capabilities of EwE such as multi-stanza age structure and seasonal forcing functions, that are needed for this particular application. Further development of Ecostate and/or the EwE software would help us move beyond the approaches used here to a more robust estimation of model parameters. Uncertainty analyses could then be developed based on uncertainty in parameter estimates as is typically done with age-structured stock assessments.
3. It is possible to add forcing functions to EwE models that represent changes to productivity, recruitment, mortality, or trophic dynamics, as was done with Atlantic herring in the NWACS-MICE model. Including recruitment forcing functions for other ERP species in the NWACS-MICE model should be explored in the near future. Other forcing functions should be explored as data become available. The ERP WG recommends exploring these changes in the NWCAS-MICE model to inform/prioritize what forcing functions to use in NWACS-FULL model.

15.2.2.2 Long Term

1. Developing more spatially explicit ERPs has been a priority for managers and stakeholders. This could be accomplished with the NWACS models via Ecospace. We recommend pursuing a spatial version of the NWACS-MICE model first, before attempting this approach with the Full model.
2. Depending on the management goals of spatial ERPs, the development of spatially explicit single-species models for Atlantic menhaden and other ERP species might be required.

15.2.3 Management Process Needs

15.2.3.1 Short Term

1. The ERP WG recommends convening a workshop with Board members and stakeholders similar to the 2015 Ecosystems Management Objectives workshop to identify the goals and objectives that spatial ERPs should address. This information is needed to develop a timeline for data collection and model development for spatial ERPs.

15.3 Timing of Future Assessments

The ERP WG noted that there are several factors that will influence the timing of the next update and benchmark assessments for ERPs, making it difficult to recommend a specific timeline. MRIP estimates of recreational effort and catch are in the process of being recalibrated, with a new time-series of data for all species scheduled to be released in mid-2026. The scale of the changes is not expected to be as significant as the changes in 2017 but will likely result in some changes to the assessments of bluefish, weakfish, and striped bass. In addition, striped bass will be undergoing a benchmark assessment in 2027. The ERP WG recommends updating the NWACS-MICE model in conjunction with the next menhaden single-species update after that, if the 2027 striped bass benchmark results in a change to striped bass reference points or management objectives that could be accommodated within the current NWACS-MICE trade-off analysis framework. Otherwise, the ERP WG recommends only updating the single-species menhaden assessment until the next benchmark.

Additionally, managers and stakeholders have expressed strong interest in spatial ERPs for menhaden in the past. To develop a timeline for the next benchmark, the management goals and objectives for spatial ERPs need to be defined, so that the data and model needs to address those goals and objectives can be identified. To do so the ERP WG recommends a workshop to address this need. As such, the timing of the next benchmark will depend on the timing of the proposed workshop.

That said, the short-term capacity of the SEFSC to dedicate time and resources to developing a spatially explicit version of BAM is unclear. Depending on the spatial ERP objectives, ASMFC and the Atlantic Menhaden SAS may have to take the lead on developing a spatially explicit single-species assessment framework for menhaden, which would affect the timing of the next benchmark.

The ERP WG also recommends that work progress on the ERP models outside the benchmark timeline to allow for full development and exploration of these models; work on an Ecospace version of the NWACS-MICE model is already planned to continue after the 2025 benchmark. Depending on the spatial ERP objectives and whether they would require a spatially explicit single-species assessment (likely if the Board and stakeholders are interested in setting the TAC at the regional level), the next ERP benchmark assessment should occur after the spatially explicit single-species assessment is successfully peer reviewed.

Lastly, a significant amount of funding external to the ASMC was critical to the completion and development of the EwE models, both in the past and for the current assessment, including competitive grants from NOAA and the Lenfest Ocean Program. Sustained financial investment is needed to develop and implement the modeling tools, specifically as we move towards using spatial models to define the ERPs and improving parameter estimation procedures. The amount of investment is expected to decrease with fully developed tools. Training of ASMFC, federal, and state agency staff on model use is one avenue to make things more cost-effective. In the current assessment, we developed code repositories on GitHub to facilitate future updates and reproducibility by agency scientists. This co-development should be fully codified in future updates.

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17 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	Evaluate trade-offs between menhaden harvest and predator abundance	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	•	•	•	•					•			ǂ	
Multi-Species Statistical Catch-at-Age (MSSCAA)													
VADER	•	•	•	•	○	○	○ (proxy)	○	•	○		ǂ	ǂ
Ecopath with Ecosim (EwE)													
NWACS-MICE	•	•	•	•	○	○	○ (proxy)	○	•	○	○	ǂ	ǂ
NWACS-FULL	•	•	•	•	•	•	• (proxy)	•	•	•	•	ǂ	ǂ

○ For menhaden or key finfish predators

• For wide range of predator and prey species, including birds and mammals

ǂ: Model could be developed further 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 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 $F_{40\%SPR}$	186,367 mt	464,425 mt
Bluefish	Projected SSB when fishing at $F_{35\%SPR}$	88,131 mt	101,106 mt
Spiny dogfish	Spawning output at $F_{60\%SPR}$	188 million pups	505,126 mt
Striped bass	125% of female SSB in 1995	111,892 mt	311,691 mt
Weakfish	Not defined	N/A	N/A

Species	$SSB_{THRESHOLD}$ Definition	$SSB_{THRESHOLD}$	$B_{THRESHOLD}$ Proxy
Atlantic herring	½ SSB_{TARGET}	93,684 mt	232,213 mt
Bluefish	½ SSB_{TARGET}	44,066 mt	50,553 mt
Spiny dogfish	½ SSB_{TARGET}	94 million pups	252,563 mt
Striped bass	Female SSB in 1995	89,513 mt	249,353 mt
Weakfish	30% of unexploited SSB	14,570 mt*	17,183 mt*

Species	F_{TARGET} Definition	F_{TARGET}
Atlantic herring	Not defined	
Bluefish	Not defined	
Spiny dogfish	Not defined	
Striped bass	*F* rate projected to achieve SSB target	0.17
Weakfish	$Z_{SPR30\%}$ =0.91; based on M=0.43	F=0.48*

Species	$F_{THRESHOLD}$ Definition	$F_{THRESHOLD}$
Atlantic herring	$F_{40\%SPR}$	0.45
Bluefish	$F_{35\%SPR}$	0.239
Spiny dogfish	$F_{60\%SPR}$	0.025
Striped bass	*F* rate projected to achieve $SSB_{THRESHOLD}$	0.20
Weakfish	$Z_{SPR20\%}$=1.19; based on M=0.43	F=0.76*

Table 4. Single-species estimates of total biomass and *F* in 2023 and percent change needed to achieve target and threshold values.

Species	*F* to achieve SSB_{TARGET}	2023 *F* (status quo *F*)	Multiplier to reach target *F*
Atlantic herring	0.45	0.263*	1.714*
Bluefish	0.239	0.127**	1.878**
Spiny dogfish	0.025	0.025**	1.000**
Striped bass	0.17	0.18	0.934
Weakfish	0.76‡	0.25	3.04

Species	Status Quo Biomass	Multiplier to Reach	
		$B_{THRESHOLD}$	B_{TARGET}
Atlantic herring	90,839 mt*	2.556*	5.113*
Bluefish (2023) (ASMFC)	75,477 mt**	0.670**	1.339**
Spiny dogfish	728,168 mt**	0.347**	0.694**
Striped bass	186,509 mt	1.343	1.679
Weakfish	8,766 mt‡	1.960	N/A

*: A retrospective adjustment has been applied to the 2023 *F* and biomass estimates and the multiplier is based on the adjusted 2023 values.
**: 2023 values for bluefish and spiny dogfish are based on projected values, since the terminal year of their assessments is 2022.
‡: Estimates for weakfish are based on preliminary assessment updates and may not match final values used in management.

Table 5. ERP species stock assessments

Species	Assess. Year	Term. Year
Atlantic herring - ASAP	2024	2023
Atlantic menhaden – BAM (*M*=0.925 base run)	2024	2023
Bluefish - WHAM	2023	2022
Striped bass - SCA	2024	2023
Spiny dogfish – SS3	2023	2022
Weakfish - ASAP	2024	2023

Table 6. Average numerical and biomass density by family from the NBI datasets.

Phylum	Class	Density (n/m^2)	Density (g/m^2)
Mollusca	Bivalvia	463.326	25.483
Annelida	Polychaeta	1759.005	9.205
Arthropoda	Malacostraca	927.081	4.037
Annelida	Clitellata	377.941	1.935
Mollusca	Gastropoda	147.081	1.741
Echinodermata	Ophiuroidea	17.025	0.936
Chordata	Ascidiacea	15.294	0.841
Cnidaria	Anthozoa	11.867	0.653
Nemertea	Palaeonemertea	8.529	0.469
Nemertea	Neonemertea	5.498	0.302
Sipuncula	Sipunculidea	5.204	0.286
Echinodermata	Echinoidea	4.559	0.251
Arthropoda	Thecostraca	3.982	0.219
Echinodermata	Holothuroidea	3.529	0.194
Arthropoda	Insecta	78.043	0.187
Sipuncula	Phascolosomatidea	2.500	0.138
Mollusca	Scaphopoda	2.059	0.113
Bryozoa	Gymnolaemata	1.719	0.095
Mollusca	Polyplacophora	2.048	0.093
Hemichordata	Enteropneusta	4.446	0.067
Cnidaria	Hydrozoa	0.498	0.027
Arthropoda	Pycnogonida	0.385	0.021
Echinodermata	Asteroidea	0.317	0.017
Arthropoda	Arachnida	0.204	0.011
Arthropoda	Branchiopoda	0.204	0.011
Arthropoda	Ostracoda	17.206	0.009
Arthropoda	Copepoda	0.305	0.002
Arthropoda	Branchiura	0.023	0.001
Brachiopoda	Lingulata	0.023	0.001
Chordata	Teleostei	0.011	0.001
Arthropoda	Merostomata	0.011	0.001

Table 7. Number of individual stomachs contained in each of the diet data sources.

Predator	ChesMMAP	MSVPA	NEAMAP	NEFSC	NJOT	RIDEM	Total
anchovy				15			15
Atlantic herring 0-1	1		147				148
Atlantic herring 2	15		333	23491			23839
bluefish adult	29	1283	378	5492	57	210	7449
bluefish juv	553	5661	3434		111	22	9781
menhaden adult	2			205			207
spiny dogfish	145		2710	71431	304	15	74605
striped bass 0-1	2747	2460	57		4	12	5280
striped bass 2-5	1436	7166	169	984	44	235	10034
striped bass 6	91	2578	161	614	12	53	3509
weakfish adult	1164	1459	1267	5477	80	161	9608
weakfish juv	5874	535	6925		266	35	13635

Table 8. Final diet matrix for the NWACS-MICE model.

Prey	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
1. striped bass 0-1	0.000	0.000	0.000				0.003	0.000	0.000						
2. striped bass 2-5						0.003									
3. striped bass 6+						0.002									
4. menhaden juv	0.046	0.210	0.050			0.013	0.017	0.013	0.002	0.006					
5. menhaden adult		0.008	0.407			0.114		0.104		0.049					
6. spiny dogfish						0.000		0.000							
7. bluefish juv	0.000	0.001	0.000				0.001	0.002		0.000					
8. bluefish adult						0.004		0.022							
9. weakfish juv	0.002	0.000					0.001	0.000	0.002	0.002					
10. weakfish adult	0.002	0.007	0.000			0.004		0.008		0.048					
11. Atlantic herring 0-1		0.010	0.003			0.001	0.001	0.006							
12. Atlantic herring 2+		0.001	0.035			0.023		0.011		0.000					
13. anchovies	0.320	0.143	0.077			0.018	0.522	0.165	0.298	0.274	0.000	0.020			
14. benthic inverts	0.452	0.349	0.106			0.187	0.033	0.019	0.311	0.180	0.228	0.256	0.101	0.090	0.001
15. zooplankton	0.041	0.005	0.000	0.540	0.660	0.101	0.005	0.003	0.197	0.043	0.772	0.713	0.684	0.021	0.261
16. phytoplankton				0.240	0.175								0.155	0.229	0.490
17. Detritus	0.000	0.001		0.220	0.165	0.000	0.000		0.002	0.000			0.060	0.413	0.199
Import	0.137	0.264	0.321			0.530	0.418	0.648	0.187	0.397	0.000	0.011		0.246	0.048

Table 9. Unbalanced (*U*) and balanced (*B*) values input or calculated by the NWACS-MICE model. Biomass, biomass accumulation rate (BA), and *Z* (or P/B) were input to the model while fishing mortality (*F*) was calculated based on landings input, predation mortality (M_2) was calculated from predator consumption, total natural mortality (*M*) was taken as *Z-F*, and *EE* is calculated by Ecopath (Christensen & Walters, 2004).

Group name	Biomass		*Z* or P/B		*F*		M_2		*M*		BA		*EE*	
	U	*B*	*U*	*B*	*U*	*B*	*U*	*B*	*U*	*B*	*U*	*B*	*U*	*B*
stiped bass 0-1	0.008	0.008	1.040	1.040	0.035	0.035	2.398	0.048	1.005	1.005	0.168	0.168	2.339	0.080
striped bass 2-5	0.041	0.041	0.420	0.420	0.046	0.046		0.221	0.374	0.374	0.168	0.168	0.109	0.636
striped bass 6+	0.040	0.040	0.153	0.153	0.003	0.003		0.112	0.150	0.150	0.168	0.168	0.018	0.750
menhaden juv	0.364	0.313	1.398	1.398	0.031	0.036	1.291	0.296	1.366	1.361	0.223	0.129	0.946	0.238
menhaden adult	2.525	2.525	1.295	1.295	0.312	0.312	0.064	0.212	0.983	0.983	0.223	0.129	0.290	0.405
spiny dogfish	1.668	1.668	0.201	0.201	0.032	0.032	0.000	0.000	0.168	0.168	0.000	0.000	0.163	0.163
bluefish juv	0.007	0.007	0.934	0.934	0.247	0.247	6.185	0.434	0.686	0.686	-0.176	-0.176	6.890	0.730
bluefish adult	0.465	0.465	0.749	0.749	0.250	0.250	0.003	0.094	0.499	0.499	-0.176	-0.176	0.338	0.459
weakfish juv	0.005	0.002	0.726	0.726	0.244	0.677	8.794	0.576	0.482	0.049	0.563	-0.535	12.448	0.990
weakfish adult	0.039	0.039	1.146	1.146	0.613	0.613	0.093	0.842	0.533	0.533	0.563	-0.321	0.616	0.990
Atlantic herring 0-1	0.067	0.023	1.021	1.021	0.131	0.386	1.734	0.559	0.890	0.635	0.983	0.331	1.827	0.925
Atlantic herring 2+	0.318	0.318	0.759	0.759	0.409	0.409	0.114	0.279	0.350	0.350	0.983	0.331	0.689	0.906
anchovies	0.635	0.635	2.200	2.200	0.000	0.000	0.717	0.704	2.200	2.200	0.000	0.000	0.326	0.320
benthic inverts	51.910	51.910	2.432	2.432	0.000	0.000	1.210	1.208	2.432	2.432	0.000	0.000	0.498	0.497
zooplankton	24.470	24.470	45.850	45.850	0.000	0.000	41.513	41.492	45.850	45.850	0.000	0.000	0.905	0.905
phytoplankton	25.450	25.450	186.436	186.436	0.000	0.000	78.797	78.793	186.436	186.436	0.000	0.000	0.423	0.423
Detritus	29.420	29.420	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.234	0.234

Table 10. List of annual observed relative (rel) and absolute (abs) timeseries used to calibrate the NWACS-MICE Ecosim model.

NWACS species	Label	Type	weight	Years
striped bass 0-1	SB_compos	rel B	3.21	1985-2023
striped bass 0-1	SB_stanza1_biomass_mtkm2	rel B	0.00	1985-2023
striped bass 0-1	SB_stanza1_landings_mtkm2	rel C	10.00	1985-2023
striped bass 2-5	SB_MRIP	rel B	5.45	1985-2023
striped bass 2-5	SB_CTLIST	rel B	5.38	1987-2023
striped bass 2-5	SB_stanza2_biomass_mtkm2	abs B	0.00	1985-2023
striped bass 2-5	SB_stanza2_landings_mtkm2	abs C	10.00	1985-2023
striped bass 6+	SB_MDSSN	rel B	4.00	1985-2023
striped bass 6+	SB_stanza3_biomass_mtkm2	rel B	0.00	1985-2023
striped bass 6+	SB_stanza3_landings_mtkm2	rel C	10.00	1985-2023
menhaden juv	AM_lowM_U.jai.ob	rel B	1.99	1985-2000
menhaden juv	AM_lowM_stanza1_biomass_mtkm2	rel B	0.00	1985-2023
menhaden juv	AM_stanza1_landings_mtkm2	rel C	10.00	1985-2023
menhaden adult	AM_lowM_U.nad.ob	rel B	1.65	1990-1992
menhaden adult	AM_lowM_U.mad.ob	rel B	1.58	1985-2023
menhaden adult	AM_lowM_U.sad.ob	rel B	1.76	1990-2023
menhaden adult	AM_lowM_stanza2_biomass_mtkm2	abs B	0.00	1985-2023
menhaden adult	AM_stanza2_landings_mtkm2	abs C	10.00	1985-2023
spiny dogfish	SDF_NEFSC_Spring_BTS	rel B	3.46	1985-2022
spiny dogfish	SDF_stanza1_biomass_mtkm2	abs B	0.00	1985-2023
spiny dogfish	SDF_stanza1_landings_mtkm2	abs C	10.00	1985-2023
bluefish juv	BF_CompYOY	rel B	3.46	1985-2023
bluefish juv	BF_stanza1_biomass_mtkm2	rel B	0.00	1985-2023
bluefish juv	BF_stanza1_landings_mtkm2	rel C	10.00	1985-2023
bluefish adult	BF_MRIP_CPUE	rel B	3.81	1985-2020
bluefish adult	BF_NC_PSIGNS	rel B	7.62	2001-2023
bluefish adult	BF_NEFSC_A	rel B	1.45	1985-2008
bluefish adult	BF_stanza2_biomass_mtkm2	abs B	0.00	1985-2023
bluefish adult	BF_stanza2_landings_mtkm2	abs C	10.00	1985-2023
weakfish juv	WF_Composite_YOY	rel B	2.56	1988-2022
weakfish juv	WF_stanza1_biomass_mtkm2	rel B	0.00	1985-2023
weakfish juv	WF_stanza1_landings_mtkm2	rel C	10.00	1985-2023
weakfish adult	WF_MRIP_CPUE	rel B	3.57	1985-2023
weakfish adult	WF_DE_30ft_Trawl	rel B	4.30	1991-2023
weakfish adult	WF_NJ_Ocean_Trawl	rel B	3.43	1985-1986
weakfish adult	WF_NC_PSIGNS	rel B	3.09	1990-2023
weakfish adult	WF_stanza2_biomass_mtkm2	abs B	0.00	1985-2023
weakfish adult	WF_stanza2_landings_mtkm2	abs C	10.00	1985-2023
Atlantic herring 0-1	AH_stanza1_biomass_mtkm2	rel B	0.00	1985-2023
Atlantic herring 0-1	AH_stanza1_landings_mtkm2	rel C	10.00	1985-2019
Atlantic herring 2+	AH_Shrimp	rel B	1.20	1985-2023
Atlantic herring 2+	AH_FallAlb85	rel B	1.41	1985-2008
Atlantic herring 2+	AH_FallBig	rel B	1.84	2009-2023
Atlantic herring 2+	AH_stanza2_biomass_mtkm2	abs B	0.00	1985-2023
Atlantic herring 2+	AH_stanza2_landings_mtkm2	abs C	10.00	1985-2023
anchovies	Anchovy _SDM	rel B	1.00	1985-2023
zooplankton	ZP NEFSC biovolume	rel B	1.81	1985-2023

Table 11. Monthly egg production forcing functions for the ERP multistanza species.

	striped bass	menhaden	bluefish	weakfish	Atlantic herring
Jan	0.00	0.41	0.19	0.00	0.00
Feb	0.00	0.36	0.18	0.00	0.00
Mar	0.05	0.66	0.25	0.00	0.00
Apr	1.00	0.58	0.50	0.10	0.00
May	0.63	0.39	0.56	1.00	0.00
Jun	0.05	0.29	1.00	0.71	0.00
Jul	0.00	0.27	0.75	0.41	0.00
Aug	0.00	0.39	0.25	0.34	0.50
Sep	0.00	0.38	0.13	0.00	1.00
Oct	0.00	1.00	0.08	0.00	1.00
Nov	0.00	0.70	0.13	0.00	0.50
Dec	0.00	0.67	0.15	0.00	0.00
Source	Brown et al. 2024	Latour et al. 2023	Robillard et al. 2008	Lowerre-Barbieri et al. 1996	Fishbase

Table 12. Summary of NWACS-MICE Ecosim models evaluated.

Model num	Annual F	Monthly F	Annual PP	Monthly PP	Monthly vul ff	Monthly egg ff	AH Rdev ff	AH Ndev ff	Number of runs
1	X								30
2	X		X						39
3	X			X					32
4	X					X			7
5	X				X				6
6		X							6
7		X		X	X	X			6
8	X			X	X	X			4
9	X		X		X				2
10	X			X	X				3
11	X				X	X			4
12	X				X	X	X		6
13	X				X	X		X	5

Table 13. NWACS-MICE model runs with SS falling in the lowest 10th percentile (1,895). Runs 86 and 139 (*) were not in the top 10% but are included here because they were considered when determining the base run.

Run num	Model num	Model label	Initi k_{ij}	Max Rel PB	P_j	Final SS	Final AIC	N est k_{ij}	N low	N high
145	13	seasonal eggs, vuls, AH NAAdev	B_{unf}/B_o	2	1	1791	151	82	24	6
144	12	seasonal eggs, vuls, AH Rdev	B_{unf}/B_o	2	mix	1800	160	93	13	2
6	1	continuity	B_{unf}/B_o	2	1	1824	183	78	20	2
137	11	seasonal eggs and vuls	V_{max}	2	1	1837	NA	78	19	9
136	11	seasonal eggs and vuls	B_{unf}/B_o	2	1	1860	217	82	19	7
142	12	seasonal eggs, vuls, AH Rdev	B_{unf}/B_o	2	1	1864	220	67	12	1
95	1	continuity	V_{max}	2	1	1865	222	63	15	3
147	13	seasonal eggs, vuls, AH NAAdev	B_{unf}/B_o	2	1	1872	276	84	21	11
150	12	seasonal eggs, vuls, AH Rdev	B_{unf}/B_o	2	mix	1874	167	93	12	2
138	11	seasonal eggs and vuls	B_{unf}/B_o	2	mix	1884	NA	88	17	3
92	5	seasonal vuls	B_{unf}/B_o	2	1	1888	242	88	17	3
105	2	PP annual	B_{unf}/B_o	3	0	1890	244	79	21	22
149	13	seasonal eggs, vuls, AH NAAdev	B_{unf}/B_o	2	1	1892	246	92	12	2
85	4	seasonal eggs	B_{unf}/B_o	2	0	1892	246	76	17	7
77	2	PP annual	B_{unf}/B_o	2	0	1895	249	77	19	9
86*	4	seasonal eggs	B_{unf}/B_o	2	0.5	1934	284	85	11	3
139*	11	seasonal eggs and vuls	B_{unf}/B_o	2	NA	1971	NA	90	9	2

Table 14. Estimated k_{ij} for ERP species from select model runs. Green colors indicate low estimated k_{ij} (<0), yellow values are intermediate (10-1e5), and red values are high (>1e6). Numbers in italic are considered to be on bounds.

pred	prey	run 6	run 77	run 85	run 92	run 95	run 105	run 136	run 137	run 139	run 142	run 144	run 150
AH 1	ANC	1.1E+01	1.1E+01	1.1E+01	1.1E+01	6.1E+04	1.1E+01	1.1E+01	6.1E+04	1.1E+01	1.1E+01	1.1E+01	1.1E+01
AH 1	BI	4.3E+02	8.8E+01	1.1E+01	6.7E+07	2.4E+03	1.4E+03	4.7E+05	2.5E+03	2.3E+02	2.1E+00	1.2 E+02	1.2E+02
AH 1	ZOO	2.8E+05	7.3E+03	1.7E+04	8.8E+06	6.3E+03	2.6E+06	7.1E+04	7.0E+03	9.9E+09	1.9E+00	9.9E+09	9.9E+09
AH 2	ANC	2.5E+08	9.7E+09	*1.0E+00*	9.8E+09	3.2E+03	*1.0E+10*	3.3E+04	2.5E+04	*1.0E+10*	2.3E+07	*1.0E+10*	*1.0E+10*
AH 2	BI	4.6 E+04	8.7E+09	8.3E+09	4.2E+03	1.9E+05	*1.0E+10*	4.9E+03	1.9E+04	8.7E+09	1.6 E+01	8.7E+09	8.7E+09
AH 2	ZOO	3.1E+04	3.7E+00	2.5E+00	4.3E+04	9.3E+02	3.7E+00	1.0E+01	1.7E+01	4.7 E+00	2.3E+02	5.0E+00	5.0E+00
AM 1	DET	2.5E+01	2.0E+00	*1.0E+00*	1.4E+00	*1.0E+00*	2.0E+00	2.5E+01	*1.0E+00*	9.0E+00	2.5E+01	1.0E+00	1.0E+00
AM 1	PHY	2.5E+01	1.7 E+00	*1.0E+00*	1.1E+03	6.6E+03	2.5E+00	2.5 E+01	6.7E+03	1.6E+02	2.5E+01	1.7 E+02	1.7E+02
AM 1	ZOO	*1.0E+00*	3.3 E+00	4.3E+00	*1.0E+00*	5.5E+02	3.0E+00	*1.0E+00*	1.9E+03	2.6E+00	*1.0E+00*	2.0E+00	2.0E+00
AM 2	DET	6.5E+00	1.4E+08	*1.0E+10*	1.1E+00	*1.0E+00*	*1.0E+10*	*1.0E+00*	*1.0E+00*	3.6E+09	9.0E+00	3.6E+09	3.6E+09
AM 2	PHY	1.1E+03	7.8E+01	5.8E+00	3.1E+03	2.8E+03	1.9E+04	*1.0E+00*	2.8E+03	1.0E+10	2.3E+00	1.0E+10	1.0E+10
AM 2	ZOO	1.0E+00	1.3 E+08	2.3E+00	*1.0E+00*	1.2E+00	*1.0E+10*	2.0E+00	1.4E+00	1.7E+00	1.0E+00	1.7E+00	1.7E+00
BF 1	AH 1	6.4 E+03	5.3E+00	5.3E+00	9.4E+01	1.6E+02	5.3E+00	5.3E+00	1.4E+02	5.3E+00	5.3E+00	5.8E+00	5.8E+00
BF 1	BF 1	5.3E+00	5.3E+00	5.3E+00	5.3E+00	5.8E+01	5.3E+00	5.3E+00	5.8E+01	3.9E+00	5.3E+00	3.9E+00	3.9E+00
BF 1	AM 1	6.5E+01	5.3E+00	5.3E+00	*1.0E+10*	3.2E+02	3.4 E+01	5.3E+00	3.2E+02	1.1E+02	9.8E+04	1.0E+02	1.0E+02
BF 1	SB 1	4.2E+02	1.5E+03	5.7E+00	1.8 E+00	2.6E+01	3.5 E+01	8.0E+03	1.7E+06	8.7E+09	2.2E+00	8.7E+09	8.7E+09
BF 1	WF 1	5.3E+00	5.3E+00	5.3E+00	4.1E+01	2.0E+00	5.3E+00	5.3E+00	4.8 E+08	5.3E+00	5.3E+00	4.0E+01	4.0E+01
BF 2	AH 1	*1.0E+00*	*1.0E+00*	*1.0E+00*	1.2E+02	*1.0E+00*	*1.0E+00*	5.5E+05	*1.0E+00*	2.0E+04	1.9 E+00	2.0E+04	2.0E+04
BF 2	AH 2	*1.0E+00*	*1.0E+00*	1.4 E+00	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	9.6E+09	*1.0E+00*	9.6E+09	9.6E+09
BF 2	BF 2	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	1.0E+00	*1.0E+00*	*1.0E+00*	*1.0E+00*
BF 2	BF 1	6.7E+00	*1.0E+00*	6.7E+08	1.5E+02	1.2E+00	*1.0E+00*	1.1E+03	2.2E+08	2.2E+00	*1.0E+00*	6.5 E+01	6.5E+01
BF 2	AM 2	9.6E+03	7.0E+09	*1.0E+10*	1.1E+07	1.5E+05	*1.0E+10*	1.2E+00	1.6E+04	9.2E+09	9.2E+02	9.2E+09	9.2E+09
BF 2	AM 1	4.5E+02	2.8E+03	1.7E+04	3.1E+04	2.3E+01	1.4E+03	5.7E+02	2.9E+03	9.8E+09	1.4 E+02	9.8E+09	9.8E+09
BF 2	SDF	2.3E+00	2.3E+00	2.3E+00	2.3E+00	1.9E+03	2.3E+00	2.3E+00	1.9E+03	1.1E+00	2.3E+00	1.1E+00	1.1E+00
BF 2	SB 1	*1.0E+00*	2.0E+02	*1.0E+00*	2.6E+03	*1.0E+00*	2.0E+00	2.4E+01	*1.0E+00*	2.3E+02	3.0E+01	2.3E+02	2.3E+02
BF 2	WF 2	*1.0E+00*	1.0E+10	7.0E+09	*1.0E+00*	*1.0E+00*	*1.0E+10*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*
BF 2	WF 1	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*
SB 1	BF 1	3.6E+01	4.8E+01	3.6E+01	1.7E+00	8.7E+02	1.2E+01	3.6E+01	7.7E+02	2.6 E+00	3.6E+01	8.4E+00	8.4E+00
SB 1	AM 1	4.7E+00	1.8E+01	*1.0E+00*	*1.0E+00*	1.8E+03	2.1E+01	8.1E+01	2.2E+06	2.1E+00	5.2E+00	4.7E+00	4.7E+00
SB 1	SB 1	2.0E+00	2.1E+00	*1.0E+10*	1.4E+00	1.5E+03	*1.0E+00*	*1.0E+00*	5.8E+00	1.5 E+00	3.5E+00	2.1E+00	2.1E+00
SB 1	WF 2	4.6E+06	8.2E+09	*1.0E+10*	5.1E+02	1.8 E+02	*1.0E+10*	*1.0E+10*	6.5E+05	1.8E+02	3.3E+02	4.6E+04	4.6E+04
SB 1	WF 1	3.7E+04	9.6E+09	7.3E+09	2.1 E+06	2.7 E+07	*1.0E+10*	2.0E+06	*1.0E+10*	1.0E+00	9.7E+04	1.8E+01	1.8E+01
SB 2	AH 1	1.1E+00	*1.0E+10*	8.5E+05	2.5E+00	1.1E+00	*1.0E+10*	*1.0E+00*	2.0E+00	1.7E+02	3.0E+00	9.4E+01	9.4E+01
SB 2	AH 2	1.4E+01	5.4E+02	1.7E+02	7.4E+01	1.2E+03	2.4E+05	5.3E+01	2.6E+02	7.4E+01	7.4E+01	7.4E+01	7.4E+01
SB 2	BF 1	1.1E+00	*1.0E+00*	5.9E+00	7.4E+01	3.4E+01	*1.0E+00*	1.1E+01	8.7E+02	1.8E+00	7.4E+01	*1.0E+00*	*1.0E+00*
SB 2	AM 2	*1.0E+00*	2.1 E+00	8.0E+01	2.0E+01	2.3E+03	2.2E+01	8.9E+02	2.3E+03	7.4E+01	7.4E+01	4.1 E+02	4.1E+02
SB 2	AM 1	*1.0E+00*	6.2E+00	4.8E+00	6.0E+00	1.3E+03	5.9E+00	1.5 E+00	2.2 E+01	1.6E+01	1.9 E+00	8.3E+00	8.3E+00
SB 2	SB 1	*1.0E+00*	*1.0E+00*	1.6E+00	2.0E+00	1.2E+00	*1.0E+00*	2.9E+00	1.5E+00	1.0E+00	2.1 E+00	1.0E+00	1.0E+00
SB 2	WF 2	7.4E+06	9.3E+09	*1.0E+10*	3.6E+02	*1.0E+10*	*1.0E+10*	4.7E+07	*1.0E+10*	2.3E+01	1.1E+03	2.1E+04	2.1E+04
SB 2	WF 1	4.2 E+02	9.0E+07	6.3E+04	7.4E+01	1.5E+03	*1.0E+00*	1.1 E+02	*1.0E+10*	2.3E+00	7.4E+01	1.4E+05	1.4E+05
SB 3	AH 1	7.1E+02	7.1E+02	7.1E+02	6.8E+02	2.6E+07	1.2E+03	5.0E+02	1.9E+00	4.8E+02	7.1E+02	4.8E+02	4.8E+02
SB 3	AH 2	3.9E+01	6.4E+05	2.4E+07	1.9E+01	3.3E+01	5.1E+08	8.4E+01	3.0E+05	9.4E+02	7.1E+02	1.7E+03	1.7E+03
SB 3	BF 1	7.1 E+02	7.1E+02	7.1E+02	7.1E+02	6.7E+02	7.1E+02	7.1E+02	6.6 E+02	6.6E+02	7.1E+02	6.6E+02	6.6E+02
SB 3	AM 2	1.1E+03	1.8E+05	6.4E+05	1.3E+01	2.8E+03	*1.0E+10*	5.0E+05	*1.0E+10*	8.1E+01	3.1E+03	2.0E+01	2.0E+01
SB 3	AM 1	7.1E+02	7.1E+02	*1.0E+00*	1.6E+01	1.6E+03	4.6E+02	7.1E+02	1.2E+02	1.1E+02	7.1E+02	1.9E+00	1.9E+00
SB 3	SB 1	3.5E+03	7.1E+02	7.1E+02	2.2E+00	1.0E+03	7.1E+02	3.5E+00	2.5E+00	1.7E+01	7.1E+02	2.3E+00	2.3E+00
SB 3	WF 2	7.1E+02	7.1E+02	7.1E+02	7.1E+02	2.8E+03	7.1E+02	7.1E+02	2.8E+03	7.1E+02	7.1E+02	7.1E+02	7.1E+02
SDF	AH 1	1.1 E+05	*1.0E+10*	1.2E+06	8.9E+06	1.2E+08	*1.0E+10*	7.6E+00	*1.0E+10*	9.7E+09	2.6E+01	9.7E+09	9.7E+09
SDF	AH 2	1.1 E+05	1.0E+10	1.0E+10	3.7E+06	*1.0E+10*	*1.0E+10*	3.3E+03	*1.0E+10*	1.0E+10	4.2E+00	1.0E+10	1.0E+10
SDF	BF 2	1.0E+00	9.1E+02	6.1E+06	*1.0E+00*	1.9E+01	6.2E+04	4.9E+02	3.8E+00	*1.0E+00*	2.0E+00	*1.0E+00*	*1.0E+00*
SDF	AM 2	3.8E+06	*1.0E+10*	2.1E+09	4.9E+05	9.9E+02	*1.0E+10*	*1.0E+10*	*1.0E+10*	7.2 E+00	5.2E+01	2.1E+01	2.1E+01
SDF	AM 1	2.9E+03	*1.0E+10*	1.0E+00	3.9E+06	5.8E+02	*1.0E+10*	1.2E+03	4.9E+08	1.6 E+00	6.5E+00	8.8E+01	8.8E+01
SDF	SDF	6.5E+00	6.5E+00	6.5E+00	*1.0E+00*	9.3E+02	6.5E+00	9.2E+00	9.3E+02	8.8E+00	6.5E+00	8.8E+00	8.8E+00
SDF	SB 2	2.6E+00	*1.0E+00*	*1.0E+00*	*1.0E+00*	2.2 E+00	1.1 E+00	2.0E+00	2.7E+00	3.5E+00	2.0E+00	*1.0E+00*	1.5E+00
SDF	SB 3	1.0E+00	1.0E+10	6.7E+09	1.1 E+00	*1.0E+00*	*1.0E+10*	1.4E+00	*1.0E+00*	*1.0E+00*	1.3E+00	*1.0E+00*	*1.0E+00*
SDF	WF 2	*1.0E+10*	4.6E+00	*1.0E+10*	9.8E+09	1.2 E+00	*1.0E+10*	*1.0E+10*	2.7E+09	8.6E+09	*1.0E+10*	8.6E+09	8.6E+09
WF 1	AM 1	5.5E+00	5.5E+00	5.5E+00	5.5E+00	1.1E+04	5.5E+00	5.5E+00	1.1E+04	5.5E+00	5.5E+00	5.5E+00	5.5E+00
WF 1	SB 1	5.5E+00	5.5E+00	5.5E+00	1.3E+01	3.8E+03	5.5E+00	3.2E+03	3.8E+03	5.5E+00	5.5E+00	5.5E+00	5.5E+00
WF 1	WF 1	5.5E+00	5.5E+00	5.5E+00	5.5E+00	1.1E+00	5.5E+00	4.5E+02	*1.0E+00*	5.5E+00	5.5E+00	5.5E+00	5.5E+00
WF 2	AH 2	2.6E+00	2.6E+00	2.6E+00	2.6E+00	7.6E+03	2.6E+00	2.6E+00	7.6E+03	2.6E+00	2.6E+00	2.6E+00	2.6E+00
WF 2	BF 1	2.6E+00	2.6E+00	2.6E+00	2.6E+00	3.4E+02	2.6E+00	2.6E+00	3.4E+02	6.0E+02	2.6E+00	6.0E+02	6.0E+02
WF 2	AM 2	*1.0E+00*	1.7 E+00	3.1E+09	*1.0E+00*	3.4E+02	2.2 E+00	*1.0E+00*	3.4E+02	*1.0E+10*	*1.0E+00*	*1.0E+10*	*1.0E+10*
WF 2	AM 1	2.4E+02	1.6 E+00	*1.0E+00*	2.9E+02	4.8E+02	3.1E+00	9.7E+05	4.8E+02	7.7E+00	*1.0E+00*	1.6E+01	1.6E+01
WF 2	WF 2	*1.0E+00*	6.4E+01	*1.0E+10*	*1.0E+00*	*1.0E+00*	*1.0E+10*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*
WF 2	WF 1	*1.0E+00*	2.6E+00	3.2E+00	1.9E+01	*1.0E+00*	2.6E+00	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*	*1.0E+00*

Table 15. Summary table for scoring the performance of the equilibrium diagnostic for the top 10% of runs. 0 indicates possible unstable dynamics, 1 indicates an asymptotic relationship where F_{MSY} is poorly defined, and 2 indicates a dome-shaped equilibrium yield curve where a maximum yield can be defined.

Run num	Model num	Model label	Final SS	N est kij	N low	N high	SB	AM	SDF	BF	WF	AH
144	12	seasonal eggs, vuls, AH Rdev	1800	93	13	2	2	2	2	2	2	2
6	1	continuity	1824	78	20	2	2	2	1	1	1	0
137	11	seasonal eggs and vuls	1837	78	19	9	2	1	2	1	1	1
95	1	continuity	1865	63	15	3	2	1	2	2	1	1
150	12	seasonal eggs, vuls, AH Rdev	1874	93	12	2	2	2	2	2	2	2
138	11	seasonal eggs and vuls	1884	88	17	3	2	1	1	2	1	1
92	5	seasonal vuls	1888	88	17	3	2	1	0	2	1	1
105	2	PP annual	1890	79	21	22	2	1	2	1	2	2
77	2	PP annual	1895	77	19	9	2	1	2	2	2	2
86*	4	seasonal eggs	1934	85	11	3	2	2	2	2	2	2
139*	11	seasonal eggs and vuls	1971	90	9	2	2	2	2	2	2	2

Table 16. Proof-of-concept ecological reference points based on the NWACS-MICE tradeoff analysis.

Menhaden *F* reference point	**Run 150**
$F_{current}$ Ecosim	0.127
$F_{current}$ BAM F_{FULL}	0.256
ERP F_{TARGET} multiplier	0.737
ERP $F_{THRESHOLD}$ multiplier	1.916
ERP F_{TARGET} Ecosim	0.094
ERP $F_{THRESHOLD}$ Ecosim	0.244
ERP F_{TARGET} BAM F_{FULL}	0.189
ERP $F_{THRESHOLD}$ BAM F_{FULL}	0.490

Table 17. Comparison of ERP estimates for NWACS-MICE model run 150 under base and sensitivity run recruitment scenario for Atlantic herring.

	Run 150 (recent low recruitment deviations persist in future for Atlantic herring)	Run 150b (recruitment returns to long-term mean)
ERP F_{TARGET} multiplier	0.737	0.884
ERP $F_{THRESHOLD}$ multiplier	1.916	1.989
ERP F_{TARGET} (Ecosim units)	0.094	0.113
ERP $F_{THRESHOLD}$ (Ecosim units)	0.244	0.254
ERP F_{TARGET} (BAM F_{FULL} units)	0.189	0.226
ERP $F_{THRESHOLD}$ (BAM F_{FULL} units)	0.490	0.509

Table 18. Trophic group definitions for the NWACS-FULL ecosystem model.

Category	Node	GroupName	ParentCode	Code	Age (yr)	Length (cm)	Biomass time-series	Catch time-series
Primary producers	1	Phytoplankton	PHYT	PHYT.1	--	--		
Primary producers	2	Other primary producers	OPP	OPP.1	--	--		
Bacteria	3	Bacteria	BACT	BACT.1	--	--		
Zooplankton	4	Microzooplankton	MICZOO	MICZOO.1	--	--	INDEX	
Zooplankton	5	Small copepods	COPS	COPS.1	--	--	INDEX	
Zooplankton	6	Large copepods	COPL	COPL.1	--	--	INDEX	
Zooplankton	7	Gelatinous zooplankton	GELZOO	GELZOO.1	--	--	INDEX	
Zooplankton	8	Micronekton	MINEK	MINEK.1	--	--	INDEX	
Benthic Inverts	9	Macrobenthos - polychaetes	MACPOLY	MACPOLY.1	--	--		
Benthic Inverts	10	Macrobenthos - crustaceans	MACCRUS	MACCRUS.1	--	--		NOAA
Benthic Inverts	11	Macrobenthos - molluscs	MACMOLL	MACMOLL.	--	--		NOAA
Benthic Inverts	12	Macrobenthos - other	MACOTH	MACOTH.1	--	--		
Benthic Inverts	13	Megabenthos - filterers	MEGFILT	MEGFILT.1	--	--		NOAA
Benthic Inverts	14	Megabenthos - other	MEGOTH	MEGOTH.1	--	--		NOAA
Benthic Inverts	15	Shrimp and Similar Species	SHRIMP	SHRIMP.1	--	--		NOAA
Forage fishes	16	Mesopelagics	MESOPEL	MESOPEL.1	--	--		
Forage fishes	17	Atlantic herring (S)	AHERR	AHERR.1	0-1	<=13.5	SAR	SAR
Forage fishes	18	Atlantic herring (L)	AHERR	AHERR.2	2+	>13.5	SAR	SAR
Forage fishes	19	Alosines	ALOS	ALOS.1	--	--	NEFSC	NOAA
Forage fishes	20	Atlantic menhaden (S)	MENH	MENH.1	0	<1.52	SAR	SAR
Forage fishes	21	Atlantic menhaden (L)	MENH	MENH.2	1+	>=1.52	SAR	SAR
Forage fishes	22	Anchovies	ANCH	ANCH.1	--	--	INDEX	
Forage fishes	23	Atlantic mackerel	AMACK	AMACK.1	--	--	SAR	SAR
Forage fishes	24	Squid	SQUID	SQUID.1	--	--	NEFSC	NOAA
Forage fishes	25	Butterfish	BTRF	BTRF.1	--	--	SAR	SAR
Forage fishes	26	Small pelagic - other	SMPEL	SMPEL.1	--	--	NEAMAP	NOAA
Fishes	27	Bluefish (S)	BLUE	BLUE.1	0	<=32.3	SAR	SAR
Fishes	28	Bluefish (L)	BLUE	BLUE.2	1+	> 32.3	SAR	SAR
Fishes	29	Striped bass (S)	STBASS	STBASS.1	0-1	<=37.3	SAR	SAR
Fishes	30	Striped bass (M)	STBASS	STBASS.2	2-4	37.4-70.8	SAR	SAR
Fishes	31	Striped bass (L)	STBASS	STBASS.3	5+	>70.8	SAR	SAR
Fishes	32	Weakfish (S)	WEAK	WEAK.1	0	<=26.2	SAR	SAR
Fishes	33	Weakfish (L)	WEAK	WEAK.2	1+	>26.2	SAR	SAR
Fishes	34	Spiny dogfish	SPDOG	SPDOG.1	--	--	SAR	SAR
Fishes	35	Atlantic cod (S)	COD	COD.1	0-1	<=20	SAR	SAR
Fishes	36	Atlantic cod (M)	COD	COD.2	2-3	20.1-50	SAR	SAR
Fishes	37	Atlantic cod (L)	COD	COD.3	4+	>50	SAR	SAR
Fishes	38	Haddock	HAD	HAD.1	--	--	SAR	SAR
Fishes	39	Hakes	HAKE	HAKE.1	--	--	SAR	NOAA
Fishes	40	Atlantic croaker	CROAK	CROAK.1	--	--	NEAMAP	NOAA
Fishes	41	Yellowtail flounder (S)	YTF	YTF.1	0	<=20	SAR	SAR
Fishes	42	Yellowtail flounder (L)	YTF	YTF.2	1+	>20	SAR	SAR
Fishes	43	Summer flounder (S)	SUMFL	SUMFL.1	0	<=25	SAR	SAR
Fishes	44	Summer flounder (L)	SUMFL	SUMFL.2	1+	>25	SAR	SAR
Fishes	45	Skates	SKATE	SKATE.1	--	--	NEFSC	NOAA
Fishes	46	Demersal benthivores - other	DBOTH	DBOTH.1	--	--	NEFSC	NOAA
Fishes	47	Demersal piscivores - other	DPOTH	DPOTH.1	--	--	NEFSC	NOAA
Fishes	48	Demersal omnivores - other	DOOTH	DOOTH.1	--	--	NEFSC	NOAA
Fishes	49	Medium pelagic - other	MEDPEL	MEDPEL.1	--	--		NOAA
Apex predators	50	Sharks - coastal	SHARKC	SHARKC.1	--	--		NOAA
Apex predators	51	Sharks - pelagic	SHARKP	SHARKP.1	--	--		NOAA
Apex predators	52	Large pelagics (HMS)	HMS	HMS.1	--	--	SAR	SAR
Apex predators	53	Pinnipeds	PINN	PINN.1	--	--		
Apex predators	54	Baleen whales	WHALEB	WHALEB.1	--	--		
Apex predators	55	Odontocetes	WHALET	WHALET.1	--	--		
Apex predators	56	Seabirds	SBIRD	SBIRD.1	--	--		
Apex predators	57	Nearshore birds - piscivorous	NSBIRD	NSBIRD.1	--	--		
Apex predators	58	Osprey	OSPREY	OSPREY.1	--	--	INDEX	
Detritus	59	Detritus	DETRITUS	DETRITUS.1	--	--		

Time-series data sources: NOAA=NOAA/ACCSP/MRIP landings database, SAR=Stock assessment report, NEFSC=NEFSC Trawl Survey, NEAMAP=NEAMAP Trawl Survey, INDEX=other survey

Table 19. Stock assessment information used for Ecological Reference Point (ERP) and non-ERP fish species in the NWACS-FULL model. Stocks include Gulf of Maine (GOM), Georges Bank (GB), Southern New England (SNE), Mid Atlantic Bight (MAB) and Cape Cod (CC). Model Platforms include the Age Structured Assessment Program (ASAP), Beaufort Assessment Model (BAM), Woods Hole Assessment Model (WHAM), Stock Synthesis 3 (SS3), and Statistical Catch at Age (SCA), or an empirical approach using swept-area biomass (EMP). *For Atlantic Menhaden, we used the BAM model with the *M*=0.925 base run.

Group	Species Name	Stock	Assess. Year	Term. Year	Model Platform
ERP	Atlantic Herring	UNIT	2024	2023	ASAP
	Atlantic Menhaden	UNIT	2024	2023	BAM*
	Bluefish	UNIT	2023	2022	WHAM
	Spiny Dogfish	UNIT	2023	2022	SS3
	Striped Bass	UNIT	2024	2023	SCA
	Weakfish	UNIT	2024	2023	ASAP
Non-ERP	Atlantic Cod	Western GOM	2024	2023	WHAM
		Eastern GOM	2024	2023	WHAM
		GB	2024	2023	WHAM
		SNE	2024	2023	WHAM
	Atlantic mackerel	UNIT	2023	2022	ASAP
	Bluefin Tuna (HMS)	Western Atlantic	2021	2020	SS3
	Butterfish	UNIT	2024	2023	WHAM
	Haddock	GB	2022	2023	WHAM
		GOM	2022	2023	ASAP
	Longfin (Loligo) squid	UNIT	2023	2022	EMP
	Hake, White	UNIT	2022	2021	ASAP
	Hake, Red	Northern GB / GOM	2023	2022	EMP
		Southern GB / MAB	2023	2022	EMP
	Summer flounder	UNIT	2023	2022	ASAP
	Yellowtail Flounder	CC / GOM	2022	2021	VPA
		SNE / MAB	2022	2021	ASAP

Table 20. Number of individual stomachs contained in each of the diet data sources by predator. Numbers following the predator label represent fish ages for the multistanza groups.

Predators	ChesMMAP	MSVPA	NEAMAP	NEFSC	NJOT	RI	Total
alosines	18		6593	5844			*12455*
anchovies				15			*15*
atlantic cod 0-1			11	924			*935*
atlantic cod 2-3			8	8249			*8257*
atlantic cod 4			14	10537			*10551*
atlantic croaker	4844		3429	1708	93		*10074*
atlantic herring 0-1	2		158				*160*
atlantic herring 2	16		352	23491			*23859*
atlantic mackerel				9821			*9821*
atlantic menhaden 0			2	21			*23*
atlantic menhaden 1	3		8	184			*195*
bluefish 0	560	5661	3457	2859	117	22	*12676*
bluefish 1	29	1283	380	2633	64	211	*4600*
butterfish				10892			*10892*
demersal benthivore	97		24082		149	612	*24940*
demersal omnivore	41		4237		490	1125	*5893*
demersal piscivore	57		210				*267*
haddock				14854			*14854*
hakes				119144			*119144*
medium pelagics	27		1				*28*
skates	273		18438	75586			*94297*
small pelagics	76		81				*157*
spiny dogfish	145		2758	71431	381	15	*74730*
striped bass 0	2805	2460	57	14	4	12	*5352*
striped bass 2-5	1449	7166	169	970	47	236	*10037*
striped bass 6	91	2578	161	614	12	53	*3509*
summer flounder 0	1492		1094	1695	95	1	*4377*
summer flounder 1	2364		3922	20577	350	240	*27453*
weakfish 0	6087	535	7009	1613	291	35	*15570*
weakfish 1	1185	1459	1275	3864	93	161	*8037*
yellowtail flounder 0			2	706			*708*
yellowtail flounder 1			87	9816			*9903*
Total	*21661*	*21142*	*77995*	*398062*	*2186*	*2723*	*523769*

Table 21. Basic inputs and outputs for the NWACS-FULL 2023_v2.1 model. Values include biomass (*B*; mt/km^2), biomass accumulation rates (BA; yr^{-1}), total instantaneous mortality (*Z*; yr^{-1}) or production to biomass (P/B; yr^{-1}), consumption to biomass (Q/B; /yr), total landings (mt/km^2), trophic level (TL), ecotrophic efficiency (*EE*), production to consumption ratio (P/Q; yr^{-1}), fishing mortality rate (*F*; yr^{-1}), predation mortality rate (M_2; yr^{-1}), and other mortality (M_0; yr^{-1}).

N	Group	B	BA rate	Z or P/B	Q/B	Total Landings	TL	*EE*	P/Q	*F*	M_2	M_0
1	Phytoplankton	30.000		180.700		0.000	1.000	0.722			130.461	50.239
2	Other primary producers	1.603		55.570		0.000	1.000	0.900			50.013	5.557
3	Bacteria	7.700		91.250	380.208	0.000	2.000	0.814	0.240		74.261	16.989
4	Microzooplankton	5.500		85.000	283.400	0.000	2.264	0.868	0.300		73.765	11.235
5	Small copepods	12.139	0.003	46.000	140.000	0.000	2.152	0.867	0.329		39.860	6.137
6	Large copepods	12.116	0.063	46.000	150.000	0.000	2.388	0.941	0.307		43.240	2.697
7	Gelatinous zooplankton	4.488	-0.369	40.000	145.326	0.000	3.085	0.553	0.275		22.495	17.874
8	Micronekton	3.934	0.232	14.250	85.497	0.000	2.723	0.832	0.167		11.617	2.401
9	Macrobenthos - polychaetes	17.452		2.500	17.500	0.000	2.377	0.783	0.143		1.957	0.543
10	Macrobenthos - crustaceans	7.000		3.600	21.000	0.006	2.535	0.548	0.171	0.001	1.971	1.628
11	Macrobenthos - molluscs	8.340		2.200	13.949	0.183	2.246	0.858	0.158	0.022	1.865	0.313
12	Macrobenthos - other	21.000		2.000	16.059	0.000	2.349	0.834	0.125	0.000	1.668	0.332
13	Megabenthos - filterers	5.750		1.200	6.660	1.313	2.120	0.994	0.180	0.228	0.965	0.007
14	Megabenthos - other	4.498		2.300	15.533	0.191	2.895	0.920	0.148	0.042	2.073	0.185
15	Shrimp and Similar Species	0.470		2.050	6.660	0.022	2.751	0.993	0.308	0.046	1.989	0.015
16	Mesopelagics	0.090		1.100	3.700	0.000	3.238	0.884	0.297		0.972	0.128
17	Atlantic herring (S)	0.060	0.900	1.021	8.931	0.009	3.535	0.879	0.114	0.147	0.750	0.124
18	Atlantic herring (L)	0.318	0.900	0.759	3.700	0.130	3.561	0.988	0.205	0.409	0.341	0.009
19	Alosines	0.214	-0.174	1.300	4.400	0.017	3.414	0.507	0.295	0.080	0.752	0.641
20	Atlantic menhaden (S)	0.312	0.128	1.398	16.145	0.011	2.533	0.841	0.087	0.037	1.139	0.222
21	Atlantic menhaden (L)	2.520	0.128	1.295	6.449	0.788	2.837	0.595	0.201	0.313	0.458	0.524
22	Anchovies	0.635	0.092	2.200	7.333	0.000	3.060	0.732	0.300		1.517	0.591
23	Atlantic mackerel	1.197	-0.200	0.338	2.170	0.166	3.610	0.965	0.156	0.138	0.388	0.012
24	Squid	0.650	-0.349	5.720	19.000	0.027	3.832	0.999	0.301	0.041	6.023	0.005
25	Butterfish	0.495	-0.100	1.298	4.230	0.009	3.889	0.831	0.307	0.019	1.159	0.219
26	Small pelagic - other	1.601		1.200	4.000	0.018	3.397	0.970	0.300	0.011	1.153	0.036
27	Bluefish (S)	0.013		0.933	9.407	0.002	4.296	0.893	0.099	0.139	0.695	0.100
28	Bluefish (L)	0.465		0.749	3.000	0.116	4.443	0.536	0.250	0.250	0.151	0.347
29	Striped bass (S)	0.012	0.364	1.040	7.238	0.000	3.858	0.810	0.144	0.021	0.821	0.198
30	Striped bass (M)	0.041	0.364	0.420	3.027	0.002	3.923	0.954	0.139	0.046	0.354	0.019
31	Striped bass (L)	0.014	0.364	0.153	1.824	0.000	4.064	0.329	0.084	0.008	0.043	0.103
32	Weakfish (S)	0.008	0.950	0.726	9.037	0.001	3.979	0.974	0.080	0.146	0.561	0.019
33	Weakfish (L)	0.039	0.950	1.146	3.770	0.024	4.218	0.876	0.304	0.620	0.383	0.142
34	Spiny dogfish	1.668		0.201	1.700	0.054	4.184	0.732	0.118	0.032	0.114	0.054
35	Atlantic cod (S)	0.026	-0.200	0.600	8.013	0.002	3.582	0.650	0.075	0.067	0.323	0.210
36	Atlantic cod (M)	0.065	-0.200	1.416	4.482	0.077	3.948	0.997	0.316	1.183	0.229	0.005

37	Atlantic cod (L)	0.064	-0.200	0.786	2.620	0.045	4.286	0.958	0.300	0.710	0.043	0.033
38	Haddock	0.063	-0.120	0.519	3.000	0.020	3.524	0.975	0.173	0.319	0.307	0.013
39	Hake	0.720	-0.176	1.100	3.660	0.067	4.241	0.975	0.301	0.093	1.155	0.027
40	Atlantic croaker	0.065	-0.046	0.994	3.550	0.014	3.503	0.970	0.280	0.216	0.795	0.030
41	Yellowtail flounder (S)	0.003	0.200	0.550	6.020	0.000	3.500	0.961	0.091		0.529	0.021
42	Yellowtail flounder (L)	0.025	0.200	0.881	2.900	0.014	3.464	0.754	0.304	0.555	0.109	0.217
43	Summer flounder (S)	0.005	0.200	1.100	8.257	0.004	4.347	0.973	0.133	0.792	0.279	0.030
44	Summer flounder (L)	0.059	0.200	1.204	2.900	0.056	4.518	0.924	0.415	0.945	0.168	0.091
45	Skates	2.935		0.250	0.900	0.009	3.818	0.358	0.278	0.003	0.086	0.161
46	Demersal benthivores - other	2.127	-0.090	0.600	2.000	0.147	3.510	0.606	0.300	0.069	0.385	0.236
47	Demersal piscivores - other	1.356		0.450	1.500	0.050	4.068	0.372	0.300	0.037	0.131	0.282
48	Demersal omnivores - other	1.400		0.800	2.700	0.097	3.768	0.920	0.296	0.069	0.667	0.064
49	Medium pelagic - other	0.060		0.500	1.838	0.006	4.723	0.902	0.272	0.092	0.359	0.049
50	Sharks - coastal	0.025		0.200	1.247	0.004	4.611	0.981	0.160	0.146	0.051	0.004
51	Sharks - pelagic	0.050		0.200	0.690	0.009	4.656	0.955	0.290	0.172	0.019	0.009
52	Large pelagics (HMS)	0.080	-0.059	0.237	6.794	0.008	4.427	0.196	0.035	0.095	0.011	0.191
53	Pinnipeds	0.035		0.075	5.581	0.000	4.609	0.370	0.013		0.028	0.047
54	Baleen whales	0.373		0.040	3.217	0.000	4.138	0.048	0.012		0.002	0.038
55	Odontocetes	0.087		0.040	14.301	0.000	4.708	0.309	0.003		0.012	0.028
56	Seabirds	0.007		0.279	80.000	0.000	4.219	0.798	0.003		0.223	0.056
57	Nearshore birds - piscivorous	0.007		0.279	80.000	0.000	4.026	0.016	0.003		0.004	0.275
58	Osprey	0.000	0.018	0.279	80.000	0.000	4.439	0.065	0.003			0.261
59	Detritus	52.600		0.000		0.000	1.000	0.698				

Table 22. Diet composition for the NWACS-FULL 2023_v2.1 model. Columns indicate the predators (labeled by node number) and rows are prey. Values <0.0005 are listed as 0.000. The table is broken up into 3 sections to fit onto the pages.

N	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		0.000		0.278
2	Other primary producers	0.023						0.015	0.012	0.010	0.013	0.006					0.000		0.023
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		0.000		0.140
5	Small copepods			0.010	0.114	0.303	0.149		0.015						0.439			0.315	0.140
6	Large copepods				0.065	0.432	0.323		0.033						0.429	0.425	0.360	0.315	0.140
7	Gelatinous zooplankton				0.042	0.035											0.011	0.000	
8	Micronekton						0.060	0.015	0.014	0.009	0.019			0.134	0.013	0.151	0.248	0.238	
9	Macrobenthos - polychaetes							0.005	0.099		0.021		0.133			0.000	0.000	0.000	
10	Macrobenthos - crustaceans				0.001			0.003	0.001		0.001		0.046	0.026		0.331	0.312	0.130	
11	Macrobenthos - molluscs							0.001	0.010	0.001	0.011		0.110				0.000	0.000	
12	Macrobenthos - other				0.001			0.014	0.084	0.011	0.011		0.146	0.061			0.000	0.000	
13	Megabenthos - filterers							0.003	0.014	0.010	0.001		0.013				0.000		
14	Megabenthos - other							0.001	0.002	0.001	0.007		0.012			0.092	0.004	0.000	
15	Shrimp and Similar Species													0.001			0.036	0.000	
16	Mesopelagics																0.000		
17	Atlantic herring (S)																0.001		
18	Atlantic herring (L)																		
19	Alosines																		
20	Atlantic menhaden (S)																		
21	Atlantic menhaden (L)																		
22	Anchovies															0.000	0.026	0.000	
23	Atlantic mackerel																		
24	Squid					0.000											0.000	0.000	
25	Butterfish																		
26	Small pelagic - other																0.000	0.000	
27	Bluefish (S)																		
28	Bluefish (L)																		
29	Striped bass (S)																		
30	Striped bass (M)																		
31	Striped bass (L)																		
32	Weakfish (S)																		
33	Weakfish (L)																		
34	Spiny dogfish																		
35	Atlantic cod (S)																		
36	Atlantic cod (M)																		
37	Atlantic cod (L)																		
38	Haddock																		
39	Hake																0.001	0.000	
40	Atlantic croaker																		
41	Yellowtail flounder (S)																		
42	Yellowtail flounder (L)																		
43	Summer flounder (S)																		
44	Summer flounder (L)																		
45	Skates																		
46	Demersal benthivores - other																	0.000	
47	Demersal piscivores - other																0.000		
48	Demersal omnivores - other															0.000	0.000		
49	Medium pelagic - other																		
50	Sharks - coastal																		
51	Sharks - pelagic																		
52	Large pelagics (HMS)																		
53	Pinnipeds																		
54	Baleen whales																		
55	Odontocetes																		
56	Seabirds														0.001				
57	Nearshore birds - piscivorous																		
58	Osprey																		
59	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
--	Import																		

N	Prey \ predator	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
1	Phytoplankton	0.163	0.130					0.000	0.000	0.000	0.002	0.000	0.000	0.002	0.000					
2	Other primary producers	0.014	0.025					0.000	0.000	0.001	0.001	0.000	0.002	0.001	0.000					
3	Bacteria																			
4	Microzooplankton	0.220								0.000										
5	Small copepods	0.220	0.300			0.032	0.454				0.000		0.000				0.000	0.000	0.000	0.000
6	Large copepods	0.220	0.300	0.401	0.152	0.032	0.172	0.000		0.010	0.000		0.007	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7	Gelatinous zooplankton			0.000	0.008	0.643	0.018	0.000	0.000	0.001	0.001	0.000	0.000	0.000	0.085		0.000	0.000	0.000	0.000
8	Micronekton		0.084	0.454	0.525	0.088	0.186	0.007	0.002	0.101	0.026	0.000	0.296	0.111	0.013	0.134	0.053	0.010	0.079	0.166
9	Macrobenthos - polychaetes		0.010	0.000		0.000	0.001	0.002	0.000	0.173	0.122	0.008	0.014	0.008	0.044	0.000	0.019	0.013	0.138	0.005
10	Macrobenthos - crustaceans		0.091	0.101	0.118	0.202	0.135	0.014	0.003	0.131	0.096	0.013	0.135	0.062	0.028	0.767	0.163	0.029	0.228	0.193
11	Macrobenthos - molluscs			0.000				0.016	0.009	0.024	0.015	0.011	0.007	0.016	0.091	0.000	0.054	0.055	0.058	0.012
12	Macrobenthos - other			0.000	0.021	0.000	0.004	0.000	0.000	0.017	0.011	0.000	0.001	0.000	0.006	0.000	0.053	0.009	0.370	0.000
13	Megabenthos - filterers								0.000	0.000	0.001	0.000	0.000	0.000	0.018	0.000	0.029	0.033	0.005	0.000
14	Megabenthos - other			0.000	0.004			0.007	0.014	0.020	0.078	0.049	0.017	0.020	0.062	0.000	0.209	0.199	0.039	0.041
15	Shrimp and Similar Species			0.000	0.001	0.000	0.001	0.003	0.001	0.010	0.006	0.000	0.028	0.013	0.007	0.097	0.090	0.017	0.009	0.080
16	Mesopelagics				0.001			0.000	0.000						0.000					0.000
17	Atlantic herring (S)			0.001	0.000	0.000		0.001	0.001		0.009	0.013		0.000	0.001		0.005	0.024	0.019	0.005
18	Atlantic herring (L)				0.000				0.006		0.001	0.026		0.000	0.001		0.002	0.142		0.001
19	Alosines					0.000		0.002	0.009	0.004	0.006	0.004	0.000	0.000	0.012		0.000	0.001	0.000	0.000
20	Atlantic menhaden (S)				0.001	0.000		0.016	0.064	0.047	0.185	0.164	0.002	0.038	0.010					
21	Atlantic menhaden (L)				0.001				0.101		0.030	0.319		0.022	0.140					
22	Anchovies			0.005		0.000	0.024	0.528	0.137	0.325	0.146	0.082	0.306	0.285	0.023		0.000	0.000		0.000
23	Atlantic mackerel			0.005	0.001	0.000			0.006		0.001	0.000	0.000	0.000	0.016		0.000	0.016	0.000	0.024
24	Squid			0.000	0.158	0.000	0.003	0.025	0.145	0.000	0.004	0.003	0.002	0.033	0.111		0.000	0.001	0.000	0.129
25	Butterfish			0.005	0.001	0.000		0.045	0.134	0.000	0.006	0.032	0.000	0.013	0.015		0.000	0.000	0.000	0.026
26	Small pelagic - other			0.009	0.001	0.000	0.001	0.056	0.101	0.003	0.054	0.083	0.002	0.027	0.034	0.000	0.221	0.212	0.053	0.116
27	Bluefish (S)					0.000		0.000	0.002	0.000	0.001	0.000	0.000	0.000	0.000			0.000		
28	Bluefish (L)								0.024			0.000			0.004			0.000		
29	Striped bass (S)					0.000		0.002	0.000	0.000	0.000	0.000	0.000		0.003					
30	Striped bass (M)								0.000		0.000	0.000			0.005					
31	Striped bass (L)											0.000								
32	Weakfish (S)					0.000		0.005	0.000	0.004	0.001	0.000	0.004	0.010						
33	Weakfish (L)								0.005		0.000	0.000		0.004	0.001					
34	Spiny dogfish					0.000		0.000	0.000				0.000		0.000			0.000		0.000
35	Atlantic cod (S)					0.000				0.000					0.000		0.000	0.000		0.000
36	Atlantic cod (M)														0.000			0.000		
37	Atlantic cod (L)														0.000			0.000		
38	Haddock					0.000			0.001						0.000		0.000	0.002	0.000	0.000
39	Hake			0.005	0.001	0.000		0.006	0.019	0.011	0.012	0.004	0.008	0.018	0.013	0.000	0.071	0.125	0.000	0.143
40	Atlantic croaker					0.000		0.001	0.001	0.008	0.001	0.014	0.001	0.006	0.011					
41	Yellowtail flounder (S)					0.000			0.000						0.000		0.000	0.000		
42	Yellowtail flounder (L)								0.000						0.000			0.000		
43	Summer flounder (S)					0.000									0.000					
44	Summer flounder (L)														0.000					
45	Skates				0.001	0.000			0.000		0.000	0.000			0.003		0.000	0.000		0.000
46	Demersal benthivores - other			0.005	0.001	0.000		0.015	0.116	0.020	0.089	0.075	0.006	0.065	0.055	0.000	0.028	0.079	0.000	0.038
47	Demersal piscivores - other			0.005	0.001	0.000		0.000	0.000		0.000	0.000	0.000	0.001	0.008		0.000	0.009	0.000	0.020
48	Demersal omnivores - other			0.005	0.001	0.000		0.246	0.096	0.088	0.092	0.097	0.159	0.242	0.177	0.000	0.000	0.022	0.000	0.000
49	Medium pelagic - other				0.001															
50	Sharks - coastal				0.000	0.000							0.000		0.000			0.000		
51	Sharks - pelagic																			
52	Large pelagics (HMS)					0.000			0.000											
53	Pinnipeds																			
54	Baleen whales																			
55	Odontocetes																			
56	Seabirds																			
57	Nearshore birds - piscivorous																			
58	Osprey																			
59	Detritus	0.163	0.060					0.000	0.000	0.002	0.002		0.003	0.000	0.001					
--	Import																			

N	Prey \ predator	40	41	42	43	44	45	46	47	48	49	50	51	52	53	54	55	56	57	58
1	Phytoplankton					0.000		0.005		0.012										
2	Other primary producers	0.000			0.000	0.000		0.000	0.002	0.000				0.001						
3	Bacteria																			
4	Microzooplankton																			
5	Small copepods	0.000		0.000	0.000	0.000	0.000													
6	Large copepods	0.000		0.000	0.000	0.000	0.000	0.000		0.000			0.025					0.046		
7	Gelatinous zooplankton			0.000		0.000	0.000	0.006		0.000	0.001	0.001	0.003	0.000						
8	Micronekton	0.024	0.057	0.028	0.178	0.015	0.014	0.000	0.036	0.020	0.032	0.036		0.000		0.132		0.190		
9	Macrobenthos - polychaetes	0.567	0.294	0.568		0.000	0.143	0.142	0.022	0.066	0.001									
10	Macrobenthos - crustaceans	0.136	0.648	0.313	0.117	0.049	0.287	0.104	0.064	0.139	0.025	0.012	0.010	0.005					0.028	
11	Macrobenthos - molluscs	0.106	0.000	0.015	0.000	0.001	0.086	0.308	0.060	0.090		0.003								
12	Macrobenthos - other	0.000	0.000	0.043			0.001	0.105	0.001	0.000		0.019	0.010							
13	Megabenthos - filterers	0.000		0.000		0.000	0.001	0.009	0.001	0.012		0.003								
14	Megabenthos - other	0.077	0.000	0.006	0.097	0.081	0.207	0.319	0.403	0.573		0.019		0.020	0.010	0.003	0.009		0.028	
15	Shrimp and Similar Species	0.000	0.000	0.000	0.005	0.001	0.017	0.000	0.037	0.005				0.019	0.004	0.344	0.001	0.027		
16	Mesopelagics				0.000		0.000		0.001		0.055		0.005		0.002	0.006	0.046			
17	Atlantic herring (S)			0.000		0.010	0.001		0.001	0.000	0.009	0.015	0.017		0.005	0.000	0.000	0.006		0.006
18	Atlantic herring (L)						0.001					0.046	0.052	0.082	0.005	0.005	0.004	0.012		0.006
19	Alosines					0.000	0.000		0.007	0.000	0.027	0.062	0.011	0.054				0.020	0.088	0.163
20	Atlantic menhaden (S)				0.001	0.022	0.000		0.010	0.000		0.005	0.001		0.051	0.010	0.043	0.041	0.110	0.058
21	Atlantic menhaden (L)					0.007			0.051			0.059	0.037	0.247	0.051	0.090	0.043	0.089	0.220	0.058
22	Anchovies	0.076			0.165	0.071	0.000		0.004	0.008	0.006	0.062	0.021		0.014	0.013	0.012	0.122	0.276	
23	Atlantic mackerel					0.043	0.001		0.011			0.062	0.069	0.097	0.020	0.061	0.088	0.082		
24	Squid	0.002			0.101	0.224	0.013	0.000	0.041	0.018	0.283	0.126	0.155	0.070	0.059	0.035	0.490	0.074		
25	Butterfish					0.024	0.000		0.017	0.002	0.257	0.062	0.069	0.014	0.038	0.031	0.045	0.104		0.000
26	Small pelagic - other	0.002	0.000	0.027	0.001	0.099	0.139	0.000	0.021	0.040	0.083	0.062	0.021	0.198	0.333	0.163	0.030	0.122	0.110	0.138
27	Bluefish (S)					0.000			0.002				0.001							0.008
28	Bluefish (L)											0.039	0.039	0.043						0.008
29	Striped bass (S)												0.007						0.001	0.020
30	Striped bass (M)											0.009	0.009							0.020
31	Striped bass (L)											0.009	0.009							
32	Weakfish (S)						0.000			0.000			0.007							0.000
33	Weakfish (L)					0.011			0.001				0.015	0.000					0.000	0.000
34	Spiny dogfish					0.000	0.000		0.005			0.034	0.035	0.020	0.038	0.053	0.072	0.007		
35	Atlantic cod (S)								0.003			0.004	0.005					0.003		
36	Atlantic cod (M)								0.003			0.004	0.005	0.012				0.003		
37	Atlantic cod (L)											0.004	0.005	0.005						
38	Haddock					0.000	0.000		0.000			0.013	0.010	0.001	0.066	0.000	0.001			
39	Hake	0.002		0.000	0.269	0.082	0.027		0.024	0.000	0.116	0.013	0.014	0.033	0.172	0.019	0.065	0.007		
40	Atlantic croaker					0.000	0.000		0.006			0.013	0.010						0.006	0.011
41	Yellowtail flounder (S)						0.000		0.000			0.006	0.005							
42	Yellowtail flounder (L)								0.001			0.006	0.005							
43	Summer flounder (S)					0.000						0.006	0.007		0.000		0.000	0.001		0.005
44	Summer flounder (L)					0.000						0.006	0.007		0.039		0.000	0.003		0.005
45	Skates					0.000	0.000		0.054		0.059	0.020	0.019	0.008	0.025	0.035	0.048			
46	Demersal benthivores - other	0.002			0.066	0.132	0.053		0.055	0.004	0.042	0.013	0.010	0.013	0.068		0.000		0.011	0.266
47	Demersal piscivores - other	0.002				0.011	0.004		0.026			0.013	0.014					0.007	0.011	0.008
48	Demersal omnivores - other	0.002			0.001	0.116	0.004	0.000	0.028	0.010	0.005	0.062	0.080	0.015				0.037	0.011	0.086
49	Medium pelagic - other												0.021	0.010	0.000	0.001	0.002			0.000
50	Sharks - coastal											0.002	0.008	0.000						
51	Sharks - pelagic											0.010	0.019							
52	Large pelagics (HMS)											0.009	0.009							
53	Pinnipeds											0.015	0.015							
54	Baleen whales											0.012	0.010							
55	Odontocetes											0.012	0.020							
56	Seabirds											0.017	0.020							
57	Nearshore birds - piscivorous											0.001								
58	Osprey																			
59	Detritus	0.000				0.000				0.000			0.050							
--	Import													0.032					0.100	0.131

Table 23. Summary of the different NWACS-FULL models fit and compared of different "phases" of model comparison (A = initial fitting; B = modifications of best fit from phase A; C = additional ad hoc scenarios). Models differed in the forcing function used (PP = primary production), the starting value for the k_{ij} parameters (either all k_{ij} = 2, or k_{ij} estimated for ERP species based on the unfished biomass to the starting 1985 biomass [B_{unf}/B_o]). The number of iterations (Iter) used in the fitting procedure during Phase A, and the residual sum of squares (SS) for each model is included. Three different types of vulnerability adjustments were made to models: 1) v_{adj} - manual adjustments to the minimum k_{ij} for ERP species were made to improve the F_{MSY} dynamics; 2) v_{min} – a minimum k_{ij} value of 1.01 was used for all groups; and 3) v_{cap} – a maximum k_{ij} value was used for all groups. The number of k_{ij} parameters estimated (out of a total of 1083) is included as well as ratios of that number to the total number of data points in the model (n=3057) and the ratio with the total number of k_{ij} in the model (total k_{ij} = 1083). The number of k_{ij} estimated on a lower bound (k_{ij} <1.01) or upper bound (k_{ij} >=1e^9) is noted. Bold indicates the best model within each phase of fitting.

Phase	Sim	Forcing	v_{adj}	v_{min}	v_{cap}	k_{ij} start	Iter	SS	Num k_{ij} est	Est. k_{ij} / data pts	Est. k_{ij} / total k_{ij}	Num k_{ij} <1.01	Num k_{ij} >=1e9
A	1.1	none				k_{ij} =2	11	4708	239	0.08	0.22	85	32
	2.1	**none**				**B_{unf}/B_o**	**06**	**4552**	**230**	**0.08**	**0.21**	**70**	**57**
	3.1	PP				k_{ij} =2	13	4755	281	0.09	0.26	96	48
	4.1	PP				B_{unf}/B_o	15	4647	293	0.10	0.27	88	55
B	2.2	none		X		B_{unf}/B_o	--	6067	230	0.08	0.21	0	57
	2.3	none			X	B_{unf}/B_o	--	8535	229	0.07	0.21	71	0
	2.4	none		X	X	B_{unf}/B_o	--	7431	229	0.07	0.21	0	0
	2.5	**none**	**X**			**B_{unf}/B_o**	**--**	**4767**	**230**	**0.08**	**0.21**	**49**	**57**
	2.6	none	X	X		B_{unf}/B_o	--	6086	230	0.08	0.21	0	57
	2.7	none	X		X	B_{unf}/B_o	--	8819	229	0.07	0.21	50	0
	2.8	none	X	X	X	B_{unf}/B_o	--	7493	229	0.07	0.21	0	0
C	**2.9**	**none**	**X**			**B_{unf}/B_o**	**--**	**4769**	**230**	**0.08**	**0.21**	**49**	**57**

Table 24. Minimum vulnerability (k_{ij}) used for each of the six ERP species for scenarios that had a manual adjustment based on the equilibrium F_{MSY} analysis. The number of individual k_{ij} parameters that were changed is indicated within parentheses. Striped Bass (STBASS) and Spiny dogfish (SPDOG) did not have any k_{ij} on the lower bound so no changes were made. The two labels correspond to different adjustments made during Phase B (v_{adj}; for sims 2.5-2.8) and Phase C (v_{adj2}; for sim 2.9) of the calibration process (Table 23).

	Minimum kij (for each ERP spp)					
Label	MENH	AHERR	BLUE	STBASS	WEAK	SPDOG
v_{adj}	1.10 (n=2)	1.01 (n=1)	1.01 (n=5)	--	1.2 (n=13)	--
v_{adj2}	1.25 (n=2)	1.01 (n=1)	1.01 (n=5)	--	1.2 (n=13)	--

Table 25. List of predator-prey pairs where vulnerability parameters (k_{ij}) hit a lower or upper bound. ERP species in bold font, and k_{ij} adjusted manually for sim 2.9 are indicated with X.

Node	Predator	Prey	Bound	Adjusted
7	Gelatinous zooplankton	Large copepods	Lower	
8	Micronekton	Micronekton	Upper	
10	Macrobenthos - crustaceans	Detritus	Lower	
10	Macrobenthos - crustaceans	Macrobenthos - other	Lower	
14	Megabenthos - other	Macrobenthos - polychaetes	Lower	
16	Mesopelagics	Seabirds	Lower	
17	**Atlantic herring (S)**	Large copepods	**Upper**	
17	**Atlantic herring (S)**	Macrobenthos - crustaceans	**Upper**	
17	**Atlantic herring (S)**	Megabenthos - other	**Lower**	**X**
17	**Atlantic herring (S)**	Micronekton	**Upper**	
18	**Atlantic herring (L)**	Anchovies	**Upper**	
18	**Atlantic herring (L)**	**Atlantic herring (S)**	**Upper**	
18	**Atlantic herring (L)**	Large copepods	**Upper**	
19	Alosines	Macrobenthos - crustaceans	Upper	
19	Alosines	Micronekton	Upper	
21	**Atlantic menhaden (L)**	Detritus	**Lower**	**X**
21	**Atlantic menhaden (L)**	Large copepods	**Upper**	
21	**Atlantic menhaden (L)**	Microzooplankton	**Lower**	**X**
21	**Atlantic menhaden (L)**	Phytoplankton	**Upper**	
21	**Atlantic menhaden (L)**	Small copepods	**Upper**	
23	Atlantic mackerel	Large copepods	Upper	
24	Squid	**Atlantic herring (S)**	**Upper**	
24	Squid	Demersal benthivores - other	Upper	
24	Squid	Medium pelagic - other	Lower	
24	Squid	Squid	Lower	
26	Small pelagic - other	Small copepods	Lower	
27	**Bluefish (S)**	Anchovies	**Upper**	
27	**Bluefish (S)**	Demersal omnivores - other	**Upper**	
28	**Bluefish (L)**	**Atlantic herring (L)**	**Lower**	**X**
28	**Bluefish (L)**	**Atlantic menhaden (L)**	**Lower**	**X**
28	**Bluefish (L)**	**Bluefish (L)**	**Lower**	**X**
28	**Bluefish (L)**	**Bluefish (S)**	**Lower**	**X**
28	**Bluefish (L)**	Small pelagic - other	**Upper**	
28	**Bluefish (L)**	**Weakfish (L)**	**Lower**	**X**
30	**Striped bass (M)**	**Atlantic herring (S)**	**Upper**	
32	**Weakfish (S)**	Anchovies	**Lower**	**X**
32	**Weakfish (S)**	Demersal omnivores - other	**Lower**	**X**
32	**Weakfish (S)**	Macrobenthos - crustaceans	**Lower**	**X**

32	**Weakfish (S)**	Micronekton	**Lower**	**X**
32	**Weakfish (S)**	**Weakfish (S)**	**Upper**	
33	**Weakfish (L)**	Anchovies	**Lower**	**X**
33	**Weakfish (L)**	**Atlantic menhaden (L)**	**Lower**	**X**
33	**Weakfish (L)**	**Atlantic menhaden (S)**	**Lower**	**X**
33	**Weakfish (L)**	Demersal benthivores - other	**Lower**	**X**
33	**Weakfish (L)**	Demersal omnivores - other	**Lower**	**X**
33	**Weakfish (L)**	Macrobenthos - crustaceans	**Lower**	**X**
33	**Weakfish (L)**	Megabenthos - other	**Upper**	
33	**Weakfish (L)**	Micronekton	**Lower**	**X**
33	**Weakfish (L)**	Small pelagic - other	**Lower**	**X**
33	**Weakfish (L)**	Squid	**Lower**	**X**
33	**Weakfish (L)**	**Weakfish (L)**	**Upper**	
33	**Weakfish (L)**	**Weakfish (S)**	**Upper**	
34	**Spiny dogfish**	**Weakfish (L)**	**Upper**	
36	Atlantic cod (M)	**Atlantic herring (S)**	**Lower**	
37	Atlantic cod (L)	**Atlantic herring (L)**	**Lower**	
37	Atlantic cod (L)	**Atlantic herring (S)**	**Lower**	
37	Atlantic cod (L)	Demersal piscivores - other	Lower	
37	Atlantic cod (L)	Hake	Lower	
37	Atlantic cod (L)	Macrobenthos - molluscs	Lower	
37	Atlantic cod (L)	Megabenthos - filterers	Lower	
37	Atlantic cod (L)	Megabenthos - other	Lower	
37	Atlantic cod (L)	Small pelagic - other	Lower	
38	Haddock	**Atlantic herring (S)**	**Upper**	
38	Haddock	Macrobenthos - crustaceans	Lower	
38	Haddock	Macrobenthos - other	Upper	
38	Haddock	Macrobenthos - polychaetes	Upper	
39	Hake	**Atlantic herring (L)**	**Upper**	
39	Hake	**Atlantic herring (S)**	**Upper**	
42	Yellowtail flounder (L)	Macrobenthos - crustaceans	Lower	
42	Yellowtail flounder (L)	Macrobenthos - other	Lower	
42	Yellowtail flounder (L)	Macrobenthos - polychaetes	Lower	
43	Summer flounder (S)	Anchovies	Lower	
43	Summer flounder (S)	Hake	Lower	
43	Summer flounder (S)	Micronekton	Lower	
44	Summer flounder (L)	**Atlantic herring (S)**	**Upper**	
44	Summer flounder (L)	**Atlantic menhaden (S)**	**Lower**	
44	Summer flounder (L)	Demersal benthivores - other	Lower	
44	Summer flounder (L)	Demersal omnivores - other	Lower	
44	Summer flounder (L)	Macrobenthos - crustaceans	Lower	
44	Summer flounder (L)	Small pelagic - other	Lower	
44	Summer flounder (L)	Squid	Lower	
44	Summer flounder (L)	**Weakfish (L)**	**Upper**	
45	Skates	**Atlantic herring (L)**	**Upper**	

45	Skates	**Atlantic herring (S)**	**Upper**
45	Skates	Demersal benthivores - other	Lower
45	Skates	Haddock	Lower
45	Skates	Macrobenthos - crustaceans	Upper
45	Skates	Megabenthos - other	Lower
46	Demersal benthivores - other	Gelatinous zooplankton	Lower
46	Demersal benthivores - other	Macrobenthos - other	Upper
47	Demersal piscivores - other	Alosines	Upper
47	Demersal piscivores - other	Atlantic cod (M)	Upper
47	Demersal piscivores - other	Atlantic cod (S)	Upper
47	Demersal piscivores - other	Atlantic croaker	Lower
47	Demersal piscivores - other	**Atlantic herring (S)**	**Upper**
47	Demersal piscivores - other	**Bluefish (S)**	**Lower**
47	Demersal piscivores - other	Demersal benthivores - other	Upper
47	Demersal piscivores - other	Demersal omnivores - other	Upper
47	Demersal piscivores - other	Haddock	Lower
47	Demersal piscivores - other	Macrobenthos - molluscs	Lower
47	Demersal piscivores - other	Squid	Upper
47	Demersal piscivores - other	**Weakfish (L)**	**Upper**
47	Demersal piscivores - other	Yellowtail flounder (L)	Upper
48	Demersal omnivores - other	Megabenthos - other	Lower
49	Medium pelagic - other	**Atlantic herring (S)**	**Upper**
50	Sharks - coastal	**Atlantic herring (L)**	**Upper**
50	Sharks - coastal	Seabirds	Lower
51	Sharks - pelagic	**Atlantic herring (L)**	**Lower**
51	Sharks - pelagic	**Striped bass (L)**	**Lower**
52	Large pelagics (HMS)	**Atlantic herring (L)**	**Upper**
52	Large pelagics (HMS)	**Atlantic menhaden (L)**	**Lower**
52	Large pelagics (HMS)	Medium pelagic - other	Upper
53	Pinnipeds	Haddock	Lower
53	Pinnipeds	Summer flounder (L)	Lower
54	Baleen whales	**Atlantic herring (L)**	**Upper**
55	Odontocetes	**Atlantic herring (L)**	**Upper**
55	Odontocetes	Haddock	Lower
55	Odontocetes	**Spiny dogfish**	**Lower**
55	Odontocetes	Squid	Upper
56	Seabirds	**Atlantic herring (L)**	**Upper**
56	Seabirds	**Atlantic herring (S)**	**Upper**
56	Seabirds	Micronekton	Upper
57	Nearshore birds - piscivorous	Anchovies	Lower
57	Nearshore birds - piscivorous	**Atlantic menhaden (L)**	**Upper**

Table 26. Effects of three different menhaden fishing mortality rates on the equilibrium (i.e., 2063) biomass and catch of different trophic groups. Biomass is expressed relative to the 2063 equilibrium biomass under the status quo menhaden fishing scenario (B_{2063}/B_{SQ}) while catch is relative to the maximum equilibrium 2063 catch across all menhaden fishing scenarios (C_{2063}/C_{max}). Non-menhaden ERP species were kept at their target *F* for these projections and all other groups and fishing fleets were maintained at 2023 levels. Menhaden fishing mortality rates evaluated were: no menhaden fishing (F_0), status quo (2023) menhaden fishing (F_{SQ}), and the maximum menhaden *F* (F_{max}) which is twenty times greater F_{SQ}. Groups with a direct trophic link with menhaden are marked with "yes". The dashed line separates relative biomass "losers" from "winners" (B_{2063}/B_{SQ}) for the F_{max} scenario, and only groups experiencing at least a 25% change are shown. ERP groups are in bold italics, and values differing from 1 by more than 10% are in bold.

	Menh.	F_0		F_{SQ}		F_{max}	
Group	link	B_{2063}/B_{SQ}	C_{2063}/C_{max}	B_{2063}/B_{SQ}	C_{2063}/C_{max}	B_{2063}/B_{SQ}	C_{2063}/C_{max}
Atlantic menhaden	yes	1.08	**0.00**	1.00	**0.32**	**0.03**	**0.19**
Medium pelagic - other		1.04	1.00	1.00	0.96	**0.10**	**0.09**
Nearshore birds - piscivorous	yes	**1.34**	--	1.00	--	**0.30**	--
Osprey	yes	1.07	--	1.00	--	**0.33**	--
Haddock		1.04	1.00	1.00	0.97	**0.36**	**0.35**
Striped bass	yes	1.06	1.00	1.00	0.96	**0.39**	**0.45**
Seabirds	yes	1.04	--	1.00	--	**0.56**	--
Butterfish		1.01	1.00	1.00	0.99	**0.61**	**0.60**
Large pelagics (HMS)	yes	1.02	1.00	1.00	0.98	**0.66**	**0.65**
Sharks - pelagic	yes	1.02	1.00	1.00	0.98	**0.73**	**0.72**
Spiny dogfish	yes	1.05	1.00	1.00	0.95	**0.81**	**0.77**
Atlantic herring		1.00	**0.79**	1.00	**0.79**	**0.82**	**0.64**
Weakfish	yes	0.99	**0.87**	1.00	**0.88**	1.09	0.97
small pelagic - other		1.00	**0.76**	1.00	**0.77**	**1.28**	0.98
Macrobenthos - crustaceans		1.00	**0.78**	1.00	**0.78**	**1.28**	1.00
Macrobenthos - polychaetes		1.00	--	1.00	--	**1.29**	--
Skates		0.99	**0.76**	1.00	**0.77**	**1.30**	1.00
Megabenthos - other		0.99	**0.76**	1.00	**0.77**	**1.31**	1.00
Yellowtail flounder		1.00	**0.69**	1.00	**0.69**	**1.43**	1.00
Bluefish	yes	0.99	**0.41**	1.00	**0.42**	**2.38**	1.00
Demersal benthivores - other		0.99	**0.40**	1.00	**0.41**	**2.46**	1.00

Table 27. Stock status of Atlantic menhaden based on proof-of-concept ERPs. Fishing mortality is the full fishing mortality. Fecundity is in billions of eggs.

Reference Point	ERP Value	2023 Value	Stock Status
$F_{THRESHOLD}$	0.490	0.26	Not Overfishing
F_{TARGET}	0.189		
$FEC_{THRESHOLD}$	1,142,453	1,240,272	Not Overfished
FEC_{TARGET}	1,666,030		

Table 28. Probability of exceeding the proof-of-concept ERP $F_{THRESHOLD}$ and F_{TARGET} for 2024-2028 under a constant status quo TAC.

	2024	2025	2026	2027	2028
ERP $F_{THRESHOLD}$	0.0%	0.0%	0.0%	0.5%	0.5%
ERP F_{TARGET}	100.0%	99.5%	99.5%	99.5%	99.5%

18 FIGURES

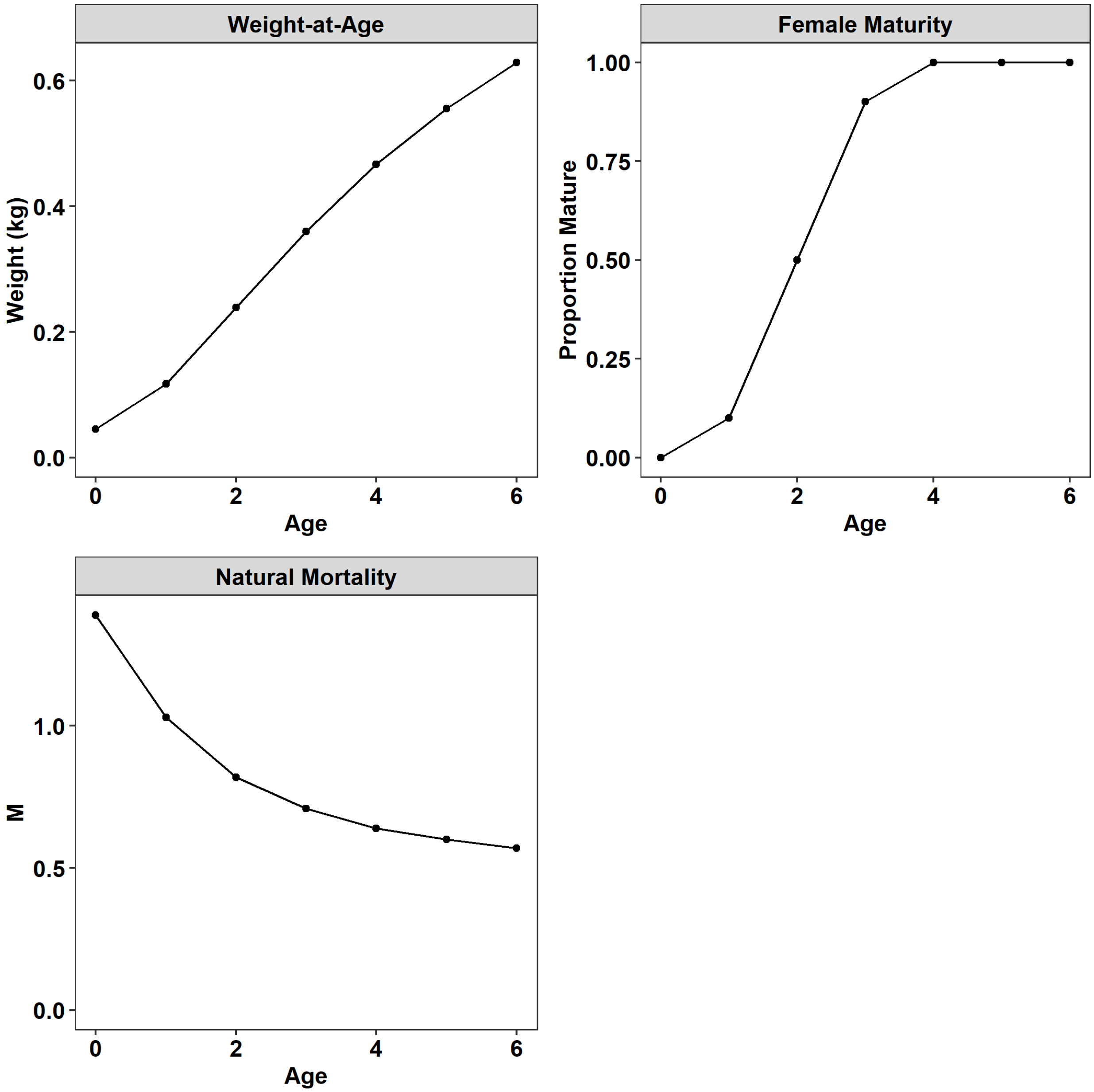


Figure 1. Time-invariant life history parameters for Atlantic menhaden.

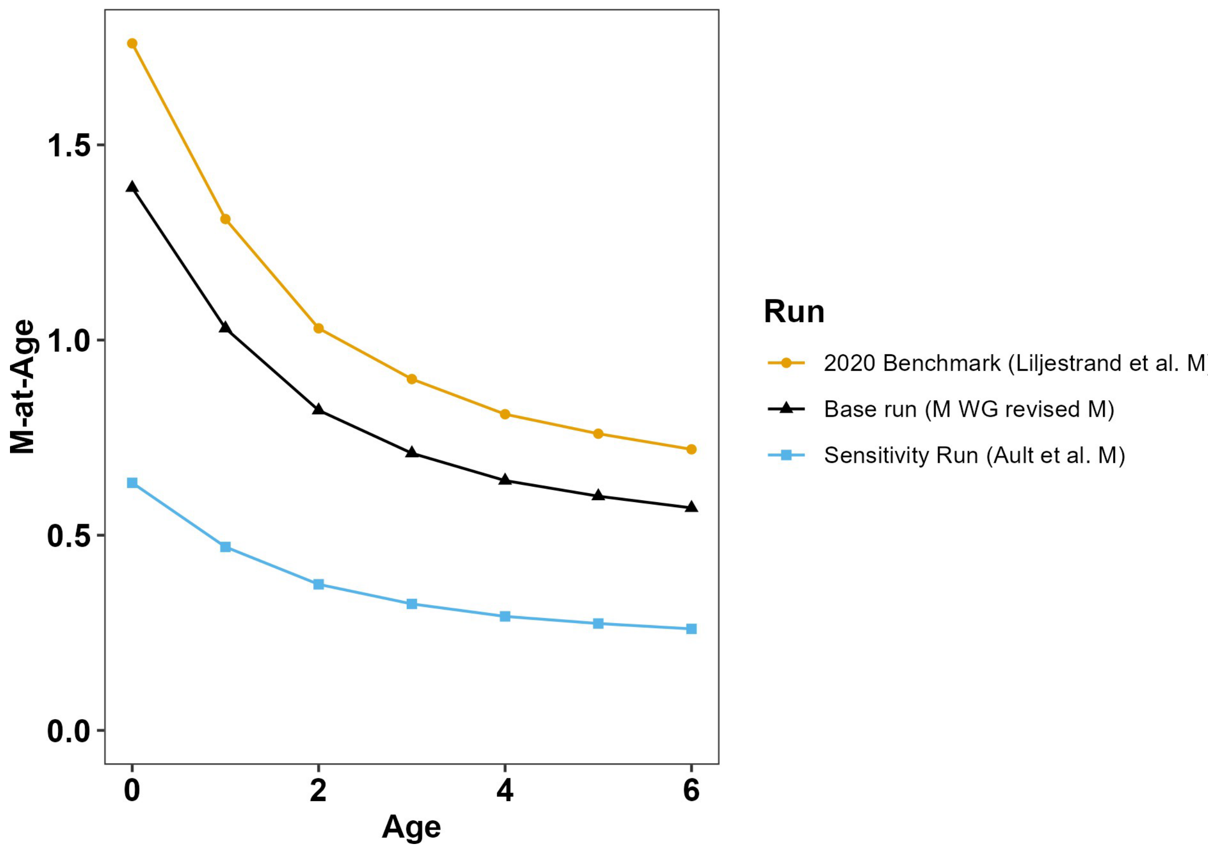


Figure 2. The base run estimates of *M*-at-age used in the 2025 Atlantic menhaden assessments plotted with the 2020 benchmark estimates and the 2025 sensitivity run.

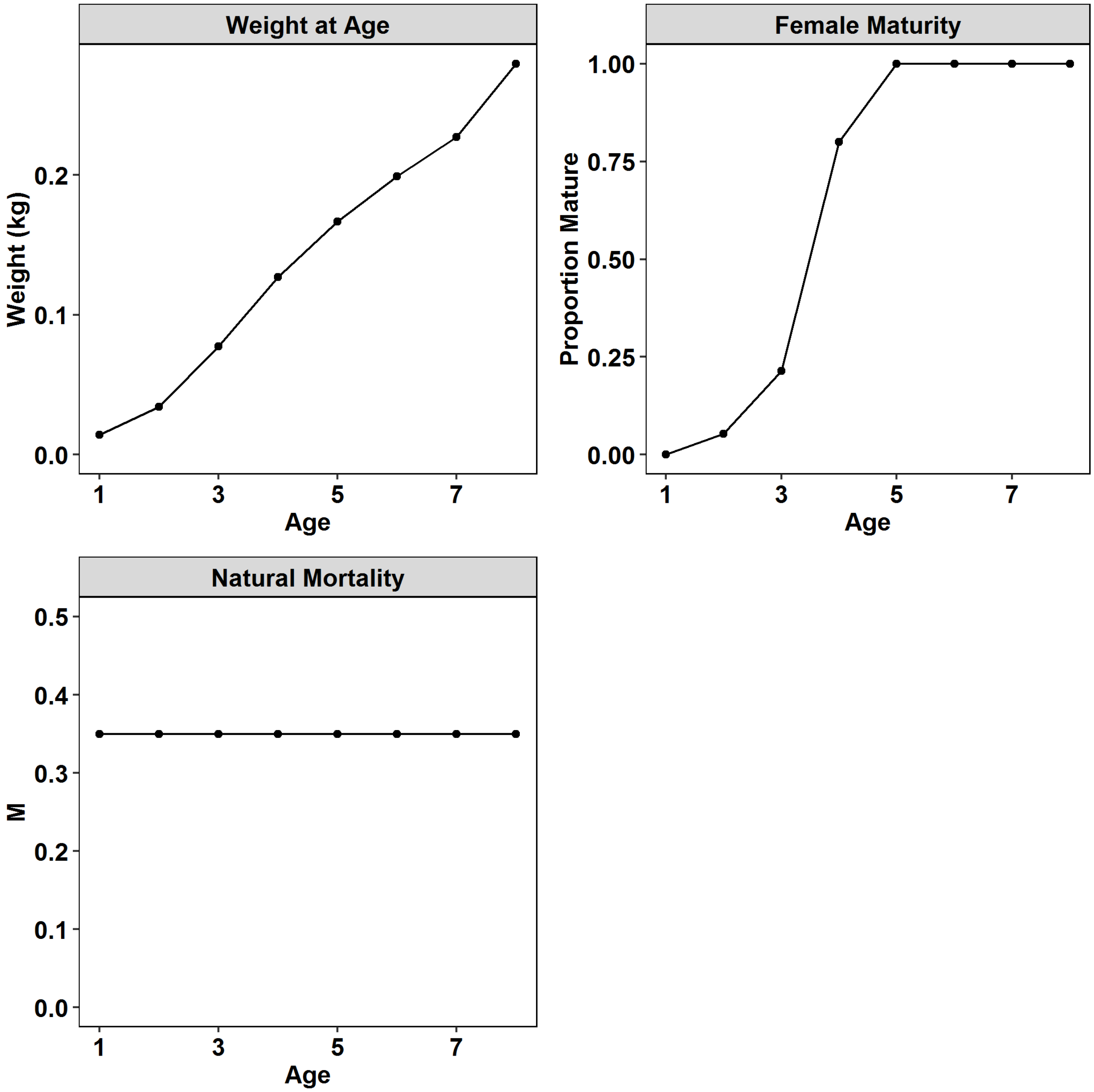


Figure 3. Time-invariant life history parameters for Atlantic herring.

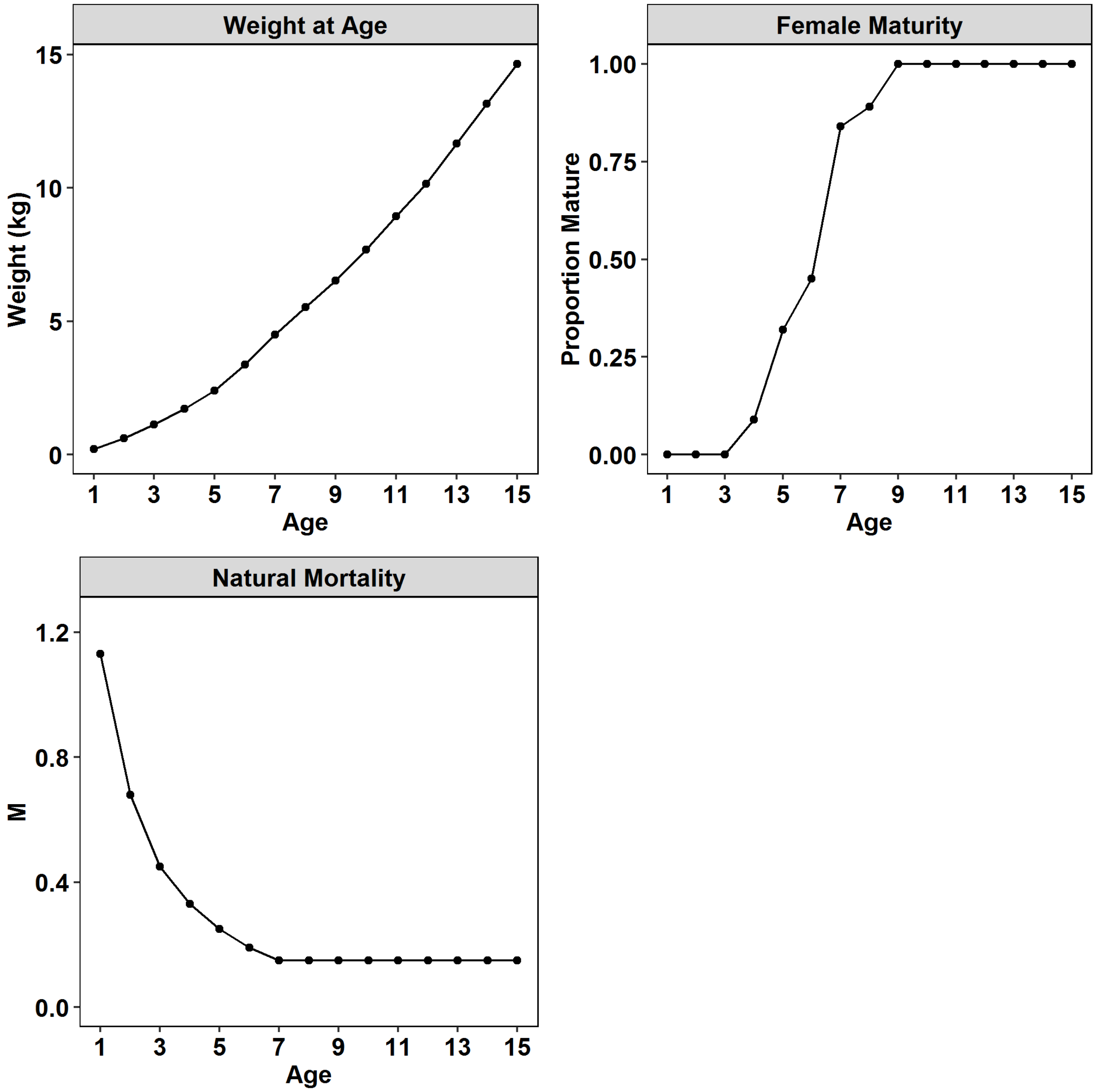


Figure 4. Time-invariant life history parameters for Atlantic striped bass.

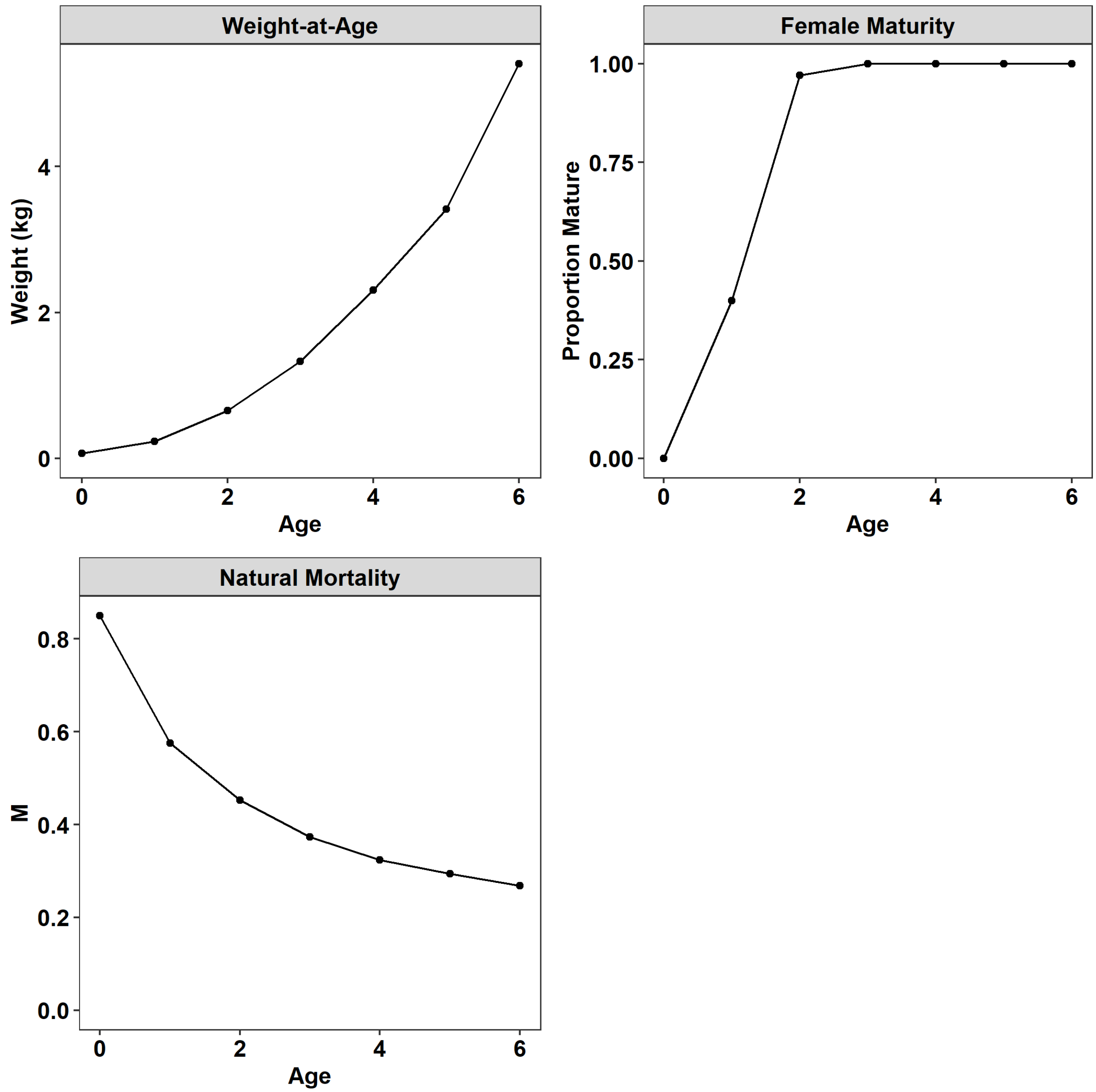


Figure 5. Time-invariant life history parameters for bluefish.

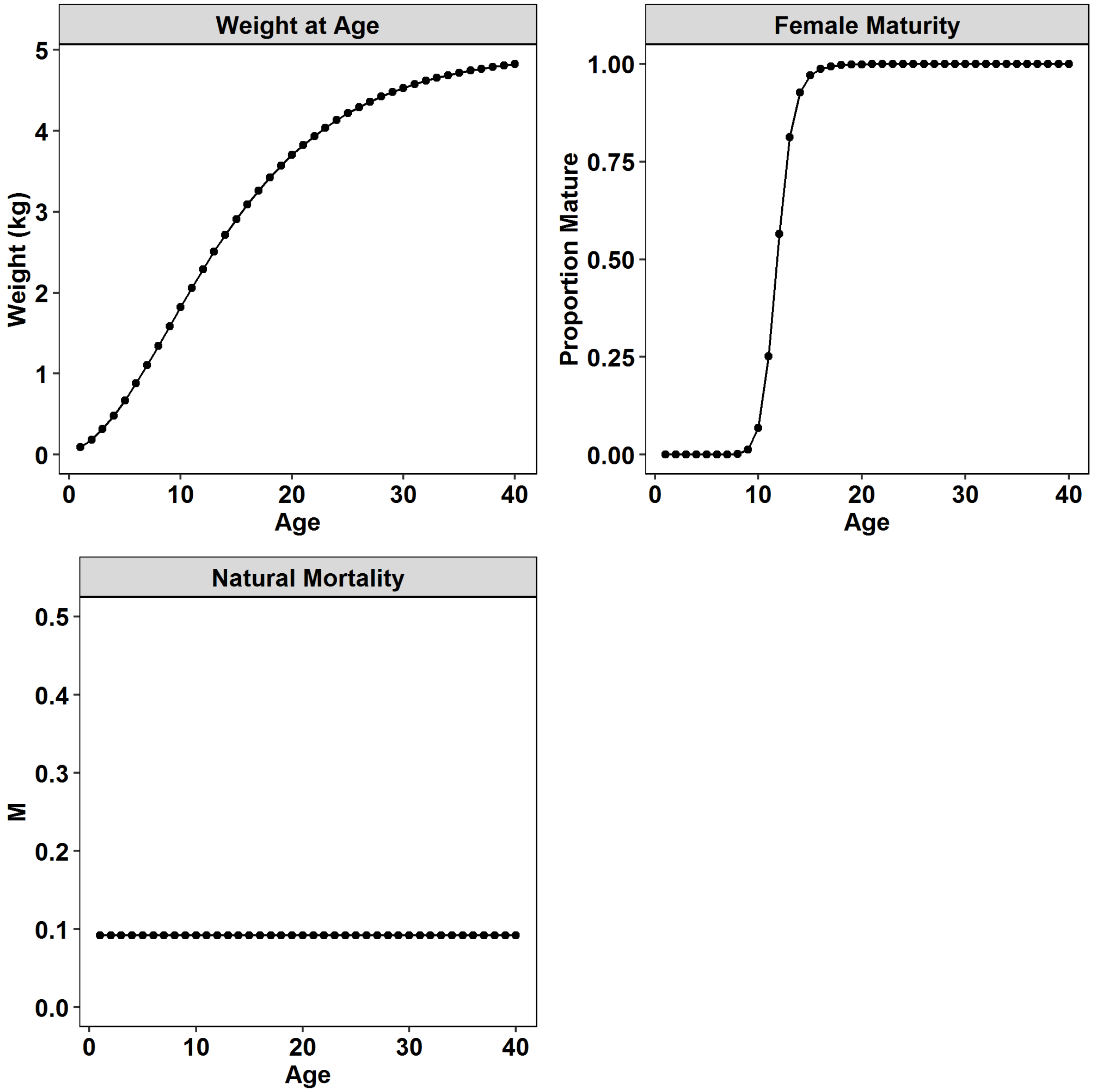


Figure 6. Time-invariant life history parameters for spiny dogfish.

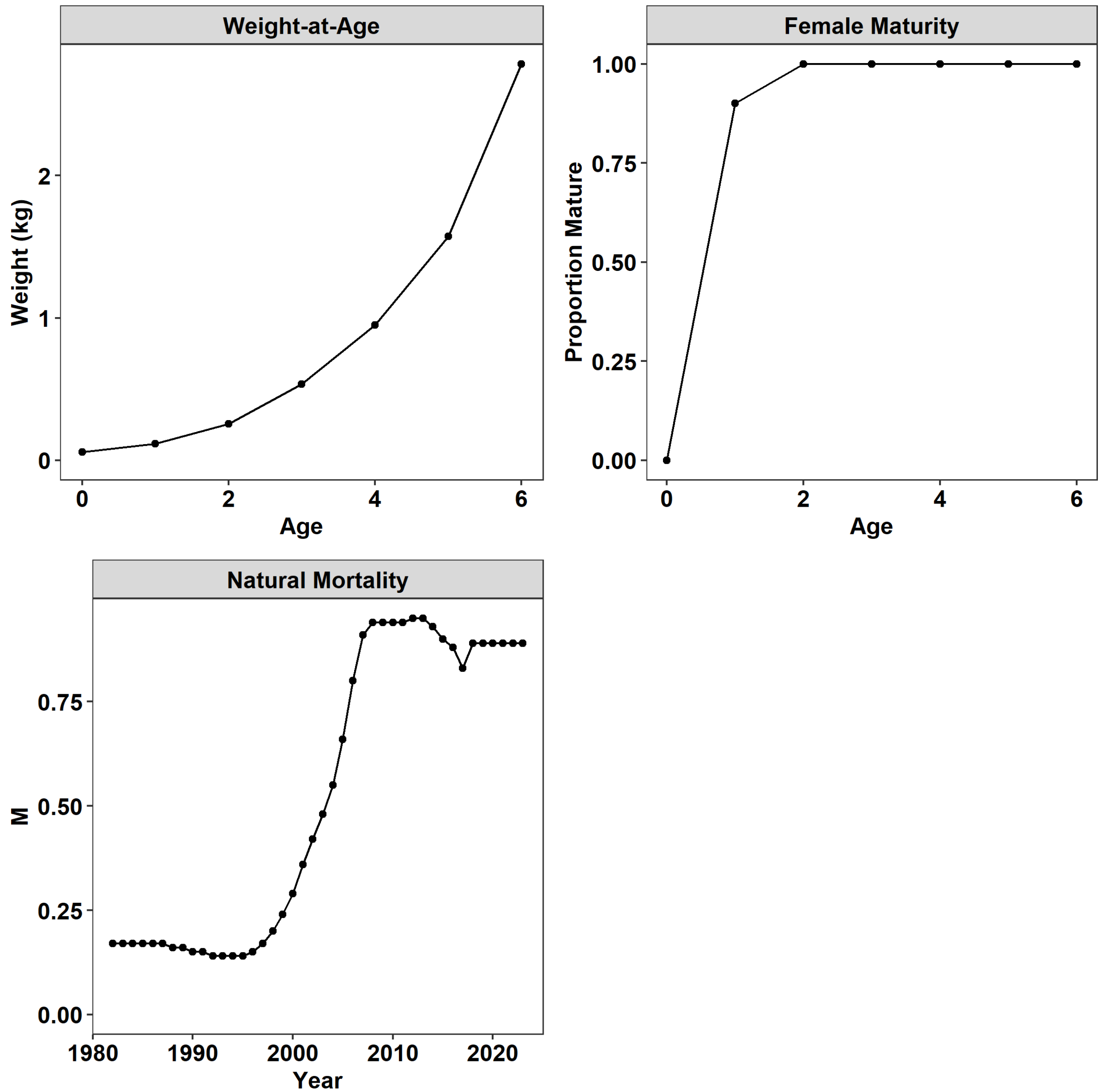


Figure 7. Time-invariant weight at age and maturity at age parameters, and time-varying natural mortality estimates for weakfish.

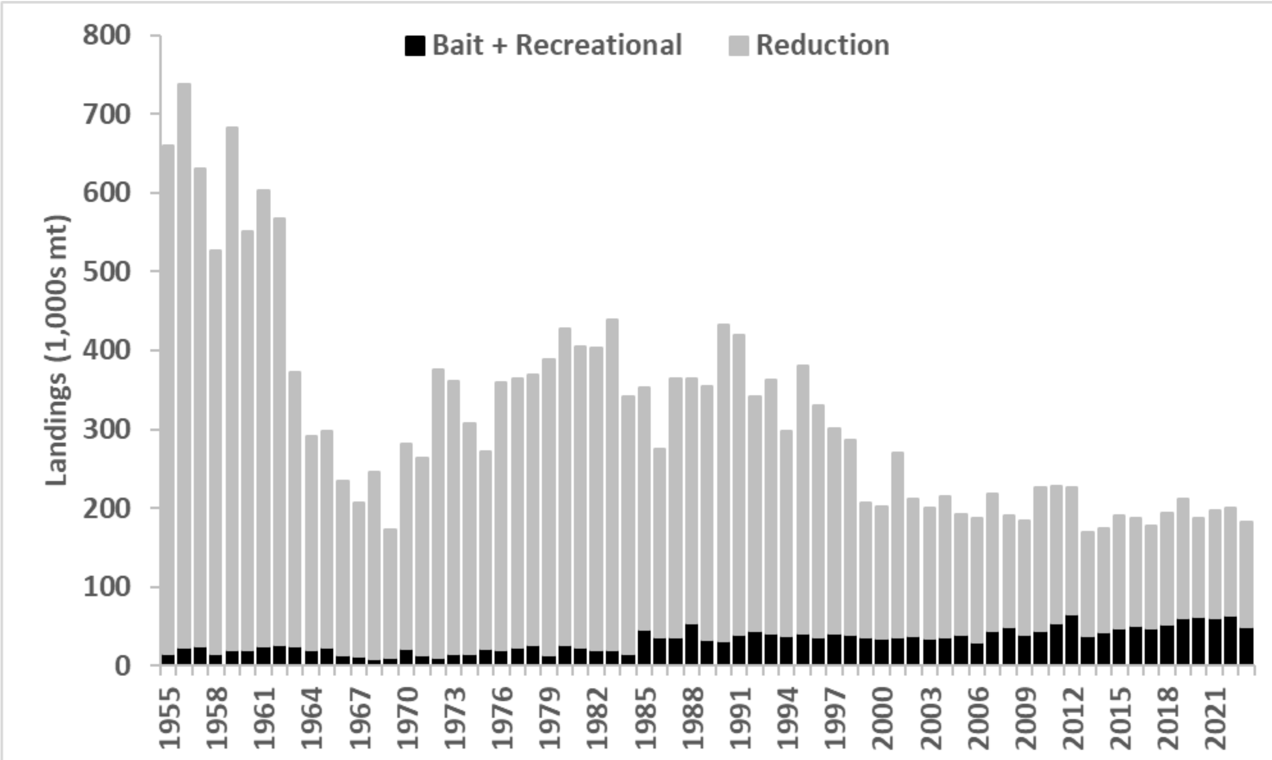


Figure 8. Total removals of Atlantic menhaden by sector.

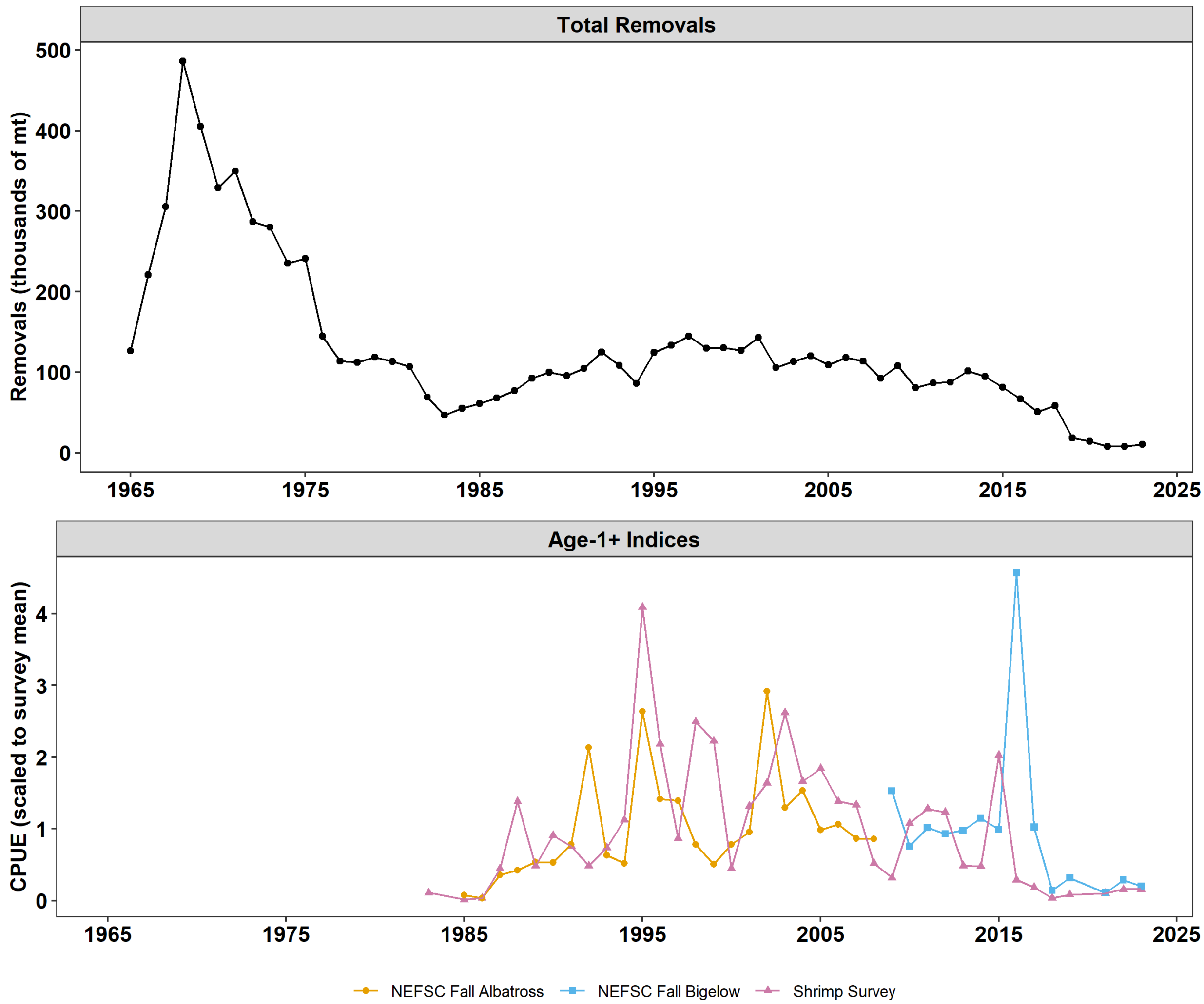


Figure 9. Total removals (top) and indices of abundance (bottom) for Atlantic herring.

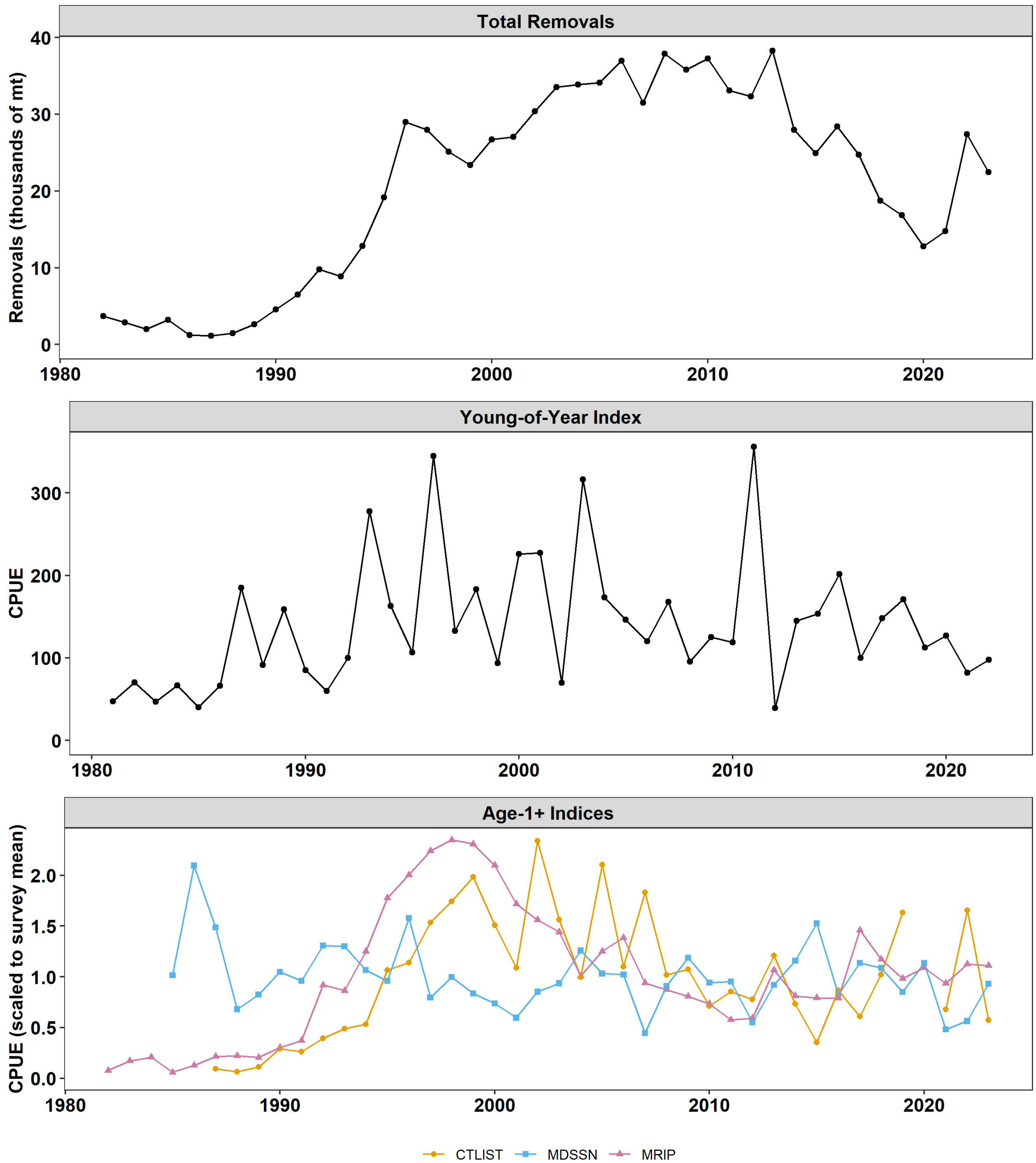


Figure 10. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for Atlantic striped bass.

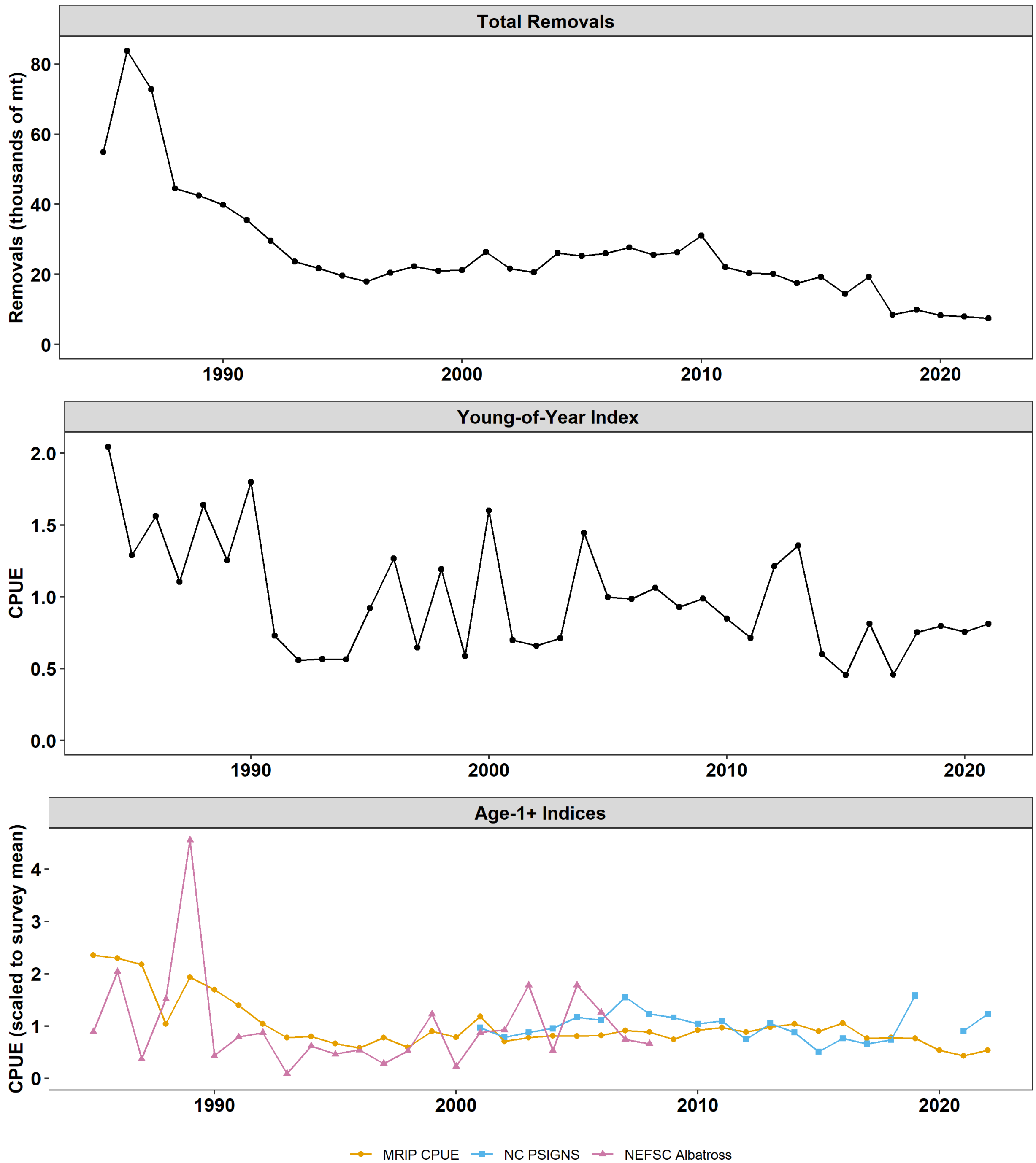


Figure 11. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for bluefish.

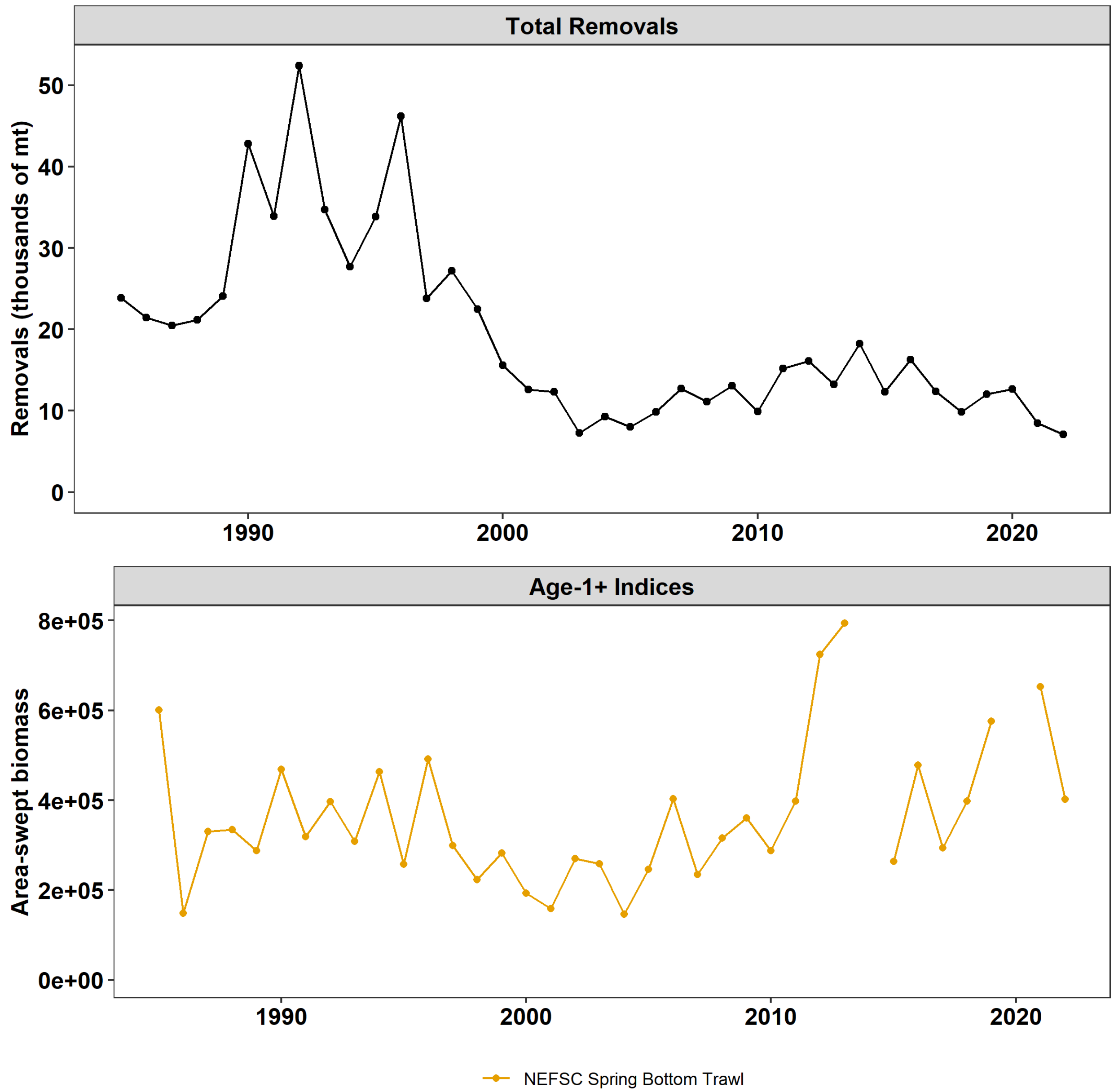


Figure 12. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for spiny dogfish.

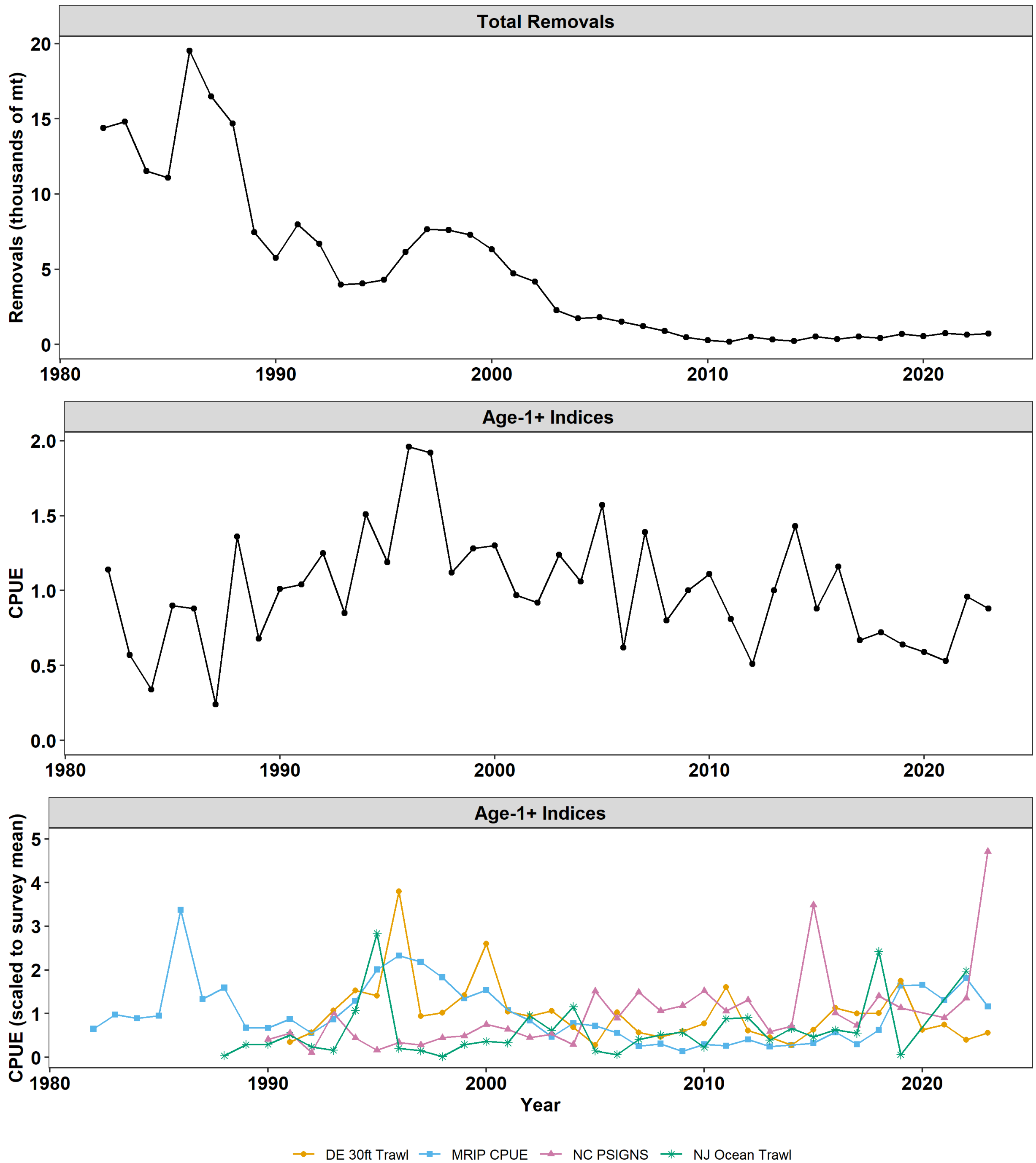


Figure 13. Total removals (top) and indices of recruitment (middle) and age-1+ abundance (bottom) for weakfish.

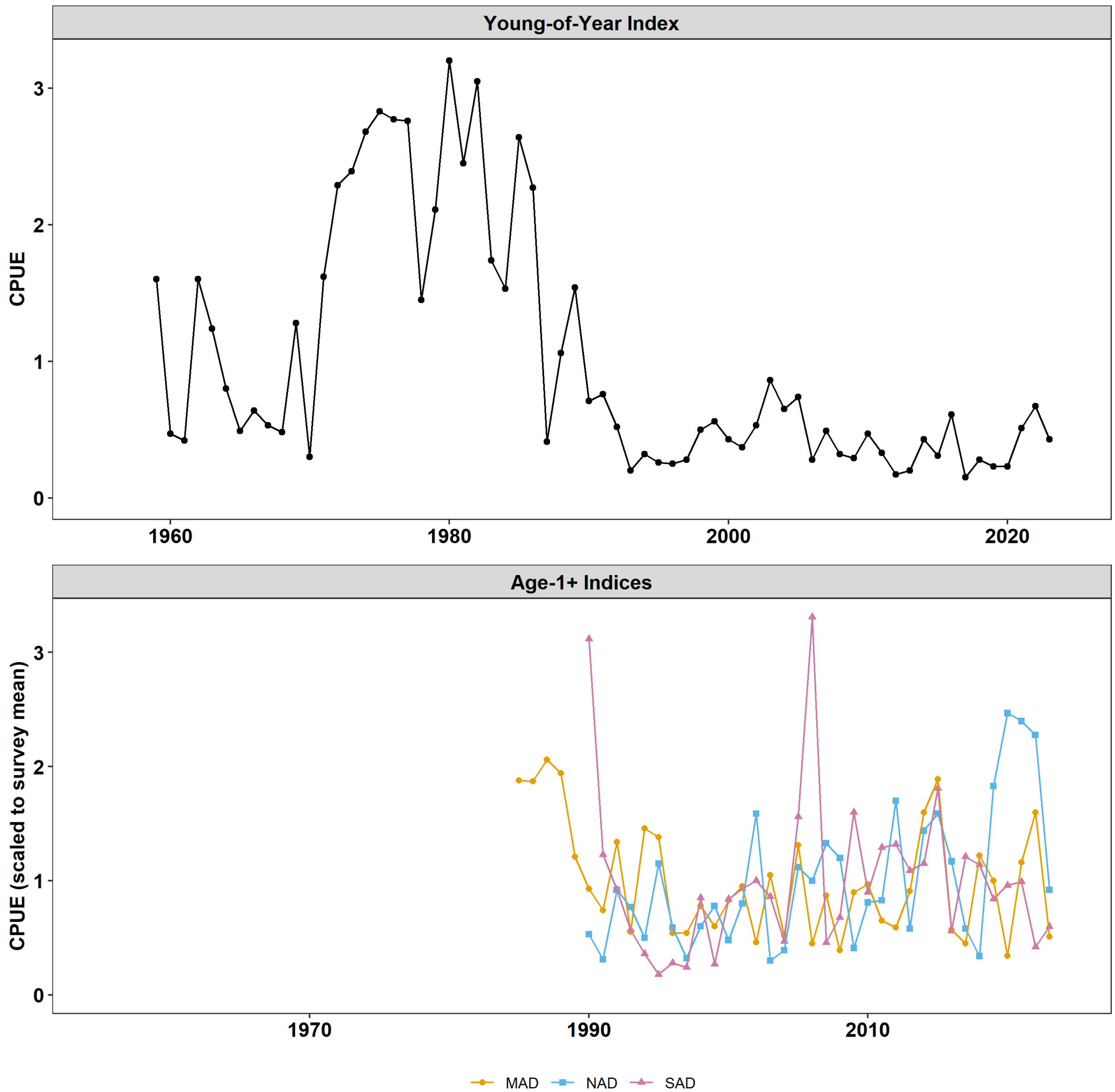


Figure 14. Fishery independent (top) and fishery dependent (bottom) indices of abundance for Atlantic menhaden.

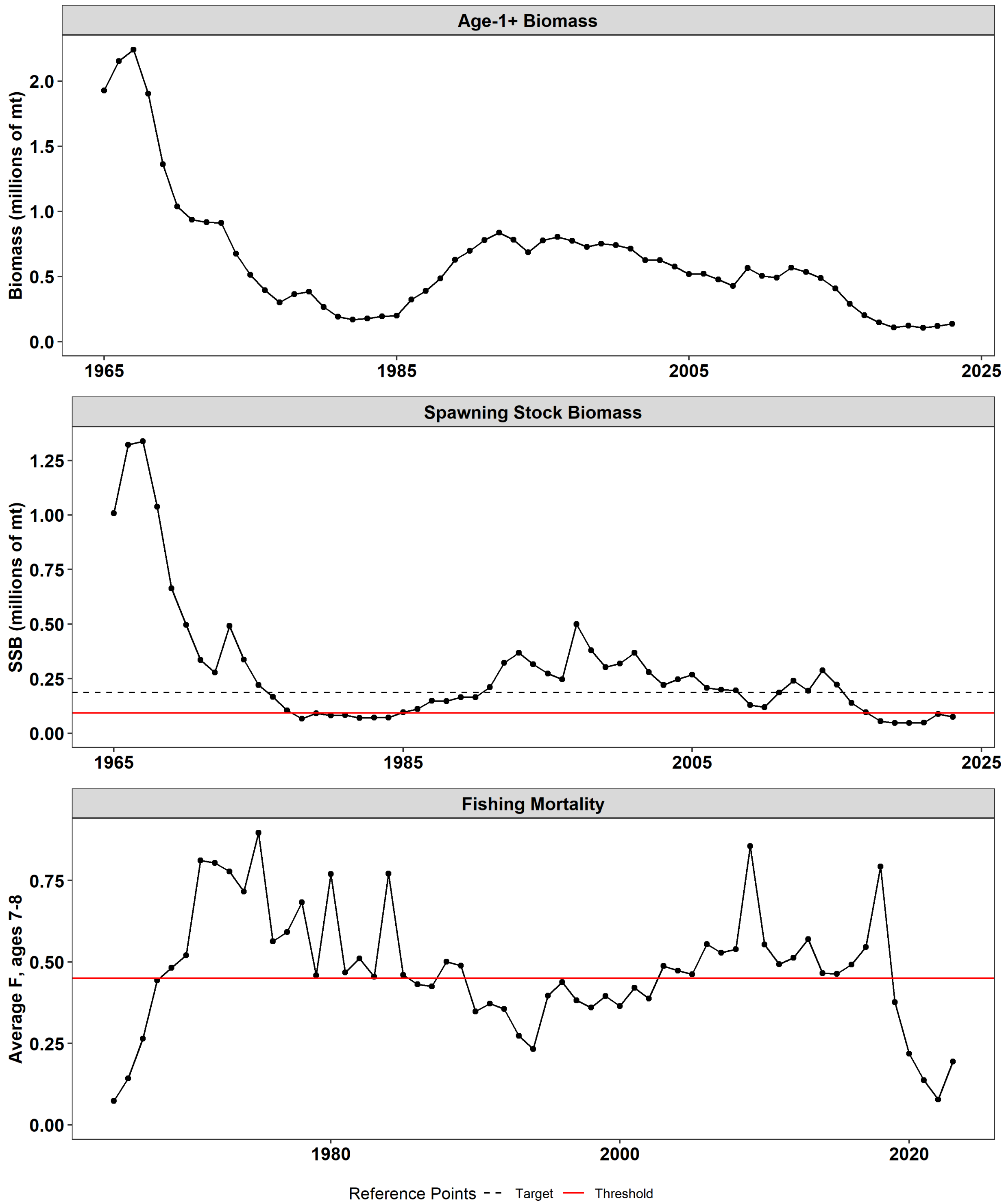


Figure 15. Age-1+ biomass, spawning stock biomass, and average *F* for Atlantic herring, plotted with their respective targets and thresholds, where defined.

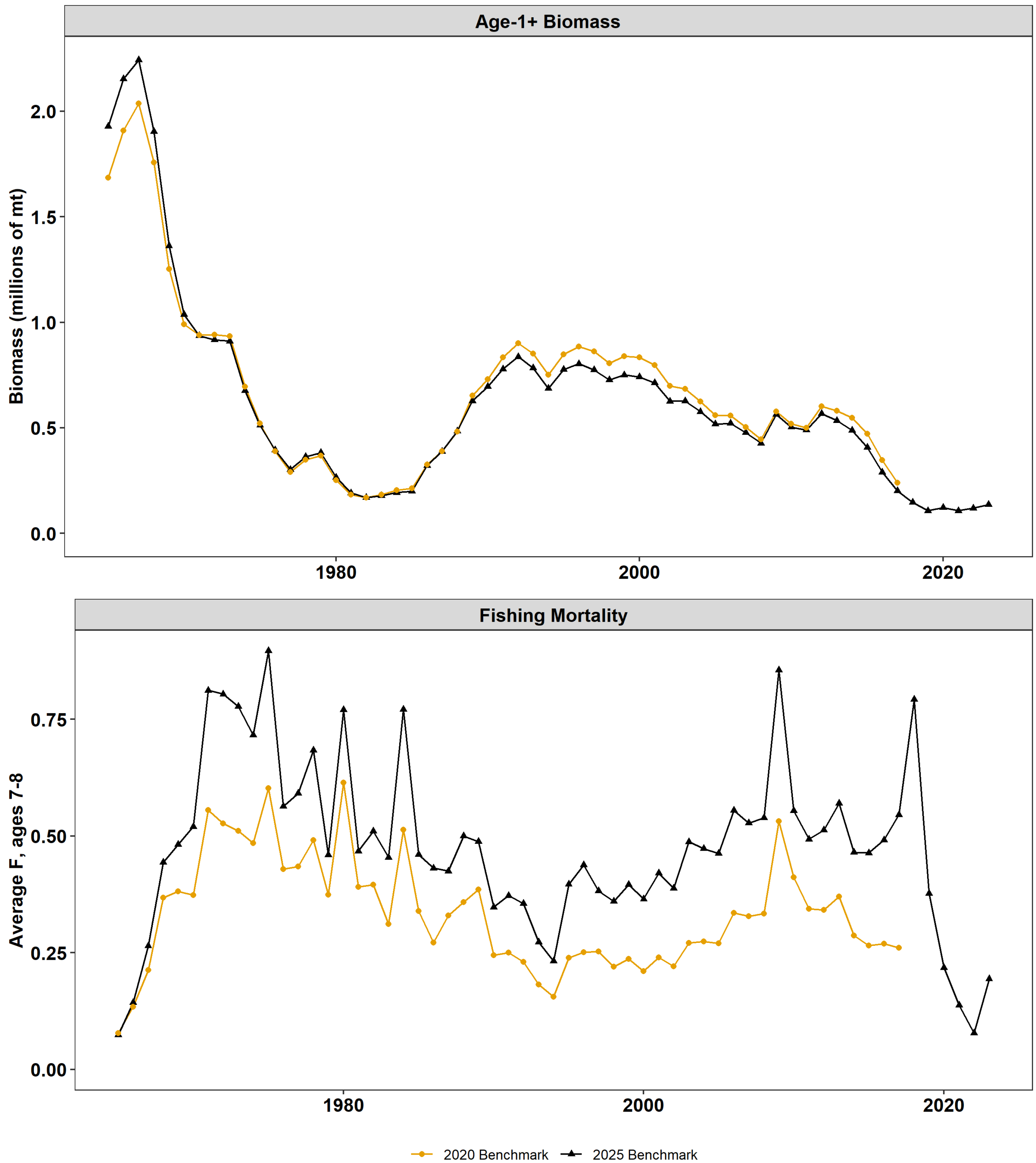


Figure 16. Comparison of age-1+ biomass and average *F* estimates for Atlantic herring used in the 2020 and 2025 ERP benchmark assessments.

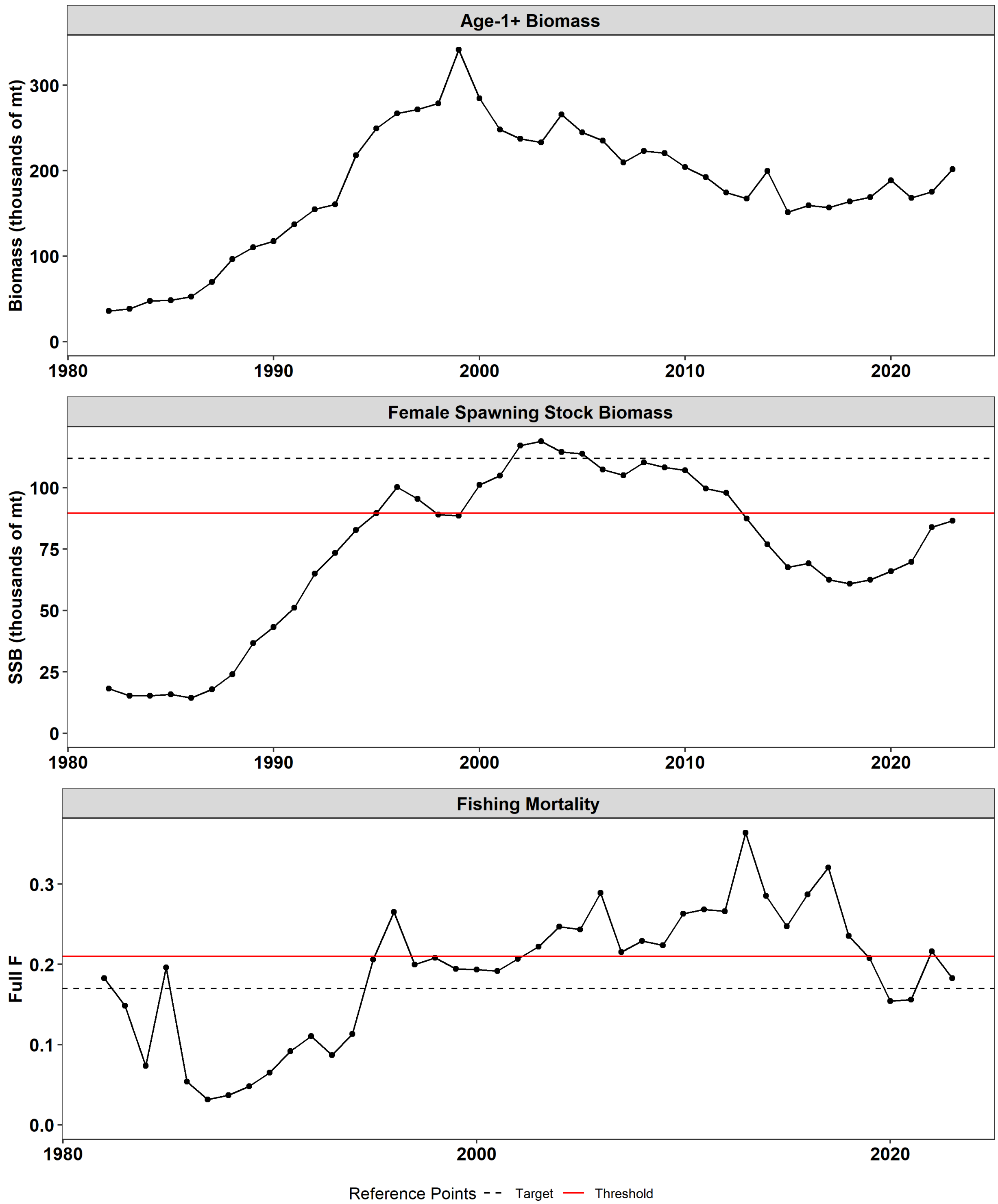


Figure 17. Age-1+ biomass, female spawning stock biomass, and full *F* for Atlantic striped bass, plotted with their respective targets and thresholds, where defined.

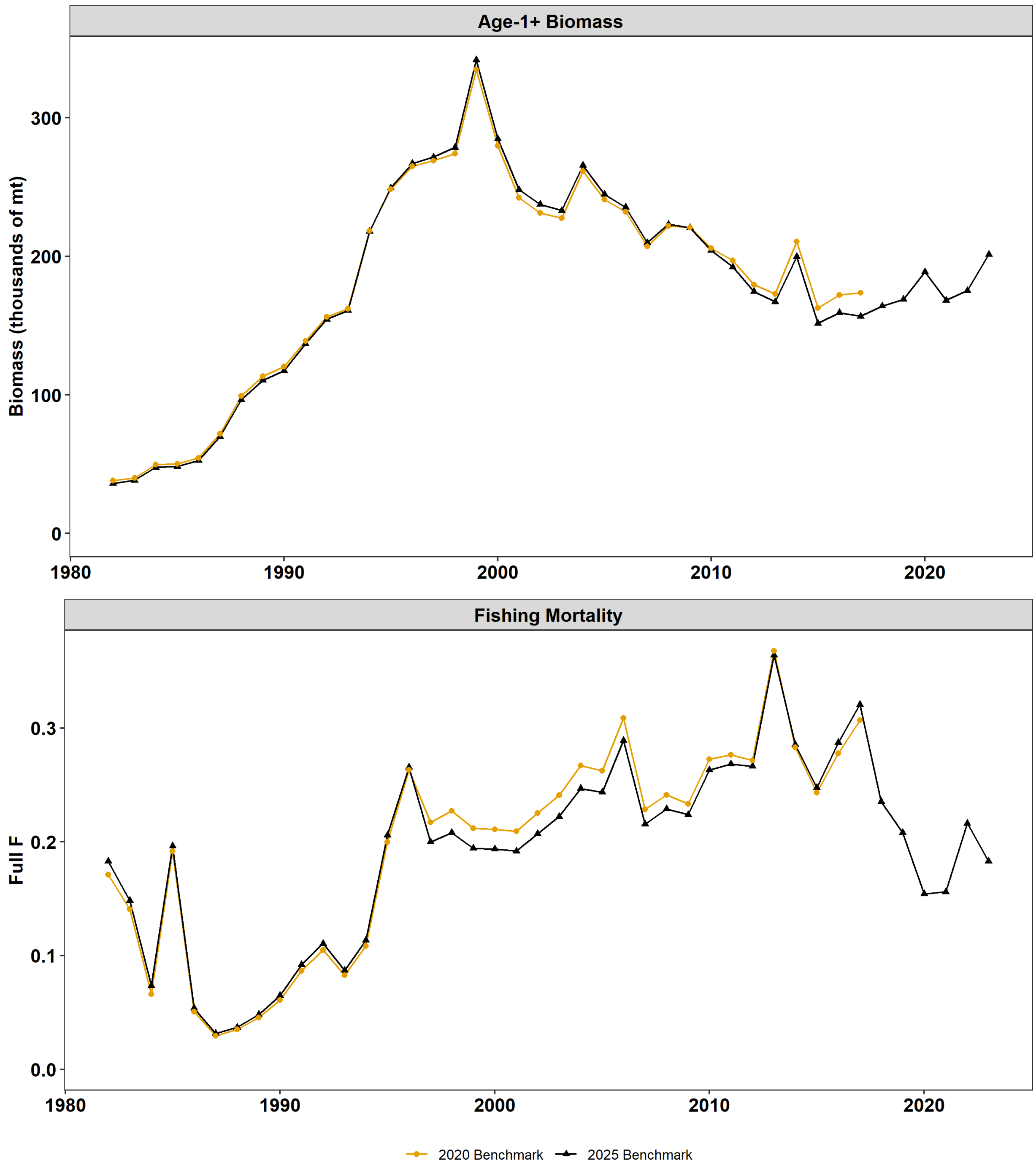


Figure 18. Comparison of age-1+ biomass and full *F* estimates for Atlantic striped bass used in the 2020 and 2025 ERP benchmark assessments.

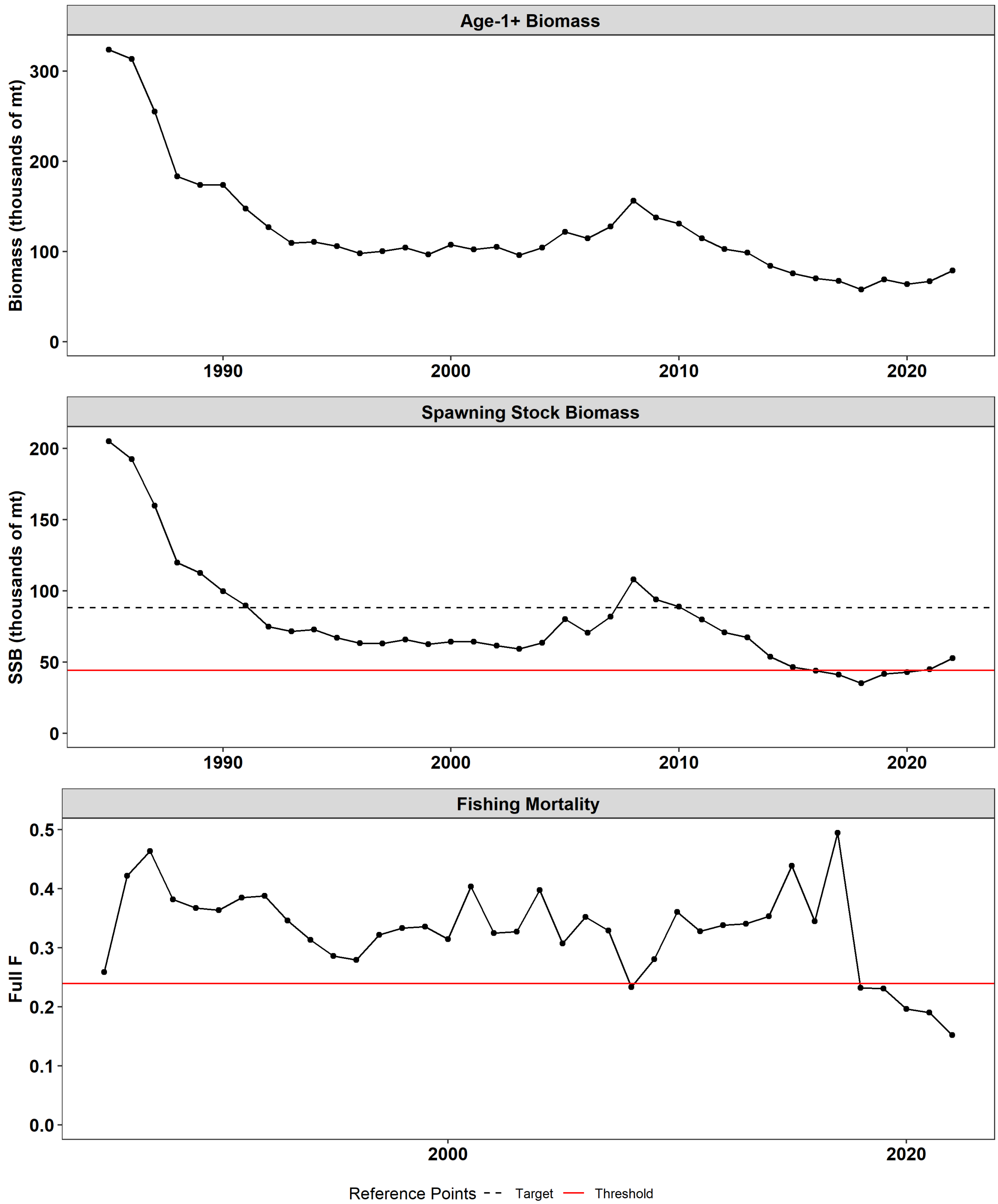


Figure 19. Age-1+ biomass, spawning stock biomass, and full *F* for bluefish, plotted with their respective targets and thresholds, where defined.

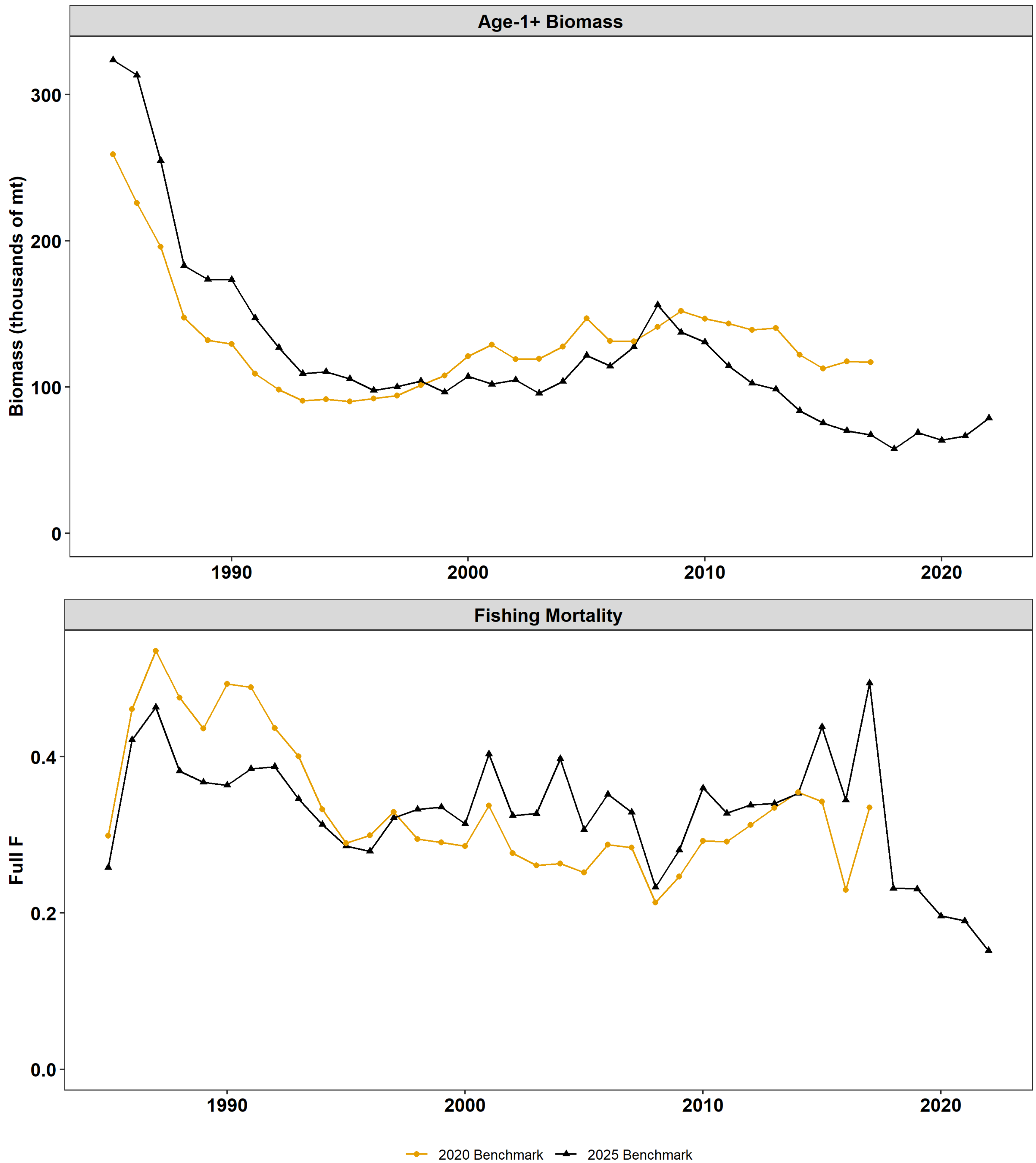


Figure 20. Comparison of age-1+ biomass and full *F* estimates for bluefish used in the 2020 and 2025 ERP benchmark assessments.

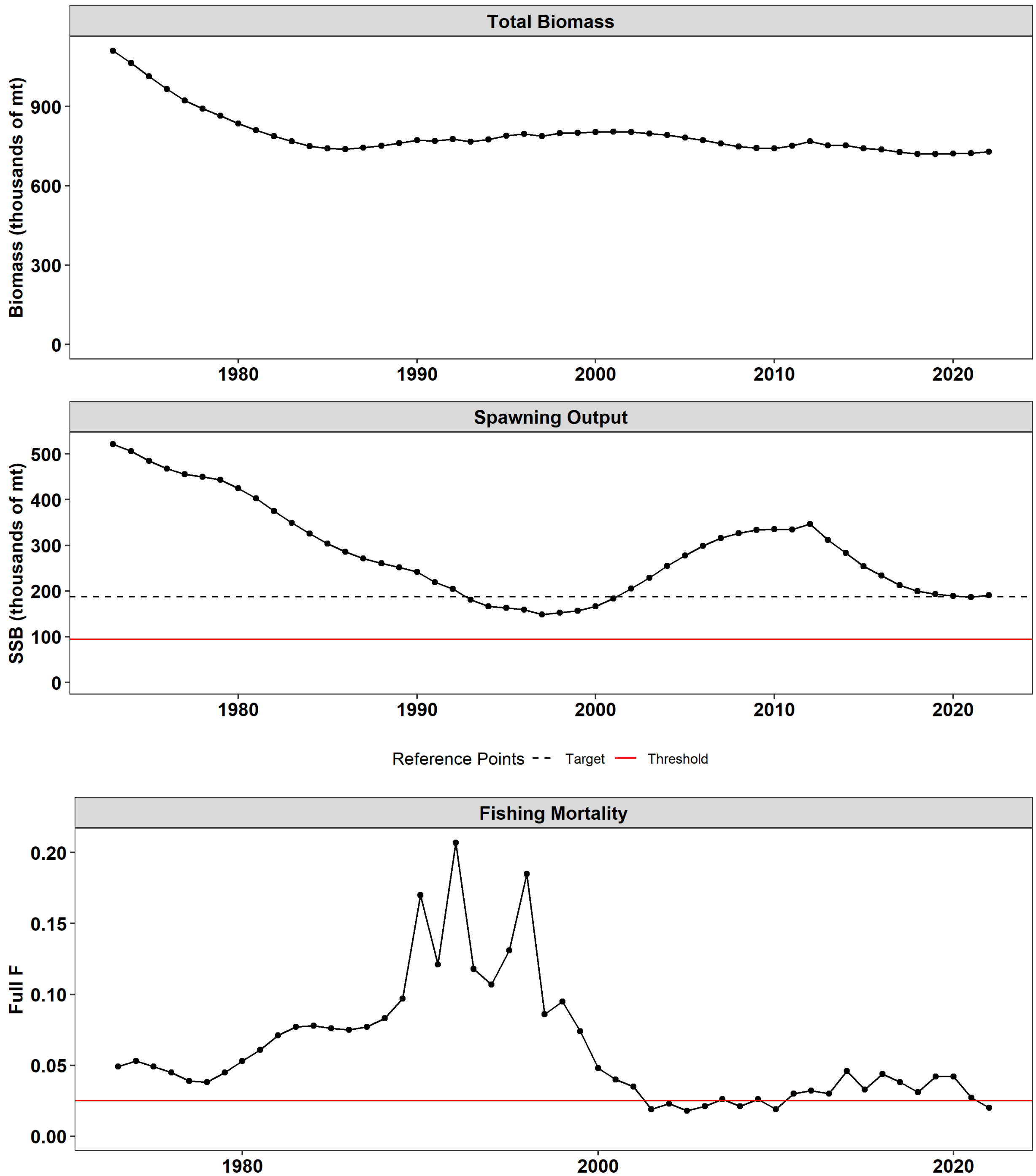


Figure 21. Total biomass, spawning output, and full *F* for spiny dogfish, plotted with their respective targets and thresholds, where defined.

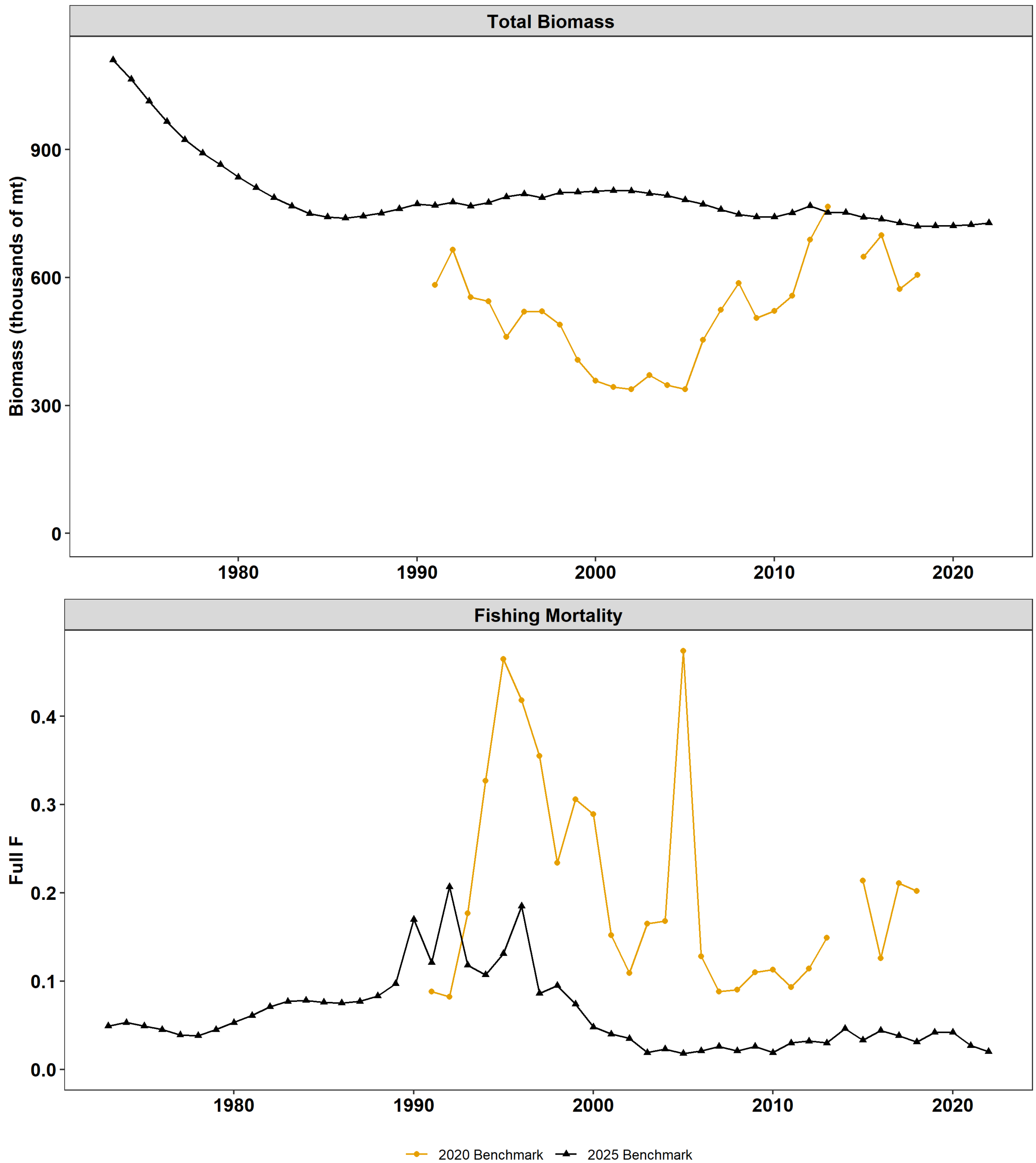


Figure 22. Comparison of total biomass and full *F* estimates for spiny dogfish used in the 2020 and 2025 ERP benchmark assessments.

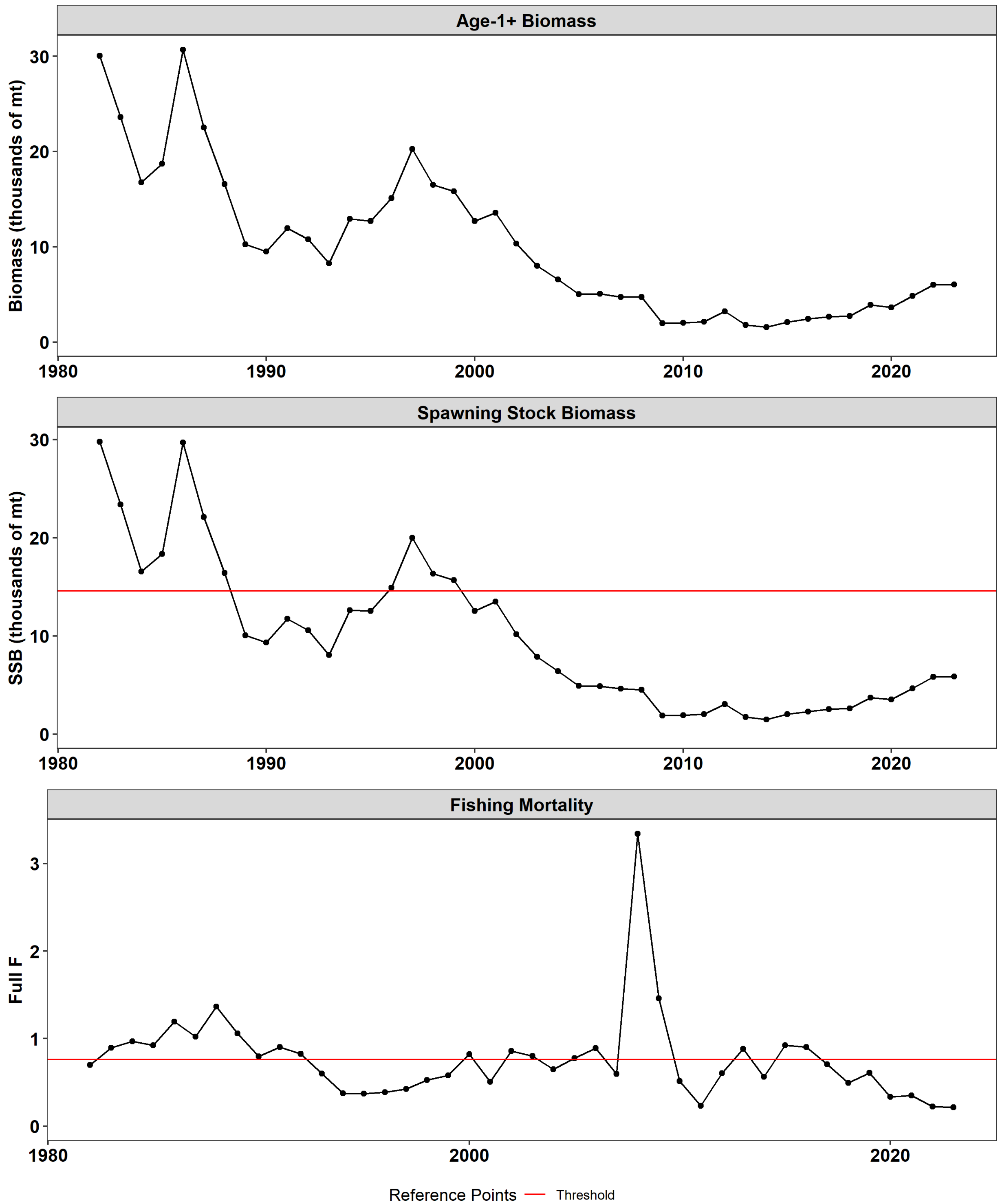


Figure 23. Age-1+ biomass, spawning stock biomass, and full *F* for weakfish, plotted with their respective thresholds, where defined. Estimates are from a preliminary assessment update with data through 2023 and may not match values used for management.

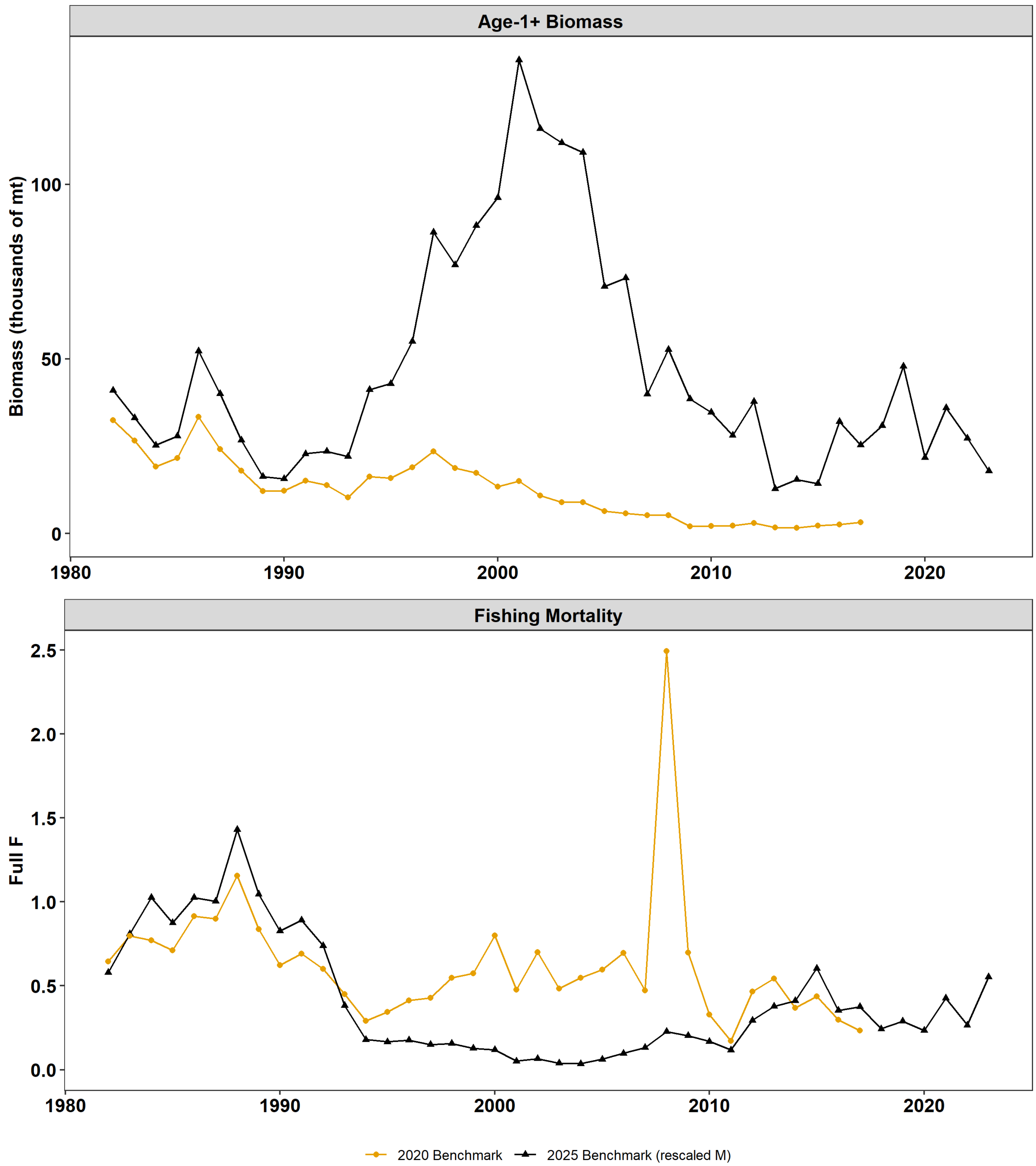
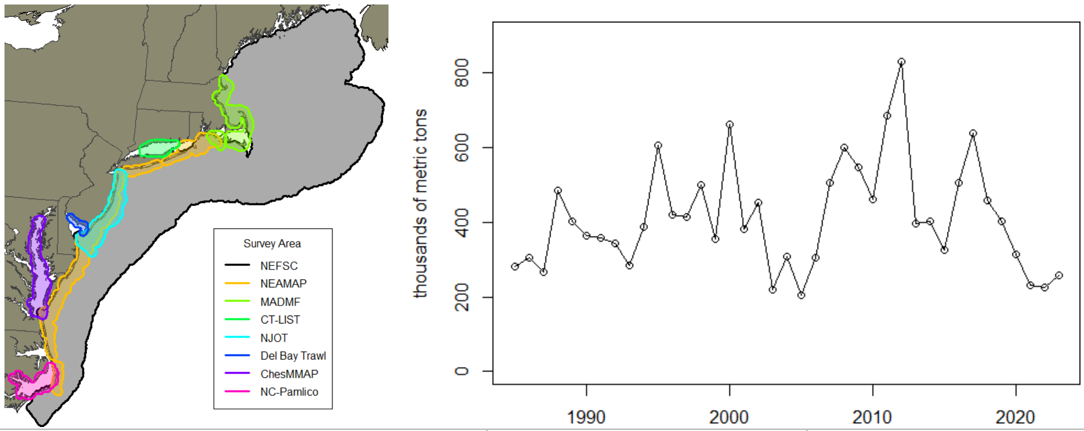


Figure 24. Comparison of age-1+ biomass and full *F* estimates for weakfish used in the 2020 and 2025 ERP benchmark assessments. The estimates of biomass and *F* used in the 2025 assessment are from the ASAP run with the estimated *M* rescaled to an empirical estimate of *M* from a tagging model.



Survey	n Tows	% Positive
NEFSC	22,833	10%
NEAMAP	4,724	51%
MADMF	7,358	5%
CT-LISTS	4,122	28%
NJOT	5,813	39%
DBTrawl	1,863	95%
ChesMMAP	7,440	22%
NC-Pamlico	4,966	63%
total	59,119	25%

Figure 25. Survey coverage trajectory of average annual anchovy biomass within the NWACS model domain.

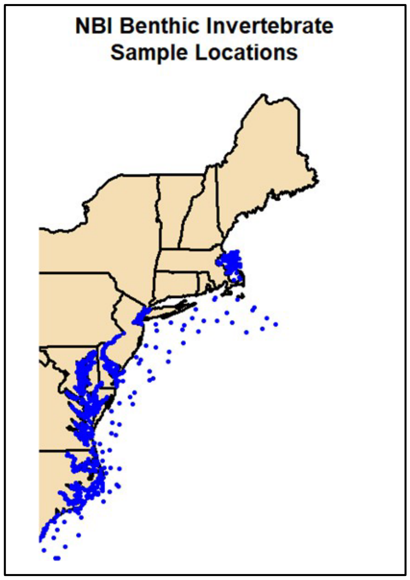


Figure 26. Location of benthic grab samples in the NBI repository used to estimate benthic invertebrate biomass.

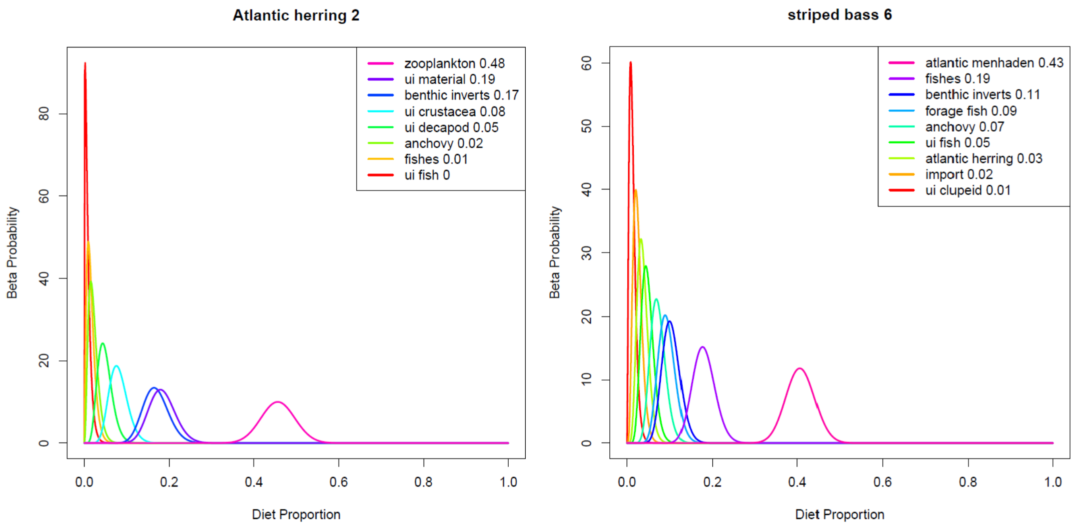


Figure 27. Example of estimated posterior distribution functions from the Dirichlet multinomial for adult Atlantic herring and striped bass age 6+.

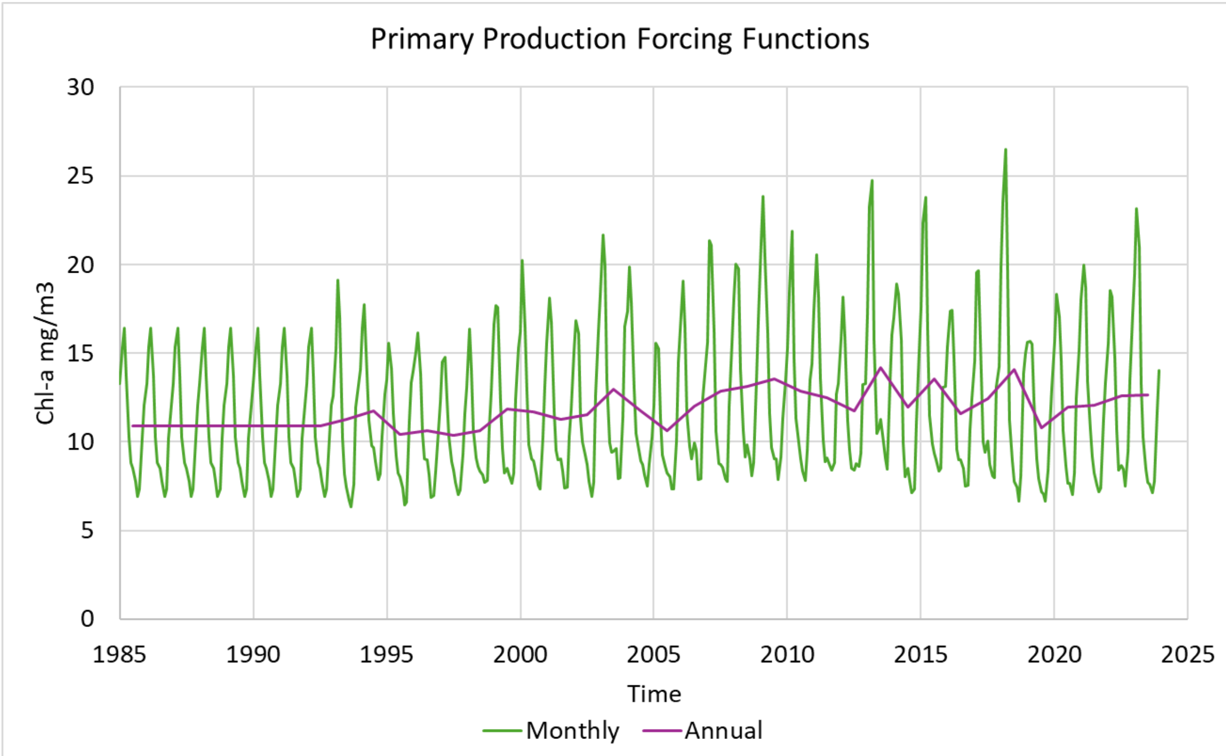


Figure 28. Chlorophyll-a timeseries from the GLORYS dataset used to force primary production in NWACS-MICE.

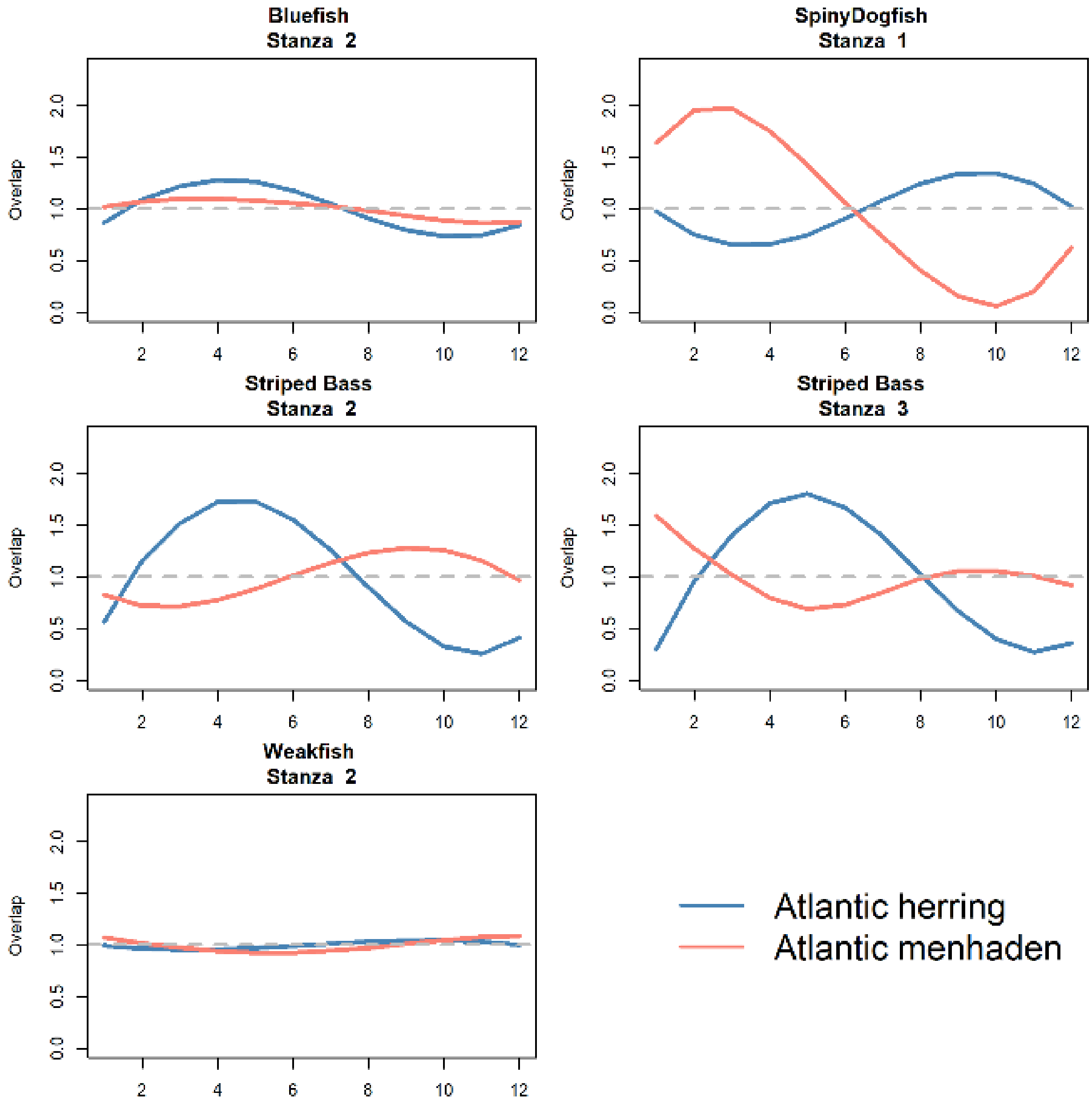


Figure 29. Input monthly vulnerability forcing functions intended to represent the seasonal overlap of key predator-prey interactions.

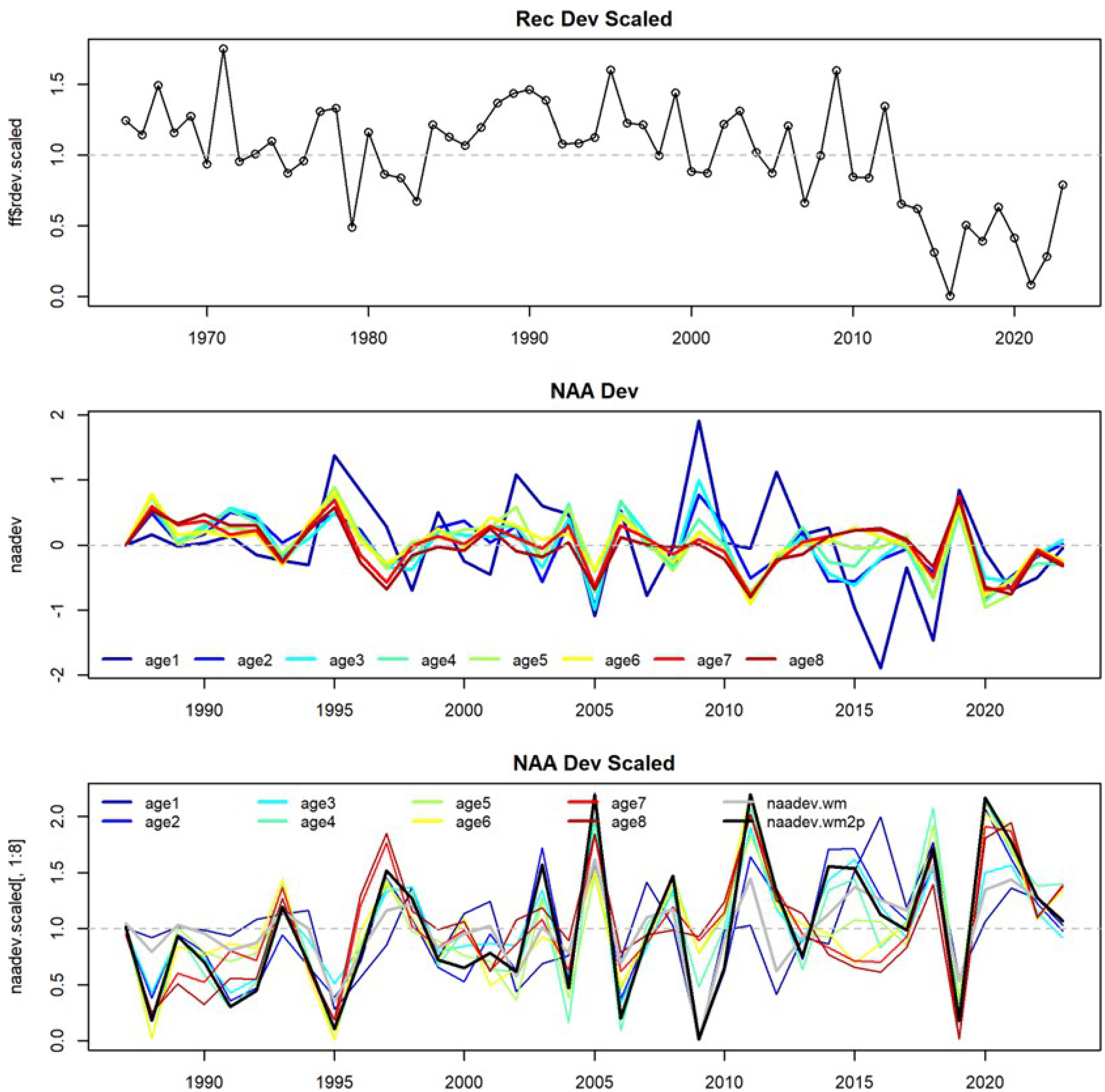


Figure 30. Atlantic herring forcing functions.

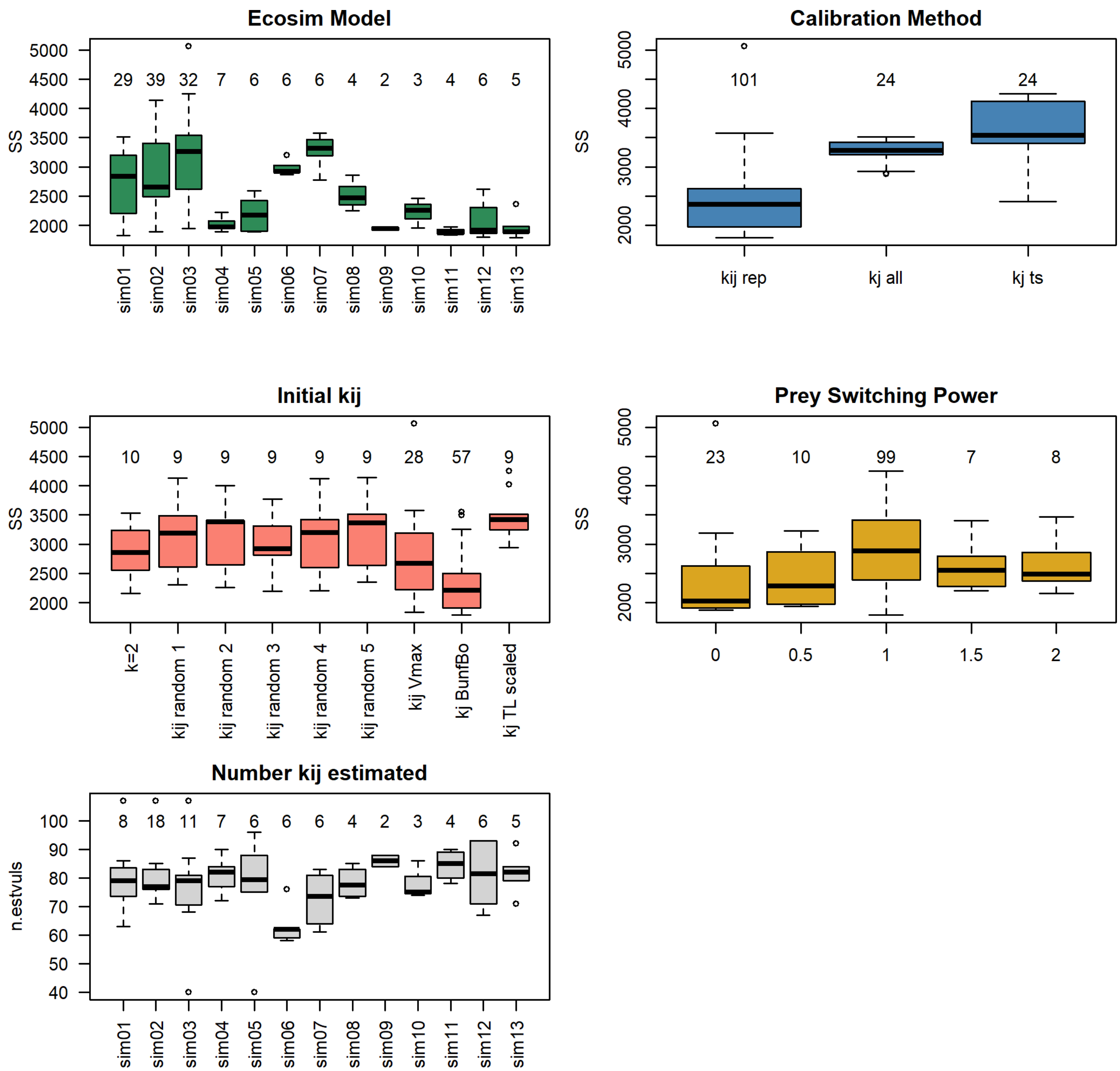


Figure 31. Summary sum-of-squared residuals (SS) from 129 calibrated NWACS-MICE Ecosim models. Number of runs is indicated above each boxplot.

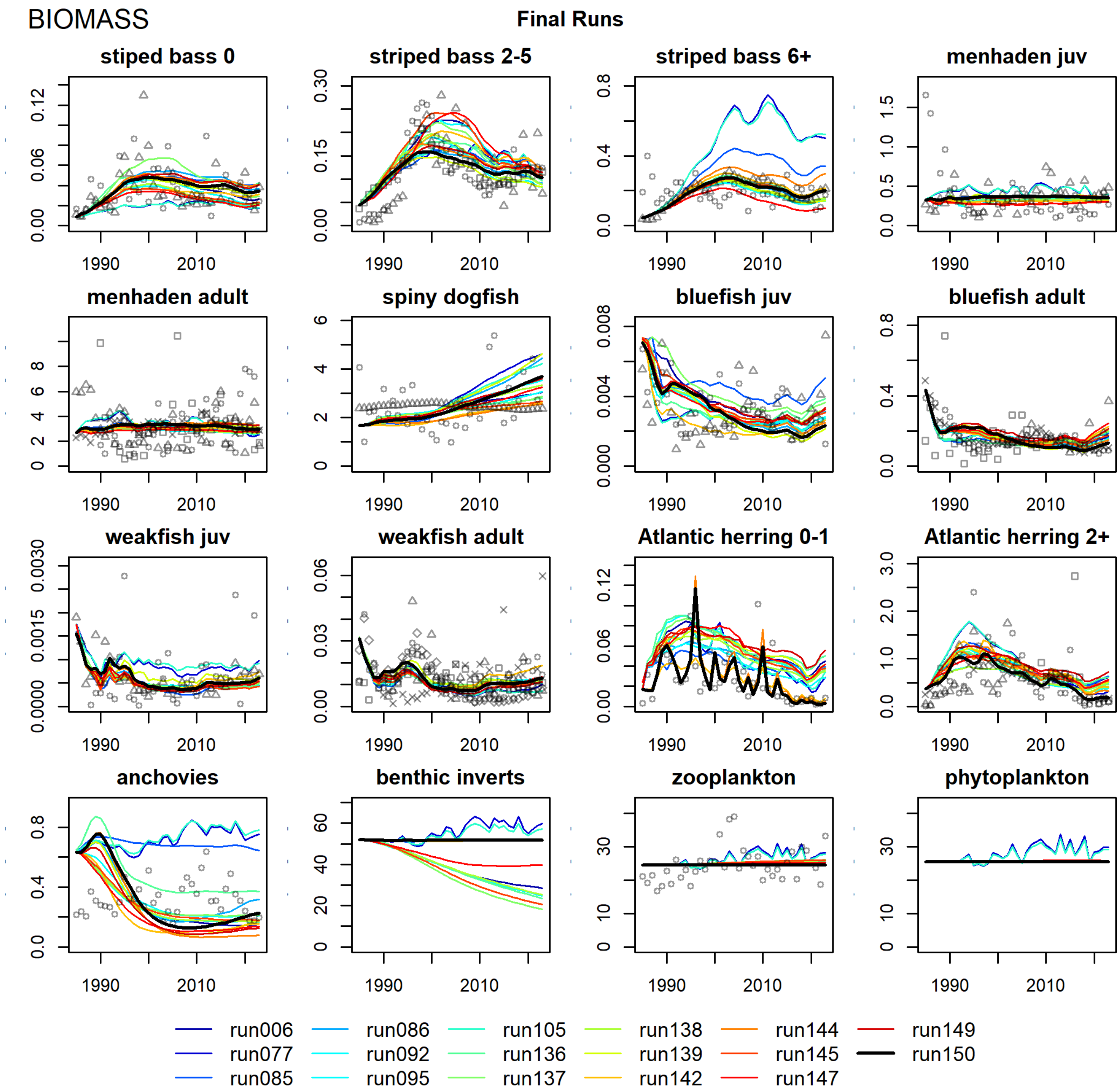


Figure 32. Fits to biomass from the top 10% of model runs for the NWACS-MICE model. The points represent observed abundance time series data, each with a different symbol.

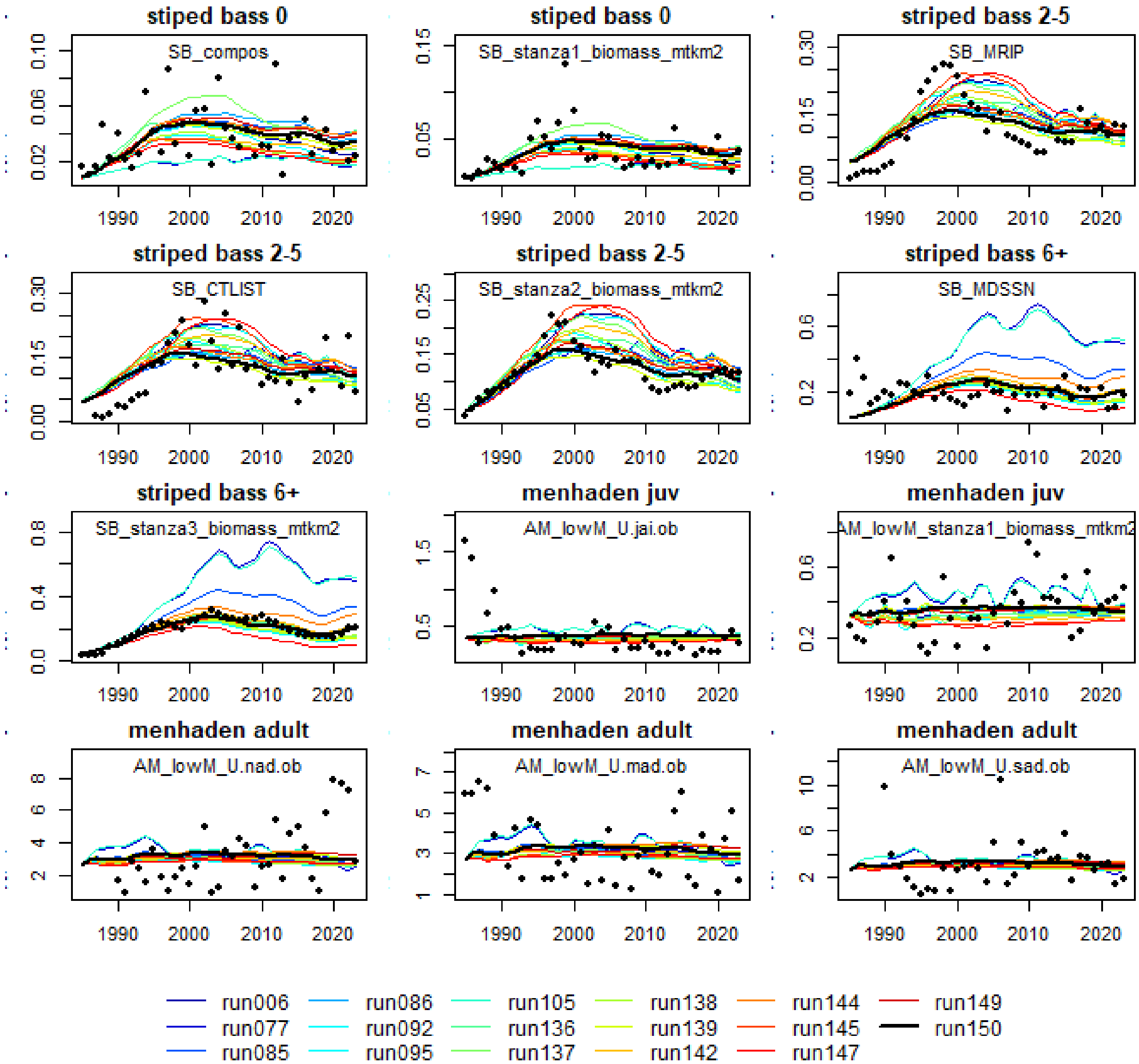


Figure 33. Fits to individual indices and time series of biomass or relative abundance for the NWACS-MICE model.

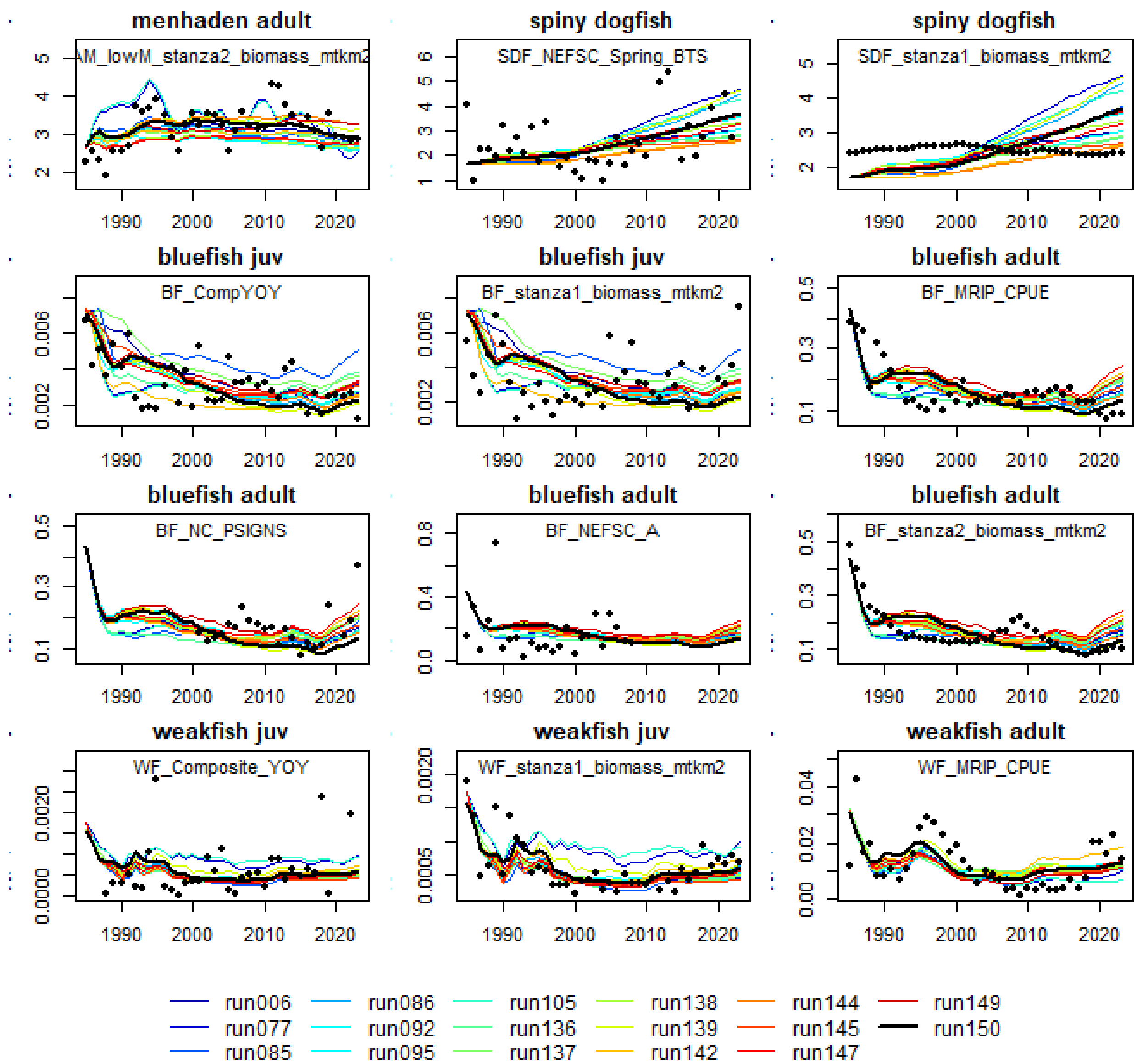


Figure 34 cont. Fits to individual indices and time series of biomass or relative abundance for the NWACS-MICE model.

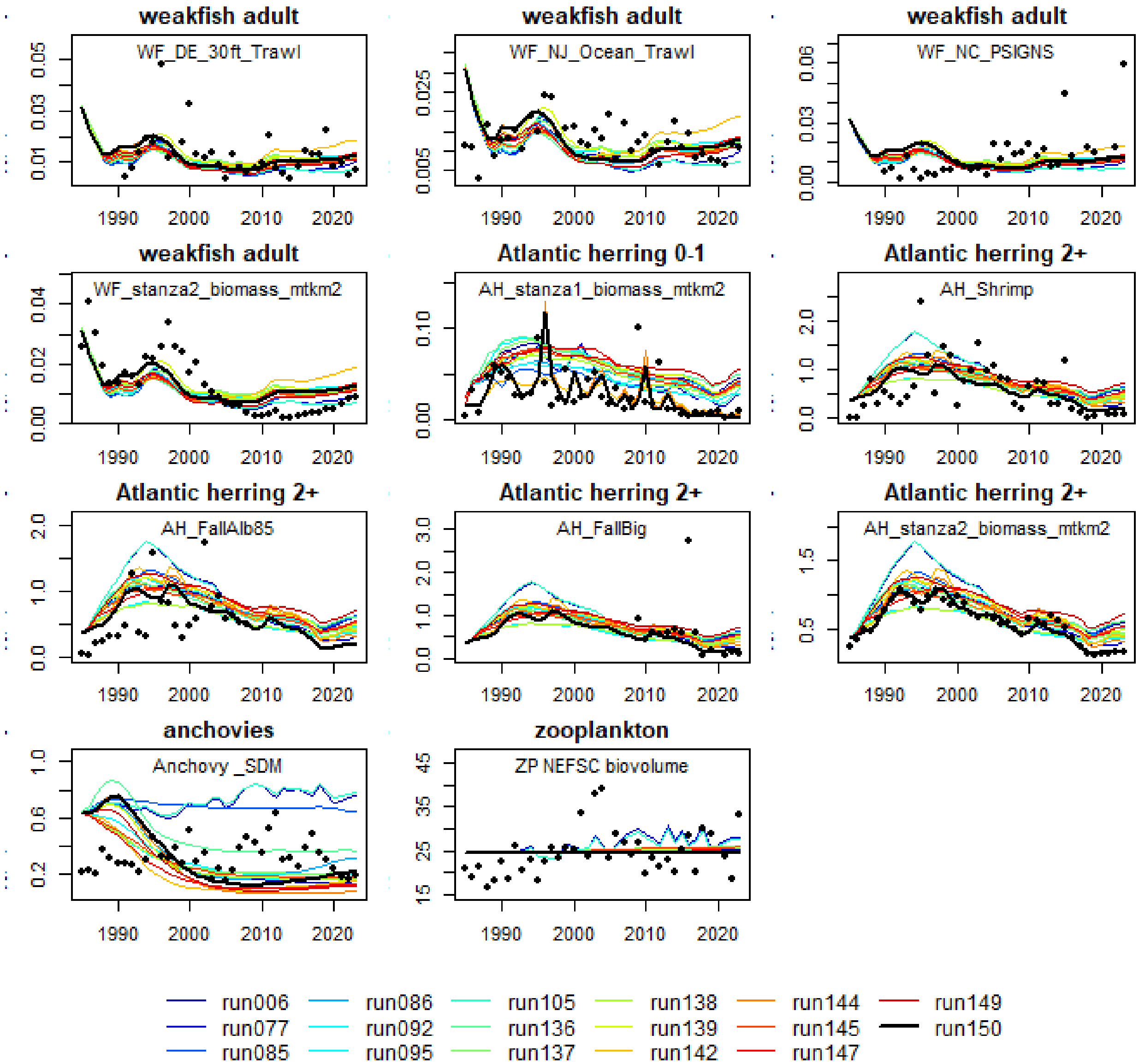


Figure 34 cont. Fits to individual indices and time series of biomass or relative abundance for the NWACS-MICE model.

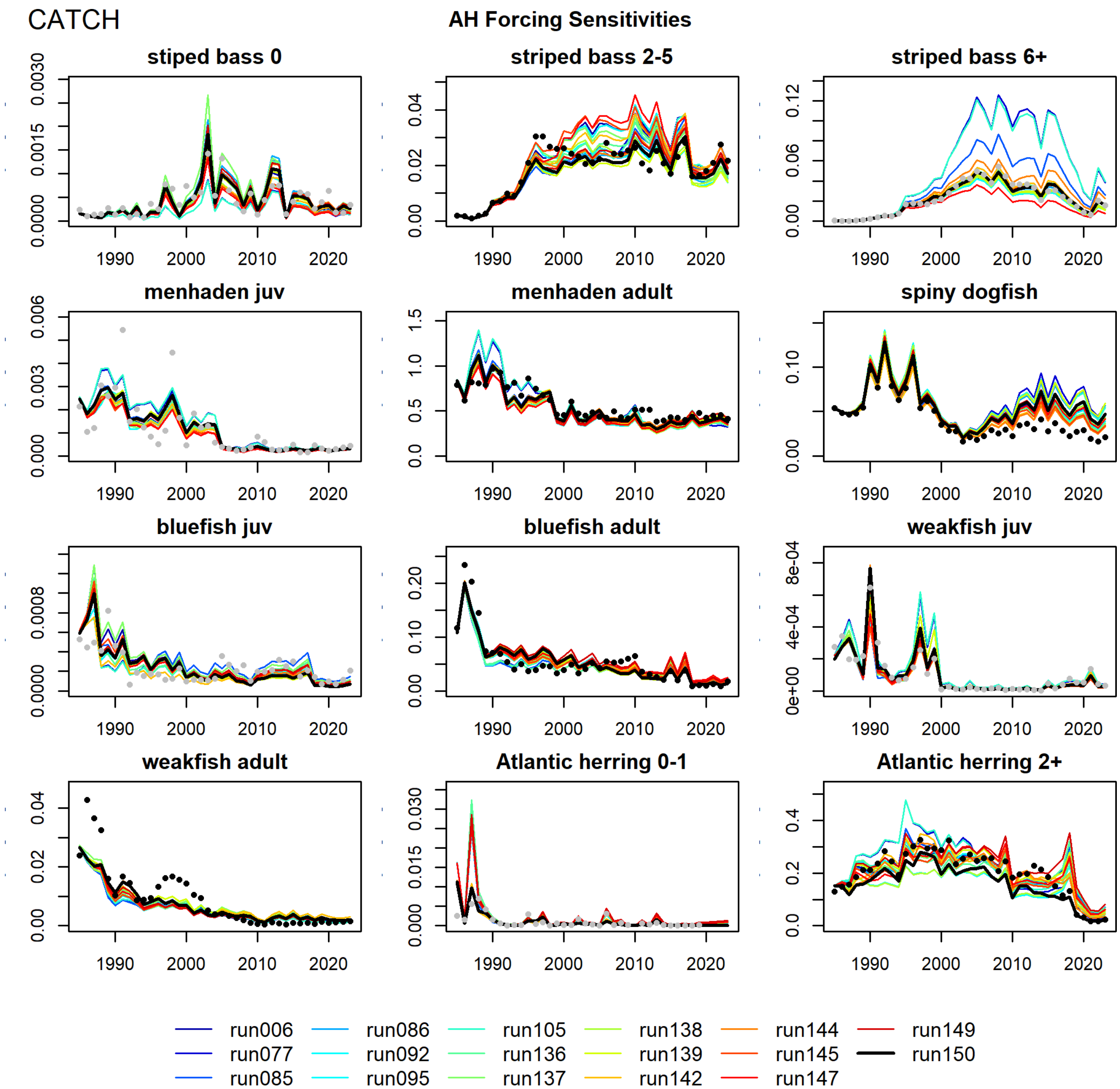


Figure 34. Fits to catch from the top 10 percent of model runs for the NWACS-MICE model. Gray points represent 'relative catch' time series.

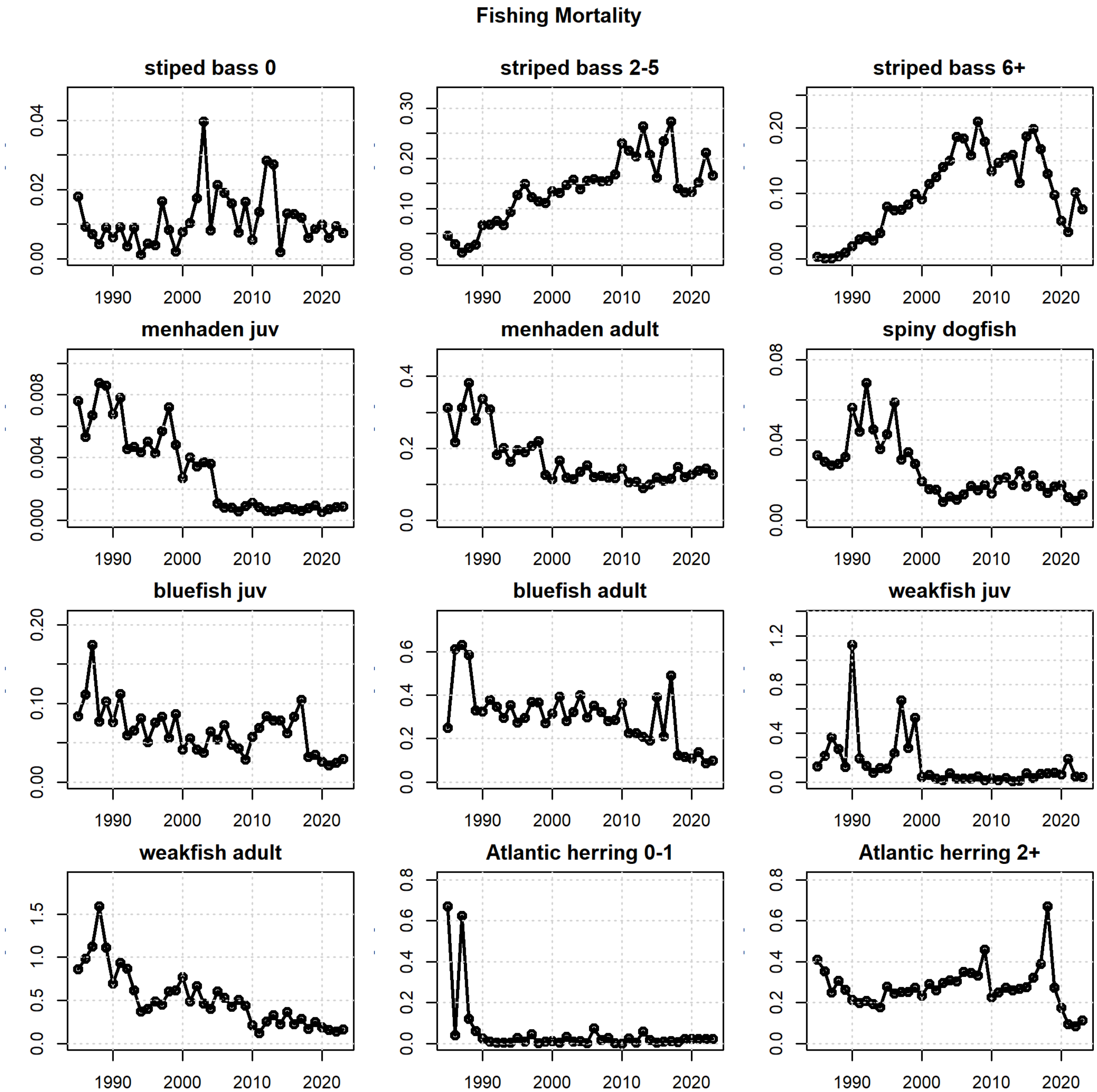


Figure 35. Input fishing mortality timeseries for the NWACS-MICE Ecosim model. All data were converted from stock assessment estimates to stanza-specific units in weight.

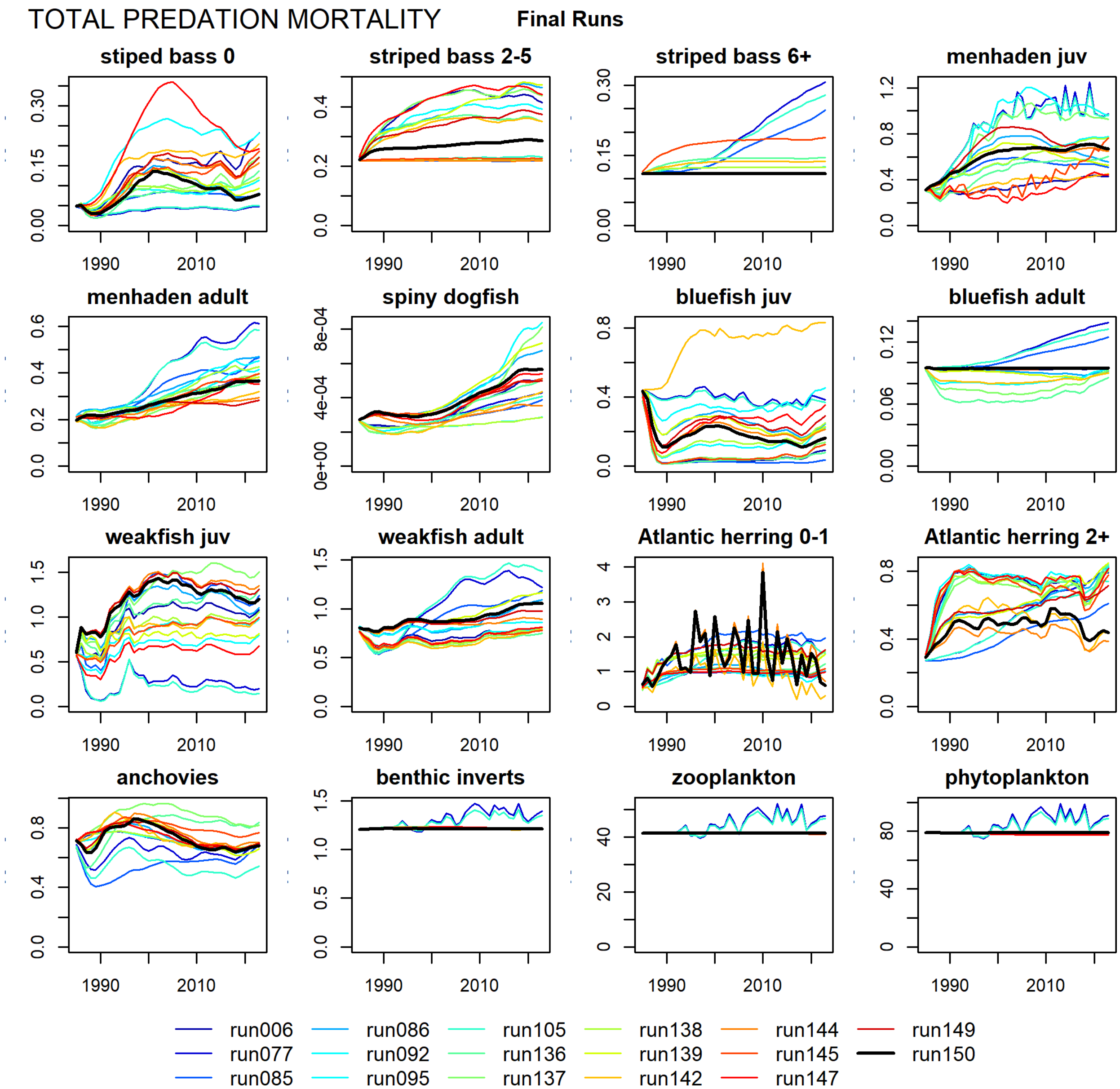


Figure 36. Total predation mortality (summed across predators) by year estimated by the NWACS-MICE model.

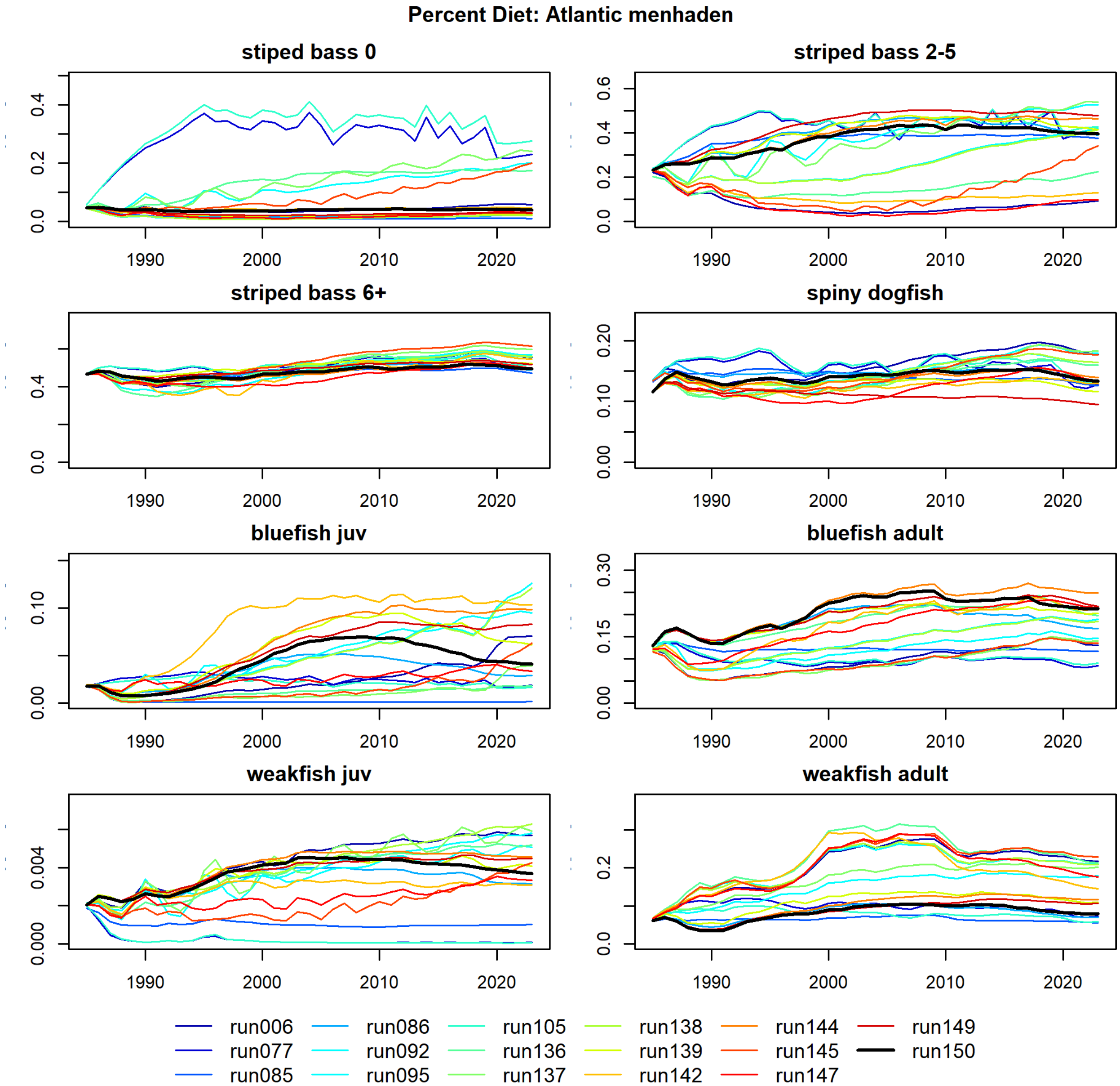


Figure 37. Contribution of Atlantic menhaden (all stanzas combined) to the diets of ERP predators for the NWACS-MICE model, averaged for each year in the model.

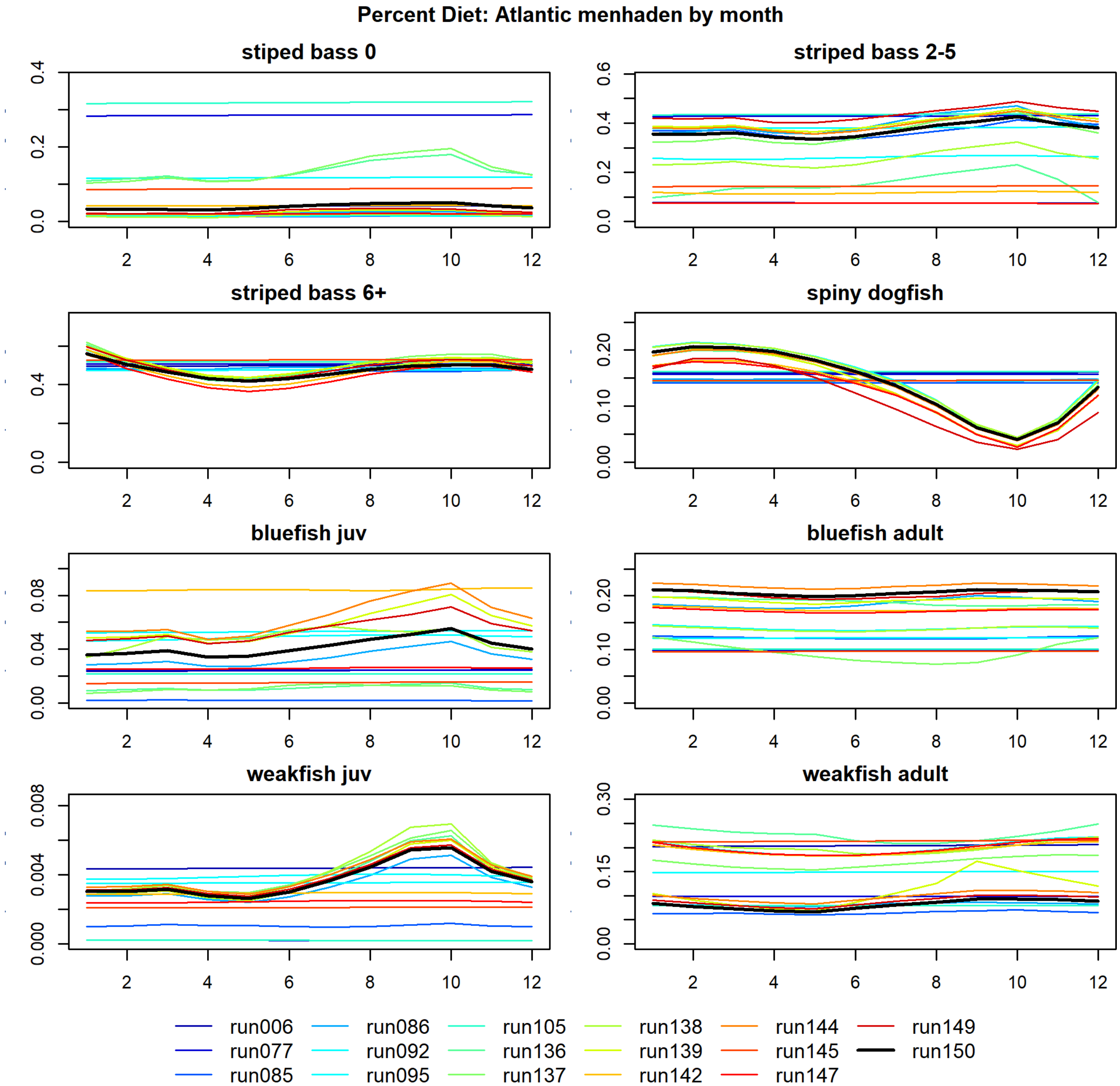


Figure 38. Contribution of Atlantic menhaden (all stanzas combined) to the diets of ERP predators for the NWACS-MICE model, averaged for each month and model run.

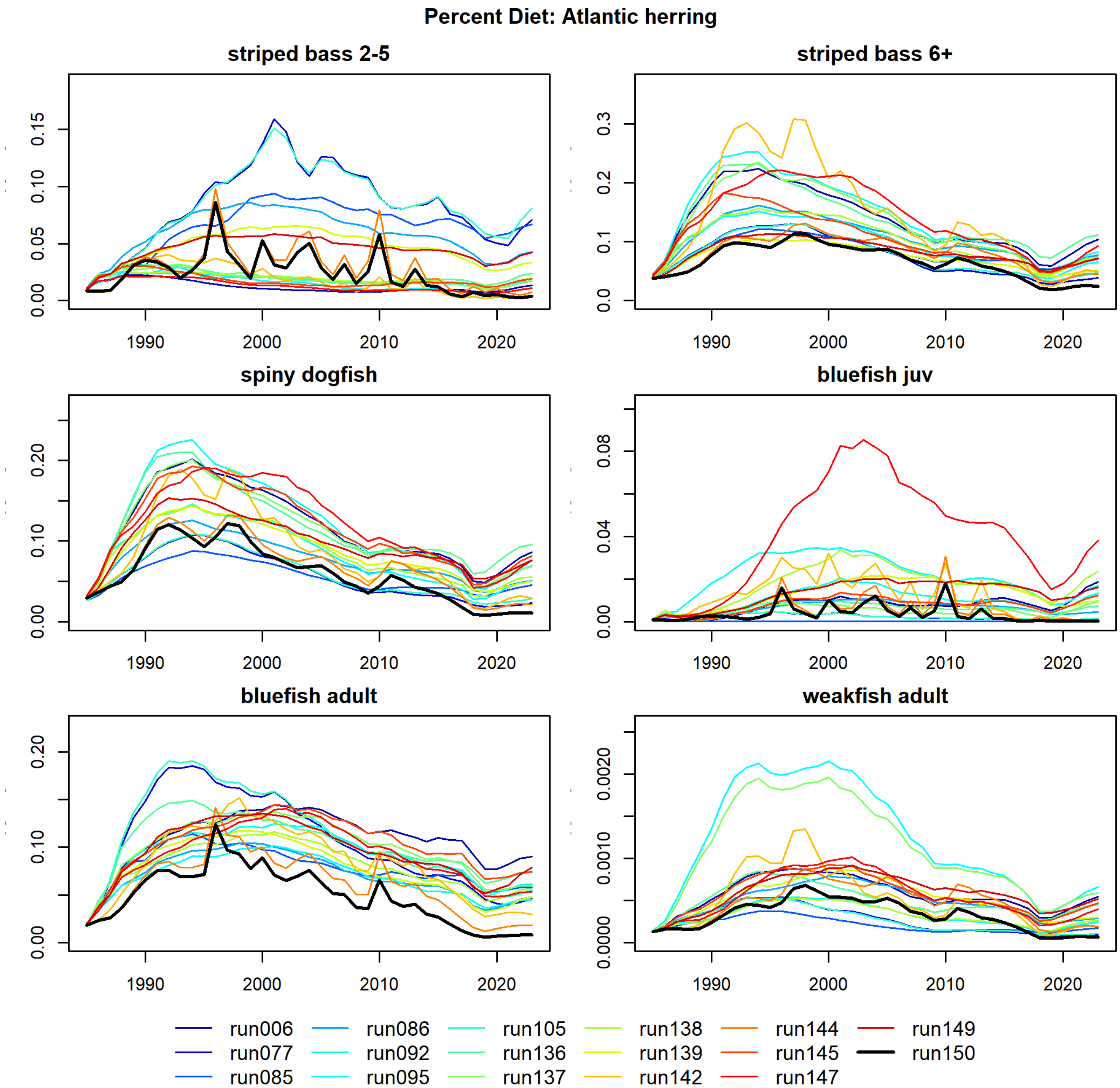


Figure 39. Contribution of Atlantic herring (all stanzas combined) to the diets of ERP predators for the NWACS-MICE model, averaged for each year in the model.

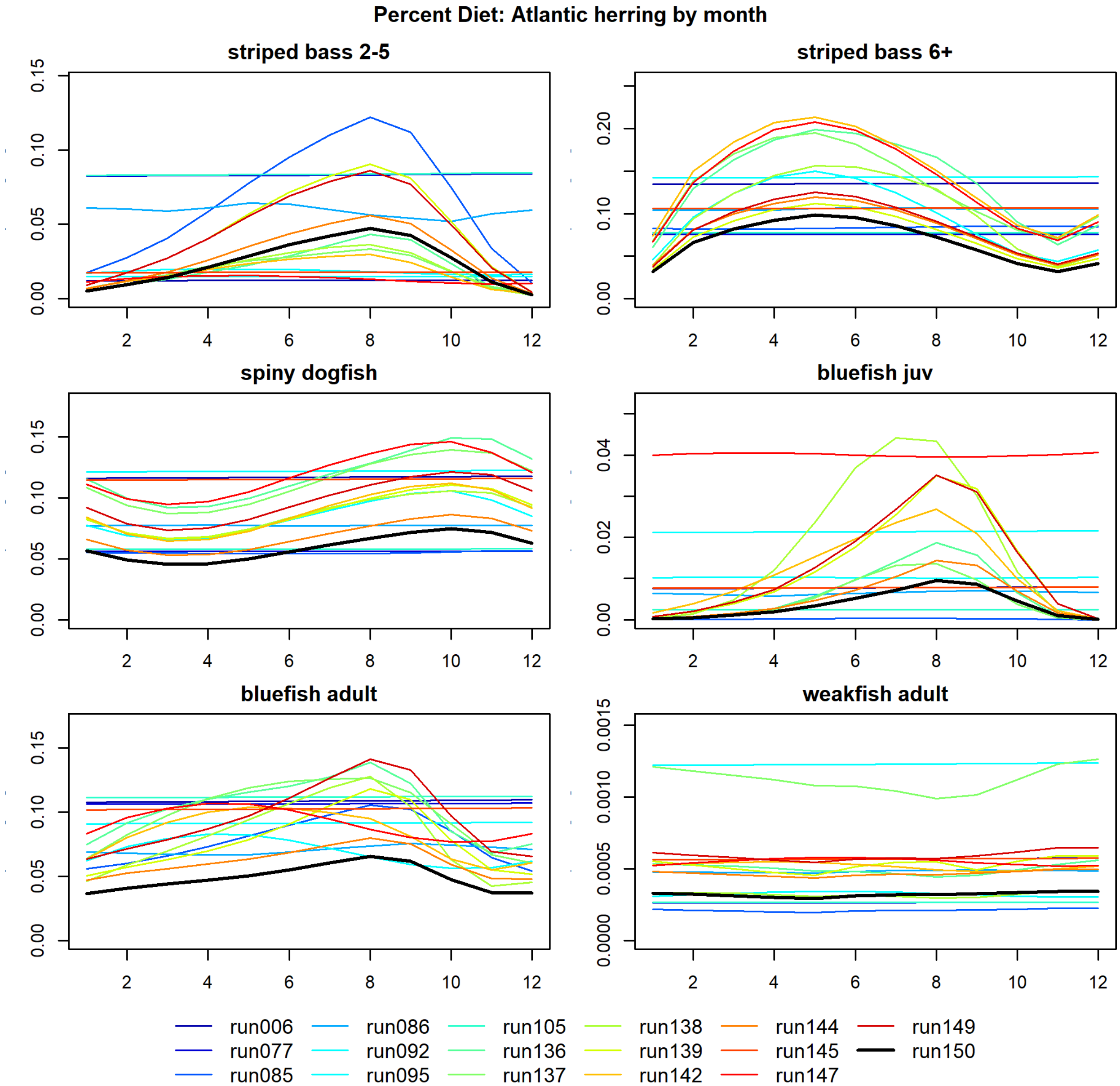


Figure 40. Contribution of Atlantic herring (all stanzas combined) to the diets of ERP predators for the NWACS-MICE model, averaged for each month and model run.

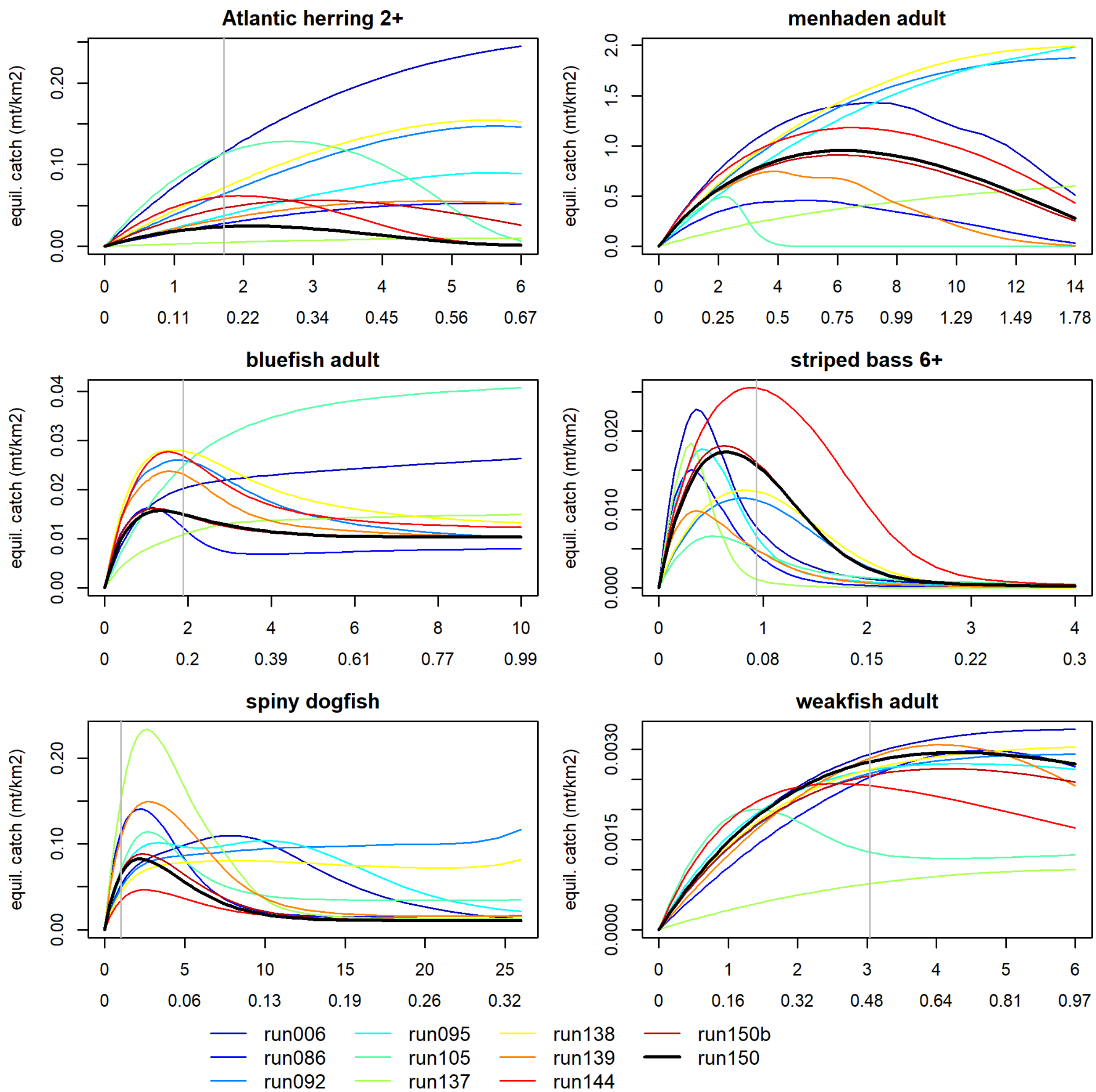


Figure 41. NWACS-MICE equilibrium yield curves. Model runs 'b' assume normal (higher) recruitment for Atlantic herring in the equilibrium projections. The top row of the x-axis is the multiplier on current F and the bottom row is the actual F (in Ecosim units).

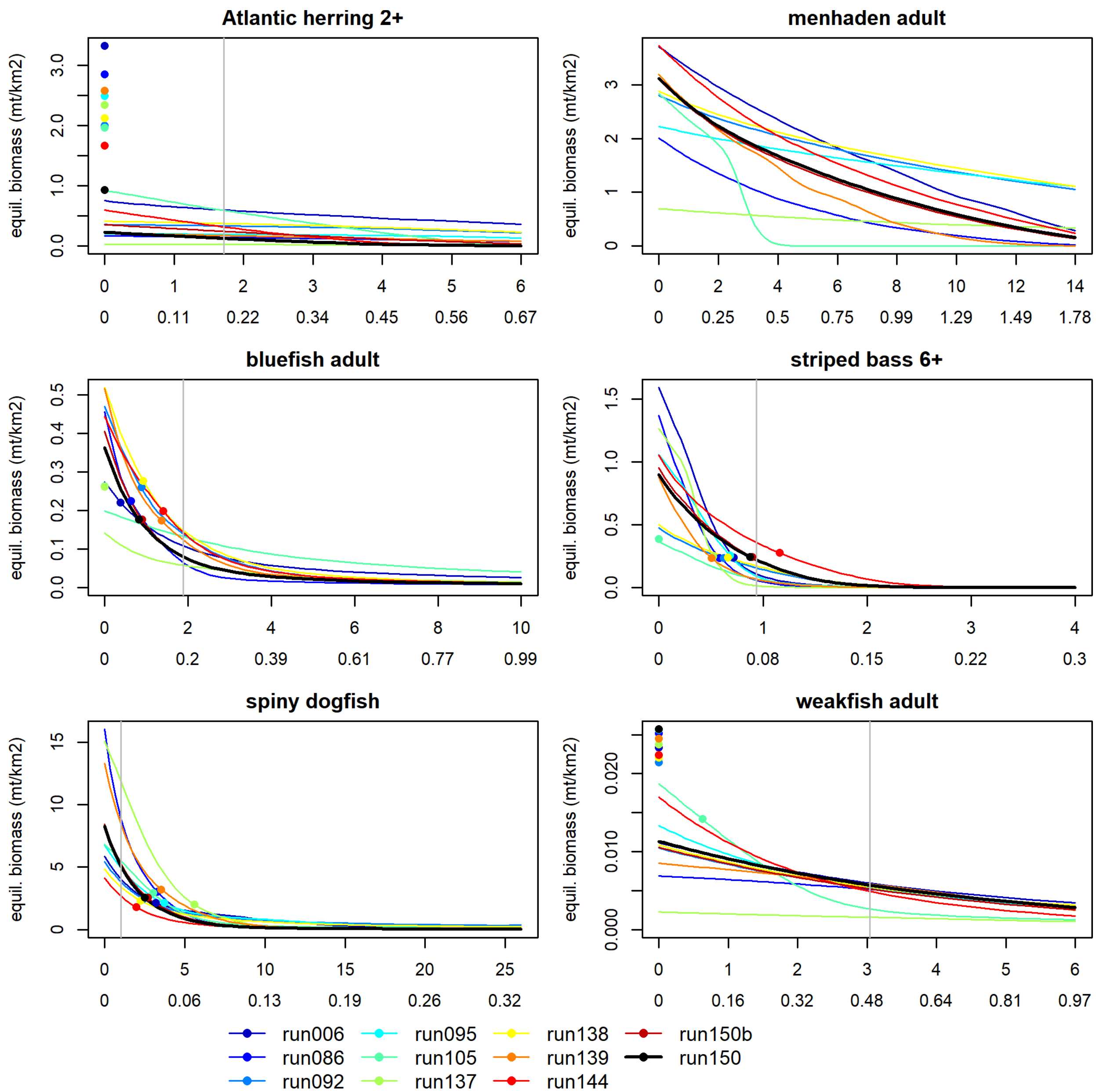


Figure 42. NWACS-MICE equilibrium biomass by fishing mortality rate. The top row of the x-axis is the multiplier on current F and the bottom row is the actual F (in Ecosim units). The equivalent single species F_{TARGET} is indicated by the vertical gray line. The points represent the location of B_{TARGET} specific to each run (determined by multipliers on current biomass) and *F* rate to achieve it. For Atlantic herring and weakfish, the biomass target is not met even at *F*=0 for almost all runs.

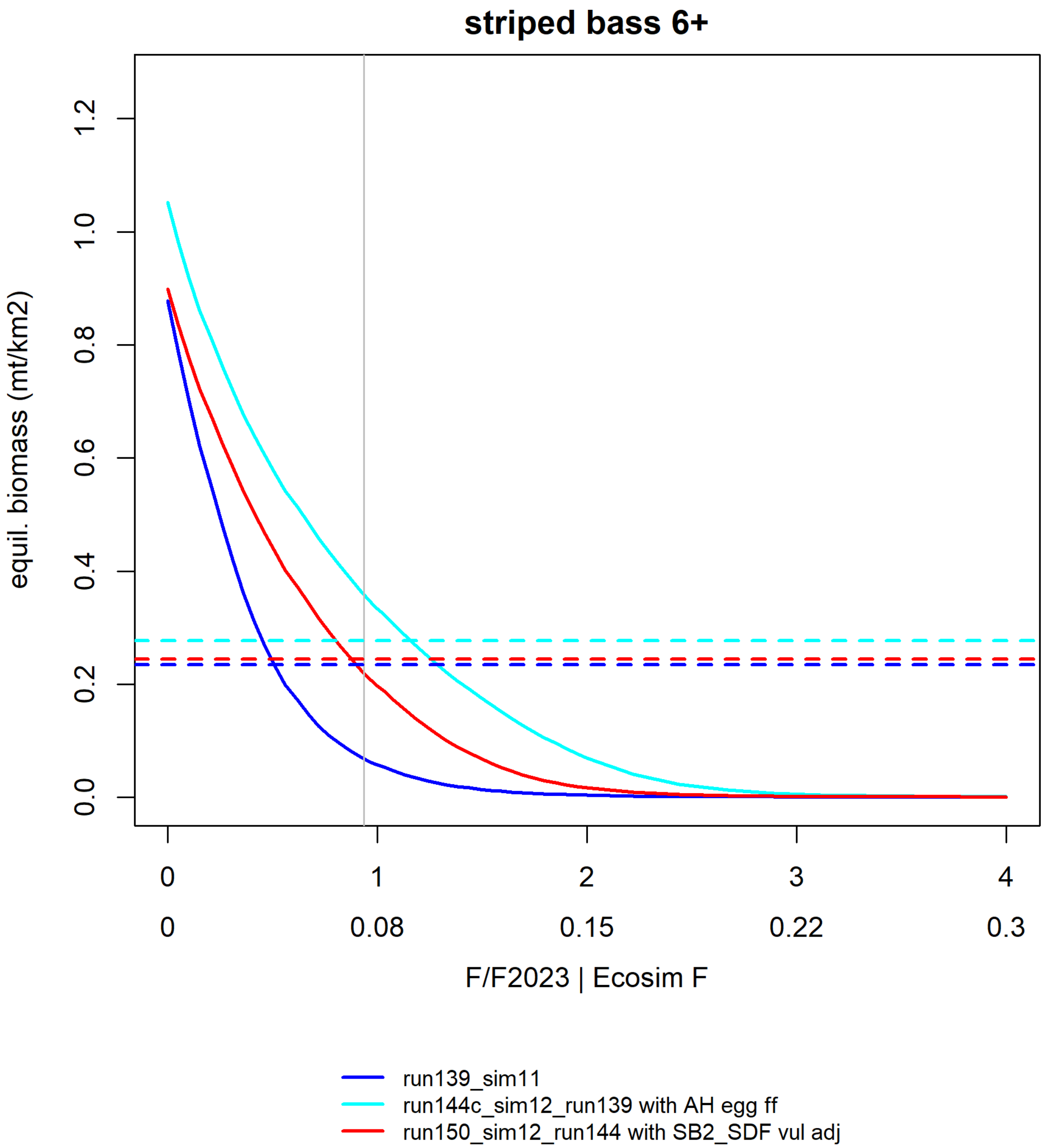


Figure 43. Equilibrium biomass for striped bass age 6+ as a function of menhaden *F* from the NWACS-MICE model, with F_{TARGET} indicated by the vertical gray line and B_{TARGET} for each model run (relative to 1995 biomass) indicated by the dashed lines.

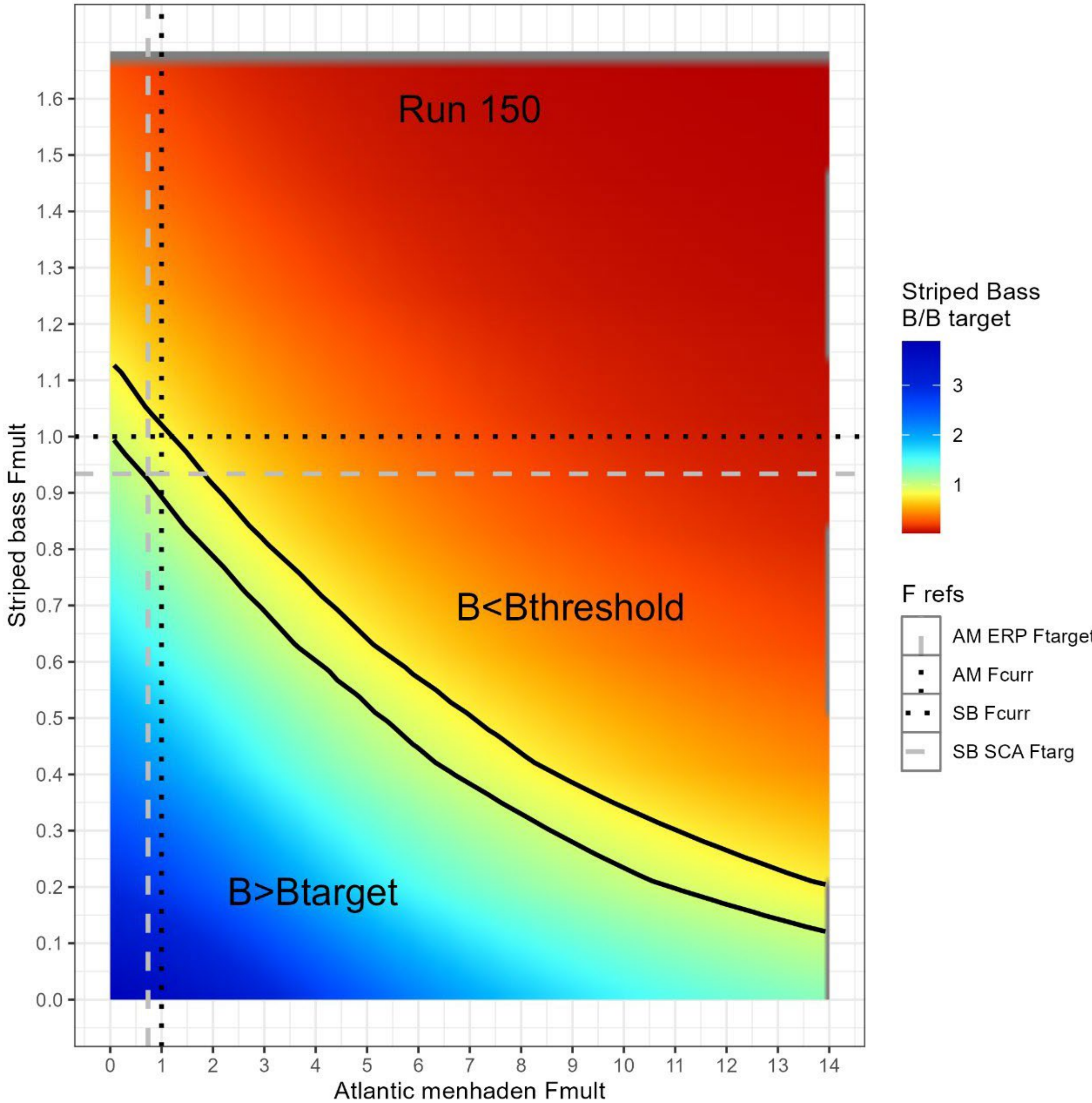


Figure 44. Tradeoff analysis surface plot from run 150 showing the striped bass age 6+ biomass ratio (B/B_{TARGET}) in the terminal year of the NWACS-MICE projections as a function of fishing mortality on both Atlantic menhaden and striped bass. The solid black lines represent the contours where striped bass B=$B_{THRESHOLD}$ and B=B_{TARGET}. The dashed lines highlight specific F scenarios where F is equivalent to the F in 2023 (striped bass = SB F_{curr}, Atlantic menhaden = AM F_{curr}) or the F_{TARGET} for each species (proof-of-concept ERP for Atlantic menhaden = AM ERP F_{TARGET}, striped bass = SB SCA F_{TARG}).

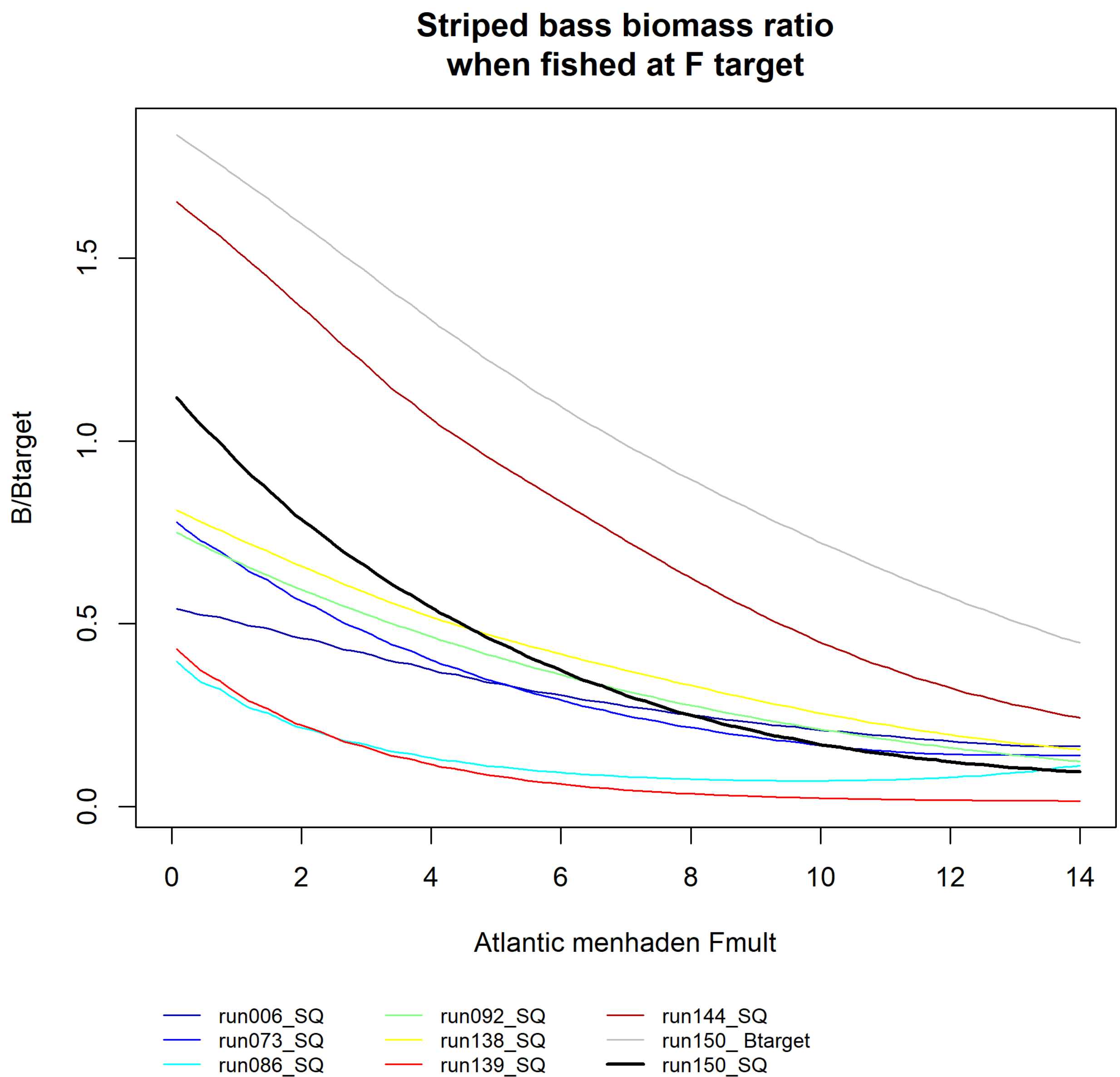


Figure 45. Comparison of tradeoff curves for striped bass, when fished at their F_{TARGET} (F_{MULT}=0.934)

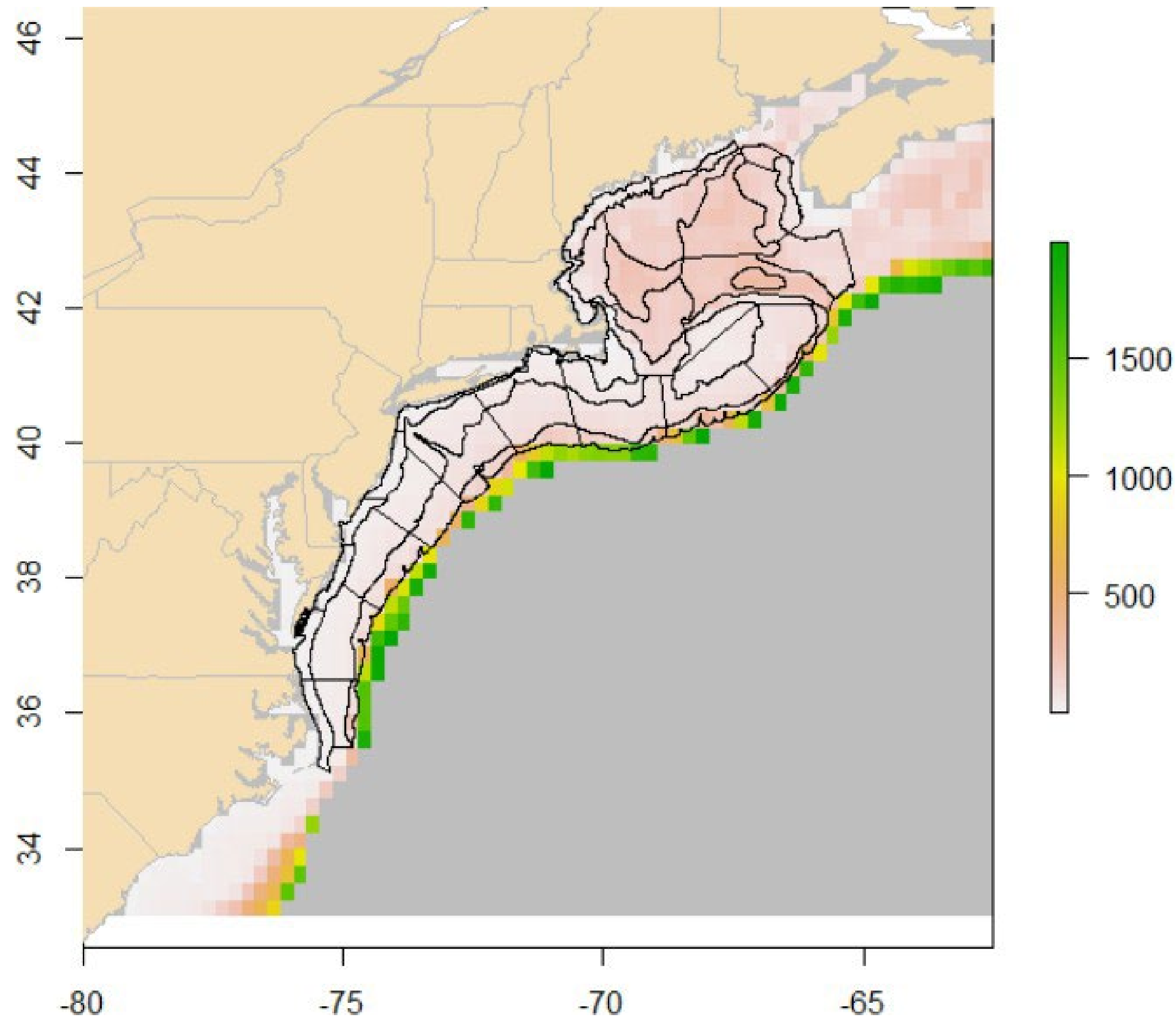


Figure 46. Spatial extent of the Northwest Atlantic Continental Shelf (NWACS) Full Ecosystem model (colored raster), extending from just south of North Carolina, USA to Nova Scotia, Canada. Black lines represent the survey strata for the NEFSC bottom trawl fisheries survey and the general spatial extent of the previous NWACS-FULL ecosystem. Color scale represents depth in meters.

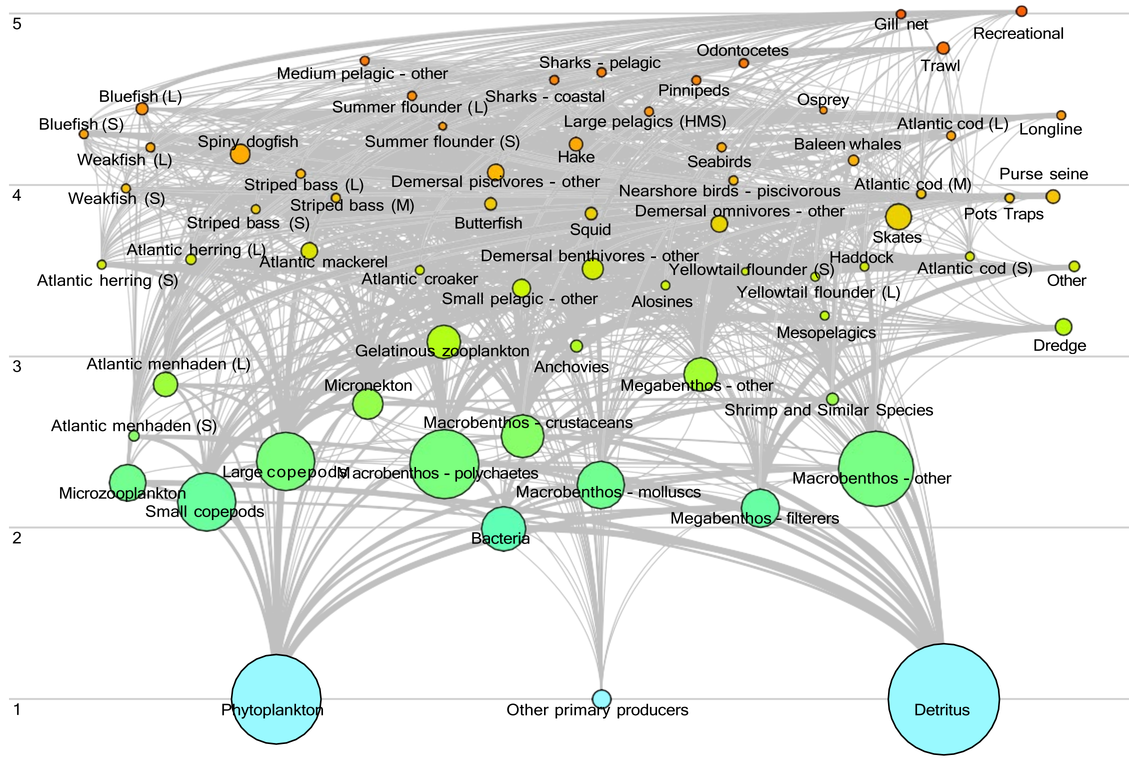


Figure 47. Flow diagram for the NWACS-FULL Model 2023 v2.1. Nodes represent biomass of modeled trophic groups (scaled to the logarithm of the group's biomass). Lines represent trophic linkages (with a scaled thickness). Colors and horizontal lines denote trophic levels.

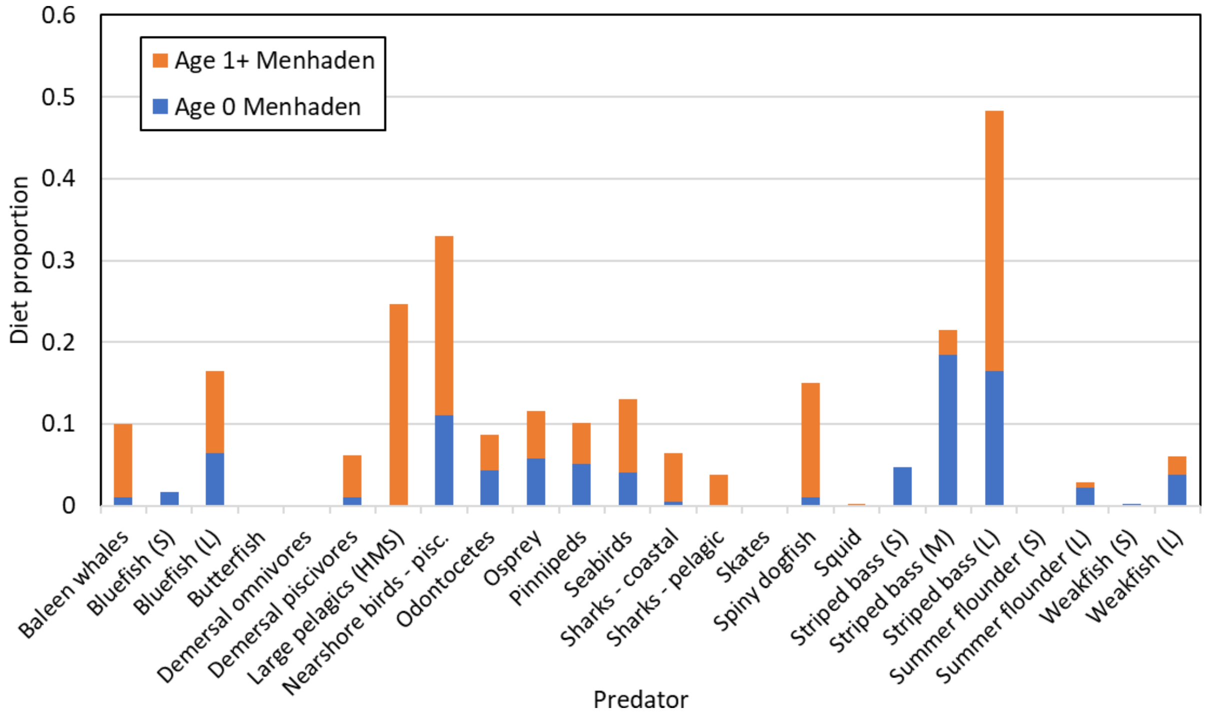


Figure 48. Contributions of age-0 and age-1+ Atlantic menhaden in the diets of all menhaden predators in the balanced NWACS FULL Ecosystem model. Six predators had diet contributions <0.05%.

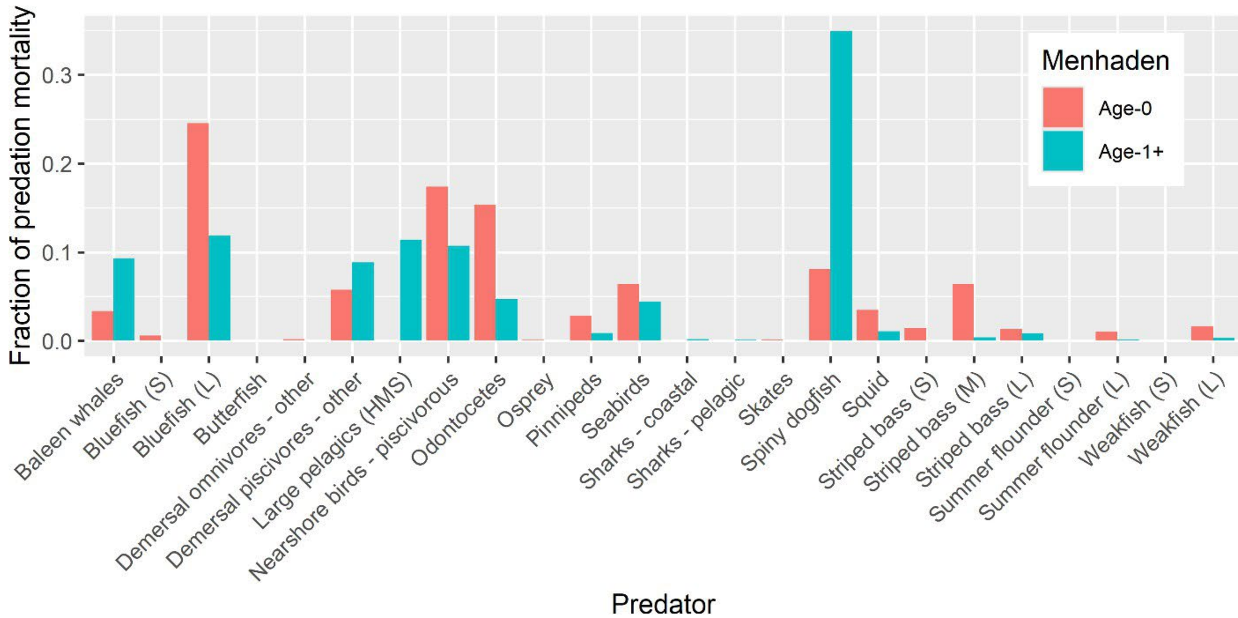


Figure 49. Predator contributions to age-0 and age-1+ Atlantic menhaden predation mortality (as a fraction of total M_2) in 1985. Results are based on sim 2.9 of the NWACS-FULL model in 1985.

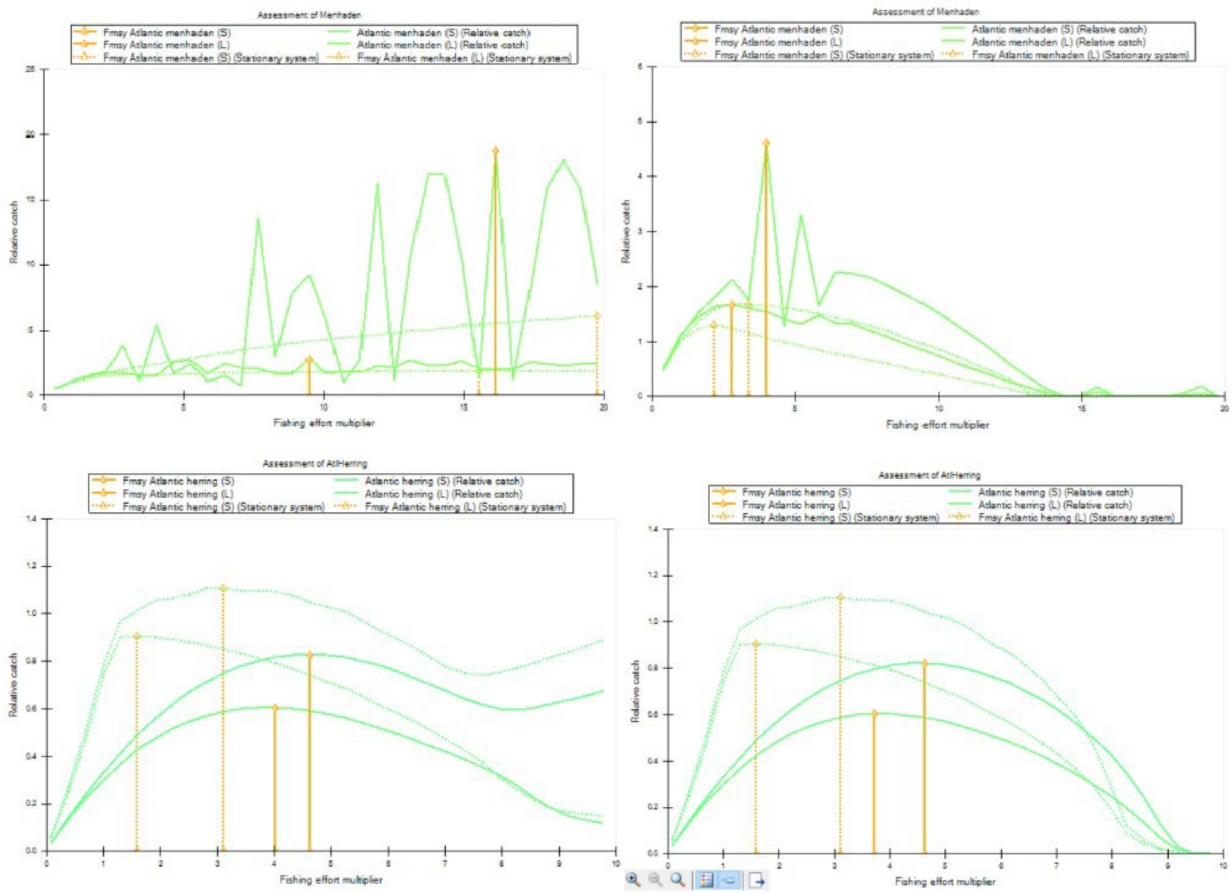


Figure 50. Effect of specifies-specific fishing effort on the relative equilibrium catch of that species, based on sim 2.1 from the NWACS-FULL model. Each row of plots is for a different ERP species whose minimum k_{ij} parameter was modified (Atlantic menhaden, Atlantic herring, Bluefish, Weakfish). The left plot uses the vulnerabilities as estimated for sim 2.1. The right plot is the result after increasing the minimum k_{ij} parameter from k_{ij} =1 to a different value for each species (ranging from 1.01 to 1.2; see Table 24). Dashed lines represent model predictions under a stationary system and solid lines are for predictions with full compensation in the system which are expected to be more realistic.

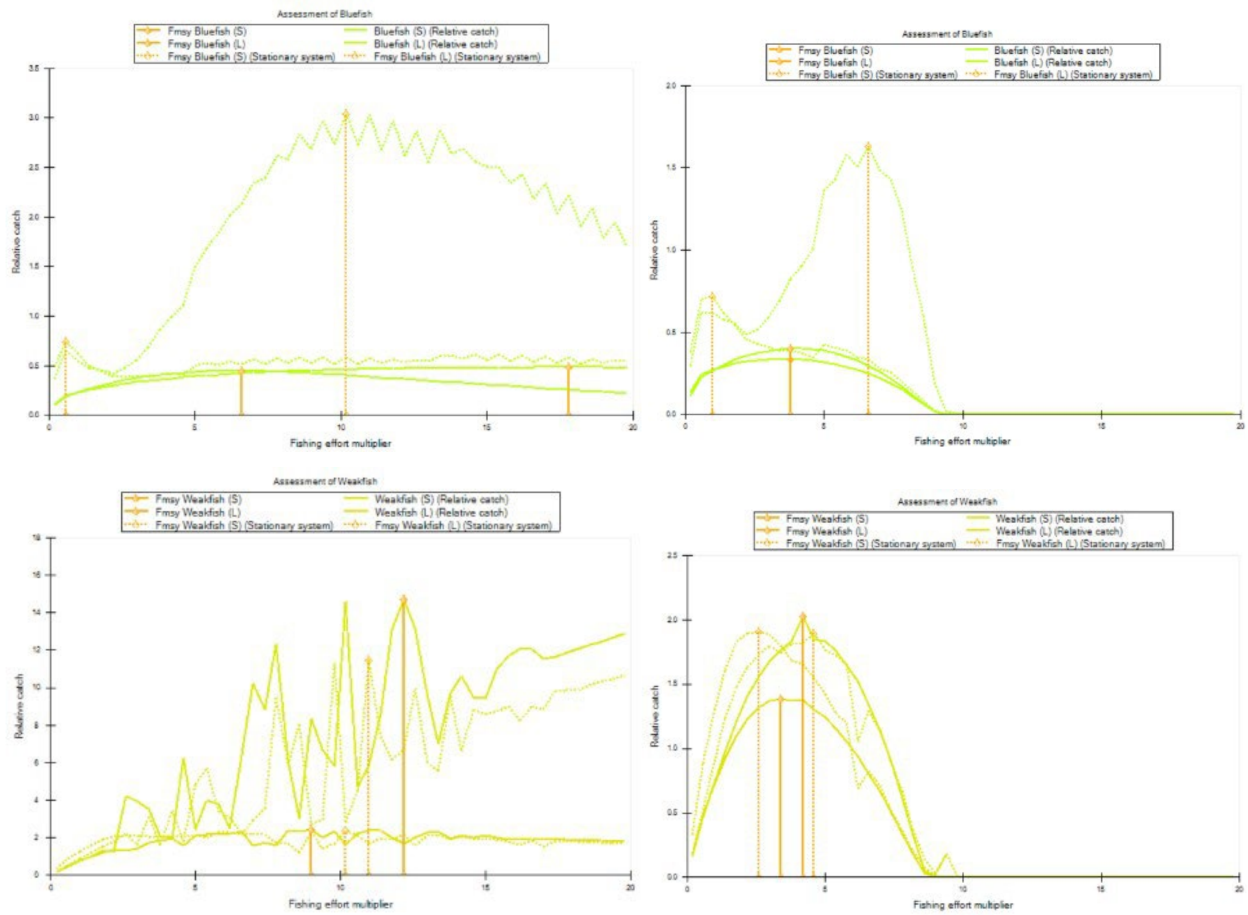


Figure 50 (cont.)

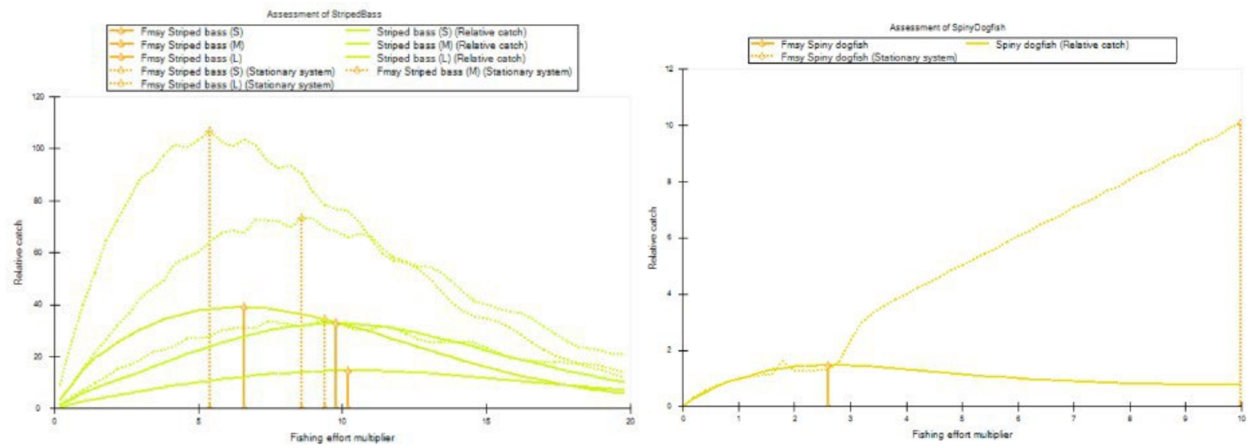


Figure 51. Effect of specifies-specific fishing effort on the relative equilibrium catch of Striped Bass (left) and Spiny Dogfish (right), based on sim 2.1 of the NWACS-FULL model. These species did not have any modifications made to their minimum k_{ij} parameter because all k_{ij} were > 2. Dashed lines represent model predictions under a stationary system and solid lines are for predictions with full compensation in the system which are expected to be more realistic.

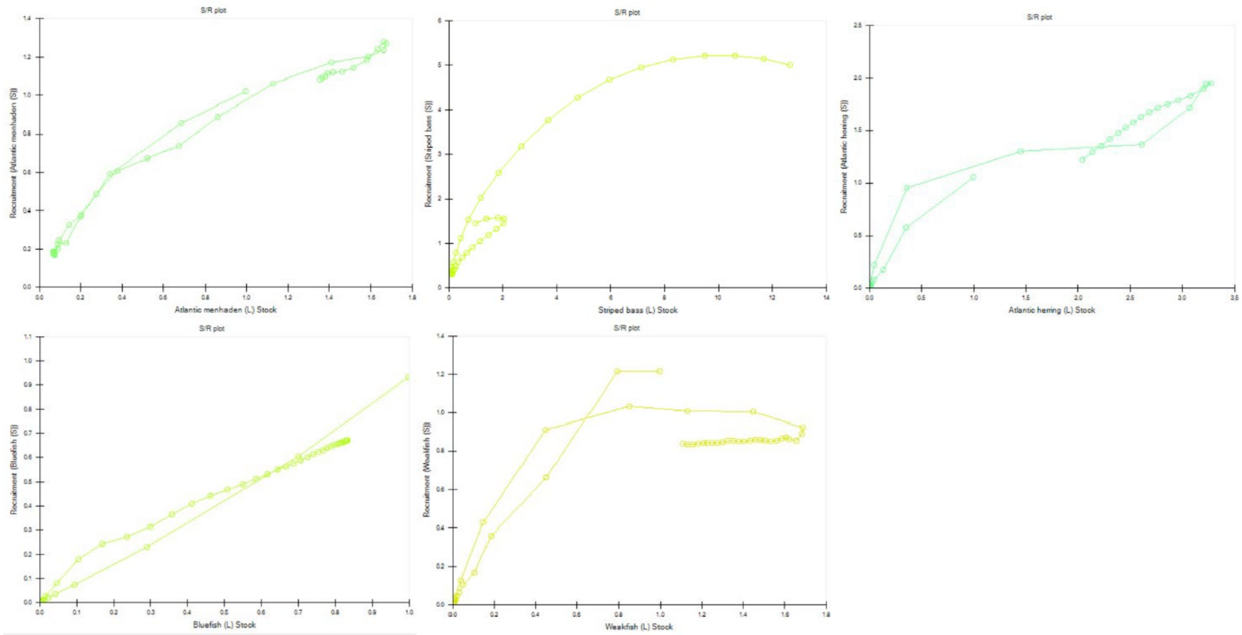


Figure 52. Emergent stock recruitment relationships in the NWACS Full Ecosystem model for the ERP species with multiple stanzas (Atlantic menhaden, striped bass, Atlantic herring, Bluefish, and Weakfish). These S-R relationships were generated by simulating a substantial increase in fishing effort for each species followed by a decline in effort to zero. The points represent the model predictions over time, and the goal is to see relationships that approximate a Beverton-Holt curve with some indication of compensation during the recovery period.

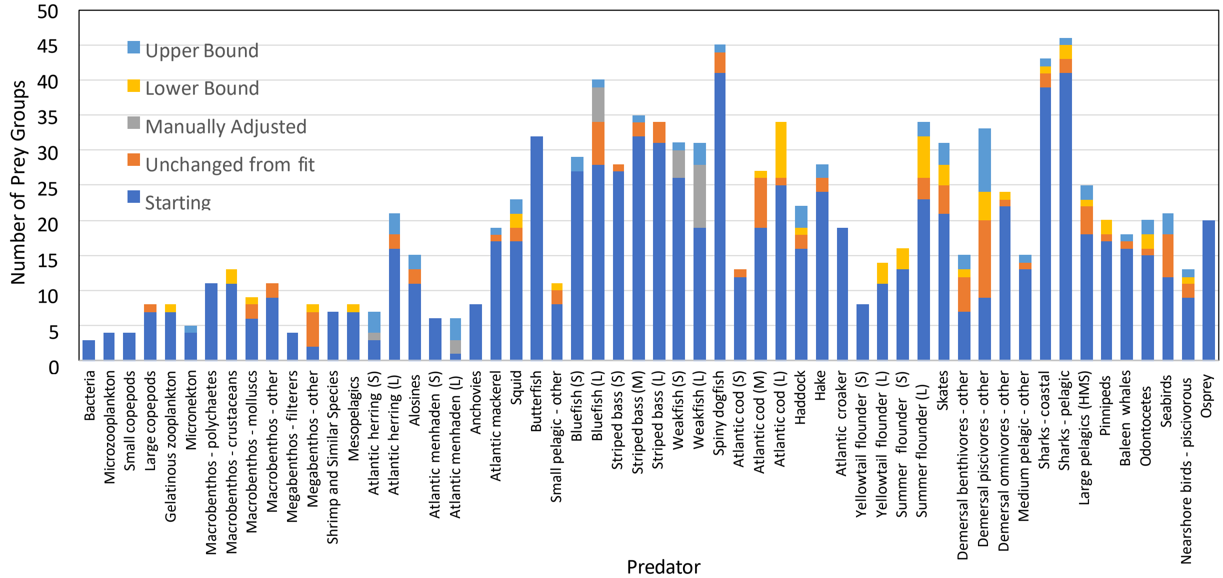


Figure 53. Number of total prey groups (and vulnerability parameters) for each NWACS-FULL predator. Colors indicate the number of vulnerability parameters (k_{ij}) from sim 2.9 that were: equal to the starting value (dark blue), unchanged from the initial fit (orange), manually adjusted away from a lower bound to improve diagnostics for ERP species (gray), on a lower bound < 1.01 (yellow), or on an upper bound >1e+9 (light blue).

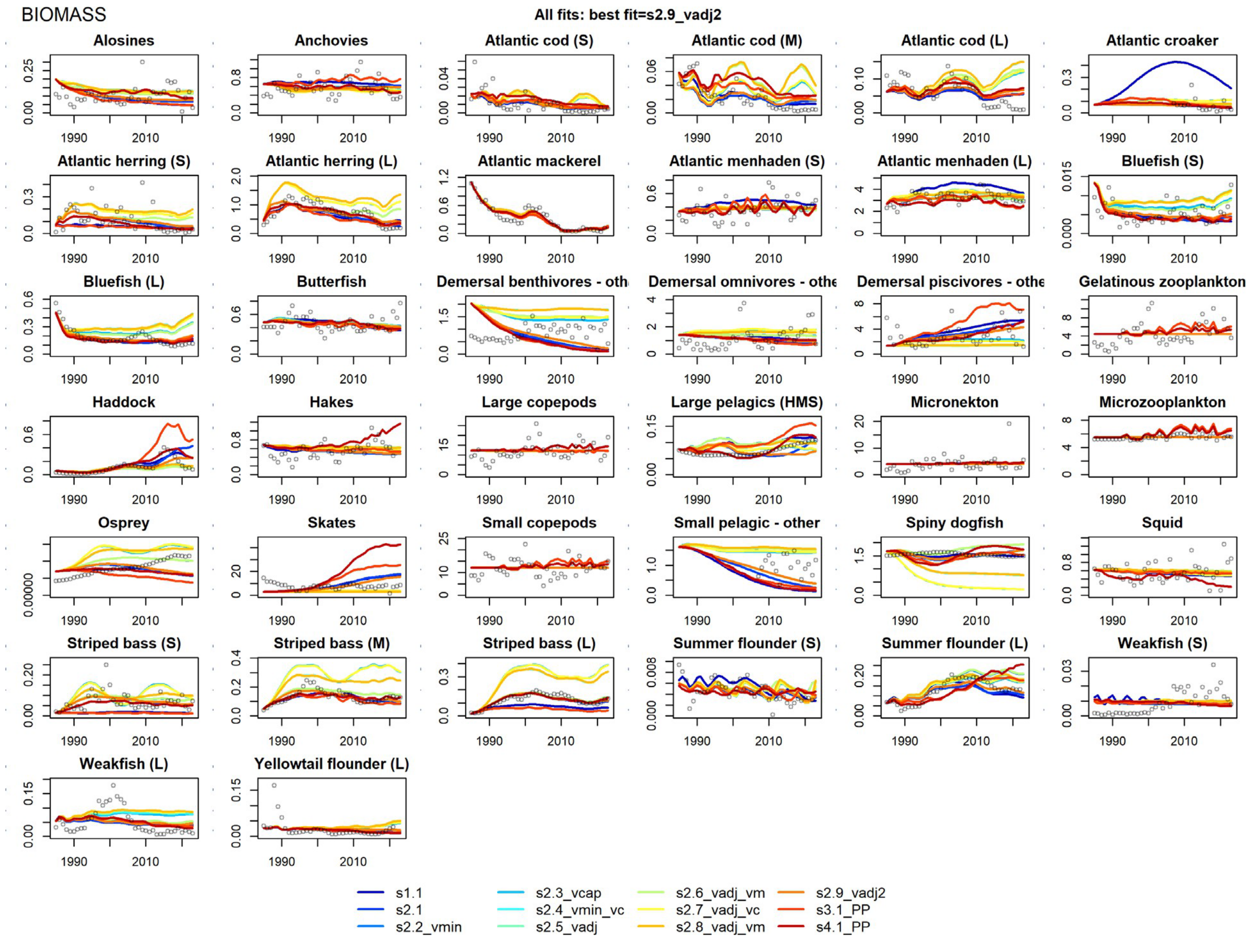


Figure 54. Biomass fits for the NWACS-FULL model. Lines depict predicted biomass estimates by year for different model simulations (see legend). Points depict time series of absolute biomass (mt/km²) from stock assessments or relative biomass from fisheries surveys (magnitude of points is scaled based on sim 2.9). Panels are labeled by trophic group code and numeric stanza. Trophic groups without observed, empirical data are excluded.

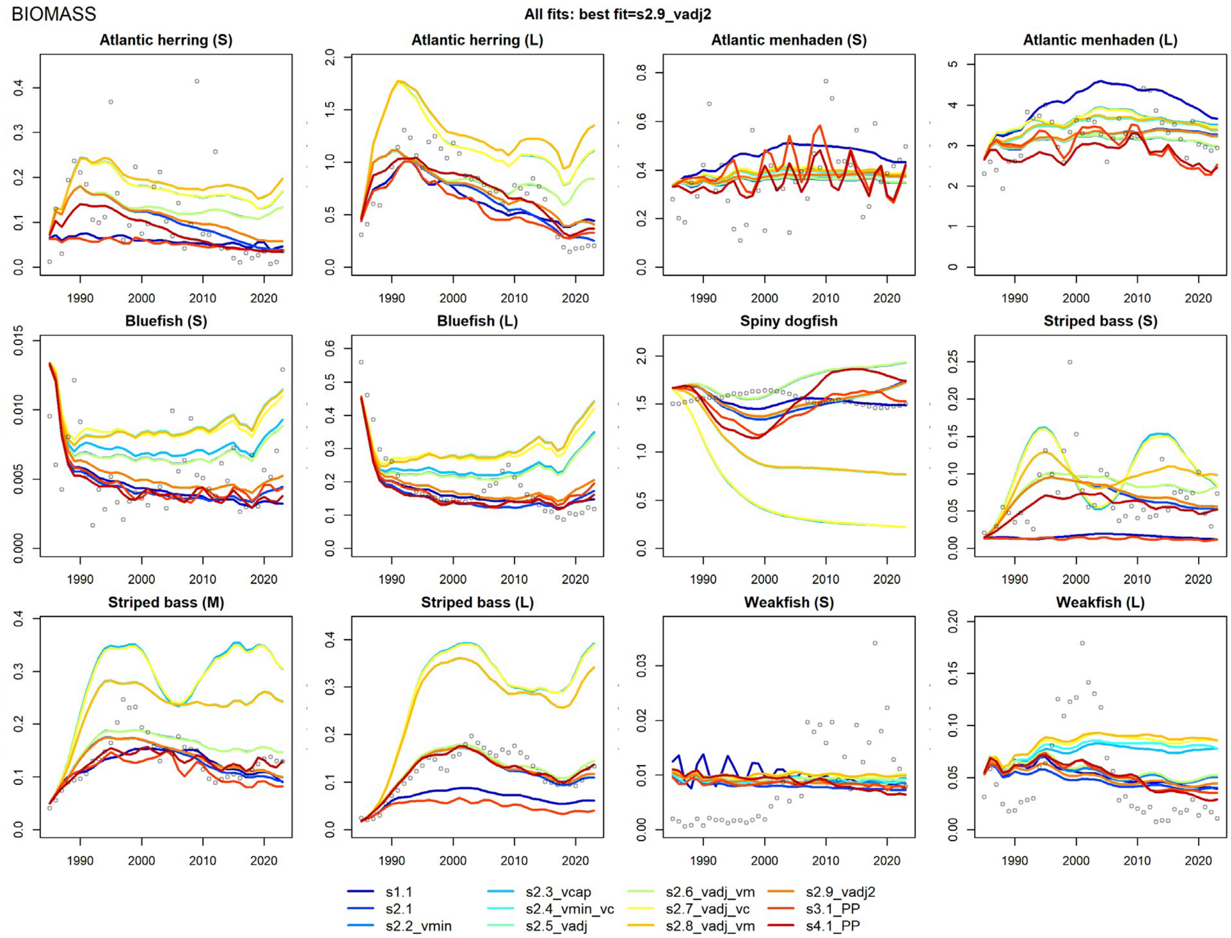


Figure 55. Biomass (mt/km^2) fits for the NWACS-FULL model for ERP species (Atlantic Herring [AHERR], Bluefish [BLUE], Atlantic Menhaden [MENH], Spiny Dogfish [SPDOG], Striped Bass [STBASS], Weakfish [WEAK]). See Figure 54 for full description.

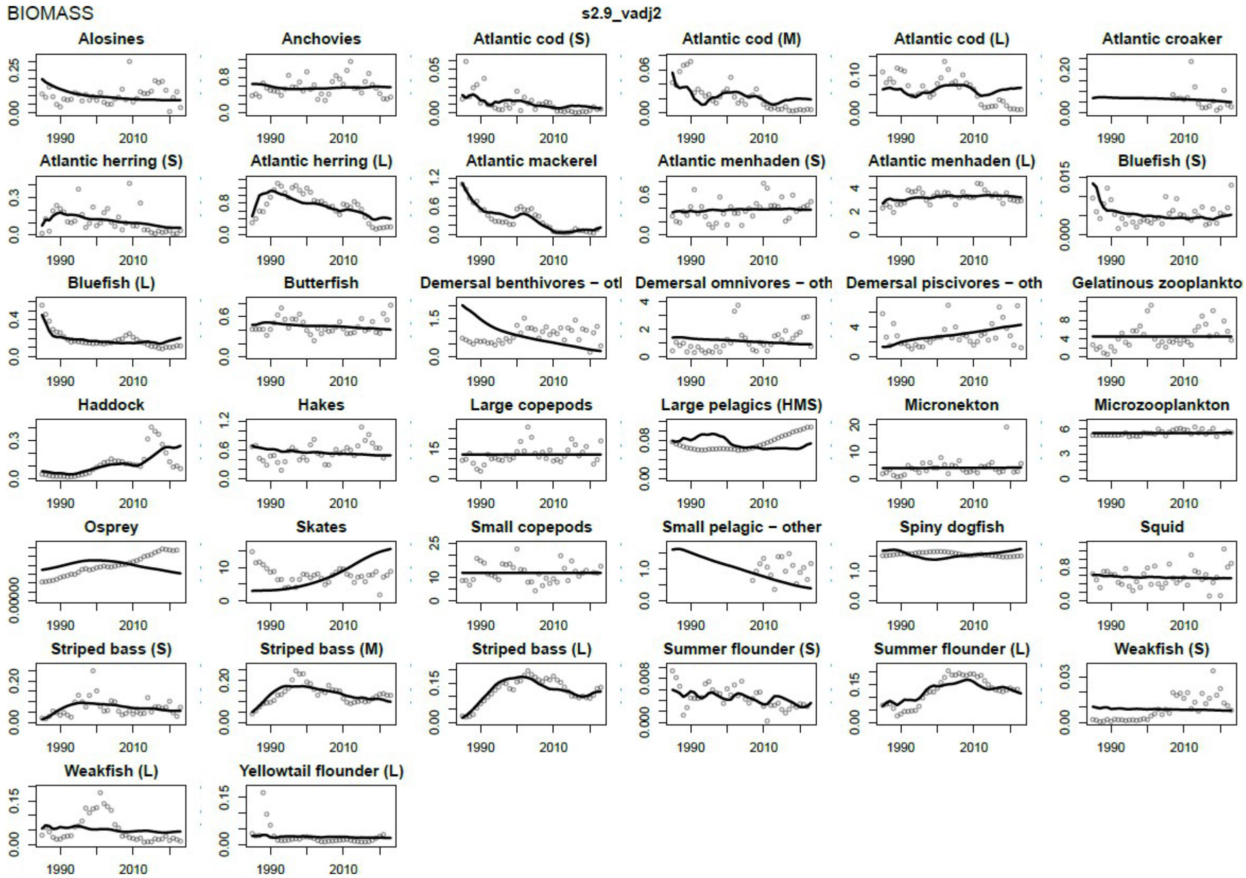


Figure 56. Biomass fits for NWACS-FULL model for the base run, sim 2.9. Lines depict predicted biomass estimates by year. Points depict time series of absolute biomass (mt/km^2) from stock assessments or relative biomass from fisheries surveys. Panels are labeled by trophic group code and numeric stanza. Trophic groups without observed, empirical data are excluded.

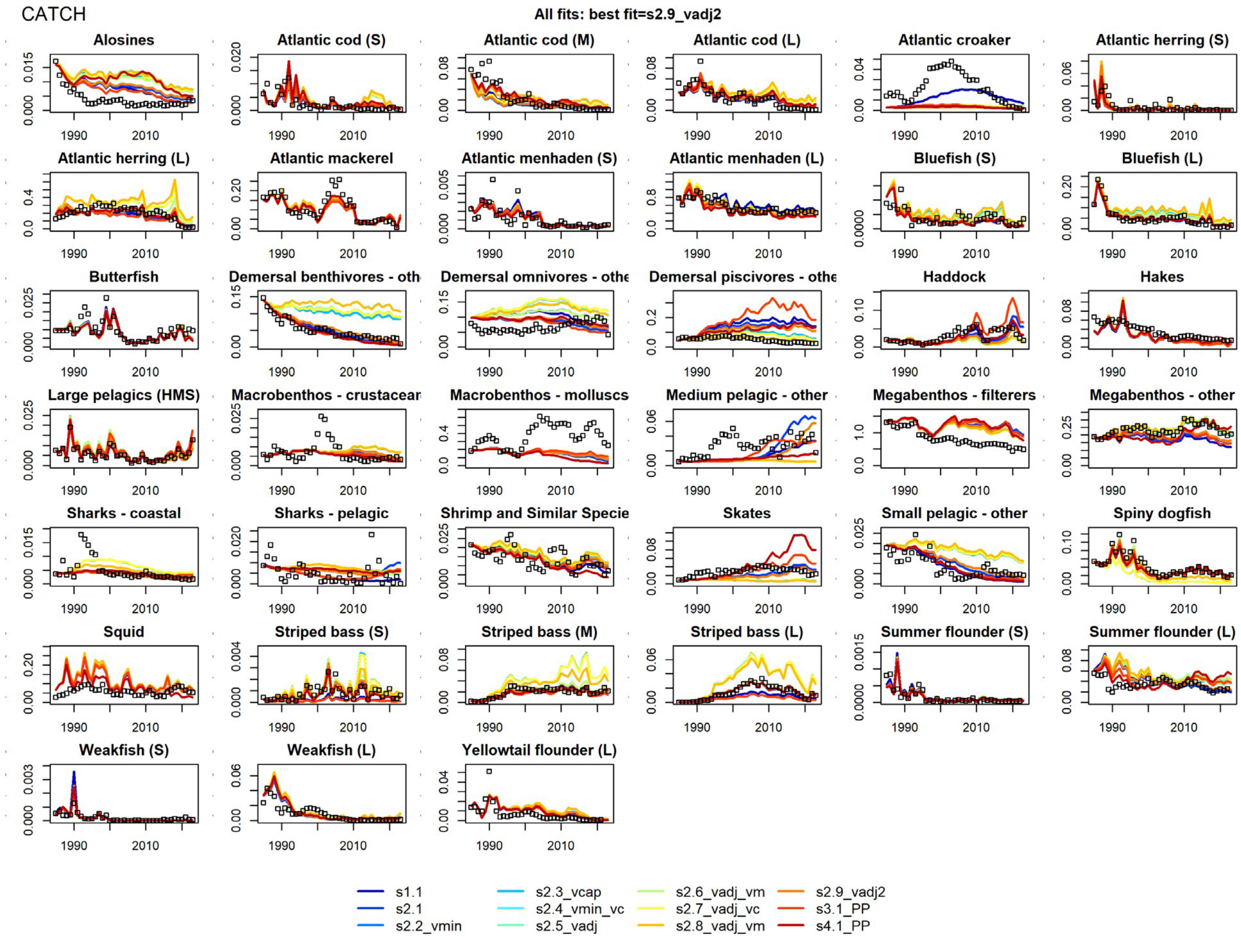


Figure 57. Catch fits for NWACS-FULL model. Points are the observed catches (mt/km²) and lines depict predicted catch by year for different model simulations (see legend). Panels are labeled by trophic group code and numeric stanza. Trophic groups without observed, empirical data are excluded.

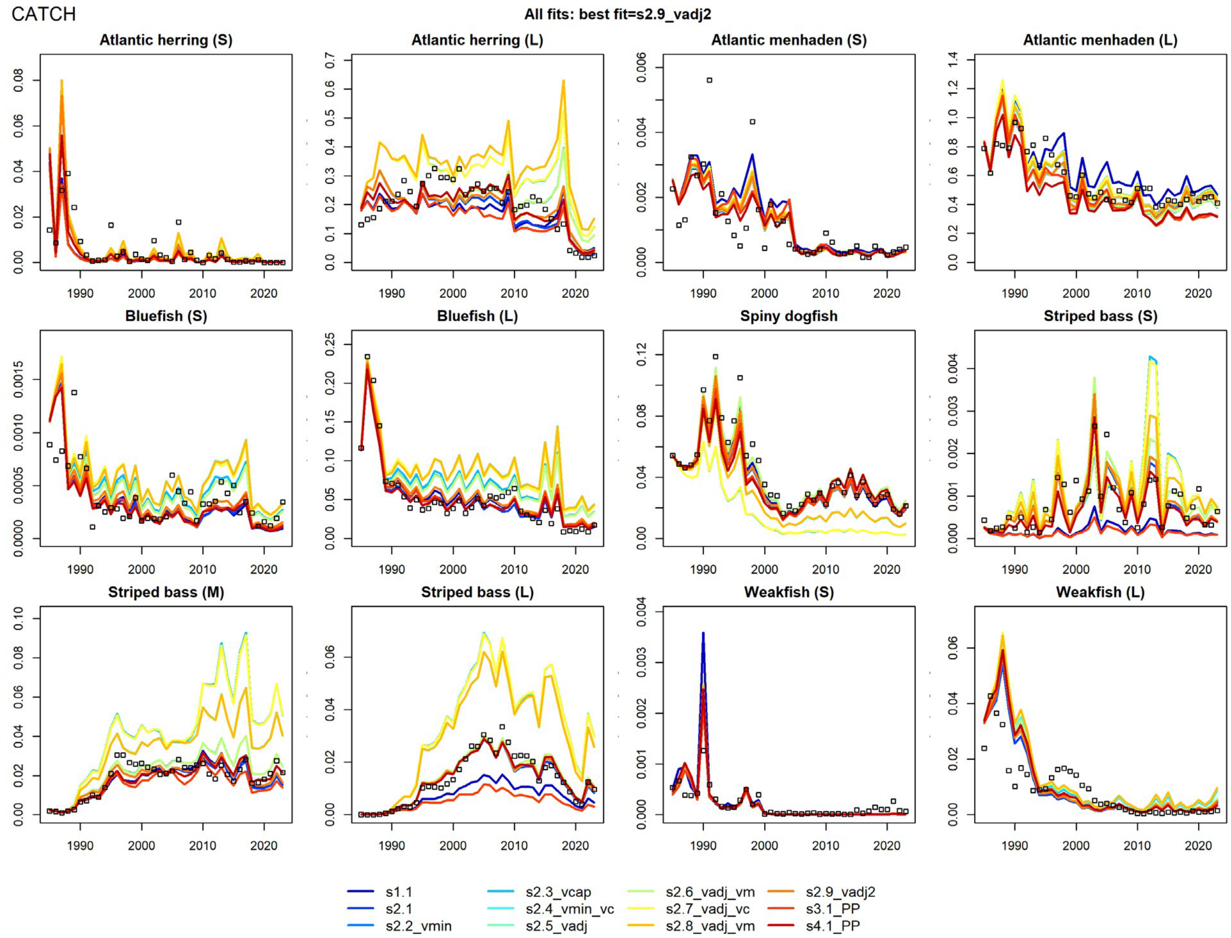


Figure 58. Catch (mt/km^2) fits for the NWACS-FULL model for ERP species (Atlantic Herring [AHERR], Bluefish [BLUE], Atlantic Menhaden [MENH], Spiny Dogfish [SPDOG], Striped Bass [STBASS], Weakfish [WEAK]). See Figure 57 for full description.

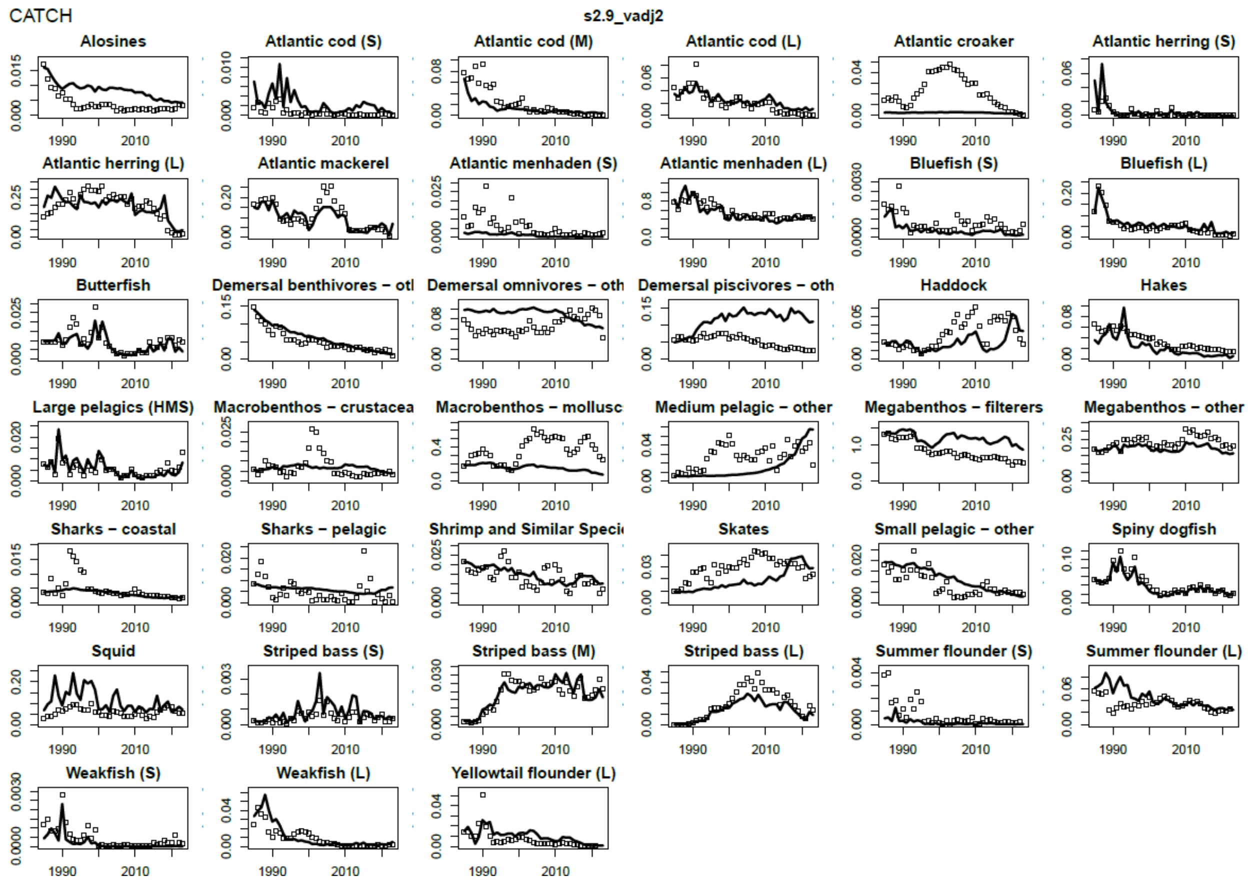


Figure 59. Catch fits for the NWACS-FULL model for the base run, sim 2.9. Points are the observed catches (mt/km^2) and lines depict predicted catch by year. Panels are labeled by trophic group code and numeric stanza. Trophic groups without observed, empirical data are excluded.

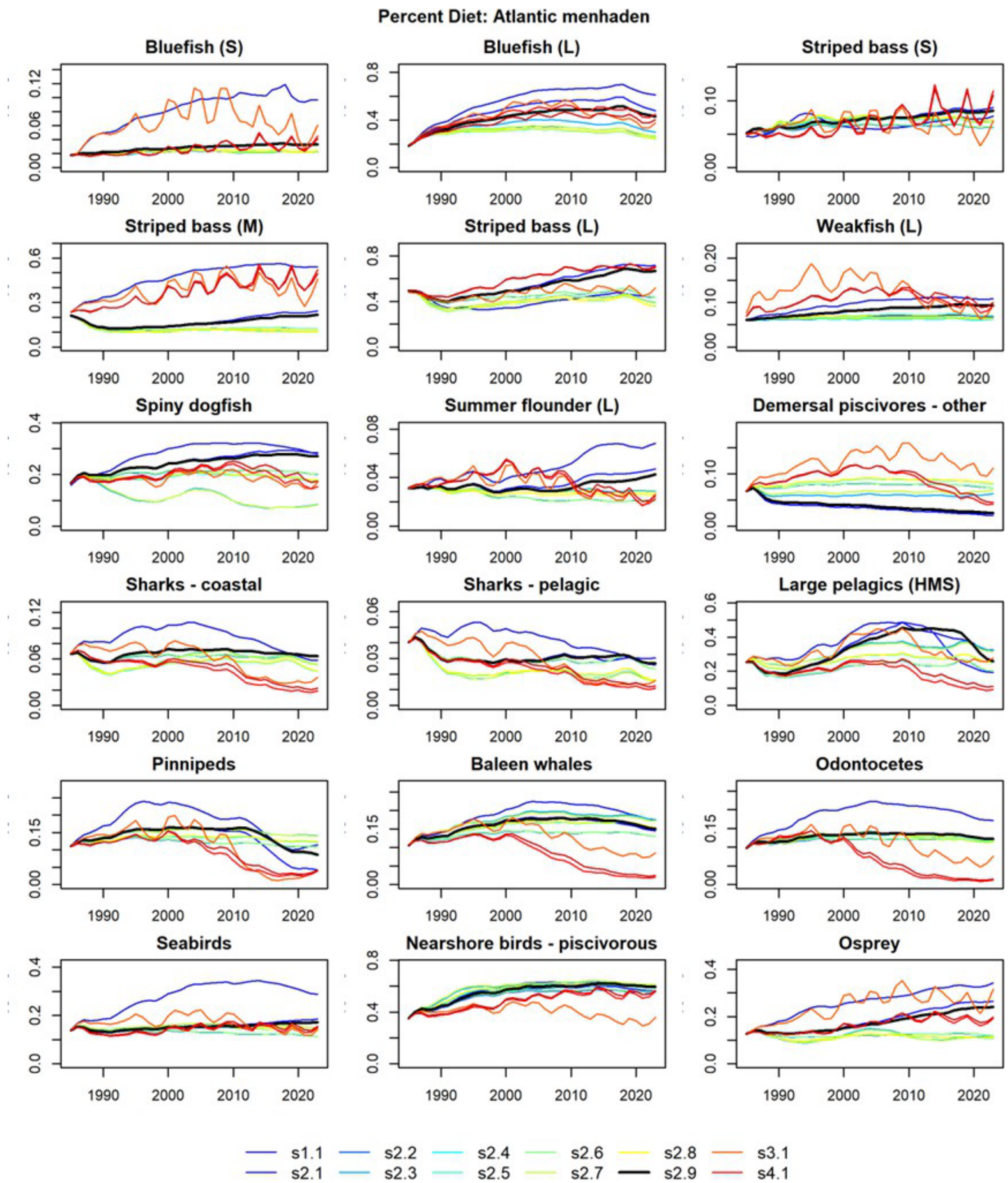


Figure 60. Annual contribution of Atlantic menhaden (all stanzas combined) to the diets of predators for the NWACS-FULL model. Only predators that had >1% of Atlantic menhaden in the input diet matrix are shown.

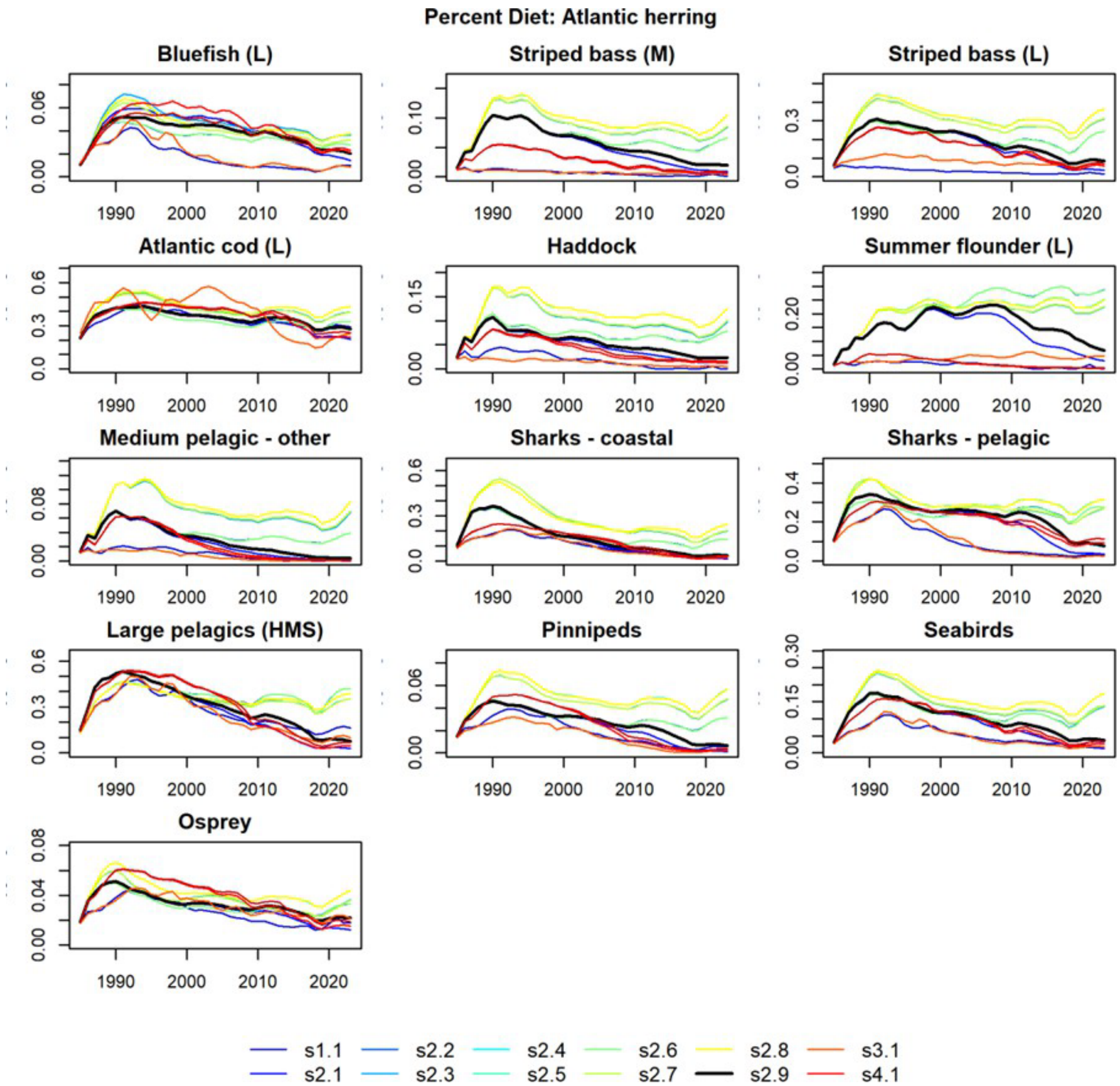


Figure 61. Annual contribution of Atlantic herring (all stanzas combined) to the diets of predators for the NWACS-FULL model. Only predators that had >1% of Atlantic herring in the input diet matrix are shown.

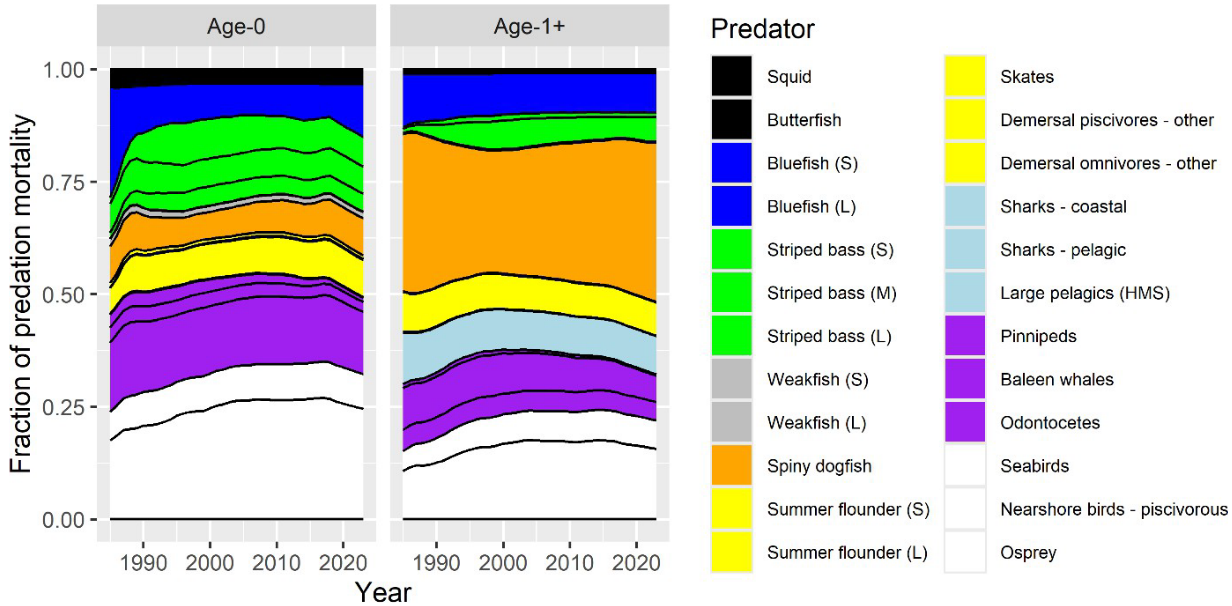


Figure 62. Predator contributions to age-0 and age-1+ Atlantic menhaden predation mortality (as a fraction of total M_2) by year. Results are based on sim 2.9 of the NWACS-FULL model. Predators and size classes are grouped by color and ordered as listed. Predator groups contributing the most to M_2 for colors that are not discernable are: Bluefish (L) – blue; demersal piscivores – yellow; large pelagics – light blue; nearshore piscivorous birds – white.

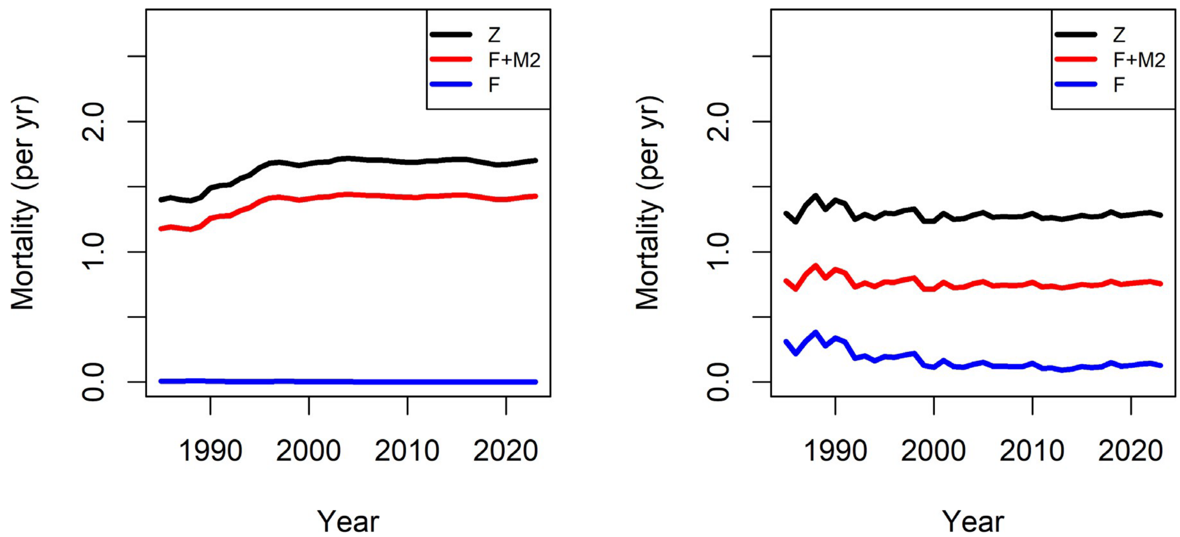


Figure 63. Instantaneous mortality rates for two age-classes of Atlantic menhaden (left: Small or age-0, right: large or age1+) based on sim 2.9 of the NWACS-FULL model.

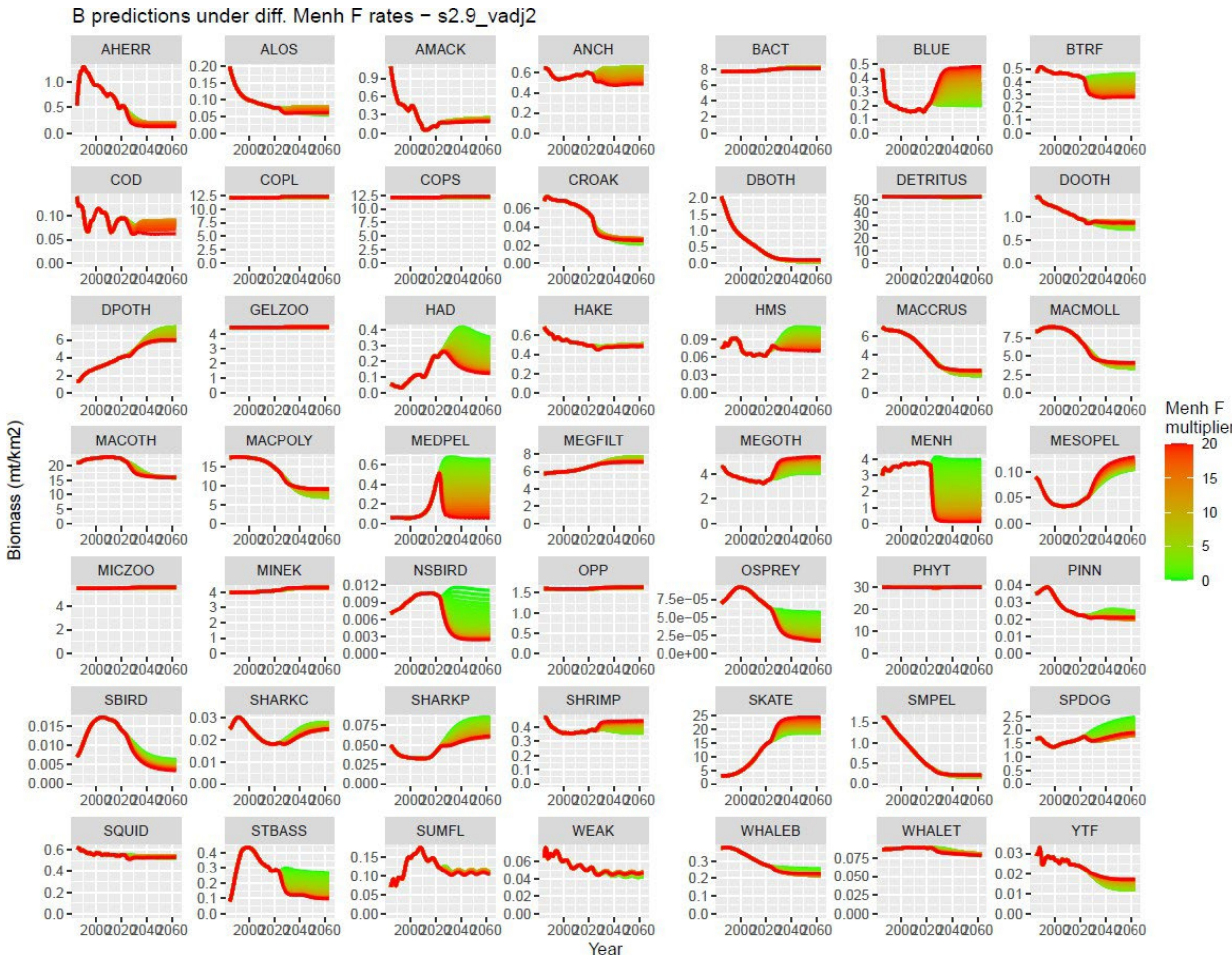


Figure 64. Biomass predictions for all model groups under different menhaden *F* rates while fishing the ERP species at their respective *F* targets from the NWACS-FULL model. Menhaden *F* rates were scaled from F_{2023} using an *F*-multiplier for each simulation.

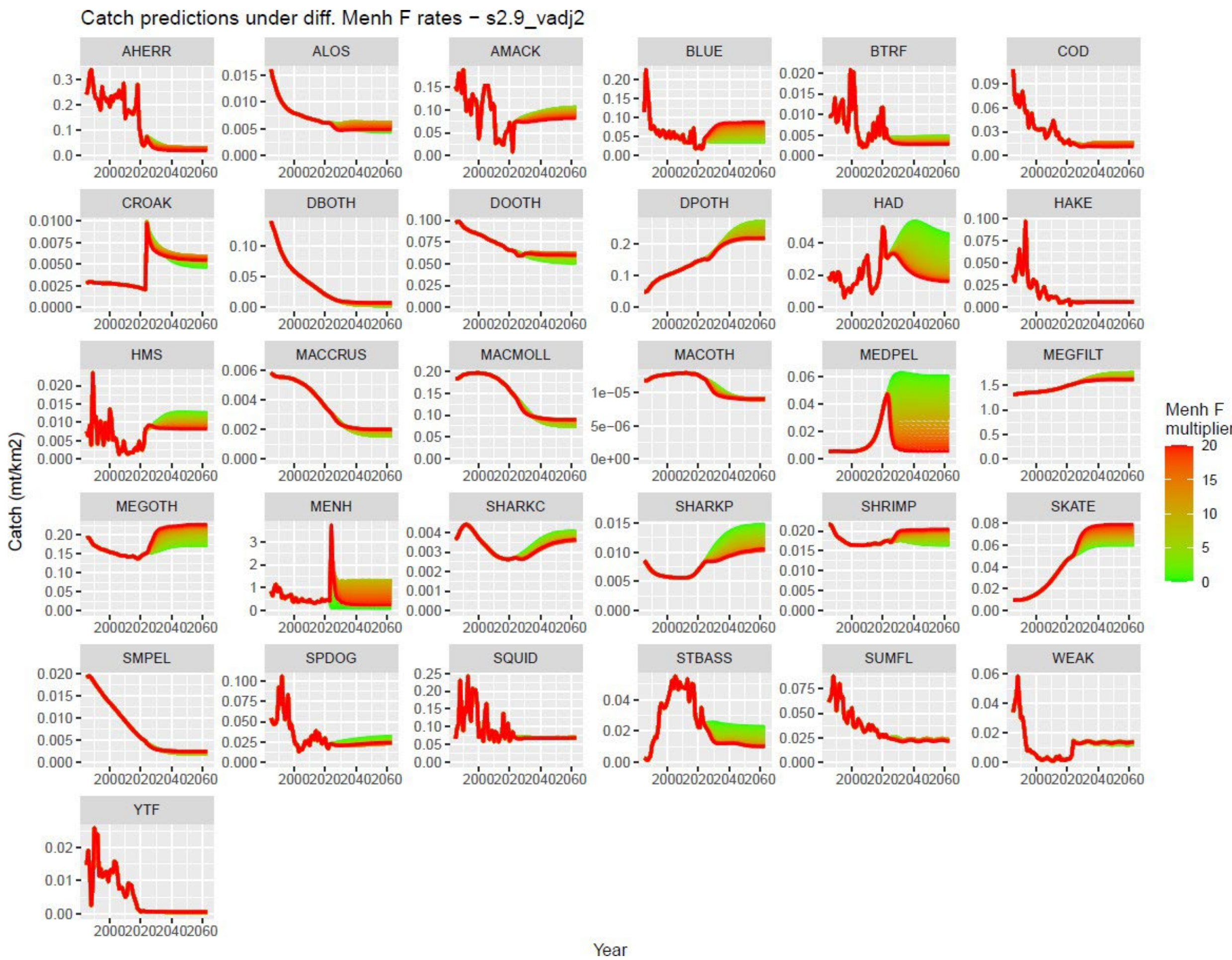


Figure 65. Catch predictions for all species under different menhaden *F* rates while fishing the ERP species at their respective F targets from the NWACS-FULL model. Menhaden *F* rates were scaled from F_{2023} using an *F*-multiplier for each simulation.

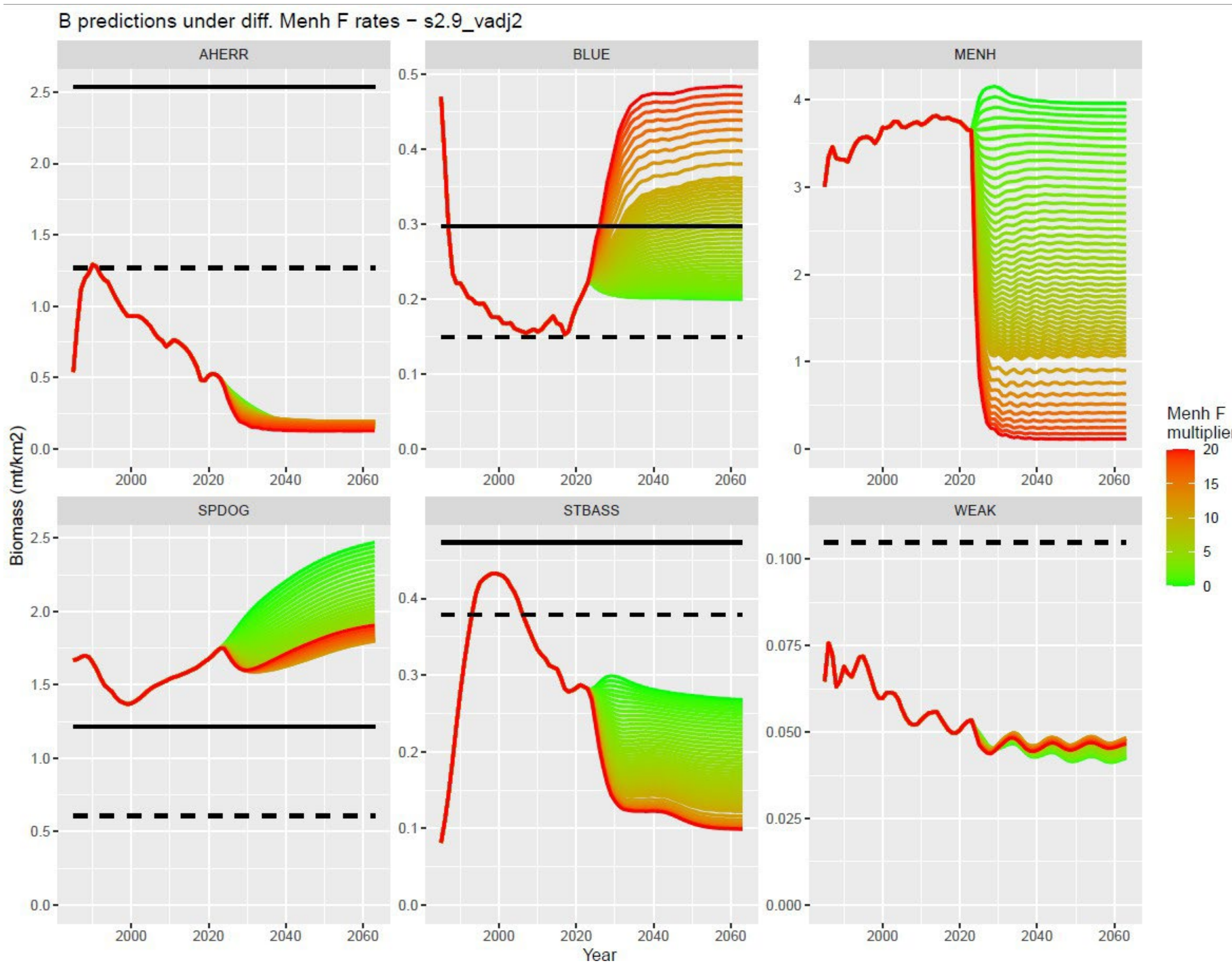


Figure 66. Biomass predictions for ERP species groups based on the NWACS-FULL model under different menhaden *F* rates. Projections were conducted holding ERP species at their *F* targets and all other groups and fisheries at status quo 2023 levels. Menhaden *F* rates were scaled from F_{2023} using an *F*-multiplier for each simulation. Black horizontal lines denote the Biomass thresholds (dashed) and targets (solid) for each species (as available).

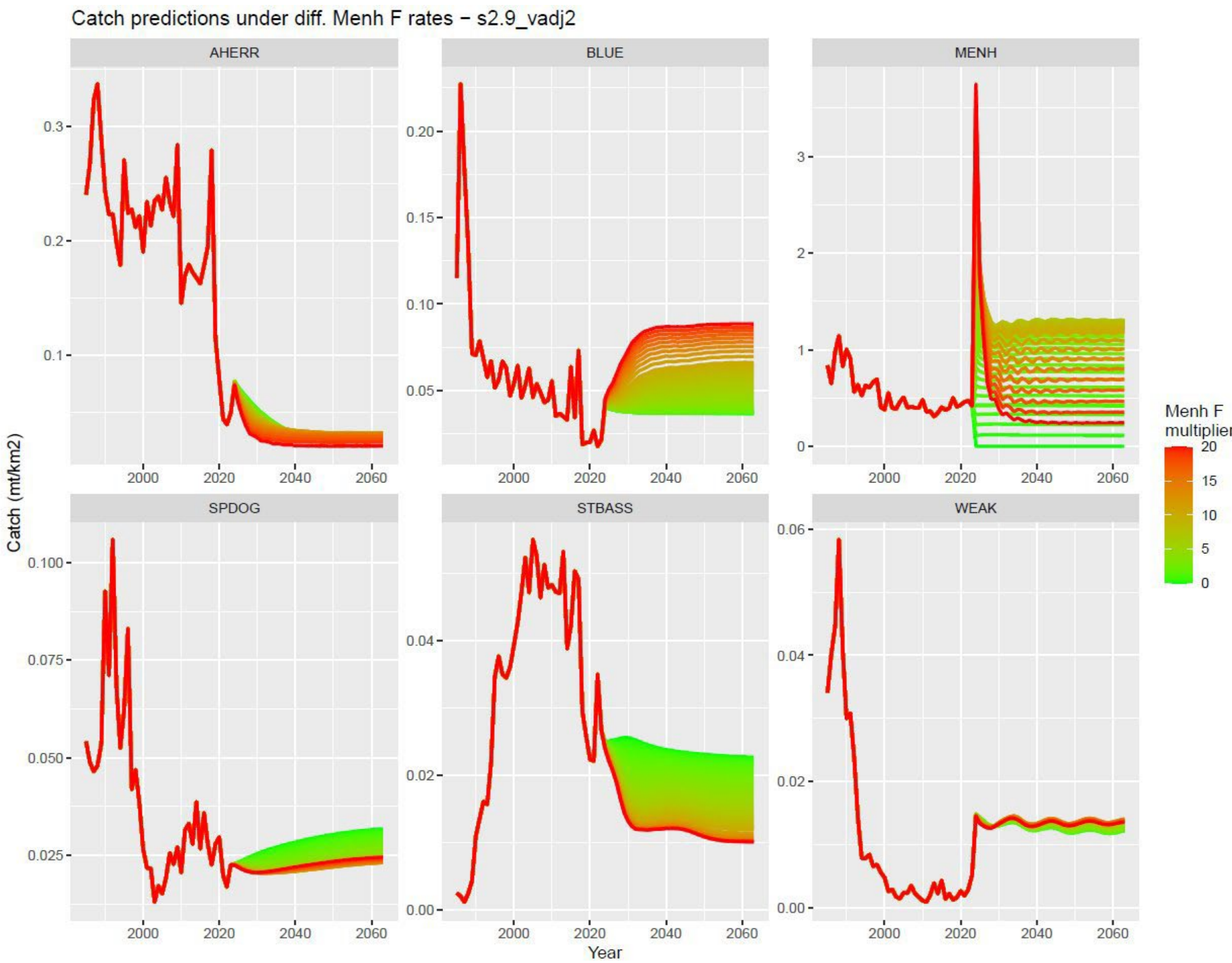


Figure 67. Catch predictions for all species under different Menhaden *F* rates while fishing the ERP species at their respective *F* targets from the NWACS-FULL model. Menhaden *F* rates were scaled from F_{2023} using an F-multiplier for each simulation.

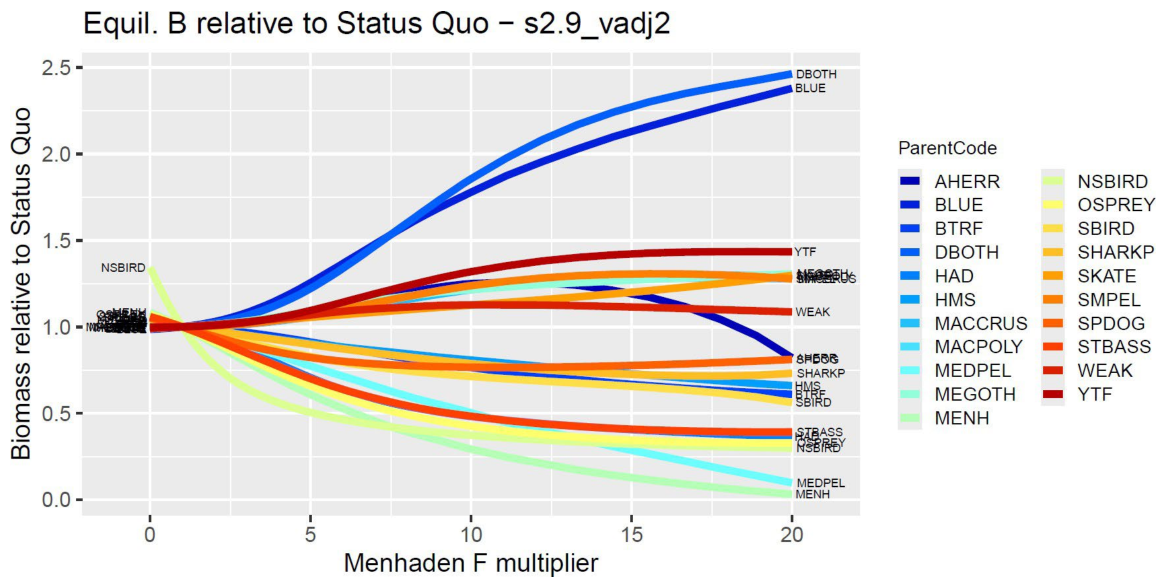


Figure 68. Effect of menhaden fishing on equilibrium biomass of select trophic groups (projected for 40 years) relative to their equilibrium biomass under status quo menhaden fishing rates from the NWACS-FULL model. ERP species were fished at their target *F* while menhaden *F* rates were scaled from 0 to 20 times the 2023 values using an *F*-multiplier. Biomasses for all species were summed across age-stanzas (if applicable). Lines are plotted for all ERP species and other trophic groups with non-negligible responses (>25% at F_{max}).

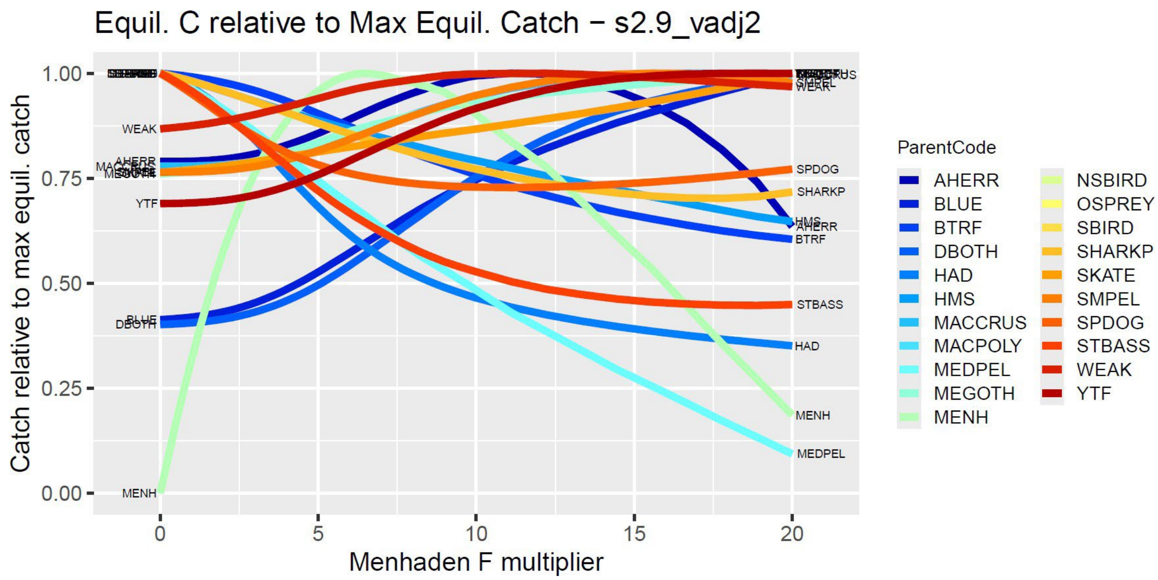


Figure 69. Effect of different menhaden fishing mortality rates on the equilibrium (40-yr) catch of selected trophic groups relative to the maximum equilibrium catch observed across all menhaden fishing scenarios from the NWACS-FULL model. Non-menhaden species were kept at their target *F* while menhaden *F* rates were scaled from 0 to 20 times the 2023 values using an *F*-multiplier. Catches for all species were summed across age-stanzas (if applicable). Lines are plotted for all ERP species and other trophic groups with non-negligible responses (>25% at F_{max}).

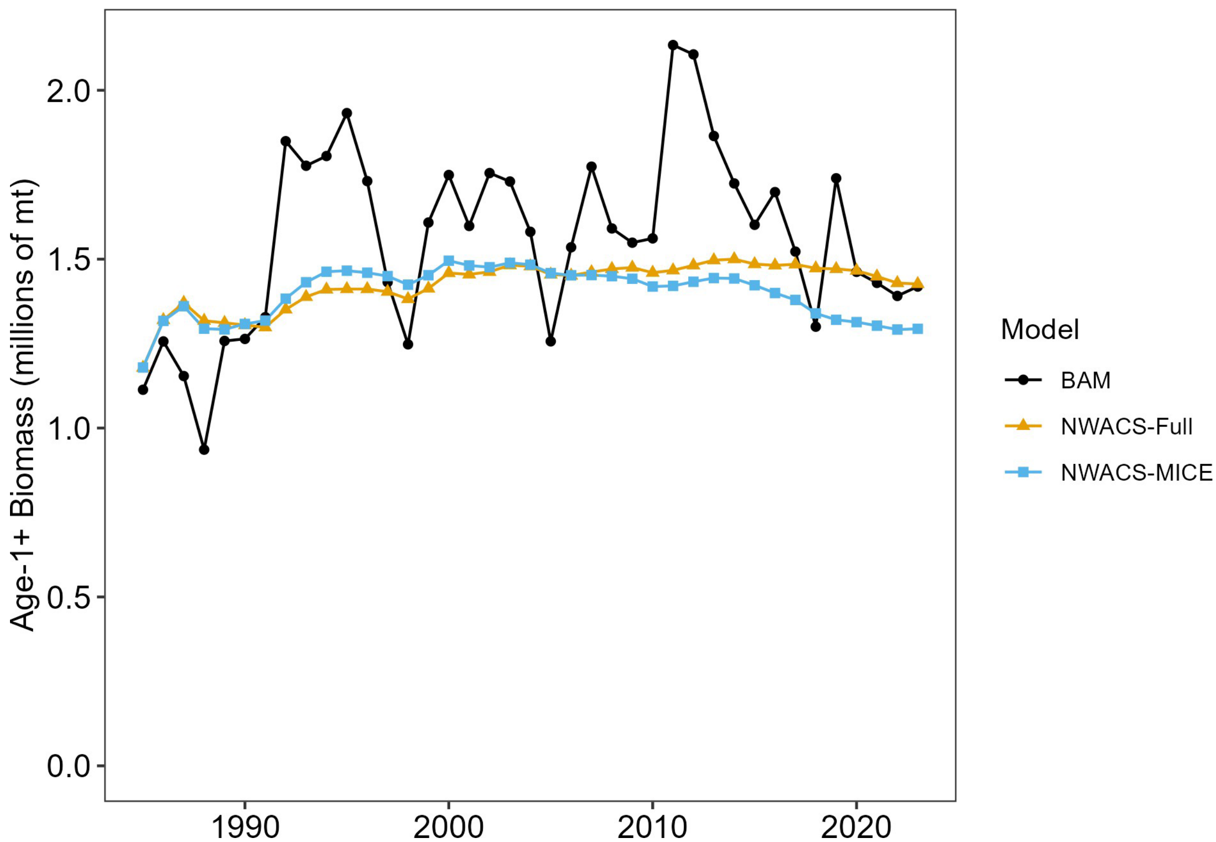


Figure 70. Age-1+ biomass of Atlantic menhaden from the single-species assessment (BAM) and the NWACS ecosystem models, 1985-2023.

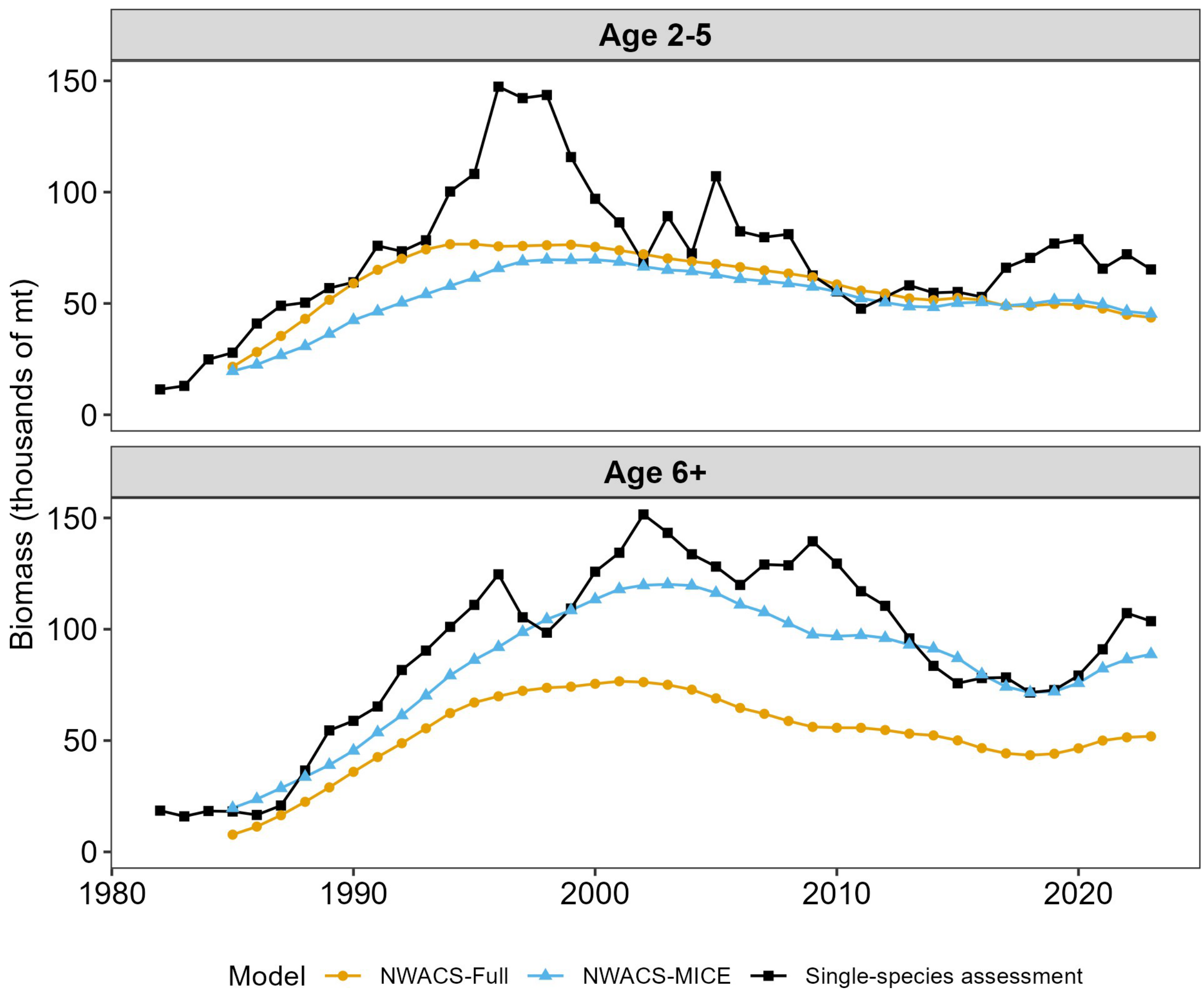


Figure 71. Comparison of striped bass biomass estimates for the non-recruit stanzas for the single-species assessment and the NWACS ecosystem models.

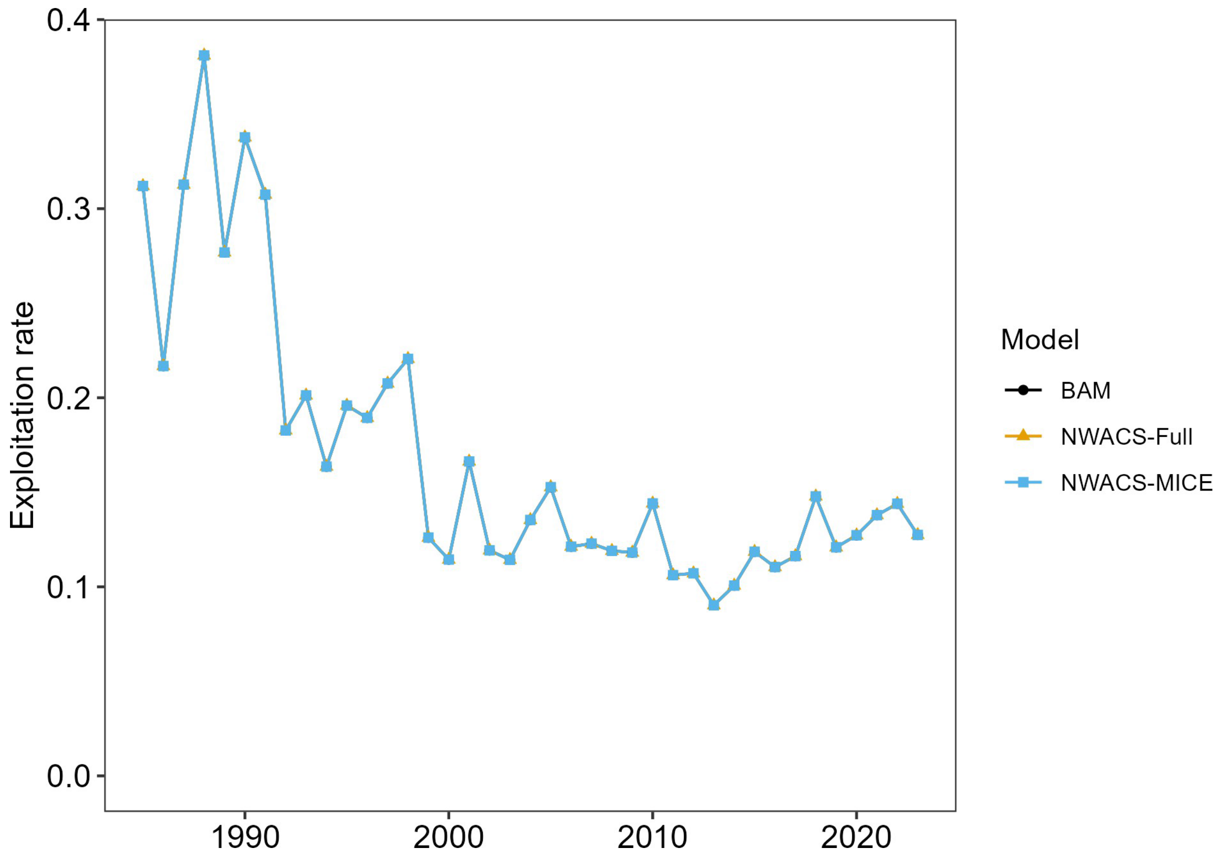


Figure 72. Age-1+ exploitation rates for Atlantic menhaden from the single-species model (BAM) and the NWACS ecosystem models.

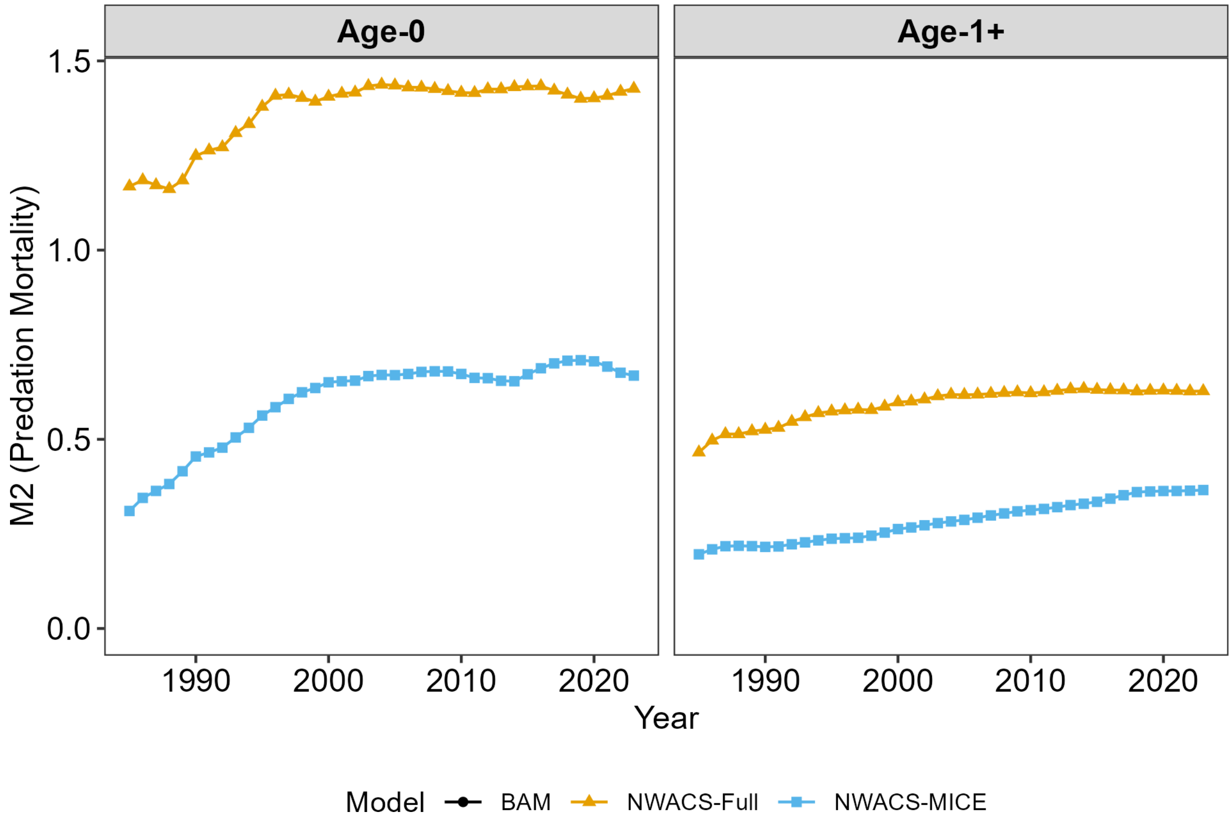


Figure 73. Estimates of M_2 (predation mortality) by stanza for Atlantic menhaden from the NWACS ecosystem models, 1985-2023.

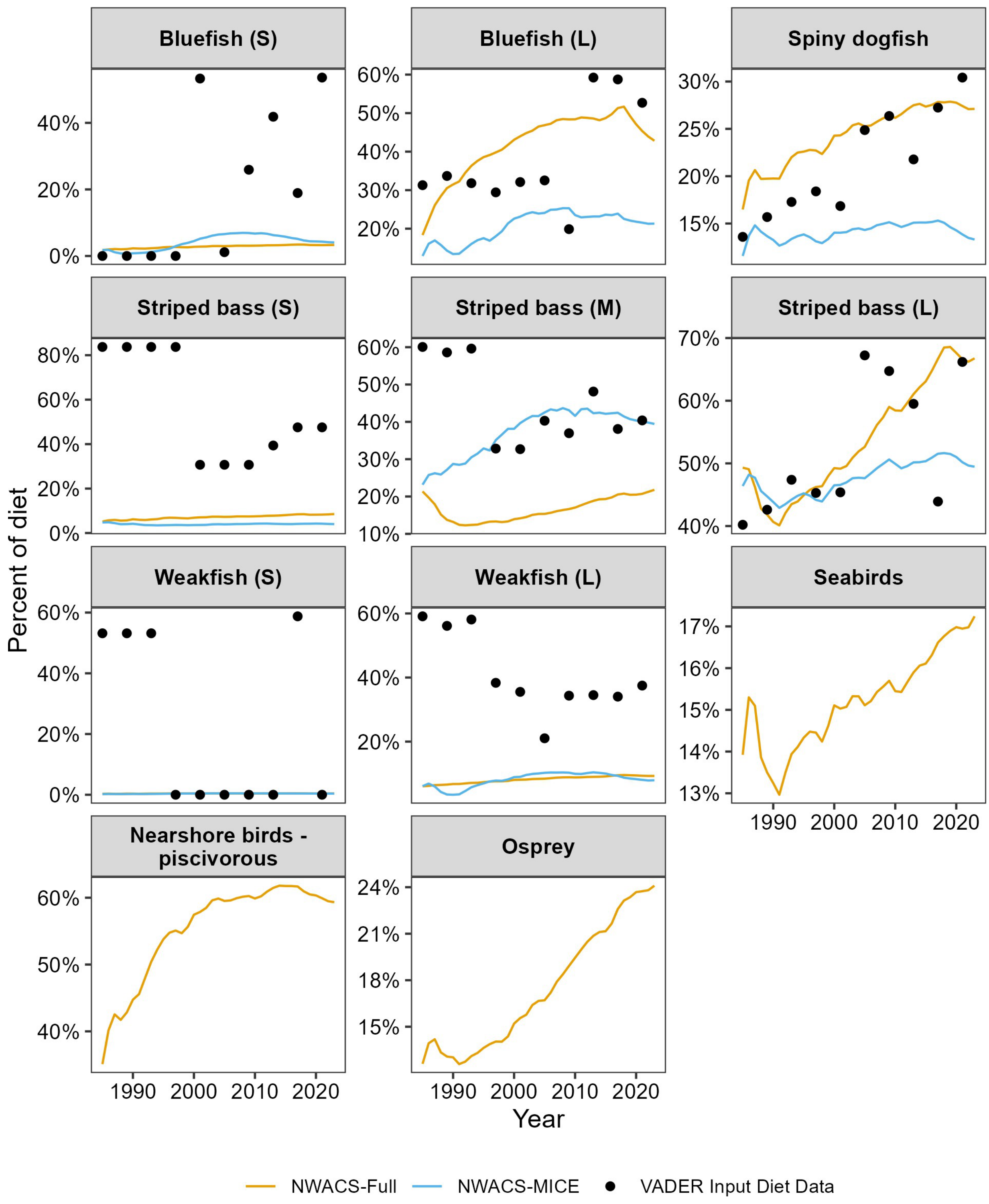


Figure 74. Annual predicted contribution of Atlantic menhaden (all stanzas combined) to the diets of ERP species and birds for the NWACS-MICE and NWACS-FULL models, plotted with the observed time-series of diet composition developed for the VADER model.

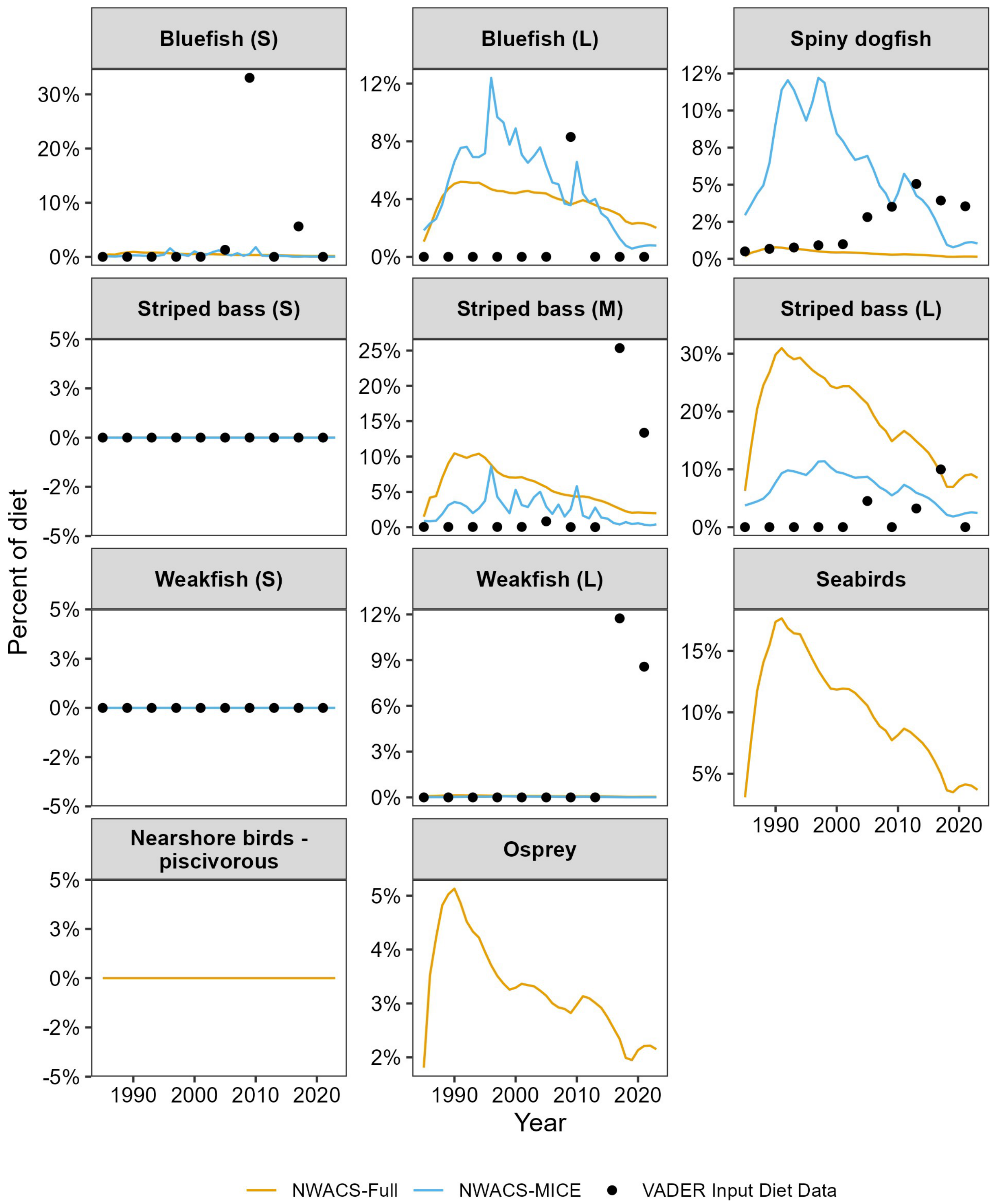


Figure 75. Annual contribution of Atlantic herring (all stanzas combined) to the diets of ERP species and birds for the NWACS-MICE and NWACS-FULL models, plotted with the observed time-series of diet composition developed for the VADER model.

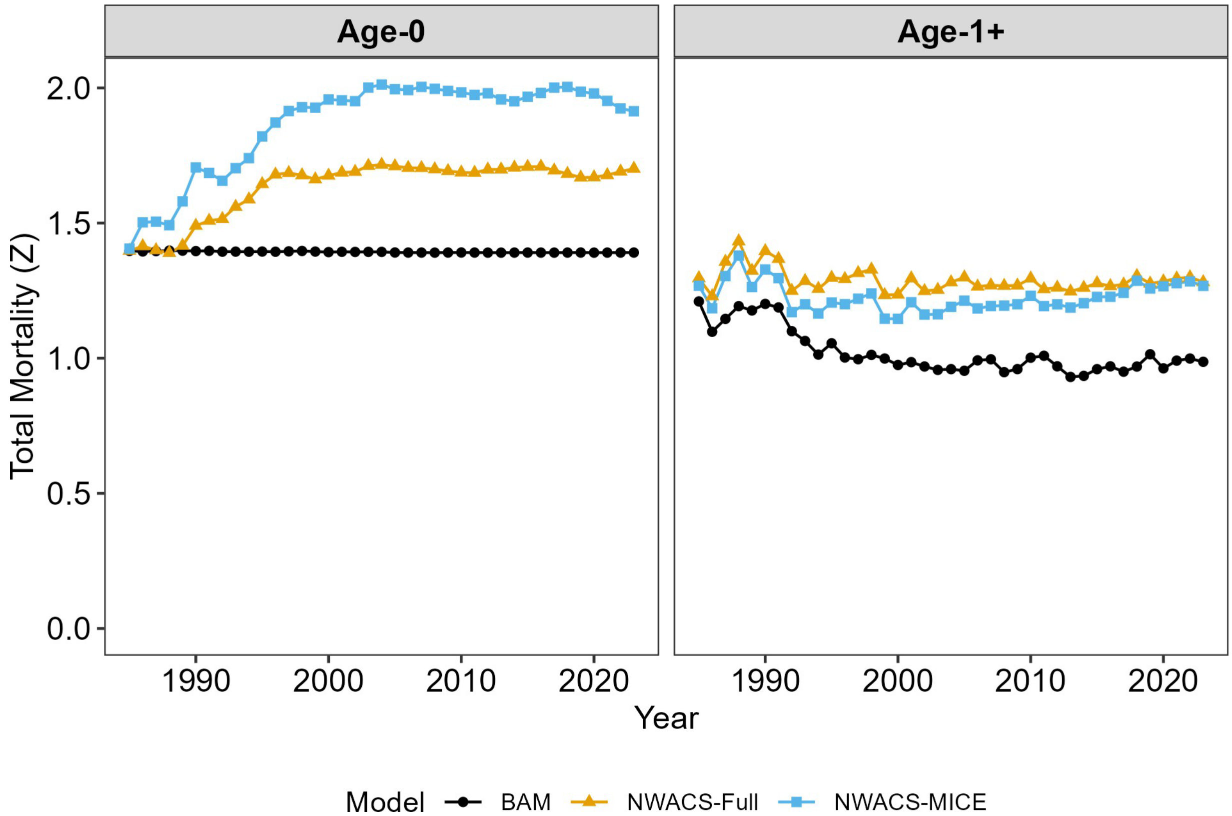


Figure 76. Estimates of total mortality (*Z*) for Atlantic menhaden by stanza from the single-species model (BAM) and the NWACS ecosystem models. Single-species *Z* estimates for age-1+ are biomass-weighted averages.

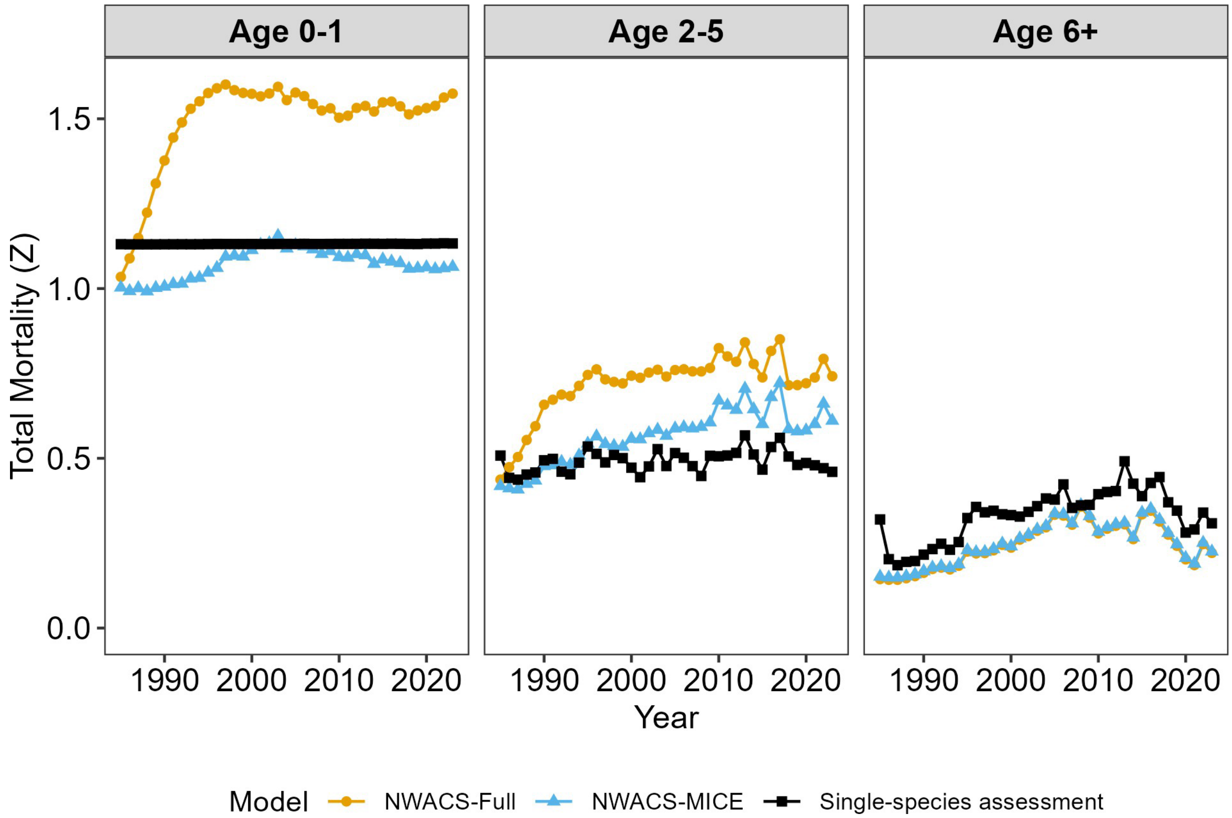


Figure 77. Estimates of total mortality (*Z*) for striped bass by stanza from the single-species model and the NWACS ecosystem models. The Age 0-1 stanza for the single-species model includes only age-1 fish; the *Z* estimates for the single-species model for the non-recruit stanzas are biomass-weighted averages.

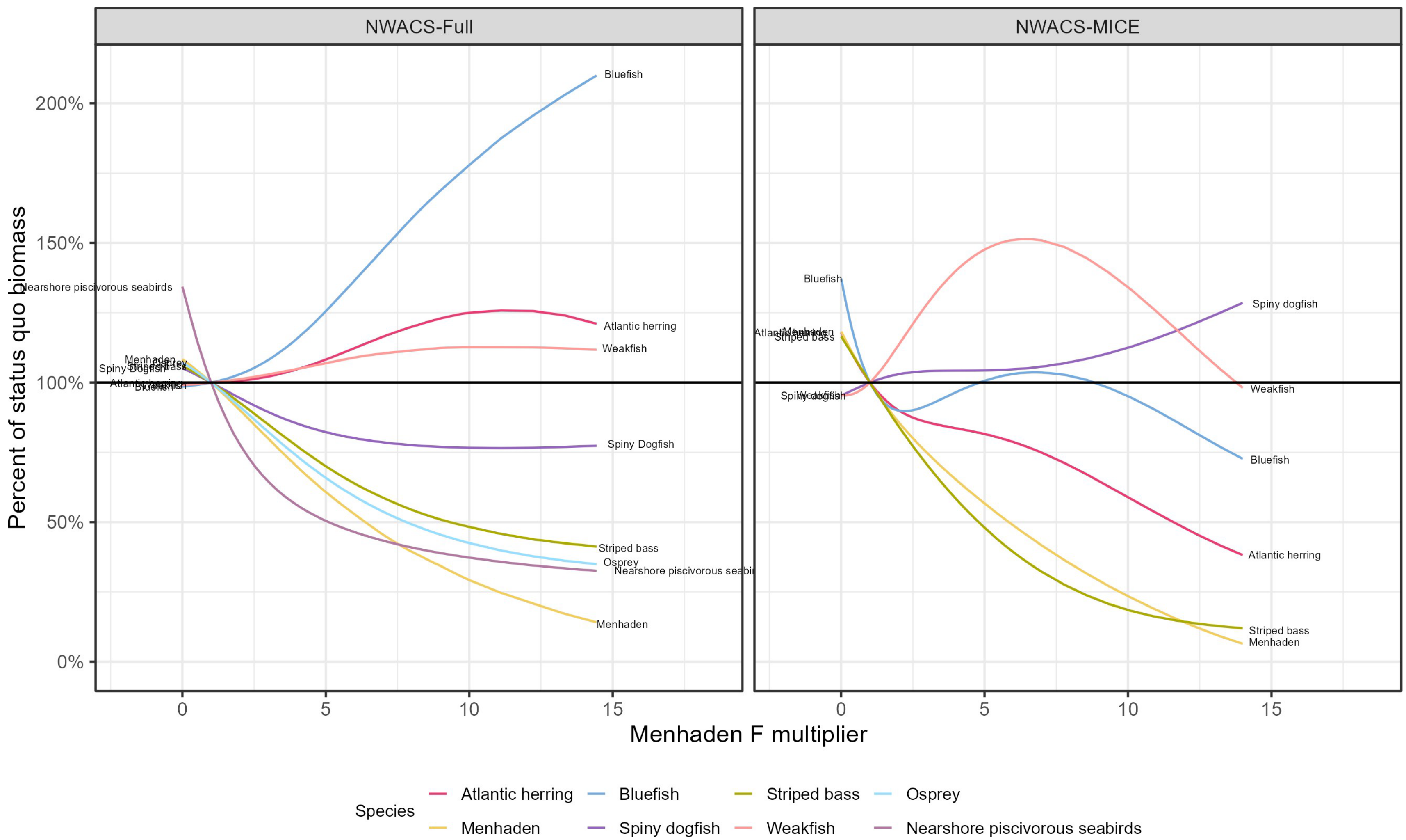


Figure 78. Long-term biomass of key species under different levels of Atlantic menhaden *F* relative to biomass under status quo *F* for the NWACS models.

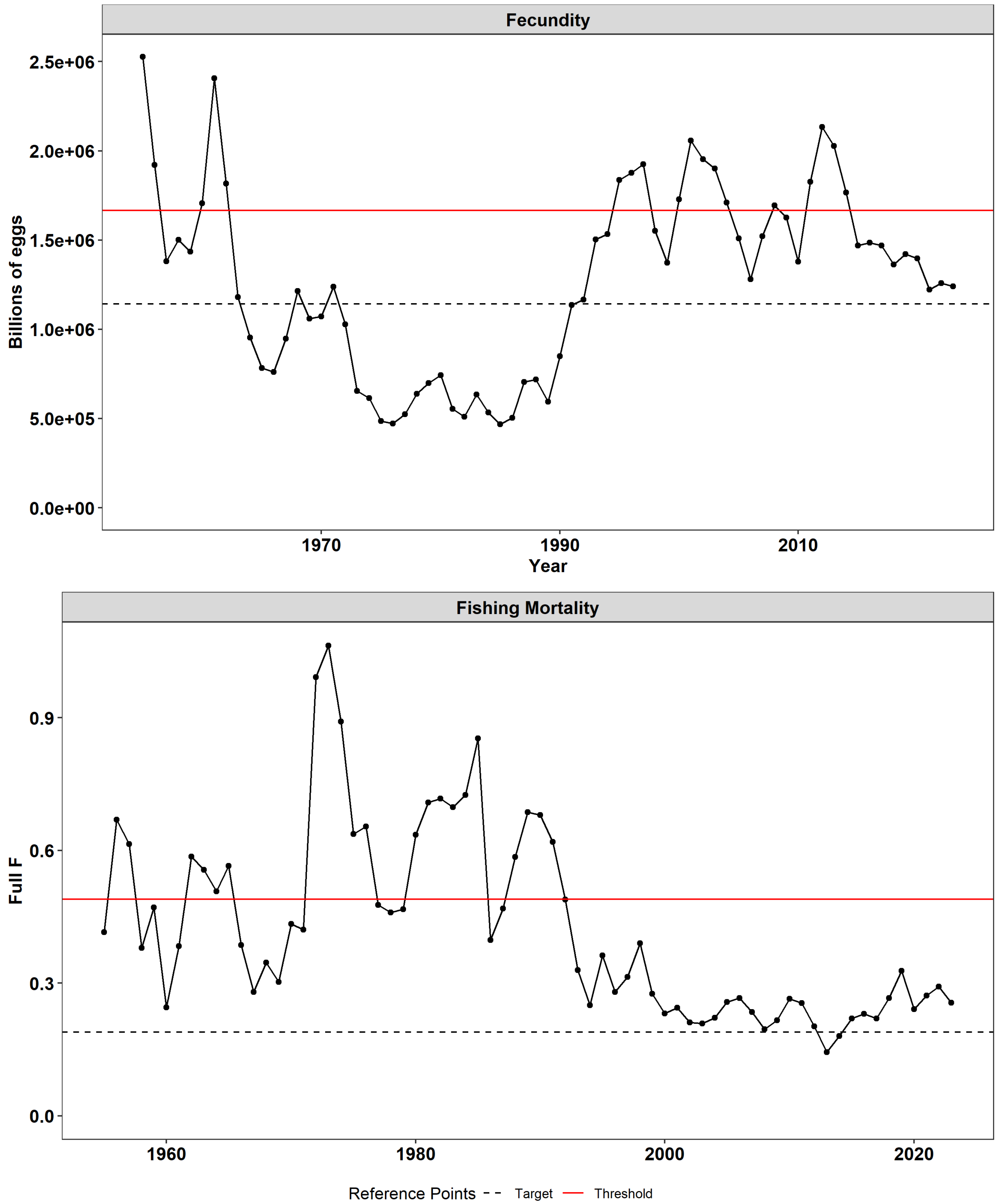


Figure 79. Atlantic menhaden fecundity (top) and fishing mortality (bottom) from the single-species assessment plotted with the proof-of-concept ERP target and thresholds from the 2025 ERP assessment.

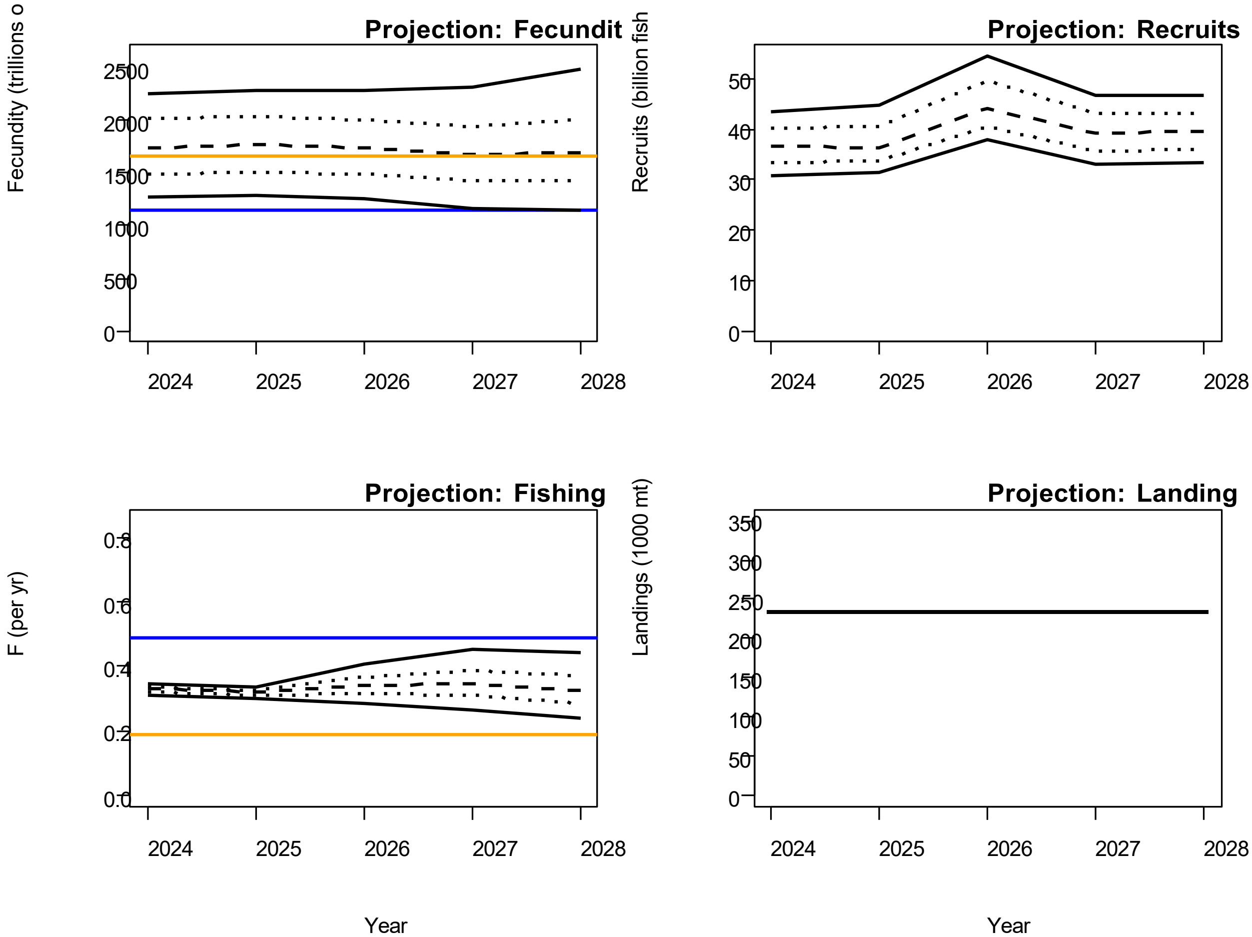
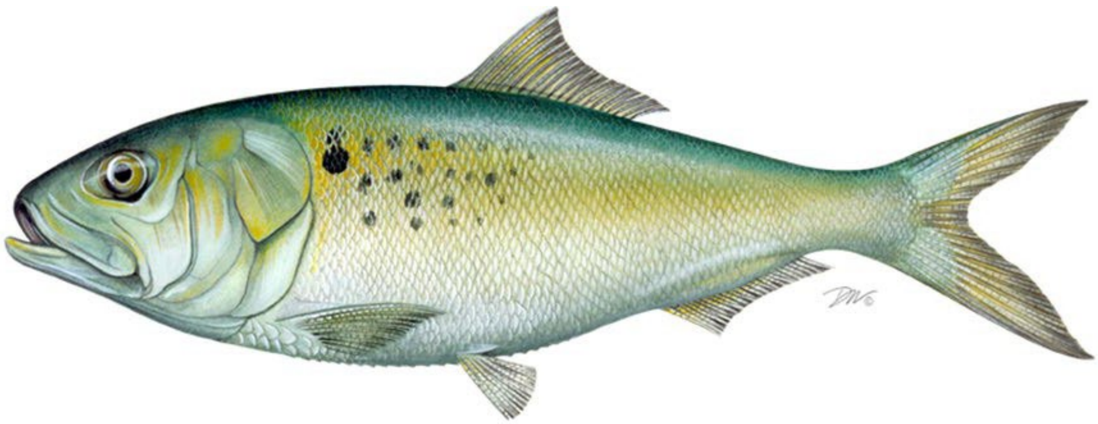


Figure 80. Fecundity, full fishing mortality rate, and recruits projected from 2024 to 2028 for a coastwide total allowable catch of 233,550 mt using the single-species model (BAM). The orange lines represent the proof-of-concept target fishing mortality rate and fecundity for the ecological reference points, while the blue lines represent the proof-of-concept threshold fishing mortality rate and fecundity for the ecological reference points. The dashed black line is the 50th percentile (median), the dotted black lines are the 25th and 75th percentiles, and the solid black lines are the 5th and 95th percentiles.

APPENDIX I: SINGLE-SPECIES ASSESSMENT UPDATE

Atlantic States Marine Fisheries Commission

2025 Atlantic Menhaden Stock Assessment Update



Sustainable and Cooperative Management of Atlantic Coastal Fisheries

Atlantic States Marine Fisheries Commission

Atlantic Menhaden Stock Assessment Update

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EXECUTIVE SUMMARY

The purpose of this assessment was to update the 2020 Atlantic Menhaden Single-Species Benchmark Stock Assessment (SEDAR 2020a) and recent stock assessment update (ASMFC 2022) with data from 2022-2023. The stock assessment update reran the peer-reviewed Beaufort Assessment Model (BAM) with a terminal year of 2023.

As part of the assessment process, the Atlantic Menhaden Stock Assessment Subcommittee (SAS) identified an error in the publication used to estimate the natural mortality rate used in the assessment. The SAS developed a revised estimate of M to use in the base run of the assessment, which was lower than the estimate used in the 2020 benchmark. This resulted in a lower estimate of biomass and fecundity and a higher estimate of fishing mortality over the time-series compared to the previous assessment update. These estimates were no longer directly comparable to the ecological reference points (ERPs) from the 2020 ERP benchmark assessment used in management, so the proof-of-concept ERPs developed during the 2025 ERP benchmark assessment were used to evaluate stock status for this report. The final ERP values used in management going forward may be different based on the outcome of the peer review and the decisions of the Atlantic Menhaden Management Board, in which case, this assessment update will be revised to use the final reference points.

Landings

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. The first coastwide total allowable catch (TAC) for commercial landings of Atlantic menhaden was implemented in 2013 and has changed in value depending on the most recent stock assessment and management document. Incidental catch and recreational harvest are not counted toward the TAC. The current TAC for the 2023 – 2025 fishing seasons is 233,550 mt. Reduction landings have been steady since the implementation of the TAC, while bait landings have increased, particularly in the northern states. For 2022-2023, reduction landings comprised about 70% of the coastwide landings. In 2023, bait and recreational landings were approximately 50,000 mt and reduction landings were approximately 131,800 mt.

Indices of Relative Abundance

The juvenile Atlantic menhaden index developed from 16 fishery-independent surveys showed the highest young-of-year abundance occurred during the 1970s and 1980s. Abundance has been lower since the 1990s with some moderate increases in the mid-2000s, 2016, and 2021-2023.

Three coastwide indices of adult abundance were developed from eight fishery independent survey data sets: northern (NAD; age-2+), Mid-Atlantic (MAD; age-1+), and southern (SAD; age-1) adult indices. The NAD indicated that age-2+ relative abundance has been variable, but abundance was high in 2012 and 2019-2022 before declining again in 2023. The MAD showed high relative abundance in the late 1980s and then variable abundance with peaks in recent years, including 2022 before declining again in 2023. The SAD indicated that age-1 abundance was high in 1990 and then declined through the 1990s. Abundance peaked again in 2006 and then remained variable with low abundance in the terminal years.

Fishing Mortality

Highly variable fishing mortalities were noted throughout the entire time series and are dependent upon fishing and management policies, as well as stock biomass. The fishing mortality rate was highest in the 1970s and 1980s and has been relatively stable since the early 2000s.

Biomass

Age-1+ biomass has fluctuated over time with a time-series high in 1959 and a time-series low in 1973. During the 1990s, age-1+ biomass increased and has remained relatively stable over the past decades.

Fecundity

Population fecundity (i.e., number of maturing ova), used as a measure of spawning potential, was highest in the early 1960s, low in the 1970s and 1980s, and high again from the 1990s to the present. The largest values of population fecundity were in 1955, 1961, and 2012. Fecundity estimates have been declining since the high in 2012.

Stock Status

Based on the proof-of-concept ERPs developed for the 2025 ERP benchmark assessment, the Atlantic menhaden population is not overfished and overfishing is not occurring. The fishing mortality rate is just above the target and well below the threshold value, while the fecundity is below the target but above the threshold.

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INTRODUCTION

This Terms of Reference (TOR) report describes the update to the single-species benchmark stock assessment for Atlantic menhaden (SEDAR 2020a). The benchmark was updated in 2022 (ASMFC 2022) to extend the fishery-independent and -dependent data for Atlantic menhaden through 2021, rerun the peer-reviewed Beaufort Assessment Model (BAM), and determine stock status of Atlantic menhaden using the ecological reference points (ERPs) defined in SEDAR (2020b) and accepted for management use in 2020. This update further extends the data, model, and assessment through 2023. This update includes a revised estimate of M, which will be peer-reviewed through the concurrent 2025 ERP benchmark assessment, and uses preliminary, proof-of-concept ERPs based on the Board's definition of the ERP target and threshold to determine stock status.

TOR 1. Fishery-Dependent Data

Update fishery-dependent data (landings, discards, catch-at-age, etc.) that were used in the previous peer-reviewed and accepted benchmark stock assessment.

The commercial reduction, commercial bait, and recreational landings time series were extended from the previous assessment (SEDAR 2020a; ASMFC 2022) through 2023, along with the associated age compositions from the reduction and bait fisheries. For use in the BAM, landings were split into northern and southern regions as defined by waters north and south of Machipongo Inlet, Virginia, where the Chesapeake Bay is in the southern region.

Reduction landings were provided by the NOAA Fisheries Beaufort Lab. Reduction landings in the southern region have been increasing since the last assessment update while the northern reduction landings were decreasing. Southern landings are consistently larger than those in the north (Figure 1). Total reduction landings in 2023 were 131,800 mt.

Bait landings from 1955-1984 were compiled from historic records and considered incomplete, whereas bait landings for 1985-2023 were validated with the states by the Atlantic Coastal Cooperative Statistics Program (ACCSP). Bait landings in the north increased in recent years and were over twice as much as bait landings in the south for the last four years (Figure 2). Total bait landings were relatively constant for 2019-2022, averaging 57,140 mt, but decreased in 2023, in both the north and south, to 48,550 mt.

The Marine Recreational Fisheries Statistics Survey (MRFSS, 1981-2003) and the Marine Recreational Information Program (MRIP, 2004-2023) data sets were used to derive a time series of recreational landings of Atlantic menhaden. The uncertainty associated with recreational estimates for Atlantic menhaden is high and the landings are variable but have increased approximately 2-3 times in the last decade compared to earlier years. (Figure 3). For use in the BAM, recreational harvest, which comprises less than 1% of coastwide harvest, was added to the bait landings.

Coastwide reduction landings have remained relatively steady since 2000 with bait landings increasing over time, comprising 27% of coastwide landings in 2023 (Figure 4).

Commercial reduction and bait catch-at-age matrices were developed from the available biological data collected in each fishery by region. Age proportions of the bait catch were applied to the MRIP estimates of recreational catch and pooled with the bait catch-at-age.

See Appendix for supplemental tables (Table A1 – Table A5) for TOR 1.

TOR 2. Fishery-Independent Data

Update fishery-independent data (abundance indices, age-length data, etc.) that were used in the previous peer-reviewed and accepted benchmark stock assessment.

Sixteen fishery-independent surveys from Rhode Island to South Carolina were used to develop young-of-year (YOY) abundance indices, which were then combined into a coastwide index of relative YOY abundance using the Conn method (Conn 2010; Table 1). Eight fishery-independent surveys from Connecticut to Georgia were developed into age 1+ abundance indices and were combined into three regional adult surveys: a northern adult index (NAD), a Mid-Atlantic adult index (MAD), and a southern adult index (SAD). Several surveys were affected by the COVID-19 pandemic and had no or limited sampling in 2020 and 2021 (Table 1).

The coastwide YOY index of relative abundance for Atlantic menhaden indicated high abundance in the 1970s and 1980s, with declines through the 1990s before stabilizing at pre-1970s levels (Figure 5). YOY abundance remained low but there was a slight increase in the terminal years of 2021-2023. The NAD index predicted variable abundance throughout the time series with high abundance occurring in the recent years of 2019-2022 before declining again in the terminal year of 2023 (Figure 6). The MAD index predicted higher than average abundance in the beginning of the time series followed by a lower but variable abundance through the late 1990s-early 2010s (Figure 7). Abundance in the Mid-Atlantic region began to increase in the mid-2010s but then decreased and was variable through the terminal years with 2020 representing a time series low. Abundance increased in 2021-2022 but declined in 2023. The SAD index predicted high abundance in 1990 followed by low abundance through the mid-2000s (Figure 8). The index peaked again in 2006 but then decreased and was variable through the terminal year. The terminal years of 2022-2023 both indicated relatively low abundance in the region.

For the adult indices, length compositions were developed by combining data from each of the surveys and weighting the data by the inverse of the squared sigma values outputted from the Conn method.

An index of Atlantic menhaden spawning biomass was developed using larval abundance data collected from two regional ichthyoplankton surveys (MARMAP and EcoMon; Figure 9). The index increased in the last few years to an EcoMon time series peak in 2019, after which it started to decline again. The index was updated through 2022, although data from 2021 were not available due to COVID. This index was included in the base run of the assessment model in SEDAR 2020a but was excluded from the 2022 update and this update's base run due to issues with model fitting (ASMFC 2022).

See Appendix for supplemental tables (Table A6 – Table A7) and figures (Figure A1- Figure A4) for TOR 2.

TOR 3. Life History Information and Model Parameterization

Tabulate or list the life history information used in the assessment and/or model parameterization (M, age plus group, start year, maturity, sex ratio, etc.) and note any differences (e.g., new selectivity block, revised M value) from benchmark.

Tabulated life history information and model inputs can be found in Table 2. The benchmark assessment was updated with all available data through the terminal year of 2023. The same time blocks for catch selectivity estimations used in SEDAR 2020a were used in this update. Since the last assessment (SEDAR 2020a), the fecundity information was updated by the Virginia Institute of Marine Science (R. Latour and J. Gartland, VIMS, unpublished data; Latour et al 2023) using the same methods as was used for the benchmark.

Three changes were made to the updated run from the benchmark assessment during the 2022 assessment update which were carried through to this update:

1. Censoring of the MARMAP/EcoMon (MARECO) ichthyoplankton index;
2. Censoring of the commercial bait south age compositions for 2020;
3. The inclusion of penalties on some of the selectivity parameters that were hitting bounds during the estimation process.

These changes to the assessment update were considered thoroughly during the last assessment update and were discussed thoroughly in that documentation (ASMFC 2022). Briefly, the quality and quantity of data during the COVID-19 pandemic years caused some problems with estimation of parameters and the determination of year-class strength (recruitment). This update assessment retained the same method of recruitment estimation as used during the benchmark assessment. There is no formal stock-recruitment structure, rather median recruitment is estimated along with annual recruitment deviations from that median for the duration of the time series.

The only new change for this update assessment is the inclusion of a new vector of natural mortality based on a revised analysis of the historical tagging data that was completed by the M working group. The 2020 benchmark assessment used the estimate of M from Liljestrand et al.'s (2019) analysis of the tagging data to scale the Lorenzen (1996) curve of M-at-age, assuming that the M estimated from the tagging data represented the M for age-1.5 menhaden, based on the size of the tagged fish. During the 2025 benchmark assessment process, Ault et al. (2023) submitted a working paper to the Atlantic menhaden SAS and the Ecological Reference Points Work Group (ERP WG) that re-analyzed the historical tagging data and produced an estimate of M = 0.56, significantly lower than the M = 1.17 reported by Liljestrand et al. (2019).

However, Ault et al. (2023) had used a different subset of the data and a different approach to handling key parameters, which made direct comparisons with Liljestrand et al. (2019) difficult. The SAS formed a working group to review the datasets and methods in consultation with the primary authors to determine the best estimate of M for use in the Atlantic menhaden stock assessment. The M WG and SAS determined that the main cause of the difference in the M estimates was the handling of the magnet efficiency parameter, which was equivalent to the tag reporting rate in conventional tagging models. The M WG and SAS found that Liljestrand et al. (2019) had overestimated the magnet efficiency rate in their analysis, but did not agree with the stepwise estimation approach proposed by Ault et al. (2023) to estimate this parameter. In the end, the M WG and SAS recommended a revised estimate of M = 0.92 from the tagging study, based on the corrected magnet efficiency rate and updated effort and landings datasets, which was lower than the value used in the 2020 benchmark, but higher than the value estimated from Ault et al.'s (2023) method. This revised estimate of M was used to scale the Lorenzen (1996) curve to develop M-at-age estimates so that the estimate of M-at-age for age-1.5 was equal to the estimate from the tagging model, based on the size of the tagged fish, as was done for the benchmark (Table 2). The estimate developed using the Ault et al. (2023) stepwise approach was used as a sensitivity run (Table 2). See the working paper SEDAR 102 WP-01 for a full description of the data, methods, and M WG findings.

TOR 4. Updated Beaufort Assessment Model

Update accepted model(s) or trend analyses and estimate uncertainty. Include sensitivity runs and retrospective analysis if possible and compare with the benchmark assessment results. Include bridge runs to sequentially document each change from the previously accepted model to the updated model.

In order to bridge from the benchmark assessment to the current updated assessment with the new M vector, we provided bridging runs including the benchmark assessment, the 2022 update assessment, and this update assessment both with the old natural mortality vector and the new vector.

In general, the updated base run assessment is similar to the 2020 benchmark assessment, the 2022 update assessment, and the continuity run for this assessment with main differences being in the scale of this assessment given the difference in the scale of natural mortality. Generally, the trends over time are similar across metrics, and the largest change is in the estimation of mean recruitment from the time series, which is an expected change. The model fit well to the landings for all four fleets. In general, the patterns in the age compositions were random and did not exhibit any patterning. The fits to the indices were similar to the fits during the benchmark assessment and did not have runs in residuals. The fits to the NAD and MAD length compositions were also similar to the fits during the benchmark assessment. Selectivity for the fisheries and the indices were similar to the last assessment.

The fishing mortality rate (*F*) increased slightly in 2022 and then decreased again in 2023 and has been relatively stable since 2000 (Figure 10). The recruitment classes for 2022 and 2023 appear to be slightly larger than average over the last two decades (Figure 11). However, the model does have difficulty estimating year-classes in the terminal year of the model, as evidenced by the 2022 update to the benchmark assessment. Age-1+ biomass increased slightly during the last two years but is still below average for the last few decades (Figure 12). Finally, fecundity has been stable during the most recent years (Figure 13).

A sensitivity run was completed to show how an alternative natural mortality estimate impacted assessment outcomes. In general, natural mortality is one of the components in stock assessments that is the most uncertain. However, in the case of Atlantic menhaden the SAS has an extensive tagging study that addressed many assumptions for use in estimating the scale of natural mortality.

The sensitivity run with the lower values of M estimated by Ault et al. (2023) resulted in very similar fits to all of the indices of abundance. The largest differences between the base run and the sensitivity run with the lower M values were the estimates of the recruitment time series and the full fishing mortality rate time series, both of which scaled with assumptions about natural mortality. In general, natural mortality scales an assessment, along with landings, to give an indication of the overall mortality, Z, and thus the fishing mortality. In addition, the recruitment estimates will also scale to the appropriate level associated with the anticipated mortality rates and the catch levels. One interesting response for this sensitivity run compared to the base run was that the geometric mean fishing mortality rate was the same for both runs from the 1990s to the present. This occurred because the geometric mean fishing mortality rate is focused on age-2 to -4, and the proportion of older aged individuals was increasing in the population causing reduced fishing mortality for older ages, which was in line with the base run values.

A retrospective analysis was also completed for the update assessment. A series of runs were done removing the terminal year data in sequence. The update assessment had a terminal year of 2023, and the retrospective analysis was run back through a terminal year of 2018. Overall, the retrospective runs fall within the uncertainty bounds from the uncertainty analysis. The fits to the indices for the retrospective runs are very similar to the base run. All of the retrospective plots have good overlap in the estimated historical values across fishing mortality, fecundity, and recruitment. In general, the recruitment retrospectives did a good job estimating terminal year recruitment values, especially 2022 and 2021, which were the values estimated in the base run with the terminal year of 2023. The geometric mean fishing mortality rate and the fecundity values were generally estimated close to the base run, but the terminal geometric mean fishing mortality was generally lower in all years of the retrospective than the base run while fecundity was generally higher.

A Monte Carlo bootstrap (MCB) uncertainty analysis was completed as was done for the last benchmark assessment. The configuration was kept exactly the same with uncertainty in natural mortality and fecundity. The range of uncertainty surrounding natural mortality was updated to reflect the tagging reanalysis. Specifically, the range of natural mortality estimates for age-1.5 was [0.83, 0.97]; the Lorenzen curve for the age-varying M for each run was rescaled to the estimate of age-1.5 M drawn for that run. A total of 5,000 runs were completed. Some runs were excluded due to large gradients, leaving 4,734 MCB runs for analysis. Overall, the uncertainty was much narrower for all the metrics of interest when compared to the last update assessment and the benchmark assessment. During the benchmark and update assessments, the MCB analyses provided outcomes that were bimodal in nature. With this update, that bimodality was reduced substantially.

See Appendix for supplemental tables and figures for TOR 4: model fits to landings (Figure A5 - Figure A8) and associated age comps (Figure A9 - Figure A16), model fits to indices (Figure A17 - Figure A20) and associated length comps (Figure A21 - Figure A24), estimated selectivities (Figure A25 - Figure A30), model estimated *F*, recruitment, biomass, and fecundity (Figure A31 -

Figure A38), bridge runs (Figure A39 - Figure A46), sensitivity runs (Figure A47), and the retrospective analysis (Figure A55 - Figure A62).

TOR 5. Stock Status

Update the biological reference points or trend-based indicators/metrics for the stock. Determine stock status.

The Atlantic Menhaden Management Board (Board) adopted ERPs in Amendment 3 to account for menhaden's ecological role as a forage species. Thus, stock status is determined using ERPs. However, the ERP values adopted for management use were based on the 2020 ERP benchmark assessment and are not directly comparable to the estimates of fecundity and biomass from this assessment update, due to the change in M. The 2025 ERP benchmark assessment incorporated the revised estimate of M into the ERP models, and proof-of-concept benchmarks have been provided using the definitions adopted by the Management Board in 2020 (Table 3). These proof-of-concept values may change as the tool is further refined via the peer review process and the Management Board's use of the tool to evaluate tradeoffs in their goals and objectives. Thus, stock status may change if the final ERP values adopted for management use in 2025 change. If that occurs, this assessment will be updated to reflect the final ERP values and stock status.

Using the proof-of-concept ERP benchmarks, the Atlantic menhaden population is not overfished and overfishing is not occurring. The fishing mortality rate for the terminal year of 2023 is below the ERP threshold and just above the ERP target ($F_{2023}/F_{ERPThreshold}$ = 0.52; $F_{2023}/F_{ERPTarget}$ = 1.35; Figure 14), and the fecundity for the terminal year of 2023 is above the ERP threshold and but below the ERP target ($FEC_{2023}/FEC_{ERPThreshold}$ = 1.09; $FEC_{2023}/FEC_{ERPTarget}$ = 0.74; Figure 15). Therefore, overfishing is not occurring and the stock is not overfished (Table 3).

The uncertainty in the stock status was evaluated through the MCB analysis. The terminal year *F* was below the ERP threshold for all of the MCB runs (Figure 16) and the terminal year fecundity was above the ERP threshold for 77% of the runs (Figure 17). The SAS does note that each MCB run was not run through the ERP's Northwest Atlantic Coastal Shelf Model of Intermediate Complexity for Ecosystems (NWACS-MICE) model, thus the benchmark comparisons were to those from the base run. The MCB plots are not internally consistent for each run, but do give an idea of the uncertainty related to the ERP benchmarks, which agrees with the base run stock status determinations.

TOR 6. Projections

Conduct short term projections when appropriate. Discuss assumptions if different from the benchmark and describe alternate runs.

Short-term projections at the current Total Allowable Catch (TAC) of 233,550 mt were provided (Figure 18). Under a constant TAC of 233,550 mt, *F* will be between the *F* target and the *F* threshold, with a 0.5% probability that *F* will be above the ERP *F* threshold and a 99.5%

probability that it will be above the *F* target in 2028 (Table 4). Further projections based on different removal levels will be analyzed at the Board's request.

The projections have the same methods and assumptions as those run for the benchmark assessment. It is important to note that uncertainty is accounted for in the projections. Additionally, during the benchmark (SEDAR 2020a), the SAS used a new procedure for recruitment in the projections. Instead of assuming a static median value for recruitment, as is done for many assessment projection methodologies, recruitment was projected using nonlinear time series analysis methods (Deyle et al 2018). Specifically, projections were based on the MCB runs, which allows recruitment to change from year to year in the projections based on how recruitment has changed in the past under similar conditions. Thus, uncertainty is recognized in the recruitment time series, and the methods used for projections adequately accounted for that uncertainty using the best scientific methods available. However, the Board should still consider these uncertainties in the context of risk when using the projection information for management.

TOR 7. Research Recommendations

Comment on research recommendations from the benchmark stock assessment and note which have been addressed or initiated. Indicate which improvements should be made before the stock undergoes a benchmark assessment.

All research recommendations from SEDAR 2020a and 2020b remain important to the continued assessment of Atlantic menhaden, including those updated in this section. Please refer to the appendices at the end of this report for the complete list.

A long-standing research recommendation for Atlantic menhaden is to develop and implement a multi-year coastwide fishery-independent survey. It was noted in SEDAR 2020a that even area-specific surveys could provide substantial improvements over the indices currently used in the assessment. Pilot studies combining hydroacoustics and aerial or trawl surveys have been conducted successfully in Chesapeake Bay and mid-Atlantic ocean waters (e.g., Wilberg et al. 2020; Nesslage et al. 2024). However, no funding has been secured for long-term implementation of these projects.

Despite the research recommendation to continue the current level of sampling from the fisheries, some sampling was reduced or temporarily discontinued due to the COVID-19 pandemic. However, sample sizes have returned to pre-pandemic levels in the years since the 2022 assessment update. The TC is planning to meet this summer to evaluate the adequacy of the current bait sampling requirements for the states.

In preparation for shifting aging responsibilities to the states, ASMFC coordinated an age exchange in 2023 – 2024, with the final report due in 2025. During the exchange, 65 paired scale and otolith samples and 11 scale-only samples were read by staff from 12 states and the NOAA Beaufort lab. True age was not known for any of the samples, so comparisons only provide information on variability among users. Preliminary results indicate that precision was generally low across labs and structures, and bias was commonly detected, likely due to the fact that many of the participating labs do not regularly age menhaden. ASMFC is scheduling a follow-up meeting to review results and discuss ways to improve precision among partners before fully transitioning bait ageing to the states.

Although a seasonal and spatially-explicit model has not been developed, the SAS has recently completed a thorough review of data from an extensive mark-recapture study conducted by the NOAA Beaufort lab during the late 1960s that could provide insight into age-specific movement patterns needed for such a model (see SEDAR 102 WP-01 for more details on the dataset and estimated movement patterns).

During the next benchmark stock assessment process (scheduled for 2031), the SAS recommends that the MARECO index still be considered for inclusion in the model, but further investigation is necessary. One option the SAS could consider is using nonlinear relationships between *q* and the MARECO index. Additionally, the SAS recommends that ACCSP continues to work with the states to validate bait landings and resolve the transition in the time series from pre-1985 bait landings in the northern region.

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TABLES

Table 1. Fishery-independent surveys included in the coastwide young-of-year (YOY) and regional adult Atlantic menhaden abundance indices (Northern Adult Index, NAD; Mid-Atlantic Index, MAD; Southern Adult Index, SAD).

Conn Index	Fishery-Independent Survey (years of data)	Months	Length
NAD	CT LISTS (1996-2009, 2011-2019, 2021-2023)	Sept-lagged Jan	1990-2023
	DB Adult Trawl (1990-2023)		
	NJ Ocean Trawl (1990-1997, 1999-2019)		
MAD	MD Gill Net (1985-1995, 1998-2002, 2005-2023)	March-May	1985-2023
	VIMS Shad Gill Net (1998-2023)		
SAD	NC p915 (2008-2019, 2021-2023)	April-July	1990-2023
	SEAMAP (1990-2019, 2022-2023)		
	GA EMTS (2003-2023)		
YOY	RI Trawl (1990-2023)	Varies by survey	1959-2023
	CT LISTS (1996-2009, 2011-2019, 2021-2023)		
	CT River Alosine (1987-2023)		
	CT Thames River Alosine (1998-2016)		
	NY Juvenile Striped Bass Seine (2000-2023)		
	NY Peconic Bay Trawl (1987-1988, 1990-1992, 1994-2007, 2009-2023)		
	NY WLIS Seine (1986-2023)		
	NJ Ocean Trawl (1990-2019)		
	NJ Striped Bass YOY Seine (1986-2019, 2021-2023)		
	DB Inner Bays (1986-2023)		
	MD Coastal Trawl (1972-1992, 1994, 1998-2023)		
	MD Juvenile Striped Bass (1959-2023)		
	VIMS Juvenile Trawl (1990-2023)		
	VIMS Striped Bass Seine (1968-1972, 1980, 1982, 1985-2023)		
	NC p120 (1989-2023)		
	SC Electrofishing (2001-2023)		

Table 2. Model structure and life history information used in the stock assessment.

	Value(s)
Years in Model	1955-2023
Age Plus Group	6+
Fleets	4 (north and south regions for bait and reduction fisheries)
Fecundity	Time-varying fecundity-at-age
Natural Mortality	Age-varying natural mortality scaled to tagging based estimate, revised for 2025
Maturity	Time-varying maturity-at-age based on length-at-age
Sex Ratio	Fixed at 1:1 for males:females

	Age Group						
Natural Mortality	0	1	2	3	4	5	6+
2020 Benchmark	1.76	1.31	1.03	0.9	0.81	0.76	0.72
2025 Update Base Run	**1.39**	**1.03**	**0.82**	**0.71**	**0.64**	**0.60**	**0.57**
2025 Update Sensitivity	0.71	0.52	0.42	0.36	0.33	0.30	0.29

Table 3. Stock status based on proof-of-concept fishing mortality (*F*) and fecundity (*FEC*) ecological reference points (ERP targets and thresholds) from the 2025 ERP assessment and terminal year values from the base run of the BAM for the stock assessment update. Fishing mortality is the full fishing mortality. Fecundity is in billions of eggs.

Reference Point	ERP Value	2023 Value	Stock Status
$F_{THRESHOLD}$	0.490	0.26	Not Overfishing
F_{TARGET}	0.189		
$FEC_{THRESHOLD}$	1,142,453	1,240,272	Not Overfished
FEC_{TARGET}	1,666,030		

Table 4. Probability of exceeding the proof-of-concept ERP *F* threshold and target for 2024-2028 under a constant status quo TAC.

	2024	2025	2026	2027	2028
ERP *F* threshold	0.0%	0.0%	0.0%	0.5%	0.5%
ERP *F* target	100.0%	99.5%	99.5%	99.5%	99.5%

FIGURES

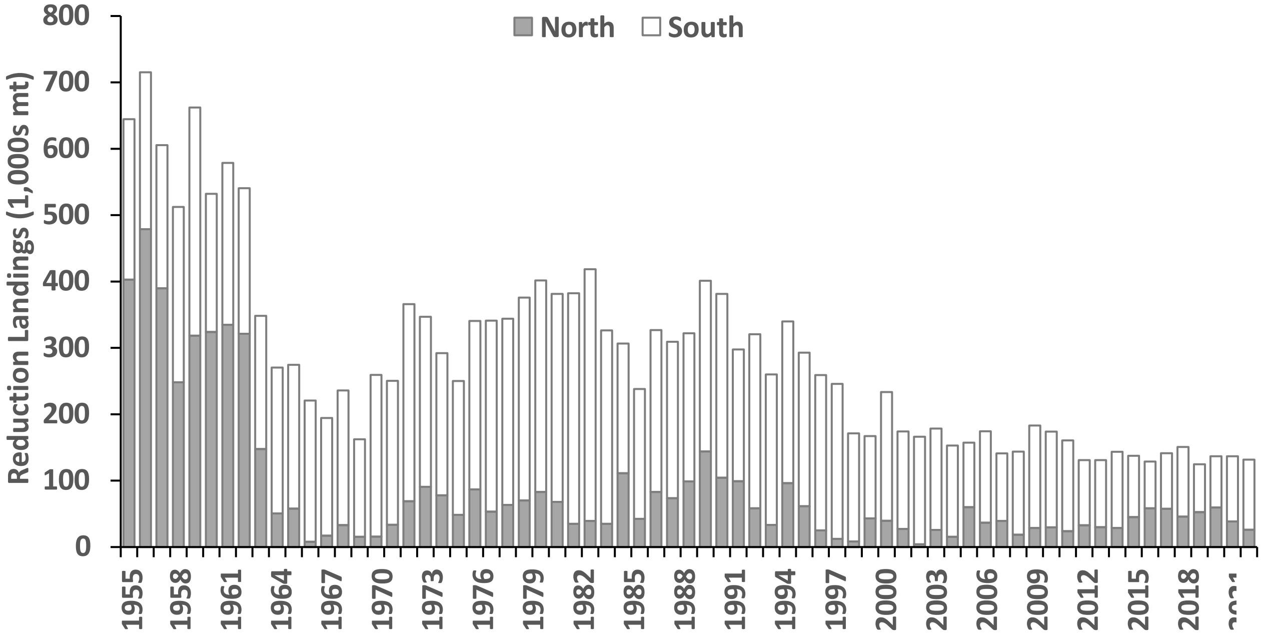


Figure 1. Atlantic menhaden reduction landings (1000s mt) from 1955-2023. The northern region is comprised of landings from Maine to Maryland's Eastern Shore, excluding the Chesapeake Bay, and the southern region is comprised of landings from Virginia Eastern Shore and Chesapeake Bay through Florida (Source: NOAA Fisheries Beaufort).

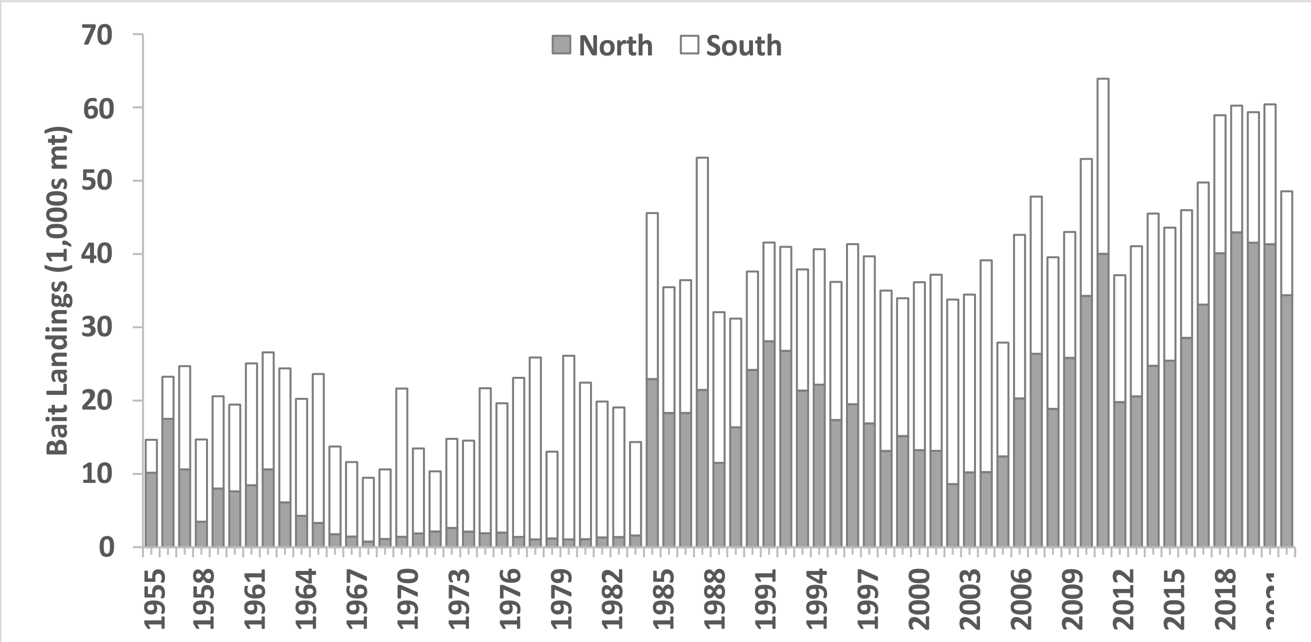


Figure 2. Atlantic menhaden bait landings (1000s mt) from 1955-2023. The northern region includes landings from Maine to Maryland's Eastern Shore, excluding the Chesapeake Bay, and the southern region is comprised of landings from Virginia Eastern Shore and Chesapeake Bay through Florida Only landings from 1985 on can be validated (Source: ACCSP).

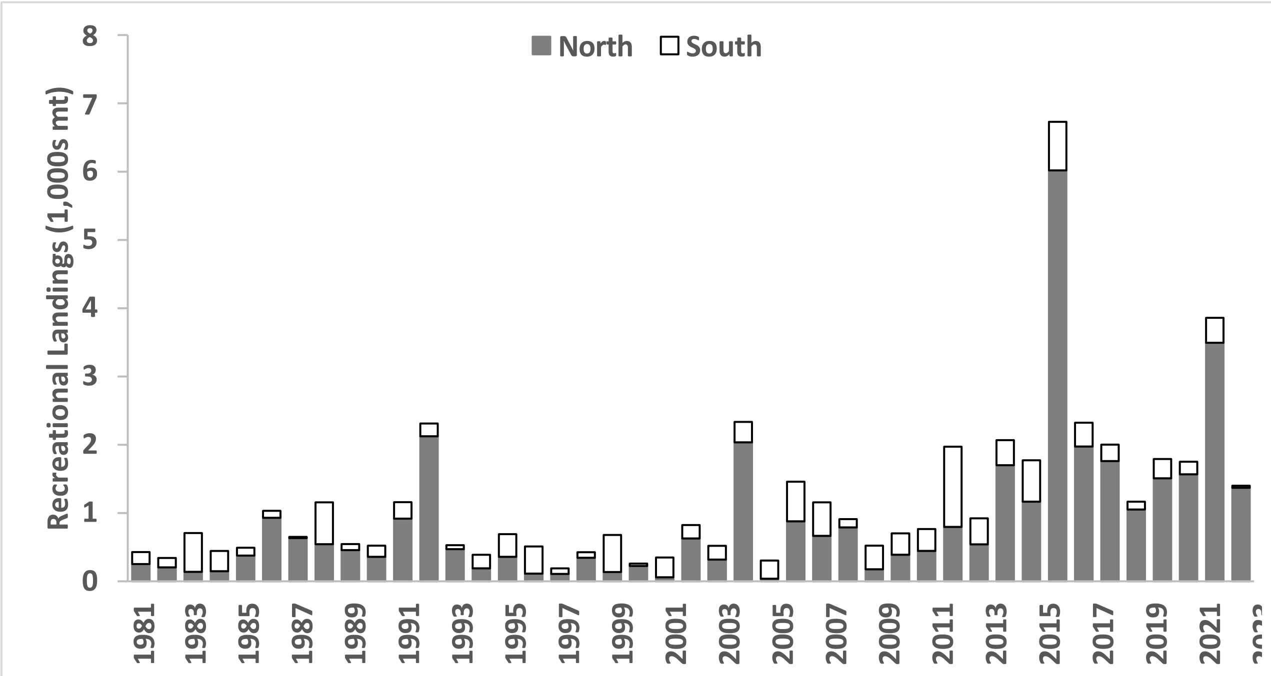


Figure 3. Atlantic menhaden recreational landings (1000s mt) from 1981-2023. The northern region includes landings from Maine to Maryland's Eastern Shore, excluding the Chesapeake Bay, and the southern region is comprised of landings from Virginia Eastern Shore and Chesapeake Bay through Florida (Source: MRIP).

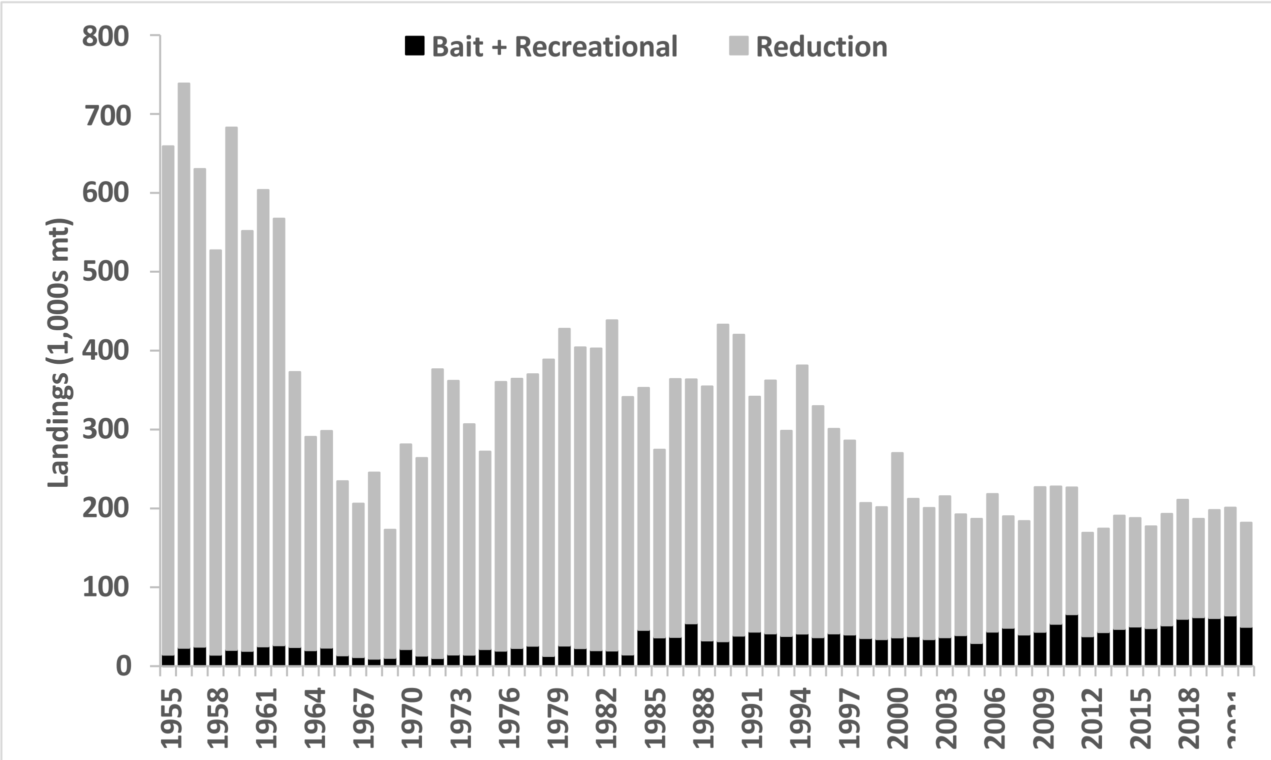


Figure 4. Coastwide Atlantic menhaden landings for the reduction and bait fisheries (1955-2023).

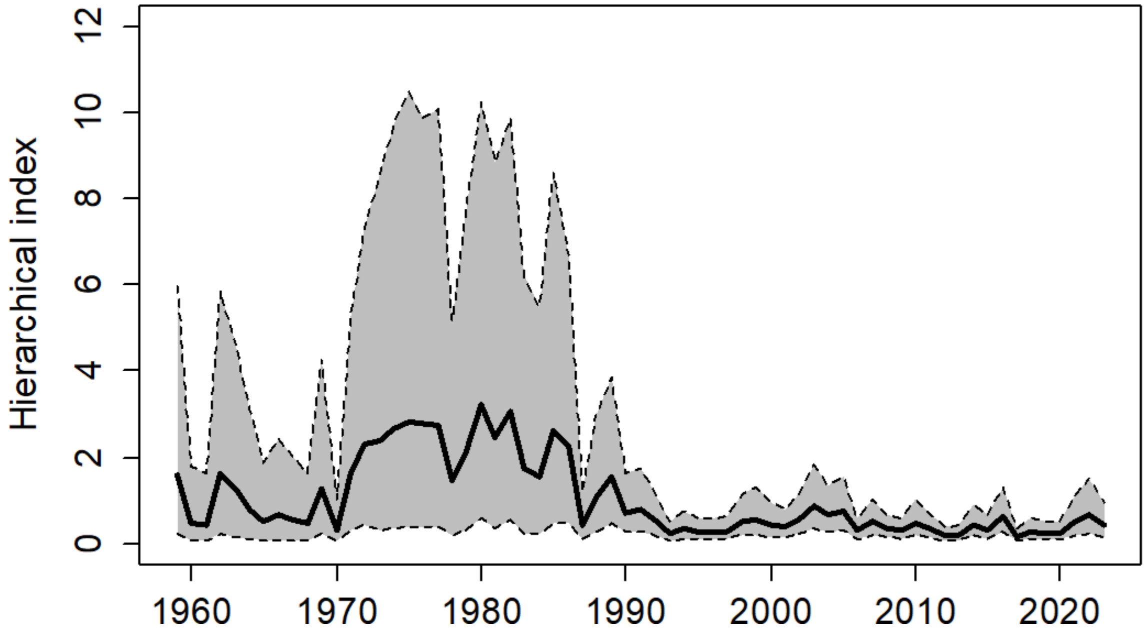


Figure 5. Time series of the young-of-year (YOY) Atlantic menhaden relative abundance index as estimated from hierarchical analysis (Conn 2010). The black line gives the posterior mean and the grey, dashed lines represent a 95% credible interval about the time series.

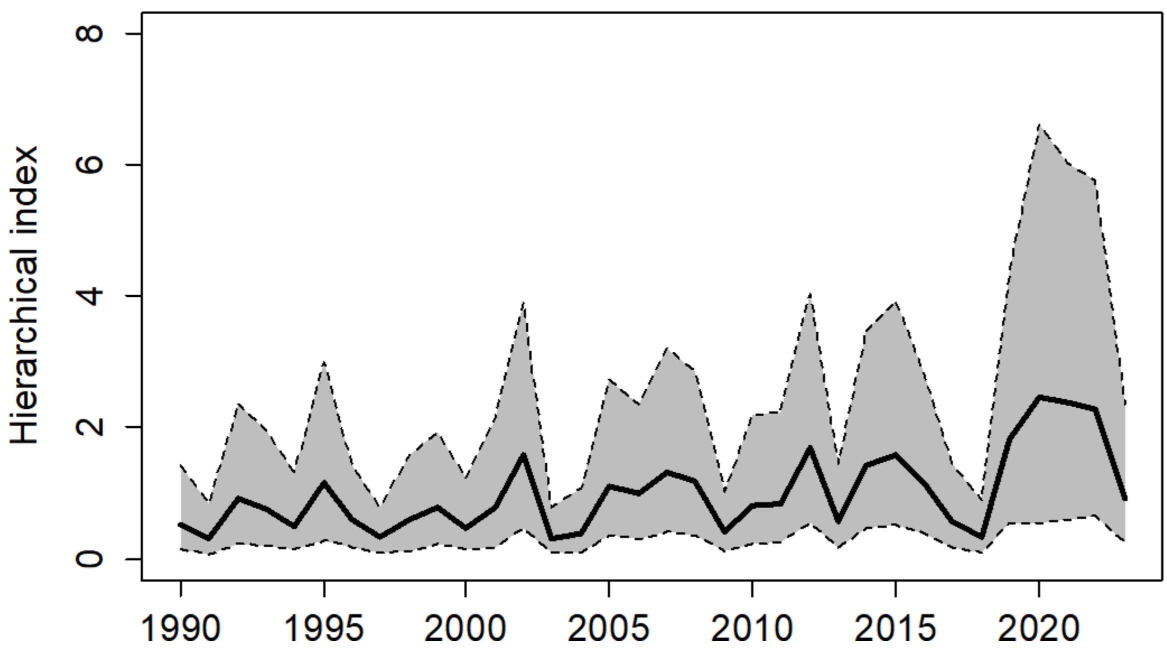


Figure 6. Time series of the northern adult Atlantic menhaden relative abundance index (NAD) as estimated from hierarchical analysis (Conn 2010). The black line gives the posterior mean and the grey, dashed lines represent a 95% credible interval about the time series.

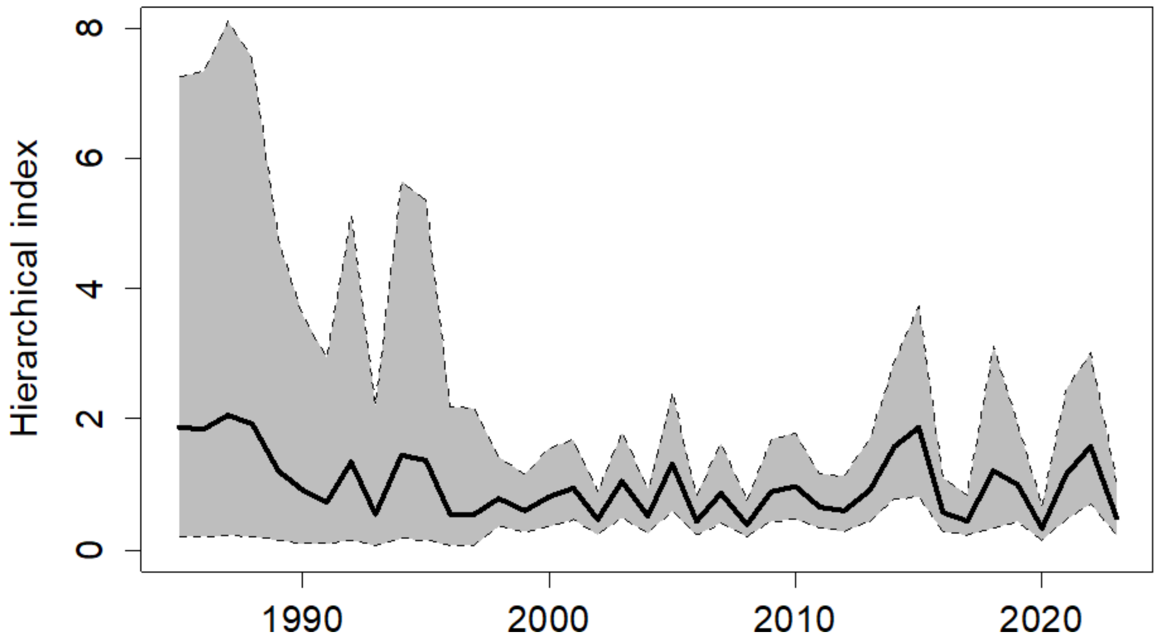


Figure 7. Time series of the Mid-Atlantic adult menhaden relative abundance index (MAD) as estimated from hierarchical analysis (Conn 2010). The black line gives the posterior mean and the grey, dashed lines represent a 95% credible interval about the time series.

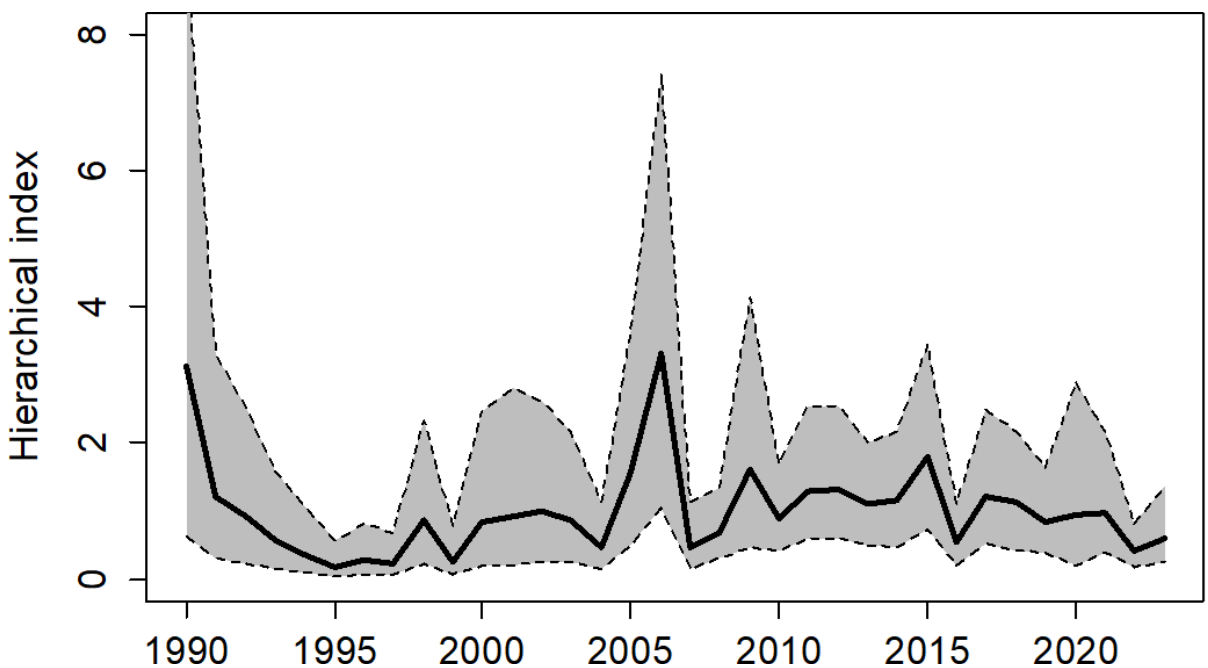


Figure 8. Time series of the southern adult Atlantic menhaden relative abundance index (SAD) as estimated from hierarchical analysis (Conn 2010). The black line gives the posterior mean and the grey, dashed lines represent a 95% credible interval about the time series.

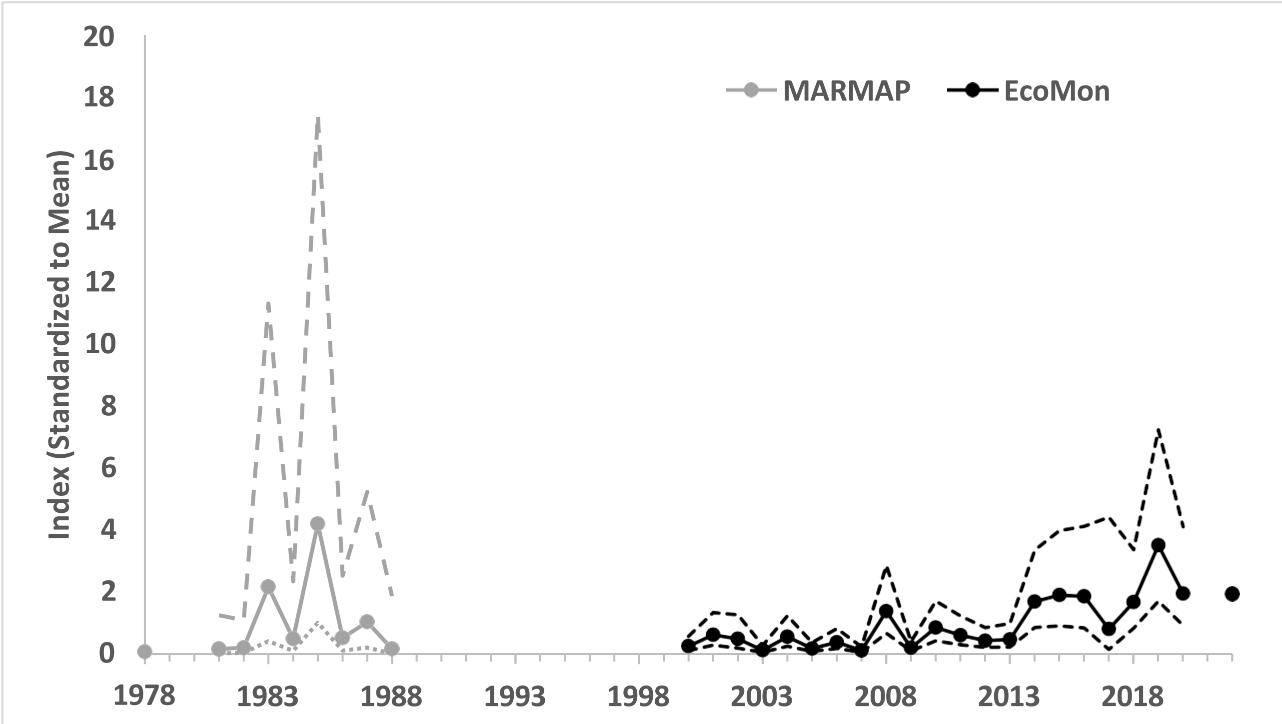


Figure 9. Standardized index of relative spawning stock biomass of Atlantic menhaden developed from the MARMAP and EcoMon ichthyoplankton surveys. Dashed lines represent 95% confidence intervals. The 1978 upper confidence interval has not been included on the graph because of its large value (94).

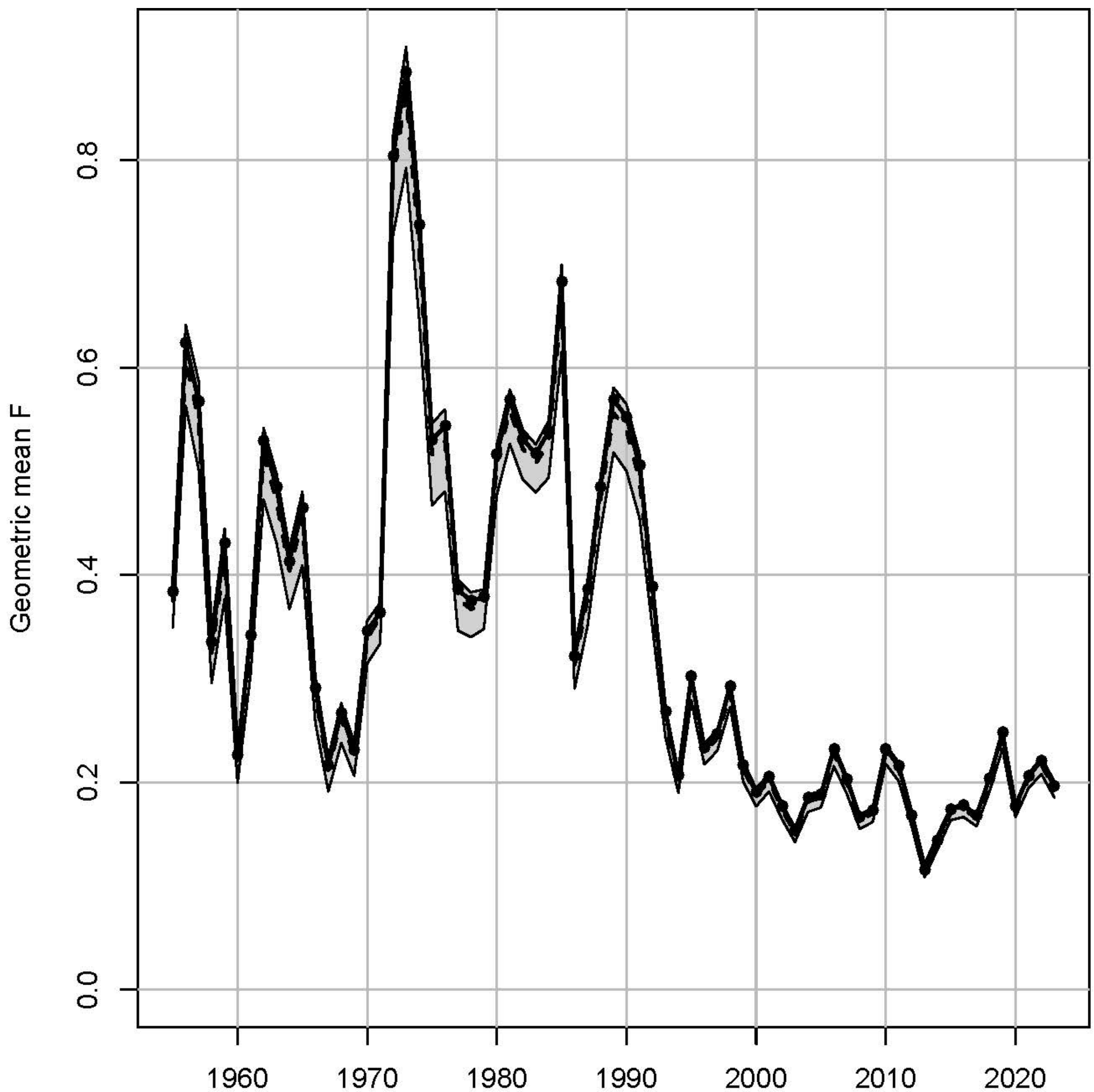


Figure 10. Time series of the geometric mean fishing mortality rate for ages-2 to 4 from 1955-2023 for the Monte Carlo bootstrap runs. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run. The dashed line represents the median of the MCB runs.

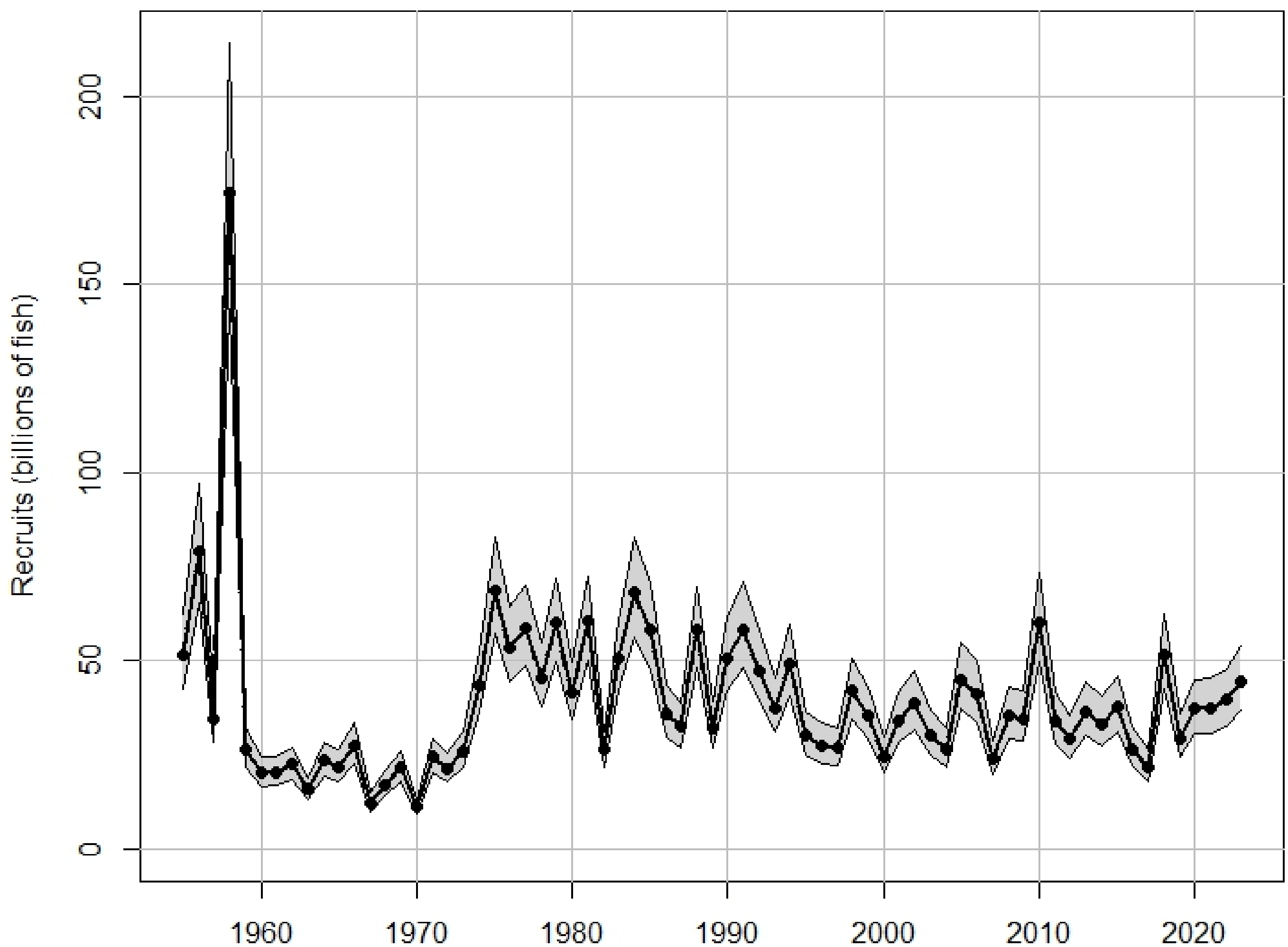


Figure 11. Estimated recruitment (billion fish) over time from 1955-2023 for the Monte Carlo bootstrap runs. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run.

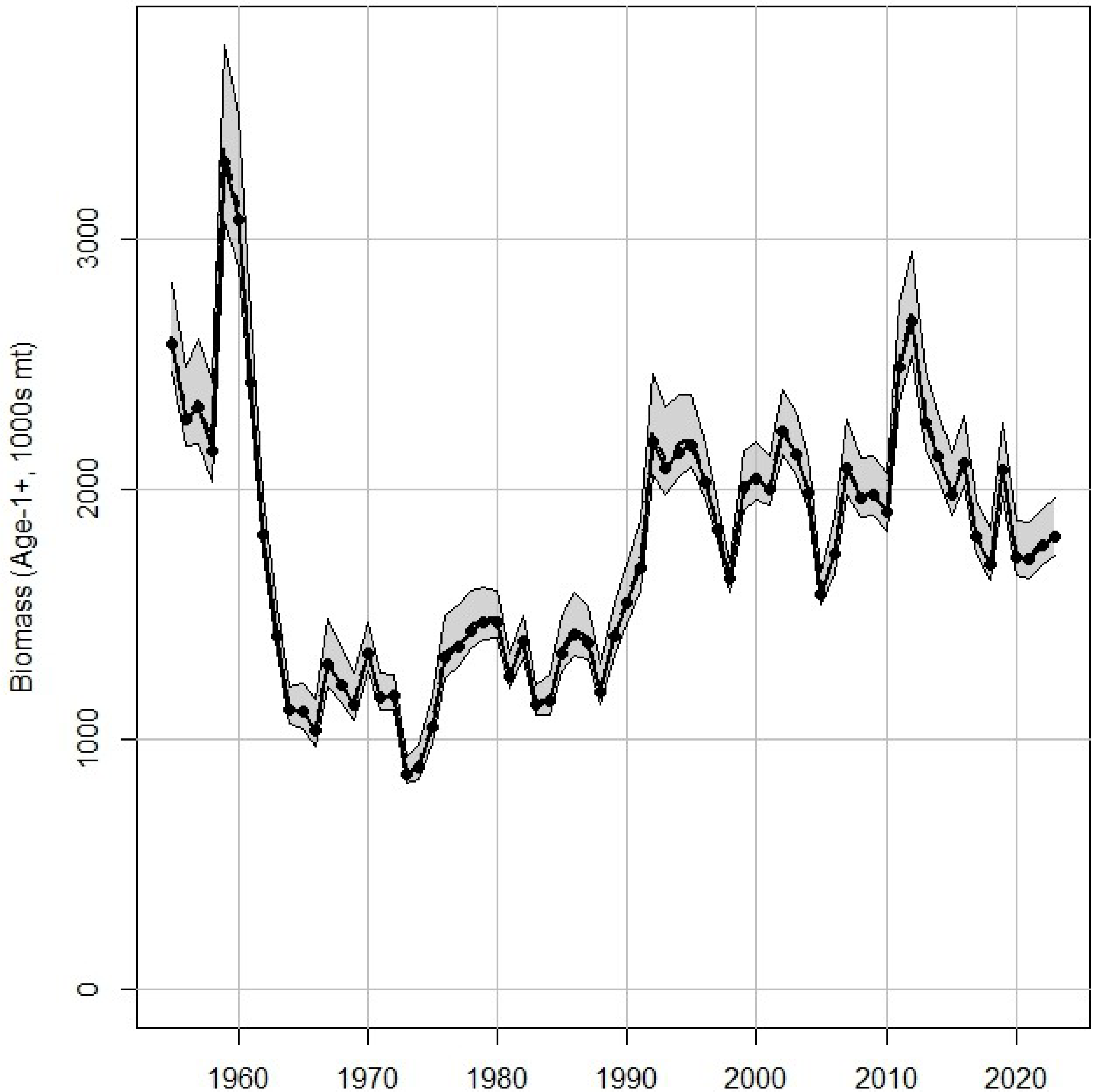


Figure 12. Time series of age-1+ biomass (1,000s metric tons) from 1955-2023 for the Monte Carlo bootstrap runs. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run. The dashed line represents the median of the MCB runs.

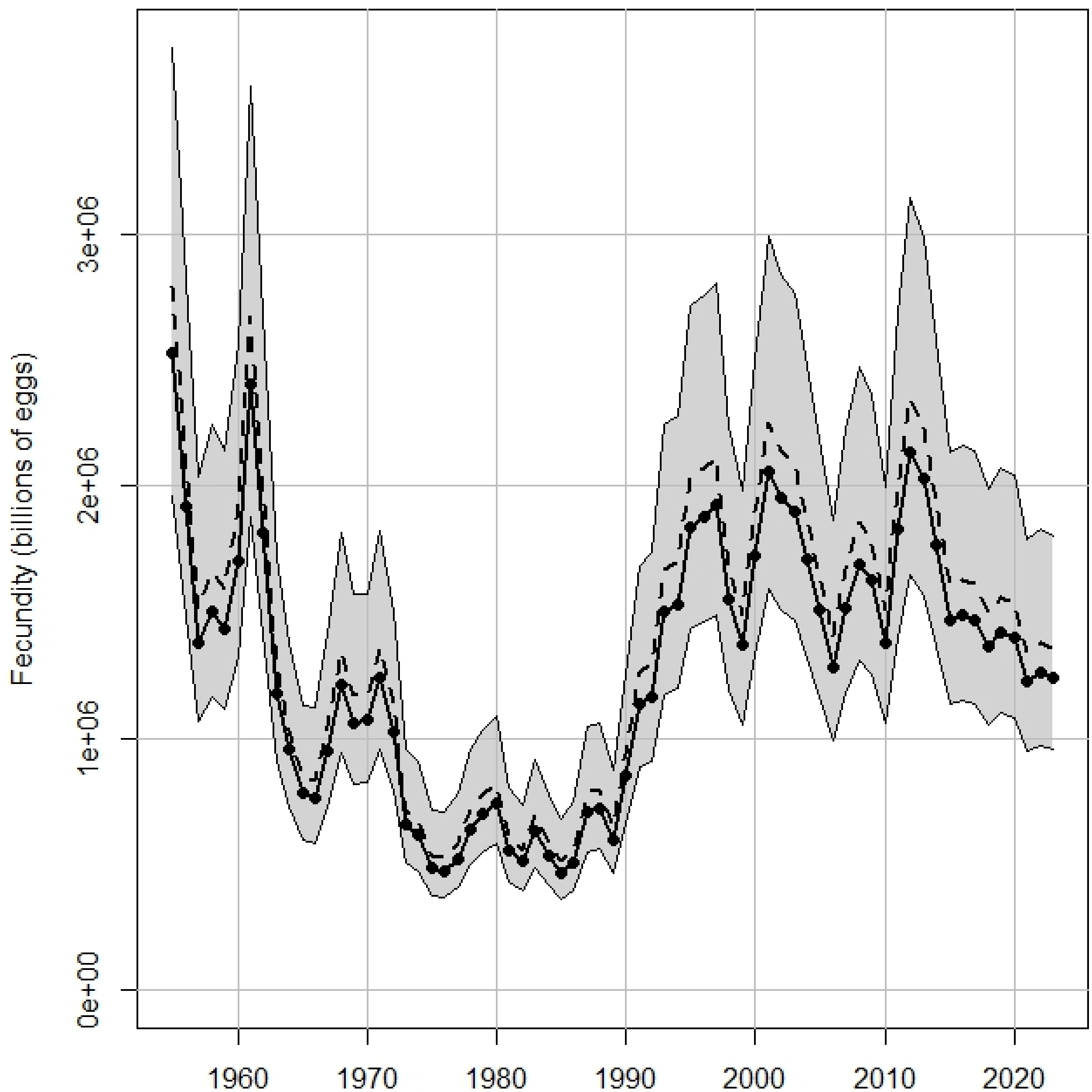


Figure 13. Time series of fecundity (billions of eggs) from 1955-2023 for the Monte Carlo bootstrap runs. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run. The dashed line represents the median of the MCB runs.

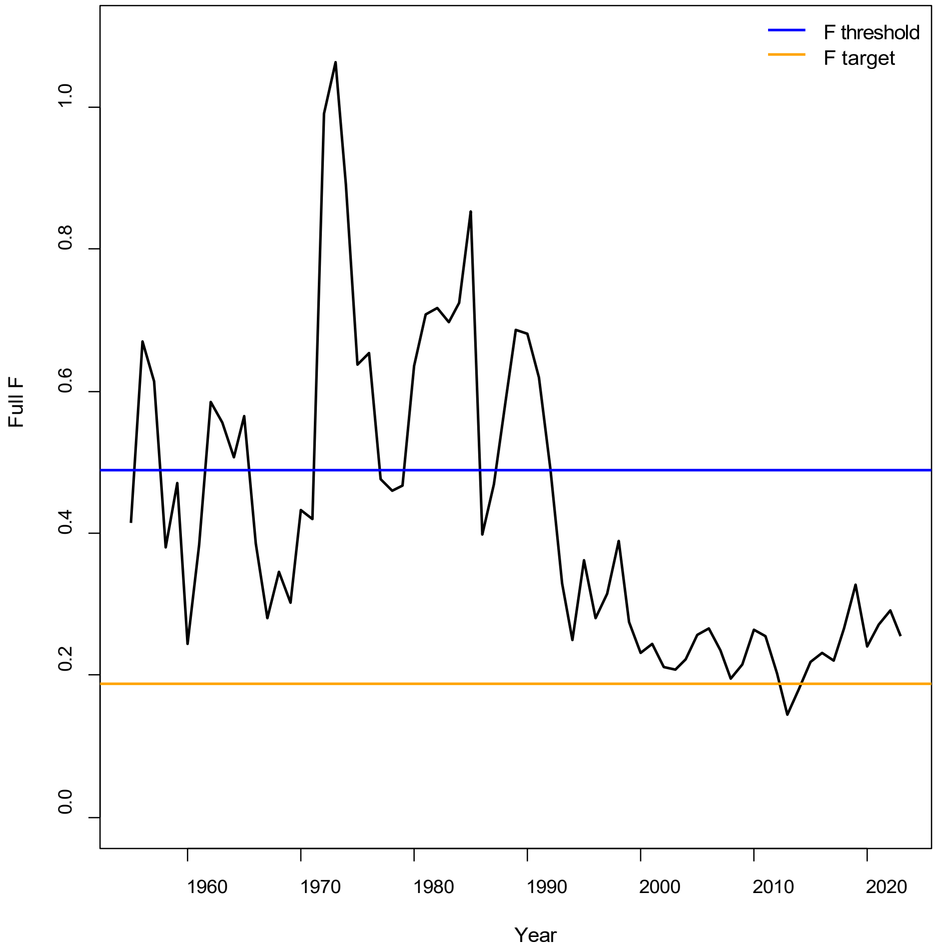


Figure 14. The full fishing mortality rate for 1955-2023 compared to the proof-of-concept ecological reference point (ERP) threshold and target for fishing mortality rate. The full fishing mortality is dependent upon selectivity for the fisheries, and thus can represent ages-2 to 4, depending upon the year.

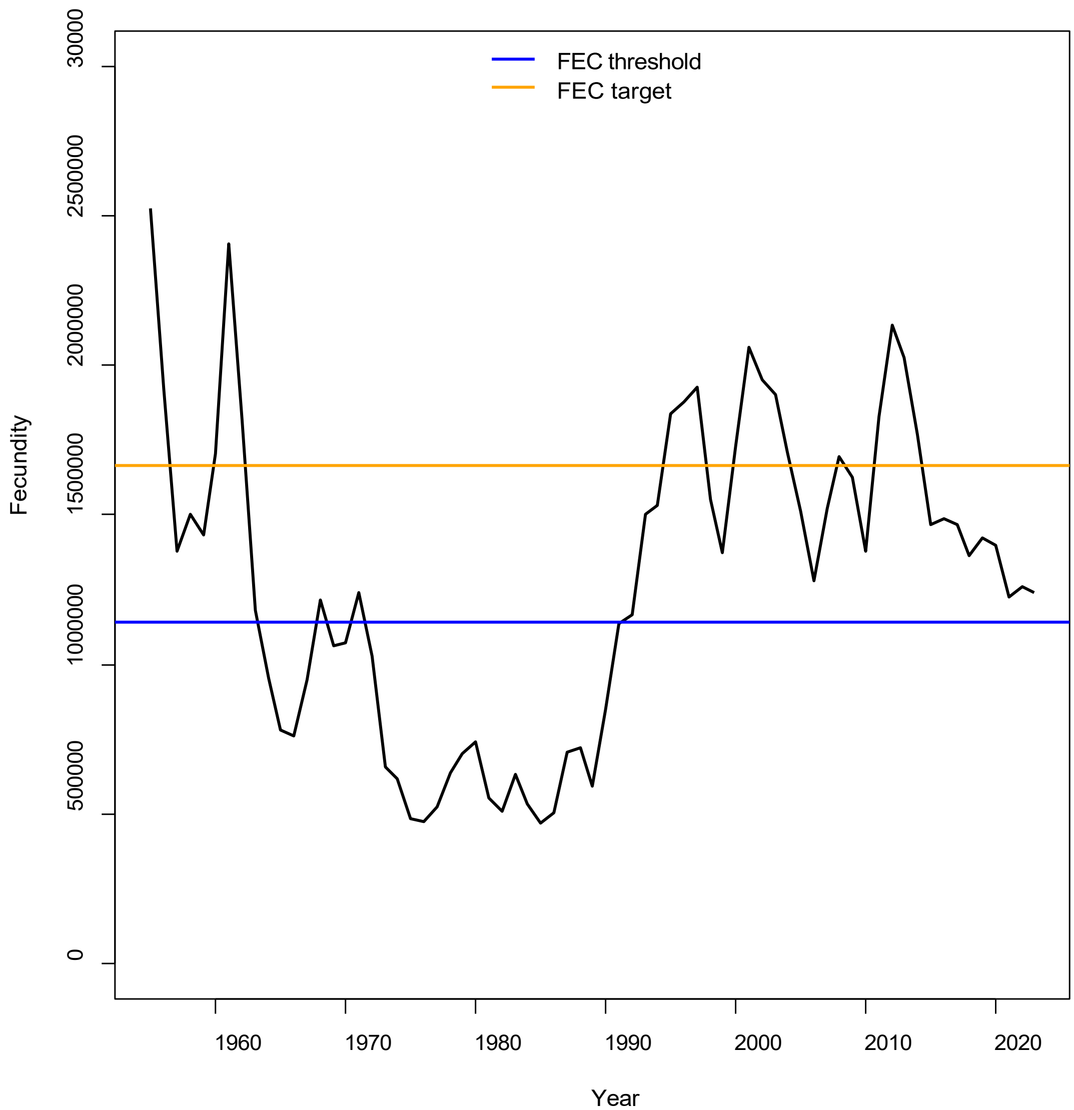


Figure 15. The fecundity for 1955-2023 compared to the proof-of-concept ecological reference point (ERP) threshold and target for fecundity.

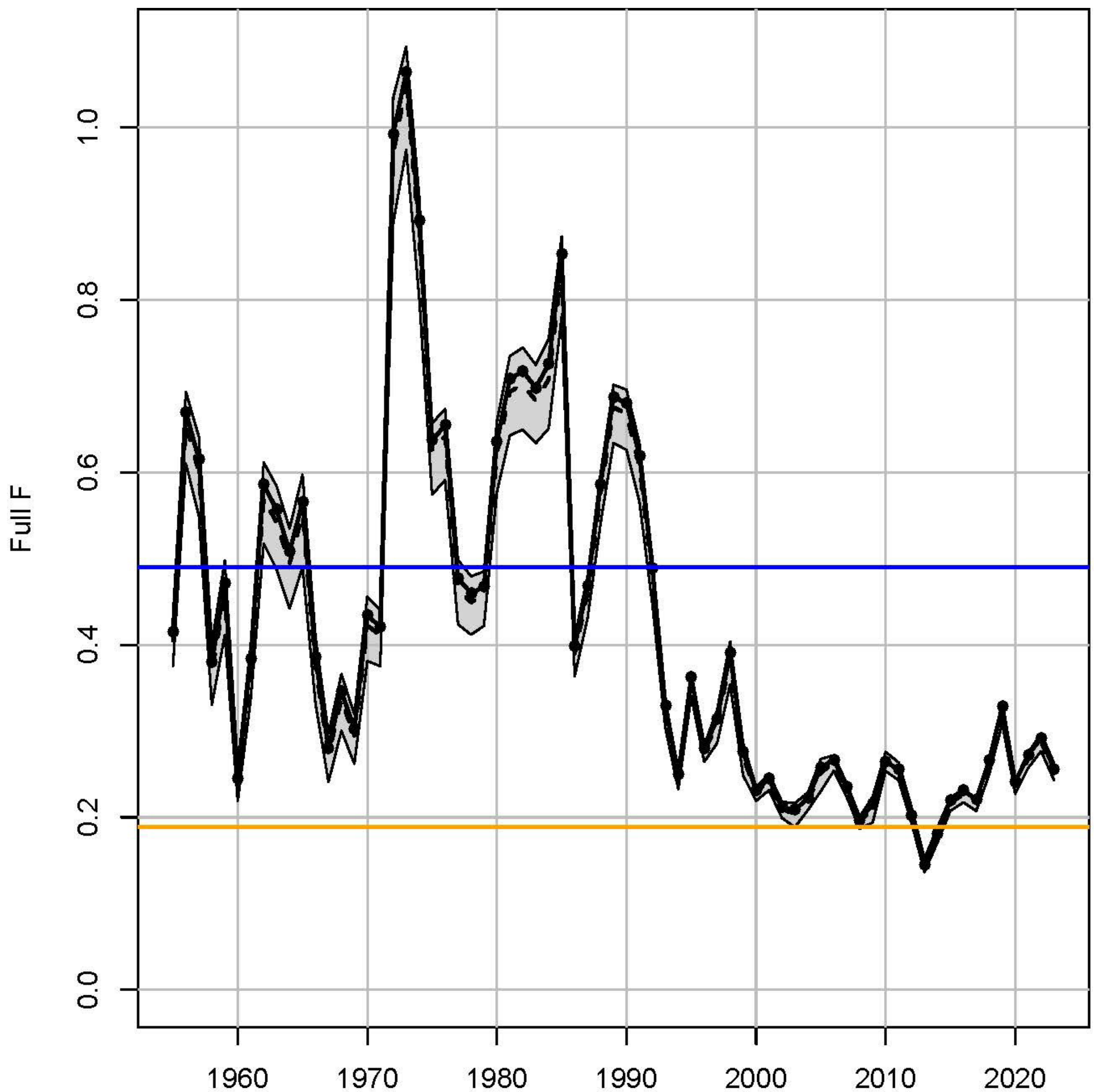


Figure 16. Fishing mortality rate from the MCB analysis plotted with the proof-of-concept ERP *F* threshold and target. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run. The dashed line represents the median of the MCB run.

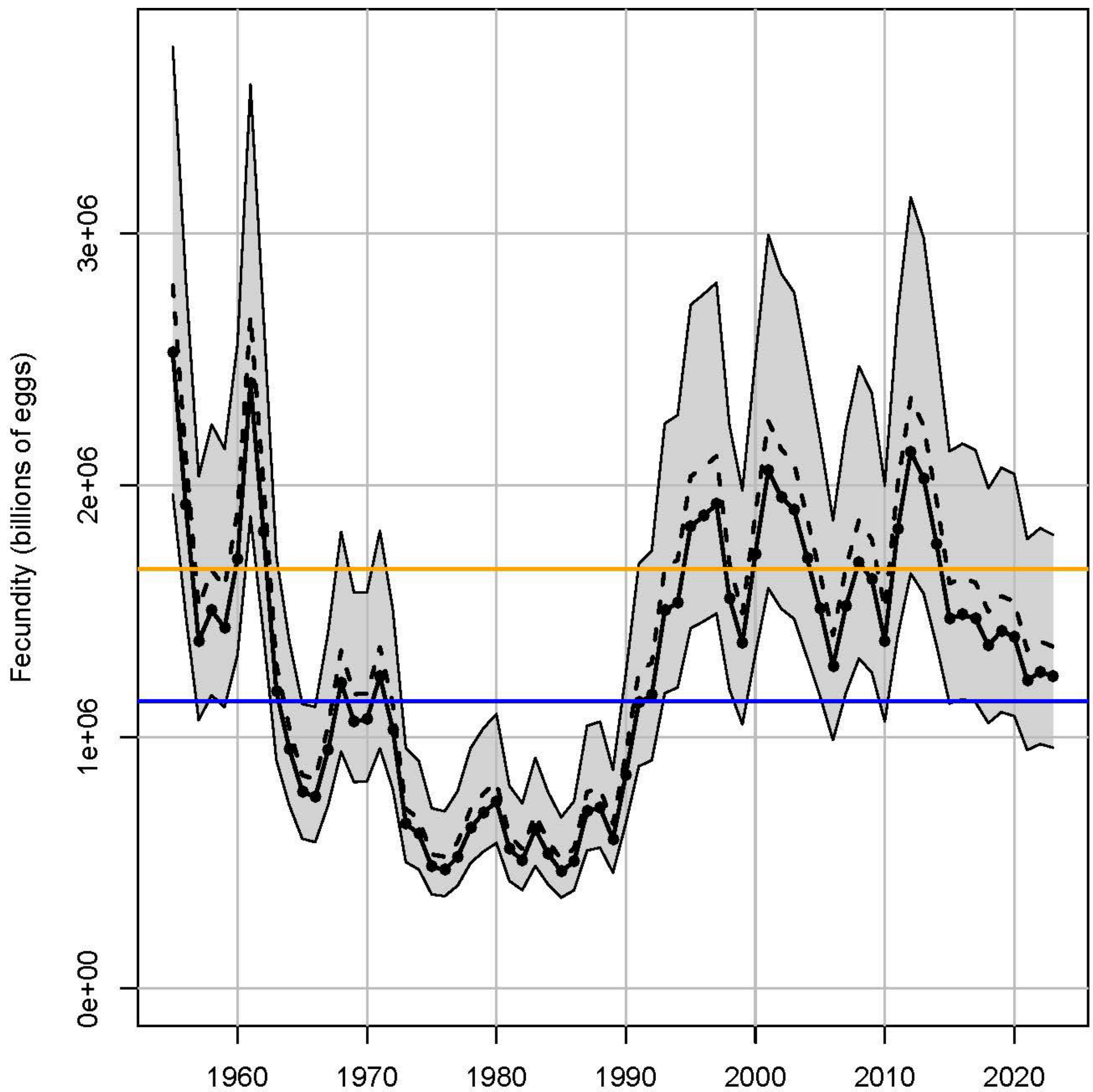


Figure 17. Fecundity from the MCB analysis plotted with the proof-of-concept ERP fecundity threshold. The grey represents the 5th and 95th percentiles across the runs, while the black line with closed black circles represents the base run. The dashed line represents the median of the MCB runs.

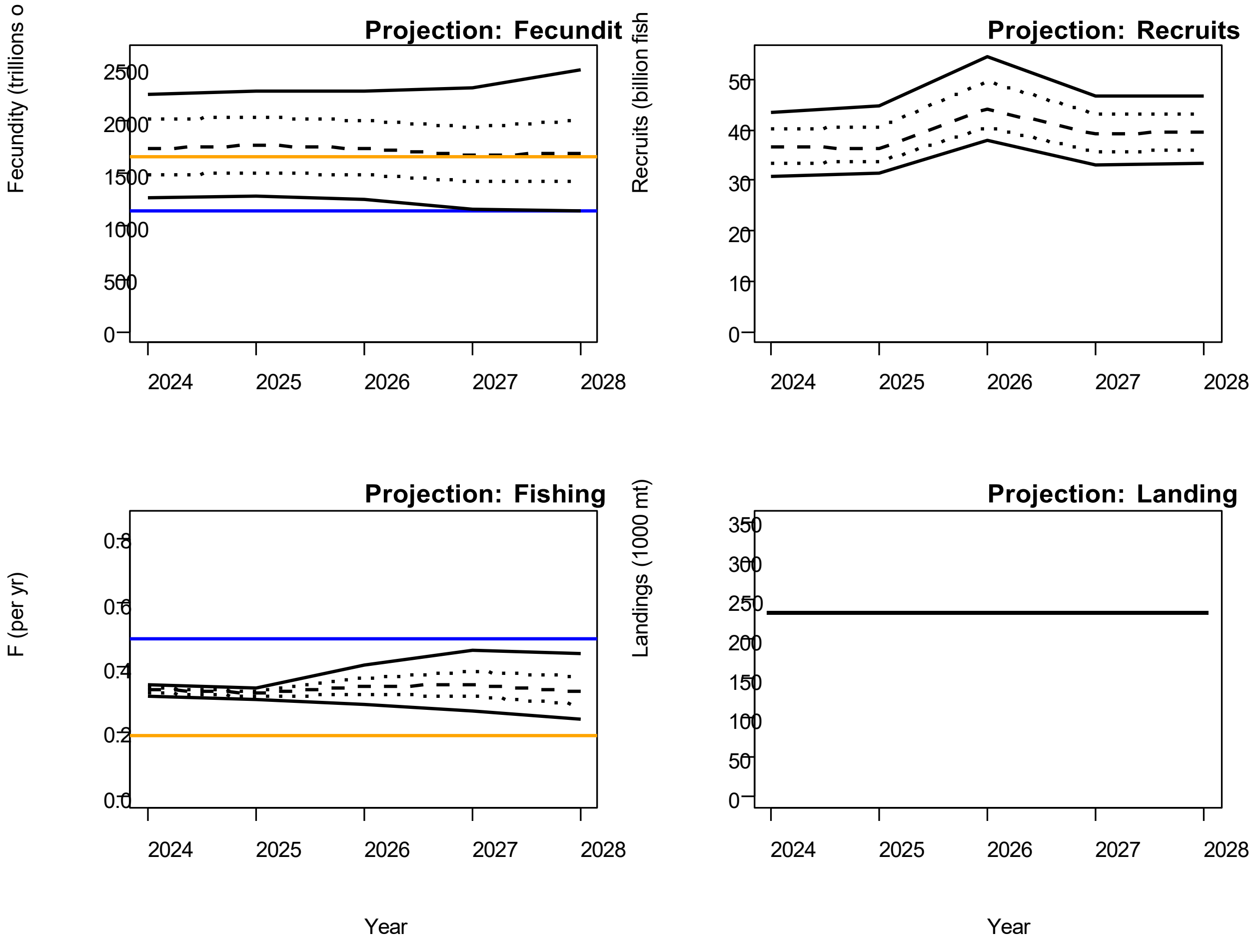


Figure 18. Fecundity, full fishing mortality rate, and recruits projected from 2024 to 2028 for a coastwide total allowable catch of 233,550 mt. The orange lines represent the proof-of-concept target fishing mortality rate and fecundity for the ecological reference points, while the blue lines represent the proof-of-concept threshold fishing mortality rate and fecundity for the ecological reference points. The dashed black line is the 50th percentile (median), the dotted black lines are the 25th and 75th percentiles, and the solid black lines are the 5th and 95th percentiles.

APPENDIX

Appendix Tables

Table A1. Atlantic menhaden landings (in 1,000s of metric tons) by fishery and region, 1955-2023. Bait landings are considered incomplete until 1985.

Year	Reduction Landings			Bait Landings			Recreational Landings			Total Landings
	Total	North	South	Total	North	South	Total	North	South	
1955	644.48	402.74	241.74	14.64	10.14	4.50				659.12
1956	715.25	478.89	236.36	23.25	17.51	5.74				738.50
1957	605.58	389.80	215.78	24.71	10.60	14.11				630.29
1958	512.39	248.34	264.05	14.69	3.46	11.23				527.07
1959	662.17	318.44	343.73	20.58	7.98	12.61				682.76
1960	532.24	323.86	208.37	19.44	7.61	11.83				551.68
1961	578.61	334.76	243.85	25.07	8.44	16.63				603.68
1962	540.66	321.36	219.31	26.58	10.60	15.98				567.24
1963	348.44	147.55	200.89	24.39	6.11	18.28				372.83
1964	270.40	50.61	219.80	20.23	4.27	15.97				290.64
1965	274.60	57.96	216.64	23.62	3.30	20.32				298.22
1966	220.69	7.89	212.80	13.72	1.76	11.96				234.41
1967	194.39	17.21	177.18	11.61	1.44	10.17				206.00
1968	235.86	33.07	202.80	9.46	0.75	8.71				245.32
1969	162.33	15.41	146.92	10.61	1.11	9.50				172.94
1970	259.39	15.80	243.59	21.64	1.41	20.23				281.03
1971	250.32	33.44	216.87	13.47	1.87	11.60				263.79
1972	365.87	69.09	296.78	10.35	2.14	8.21				376.22
1973	346.92	90.69	256.23	14.77	2.61	12.16				361.69
1974	292.20	77.90	214.31	14.54	2.11	12.43				306.74
1975	250.21	48.40	201.81	21.69	1.89	19.80				271.90
1976	340.54	86.84	253.70	19.63	1.98	17.65				360.17
1977	341.16	53.31	287.85	23.09	1.39	21.70				364.25
1978	344.08	63.53	280.55	25.87	1.07	24.80				369.95
1979	375.74	70.19	305.55	13.02	1.17	11.85				388.76
1980	401.53	83.02	318.51	26.11	1.07	25.05				427.64
1981	381.31	68.06	313.25	22.44	1.08	21.36	0.42	0.25	0.17	404.17
1982	382.46	35.08	347.38	19.86	1.32	18.54	0.34	0.20	0.14	402.66
1983	418.63	39.37	379.26	19.06	1.36	17.71	0.68	0.14	0.54	438.38
1984	326.30	34.97	291.33	14.33	1.59	12.75	0.42	0.15	0.27	341.05
1985	306.67	111.25	195.42	45.59	22.92	22.66	0.52	0.38	0.14	352.78

Table A1. ***Continued***

Year	Reduction Landings			Bait Landings			Recreational Landings			Total Landings
	Total	North	South	Total	North	South	Total	North	South	
1986	237.99	42.57	195.42	35.46	18.30	17.17	1.03	0.93	0.10	274.49
1987	326.90	82.99	243.91	36.43	18.30	18.13	0.65	0.63	0.02	363.98
1988	309.29	73.64	235.65	53.14	21.44	31.70	1.16	0.54	0.61	363.58
1989	322.00	98.82	223.18	32.06	11.49	20.57	0.54	0.46	0.09	354.61
1990	401.15	144.10	257.05	31.19	16.35	14.84	0.52	0.36	0.16	432.86
1991	381.43	104.55	276.87	37.62	24.17	13.45	1.16	0.92	0.24	420.20
1992	297.64	99.14	198.50	41.56	28.08	13.48	2.31	2.12	0.19	341.51
1993	320.60	58.37	262.23	40.98	26.76	14.22	0.53	0.47	0.06	362.11
1994	259.99	33.39	226.60	37.89	21.35	16.54	0.39	0.19	0.20	298.27
1995	339.92	96.30	243.62	40.64	22.17	18.47	0.69	0.36	0.33	381.25
1996	292.93	61.55	231.38	36.19	17.34	18.85	0.51	0.11	0.40	329.63
1997	259.14	25.17	233.98	41.35	19.49	21.86	0.19	0.11	0.08	300.68
1998	245.91	12.33	233.58	39.70	16.88	22.81	0.43	0.34	0.08	286.03
1999	171.19	8.42	162.77	35.00	13.11	21.89	0.68	0.13	0.54	206.87
2000	167.26	43.19	124.08	33.95	15.15	18.80	0.26	0.22	0.03	201.47
2001	233.56	39.62	193.94	36.14	13.24	22.91	0.35	0.06	0.29	270.05
2002	174.07	27.17	146.89	37.18	13.13	24.05	0.82	0.63	0.19	212.07
2003	166.11	4.15	161.96	33.79	8.60	25.19	0.52	0.32	0.20	200.42
2004	178.47	25.91	152.55	34.46	10.19	24.27	2.33	2.03	0.30	215.26
2005	152.85	15.37	137.48	39.15	10.23	28.91	0.30	0.04	0.27	192.30
2006	157.36	60.15	97.21	27.91	12.38	15.53	1.46	0.88	0.58	186.73
2007	174.48	36.63	137.84	42.62	20.28	22.34	1.16	0.66	0.49	218.25
2008	141.14	39.30	101.84	47.84	26.37	21.47	0.91	0.79	0.12	189.90
2009	143.75	18.66	125.09	39.55	18.87	20.68	0.52	0.18	0.35	183.82
2010	183.10	28.67	154.43	43.00	25.81	17.19	0.70	0.39	0.32	226.80
2011	174.02	29.57	144.45	52.98	34.27	18.70	0.77	0.44	0.32	227.76
2012	160.62	23.91	136.71	63.91	40.01	23.90	1.97	0.80	1.18	226.50
2013	131.02	32.70	98.32	37.10	19.77	17.32	0.92	0.54	0.38	169.04
2014	131.10	29.90	101.20	41.06	20.57	20.49	2.07	1.70	0.37	174.23
2015	143.50	28.80	114.70	45.52	24.73	20.79	1.77	1.17	0.61	190.79
2016	137.40	45.00	92.40	43.60	25.44	18.16	6.73	6.02	0.71	187.73
2017	128.92	58.45	70.47	45.97	28.54	17.42	2.32	1.97	0.35	177.21
2018	141.31	57.72	83.59	49.76	33.09	16.68	2.00	1.76	0.24	193.08
2019	150.82	45.78	105.05	58.94	40.10	18.83	1.17	1.05	0.11	210.92
2020	124.60	52.55	72.05	60.24	42.93	17.31	1.79	1.51	0.28	186.63
2021	136.69	59.62	77.07	59.36	41.54	17.82	1.75	1.57	0.19	197.80
2022	136.70	38.70	98.00	60.42	41.33	19.09	3.86	3.49	0.37	200.98
2023	131.80	26.00	105.80	48.55	34.38	14.17	1.40	1.37	0.03	181.75

Table A2. Catch-at-age for the northern commercial reduction fishery from 1955-2023.

Year	0	1	2	3	4	5	6+	# of fish sampled
1955	0	0.015	0.471	0.217	0.253	0.032	0.012	8408
1956	0	0.133	0.555	0.195	0.025	0.072	0.020	11050
1957	0	0.270	0.610	0.051	0.033	0.017	0.020	11247
1958	0	0.025	0.908	0.042	0.010	0.008	0.009	8777
1959	0	0.531	0.291	0.159	0.009	0.004	0.007	10470
1960	0	0.009	0.892	0.037	0.049	0.009	0.004	9346
1961	0	0.003	0.160	0.803	0.012	0.018	0.003	8059
1962	0	0.015	0.245	0.218	0.457	0.033	0.032	9598
1963	0	0.296	0.438	0.095	0.068	0.080	0.023	6058
1964	0	0.034	0.357	0.345	0.128	0.065	0.072	4619
1965	0	0.160	0.370	0.373	0.071	0.013	0.014	6564
1966	0	0.201	0.467	0.212	0.100	0.009	0.012	1859
1967	0	0.055	0.296	0.567	0.072	0.009	0.000	1840
1968	0	0.007	0.479	0.388	0.116	0.009	0.001	5701
1969	0	0.001	0.251	0.594	0.149	0.005	0	3621
1970	0	0.150	0.793	0.050	0.007	0	0	700
1971	0	0.126	0.288	0.433	0.137	0.017	0	760
1972	0	0.169	0.286	0.452	0.085	0.008	0	759
1973	0	0.021	0.821	0.133	0.024	0.001	0	729
1974	0	0.028	0.844	0.117	0.006	0.004	0	1280
1975	0	0	0.798	0.175	0.025	0.001	0	1850
1976	0	0.092	0.823	0.071	0.013	0	0	2010
1977	0	0.022	0.567	0.326	0.079	0.006	0.001	2200
1978	0	0	0.298	0.567	0.120	0.015	0	1861
1979	0	0.007	0.579	0.332	0.076	0.006	0	1688
1980	0	0.002	0.237	0.462	0.243	0.051	0.004	1744
1981	0	0.001	0.357	0.357	0.210	0.070	0.006	2220
1982	0	0.042	0.393	0.473	0.063	0.025	0.004	840
1983	0	0.012	0.826	0.120	0.037	0.005	0	840
1984	0	0.024	0.343	0.506	0.097	0.029	0.001	3110
1985	0	0.020	0.760	0.089	0.111	0.017	0.003	1490
1986	0	0.010	0.795	0.107	0.050	0.031	0.006	530
1987	0	0.005	0.652	0.277	0.058	0.006	0.002	940
1988	0	0	0.225	0.486	0.260	0.026	0.003	1650
1989	0	0.081	0.623	0.173	0.097	0.025	0	1360

Table A2. *Continued*

Year	0	1	2	3	4	5	6+	# of fish sampled
1990	0	0.011	0.788	0.134	0.049	0.018	0.001	1660
1991	0	0.085	0.430	0.385	0.072	0.023	0.005	1460
1992	0	0.058	0.687	0.107	0.118	0.026	0.004	1180
1993	0	0.045	0.675	0.226	0.036	0.017	0.002	640
1994	0	0.017	0.420	0.333	0.183	0.047	0	300
1995	0	0.020	0.567	0.329	0.079	0.006	0	710
1996	0	0	0.579	0.320	0.092	0.008	0	500
1997	0	0	0.495	0.293	0.158	0.055	0	130
1998	0	0	0.657	0.281	0.062	0	0	100
1999	0	0	0.389	0.428	0.168	0.015	0	120
2000	0	0.005	0.559	0.406	0.019	0.011	0	490
2001	0	0	0.150	0.796	0.055	0	0	380
2002	0	0.040	0.347	0.491	0.120	0.002	0	290
2003	0	0	0.474	0.378	0.139	0.010	0	90
2004	0	0.004	0.615	0.320	0.061	0	0	290
2005	0	0	0.219	0.605	0.174	0.002	0	240
2006	0	0.022	0.456	0.422	0.099	0.001	0	1040
2007	0	0.022	0.761	0.174	0.041	0.002	0	520
2008	0	0.002	0.216	0.668	0.106	0.008	0	550
2009	0	0.123	0.299	0.463	0.102	0.013	0	240
2010	0	0	0.456	0.348	0.193	0.003	0	380
2011	0	0.058	0.726	0.190	0.023	0.003	0	410
2012	0	0.001	0.778	0.192	0.029	0	0	330
2013	0	0.028	0.724	0.233	0.015	0	0	370
2014	0	0.085	0.518	0.274	0.119	0.004	0	290
2015	0	0.006	0.593	0.362	0.038	0	0	390
2016	0	0.075	0.413	0.481	0.031	0	0	700
2017	0	0.017	0.572	0.393	0.015	0.003	0	1070
2018	0	0.088	0.680	0.211	0.021	0	0	590
2019	0.002	0.464	0.437	0.089	0.009	0	0	640
2020								0
2021	0	0.106	0.849	0.045	0	0	0	80
2022	0	0.155	0.752	0.086	0.007	0	0	140
2023	0.009	0.167	0.674	0.130	0.020	0	0	130

Table A3. Catch-at-age for the southern commercial reduction fishery from 1955-2023.

Year	0	1	2	3	4	5	6+	# of fish sampled
1955	0.374	0.323	0.269	0.016	0.016	0.002	0	7742
1956	0.017	0.885	0.049	0.018	0.004	0.022	0.004	8831
1957	0.151	0.598	0.217	0.010	0.011	0.007	0.006	8467
1958	0.059	0.466	0.443	0.018	0.005	0.005	0.004	7008
1959	0.003	0.855	0.099	0.034	0.005	0.002	0.002	7490
1960	0.052	0.192	0.701	0.018	0.025	0.008	0.004	4167
1961	0	0.538	0.217	0.234	0.004	0.007	0	5158
1962	0.040	0.387	0.491	0.033	0.044	0.003	0.002	6197
1963	0.079	0.460	0.386	0.059	0.007	0.008	0.002	6977
1964	0.187	0.433	0.349	0.028	0.002	0	0	5824
1965	0.184	0.528	0.269	0.018	0.001	0	0	13017
1966	0.265	0.414	0.299	0.020	0.001	0	0	13848
1967	0.007	0.663	0.269	0.057	0.003	0	0	13648
1968	0.143	0.349	0.468	0.037	0.003	0	0	21168
1969	0.188	0.442	0.330	0.038	0.002	0	0	11511
1970	0.016	0.650	0.309	0.022	0.003	0	0	7761
1971	0.083	0.288	0.569	0.054	0.005	0.001	0	7510
1972	0.033	0.618	0.285	0.061	0.003	0	0	5800
1973	0.036	0.372	0.591	0.001	0	0	0	5640
1974	0.196	0.388	0.413	0.003	0	0	0	4330
1975	0.154	0.371	0.469	0.006	0.001	0	0	5450
1976	0.101	0.572	0.324	0.003	0	0	0	4720
1977	0.140	0.289	0.567	0.003	0	0	0	5080
1978	0.158	0.230	0.558	0.050	0.003	0	0	5250
1979	0.413	0.172	0.403	0.012	0.001	0	0	4680
1980	0.028	0.476	0.452	0.038	0.004	0.001	0	5548
1981	0.316	0.186	0.460	0.038	0	0	0	7000
1982	0.038	0.306	0.558	0.096	0.001	0	0	8230
1983	0.279	0.148	0.547	0.016	0.008	0.001	0	4340
1984	0.396	0.311	0.244	0.040	0.007	0.002	0	8580
1985	0.235	0.394	0.364	0.006	0	0	0	6230
1986	0.056	0.126	0.797	0.019	0.002	0.001	0	4880
1987	0.022	0.253	0.691	0.031	0.003	0	0	6460
1988	0.175	0.146	0.573	0.099	0.006	0.001	0	5708
1989	0.069	0.514	0.402	0.014	0.001	0	0	5530

Table A3. *Continued*

Year	0	1	2	3	4	5	6+	# of fish sampled
1990	0.190	0.078	0.697	0.023	0.010	0.002	0	5180
1991	0.317	0.360	0.281	0.038	0.004	0.001	0	6230
1992	0.243	0.428	0.313	0.014	0.002	0	0	4430
1993	0.049	0.266	0.608	0.074	0.003	0	0	4680
1994	0.064	0.197	0.609	0.094	0.035	0.002	0	4410
1995	0.044	0.408	0.366	0.150	0.031	0.002	0	3900
1996	0.036	0.226	0.630	0.092	0.015	0.001	0	3720
1997	0.027	0.260	0.423	0.236	0.047	0.007	0.001	3970
1998	0.073	0.187	0.535	0.123	0.073	0.009	0.001	3740
1999	0.188	0.292	0.428	0.069	0.020	0.003	0	3500
2000	0.140	0.205	0.510	0.127	0.016	0.002	0	2550
2001	0.039	0.073	0.604	0.265	0.018	0.001	0	3540
2002	0.242	0.284	0.321	0.140	0.012	0	0	3310
2003	0.088	0.185	0.643	0.073	0.010	0.001	0	3400
2004	0.020	0.234	0.670	0.060	0.015	0.001	0	3880
2005	0.020	0.131	0.618	0.210	0.018	0.003	0	3290
2006	0.016	0.525	0.378	0.072	0.008	0	0	2530
2007	0.001	0.306	0.631	0.054	0.008	0	0	3270
2008	0.017	0.115	0.812	0.053	0.003	0	0	2220
2009	0.007	0.515	0.311	0.147	0.019	0.001	0	2590
2010	0.017	0.447	0.494	0.034	0.008	0	0	2890
2011	0	0.477	0.467	0.048	0.007	0.002	0	2820
2012	0.007	0.183	0.789	0.020	0.001	0	0	2300
2013	0.043	0.457	0.388	0.095	0.016	0	0	1760
2014	0.007	0.482	0.377	0.106	0.026	0.002	0	1790
2015	0	0.141	0.759	0.092	0.009	0	0	2170
2016	0.022	0.303	0.509	0.160	0.006	0	0	1800
2017	0	0.249	0.581	0.144	0.026	0	0	1280
2018	0.036	0.334	0.479	0.136	0.015	0	0	1520
2019	0.002	0.755	0.202	0.037	0.004	0.001	0	1620
2020	0.0	0.177	0.819	0.003	0	0	0	450
2021	0.0	0.831	0.167	0.002	0.001	0	0	660
2022	0	0.530	0.412	0.047	0.010	0	0	1320
2023	0.010	0.322	0.608	0.056	0.004	0	0	915

Table A4. Catch-at-age for the northern commercial bait fishery (includes MRIP estimates of recreational catch).

Year	0	1	2	3	4	5	6+	# of fish sampled
1985	0.000	0.010	0.754	0.116	0.093	0.022	0.006	0
1986	0.000	0.001	0.207	0.563	0.116	0.091	0.023	0
1987	0.000	0.002	0.215	0.531	0.226	0.016	0.010	0
1988	0.000	0.000	0.070	0.521	0.363	0.041	0.004	0
1989	0.000	0.010	0.216	0.374	0.310	0.089	0.001	30
1990	0.000	0.003	0.534	0.262	0.144	0.053	0.005	0
1991	0.000	0.012	0.228	0.553	0.143	0.051	0.012	0
1992	0.000	0.025	0.335	0.212	0.330	0.079	0.019	0
1993	0.000	0.008	0.327	0.494	0.099	0.065	0.008	29
1994	0.000	0.000	0.098	0.505	0.347	0.045	0.004	401
1995	0.000	0.000	0.088	0.475	0.435	0.001	0.000	190
1996	0.000	0.000	0.413	0.442	0.137	0.008	0.000	203
1997	0.000	0.000	0.144	0.324	0.396	0.118	0.018	111
1998	0.000	0.000	0.103	0.379	0.420	0.084	0.013	225
1999	0.000	0.000	0.149	0.479	0.318	0.043	0.011	201
2000	0.000	0.004	0.415	0.315	0.229	0.030	0.007	266
2001	0.000	0.000	0.112	0.735	0.135	0.014	0.004	678
2002	0.000	0.000	0.053	0.552	0.336	0.058	0.000	524
2003	0.000	0.000	0.127	0.661	0.201	0.011	0.000	101
2004	0.000	0.007	0.438	0.381	0.161	0.013	0.000	29
2005	0.000	0.002	0.188	0.626	0.162	0.022	0.000	0
2006	0.000	0.004	0.278	0.566	0.147	0.001	0.004	259
2007	0.000	0.000	0.382	0.482	0.126	0.008	0.002	729
2008	0.000	0.000	0.262	0.585	0.139	0.013	0.000	973
2009	0.000	0.000	0.204	0.608	0.175	0.013	0.000	435
2010	0.000	0.000	0.365	0.380	0.228	0.025	0.002	466
2011	0.000	0.000	0.142	0.486	0.327	0.045	0.000	449
2012	0.000	0.000	0.392	0.468	0.130	0.008	0.002	547
2013	0.000	0.000	0.257	0.555	0.159	0.029	0.000	236
2014	0.000	0.000	0.066	0.525	0.387	0.020	0.002	806
2015	0.000	0.002	0.377	0.522	0.099	0.000	0.000	1291
2016	0.000	0.021	0.392	0.528	0.053	0.007	0.000	1018
2017	0.000	0.017	0.566	0.380	0.036	0.001	0.000	1487
2018	0.000	0.000	0.274	0.595	0.121	0.010	0.000	331
2019	0.000	0.037	0.356	0.446	0.142	0.015	0.004	837
2020	0.000	0.007	0.684	0.255	0.046	0.007	0.002	754
2021	0.000	0.018	0.546	0.283	0.134	0.019	0.000	471
2022	0.000	0.064	0.578	0.264	0.085	0.009	0.000	467
2023	0.000	0.132	0.435	0.352	0.077	0.005	0.000	428

Table A5. Catch-at-age for the southern commercial bait fishery (includes MRIP estimates of recreational catch).

Year	0	1	2	3	4	5	6+	# of fish sampled
1985	0.004	0.310	0.661	0.016	0.007	0.002	0.000	800
1986	0.001	0.064	0.860	0.066	0.006	0.003	0.001	420
1987	0.001	0.089	0.836	0.068	0.006	0.000	0.000	220
1988	0.004	0.060	0.663	0.232	0.038	0.003	0.000	10
1989	0.004	0.341	0.577	0.063	0.013	0.003	0.000	0
1990	0.005	0.061	0.903	0.026	0.003	0.001	0.000	10
1991	0.012	0.301	0.595	0.084	0.005	0.001	0.000	78
1992	0.000	0.554	0.446	0.000	0.000	0.000	0.000	70
1993	0.008	0.357	0.530	0.097	0.006	0.003	0.000	121
1994	0.001	0.142	0.650	0.150	0.052	0.005	0.000	139
1995	0.000	0.392	0.374	0.217	0.017	0.000	0.000	174
1996	0.000	0.006	0.757	0.199	0.037	0.000	0.000	156
1997	0.000	0.055	0.531	0.346	0.056	0.008	0.004	293
1998	0.036	0.065	0.539	0.237	0.108	0.012	0.003	411
1999	0.000	0.105	0.663	0.174	0.052	0.006	0.000	338
2000	0.008	0.222	0.659	0.112	0.000	0.000	0.000	270
2001	0.004	0.043	0.658	0.275	0.017	0.004	0.000	286
2002	0.000	0.047	0.265	0.494	0.173	0.020	0.002	180
2003	0.007	0.095	0.740	0.142	0.015	0.000	0.000	328
2004	0.000	0.066	0.733	0.167	0.031	0.003	0.000	327
2005	0.000	0.008	0.515	0.447	0.027	0.003	0.000	316
2006	0.000	0.327	0.451	0.197	0.024	0.000	0.000	220
2007	0.000	0.243	0.671	0.067	0.019	0.000	0.000	434
2008	0.005	0.044	0.809	0.112	0.017	0.013	0.000	366
2009	0.004	0.241	0.367	0.341	0.047	0.000	0.000	573
2010	0.003	0.306	0.527	0.102	0.059	0.002	0.000	435
2011	0.000	0.338	0.470	0.121	0.051	0.020	0.000	508
2012	0.000	0.068	0.825	0.085	0.017	0.002	0.002	408
2013	0.007	0.449	0.289	0.173	0.054	0.027	0.000	434
2014	0.000	0.437	0.365	0.138	0.055	0.005	0.000	559
2015	0.010	0.309	0.589	0.089	0.002	0.000	0.000	251
2016	0.000	0.225	0.423	0.324	0.021	0.007	0.000	205
2017	0.000	0.267	0.496	0.229	0.008	0.000	0.000	137
2018	0.000	0.328	0.446	0.166	0.060	0.001	0.000	280
2019	0.000	0.580	0.250	0.125	0.039	0.003	0.003	684
2020	0.000	0.004	0.023	0.972	0.000	0.000	0.000	65
2021	0.000	0.271	0.256	0.424	0.043	0.005	0.000	266
2022	0.005	0.334	0.492	0.124	0.040	0.006	0.000	233
2023	0.049	0.146	0.523	0.199	0.062	0.013	0.009	262

Table A6. Young-of-year abundance index (YOY), northern adult index (NAD), Mid-Atlantic adult index (MAD), and southern adult index (SAD) of abundance for Atlantic menhaden developed from the Conn method with associated coefficients of variation (CV).

Year	YOY		NAD		MAD		SAD	
	Index	CV	Index	CV	Index	CV	Index	CV
1959	1.60	1.03						
1960	0.47	1.07						
1961	0.42	1.10						
1962	1.60	1.03						
1963	1.24	1.08						
1964	0.80	1.15						
1965	0.49	1.06						
1966	0.64	1.09						
1967	0.53	1.10						
1968	0.48	0.90						
1969	1.28	0.90						
1970	0.30	0.90						
1971	1.62	0.86						
1972	2.29	0.84						
1973	2.39	1.00						
1974	2.68	0.99						
1975	2.83	1.00						
1976	2.77	0.98						
1977	2.76	1.01						
1978	1.45	0.99						
1979	2.11	1.00						
1980	3.20	0.83						
1981	2.45	1.01						
1982	3.05	0.84						
1983	1.74	0.99						
1984	1.53	0.98						
1985	2.64	0.86			1.88	1.09		
1986	2.27	0.76			1.87	1.13		
1987	0.41	0.72			2.06	1.13		
1988	1.06	0.69			1.94	1.11		
1989	1.54	0.59			1.21	1.12		

Table A6. *Continued*

Year	YOY		NAD		MAD		SAD	
	Index	CV	Index	CV	Index	CV	Index	CV
1990	0.71	0.51	0.53	0.67	0.93	1.12	3.12	0.75
1991	0.76	0.50	0.31	0.67	0.74	1.15	1.23	0.67
1992	0.52	0.51	0.92	0.63	1.34	1.11	0.92	0.66
1993	0.20	0.55	0.77	0.62	0.55	1.18	0.57	0.70
1994	0.32	0.52	0.50	0.63	1.46	1.12	0.36	0.79
1995	0.26	0.51	1.15	0.64	1.38	1.11	0.18	0.81
1996	0.25	0.51	0.59	0.56	0.54	1.16	0.28	0.77
1997	0.28	0.50	0.32	0.58	0.54	1.17	0.24	0.75
1998	0.50	0.50	0.60	0.65	0.78	0.36	0.85	0.68
1999	0.56	0.53	0.78	0.58	0.60	0.39	0.27	0.77
2000	0.43	0.48	0.48	0.63	0.83	0.39	0.84	0.74
2001	0.37	0.46	0.80	0.67	0.95	0.34	0.93	0.77
2002	0.53	0.44	1.59	0.58	0.46	0.39	1.00	0.66
2003	0.86	0.45	0.30	0.63	1.05	0.32	0.86	0.59
2004	0.65	0.44	0.39	0.66	0.52	0.34	0.47	0.57
2005	0.74	0.44	1.12	0.55	1.31	0.36	1.56	0.53
2006	0.28	0.44	1.00	0.54	0.45	0.37	3.31	0.50
2007	0.49	0.44	1.33	0.55	0.87	0.37	0.46	0.58
2008	0.32	0.44	1.20	0.55	0.39	0.39	0.68	0.39
2009	0.29	0.42	0.41	0.57	0.90	0.36	1.60	0.61
2010	0.47	0.45	0.81	0.68	0.97	0.36	0.90	0.37
2011	0.33	0.45	0.83	0.65	0.65	0.33	1.29	0.39
2012	0.17	0.45	1.70	0.54	0.59	0.39	1.32	0.38
2013	0.20	0.43	0.58	0.58	0.91	0.36	1.09	0.36
2014	0.43	0.43	1.44	0.56	1.60	0.34	1.15	0.38
2015	0.31	0.45	1.59	0.57	1.89	0.40	1.81	0.39
2016	0.61	0.45	1.17	0.56	0.57	0.39	0.56	0.43
2017	0.15	0.46	0.58	0.60	0.45	0.37	1.21	0.43
2018	0.28	0.44	0.34	0.63	1.22	0.61	1.14	0.40
2019	0.23	0.46	1.83	0.55	1.00	0.39	0.84	0.38
2020	0.23	0.50	2.47	0.67	0.34	0.43	0.96	0.77
2021	0.51	0.47	2.40	0.60	1.16	0.45	0.99	0.47
2022	0.67	0.50	2.28	0.60	1.60	0.38	0.42	0.39
2023	0.43	0.48	0.92	0.62	0.51	0.44	0.60	0.49

Table A7. List of surveys used in the Conn indices and their associated sigma (σ^p) values, or the standard deviation of the process error. Benchmark and update values are provided for comparison.

	Survey	2019 Benchmark	2022 Update	2025 Update
Age 1+ Surveys	CT Long Island Sound Trawl	0.96	1.90	1.20
	DE Adult Trawl	0.88	0.44	0.60
	NJ Ocean Trawl	1.53	1.15	0.80
	MD Striped Bass Spring Gill Net	2.23	2.22	2.10
	VIMS Shad and River Herring Monitoring	0.24	0.21	0.20
	NC Program 915 Pamlico Sound Gill Net	0.92	0.71	0.50
	SEAMAP	0.4	0.52	0.50
	GA Ecological Monitoring Trawl	0.5	0.73	0.90
YOY Surveys	RI Coastal Trawl	2.96	2.94	2.90
	CT River Juvenile Alosine Seine	2.5	2.52	2.70
	CT Thames River Seine	3.16	3.16	3.20
	CT Long Island Sound Trawl	1.34	1.28	1.70
	NY Peconic Bay Small Mesh Trawl	3.78	3.58	2.20
	NY Western Long Island Seine	2.99	3.10	3.00
	NY Juvenile Striped Bass Beach Seine	1.18	2.09	2.10
	NJ Ocean Trawl	1.85	1.89	1.90
	NJ Delaware River Striped Bass Seine	1.81	1.81	1.60
	DE Inland Bays	11.34	4.93	4.90
	MD Coastal Bays Trawl	2.17	1.33	4.50
	MD Juvenile Striped Bass Seine	1.64	1.44	1.50
	VIMS Juvenile Fish and Blue Crab Trawl	1.31	1.22	1.30
	VIMS Juvenile Striped Bass Seine	3.05	1.50	1.30
	NC Program 120 Estuarine Trawl	0.82	1.00	1.00
	SC Electrofishing	0.92	0.97	0.90

Appendix Figures

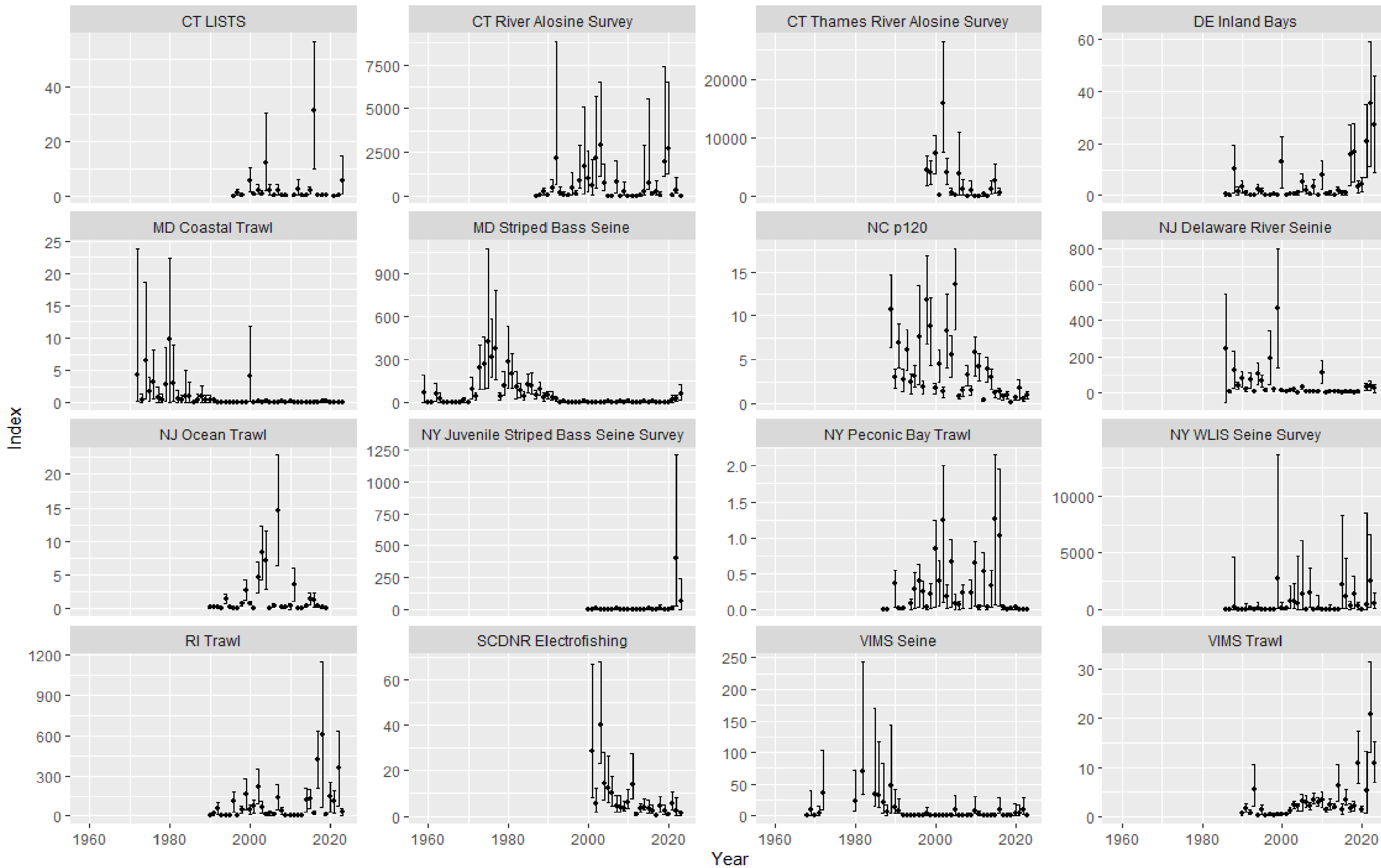


Figure A1. Individual YOY indices with 95% confidence intervals used in the coastwide YOY index.

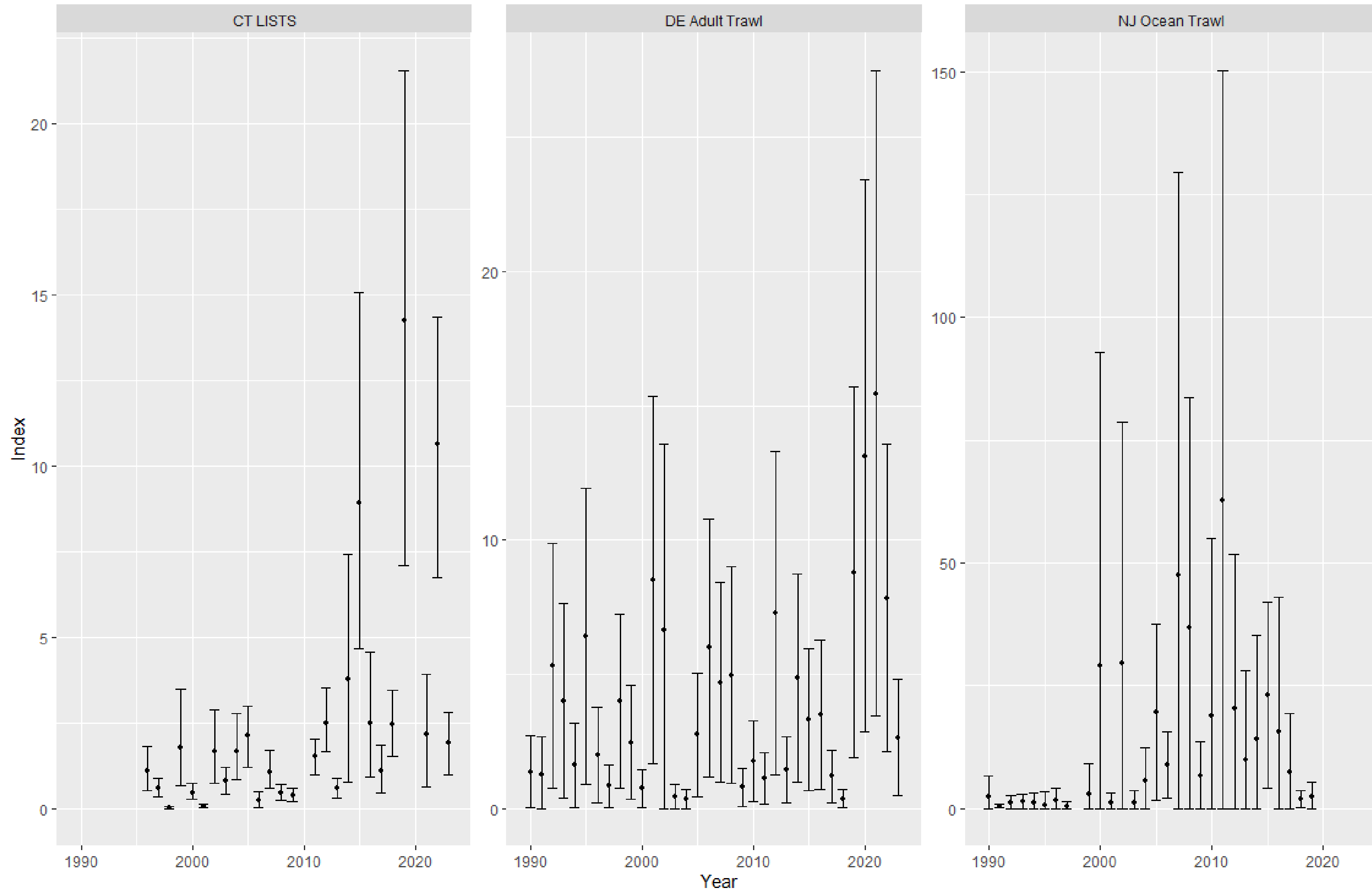


Figure A2. Individual adult indices with 95% confidence intervals used in the NAD index.

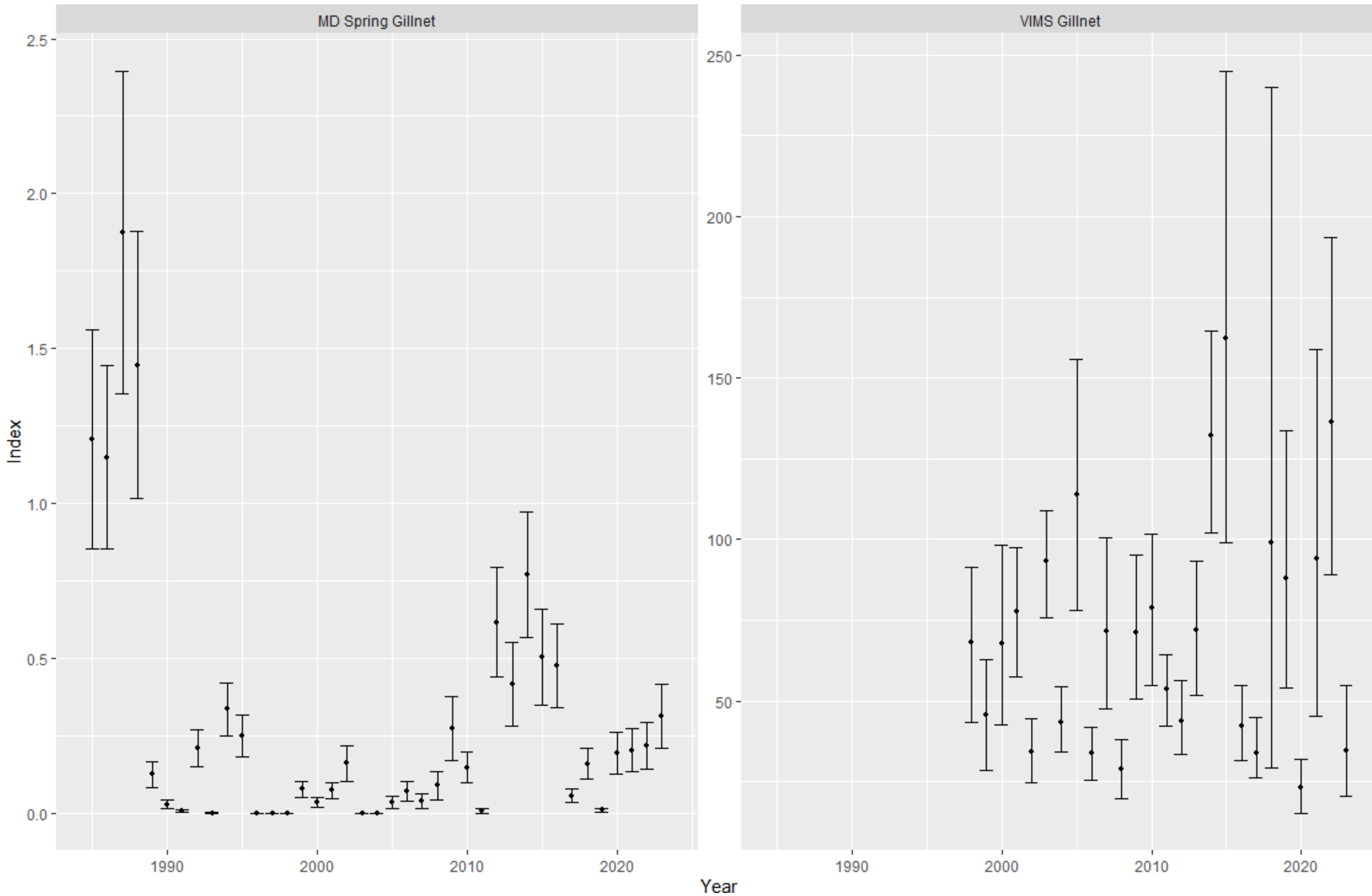


Figure A3. Individual adult indices with 95% confidence intervals used in the MAD index.

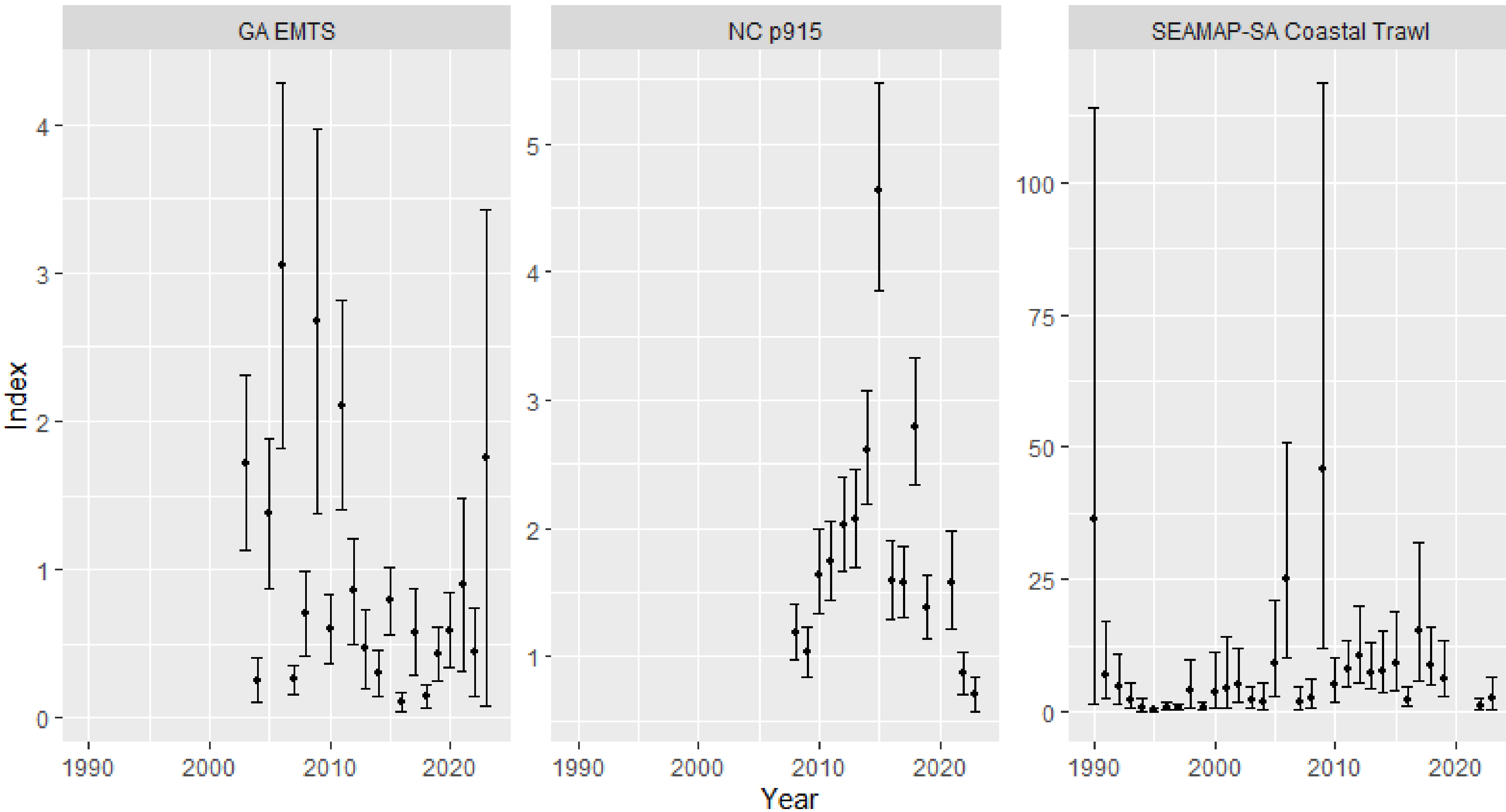


Figure A4. Individual adult indices with 95% confidence intervals used in the SAD index

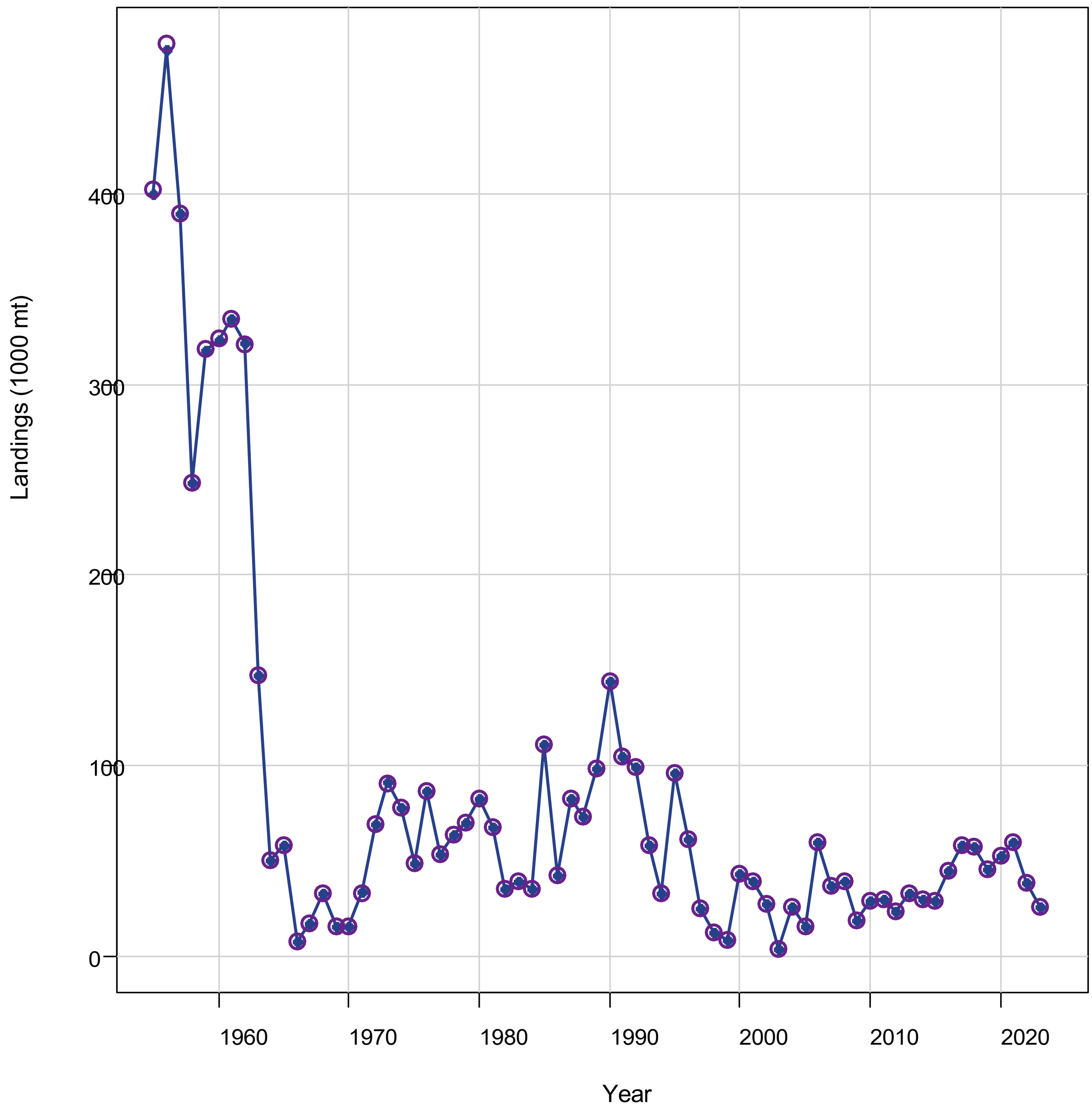


Figure A5. Predicted fit to the observed landings for the commercial reduction north fleet for 1955-2023. Predicted = solid circles + line; observed = open circles.

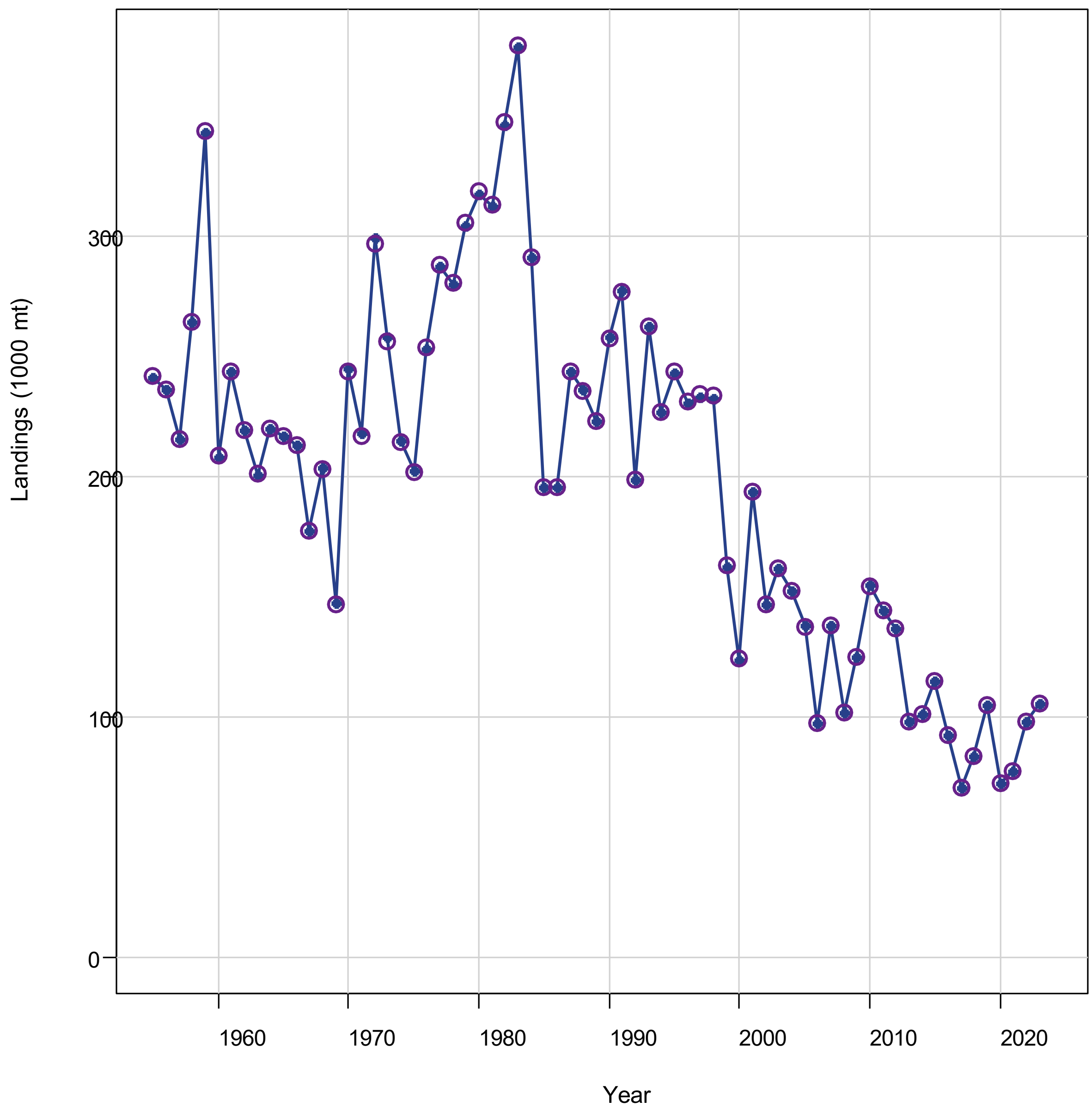


Figure A6. Predicted fit to the observed landings for the commercial reduction south fleet for 1955-2023. Predicted = solid circles + line; observed = open circles.

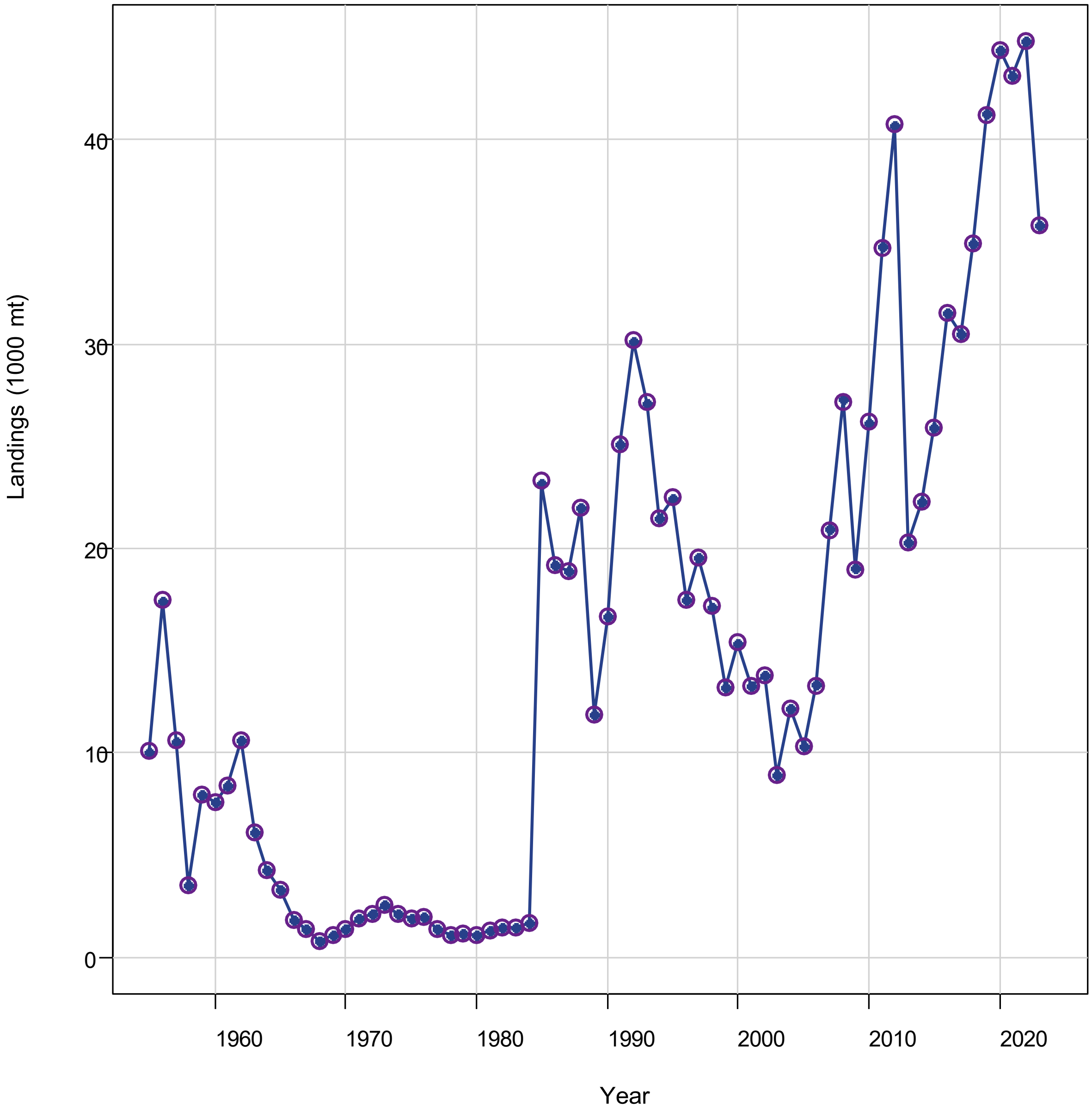


Figure A7. Predicted fit to the observed landings for the commercial bait north fleet for 1955-2023. Predicted = solid circles + line; observed = open circles.

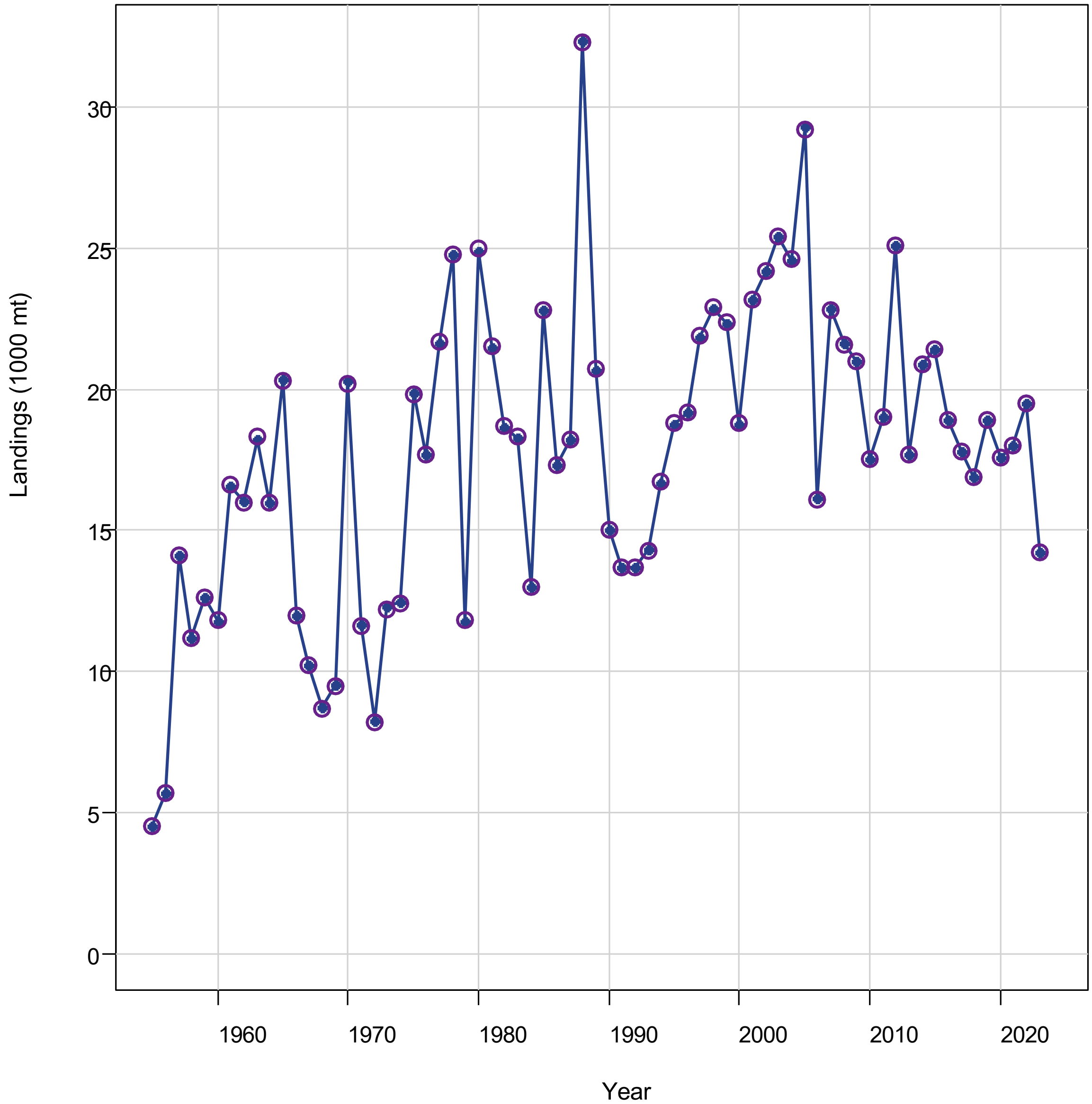


Figure A8. Predicted fit to the observed landings for the commercial bait south fleet for 1955-2023. Predicted = solid circles + line; observed = open circles.

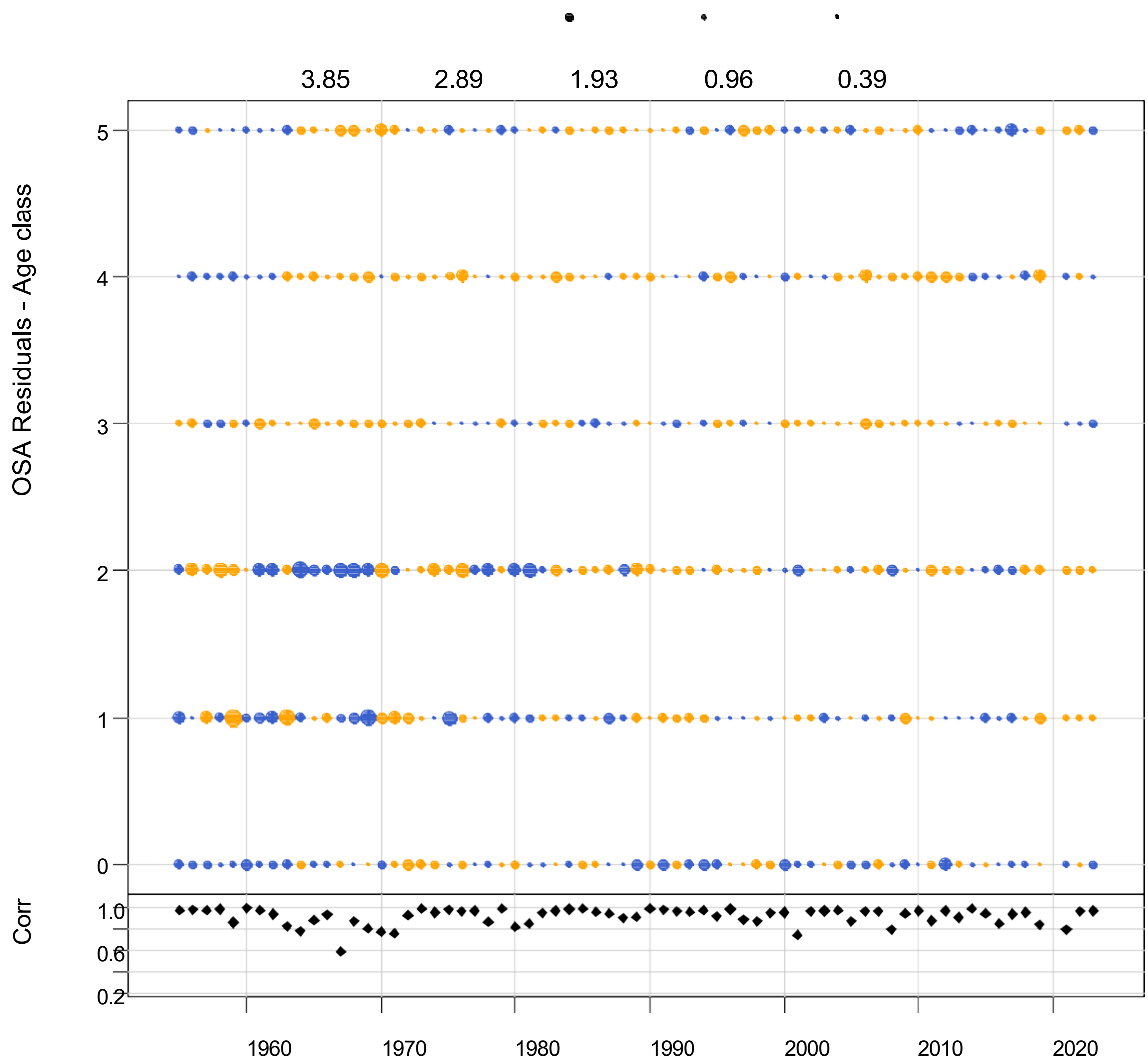


Figure A9. Bubble plot of the fits to the age compositions for the commercial reduction north fleet. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.

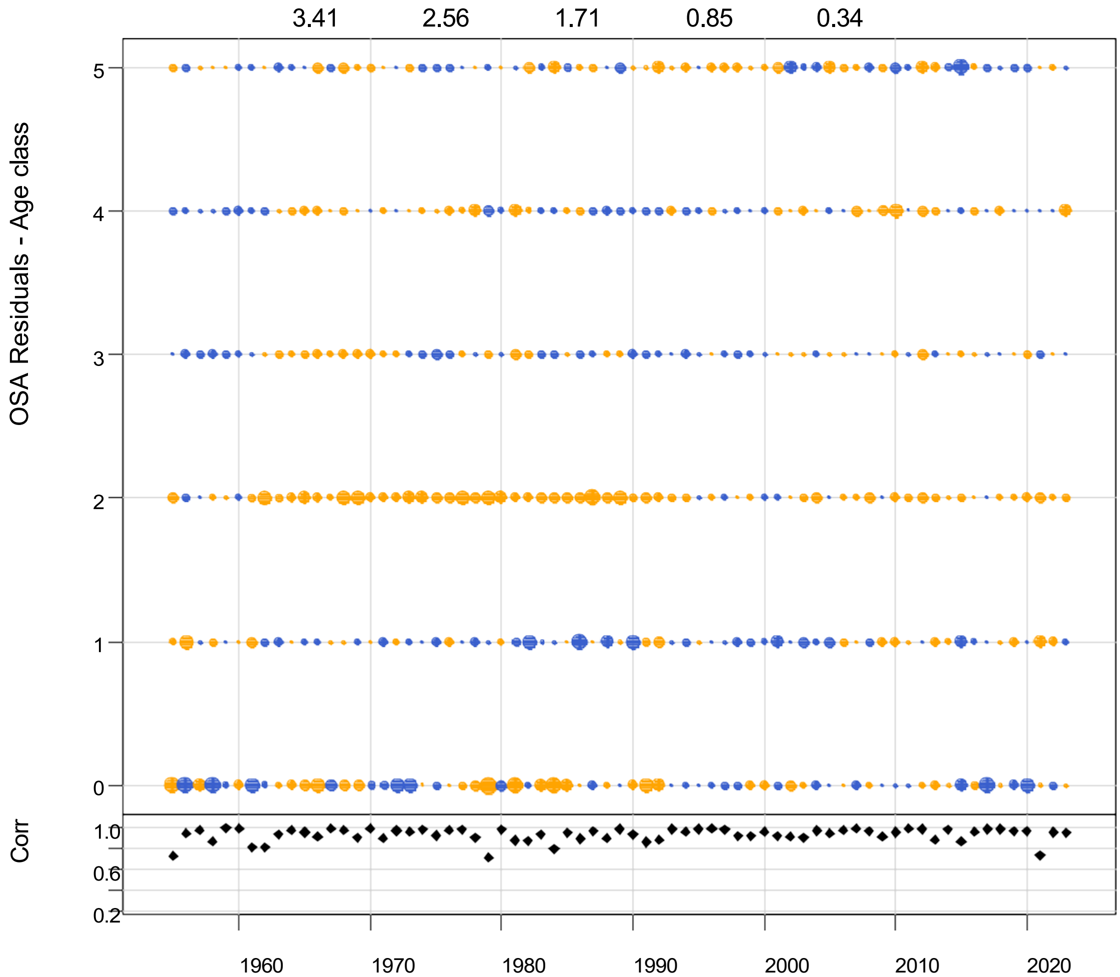


Figure A10. **Bubble plot of the fits to the age compositions for the commercial reduction south fleet. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.**

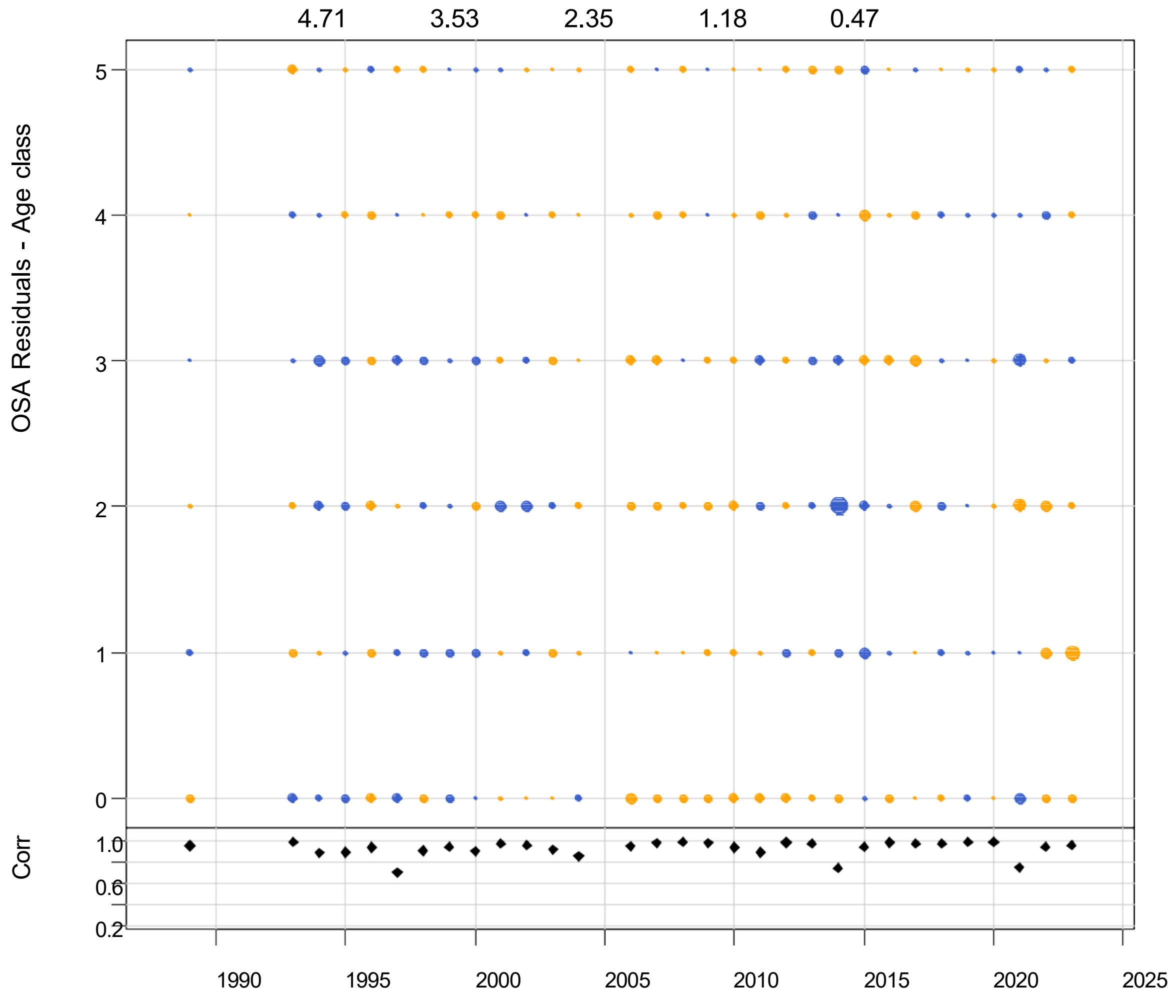


Figure A11. **Bubble plot of the fits to the age compositions for the commercial bait north fleet. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.**

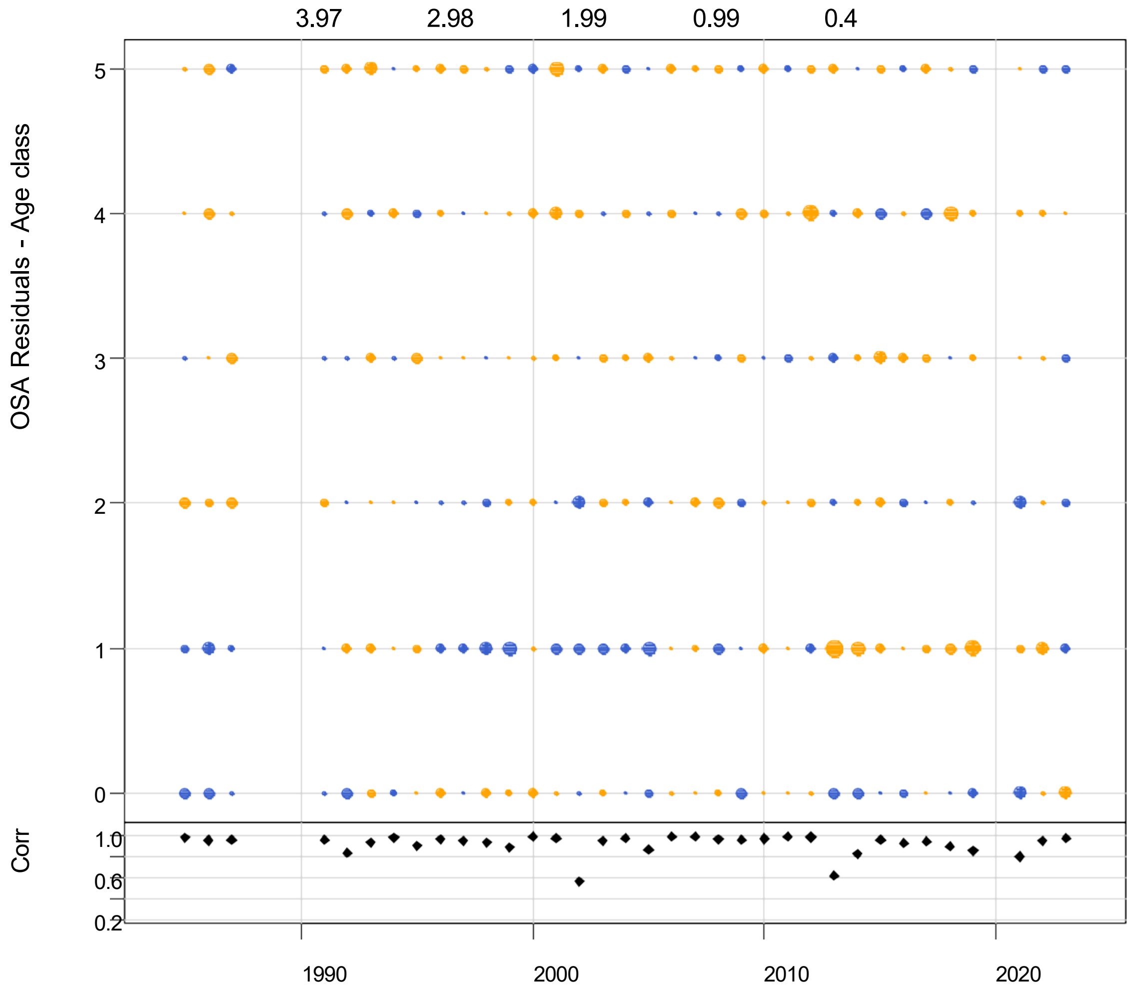


Figure A12. **Bubble plot of the fits to the age compositions for the commercial bait south fleet. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.**

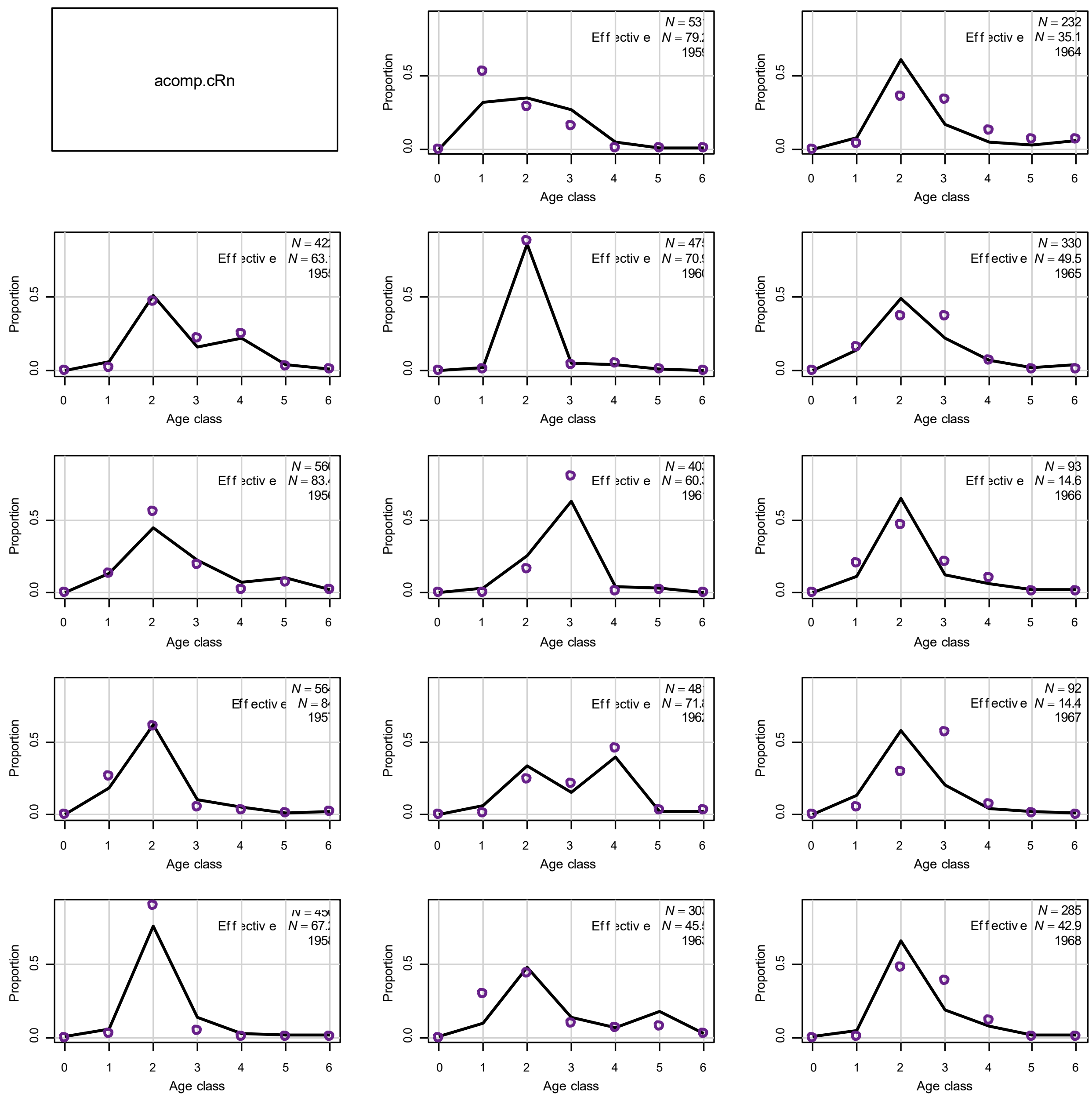
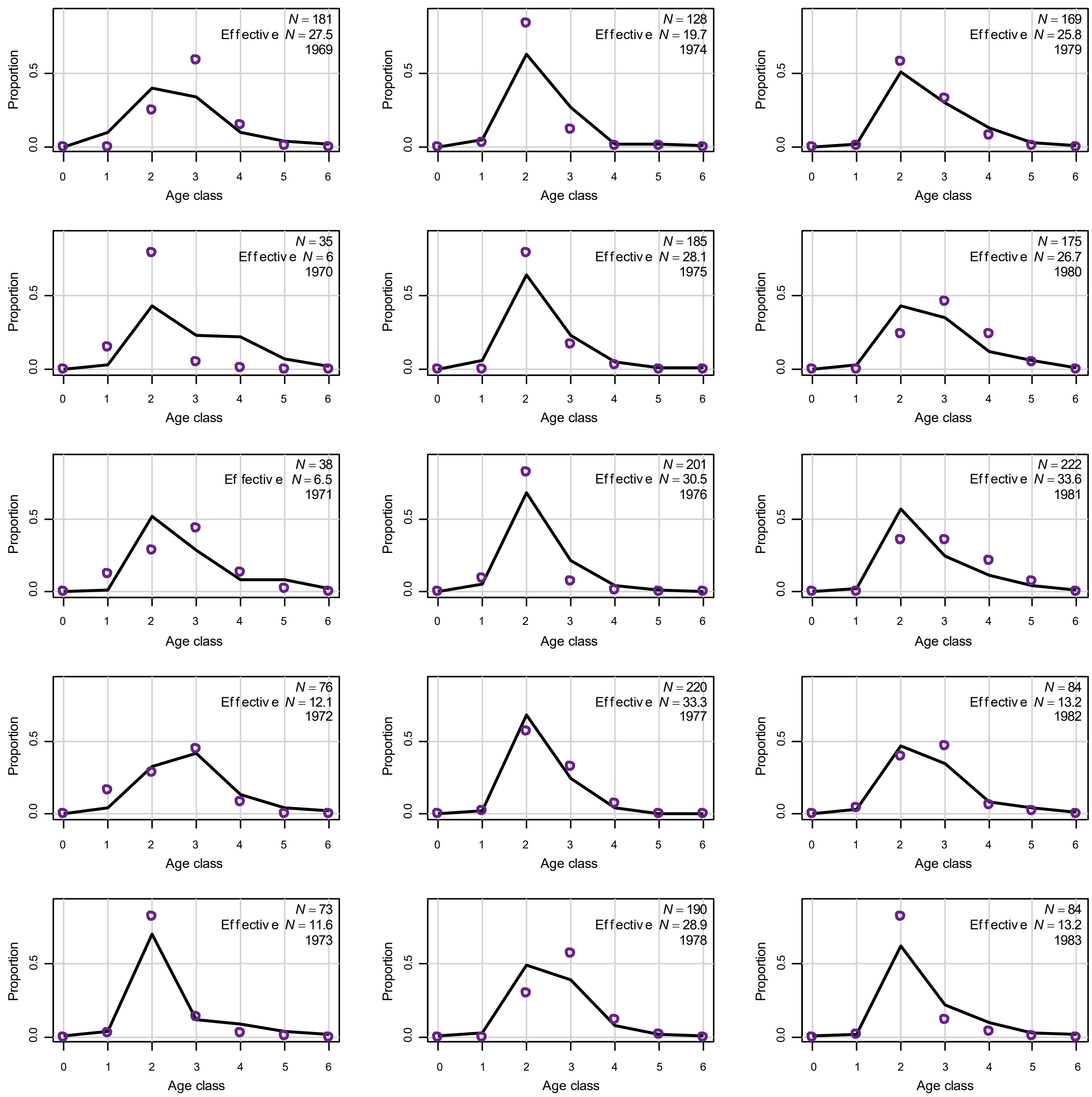


Figure A13. Annual age composition plots for the commercial reduction north fleet for 1955-2023. Open circles are the observed data, while the line indicates the model fit.



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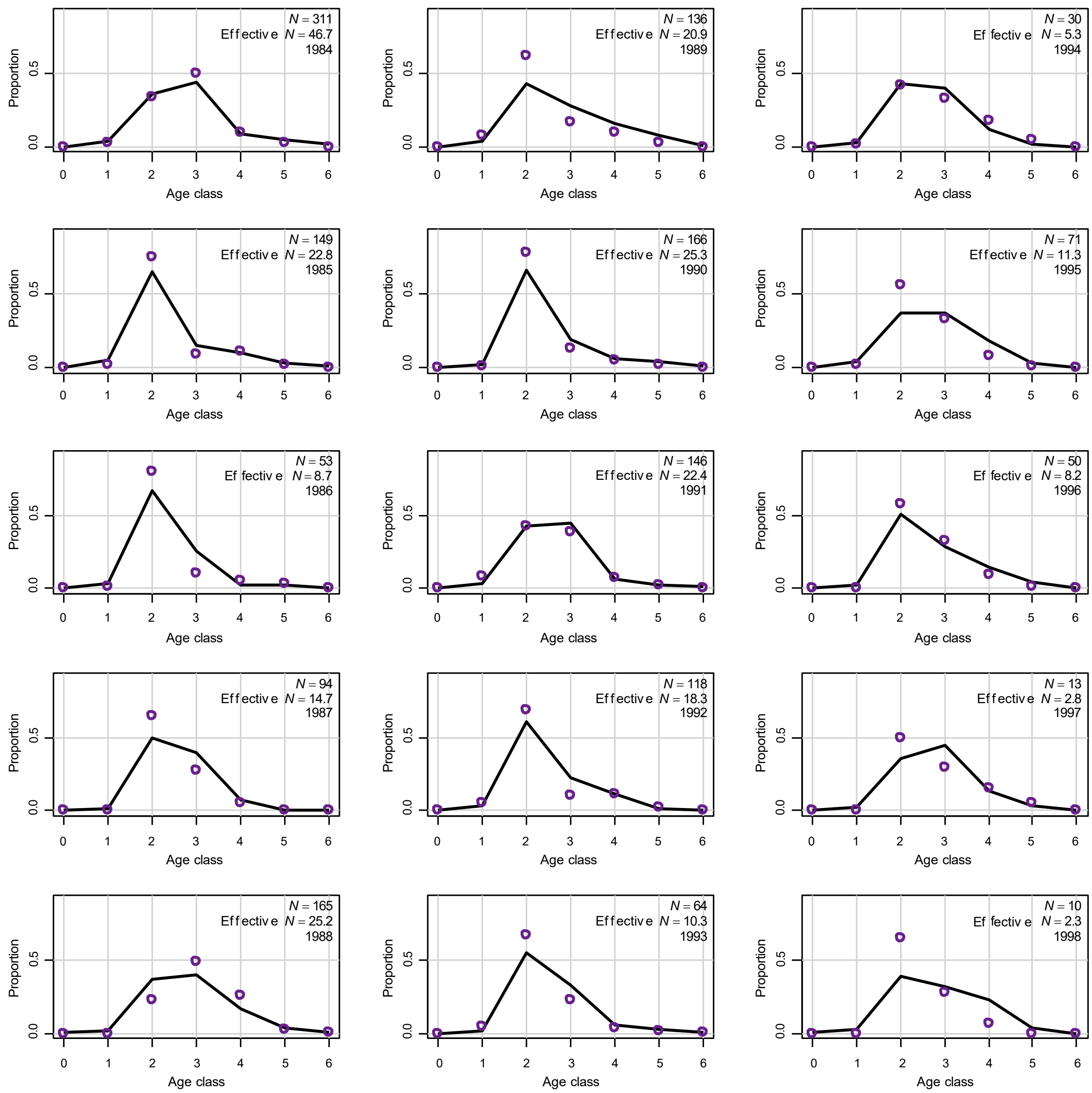

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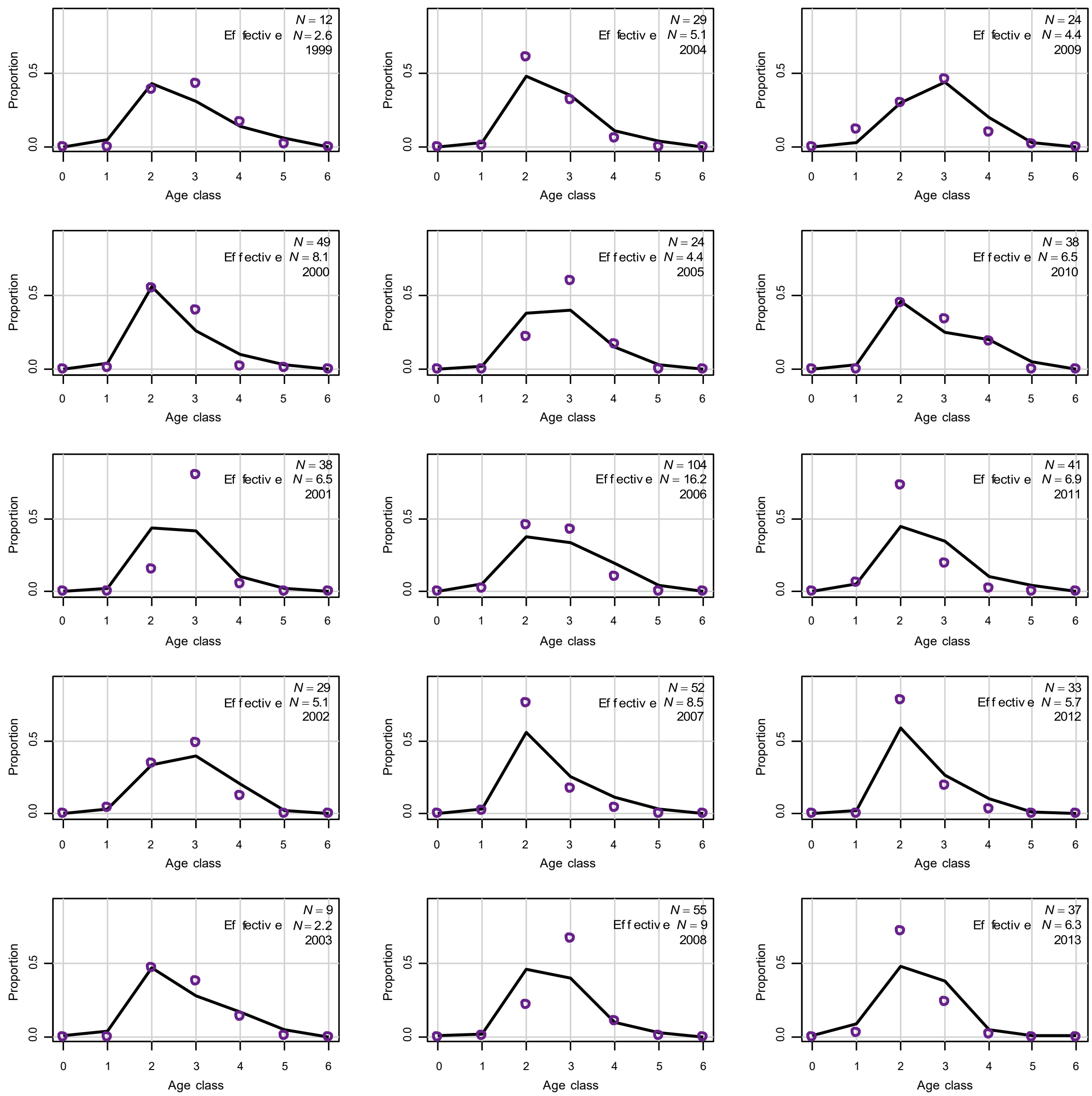

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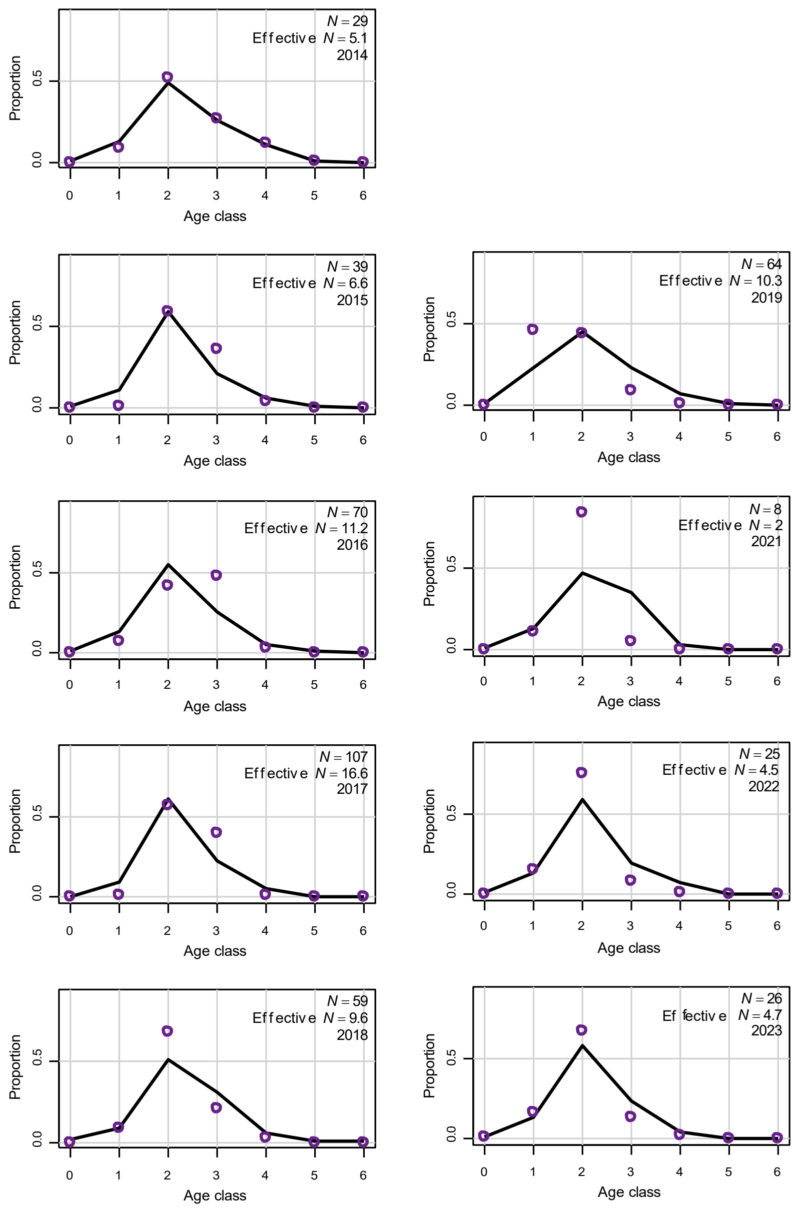

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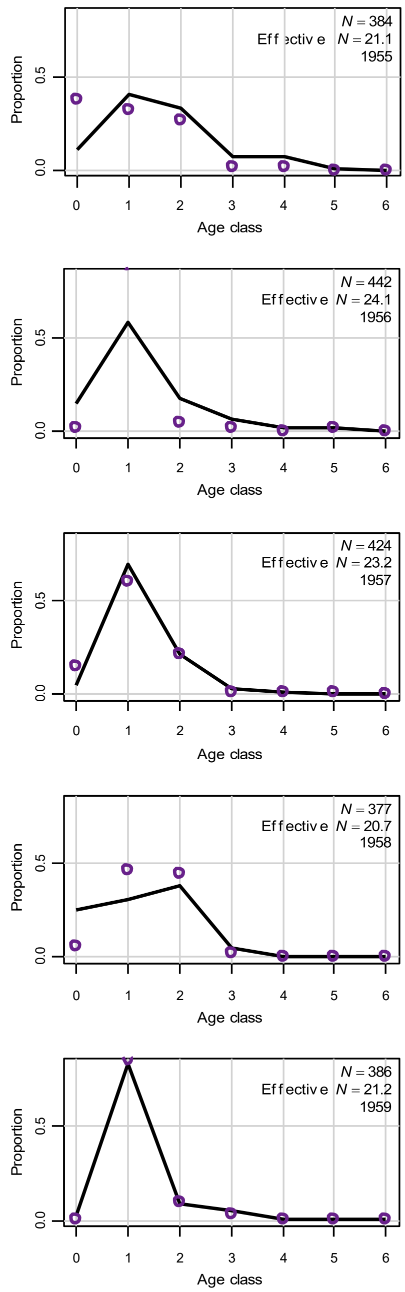
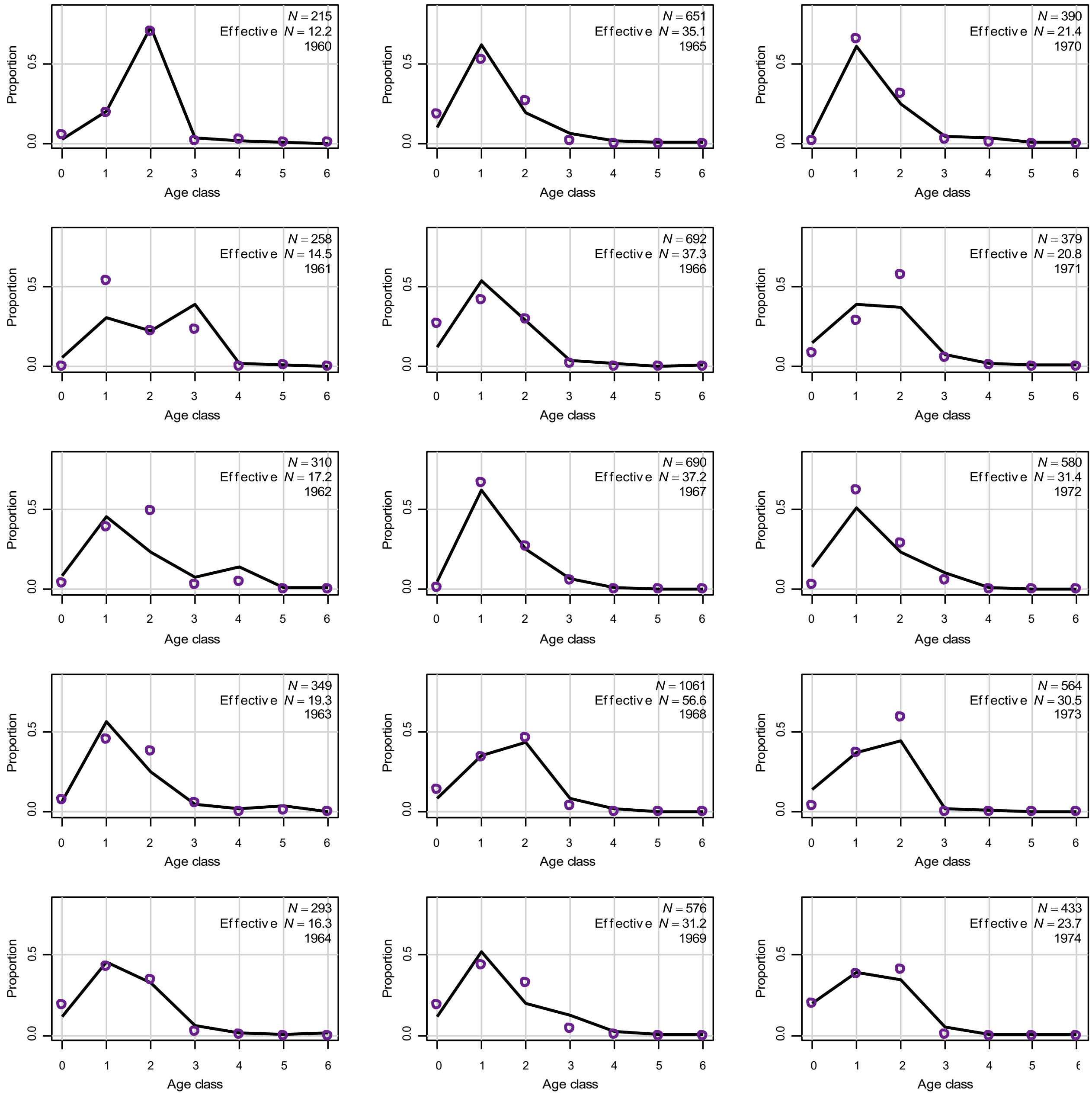


Figure A14. **Annual age composition plots for the commercial reduction south fleet for 1955-2023. Open circles are the observed data, while the line indicates the model fit.**



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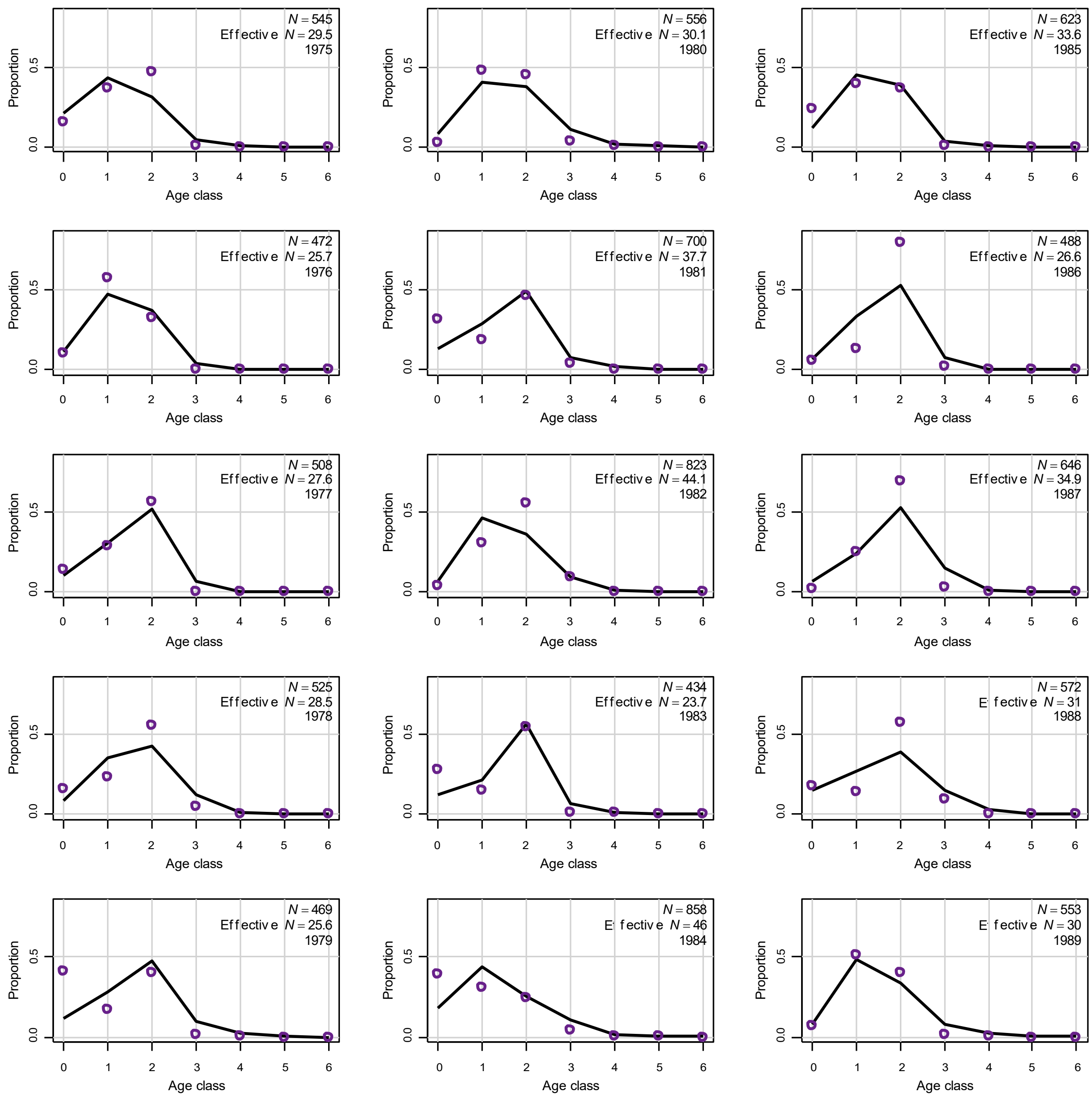

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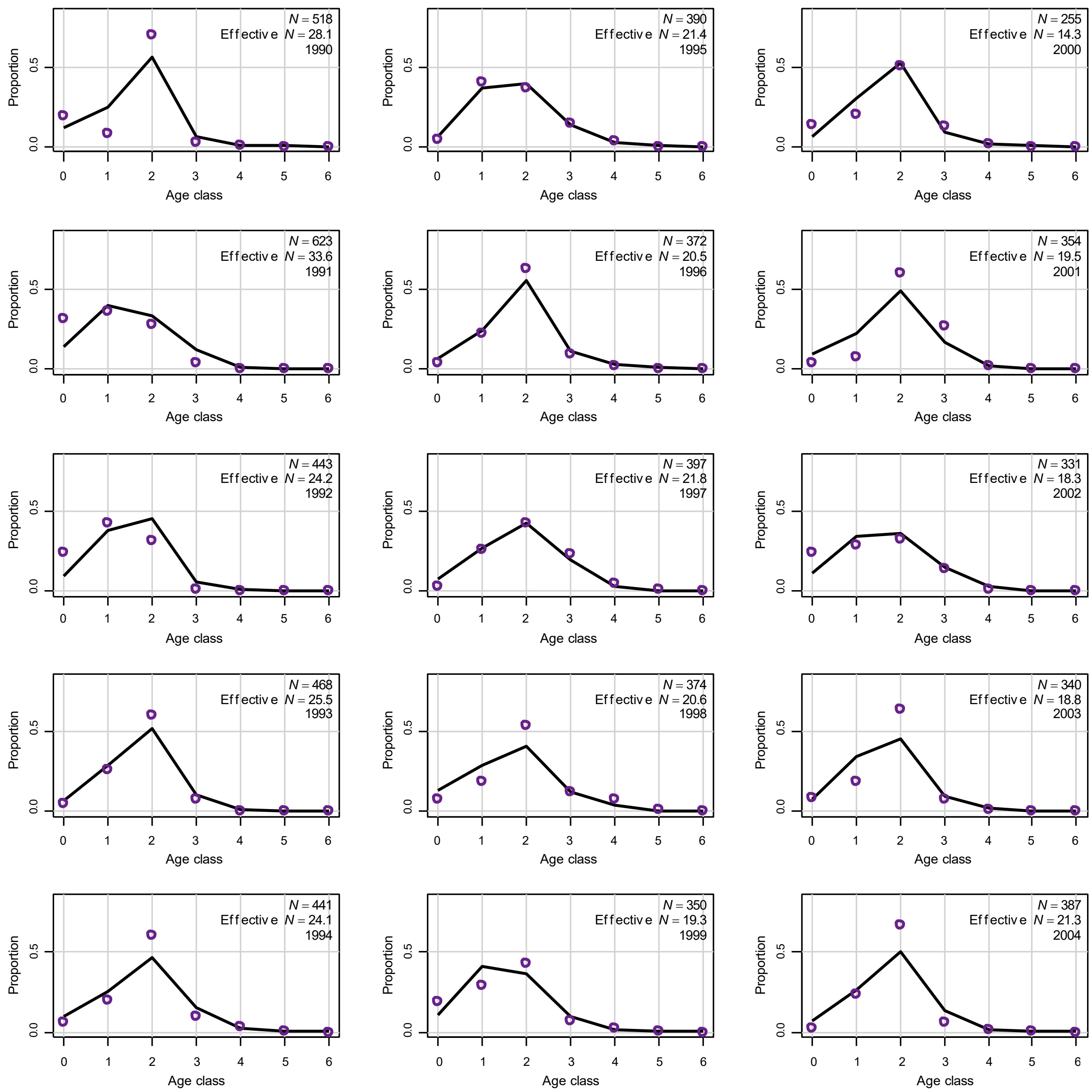

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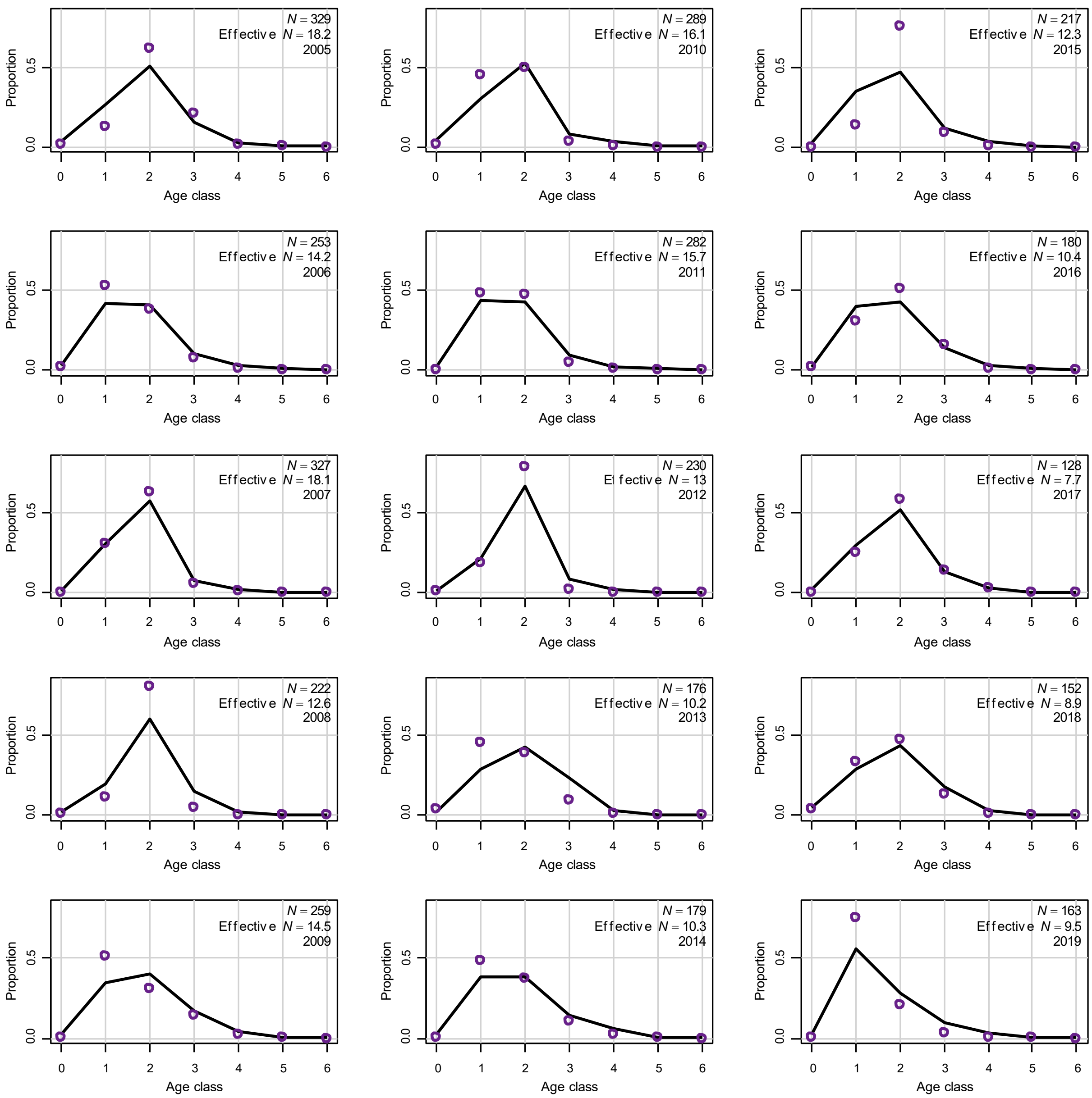

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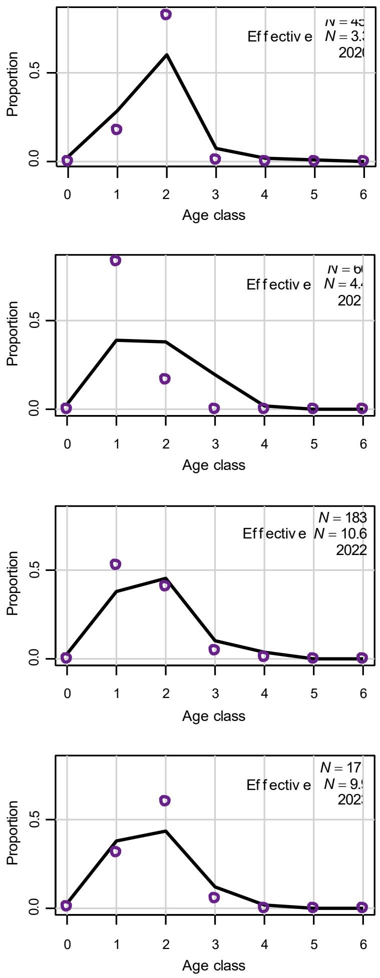

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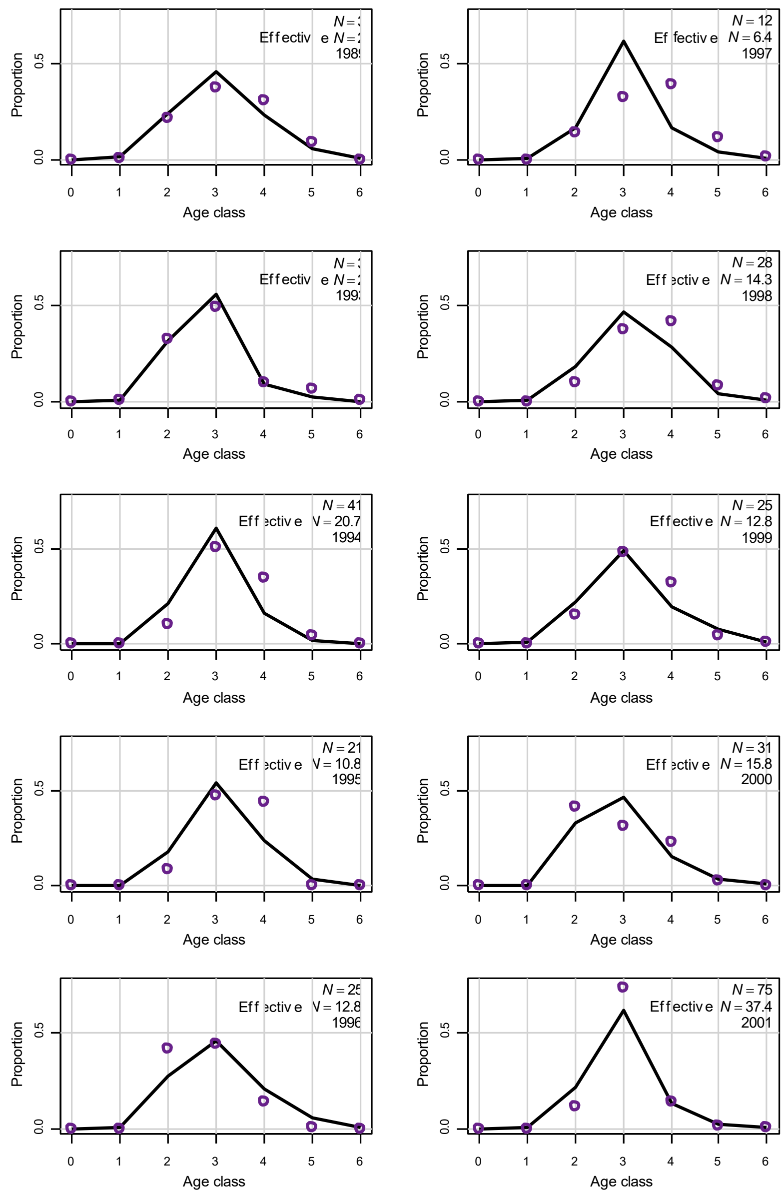
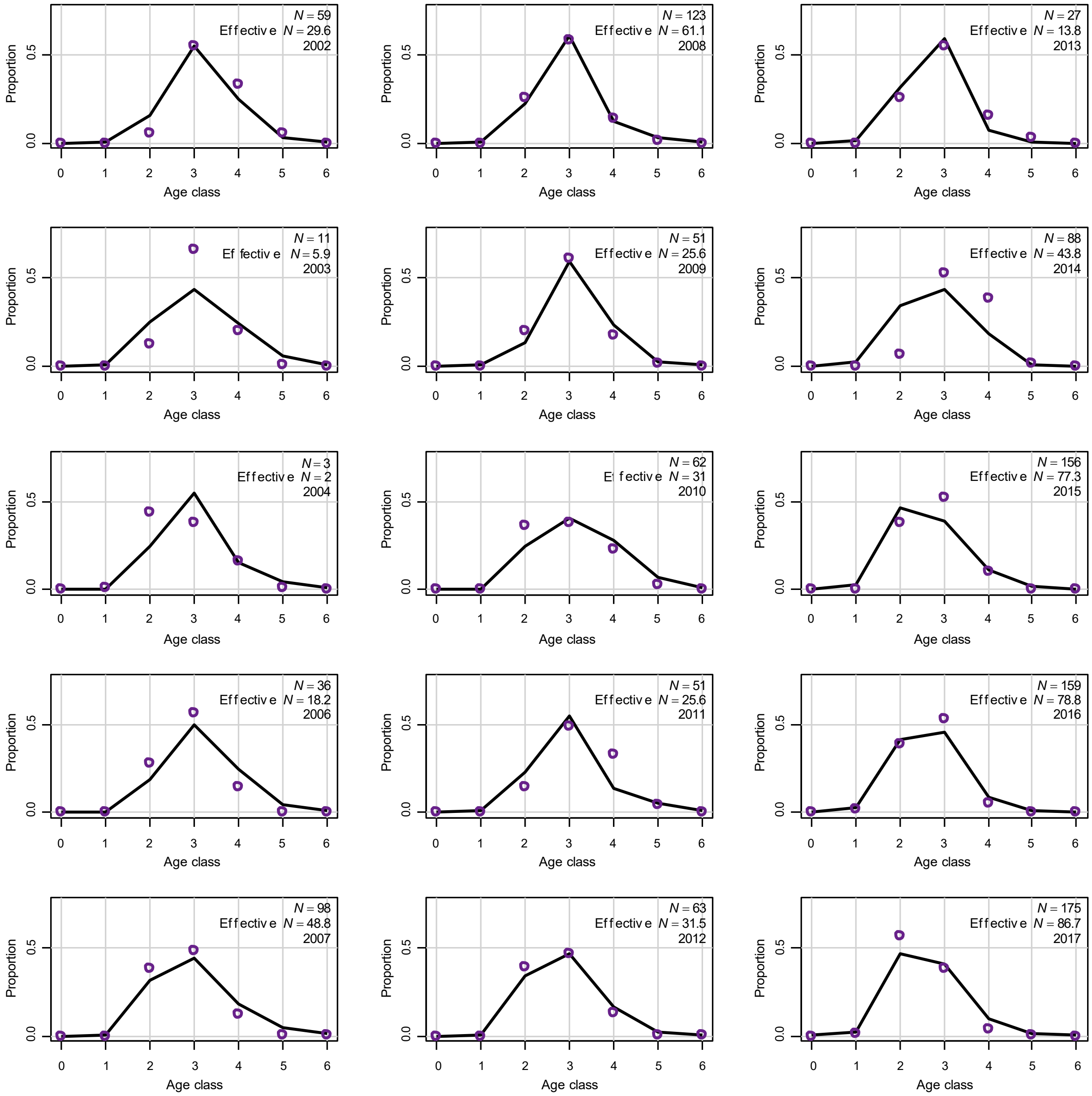


Figure A15. Annual age composition plots for the commercial bait north fleet for 1985-2023. Open circles are the observed data, while the line indicates the model fit.



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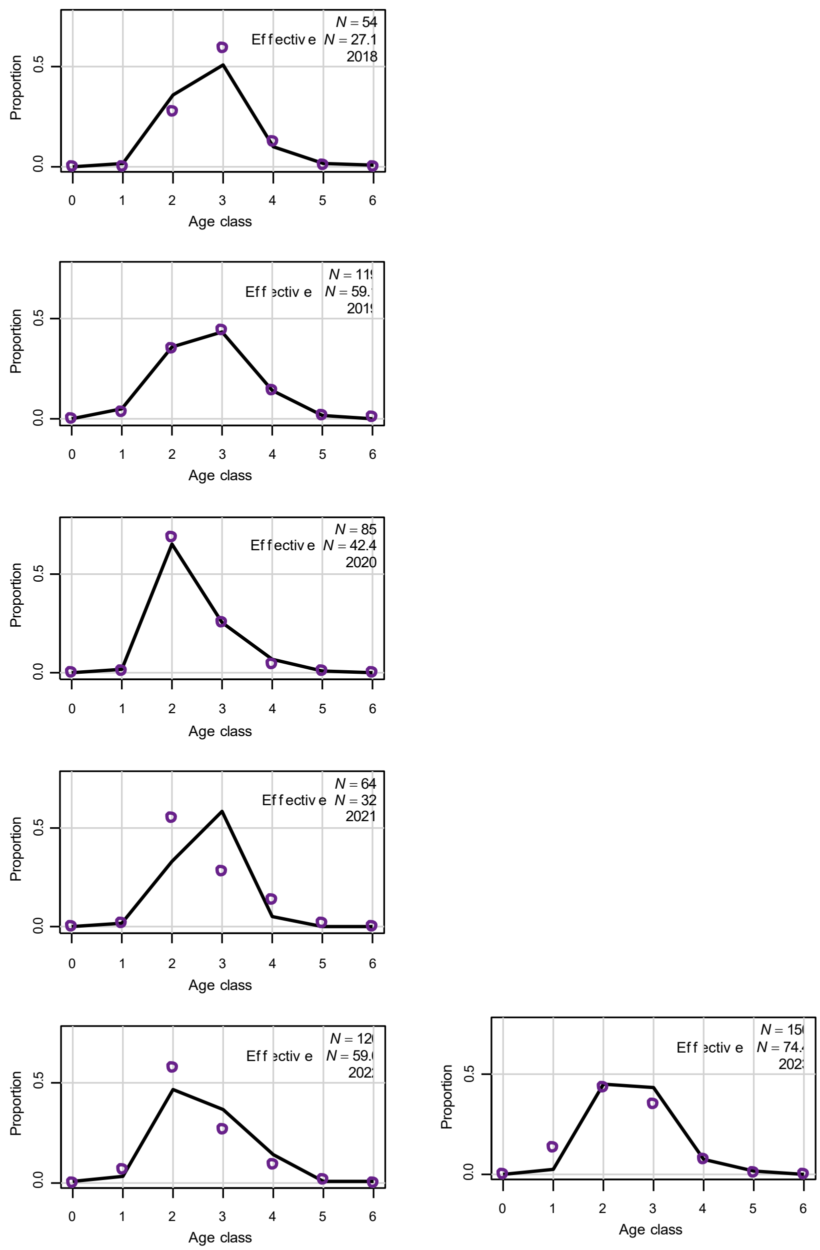

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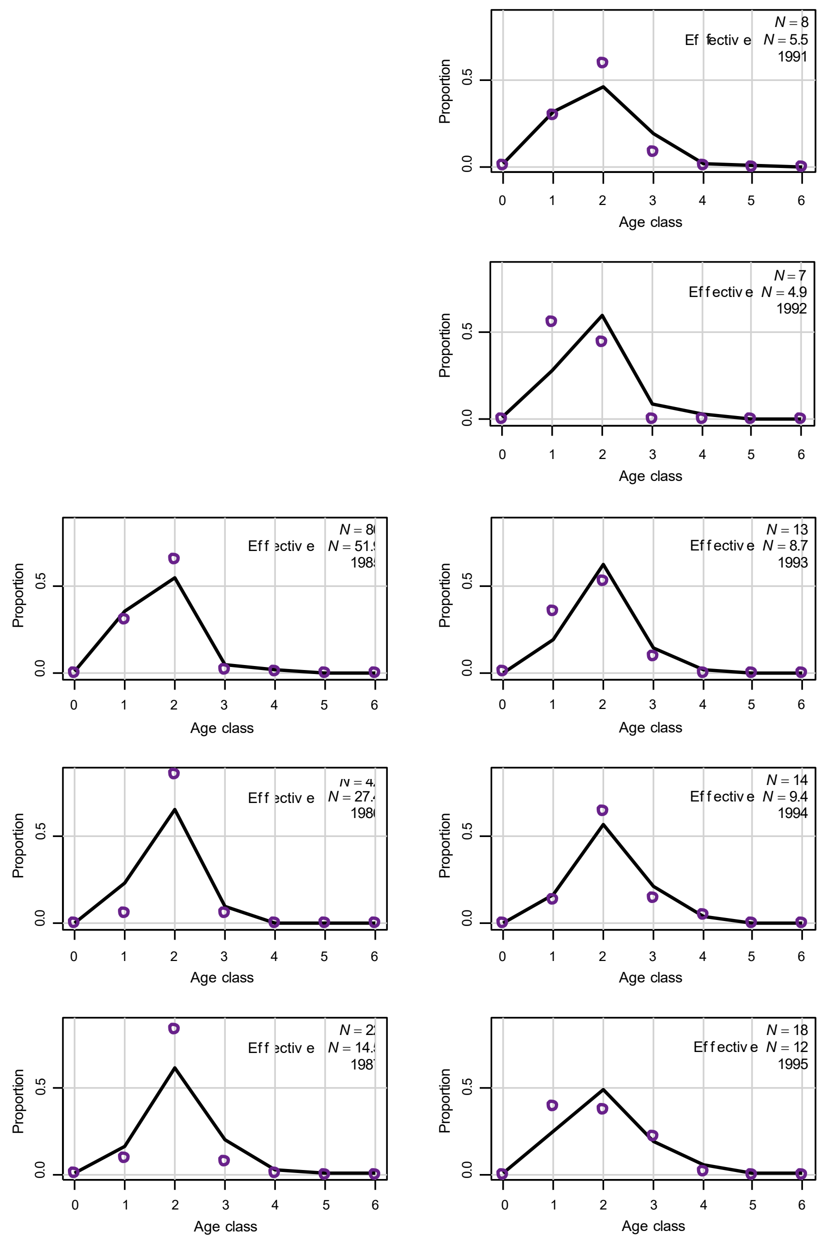
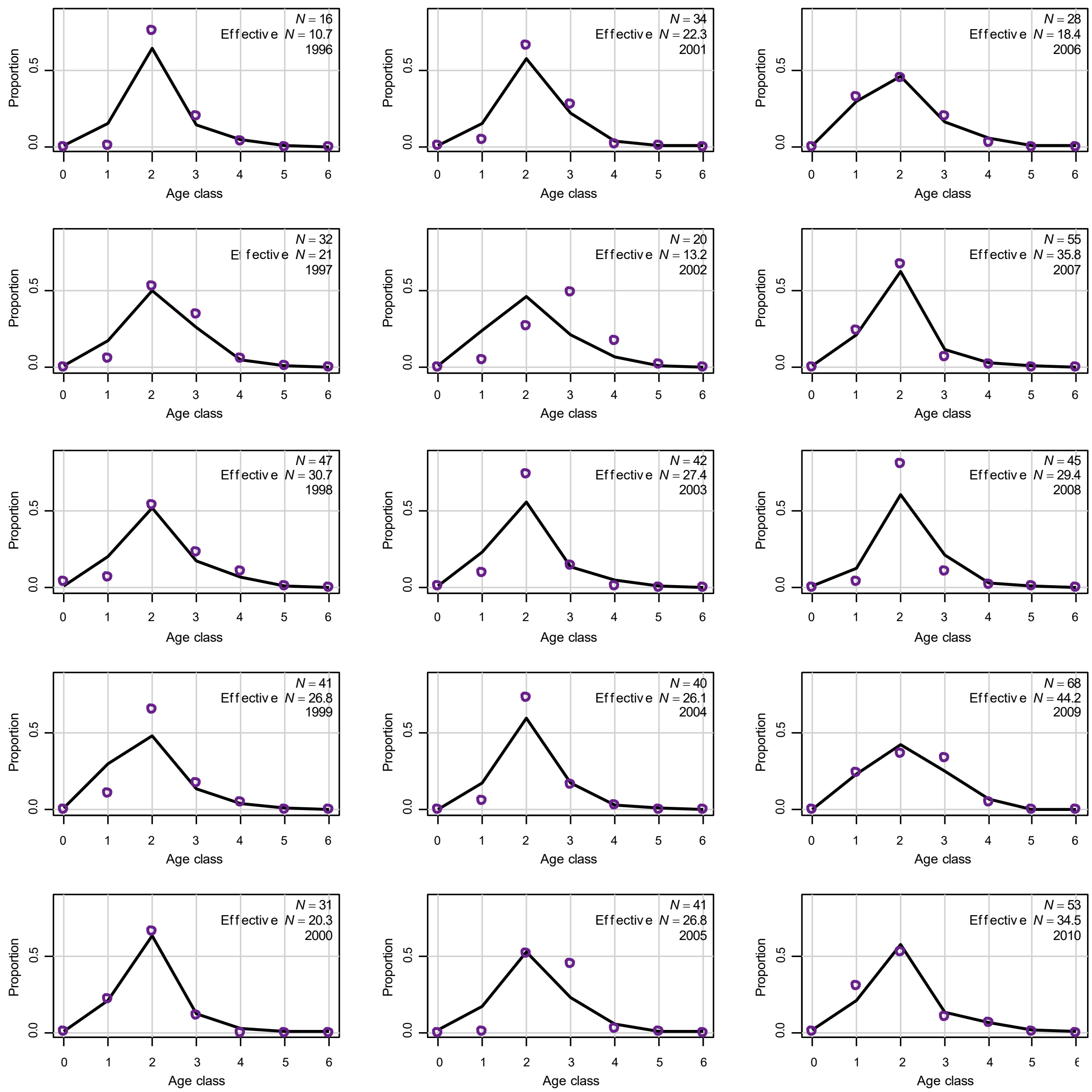


Figure A16. **Annual age composition plots for the commercial bait south fleet for 1985-2023. Open circles are the observed data, while the line indicates the model fit.**



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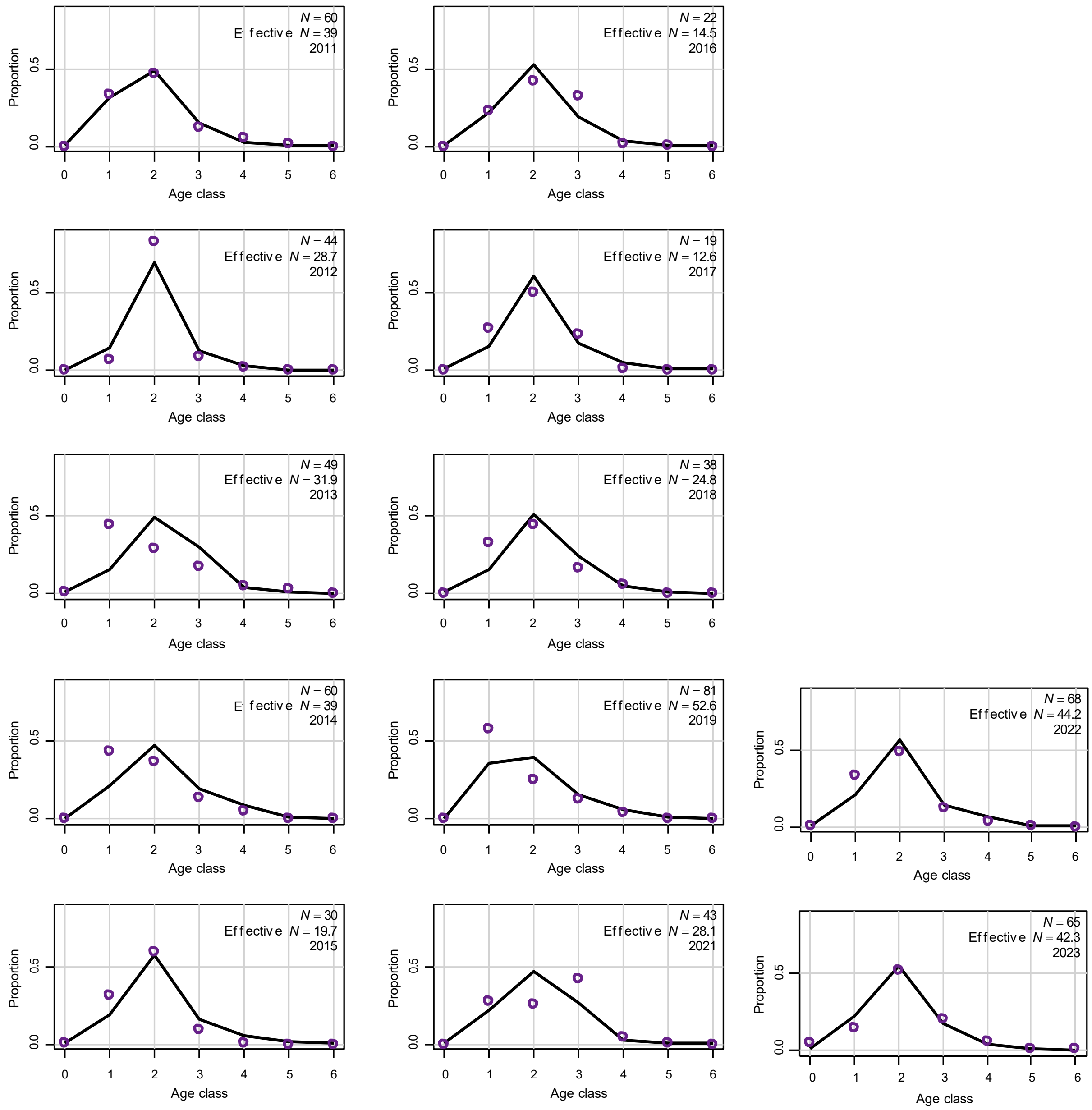

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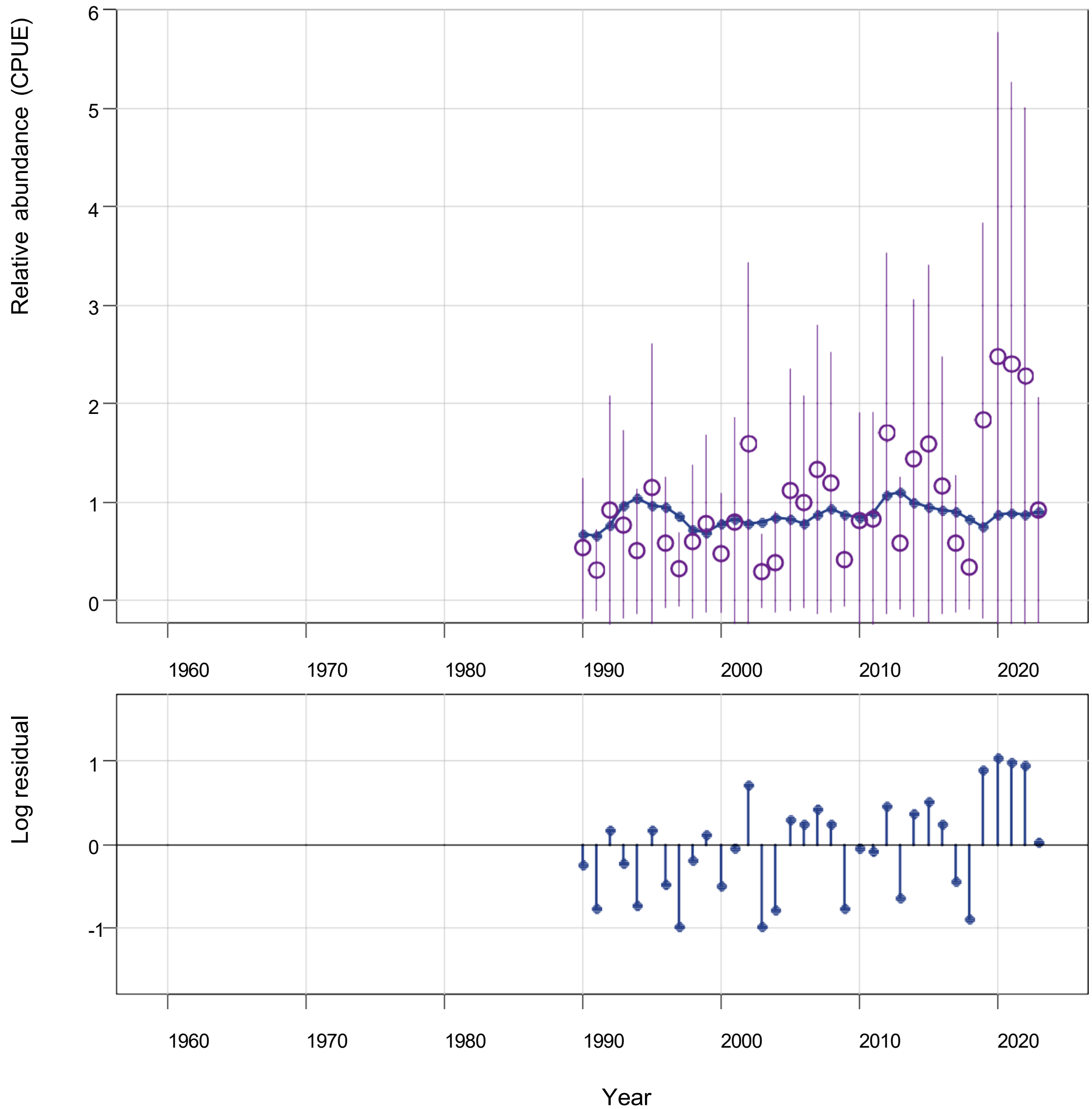


Figure A17. Predicted fit (blue, closed circle with line) to the observed (open circle) NAD index. The lower panel indicates the residual for each data point.

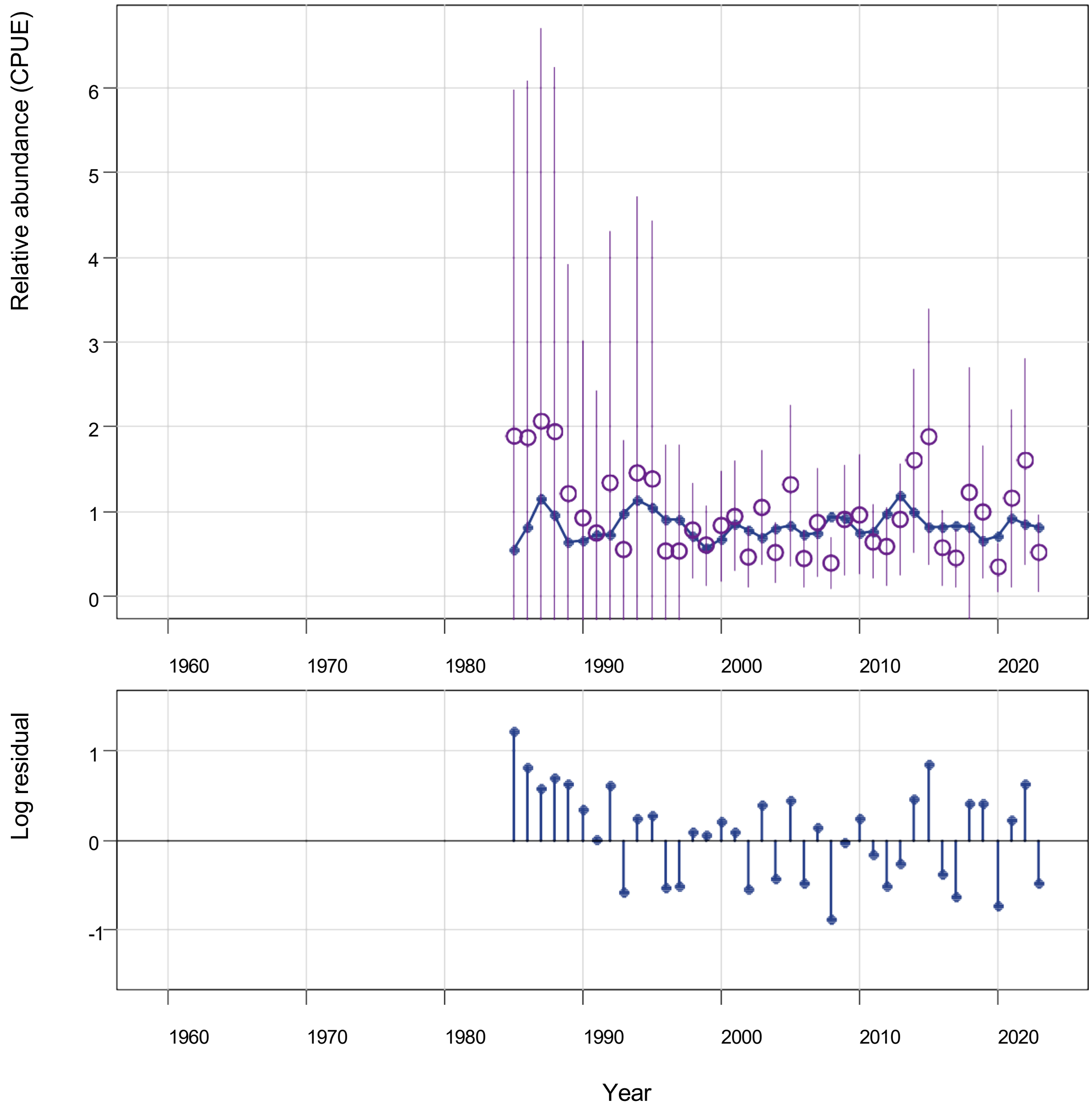


Figure A18. **Predicted fit (blue, closed circle with line) to the observed (open circle) MAD index. The lower panel indicates the residual for each data point.**

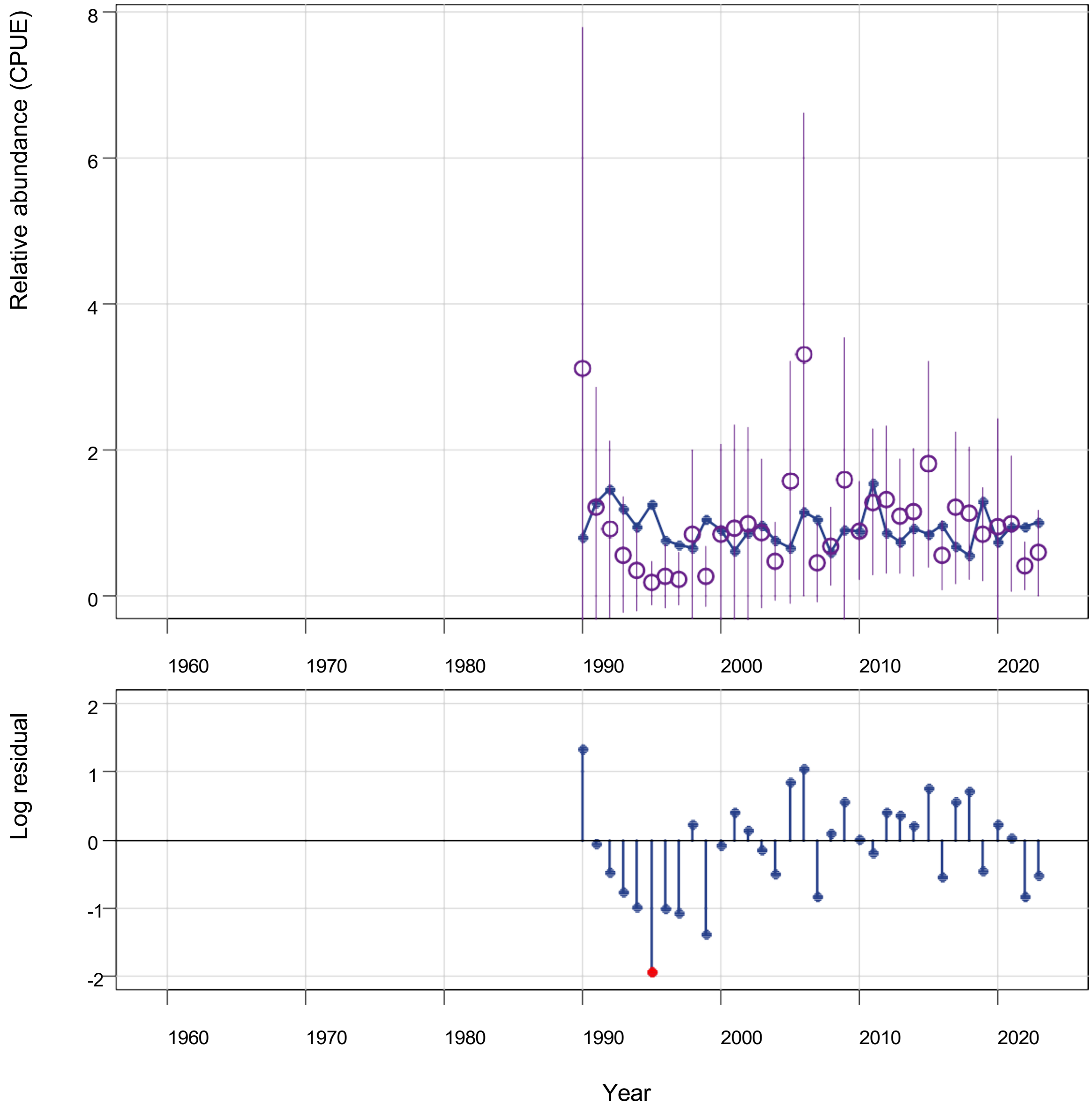


Figure A19. **Predicted fit (blue, closed circle with line) to the observed (open circle) SAD index. The lower panel indicates the residual for each data point.**

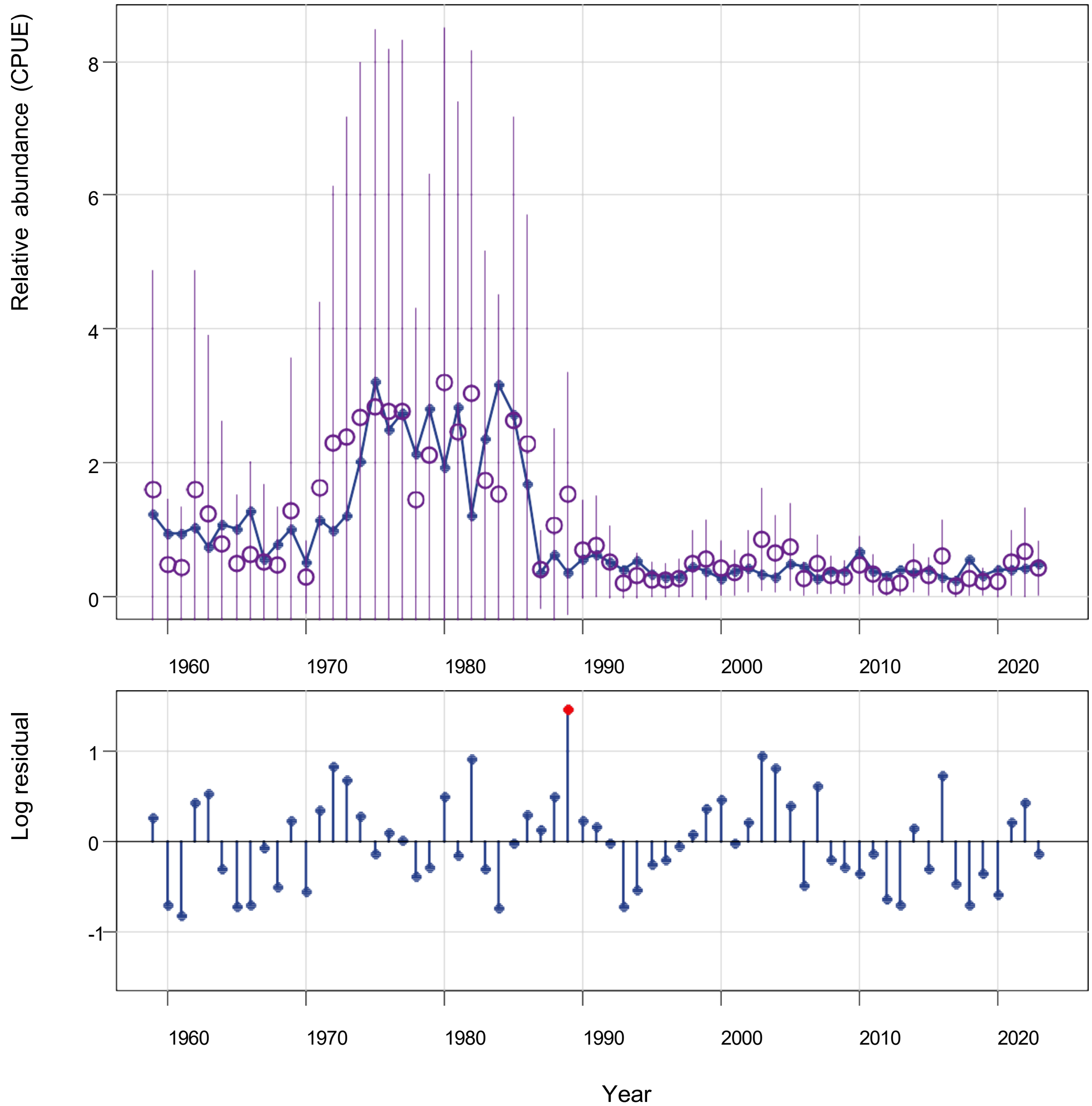


Figure A20. **Predicted fit (blue, closed circle with line) to the observed (open circle) recruitment index. The lower panel indicates the residual for each data point.**

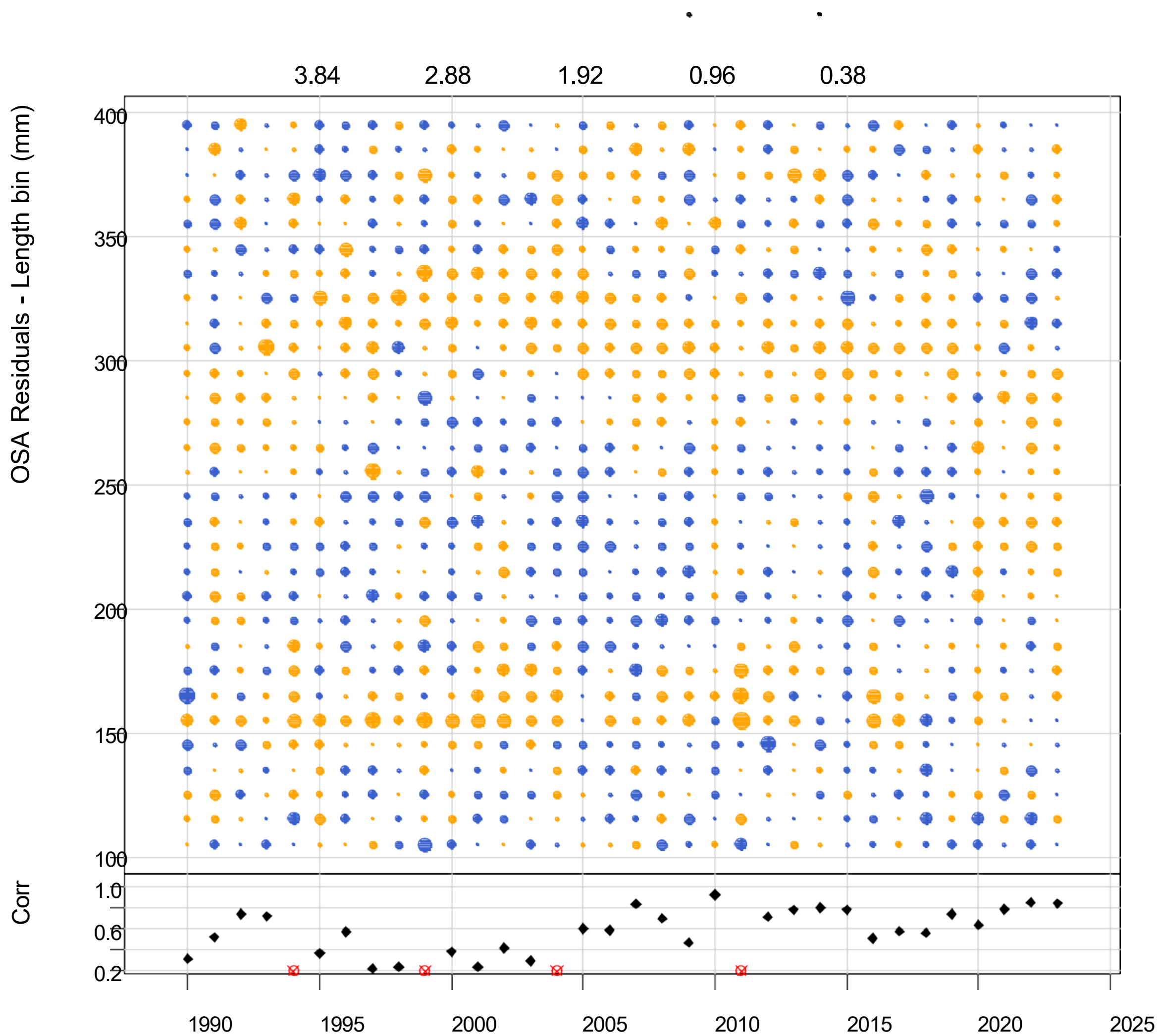


Figure A21. **Bubble plot of the fits to the length compositions for the NAD index. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.**

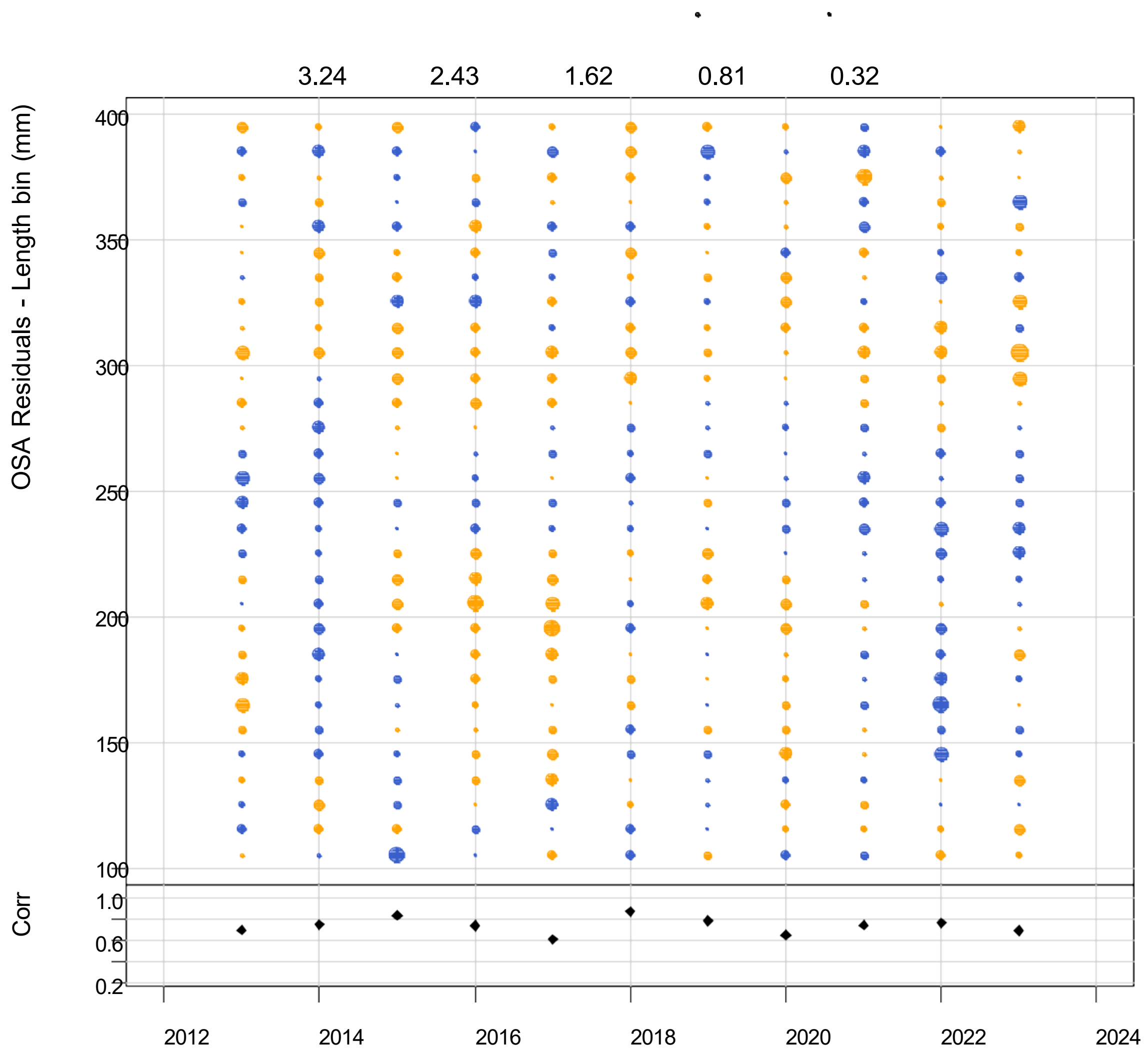


Figure A22. **Bubble plot of the fits to the length compositions for the MAD index. Orange indicates an underestimate, while blue indicates an overestimate. OSA is one step ahead residuals. The bottom panel indicates the correlation between the observed data and the model prediction.**

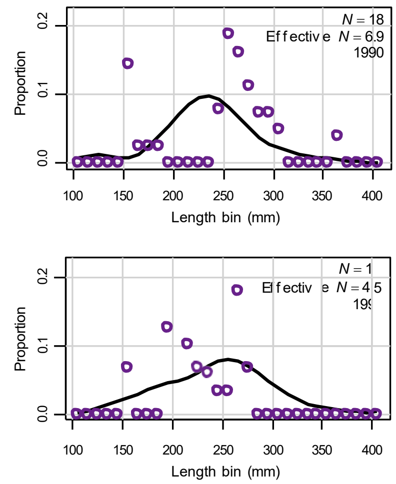
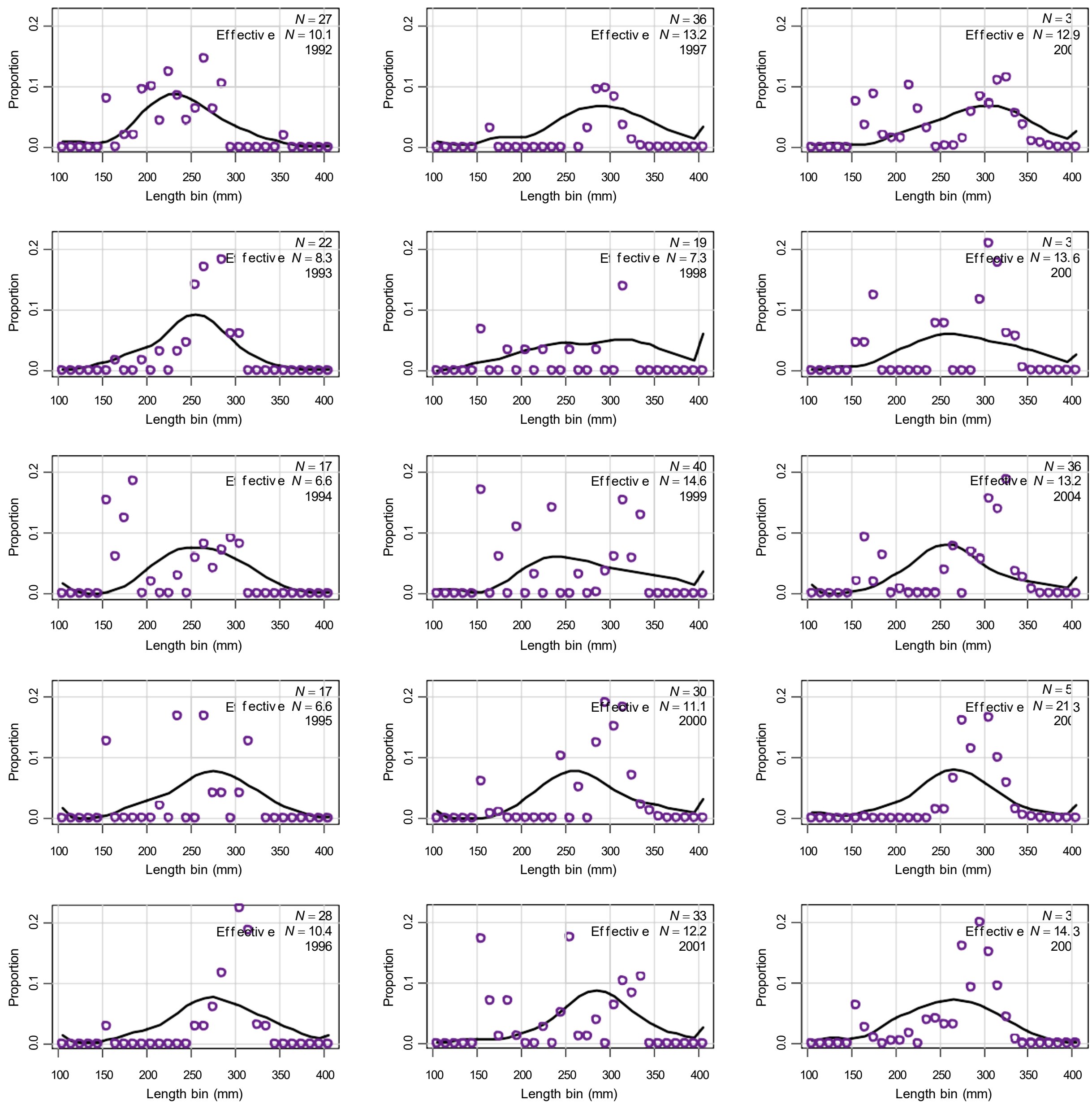


Figure A23. **Annual length composition plots for the NAD index for 1990-2023. Open circles are the observed data, while the line indicates the model fit.**



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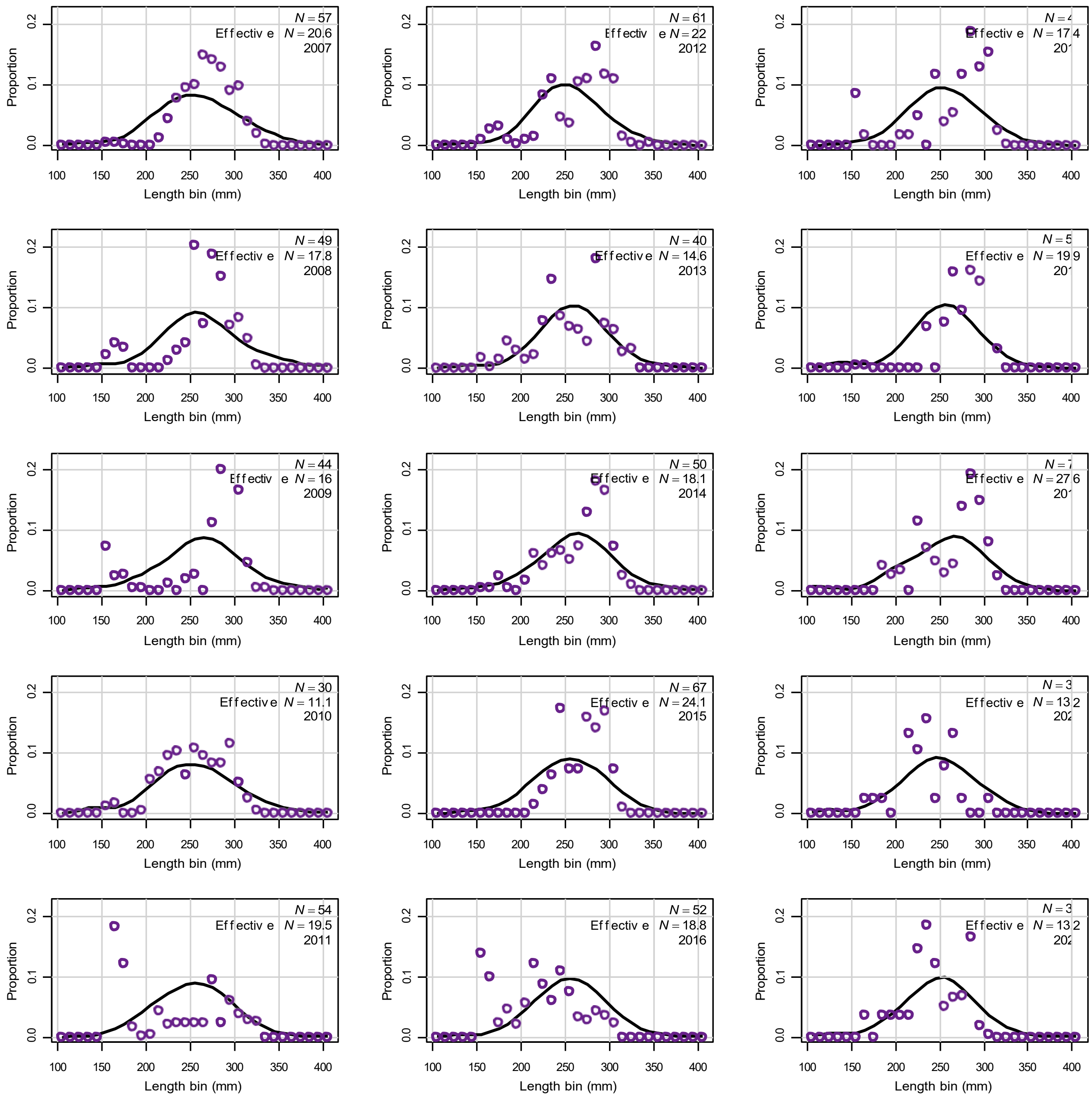

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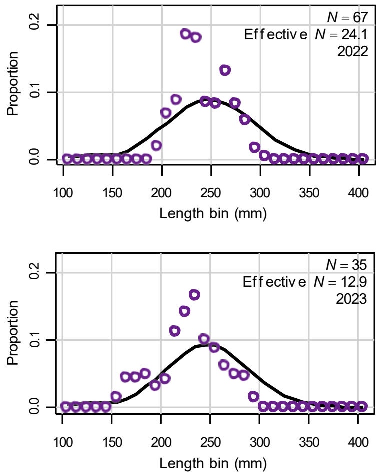

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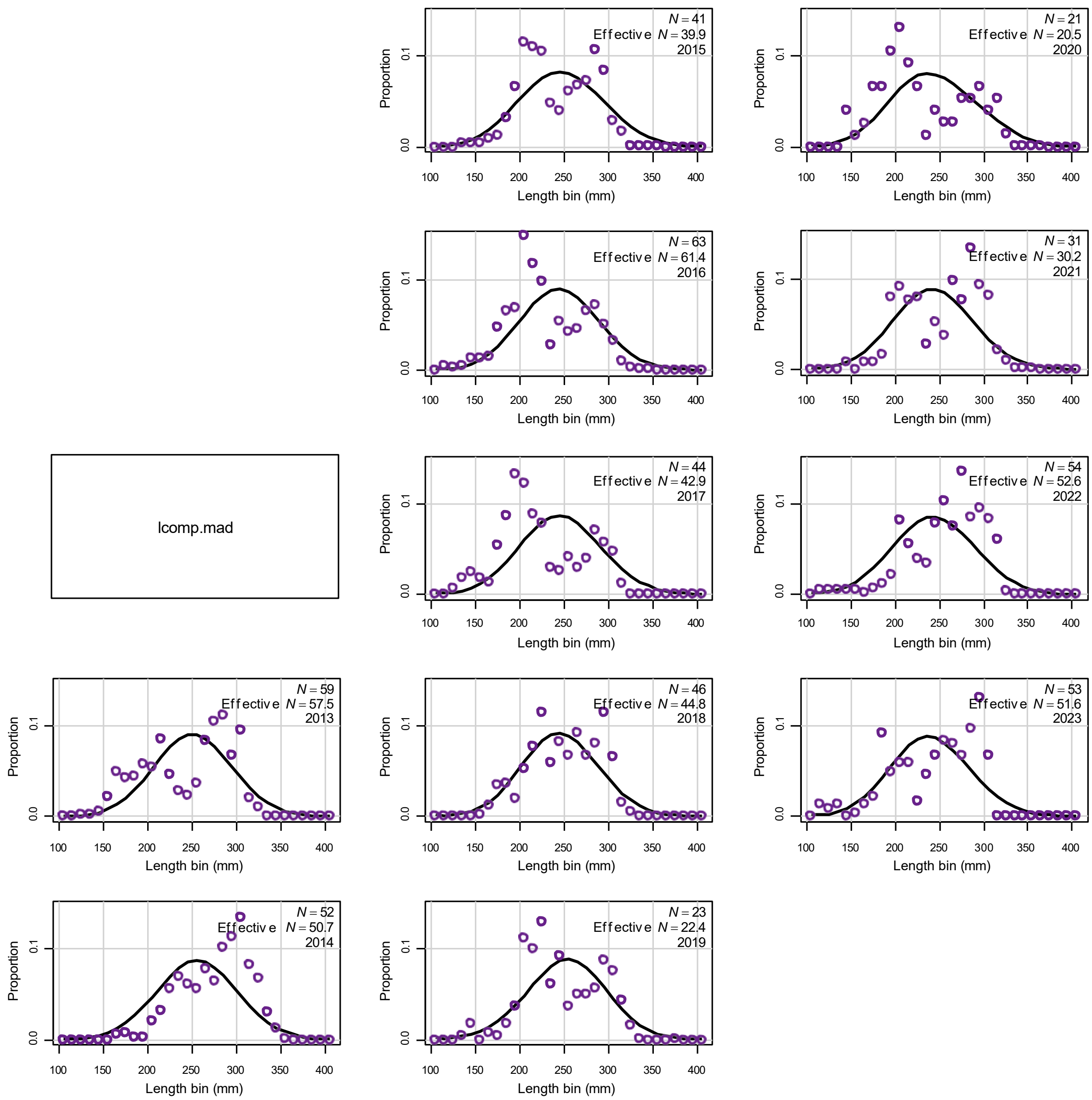

Figure A24. **Annual length composition plots for the MAD index for 2013-2023. Open circles are the observed data, while the line indicates the model fit.**

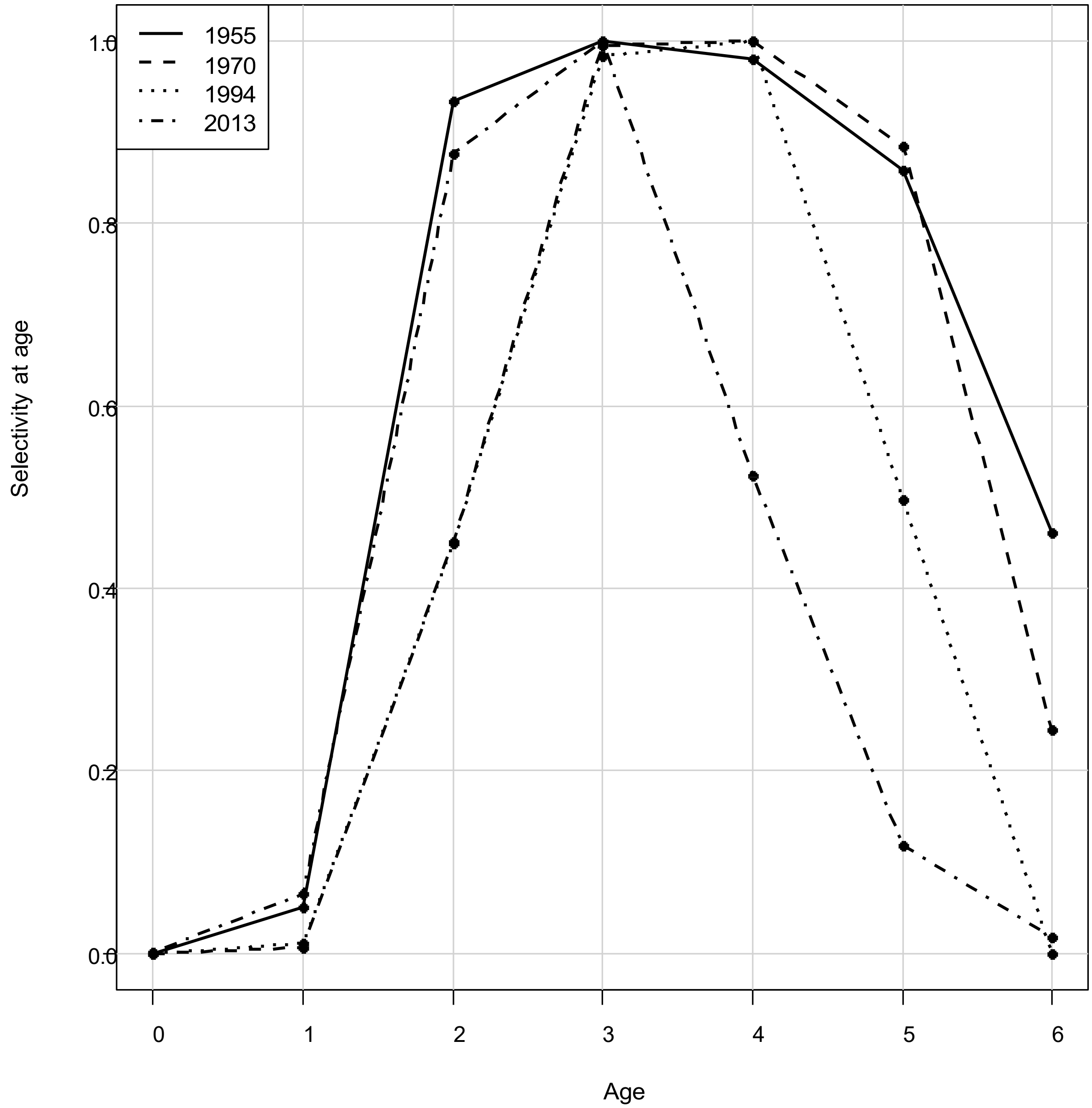

Figure A25. **Estimated selectivity of the northern commercial reduction landings for 1955-1969, 1970-1993, 1994-2012, and 2013-2023.**

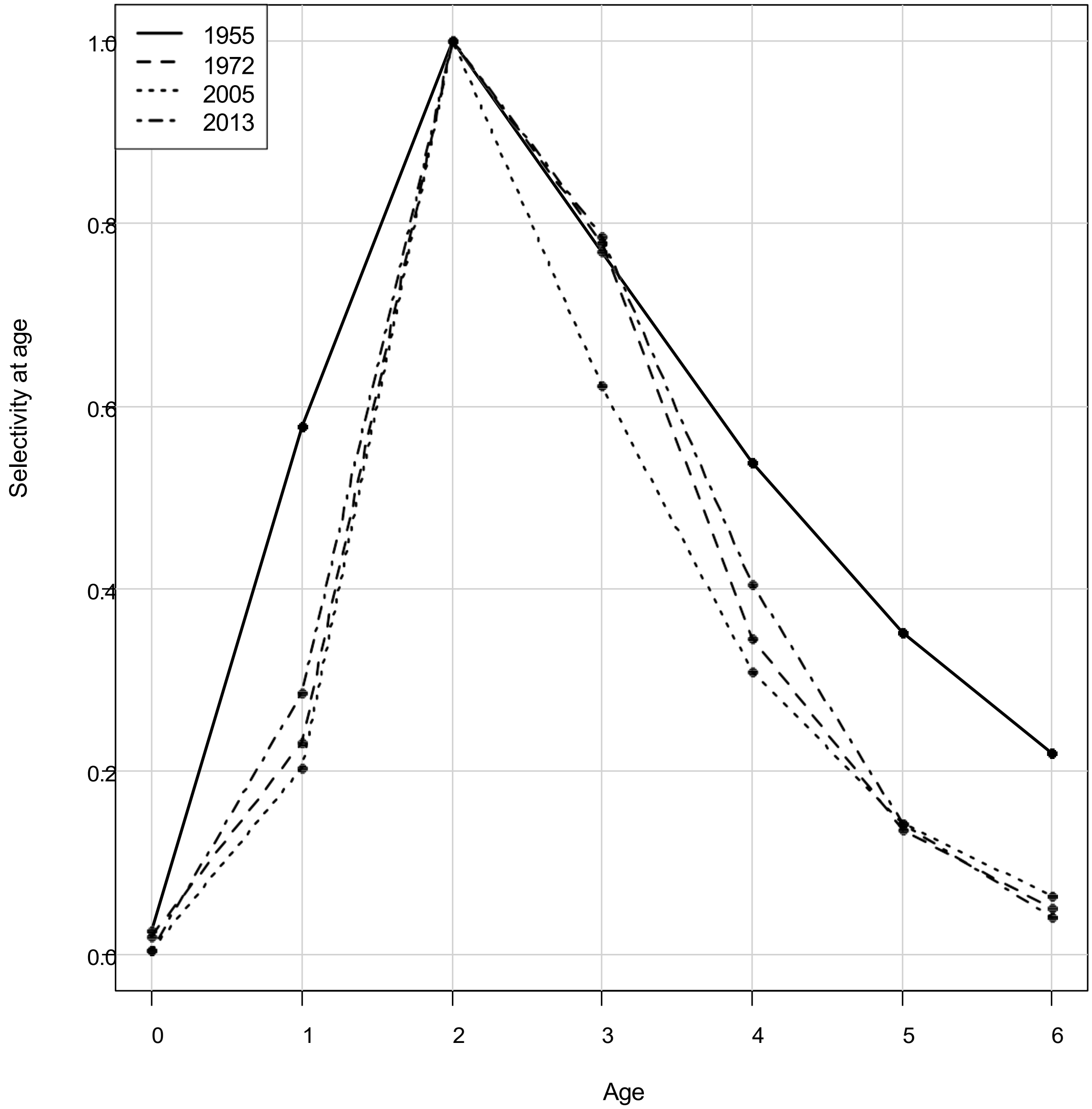

Figure A26. **Estimated selectivity of the southern commercial reduction landings for 1955-1971, 1972-2004, 2005-2012, and 2013-2023.**

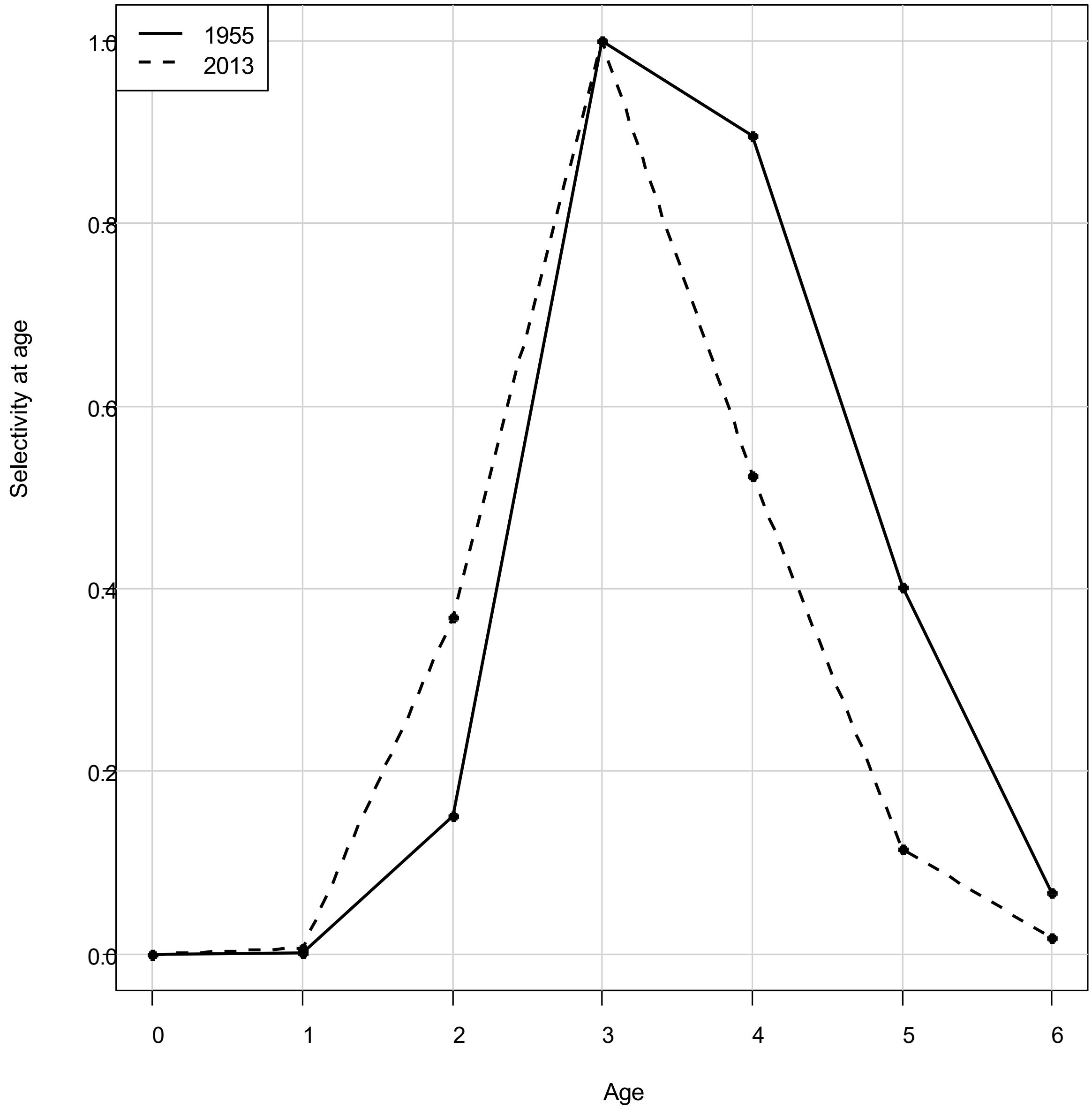

Figure A27. Estimated selectivity of the northern commercial bait landings for 1955-2012 and 2013-2023.

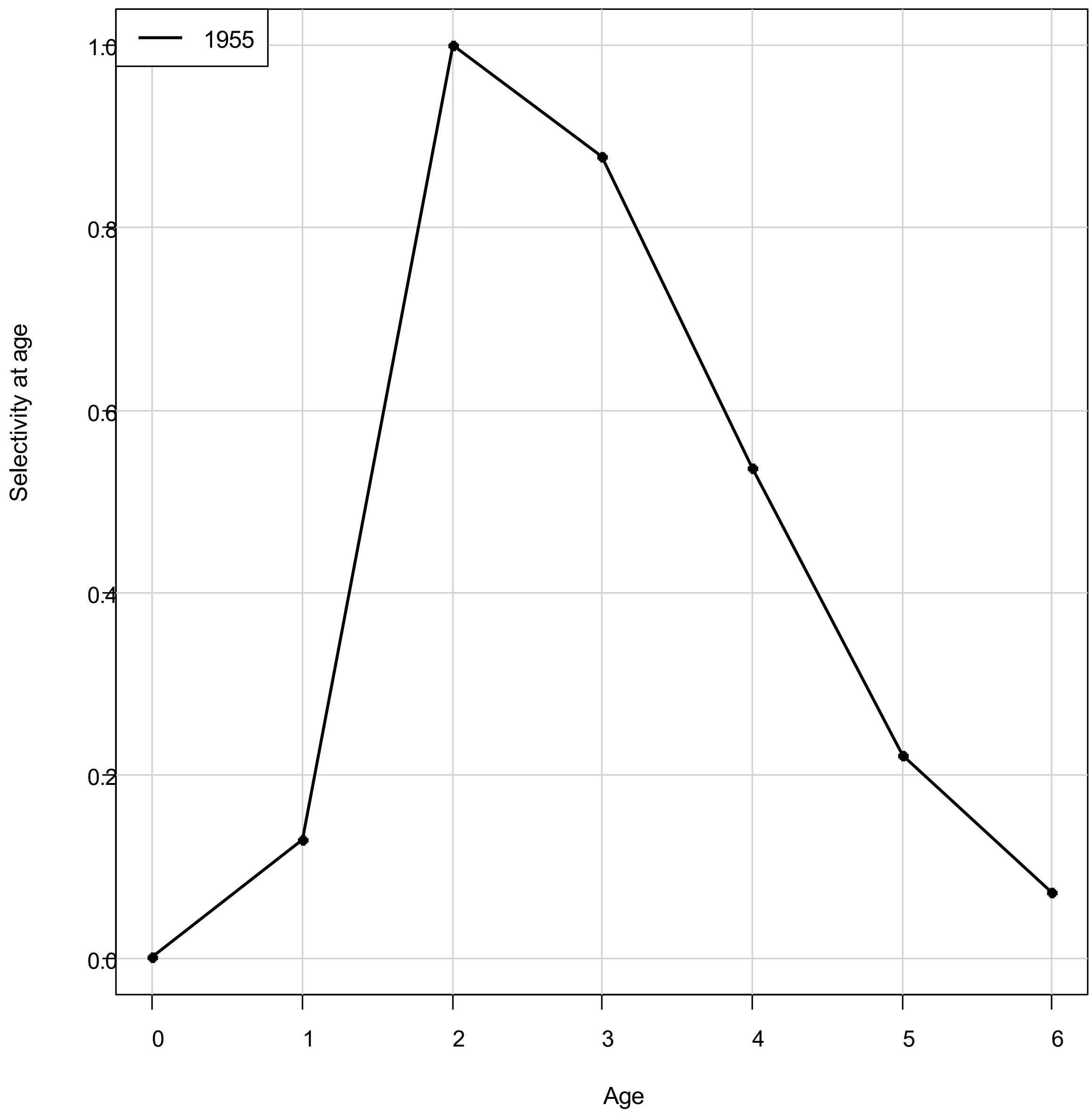

Figure A28. Estimated selectivity of the southern commercial bait landings for 1955-2023.

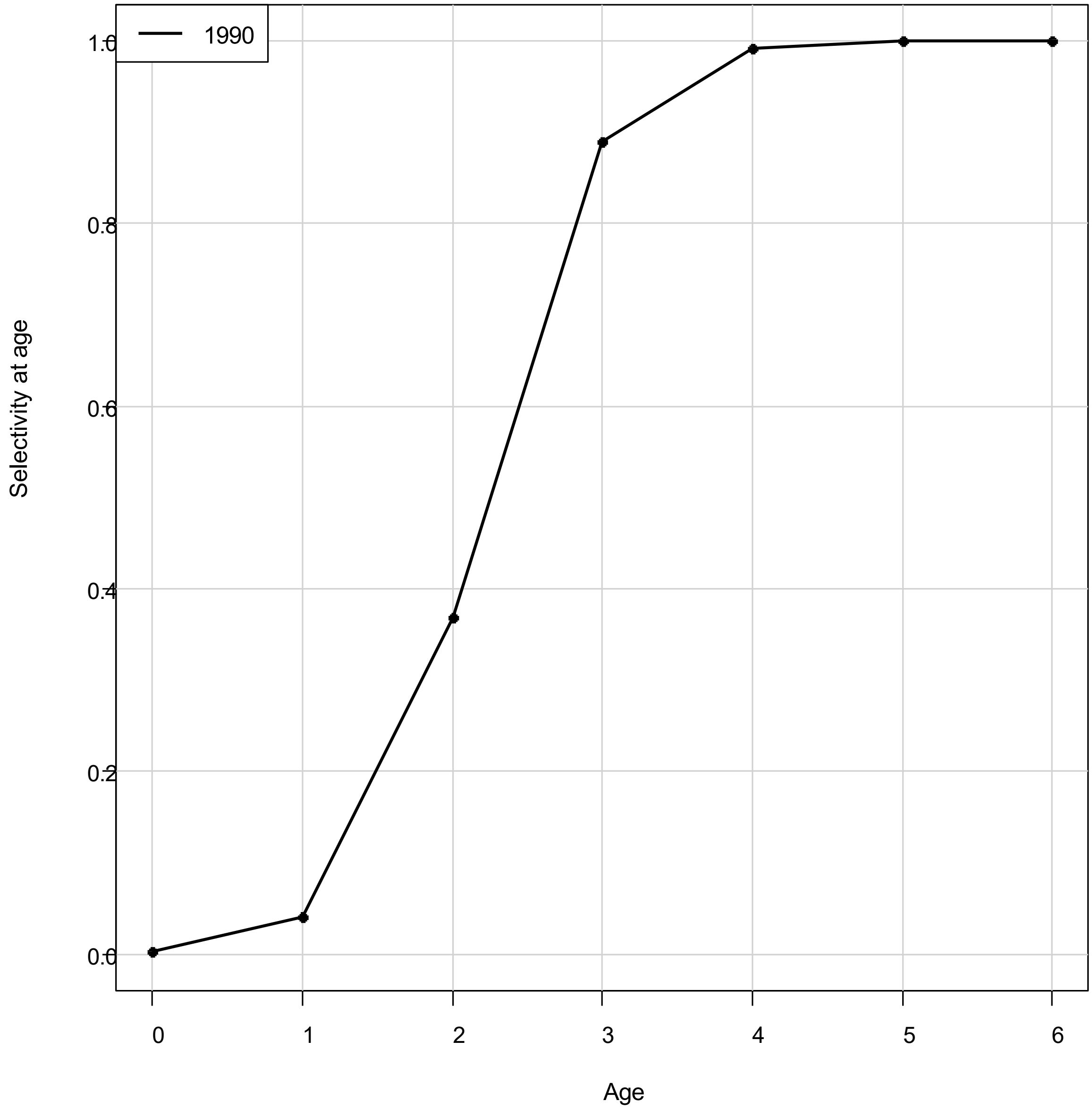

Figure A29. **Estimated selectivity for the NAD index for 1990-2023.**

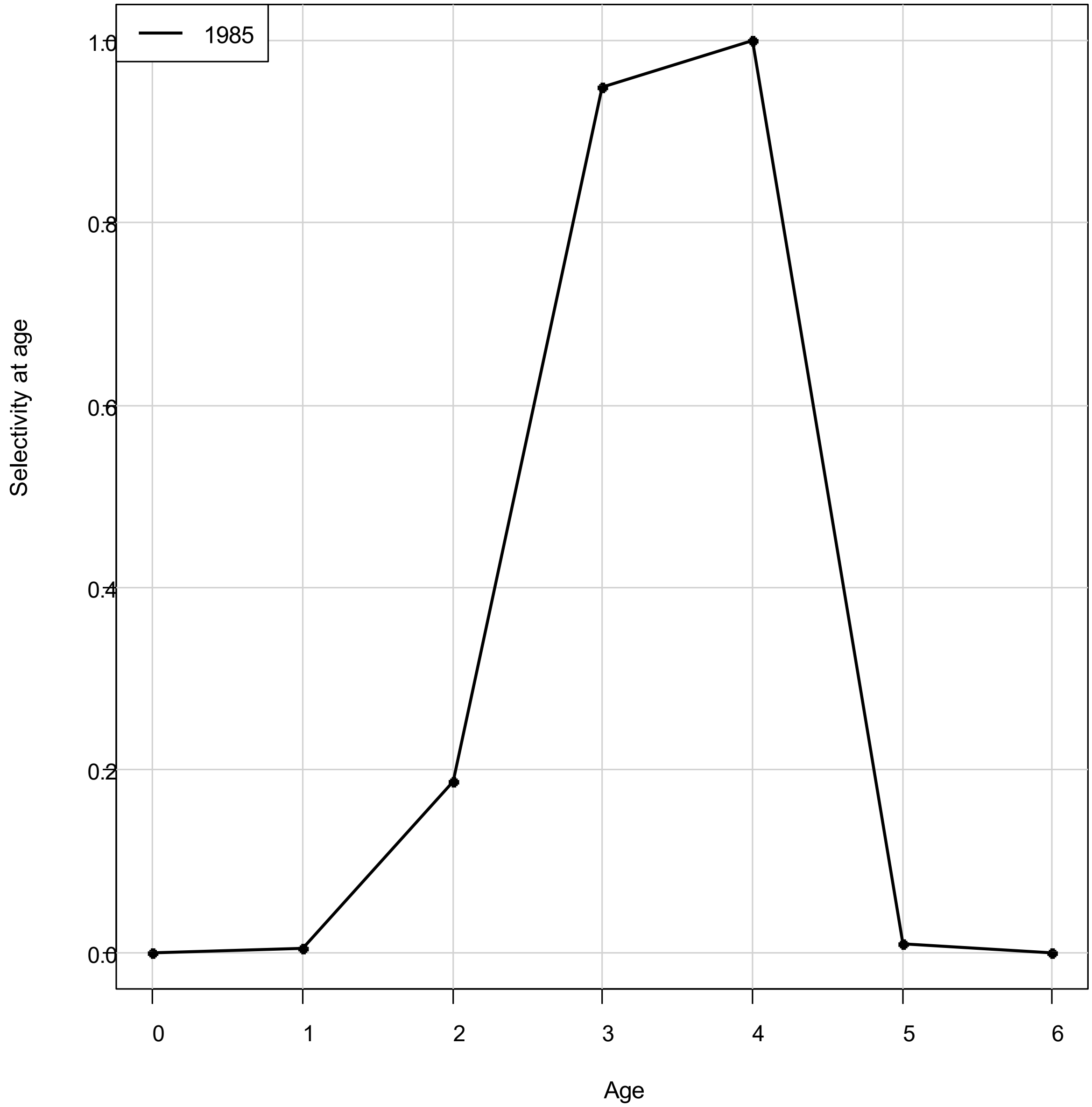

Figure A30. **Estimated selectivity for the MAD index for 1985-2023.**

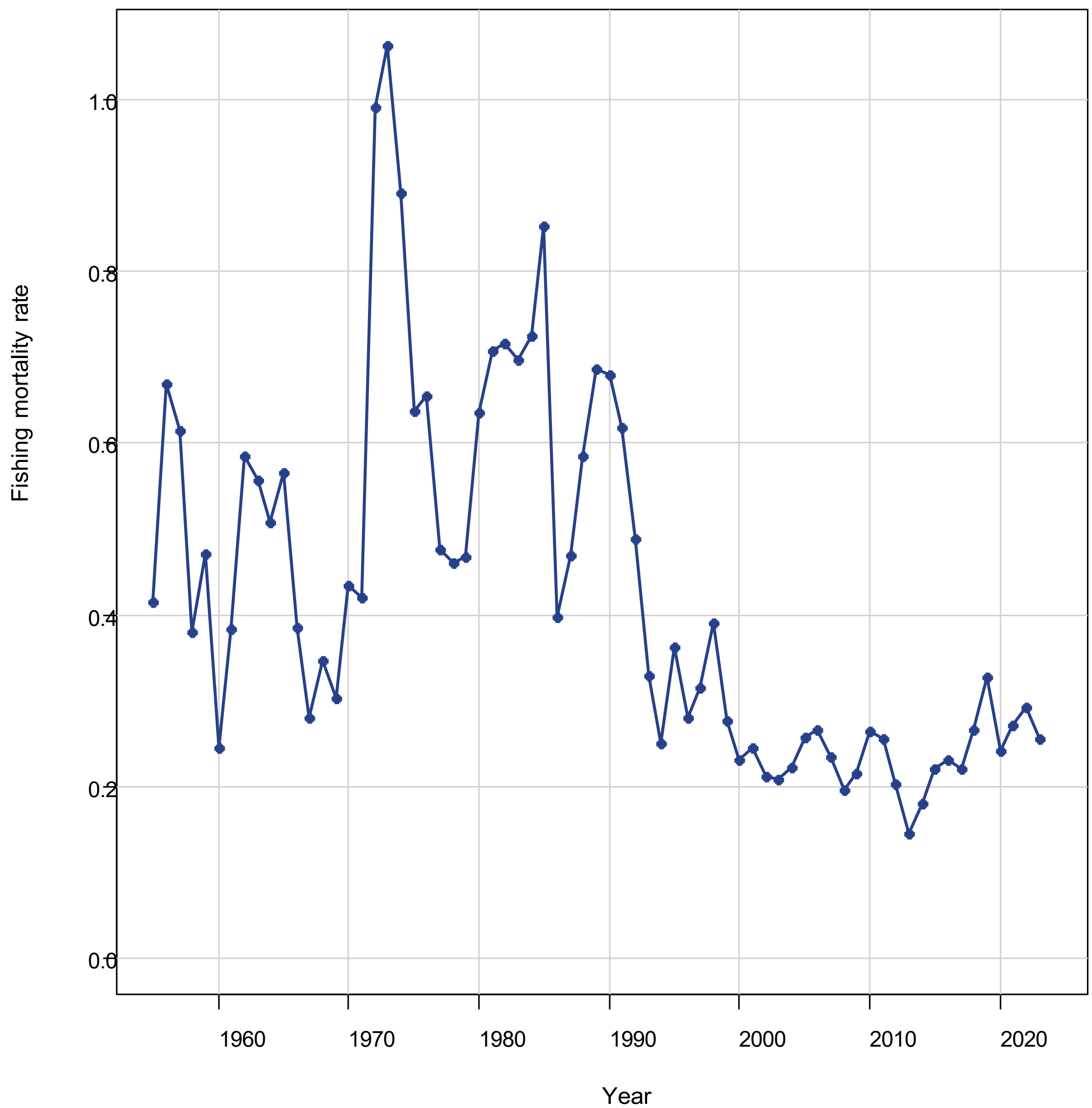

Figure A31. The full fishing mortality rate for 1955-2023.

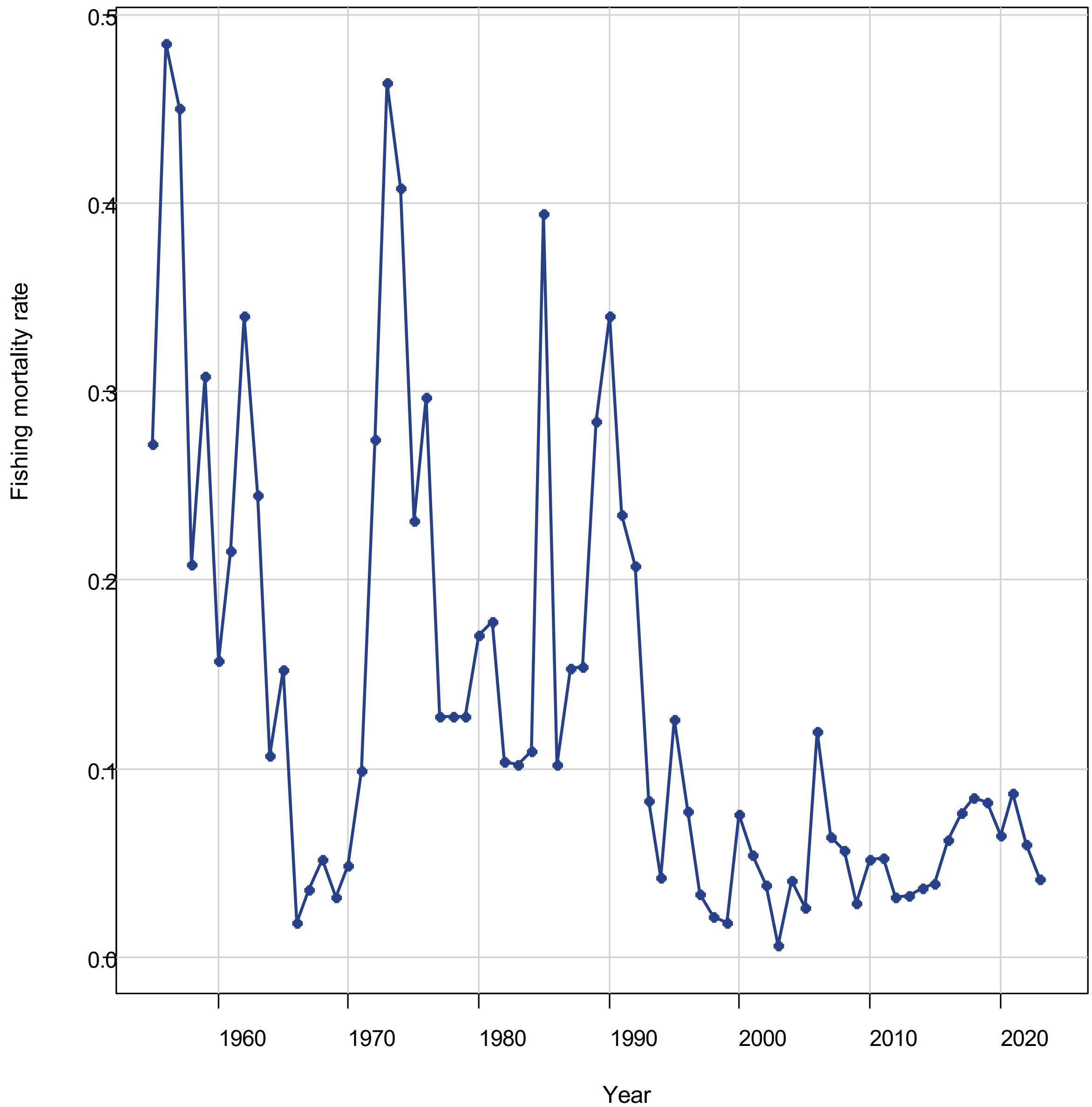

Figure A32. **The full fishing mortality rate for the commercial reduction north fleet for 1955-2023.**

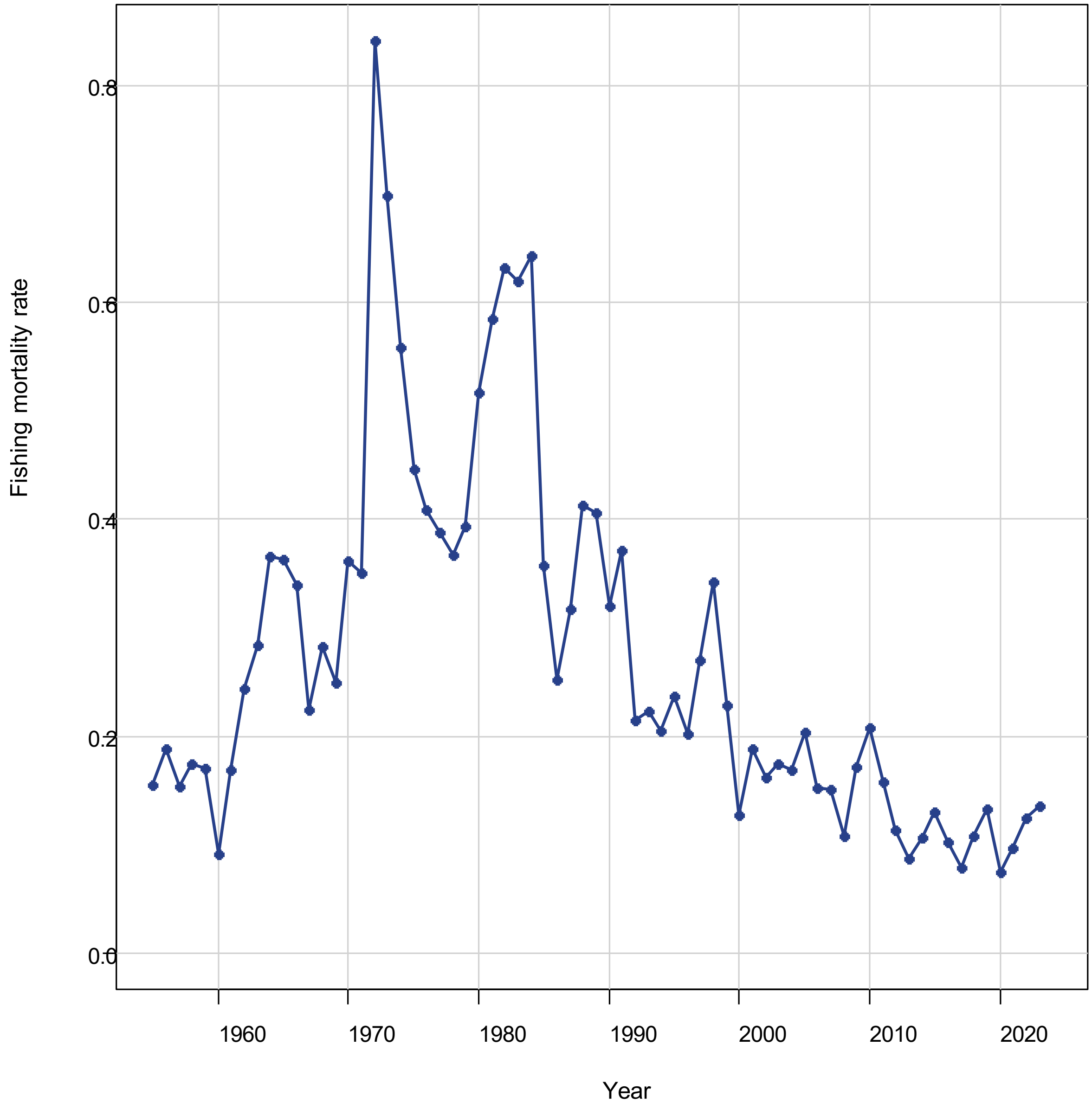

Figure A33. The full fishing mortality rate for the commercial reduction south fleet for 1955-2023.

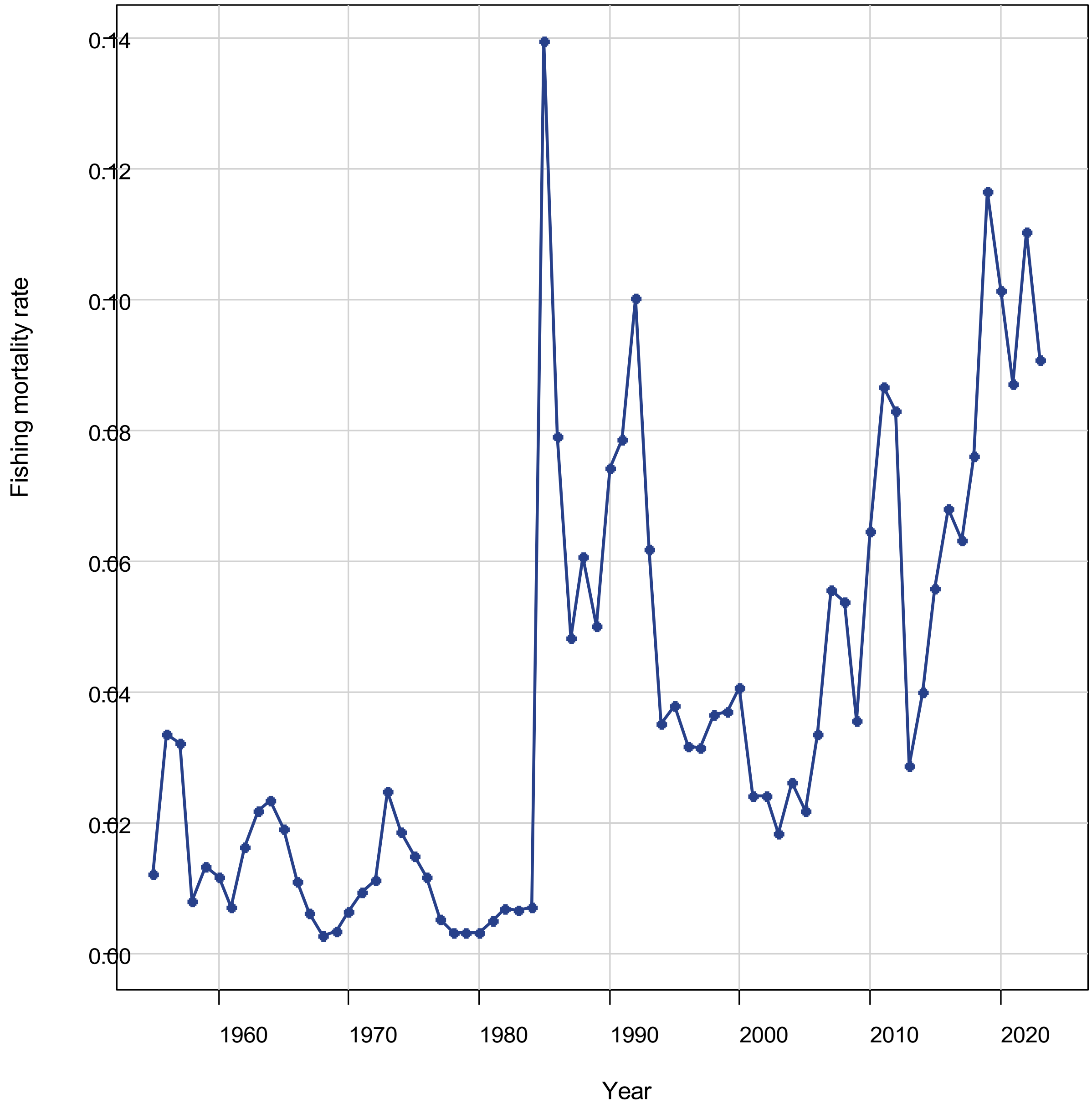

Figure A34. The full fishing mortality rate for the commercial bait north fleet for 1955-2023.

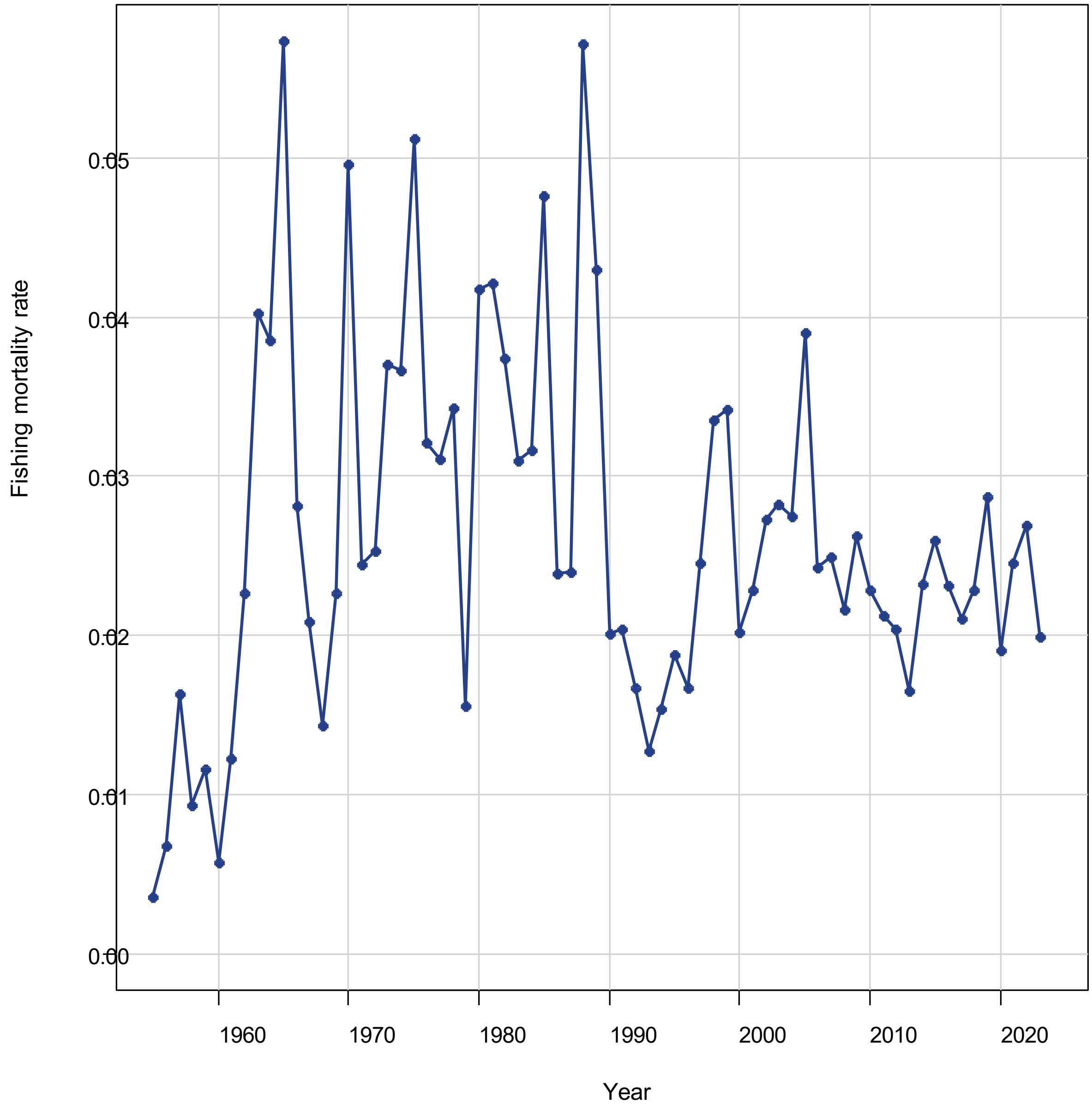

Figure A35. The full fishing mortality rate for the commercial bait south fleet for 1955-2023.

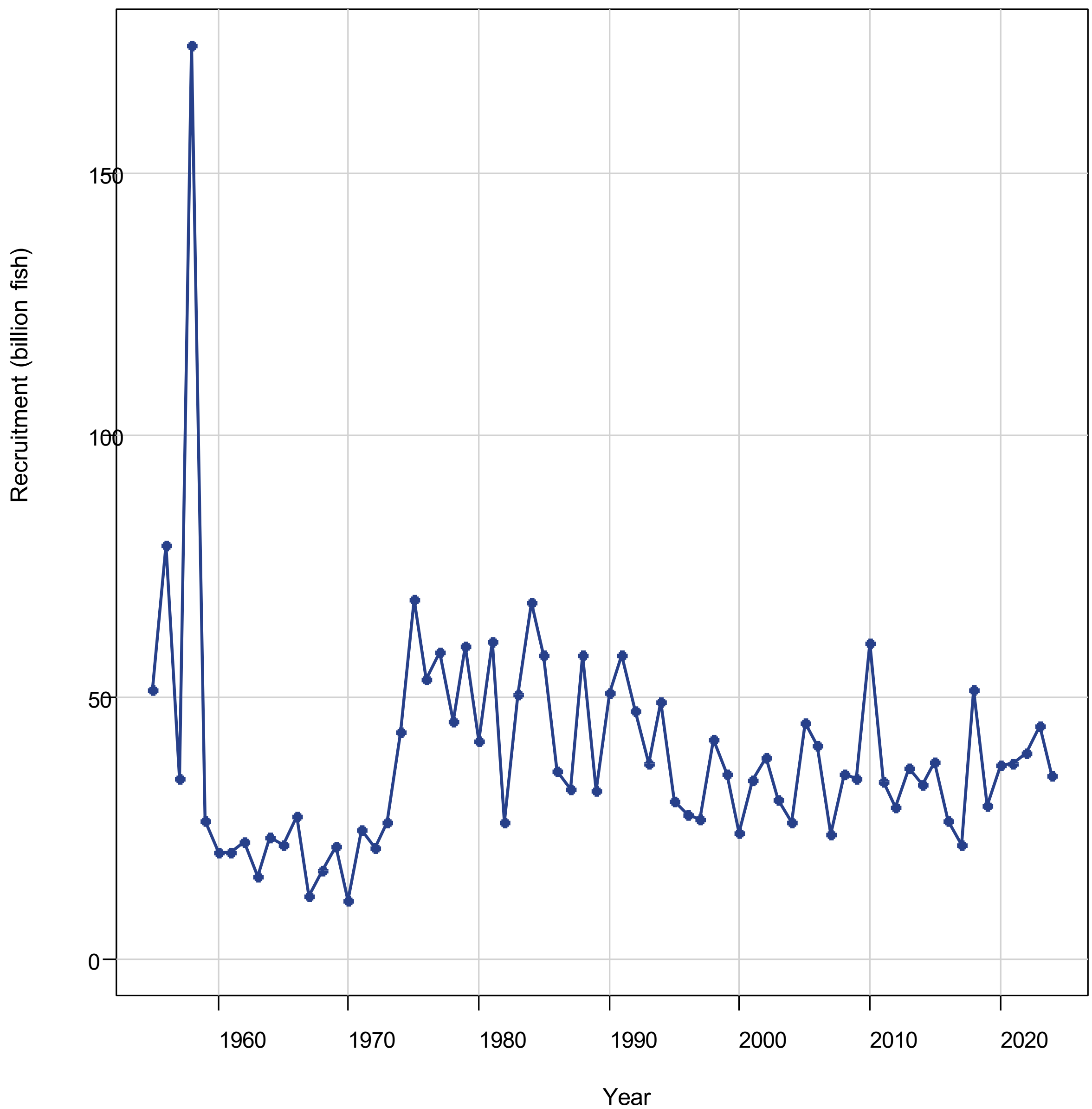

Figure A36. The estimated time series of recruitment for 1955-2023. The 2024 point is a projected recruitment point.

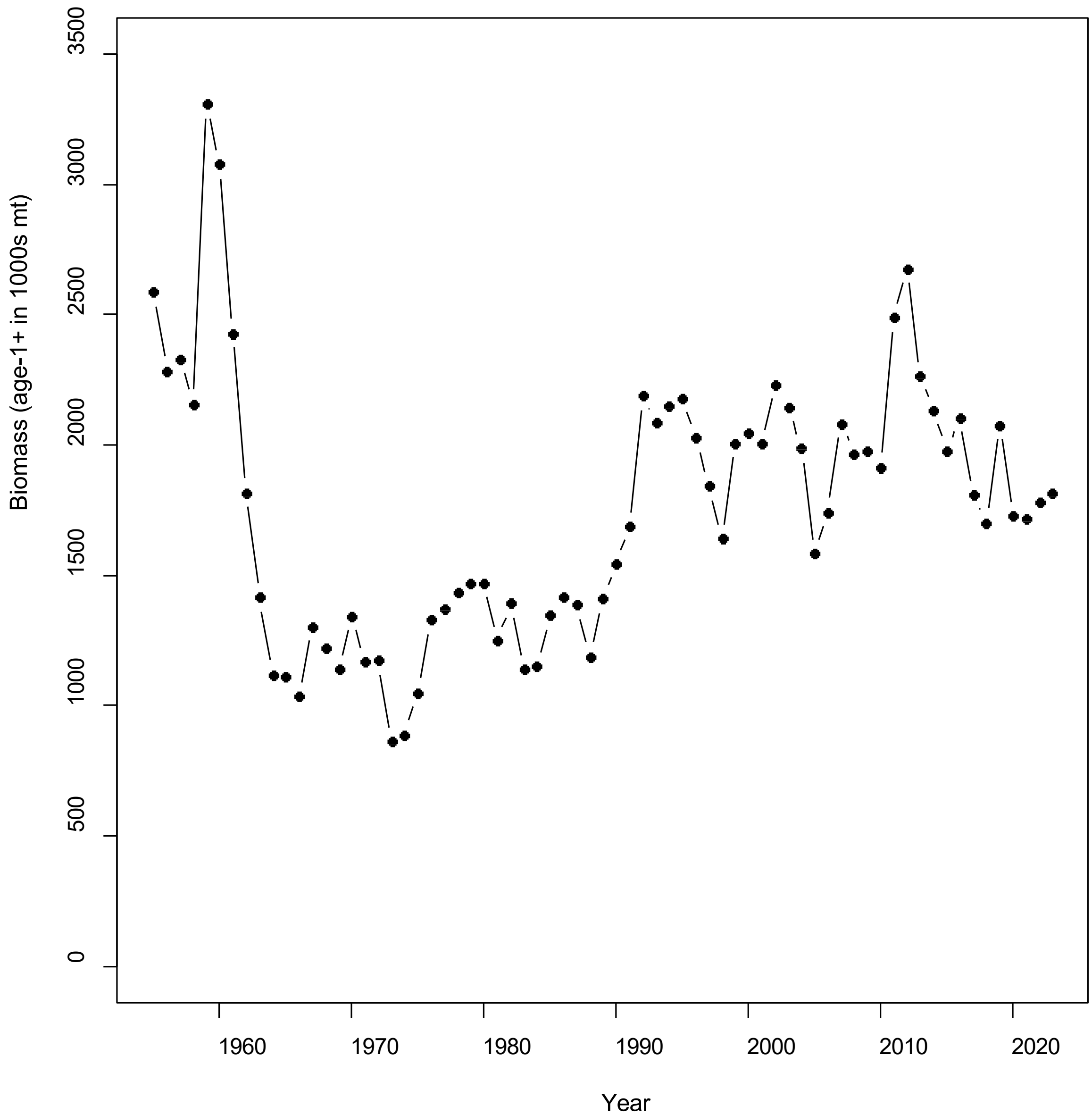

Figure A37. **Age-1+ biomass in 1000s of mt for 1955-2023.**

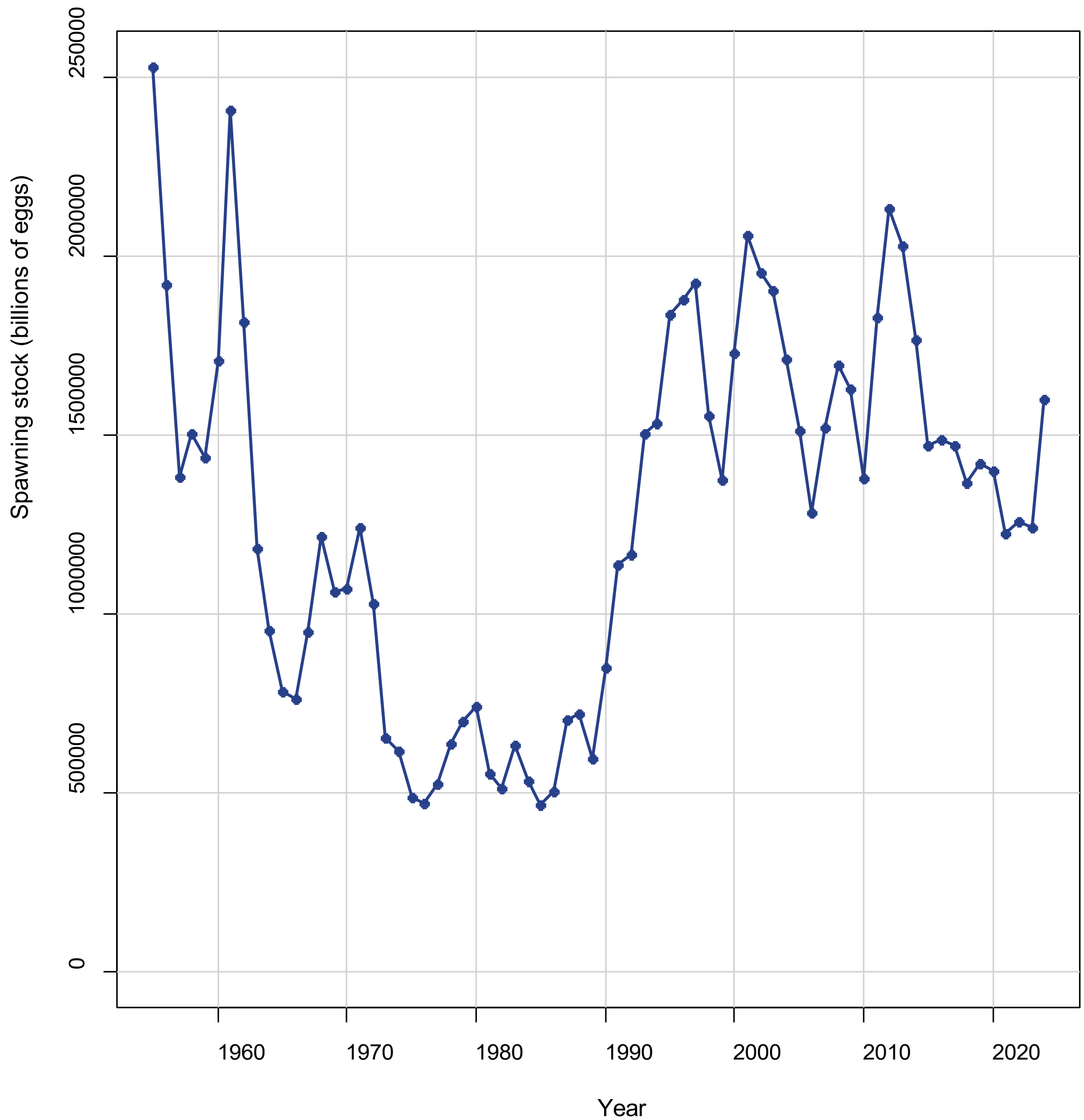

Figure A38. Fecundity in billions of ova for 1955-2023. The 2024 value is a projected value.

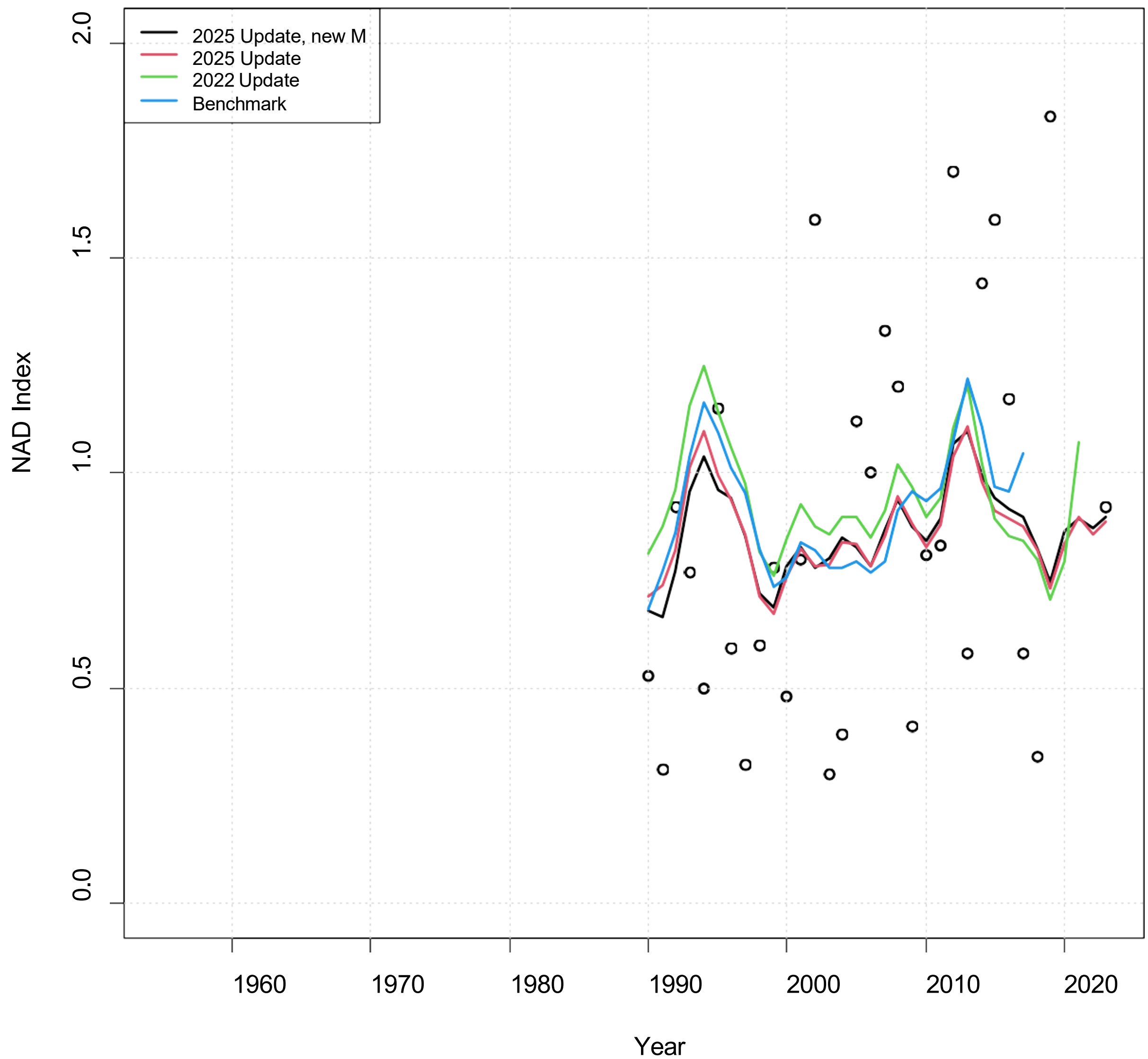

Figure A39. Fit to the observed (open circles) NAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.

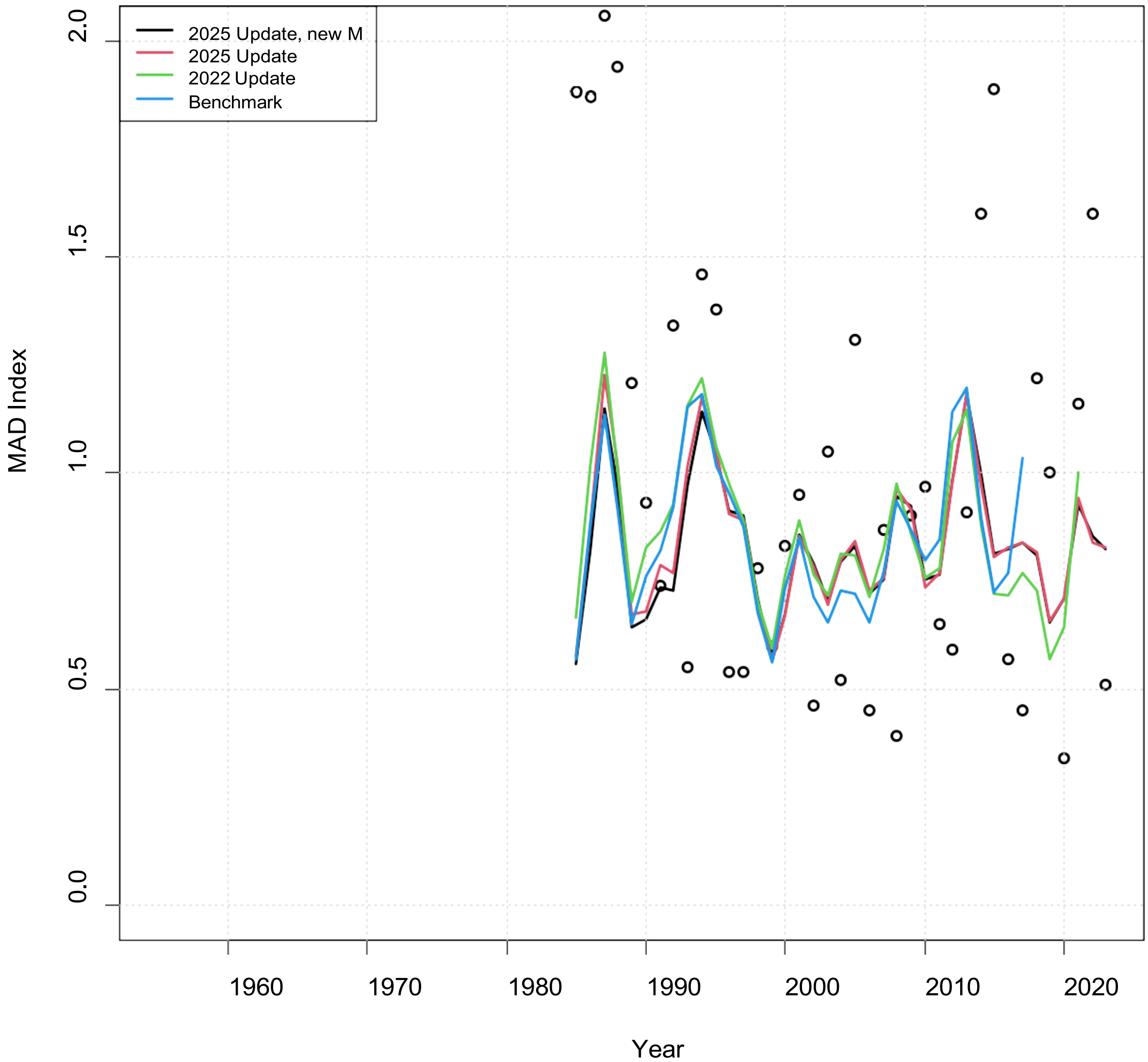

Figure A40. **Fit to the observed (open circles) MAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

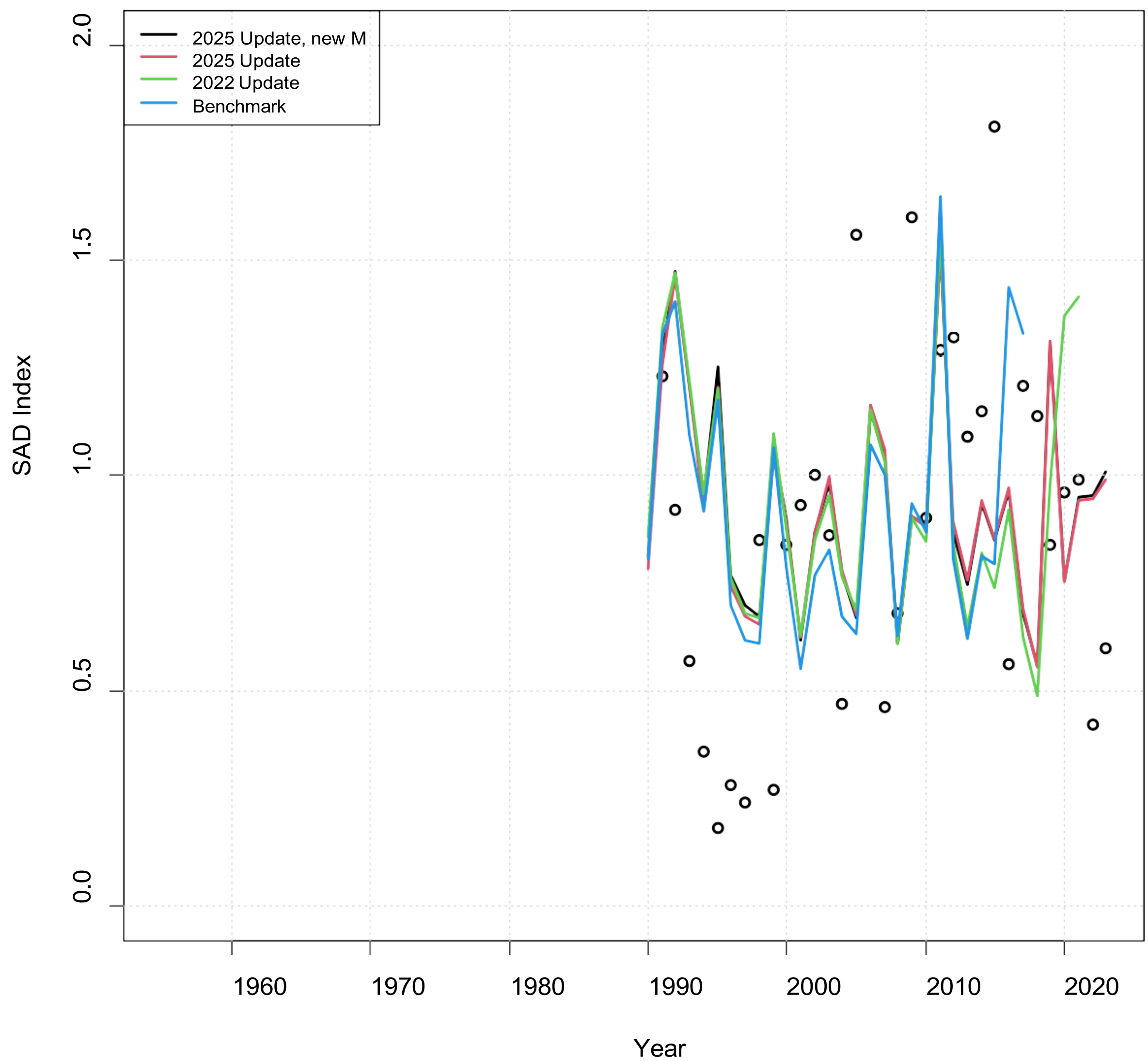

Figure A41. **Fit to the observed (open circles) SAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

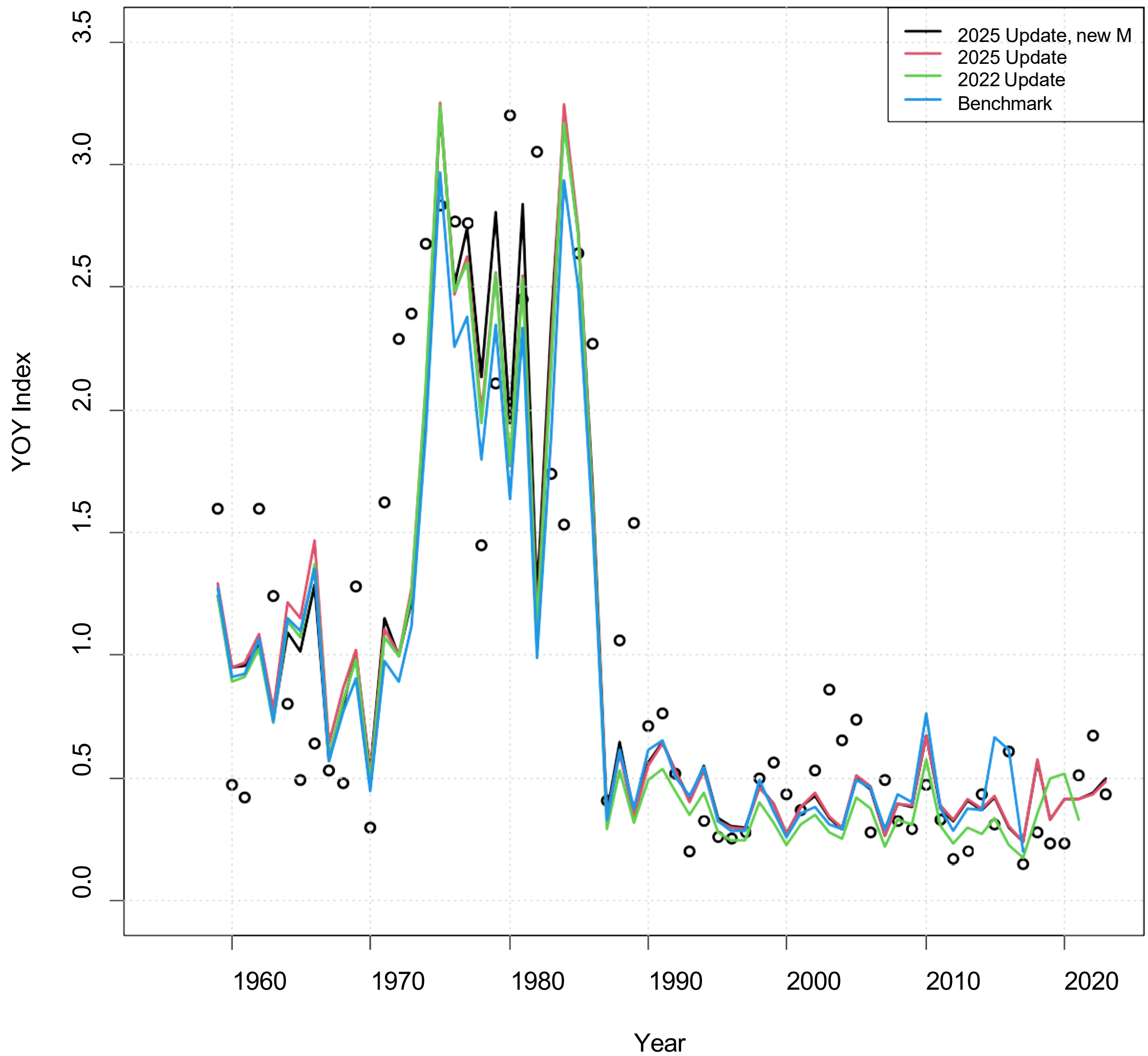

Figure A42. **Fit to the observed (open circles) recruitment index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

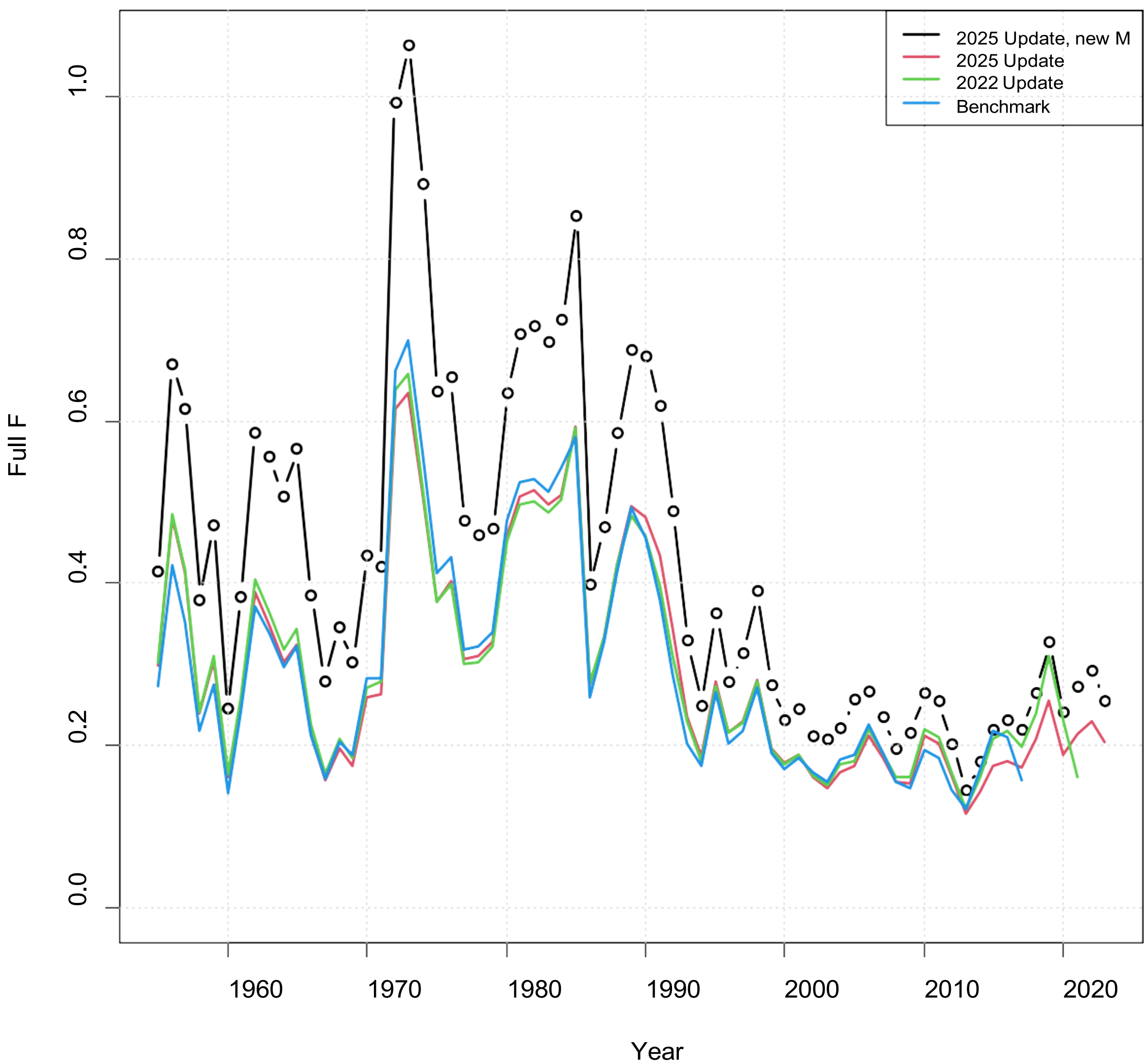

Figure A43. **Estimates of the full fishing mortality rate for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

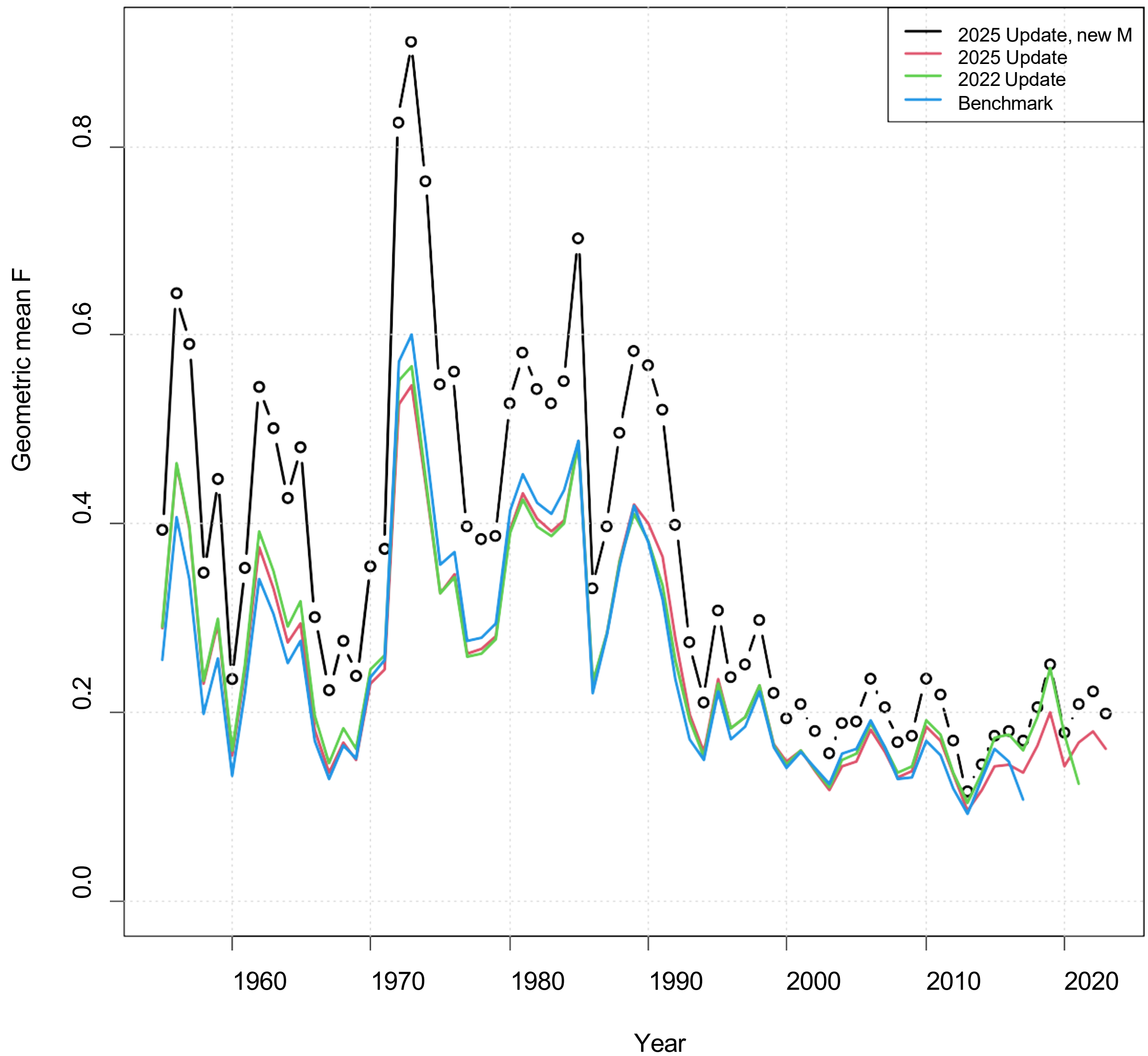

Figure A44. **Estimates of the geometric mean fishing mortality rate for ages-2 to -4 for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

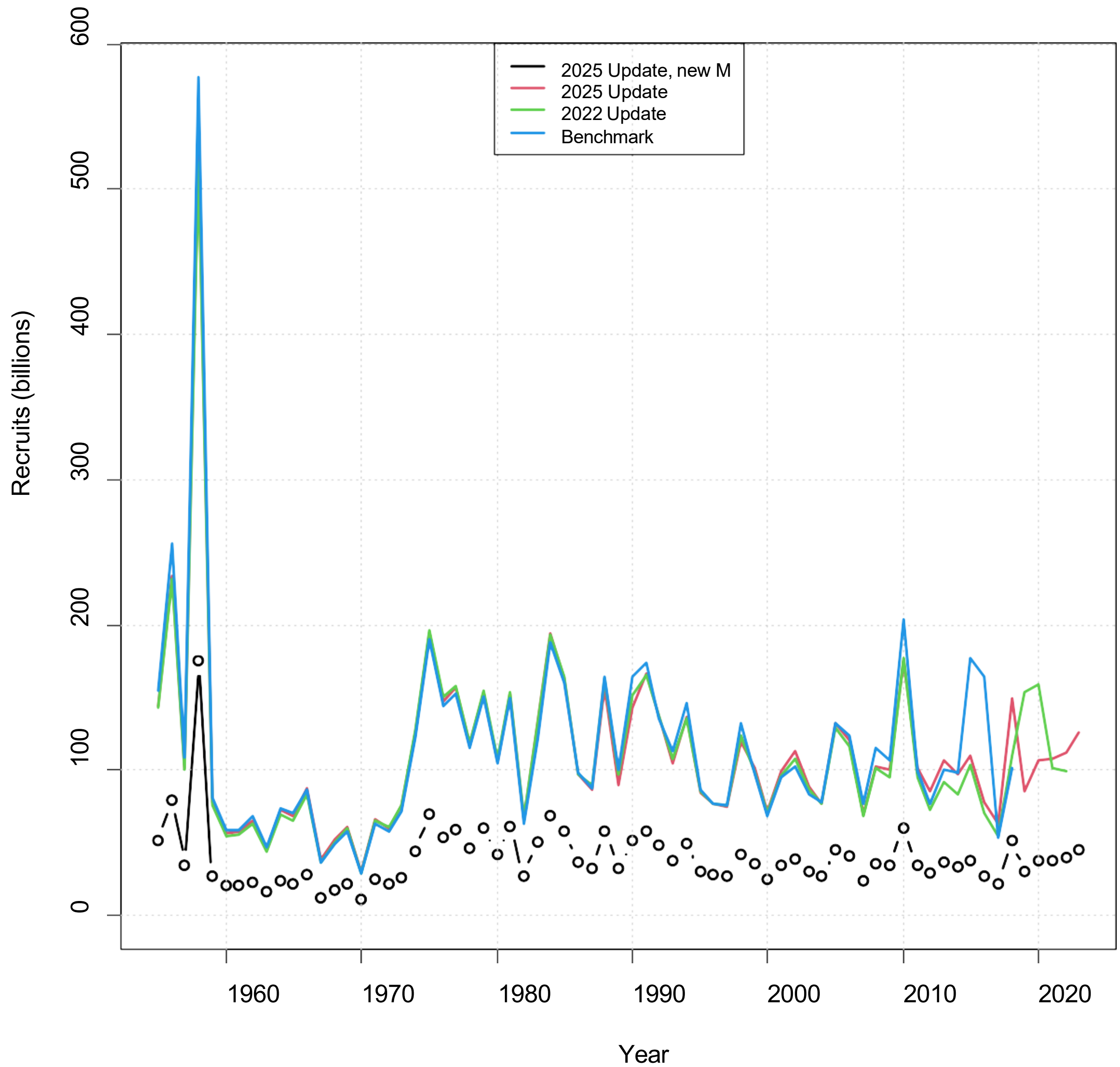

Figure A45. **Estimates of the recruitment time series for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

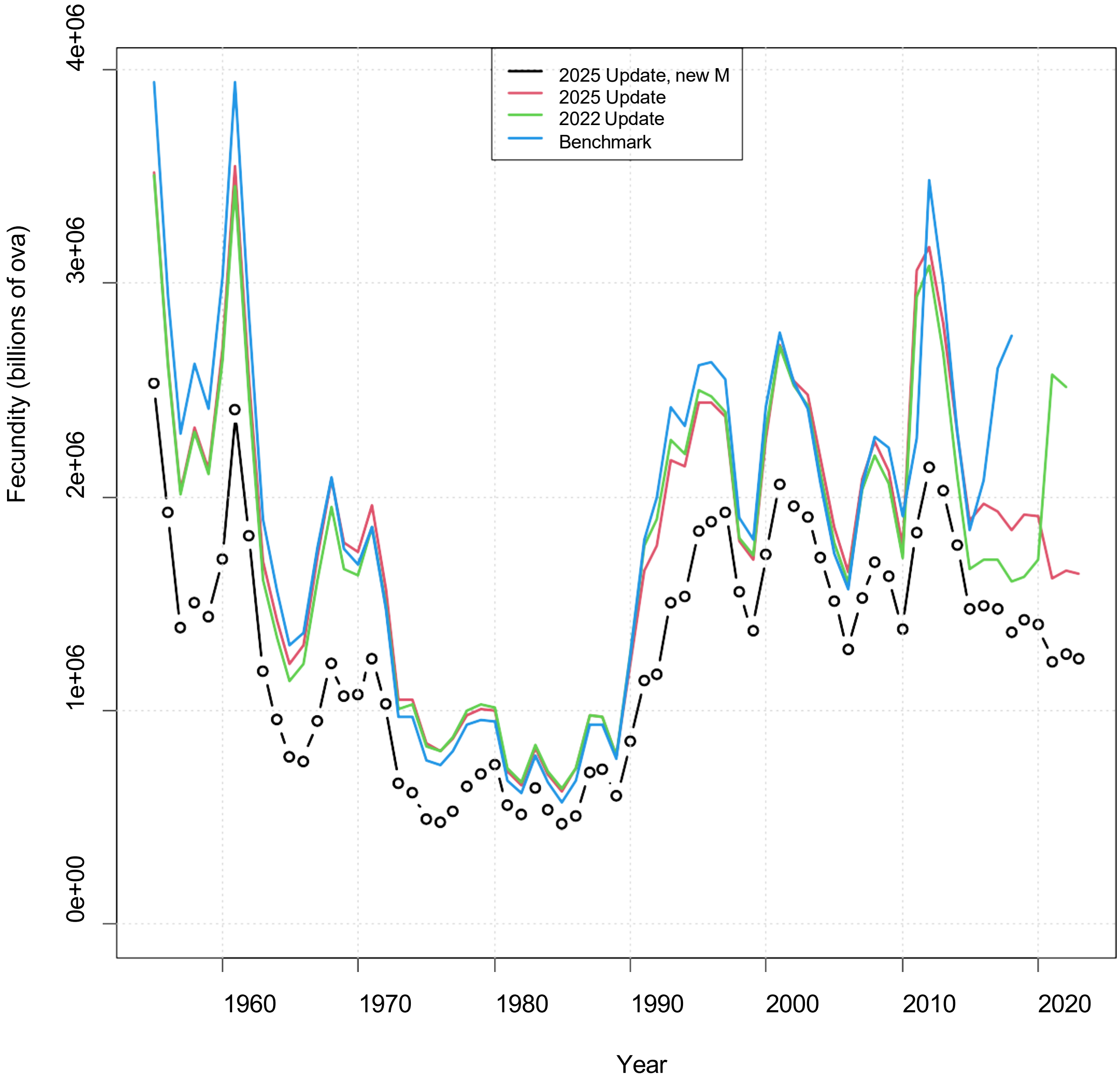

Figure A46. **Estimates of the fecundity (billions of ova) for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, the 2022 update assessment, and the last benchmark assessment.**

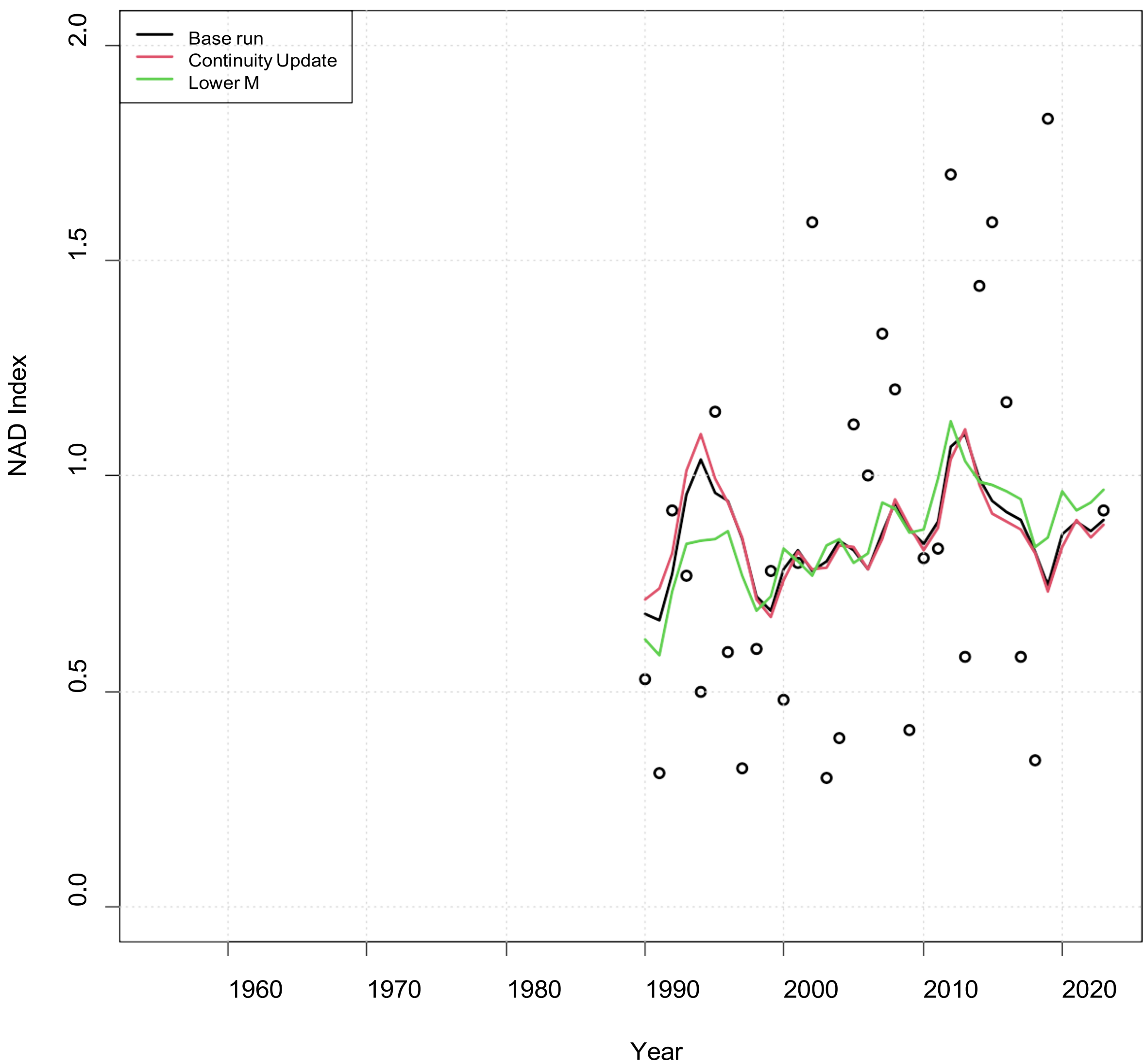

Figure A47. Fit to the observed (open circles) NAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.

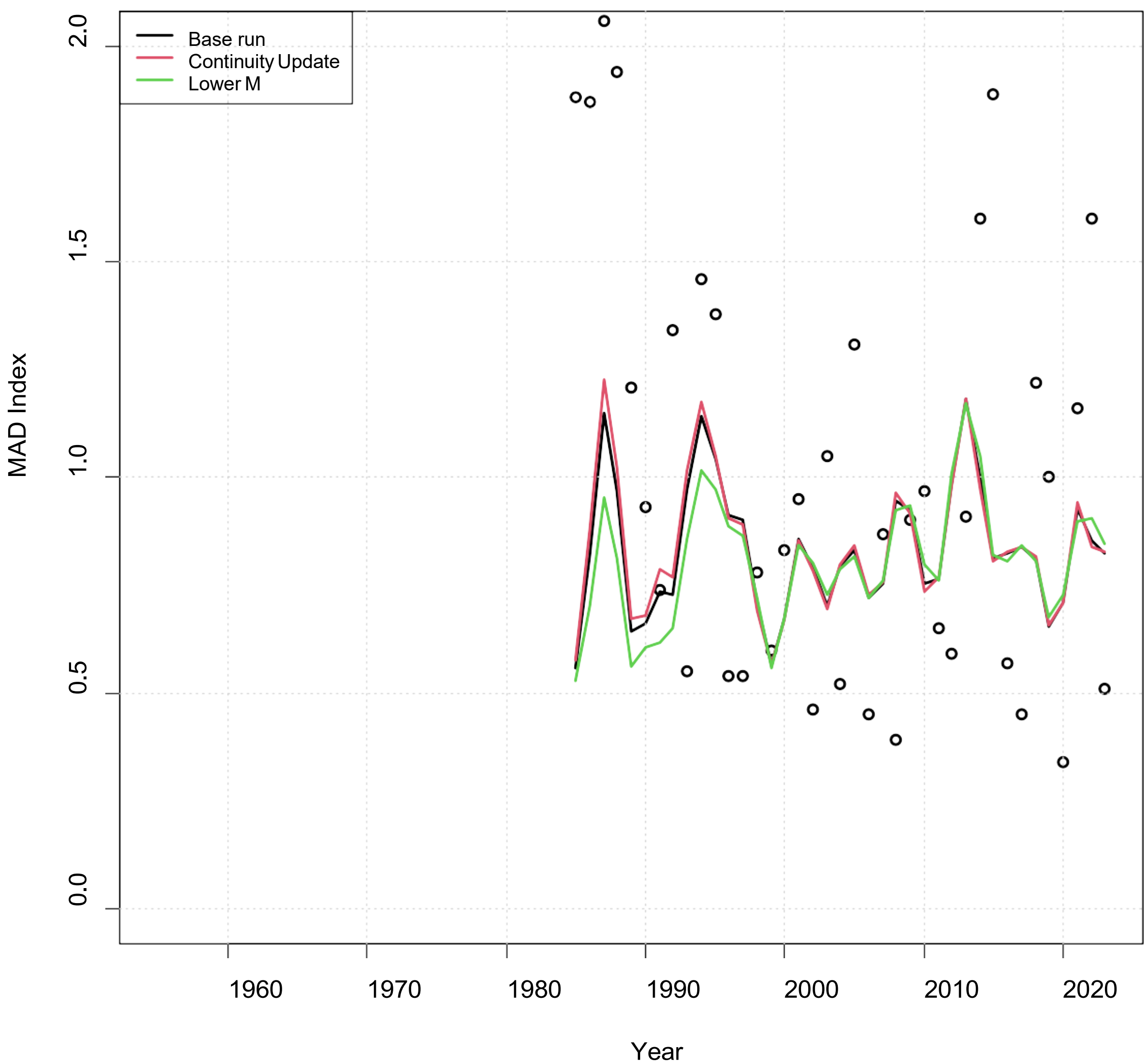

Figure A48. **Fit to the observed (open circles) MAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

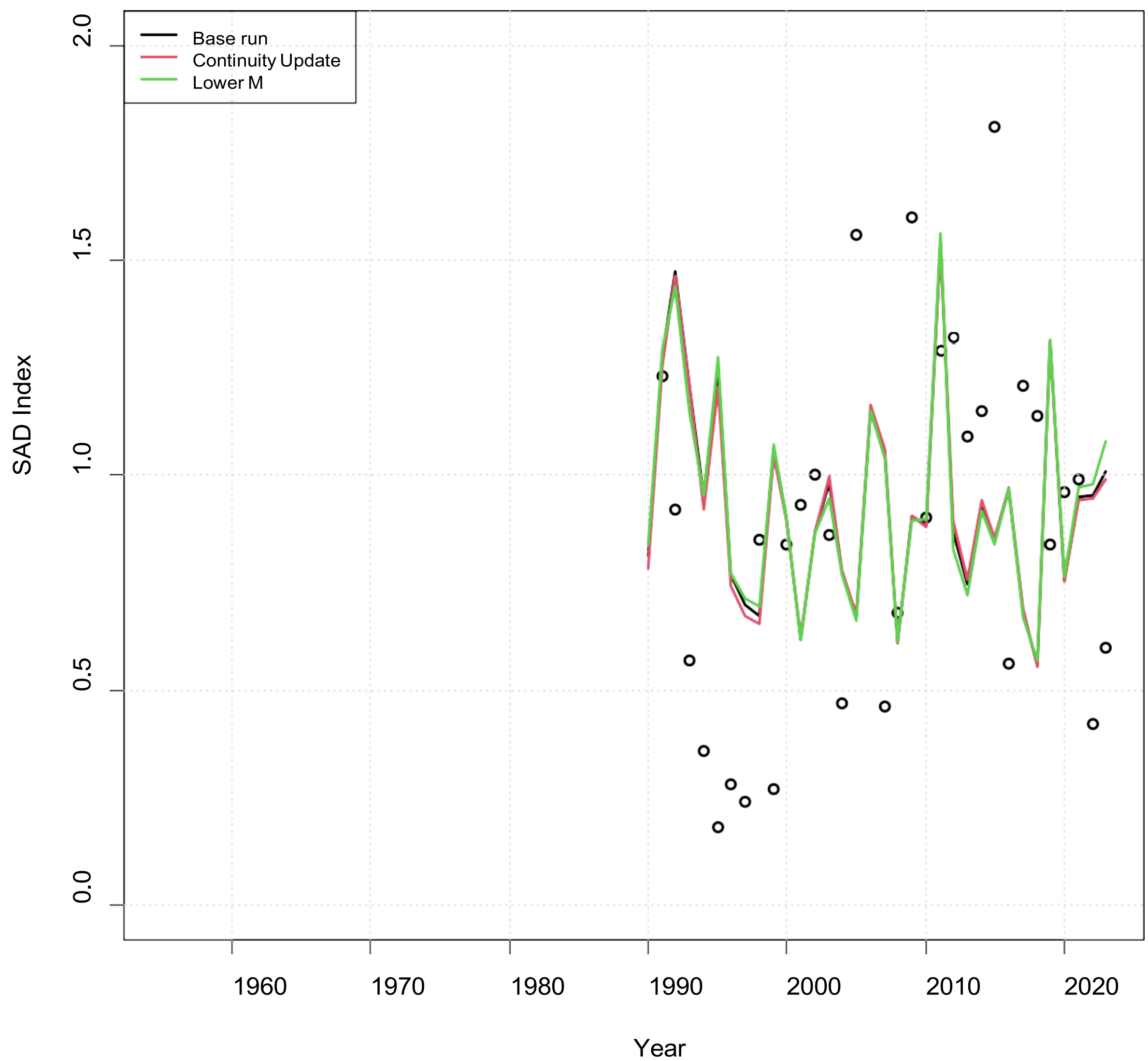

Figure A49. **Fit to the observed (open circles) SAD index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

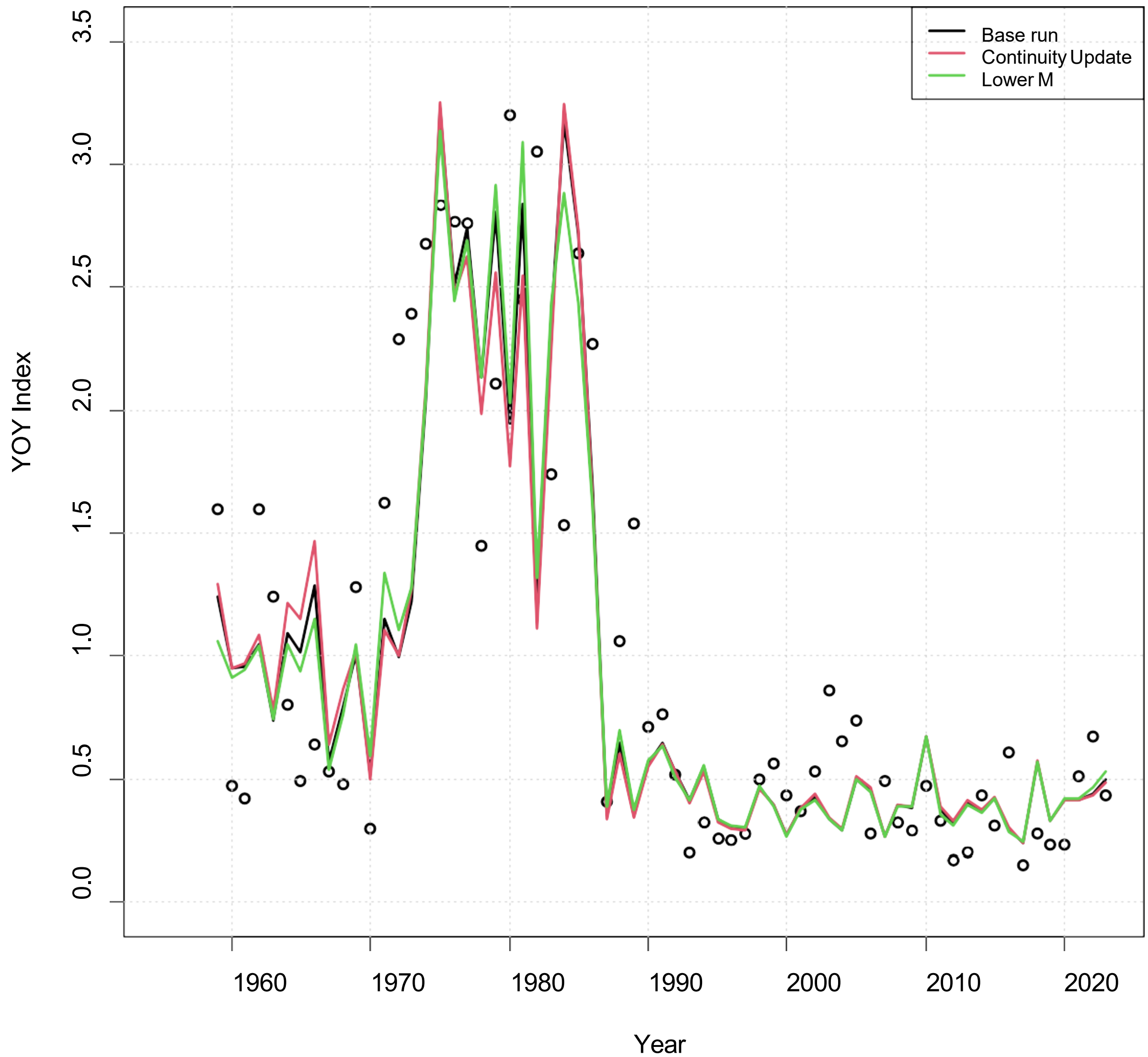

Figure A50. **Fit to the observed (open circles) recruitment index for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

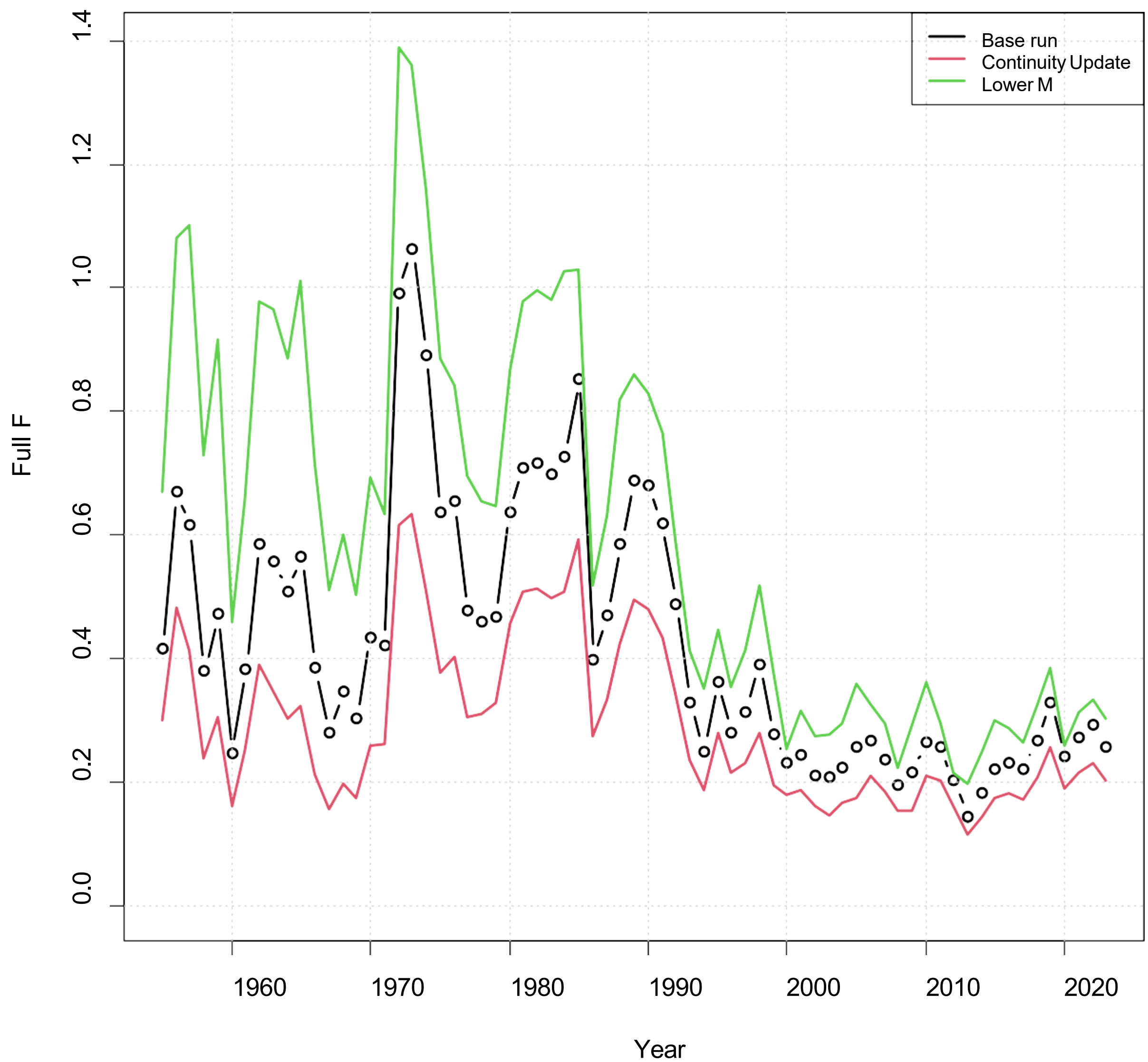

Figure A51. **Estimates of the full fishing mortality rate for the base run for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

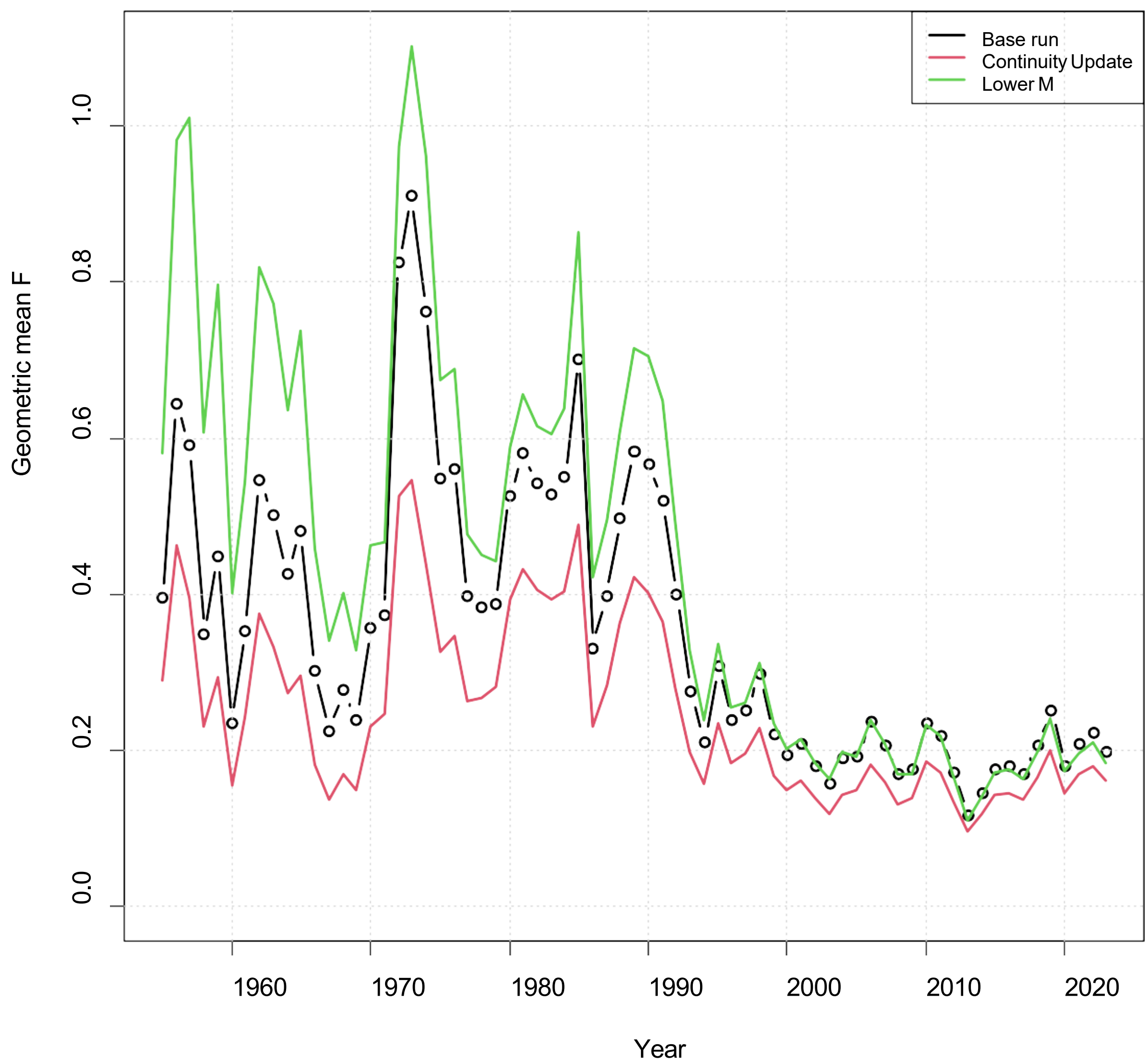

Figure A52. Estimates of the geometric mean fishing mortality rate for ages-2 to -4 for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.

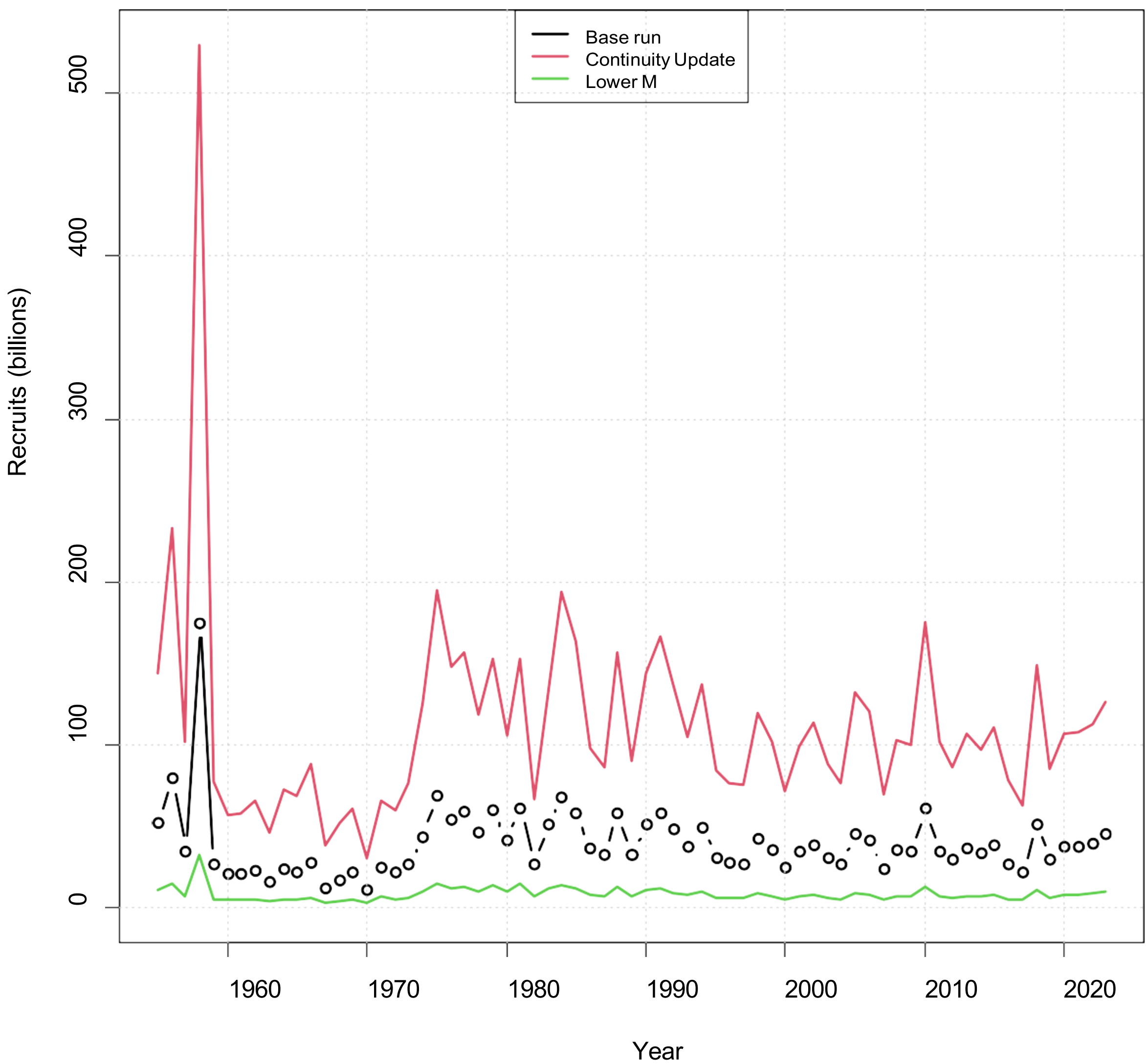

Figure A53. **Estimates of the recruitment time series for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

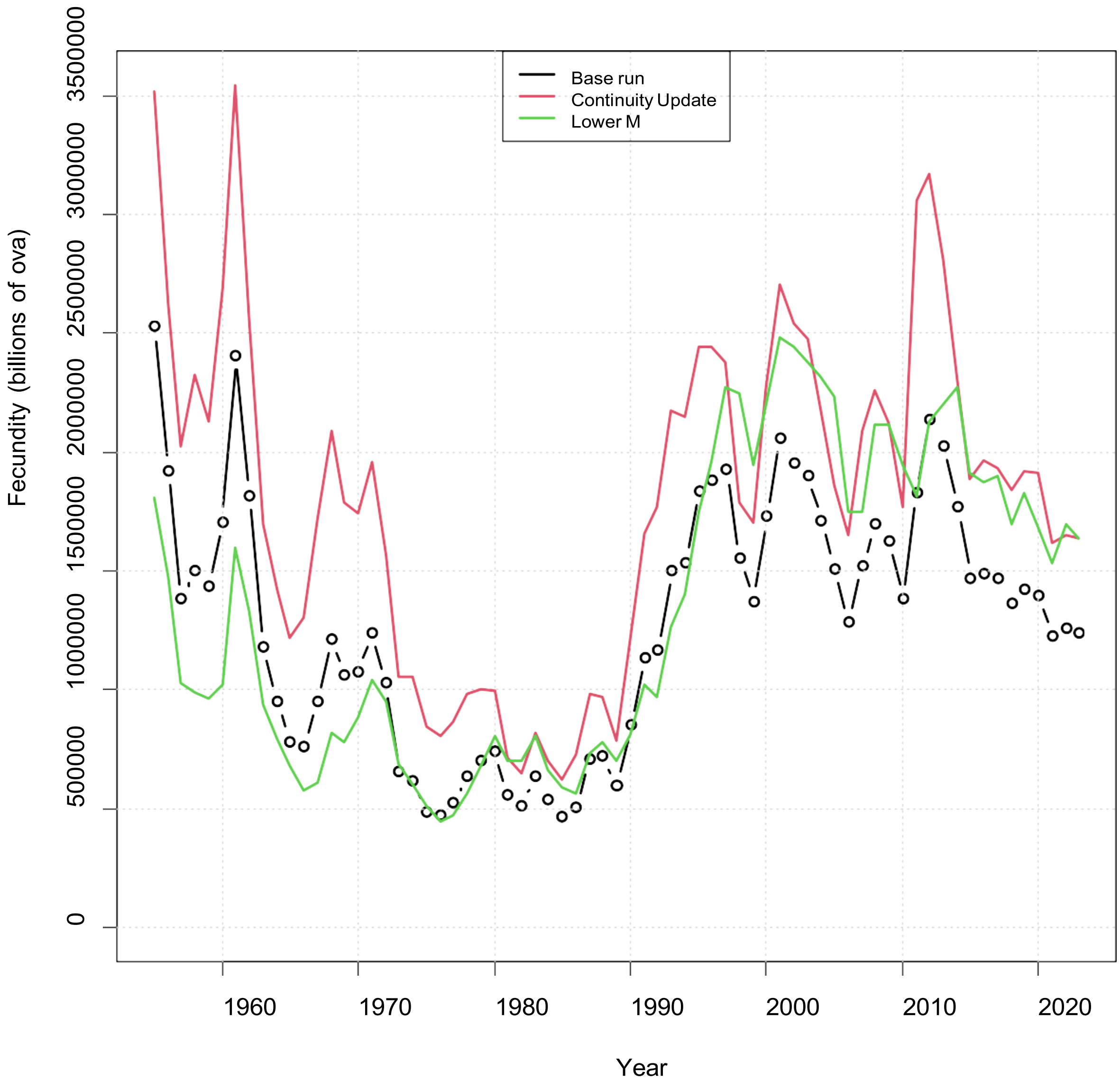

Figure A54. **Estimates of the fecundity (billions of ova) for the base run for this update assessment with a new natural mortality value (M), the 2025 update assessment as a continuity run, and a sensitivity run with a lower M.**

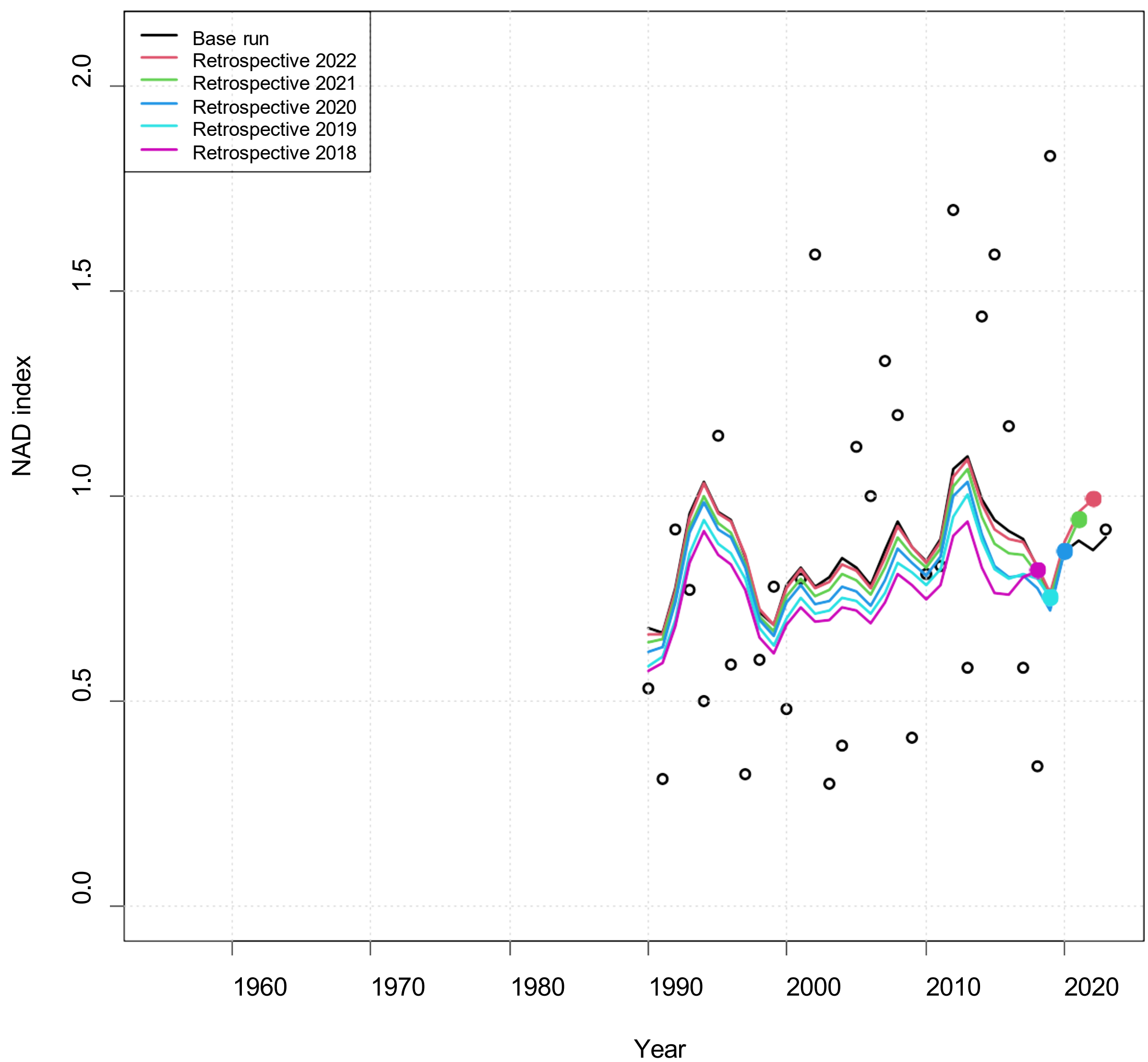

Figure A55. **Fit to the observed (open circles) NAD index for the retrospective analysis with terminal years from 2023 to 2018.**

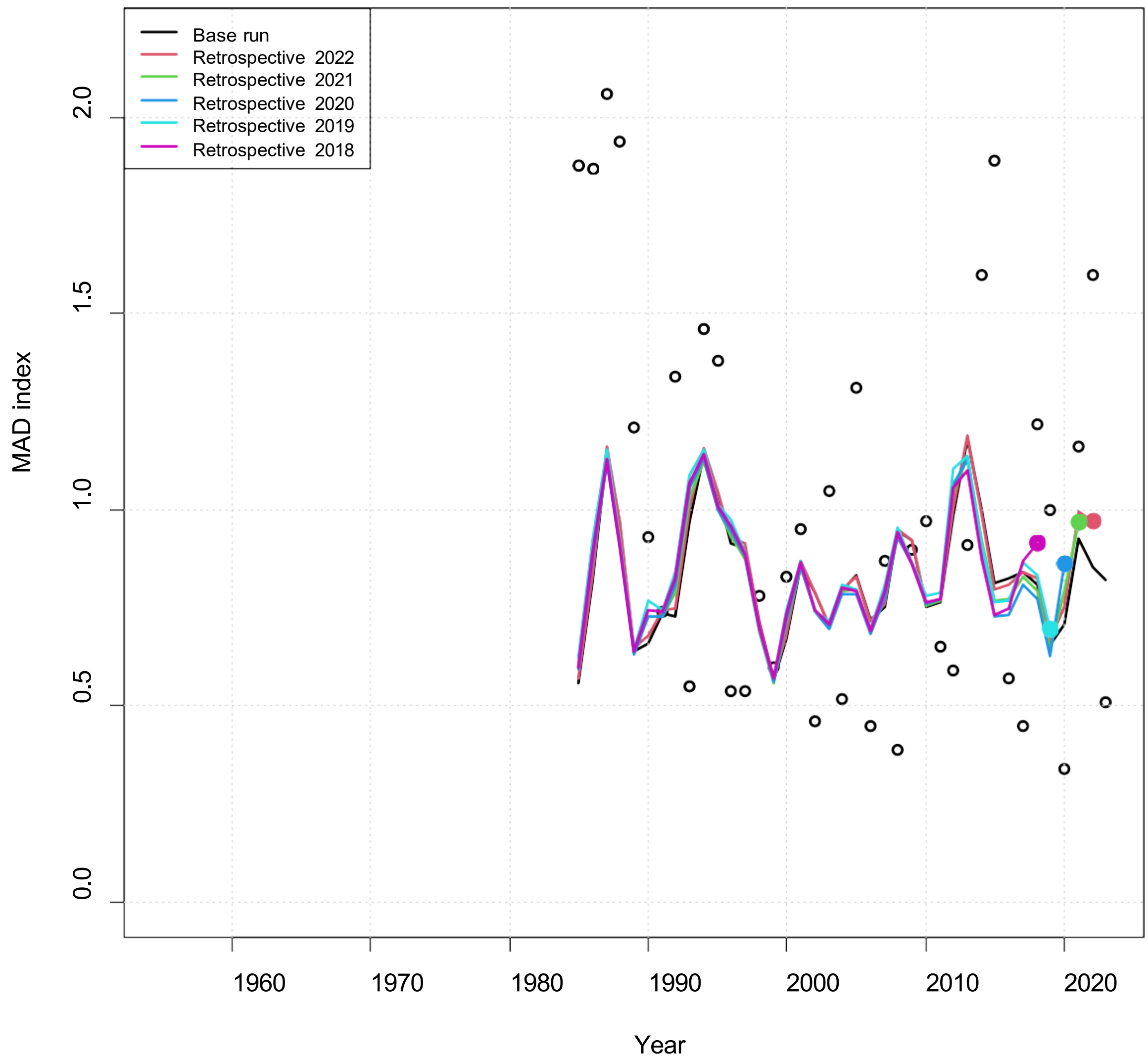

Figure A56. **Fit to the observed (open circles) MAD index for the retrospective analysis with terminal years from 2023 to 2018.**

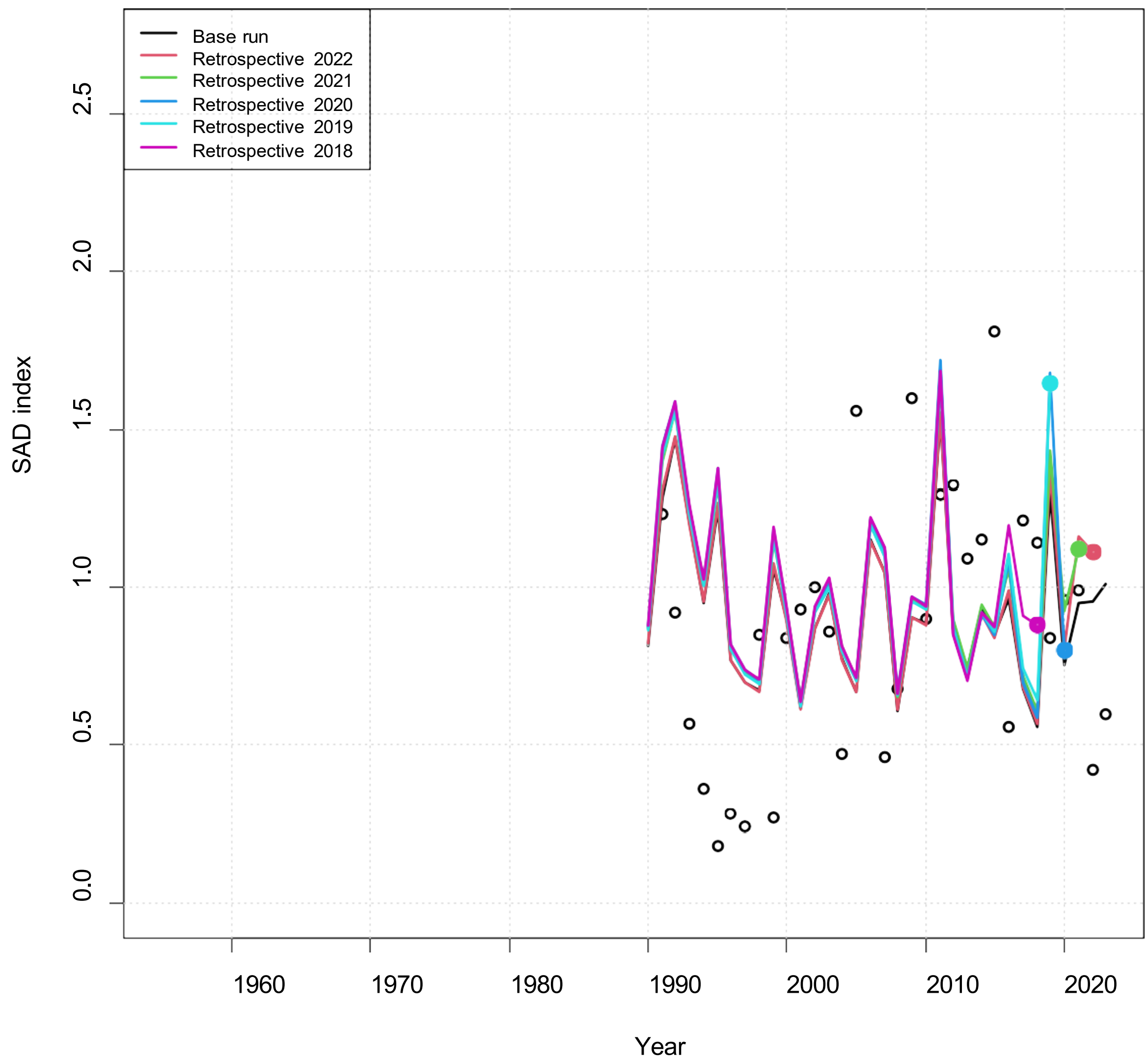

Figure A57. Fit to the observed (open circles) SAD index for the retrospective analysis with terminal years from 2021 to 2016.

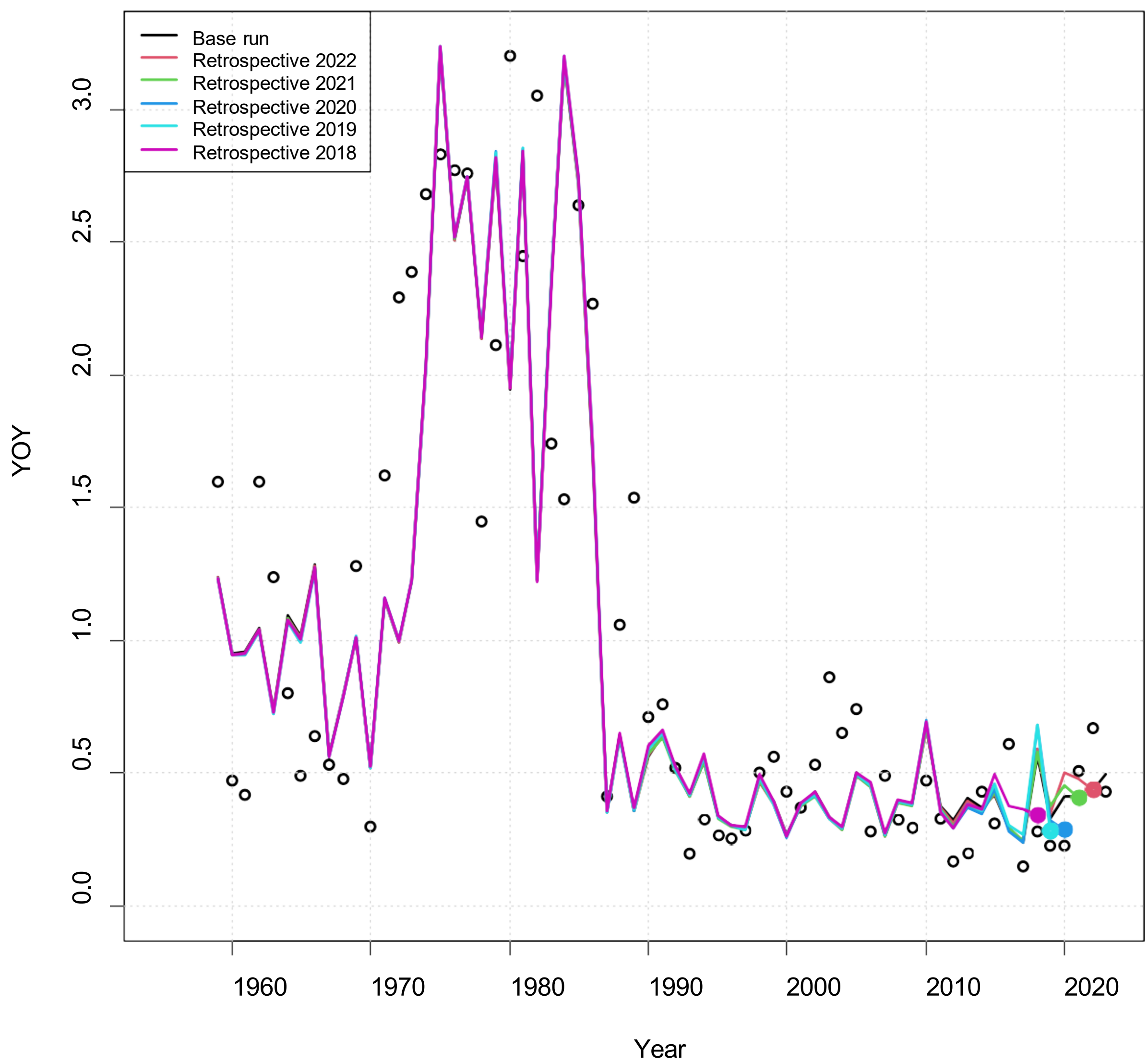

Figure A58. Fit to the observed (open circles) recruitment index for the retrospective analysis with terminal years from 2023 to 2018.

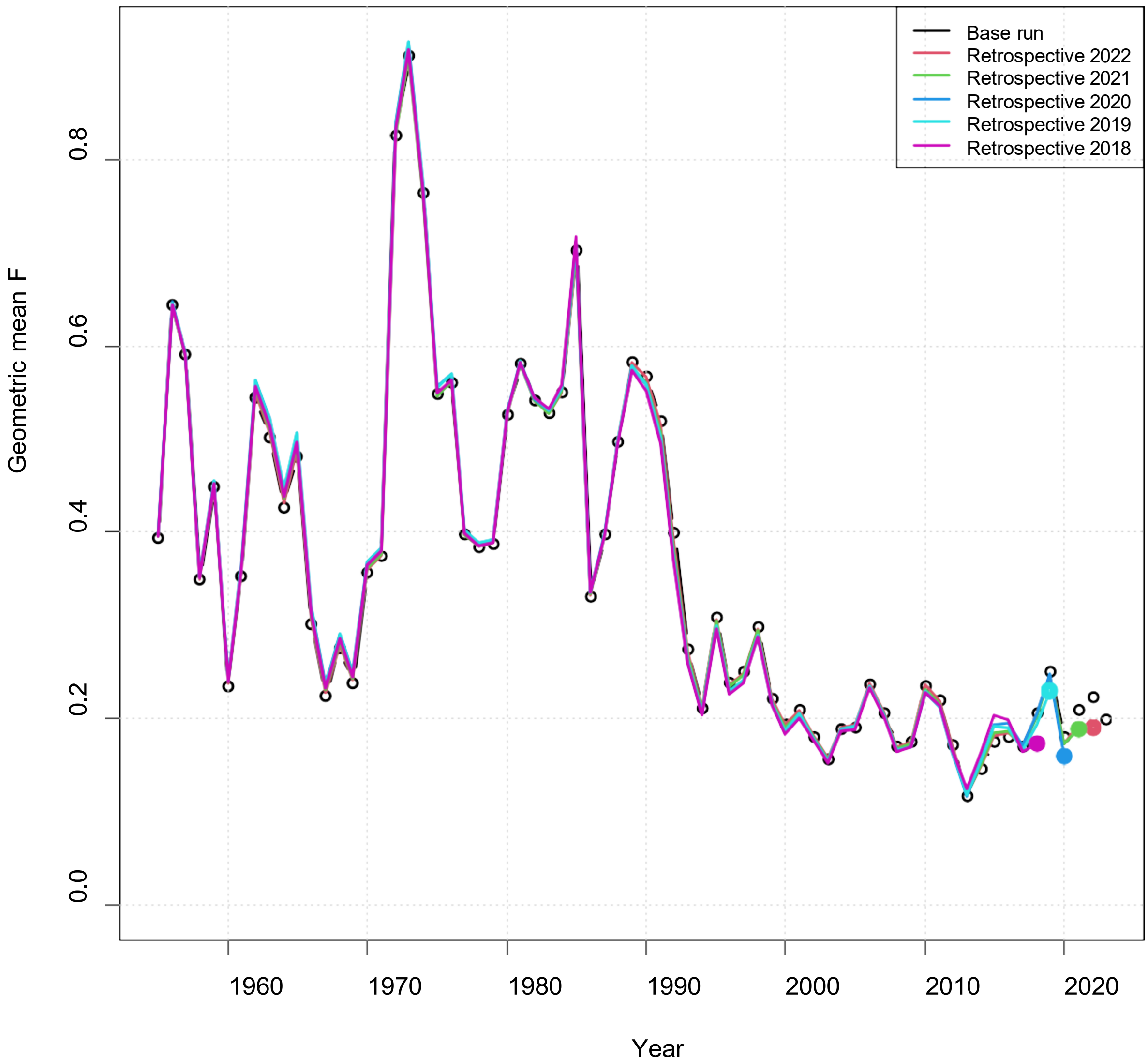

Figure A59. Estimates of the geometric mean fishing mortality rate for ages-2 to -4 for the retrospective analysis with terminal years from 2023 to 2018.

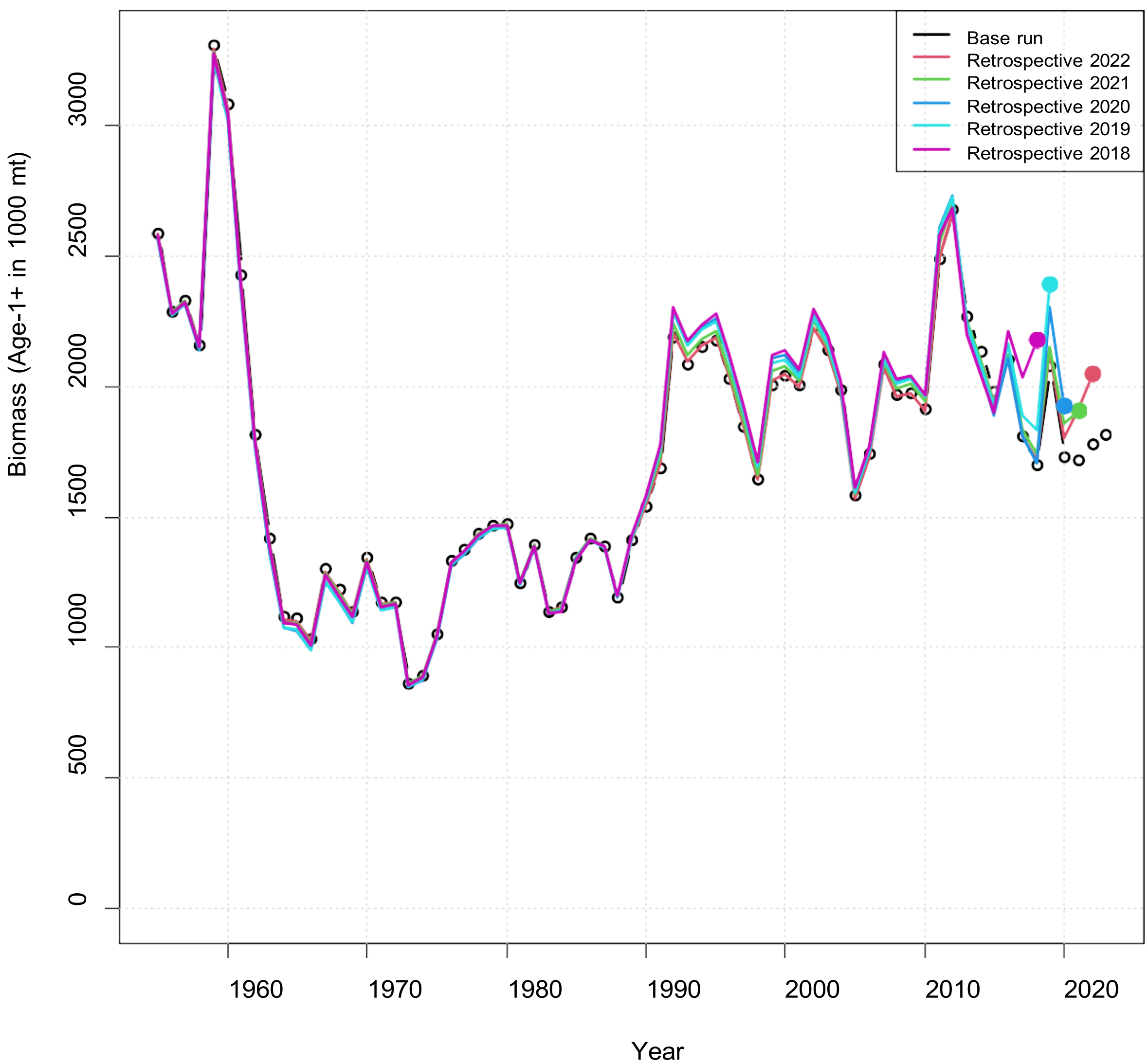

Figure A60. Estimates of the age-1+ biomass for the retrospective analysis with terminal years from 2023 to 2018.

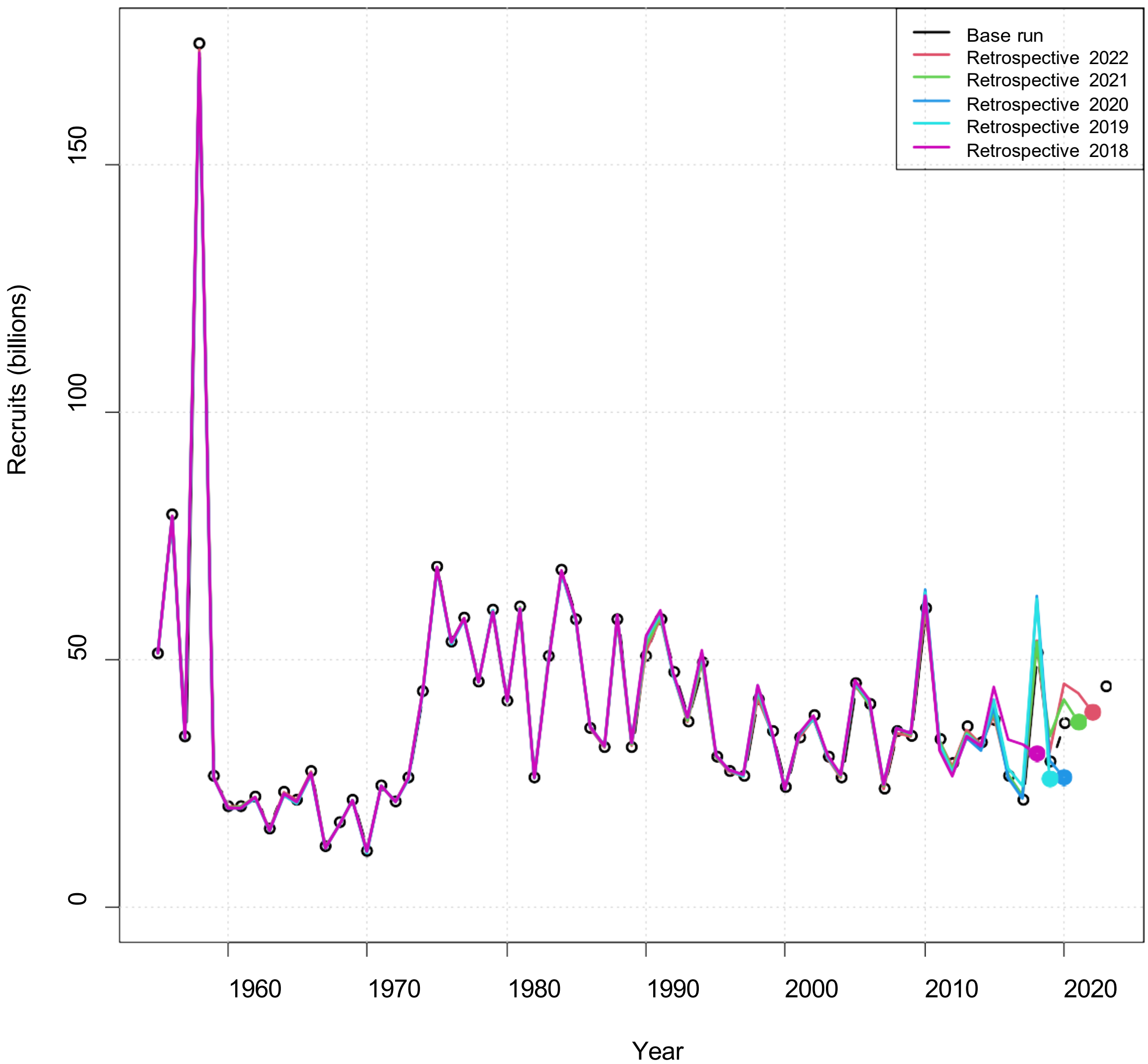

Figure A61. **Estimates of the recruitment for the retrospective analysis with terminal years from 2023 to 2018.**

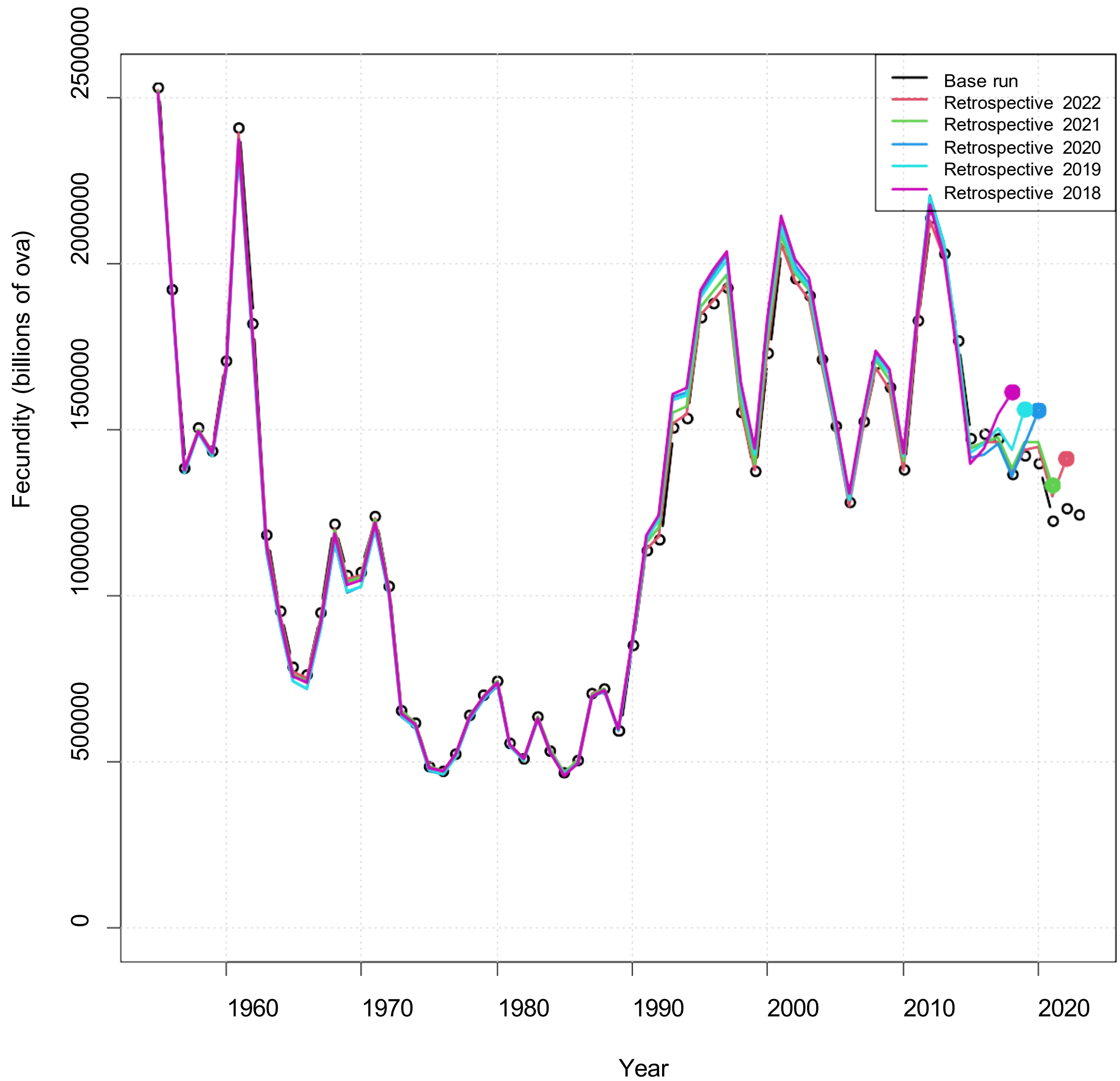

Figure A62. Estimates of the fecundity (billions of ova) for the retrospective analysis with terminal years from 2023 to 2018.

Single-Species Research Recommendations

The following is the complete list of research recommendations from the single-species benchmark assessment (SEDAR 2020a).

Research recommendations are broken down into two categories: future research and data collection, and assessment methodology. While all recommendations are high priority, the first recommendation is the highest priority. Each category is further broken down into recommendations that can be completed in the short term and recommendations that will require long term commitment. For the single-species assessment, the SAS recommends an update be considered in three years and a new benchmark be considered in six years.

Future Research and Data Collection

Short Term

1. Continue current level of sampling from bait fisheries, particularly in the Mid-Atlantic and New England. Analyze sampling adequacy of the reduction fishery and effectively sample areas outside of that fishery (e.g., work with industry and states to collect age structure data and biological data outside the range of the fishery).
2. Place observers on boats to collect at-sea samples from purse-seine sets, or collect samples at dockside during vessel pump-out operations (as opposed to current top of hold sampling) to address sampling adequacy.
3. Evaluate which proportion of bait landings by state are captured by gear versus which proportion are sampled for length and age composition to determine if current biosampling requirements are appropriate and adequate.
4. Continue to improve data validation processes for the bait fishery through ACCSP.
5. Conduct an ageing workshop to assess precision and error among readers with the intention of switching bait fishery age reading to state ageing labs.
6. Re-age historic old age samples (i.e., ages >7) to confirm the max age of Atlantic menhaden.
7. Investigate the relationship between fish size and school size to address selectivity (specifically addressing fisher behavior related to harvest of specific school sizes).
8. Investigate the relationship between fish size and distance from shore (addressing selectivity).

Long Term

1. Develop and implement a menhaden-specific, multi-year coastwide fishery-independent index of adult abundance-at-age with ground-truthing for biological information (e.g., size and age composition). A sound statistical design is essential. Ideally, it should be done coast-wide, but area-specific surveys that cover the majority of the population and are more cost-effective could provide substantial improvements over the indices currently used in the assessment.

2. Continue age-specific studies on spatial and temporal dynamics of spawning (where, how often, how much of the year, batch spawning, etc.)
3. Conduct an ageing validation study, making sure to sample older age classes.
4. Continue to investigate environmental covariates related to productivity and recruitment on a temporal and spatial scale.
5. Consider other ageing methods for the future, such as the use of Fourier transform near infrared spectroscopy (FT-NIRS).

Assessment Methods

Short Term

1. Investigate index standardization to improve CVs and explore methods of combining indices at a regional or coastwide level.
2. Explore the covariance between life history parameters to improve the understanding of uncertainty in the model.
3. Explore the error structure between MCMC and MCB.
4. Perform simulation testing on the Deyle et al. method used in the projections and determine if recruitment is accurately tracked by the method and improve short term projections.
5. Conduct a Management Strategy Evaluation (MSE).

Long Term

1. Continue to monitor model diagnostics given that the model is not robust to anomalous year-classes in the terminal year.
2. Develop a seasonal spatially-explicit model once sufficient age-specific data on movement rates of menhaden are available.

Ecological Reference Point Research Recommendations

The following is the complete list of research recommendations from the ecological reference point stock assessment (SEDAR 2020b).

The Ecological Reference Point Work Group (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.

Future Research and Data Collection

Short term

1. Expand collection of diet and nutrition 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.

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.

Modeling Needs

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.

Management Process Needs

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.

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.

SEDAR

Southeast Data, Assessment, and Review

SEDAR 102

ASMFC Atlantic Menhaden and Ecological Reference Points

SECTION III: Review Workshop Report

October 2025

SEDAR
4055 Faber Place Drive, Suite 201
North Charleston, SC 29405

Table of Contents

1. INTRODUCTION

1.1 WORKSHOP TIME AND PLACE

The SEDAR 102 Review Workshop was held in Charleston, SC August 12-15, 2025.

1.2 TERMS OF REFERENCE FOR ECOLOGICAL REFERENCE POINT EXTERNAL PEER REVIEW

1. Evaluate the justification for the inclusion, elimination, or modification of data from the Atlantic menhaden single-species assessment and the single-species assessments of the other major predator and prey species included in the ERP models.
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:
    a. Sensitivity analyses to determine model stability and potential consequences of major model assumptions
    b. 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.

1.3 LIST OF PARTICIPANTS

Review Panel

Daniel Howell ....Norwegian Institute for Marine Research /CIE Reviewer
Sarah Gaichas (Chair)...Hydra Scientific LLC
Yong Chen .... Stony Brook University/ CIE Reviewer

Analytic Team

Amy Schueller ....NOAA Southeast Fisheries Science Center
David Chagaris...University of Florida

Jainita Patel .... Atlantic States Marine Fisheries Commission
James Boyle .... Atlantic States Marine Fisheries Commission
Jason McNamee .... Rhode Island Department of Environmental Management
Katie Drew .... Atlantic States Marine Fisheries Commission
Matt Cieri .... Maine Department of Marine Resources
Patrick Campfield .... Atlantic States Marine Fisheries Commission

Staff

Emily Ott .... SEDAR
Julie Neer .... SEDAR

Observers

Jerry Ault .... University of Miami
Jiangang Luo .... University of Miami

Analytic Team via Webinar

Andre Buchheister .... California Polytechnic University Humboldt

Observers via Webinar

Addie Binstock .... Maine Department of Marine Resources
Benson Chiles ....
Brandon Muffley .... Mid-Atlantic Fishery Management Council
Brendan Turley .... NOAA Southeast Fisheries Science Center
Brooke Lowman .... Virginia Marine Resources Commission
Caitlin Craig .... New York State Department of Environmental Conservation
Catherine Wilhelm .... Virginia Marine Resources Commission
Corrin Flora .... Maine Department of Marine Resources
George Mapp .... Virginia Institute of Marine Science
Jamie Cournane .... New England Fishery Management Council
Jesse Hornstein .... New York State Department of Environmental Conservation
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Kathryn Padgett .... New York State Department of Environmental Conservation
Keilin Gamboa-Salazar .... South Carolina Department of Natural Resources
Laurie Coe ....
Manuel Coffill-Rivera .... University of South Alabama
Max Appelman .... NOAA Office of Sustainable Fisheries
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Micah Dean .... Massachusetts Division of Marine Fisheries
Michael Celestino .... New Jersey Department of Environmental Protection
Peter Himchak .... Omega Protein Corporation, Inc.
Rachael Silvas .... South Atlantic Fishery Management Council
Rob LaFrance .... Quinnipiac University

Roger Fleming ....Blue Planet Strategy
Samara Nehemiah ....Atlantic States Marine Fisheries Commission
Shanna Madsen ....Virginia Marine Resources Commission
Sharon Glen ....
Suzanna Thomas ....South Atlantic Fishery Management Council
Tina Berger ....Atlantic States Marine Fisheries Commission
Will Poston...Chesapeake Bay Foundation
Zane Ruzicka ....Wild Oceans

1.4 LIST OF REVIEW WORKSHOP WORKING PAPERS AND DOCUMENTS

Document #	Title	Authors	Date Submitted
Documents Prepared for the Review Workshop			
SEDAR102-RW-01	Revised Estimates of Natural Mortality for Atlantic Menhaden	Sydney Alhale, Jeff Brust, Caitlyn Craig, Katie Drew, Brooke Lowman, Amy Schueller, Alexei Sharov	7/25/2025
SEDAR102-RW-02	Understanding Atlantic Menhaden Population Demographics: Re-evaluation of the 1960's NMFS Tagging Data- Revised with February 2025 Supplemental Materials	Jerald S. Ault[1], Jiangang Luo[1] & Clarence E. Porch[2]	7/25/2025
SEDAR102-RW-03	Population data for including bluefin tuna in the NWACS ecosystem model	Micah Dean	7/25/2025
SEDAR102-RW-04	Blue Catfish Candidacy for the ERP Assessment	Shanna Madsen	7/25/2025
SEDAR102-RW-05	A species distribution model (SDM) approach to representing anchovies in the NWACS ecosystem model	Micah Dean and Mike Celestino	7/25/2025
SEDAR102-RW-06	Zooplankton estimates for 2025 ERP Benchmark	M Celestino, D Chagaris, A Buchheister	7/25/2025
SEDAR102-RW-07	Osprey candidacy for inclusion in the NWACS ecosystem models: a review of population and diet	Jainita Patel	7/25/2025
SEDAR102-RW-08	Virtual Assessment for the Description of Ecosystem Responses (VADER) Multispecies Statistical Catch-at-Age Model Description and Output	J. McNamee	7/25/2025
SEDAR102-RW-09	VADER Bottom-Up Feedback Data	G. Nesslage, M.	7/25/2025

	Exploration	Wilberg, J. Collie, and J. McNamee	
SEDAR102-RW-10	Investigation of Atlantic menhaden mortality rates- IN REVIEW	Jerald S. Ault and Jiangang Lou	7/25/2025
SEDAR102-RW-11	SEDAR 102 Public Comment	Public Comment	8/15/2025
SEDAR102-RW-12	NWACS FULL 2023_V2.1 Input Tables and Model	Andre Buchheister	8/18/2025
Reference Documents			
SEDAR102-RD-01	Estimation of movement and mortality of Atlantic menhaden during 1966–1969 using a Bayesian multi-state mark-recovery model	Emily M. Liljestrand[a,*], Michael J. Wilberg[a], Amy M. Schueller[b]	
SEDAR102-RD-02	Multi-state dead recovery mark-recovery model performance for estimating movement and mortality rates	Emily M. Liljestrand[a,*], Michael J. Wilberg[a], Amy M. Schueller[b]	
SEDAR102-RD-03	SEDAR 69 Atlantic Menhaden Benchmark Stock Assessment	ASMFC	
SEDAR102-RD-04	SEDAR 69 Ecological Reference Points Stock Assessment Report	ASMFC	

2. REVIEW PANEL REPORT

Executive Summary

The review panel evaluated the Atlantic menhaden Ecological Reference Points (ERP) assessment, that considers menhaden's role as a forage fish in the Northwest Atlantic ecosystem. The assessment uses ecosystem models to develop reference points that account for both predator-prey relationships and bottom-up effects of menhaden availability on predator populations. The overall process in the workflow under review involves running a single species “BAM” assessment model for Atlantic menhaden and then running an Ecopath with Ecosim (EwE) model “NWACS-MICE” to compute a suite of ERPs giving targets and thresholds for Atlantic menhaden. In addition, a larger and more complex EwE model (NWACS-Full) has been created to track impacts of harvesting menhaden across a wider range of species (though with greater uncertainties), and a statistically tuned multispecies model (VADER) is under development. All of the models were presented and reviewed as methods, but the modeling results are not being proposed for direct management use.

The ERP process was developed, reviewed, and approved previously (SEDAR 69, 2020). The current review was therefore focused on if the existing ERP procedure and updated hybrid models were still appropriate, and any changes to the underlying models, rather than as a fundamental review of all elements. While the review evaluated the proposed ERP procedure and changes in the single species stock assessment model for menhaden (principally around a revision in the value of natural mortality (M)), there is not a full evaluation of the suitability of the single species BAM assessment model in this review. Reviewers agreed with decisions made to update the single species model. However, this review is not designed to "approve" the menhaden single species assessment model.

Key review panel findings are summarized here, and full statements addressing each review Term of Reference (ToR) are in the sections below.

The panel found the use of data in the ERP models to be well justified overall. Nearly all single species stock assessment outputs used to initialize the models were previously reviewed and approved through established processes. Menhaden catch data from commercial reduction and bait fisheries are of good quality. Two notable modifications were appropriately made: menhaden natural mortality (M) estimates were updated based on a thorough re-evaluation of 1960s tagging data, and weakfish assessment inputs were modified to better reflect tagging-based mortality estimates. The review panel was impressed with the thorough and collaborative nature of the menhaden M evaluation process, and is convinced that the selection of M arising from the process is justified and reflects the best available information on menhaden natural mortality.

The panel supports the use of previously-reviewed stock assessment data sets and the consultation process with single species assessment teams for ERP model inputs. New data were also introduced in this assessment. Diet data compilation was found to be thorough and justified, incorporating five fishery-independent surveys and literature sources. Additional data compilation and analysis efforts led to significant improvements in model inputs for traditionally data-poor functional groups including anchovies, zooplankton, phytoplankton, benthic invertebrates, bluefin tuna, osprey, and marine mammals.

The panel endorses the selection of NWACS-MICE as the most appropriate ERP model. This intermediate-complexity model balances data availability with biological realism while including all key managed fish predators of menhaden, particularly striped bass (identified as the most sensitive predator). The model appropriately incorporates both top-down predator effects and bottom-up forage effects. The more complex NWACS-Full model was deemed unsuitable for tactical management due to complexity and data requirements, while the VADER model was not ready for management use.

The panel finds the ERP methodology to be sound, using a hybrid approach with NWACS-MICE to generate ecosystem reference points and the single-species BAM model for short-term projections. The method appropriately evaluates trade-offs between menhaden and striped bass fishing levels rather than providing prescriptive catch estimates, allowing managers to make decisions based on their risk tolerance and management objectives.

Model diagnostics were found to be appropriate for each model type. For the menhaden assessment, sensitivity analysis focusing on alternative natural mortality values showed similar model stability and diagnostics, with generally expected contrast in biomass and recruitment outcomes at different values. For NWACS-MICE, limited time prevented extensive sensitivity analysis, but the panel endorsed focused work on predator-prey vulnerability parameters. The panel recommends developing a suite of plausible model variants to better characterize uncertainty in future assessments.

Methods for characterizing uncertainty were appropriate within the constraints of available software and time. The menhaden assessment incorporated uncertainty in natural mortality and fecundity through Monte Carlo bootstrap analysis. However, uncertainty characterization in the ecosystem models was limited, and the panel recommends developing methods to evaluate parameter uncertainty and create multiple plausible model configurations.

The panel endorses using the menhaden single-species assessment for estimating menhaden stock biomass, abundance, and exploitation rates. Menhaden stock status will then be evaluated by comparing single-species results to ERPs derived from the NWACS-MICE model. The panel recommends this methodology as an appropriate tool for managers to evaluate trade-offs, rather than specifying exact fishing levels based on this review. The panel also recommends standardizing documentation of the modeling process to ensure consistency and transparency in identifying the base model run and sensitivity model runs.

The panel strongly supports the proposed research recommendations, emphasizing three key priorities:

1. Data Collection: Continue and expand collection of population, life history, and diet data across all ecosystem components
2. Spatial Assessment: Hold a stakeholder workshop to determine clear objectives for a potential spatial ERP assessment before model development
3. Assessment Coordination: Ensure adequate time between single-species assessments and ERP work for proper model development and analysis

The panel endorses asynchronous benchmarks for menhaden and ERP assessments, coordinated with other species assessment timelines. The next ERP benchmark assessment should occur after the 2026 MRIP recreational data recalibration and 2027 striped bass benchmark assessment, with a minimum of one year between finalized single-species assessments and ERP work.

In summary, the ERP assessment provides managers with a scientifically sound framework for evaluating ecosystem trade-offs in menhaden management. The methodology advances ecosystem-based fishery management by considering menhaden's dual role as both a harvested species and forage base for managed predators, particularly striped bass. The approach enables informed decision-making about acceptable risk levels while balancing multiple fishery management objectives. The assessment will require updates following the 2026 MRIP recalibration and 2027 striped bass benchmark, with the next full ERP benchmark recommended for 2028 or later to allow sufficient ERP model development time.

Addressing the Review Workshop Terms of Reference

ToR 1. Evaluate the justification for the inclusion, elimination, or modification of data from the Atlantic menhaden single-species assessment and the single-species assessments of the other major predator and prey species included in the ERP models.

The review panel found that overall, the use of the data is well justified. As noted in the ERP assessment report, nearly all stock assessment outputs used to initialize the ERP models (aside from weakfish) were reviewed and approved in other processes, and all time series data used to fit the ERP models were likewise reviewed and approved by the relevant assessment working groups and review panels (and SSC) for each stock. While inputs to stock assessments varied somewhat in data quality, each input has been determined by previous peer review to be best available information for each assessment. Several assessments rely on MRIP recreational CPUE data, which is scheduled for recalibration in 2026; it was noted that MRIP recalibration may change the results of future ERP species assessments (e.g., striped bass and bluefish). However, this still represents best available data in the assessments at present, and its inclusion is therefore justified in the ERP models.

Menhaden catch is mainly from the commercial reduction and bait fisheries; this data set is of good quality and its use well justified. Menhaden survey data, consisting of combined adult indices for the NAD, MAD, and SAD regions and 16 state surveys for YOY, inclusion is well justified. One possible source of uncertainty for these data sets is change in stock availability to the surveys, which may have been accounted for (at least to some extent) by the standardization the WG had done. Menhaden maturity and fecundity are age and time varying, which is well justified. Menhaden selectivity is different for reduction and bait fisheries, with the bait fishery generally harvesting larger sized individuals than the reduction fleet in the same geographic area. This has been determined by an analysis of size composition data from the fisheries, reflecting the best available information. However, changing distributions and fishing grounds as a result of changing processing facility locations, changing regulations, and changing stock distributions may affect fishery selectivity. Although this is partially accounted for with selectivity time blocks in the assessment, this may be a source of uncertainty in the menhaden stock assessment.

In two notable cases, previously used ERP assessment inputs required modification: menhaden natural mortality (M, discussed here) and assessment inputs for weakfish (discussed below). As in the previous ERP assessment, menhaden M was estimated empirically using an extensive tagging study completed during the late 1960s. The tagging study included over a million tagged menhaden landed from Florida to Massachusetts and collected information on tag shedding, tag mortality, and reduction plant magnet tag-detection efficiency. For this ERP assessment, a thorough comparative study was done to evaluate M estimates from differently parameterized tag-recapture models (Liljestrand et al. 2019, Ault et al. 2023). The menhaden stock assessment team evaluated all the available information and identified factors (i.e., spatial coverage, potential missing data, use of confidential data, different methods to estimate magnet efficiency) that may influence the analysis. Several errors in the previous analysis were discovered and corrected during this evaluation. The primary difference between proposed M estimates came down to the estimation of magnet efficiency, which functions as a tag reporting rate. Ultimately, the stock assessment team based the estimate of magnet efficiency on the extensive dataset resulting from tag seeding experiments conducted alongside the tagging study. The alternative tag-recapture model used a different method to apply the magnet efficiencies that resulted in a mismatch with empirical magnet efficiency data, resulting in lower estimated magnet efficiency than was observed in the data. In addition, the tag-recapture model parameterized with the full range of magnet efficiency data better reflected menhaden stock distributions and movements when compared with the alternative model.

The review panel was impressed with the thorough and collaborative nature of the evaluation process, and is convinced that the selection of M arising from the process is justified and reflects the best available information on menhaden natural mortality. However, the panel notes that the age-varying, time-invariant M based on the late 1960's tagging data may reflect different environmental conditions than are experienced by the stock at present (or in 1985 when the ERP models are initialized). The NWACS-MICE and NWACS-Full models both show generally increasing trends in menhaden predation mortality over time. A similar tagging study could not be conducted now, because movement estimation requires tag returns at multiple plant locations along the coast, and currently only a single plant is operational. However, age data from the survey could provide information to validate and potentially revise the estimate of M.

The review panel found that modifications to the weakfish stock assessment were justified to provide inputs to the ERP models. A weakfish tagging study estimated higher natural mortality than was used in the stock assessment. The stock assessment M was capped at an artificial level of 1.0, so the ERP WG recreated a weakfish assessment with the tagging-based M. The ERP models were initialized with the increased weakfish biomass and total mortality estimated from the modified weakfish assessment model, and found improved EwE performance with the increased weakfish mortality.

ToR 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.

This review focused on the ERP assessment rather than work completed and reviewed for the relevant single species assessments. The review panel supports the use of previously-reviewed stock assessment fishery dependent and fishery independent data sets in the ERP assessment. The panel also supports the process of the outcomes of individual reviews for the single-species assessment being used to base the data for the ERP models. The quantity and range of data is too large to be efficiently reviewed in a single meeting, and relying on previous reviews is the only viable methodology. The panel also supports the ERP WG's consultation process with each ERP species assessment team to determine which datasets to include for ERP model fitting as primary and sensitivity indices in cases where many datasets are used in assessments. This methodology ensures that efficient use is made of existing stock assessment processes, and allows ERP models to manage the number of inputs while maintaining consistency to the extent possible with existing assessments.

The panel received an overview of menhaden stock assessment inputs, and a briefer description of inputs for the assessment of all other ERP species. For existing stock assessment inputs, each available data set was evaluated during the respective species assessment processes. For the menhaden assessment, data calibration and/or standardization was well done and justified, data source uncertainty (within a survey program and across survey programs) was well described and quantified, gear selectivity for different survey programs was decided based on extensive analysis of length composition data, and composite YOY and adult indices from fishery-independent sources were well defined. The review panel assumes that vetted stock assessment inputs for the other species also meet best available data standards. While some assessments (e.g., striped bass) included survey data from many survey programs, but with limited spatial coverage, and possible hyperstability in recreational CPUE data was not considered for assessments using this data, use of all of these datasets was justified through previous assessment reviews.

The review panel noted that spatial and temporal limitations were evaluated for each dataset as part of each assessment, but temporal changes in species distributions are not always evaluated within individual assessments, which may affect the trophic interactions between menhaden and their predators. During the review, the ERP Working Group (WG) presented additional available information on potential changes in the center of gravity for bluefish and spiny dogfish, with other species distributions not fully analyzed. As the ERP continues to develop in the future, a more comprehensive multispecies data analysis may be necessary to evaluate whether ERP species distributions are changing and contributing to different trophic interactions over time.

The panel focused its review on the additional data series which did not come from the stock assessment datasets, including updated diet data for all ERP species, updated input biomass data for benthic invertebrates, and updated biomass with new time series for anchovies, zooplankton, phytoplankton, bluefin tuna, osprey, blue catfish, and marine mammals.

The review panel found the compilation and treatment of diet data in the ERP assessment to be thorough and justified. Diet data sources were expanded in this assessment relative to the 2020 ERP assessment. Five fishery independent surveys spanning the full coast and subsets of the coast and estuaries were included along with literature sources to generate diet estimates. Diet data were used in two ways: first to identify key fish predators by ranking estimated consumption of menhaden by each predator, and second to generate diet compositions for input into the NWACS models. Consumption rankings to identify fish predators focused on survey diet sources, while diet composition estimation used a bootstrap approach to resample across both survey diet sources and literature diet compositions based on the number of stomachs sampled. Resampled diets were fitted to a Dirichlet multinomial which allowed estimation of both uncertainty and point estimates of diet proportions based on a broad range of data sources.

It should be noted that the quality of data for parameterizing parts of ecosystem models is relatively poor, including unassessed fish species, lower trophic level plankton and benthic invertebrates, and upper trophic level birds and mammals. The ERP WG improved information for key groups across these traditionally data poor categories for this assessment. Survey datasets and habitat information were integrated within a spatial model to estimate annual biomass for anchovies, an important unassessed forage group. Zooplankton and phytoplankton time series were developed from existing surveys and global ocean model reanalysis data, respectively. Historical benthic datasets were analyzed to produce updated input biomass for benthic invertebrates, which represent an important component of some managed fish diets. Bluefin tuna diet studies from the Gulf of Maine were combined with Western Atlantic stock assessment estimates and a habitat utilization model to better represent this key highly migratory species in the region and its recent menhaden consumption. Time series and population survey data for migratory birds was used along with regional literature to develop biomass and diet inputs for osprey. Literature review improved diet compositions for marine mammal groups. While more can always be done, these represented substantial improvements to the input data used for NWACS model parameterization. Other potential datasets for future consideration are listed in the research recommendations section (ToR 9).

The review panel found the inclusion of new biomass estimates and biomass time series indices for anchovies, benthic invertebrates (biomass only), zooplankton, and phytoplankton to be well justified in both NWACS models to provide improved information on low trophic level dynamics important to menhaden and their ERP predator species. Replacement of a general highly migratory species (HMS) group in the NWACS-Full model with the more data informed bluefin tuna group is well justified. Inclusion of osprey and improved marine mammal groups in NWACS-Full is also well justified to evaluate potential interactions with menhaden and these groups highlighted by stakeholders. However, exclusion of bluefin tuna, osprey, and marine mammals from the NWACS-MICE model is well justified based on the combined criteria of degree of menhaden consumption, availability and quality of life history data, and management relevance to ASMFC. Exclusion of the blue catfish group from both models is well justified due to low spatial overlap with menhaden and only episodic, relatively low seasonal consumption of menhaden.

ToR 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).***

The selection of ERP model to evaluate Atlantic menhaden's role as a forage fish is evaluated here. The method used to project an ERP model under different combinations of F for menhaden and striped bass is evaluated in detail under ToR 4. The panel notes that the methodology presented here focuses on the ERP assessment. Atlantic menhaden fishing mortality, biomass, and abundance come directly from the single species model, are input to the ERP models, and as described above are not the subject of this review. The panel notes that the multispecies statistical catch at age VADER model could be producing some of these quantities for all modeled stocks if operational, but that model is not currently in a state for direct use to inform management advice.

Three models were reviewed: two Ecopath with Ecosim (EwE) models with moderate (NWACS-MICE) or high (NWACS-Full) ecological complexity, and one moderate complexity multispecies statistical catch at age model, VADER. NWACS-MICE is recommended by the ERP working group to provide ERPs by striking a balance of model complexity, data needs and biological realism. The previous version of NWACS-MICE was used to establish ERPs in 2020.

The review panel agrees that NWACS-MICE is the most appropriate ERP model given available data and the life history of all ERP species. The model includes both top down effects of predators on menhaden and bottom up effects of menhaden on predators, as required by the management objectives. NWACS-MICE uses previously vetted stock assessment inputs for initialization and model fitting (see ToRs 1 and 2), and includes all key managed fish predators of menhaden, including the most sensitive managed fish predator, striped bass. Multistanza groups are used in the NWACS-MICE model to represent ontogeny in feeding, habitat, and fishery selectivity for the focal species. NWACS-MICE is preferable at this time to both the multispecies statistical catch at age model VADER and the full ecosystem model NWACS-Full. Despite attempts to include bottom-up forcing of menhaden on predators in VADER, the model was not considered ready for management at this time by the ERP WG, and therefore cannot meet the management objectives. NWACS-Full contains the ERP groups in NWACS-MICE plus many other functional groups representing the full ecosystem. This makes updating the model on a management timeline difficult, with higher data requirements and more time required to evaluate model parameterization and fits. In addition, there are a larger number of trophic interactions with poor or limited data, which makes the overall model results more uncertain. To simplify model fitting, NWACS-Full is currently fitted to stock assessment results, rather than input indices as NWACS-MICE is. Ideally ERP models would be fit to the same indices as used in the stock assessments if possible to make them comparable.

The review panel finds that NWACS-MICE is performing adequately to produce a reasonable reconstruction of the ecosystem dynamics, especially in the menhaden and striped bass components. We note that there are issues, especially with reconciling the spiny dogfish assessment results with other assessments included in the models.

The NWACS-MICE vulnerability parameters were optimised within the model, using a weighting scheme for the different datasets. A stepwise addition of complexity was applied for calibration with 13 model types and potentially 30 variations of each type. Initial fit results eliminated some model types and variations from further consideration, resulting in a total of 150+ model configurations throughout the model development process. The review panel found this approach to be reasonable. For future efforts, the panel suggests evaluating the importance of the selection of weights for the different data sets would be useful to highlight how sensitive the model results are to small changes in those weights (see TOR 5a). In addition, robustness of the final base model parameterization to different fitting protocols should be evaluated.

Prior to use in tradeoff analysis, NWACS-MICE model projections across all species were evaluated for appropriate responsiveness to fishing pressure. Some parameterizations were unable to achieve dome shaped yield curves for ERP species, or could not achieve target biomass under any fishing mortality scenario. These parameterizations were not carried forward. In some cases, modifications to a key vulnerability parameter or to EwE inputs were necessary to achieve appropriate response to a range of future fishing while maintaining fits to historical data. The review panel agrees that projection performance is essential for the use of the NWACS-MICE model in management, and appreciated the transparency of the ERP WG’s presentation.

One key task for this review is to choose which model configuration should be used for advice. Two potentially viable model configurations (150 and 153) were presented at the review, which differed in their handling of spiny dogfish predation on striped bass. The SEDAR 102 review focused on model 150 as the base case, with 153 only presented and reviewed as a sensitivity test. The review panel endorses the choice of 150 as the model used for advice, with 153 as part of the sensitivity analysis.

The review panel endorses the choice to include current (low) status quo recruitment for Atlantic herring. This is the most realistic under current conditions, and is the clear choice for current management advice. Should herring recruitment improve in the near future, the current recruitment formulation would continue to give conservative advice until such time as the model could be revised.

In addition to the concrete management advice, the review panel recommends that the ERP WG present a preliminary evaluation of the spiny dogfish uncertainty at the next management meeting. This evaluation could use run 153 and perhaps a sensitivity test on a carefully selected range for the relevant predation vulnerability to present an idea of how the uncertainty associated with spiny dogfish predation on striped bass impacts on the overall model performance and ERP estimation.

Future work should focus on creating a suite of different model formulations which give “plausible” model outputs. These could be generated by making changes to the base run. Such a suite of models could then be used to characterize and compare the key uncertainties in the model. Once there is such a suite of models, one possibility would be to consider ensemble model averaging in future management advice. However, the suite would have clear utility in investigating uncertainty regardless of any formal ensemble modelling.

The NWACS models do estimate biomass series, and the estimated menhaden biomass time series from both models has moderate coherence with the single species assessment (noting that perfect match would not be expected because EwE models do not generally estimate annual recruitments as single species models do). In addition, both NWACS models identify striped bass as the most sensitive managed fish predator of menhaden, despite differences in model complexity, data needs, and fitting procedures. This reinforces the use of striped bass as the “indicator species” accounting for menhaden’s overall role as forage, beyond specifically striped bass-focussed questions.

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

The review panel finds that the ecological reference point methodology is sound, and derives directly from the previous review. A hybrid modeling approach is used with NWACS-MICE to generate ecosystem-level reference points (ERPs) and the menhaden assessment BAM model used to project alternative short-term operational TACs and associated risks of exceeding ERP reference points. The NWACS-MICE model is projected forwards under different combinations of F for menhaden and striped bass, which was identified as the most sensitive menhaden predator. This was then used to identify menhaden fishing levels required to reach target and threshold biomass levels for the striped bass when striped bass are fished at their F reference points. The total allowable catch then comes from the single species assessment which avoids relying directly on the more uncertain EwE model for absolute abundance estimates.

The single species BAM model projections incorporate uncertainty in menhaden M and overall stock fecundity from a bootstrap monte carlo analysis, and project recruitment using non-linear time series analysis. This allows estimation of probabilities of being within or exceeding EPR targets and thresholds for F and fecundity. The review panel finds this probabilistic method appropriate, and notes where additional uncertainty might be considered under ToR 6.

The review panel finds it appropriate that this methodology is presented as a way for managers to evaluate trade offs and to answer "what if" questions, rather than to provide prescriptive catch estimates. Quota setting will be done at the Management Board level based on the tradeoff and projection results, goals and objectives, and risk tolerance. This approach advances ecosystem-based fishery management considering both bottom up and top down feedback and managers' goals for multiple species management.

An issue was identified in the precision of the previous iteration of the scheme of an overly coarse grid of F values for striped bass and menhaden. This resulted in the estimated ERP values being different from those coming directly from the model. This has been resolved.

The NWACS-Full model is likely not suitable for development on a tactical management scale due to its complexity and data needs. However, many functional groups that feed on menhaden and or on menhaden predators such as striped bass are not included in the NWACS-MICE model. Exclusion of these groups introduces some uncertainty in ERP estimates. As a research recommendation, when the NWACS-Full model is more fully developed and parameterised using a similar approach as the NWACS-MICE model, comparing results from the two would be useful. For example, comparing the ERP values derived from the two models or the knock on effects on the wider species in the full model would be useful. Clear objectives and using input data sources and model tuning protocol similar to NWACS-MICE for the NWACS-Full model will further facilitate these comparisons.

ToR 5. Evaluate the diagnostic analyses performed as appropriate to each model, including but not limited to:

- ***a. Sensitivity analyses to determine model stability and potential consequences of major model assumptions***
- ***b. Retrospective analysis***

Overall the panel found model diagnostics to be appropriate for each model. Diagnostics in single species models have established methods, while diagnostics in ecosystem models are more diverse, and are evolving as these models are applied in different situations. The panel found the

ERP WG's NWACS-MICE diagnostics to be reasonable and transparent given the limitations of the EwE software. The panel had suggestions for future analyses for both the single species and NWACS models, as detailed below.

For the menhaden assessment model, sensitivity analysis was conducted focusing on different M values. The base model run and lower M run were very similar in model stability and diagnostics, but fishery and survey selectivity curves showed some differences, leading to differences in stock size composition, biomass, recruitment and F estimates. The panel noted that the lower M estimate resulted in more pronounced dome shaped selectivity for the northern reduction fleet and shifted other selectivities towards lower ages, resulting in the impression that there are more fish in the sea that are not being seen by the assessment relative to the base case M. Overall the temporal trends were similar between the runs. As expected, biomass and recruitment are lower with lower M. The menhaden assessment team used the base case M scenario going forward, based on the empirical tag-estimated M combined with evidence from previous tag-recapture model simulation analysis showing low bias in estimated M. Other values of menhaden M or B were not evaluated in ERP models. Research recommendations for assessing the sensitivity of all mass balance parameters conditioned on stock assessments are included in ToR 9.

Most of the work within the ERP models has been focused on creating a base run for NWACS-MICE due to the many parameterization and forcing function options requiring evaluation. The model was fit in phases including initial fitting (include primary production forcing or not, two different starting Kij values), experiment with alternative seasonal forcing, vulnerability estimation, and additional adjustment to improve menhaden dynamics. Sensitivity analysis was limited to the runs created during the tuning, but time between the assessment data becoming available and this review was too short to allow for extensive sensitivity analysis on the selected base run, 150, or the alternative run 153.

Multiple approaches are available to quantify the parameter sensitivity within EwE models. The ERP WG mainly considers the sensitivity to predator-prey vulnerability (Vij) exchange rate multipliers Kij. The review panel endorses this approach (especially given the limited time available prior to this review) because EwE models are well known to be sensitive to changes in the predator-prey vulnerability. The review panel appreciated the transparent documentation and presentation of the number of Kij parameters estimated at bounds for different NWACS runs, which was used as one diagnostic of model performance. Initial analysis of ERP sensitivity to Kij parameters for individual predator prey pairs in the base run 150 was presented, and is a promising initial step towards fuller sensitivity analysis.

Further work should examine sensitivity of the base run(s) to changes in the prey switching parameter, sensitivity to initial biomasses, and of small changes to a few key vulnerabilities (e.g. spiny dogfish on striped bass, striped bass on menhaden). The data weighting scheme could also be subject to sensitivity analysis. Sensitivity on the key parameters coming from the assessments

(M, F, B) is important, as noted above. This could include the alternate lower menhaden M values suggested by life history estimates.

A traditional analytical retrospective analysis is only feasible for the single species stock assessment model. A retrospective analysis was conducted and presented for the menhaden assessment model. There was a tendency to slightly overestimate the stock biomass in the terminal year in recent years, with an associated underestimate of fishing mortalities. However the Mohns rho values associated with this retrospective pattern were generally within acceptable limits. A fuller consideration of the retrospective pattern may be considered by managers using a framework currently in development, but at present the retrospective error adjustment was not done for the projection.

In principle a "historical retrospective" (comparing the historical model outputs from a series of assessments, regardless of methodology) can be examined for the NWACS-MICE model in the future, once such a time series exists.

ToR 6. Evaluate the methods used to characterize uncertainty in estimated parameters. Ensure that the implications of uncertainty in technical conclusions are clearly stated.

The panel found the methods used to characterize uncertainty in estimated parameters to be appropriate, as far as they go. Similar to diagnostics and sensitivity analysis, characterizing uncertainty in estimated parameters for single species models has established methods, while methods for ecosystem models are in development and are in some cases constrained by software limitations. The ERP WG characterized uncertainty in results more than in estimated parameters for the NWACS-MICE model. Ultimately, characterizing uncertainty in ERP results does communicate the implications of uncertainty for technical conclusions.

For the menhaden stock assessment model, uncertainty in M and fecundity were considered using 5000 runs of Monte Carlo bootstrap selecting from a range of both parameters, with each set estimating all other stock parameters within the BAM framework. The range of M was derived from the posterior distribution of the M estimate from the tagging model. The range of fecundity was derived from the bounds around batch frequency and number of spawns per year from Latour et al. 2023. Of the 5000 runs, 4734 (94.7%) converged and met performance standards, so these 4734 were carried forward to characterize uncertainty. The base run is similar to the median values from the bootstrap, and for most runs, the estimates have similar temporal patterns. In the 2020 menhaden assessment benchmark, additional parameters were considered in the uncertainty analysis, but M and fecundity dictated the overall width of uncertainty bounds, so only these parameters were varied here for efficiency.

These uncertainties are then carried through into projections from the menhaden assessment model and compared with ERP target and threshold values derived from the NWACS-MICE model tradeoff analysis. Uncertainty associated with this projection may be under-estimated, given that the only considered uncertainties are in M and fecundity, with recruitment projections using nonlinear time series analysis. Projections are based on a selected Total Allowable Catch (TAC) and include no changes in fishing effort, no seasonality, and no structural model uncertainty.

The panel found methods applied for the menhaden assessment parameter uncertainty to be appropriate and comparable with standard single species assessment projections, as augmented by the nonlinear time series recruitment projection, which reduces uncertainty relative to most assessments drawing from the historic distribution of recruitment values. However, the panel notes that relatively small uncertainty bounds around estimated F are driven by the relatively tight range around M derived from the tagging model posterior distributions. The panel suggests that a broader range of M could be considered in future analyses.

For NWACS-MICE, there was limited time and software capacity to analyze parameter uncertainty once the base case model was identified. Standard methods to address parameter uncertainty in ecosystem models have not been established. Uncertainties across a range of model results were presented, but not in base case model parameters. Methods of identifying the most important parts of the model for the ERP questions represent an important start to this. With an ecosystem model, it seems unlikely that analysts may achieve “95% confidence intervals” style uncertainty estimation typically evaluated in stock assessment models. The sensitivity work suggested above would partly resolve this lack. Producing a suite of “plausible” potential base model variants would give a route to looking at uncertainty (in parameters, in structural form, in hindcast biomasses and diets, and in forecasts). A qualitative or semi-qualitative analysis of what are the most sensitive and uncertain parts of the model is the best that can be achieved for a complex ecosystem model, but this is valuable.

One key parameter uncertainty identified in the development of the NWACS-MICE base case surrounds the spiny dogfish-striped bass predator prey interaction. In the base case model (150), a single vulnerability multiplier (spiny dogfish as predators and medium striped bass as prey) required manual adjustment away from the lower bound to ensure that striped bass would be at their biomass target when fished at their target F rate. In an alternative run (153), spiny dogfish input parameters for biomass and diet composition were reduced relative to run 150 to represent the possibility that the spiny dogfish stock spends less time within the model domain. Higher vulnerability of medium striped bass to dogfish was estimated in this alternative run, resulting in lower but still feasible striped bass productivity relative to the base case. Comparing ERP tradeoffs for these runs illustrates the implications of uncertainty in technical conclusions, but there are likely other feasible runs that have not yet been explored.

The implications of these uncertainties in the ecosystem model need to be tracked through to the likely uncertainty around the ERPs and hence their impacts on management. Uncertainties in input mass balance parameters can be assessed by using a data pedigree that is translated into a distribution for each input parameter (with higher quality data having tighter distributions around base parameters and lower quality data having wider distributions around base parameters).

These distributions are then used to resample base model parameters and generate a plausible range of base models (e.g. Whitehouse and Aydin 2020). The difficulty is that each of these plausible models incorporating initial parameter uncertainty (around the 1985 system state) would also need to be fitted to the time series data to estimate vulnerabilities, and then projected under different Fs to produce a range of ERPs, and there are no streamlined ways to do this in Ecosim. Both NWACS models have a data pedigree, so the basis for a broader uncertainty analysis exists, but software limitations represent a substantial impediment. The review panel makes some suggestions under ToR 9 for steps towards full uncertainty analysis, including a hybrid approach where a full base case model could be developed in EwE software, then key uncertainties could be evaluated for impacts on parameter estimates in a more flexible multispecies estimation model framework.

ToR 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.

No minority report was presented.

ToR 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.

This review did not explicitly analyze the single species menhaden model. The review panel endorses the ERP WG's recommendation to use the benchmarked single species assessment to estimate stock biomass, abundance and exploitation rates. The stock status should be evaluated by comparing the single species assessment to the ERP arising from the NWACS-MICE model and methodology reviewed here (TOR 4).

We do not recommend the specific level, rather the methodology here represents the appropriate tool for the managers to use to choose between a range of different potential fishing levels based on their goals for the striped bass and menhaden fisheries. The panel notes that the exact level needs to be re-examined following any adjustments to the ERP models.

ToR 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.

The review panel finds that the research recommendations proposed by the ERP WG for future development were reasonable, including those that have been carried over from the previous ERP benchmark. The division of short term and long term data and modeling needs in the research recommendations is extremely useful. It should be noted that the list may be longer than can be achieved, and therefore prioritizing is important. Here we suggest some priorities.

The review panel strongly supports continued and expanded seasonal, spatial, and taxonomic collection of population abundance, biomass, life history, condition, and diet data for both managed fish species, including menhaden, and functional groups throughout the ecosystem from primary producers through zooplankton and benthos to apex fish, bird, and marine mammal predators. Existing fishery independent surveys and fishery dependent data collection must be maintained both for stock assessment and for ecosystem assessment. In addition to stock assessment data, low trophic level productivity and diet data are essential to ERP model function. Improved data on prey size or age from diet collections is recommended to apportion diet across age stanzas of prey. Investigation of other existing datasets such as the Hudson River Biological Monitoring Program (HRBMP) data (1974 to 2018 for menhaden and striped bass) is also recommended.

The review panel strongly supports holding a workshop with managers and stakeholders similar to the 2015 EMO workshop to determine clear objectives for a potential spatial ERP assessment prior to any spatial model development. This workshop should also review data requirements for assessment at different spatial scales alongside existing data to ensure that expectations for spatial assessment are realistic and achievable. In particular it is important to be clear on the degree of spatial resolution possible within a proposed model, and evaluate if this would meet the wishes of the stakeholders and managers. Even if expansion of data collection as recommended above is feasible, historical data could limit the effectiveness of spatial models. In addition to considering multiple existing spatial modeling frameworks for their ability to integrate existing data and achieve management objectives, the workshop should also consider whether the objectives could be achieved with simpler non-spatial models at different scales, e.g., a model for Chesapeake Bay.

The review panel strongly supports the recommendation of the scientific group regarding assessment coordination and planning for an appropriate amount of time between the single species assessment data becoming available and the deadline for the ERP work, both for management use and especially for future benchmarks. Maintaining the current ERP assessment method requires updating and revising the NWACS model as new data and assessments become available (especially the revised MRIP recreational data and the forthcoming striped bass benchmark). However, continuous updates to the NWACS-MICE would be an inefficient use of limited modeling resources. The panel recommends conducting research on general methods for evaluating parameter uncertainty and sensitivity of ecosystem models in between benchmarks, which should be scheduled well after updated assessments for ERP focal species are completed (see ToR 10).

The review panel highlights the following recommendation for future ERP benchmarks. Once a base case model is developed, the ERP WG should focus on creating a suite of different model formulations which give "plausible" model outputs. These could be generated by making changes to the selected base run. Such a suite of models could then be used to characterize and compare the key uncertainties in the model. Once there is such a suite of models, one possibility would be to consider ensemble model averaging in future management advice. However, the suite would have clear utility in investigating uncertainty regardless of any formal ensemble modelling.

The review panel also suggests these specific research recommendations:

Continue to code and fully document model input data processing, balancing, and fitting protocols along with acceptable ranges for model diagnostics and structured sensitivity runs with the goal of fully repeatable and transparent methods that can be handed off from research teams to agency staff charged with operational assessment.

Evaluate NWACS-MICE sensitivity to initial biomass and productivity – how robust are results to revised assessment total biomass estimates and changing assumptions regarding single species M? Existing input data uncertainty methods such as Ecosampler or the R-based ecosense could be investigated for this process. A hybrid approach investigating the subset of most sensitive parameters could be implemented in alternative software such as Rpath (Lucey et al 2020) or EcoState (Thorson et al 2024) for more efficient sensitivity analysis of static food web and dynamic parameters, respectively.

Investigate NWACS-MICE model sensitivity to prey switching parameters in the absence of data, and or evaluate food habits data along with ecosystem sampling data to detect the extent to which prey switching has been observed.

Evaluate NWACS-MICE behavior with long term egg forcing using recruitment deviations for species in addition to Atlantic herring and determine whether this allows estimation of predator prey parameters away from bounds.

Develop clear objectives for NWACS-Full. If the models are to be used together for future management advice, build out NWACS-Full directly from NWACS-MICE and fit to the same time series in both models to facilitate direct comparisons between results. This may necessitate staggered model development to ensure that the final NWACS-MICE base model informs the NWACS-Full model prior to NWACS-Full fitting, but would have the additional benefit of simplifying the complex task of tuning the NWACS-Full model. The model selection process and tuning protocol should be well documented to ensure transparency and consistency

Collect age data from the menhaden fishery independent surveys to inform M (e.g., are ages up to 10 actually observed? What are the current oldest ages?).

Finally, the review panel supports the development of an MSE to identify harvest strategies maximizing the likelihood of achieving the defined ecosystem management objectives. The MSE would be most effective if it is based on coordinated management objectives between menhaden and their predator species across management Boards, which require development.

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

The review panel endorses the asynchronous benchmarks for the menhaden stock assessment and ERP models suggested by the ERP WG. Coordination with ERP species assessment timelines is also required to ensure inputs to ERP models are both reviewed within each single species assessment process and consistent with current assessments.

The review panel agrees that several upcoming events will be important to incorporate within the next ERP benchmark, including a recreational data (MRIP) recalibration scheduled for 2026 and a striped bass benchmark assessment scheduled for 2027. The MRIP recalibration is likely to affect several ERP species assessments, including striped bass, bluefish, and weakfish, so scheduling the ERP benchmark after these assessments in complete is recommended.

The most important recommendation from the review panel is to allow sufficient time between the assessment data and results becoming available and the ERP benchmark to allow for both development and analysis of the NWACS-MICE and NWACS-FULL models. The current ERP benchmark timing allowed for further necessary model development of the NWACS models, but did not allow for full exploration of sensitivities in the selected base NWACS models, nor did it allow for full development and testing of bottom up species interactions in the VADER model. Given the substantial progress made on reproducible data processing methods, base model development and selection for NWACS-MICE, it may be possible to build on the lessons learned during the current benchmark to streamline development for the next benchmark. However, the review panel recognizes that new input data from new assessments means the calibration and selection process still needs to start at the first step each time.

Therefore, the review panel recommends that the ERP benchmark be scheduled a minimum of one year after the ERP single species assessments are finalized and results are available for the ERP WG.

ToR 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.

The review panel discussed tasks to be completed following the workshop, which were minor and were not expected to result in changes to the base model results:

Implement seasonal egg production forcing in NWACS-MICE as long term forcing with a seasonal signal to ensure model behaviour is as expected. The seasonal egg forcing function in Ecosim lacked documentation and experiments with alternative scales of forcing gave unexpected results. However when combined with long term recruitment deviation forcing as for Atlantic herring, changes in seasonal forcing worked as expected. All other species should therefore have seasonal egg production forcing implemented as long term forcing with a seasonal signal (but no long term recruitment drivers at this time).

Clarify interpolation method used for the ERPs from the tradeoff projections in the documentation. Interpolation between tradeoff runs was done for two purposes: first to create a visual image, and second to select the menhaden F multipliers corresponding to striped bass biomass at striped bass F target and F threshold multipliers. Two interpolations were shown, one increasing the resolution for both striped bass F multipliers and menhaden F multipliers from 10 to 100 intervals between model run outputs, and one based on a run at the striped bass F multiplier equivalent to F target (no interpolation necessary) that increased menhaden resolution from 10 to 100 intervals between tradeoff run outputs.

The review panel will complete all other tasks as outlined in the schedule provided, including submitting this report.

Summary Results of Analytical Requests

During the review, the panel requested additional visualizations and data comparisons, as well as clarifications on implementation of some EwE model forcing functions. The ERP WG completed these requests during the workshop:

Comparisons of NWACS-MICE model inputs from the 2019 model and the currently reviewed model were provided, and additional investigation of differences for striped bass inputs between models was completed. In addition to the expected differences between previous and current striped bass stock assessment results, and the selection of "leading" stanza for calculation of ecopath inputs, there were also differences in which age classes were assigned to NWACS-MICE age stanzas between 2019 and the current model. This assignment was carried through all analyses so the current model behavior is internally consistent despite the age group labels reflecting the 2019 configuration.

Seasonal egg forcing functionality in Ecosim was investigated to evaluate whether applying different monthly scalars resulted in expected seasonal appearance of new recruits into age-0 groups. Because some unexpected results were found with forcing entered seasonally, the forcing function was changed to a long term seasonal function to ensure consistent results (see ToR 11).

All NWACS-MICE model inputs, pedigree, and parameter settings for the base run were provided as csv files. Settings for prey switching parameters were clarified in the base model.

Additional interpolation scenarios were evaluated to clarify the effects of finer resolution for both striped bass and menhaden F multipliers in selecting the menhaden F multiplier corresponding to the striped bass target biomass and target F multiplier.

All NWACS-FULL model inputs, pedigree, and parameter settings for the base run were provided as csv files.

BAM output recruitment and fecundity were provided as a csv file. Visualizations of recruitment vs fecundity and recruitment/fecundity over time were provided with different years highlighted. Pairwise comparisons of unlagged NAD, MAD, and SAD indices were plotted.

Mohn's rho values were provided for menhaden fecundity, recruitment, full F and geometric mean 2-4 F from BAM, and a supplemental flowchart of potential use of this retrospective information in the management process was provided.

SEDAR

Southeast Data, Assessment, and Review

SEDAR 102

ASMFC Atlantic Menhaden and Ecological Reference Points

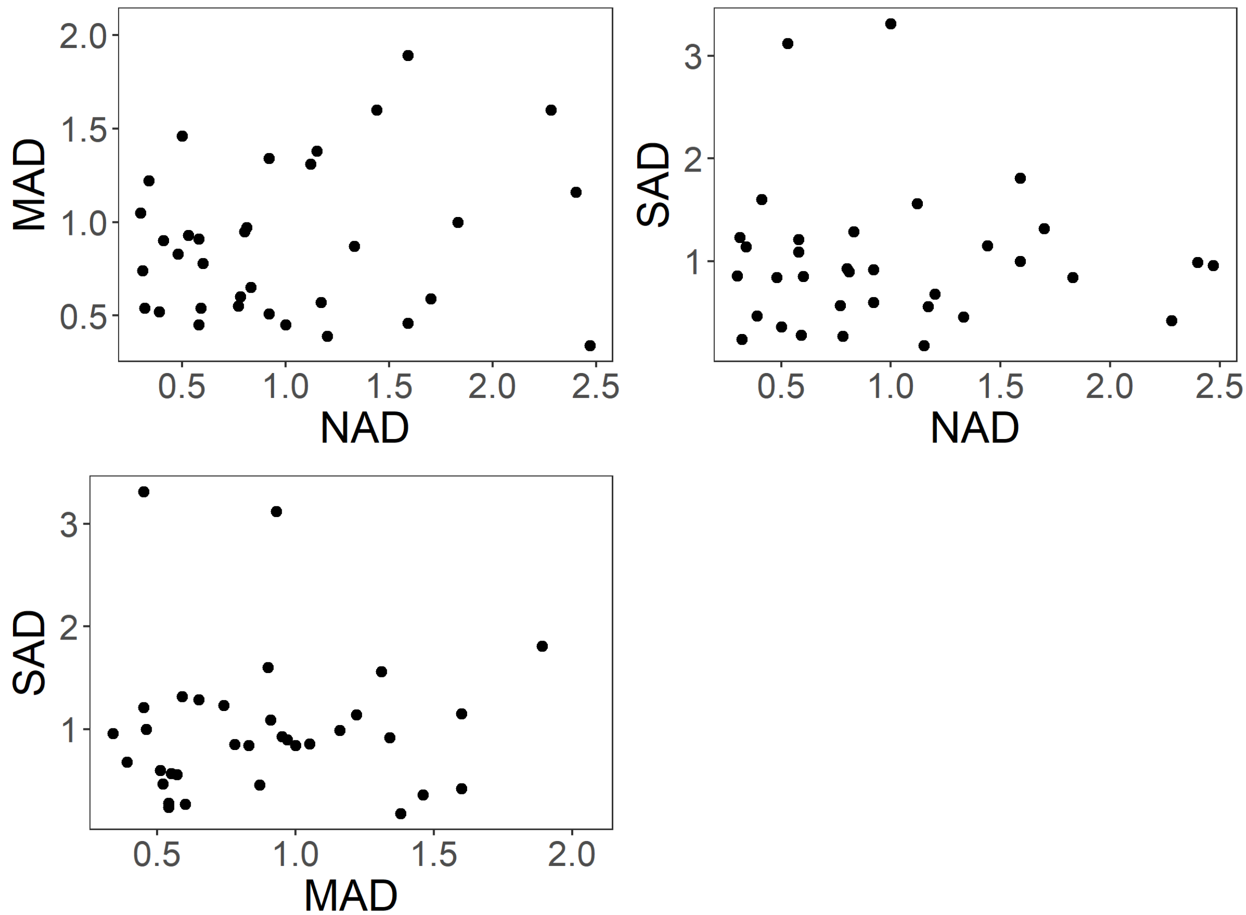
SECTION IV: Post-Review Workshop Addenda

September 2025

SEDAR
4055 Faber Place Drive, Suite 201
North Charleston, SC 29405

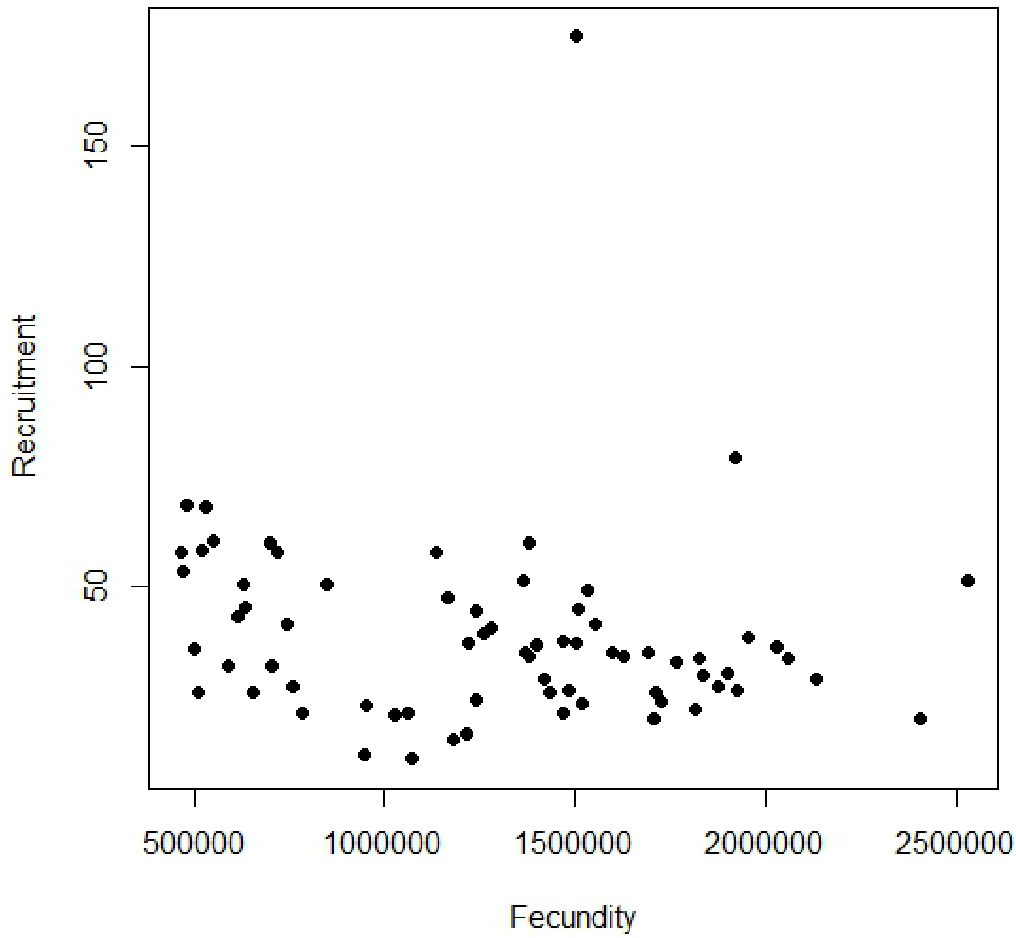
SEDAR 102 Reviewer Requests

Pairwise plot of the 3 adult indices for Atlantic menhaden, not lagged

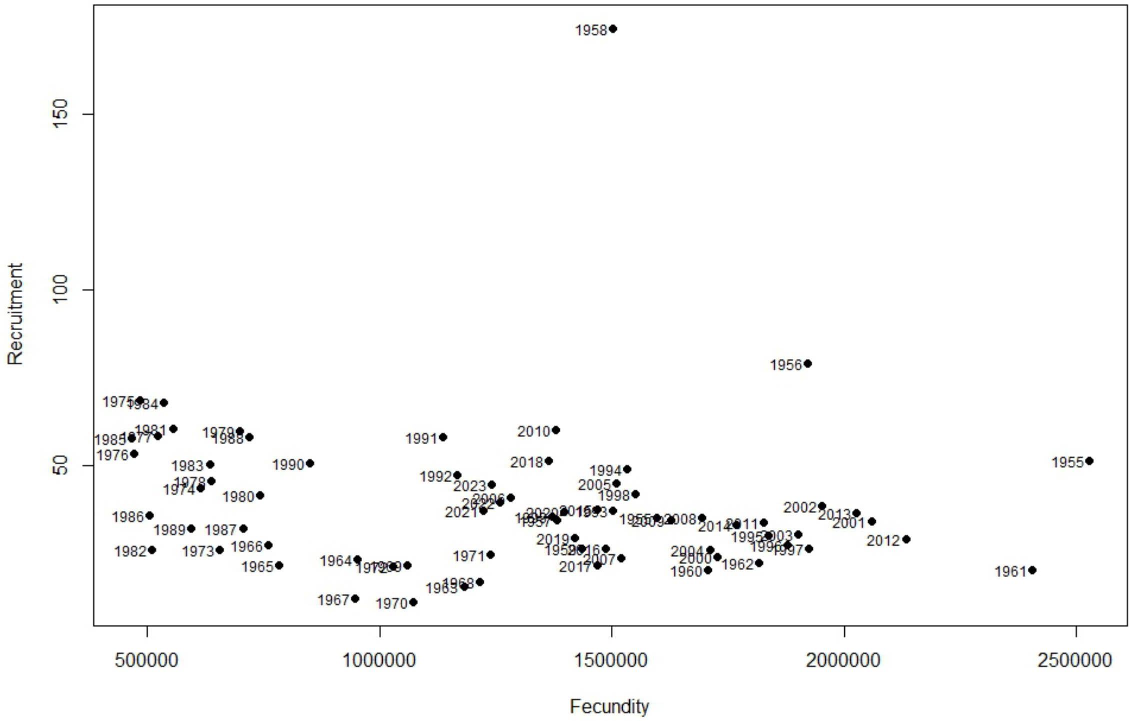


The WG noted that the length composition of each index is different, as each index captures a different subset of ages. The Atlantic Menhaden Stock Assessment Subcommittee (SAS) had explored correlations between indices in the 2020 benchmark, and the correlation between the NAD, the MAD, and the SAD were stronger when the indices were lagged to account for differences in age/length composition.

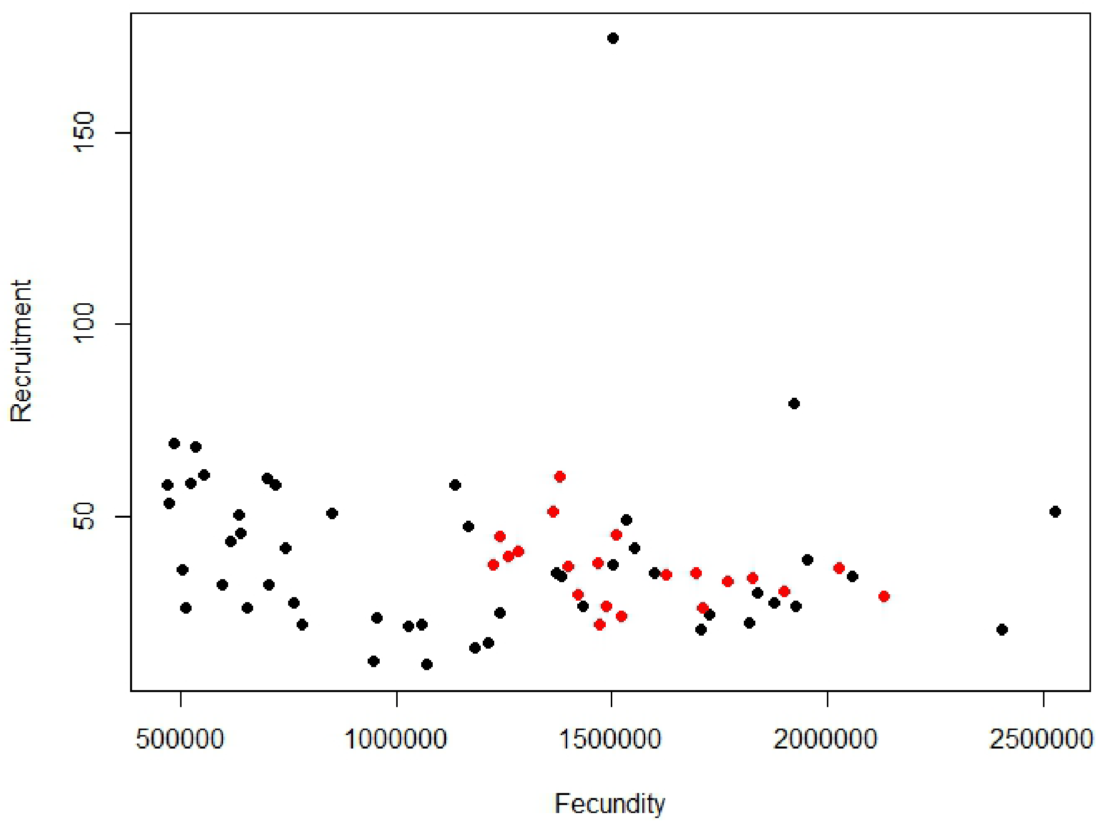
Plot of recruitment vs. fecundity



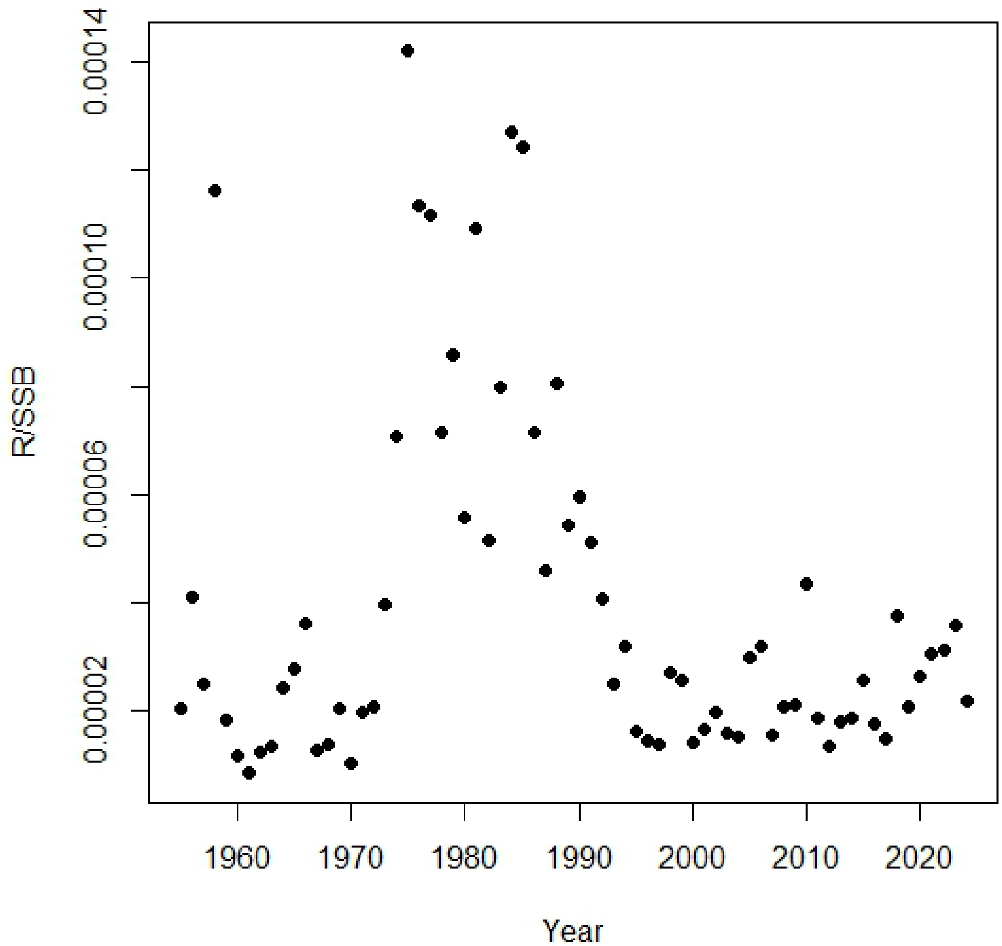
With years labeled



With last 20 years highlighted (in red)



Plot of recruitment divided by fecundity by year



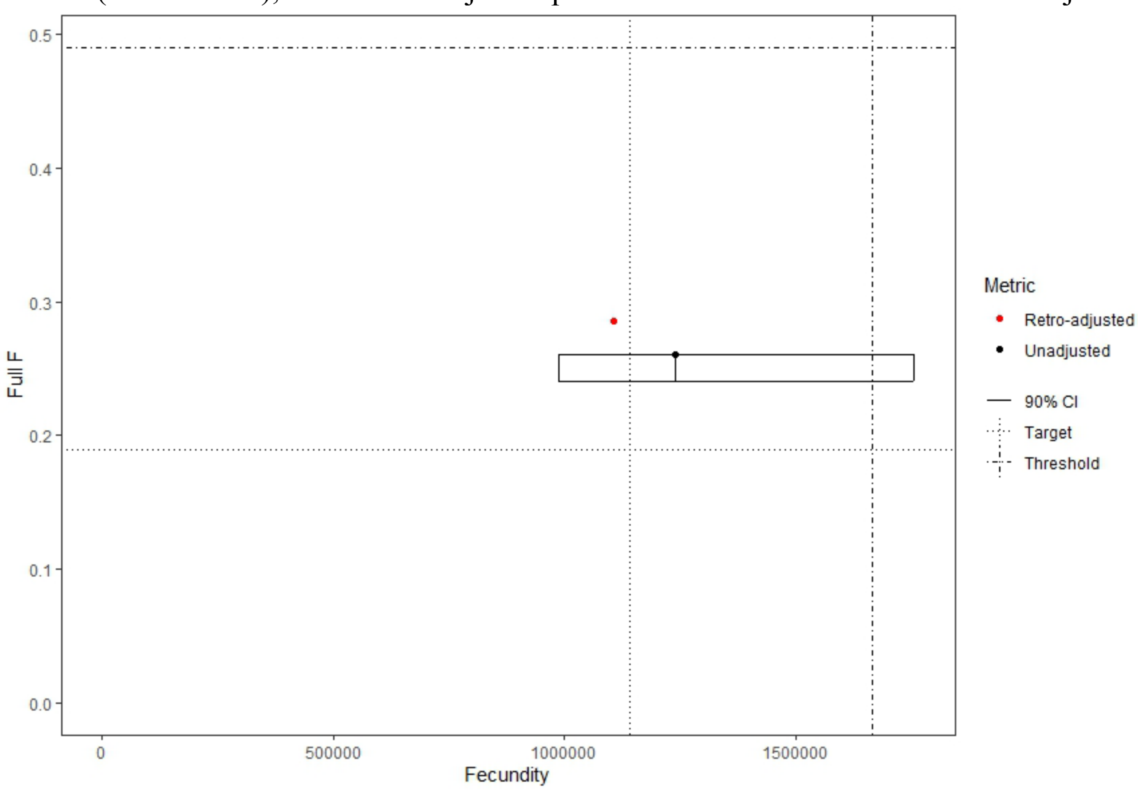
Mohn's rho values

- Age-1+ biomass: 0.16
- Fecundity: 0.12
- Recruitment: -0.16
- Full F: -0.09
- Geometric mean F: -0.12

Explanation of ASMFC's guidance for retrospective adjustments
ASMFC uses a flow chart to look at several retrospective diagnostics to evaluate whether to apply a retrospective adjustment or not. The WG walked through first two decision points of the flow chart with the Panel but noted that subsequent questions need to be discussed with the Menhaden Technical Committee (TC). This was not completed as part of the initial assessment report for the Review as some questions deal with stock status which may change based on the outcome of SEDAR 102. The flow chart will be revisited with the TC once the peer review is completed.

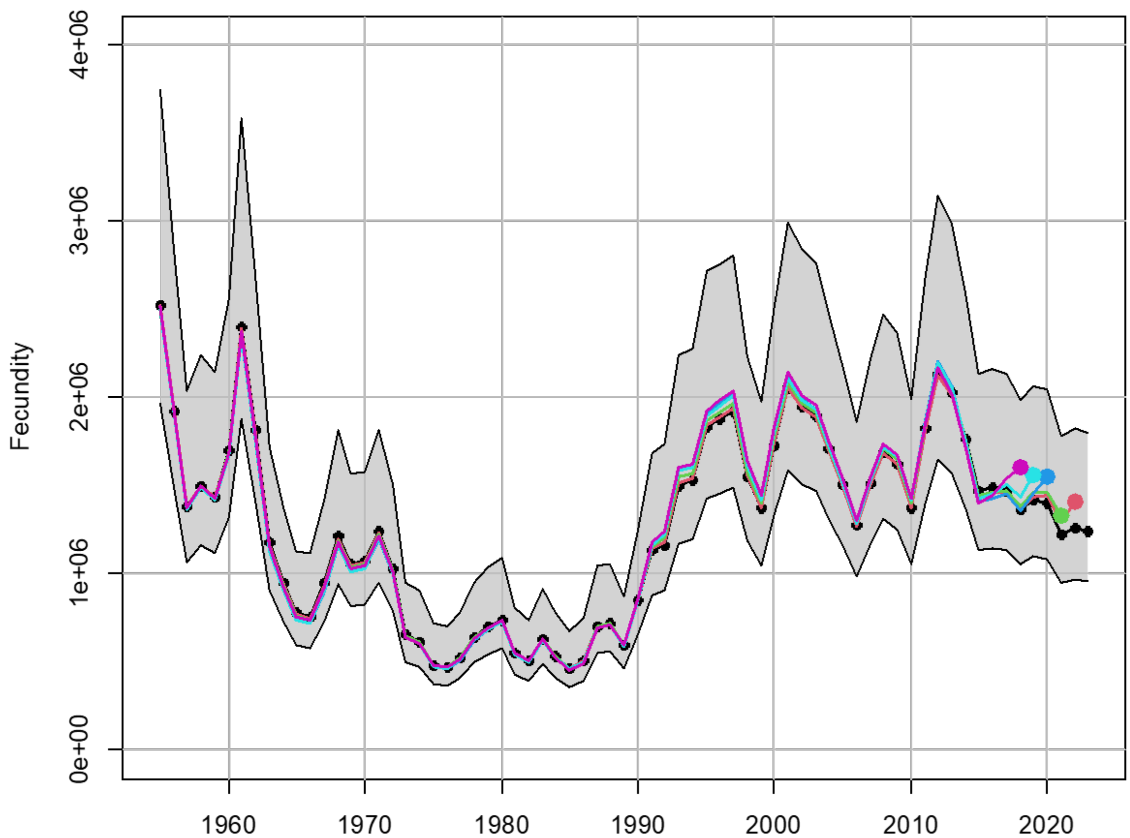
Q1. Is Mohn's rho outside the bounds recommended for a short-lived species and is the rho-adjusted value of fecundity and F outside the 90% confidence intervals of the unadjusted values?

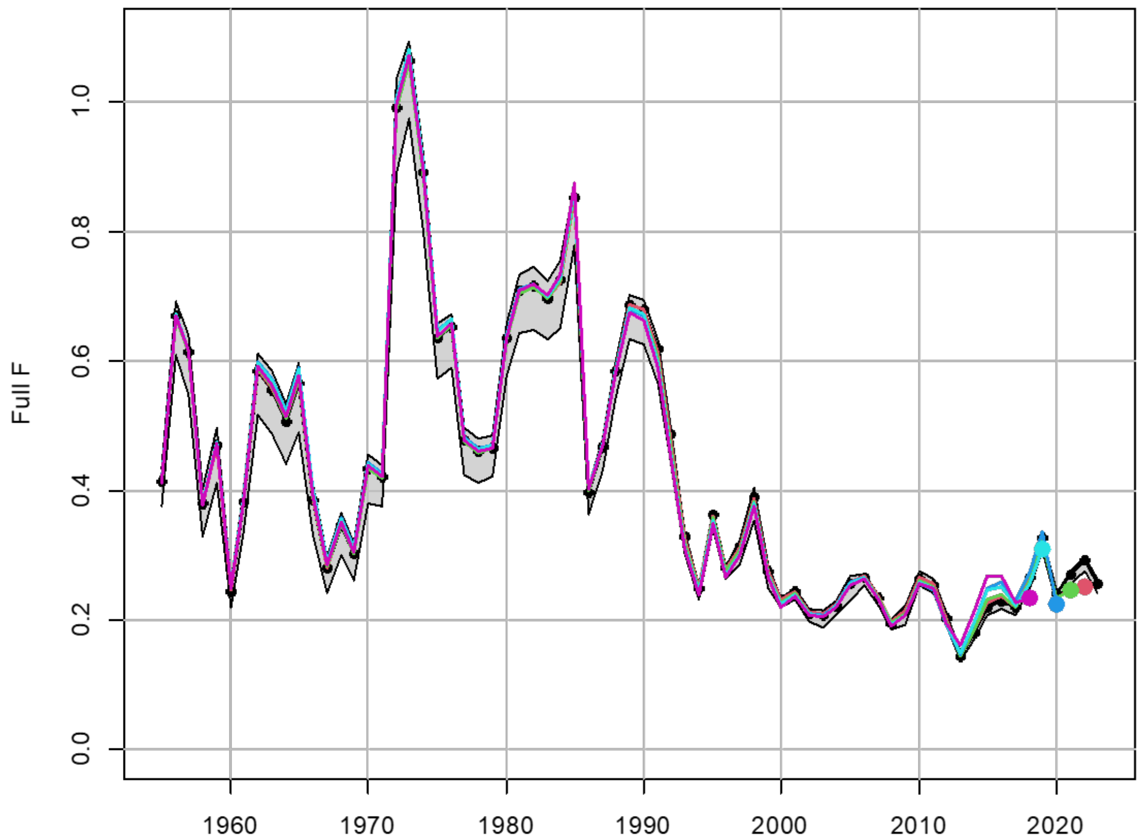
A1. Yes, one of them is: Mohn's rho for fecundity and F are not outside the recommended bounds (-0.22 – 0.30), but the rho-adjusted point is outside the 90% CIs of the unadjusted values.



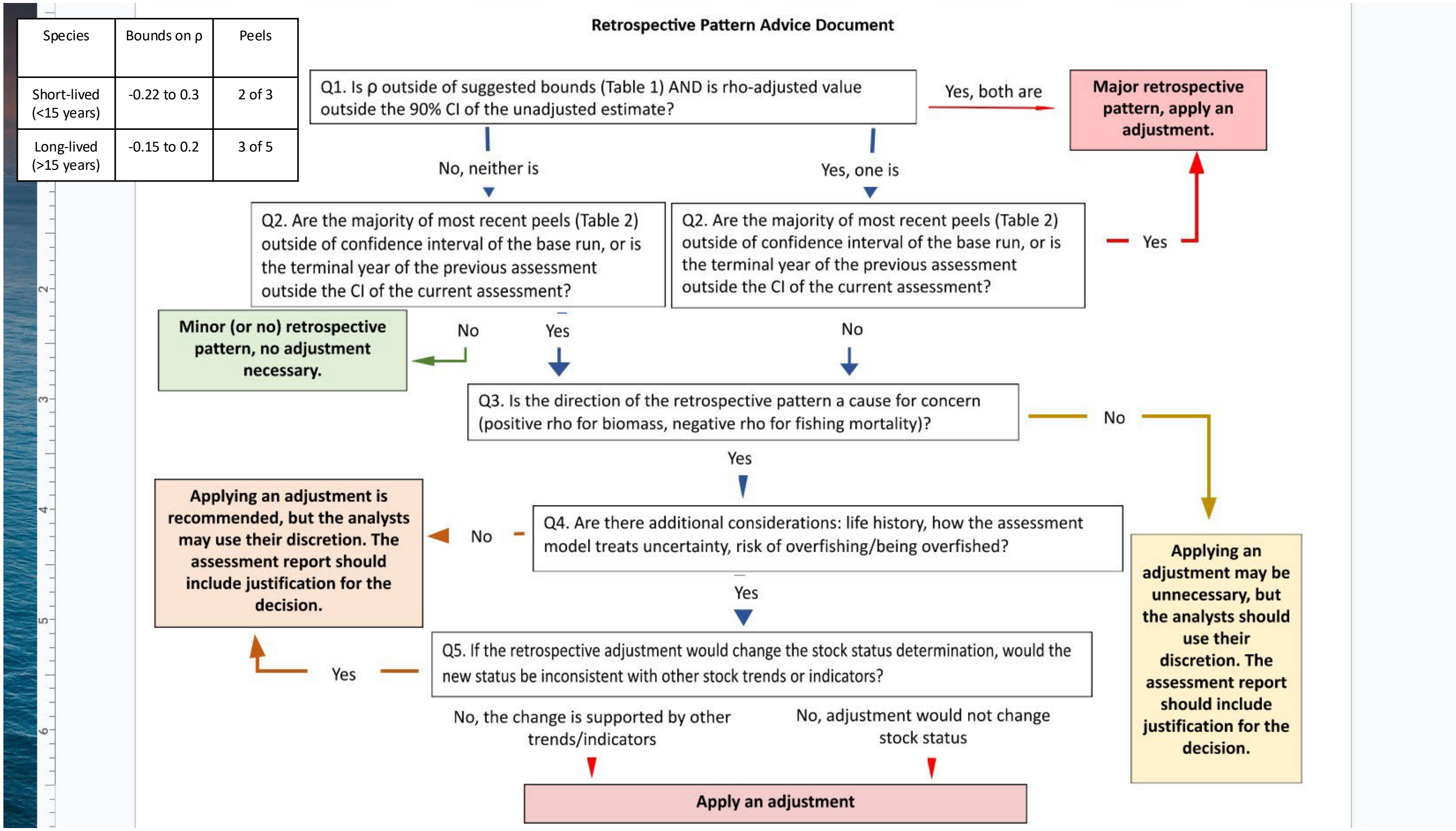
Q2: Are 2 of the 3 most recent peels outside the confidence intervals of the base run, or is the terminal year of the previous assessment outside the confidence intervals of the current assessment?

None of the fecundity peels were outside the CIs of the base run, but 2 of the 3 peels for F were, due to how narrow the CIs were for F. Because of the scale change from the 2022 update to the 2025 update, due to the new estimate of M, the terminal year of the previous assessment can't be compared directly with the CIs of the current assessment, so this diagnostic was not shown.



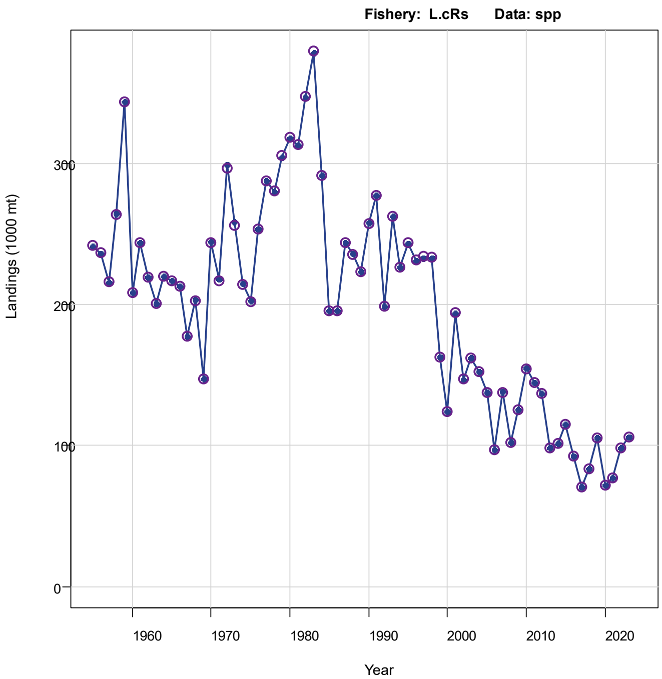


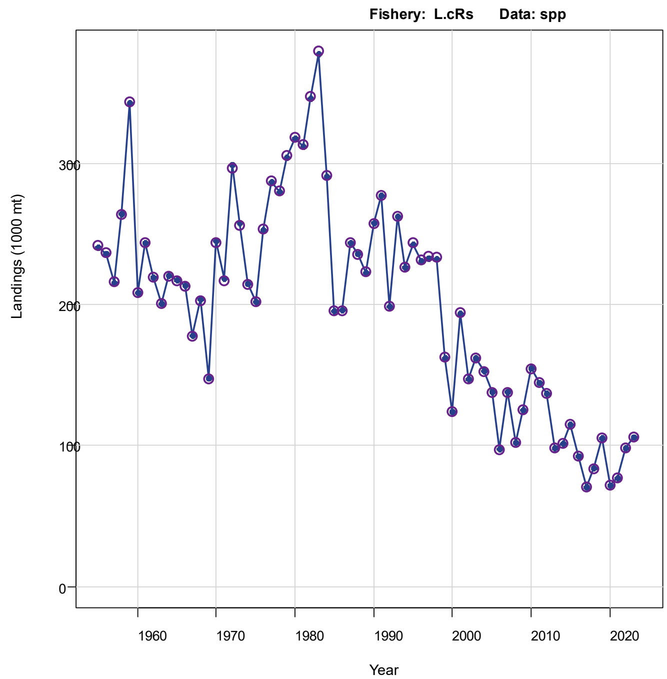
The WG noted that the flow chart does not provide guidance on what to do when the majority of peels for one of the metrics but not the other is outside the CIs, and suggested ASMFC may want to clarify this point. The retrospective guidance document was adopted by ASMFC in January 2024 and opportunities to test it in practice have been somewhat limited.


Species
Bounds on ρ
Peels
Short-lived (<15 years)
-0.22 to 0.3
2 of 3
Long-lived (>15 years)
-0.15 to 0.2
3 of 5
Retrospective Pattern Advice Document
Q1. Is ρ outside of suggested bounds (Table 1) AND is rho-adjusted value outside the 90% CI of the unadjusted estimate?
Yes, both are
Major retrospective pattern, apply an adjustment.
No, neither is
Yes, one is
Q2. Are the majority of most recent peels (Table 2) outside of confidence interval of the base run, or is the terminal year of the previous assessment outside the CI of the current assessment?
Q2. Are the majority of most recent peels (Table 2) outside of confidence interval of the base run, or is the terminal year of the previous assessment outside the CI of the current assessment?
Yes
Minor (or no) retrospective pattern, no adjustment necessary.
No
Yes
No
Q3. Is the direction of the retrospective pattern a cause for concern (positive rho for biomass, negative rho for fishing mortality)?
No
Yes
Applying an adjustment is recommended, but the analysts may use their discretion. The assessment report should include justification for the decision.
No
Q4. Are there additional considerations: life history, how the assessment model treats uncertainty, risk of overfishing/being overfished?
Applying an adjustment may be unnecessary, but the analysts should use their discretion. The assessment report should include justification for the decision.
Yes
Yes
Q5. If the retrospective adjustment would change the stock status determination, would the new status be inconsistent with other stock trends or indicators?
No, the change is supported by other trends/indicators
No, adjustment would not change stock status
Apply an adjustment

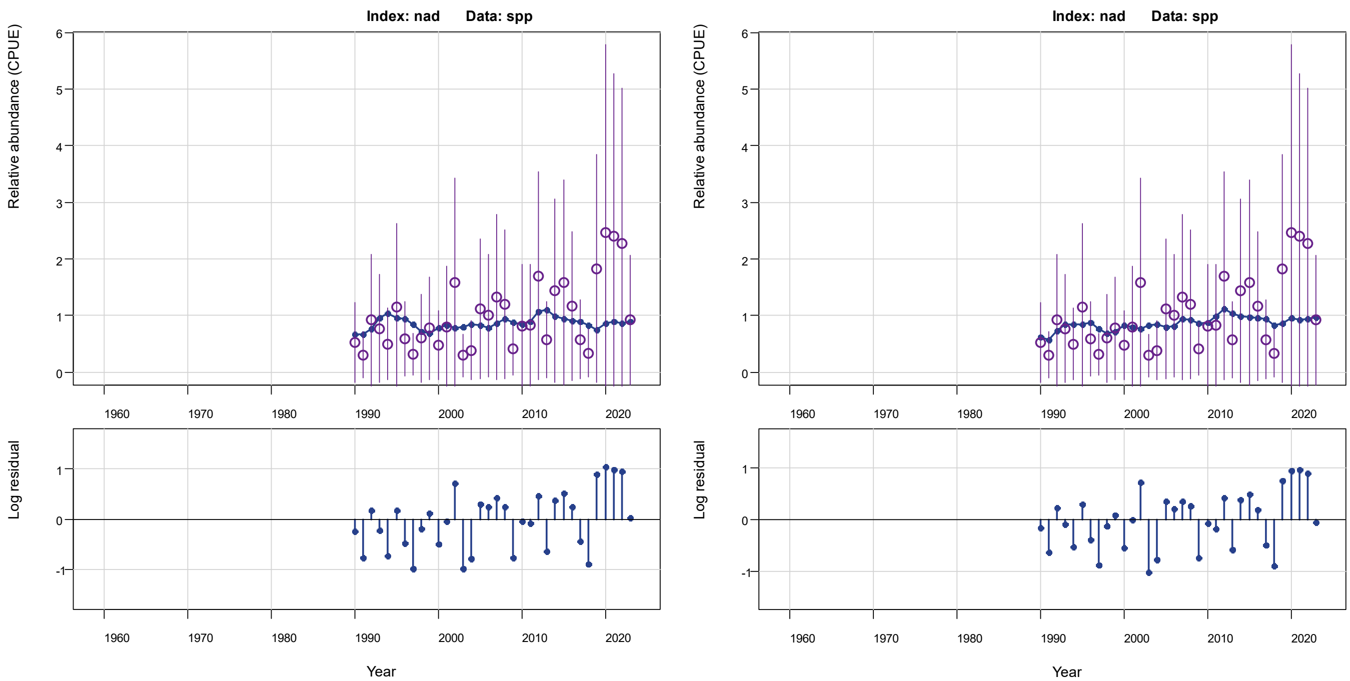
Additional BAM diagnostic plots for the base run and the sensitivity run with lower M

Fits to landings and indices were similar when comparing the base run and the sensitivity run with the lower value of M.

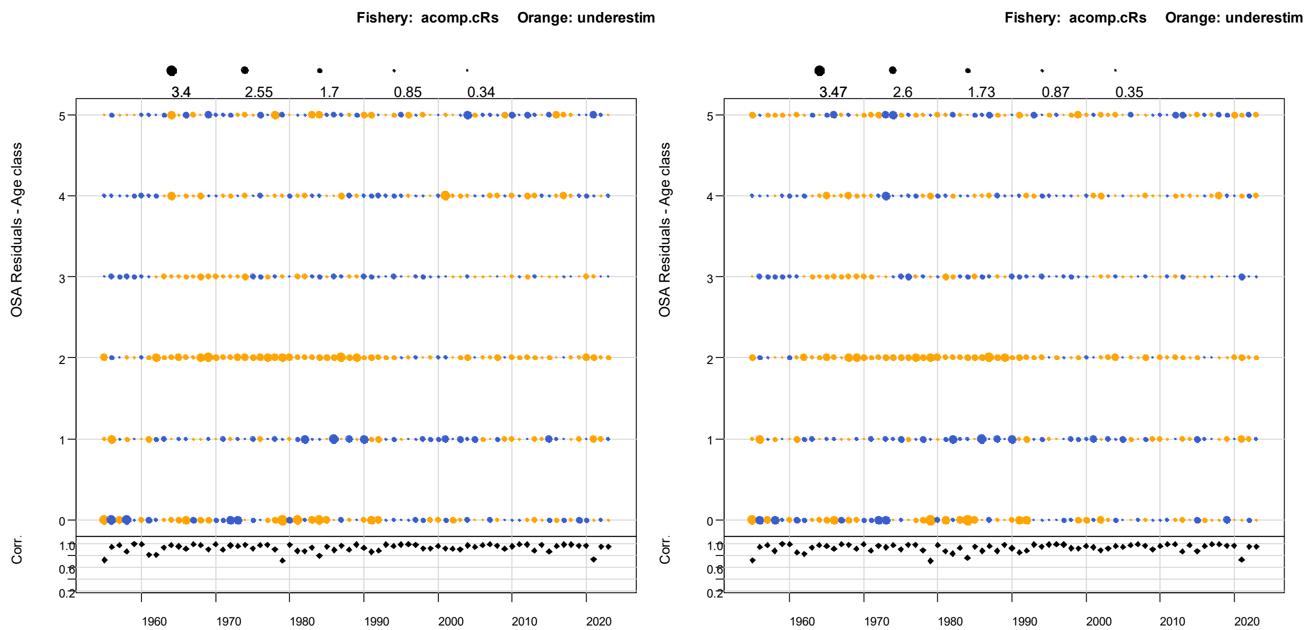




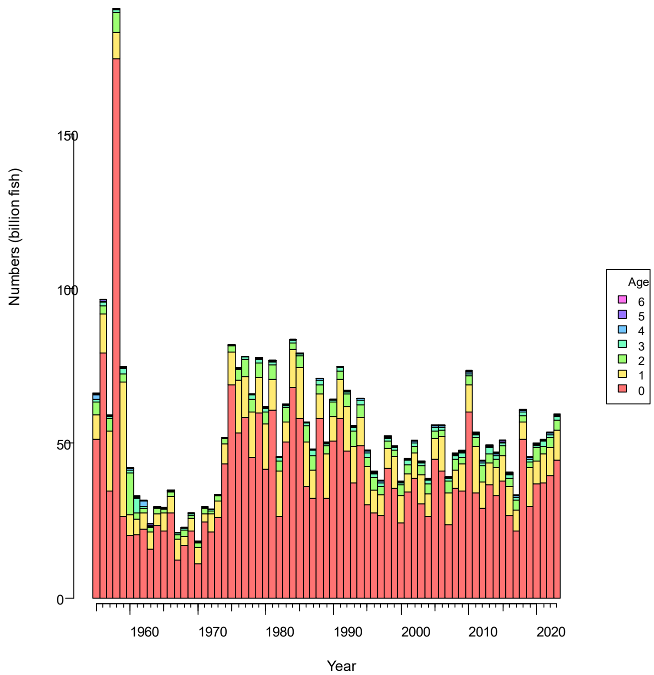
Above is an example plot of the fit to the commercial reduction landings for the southern area for the base run (left) and the sensitivity run with a lower M (right). The additional fits to the fleet landings looked as similar as this comparison.



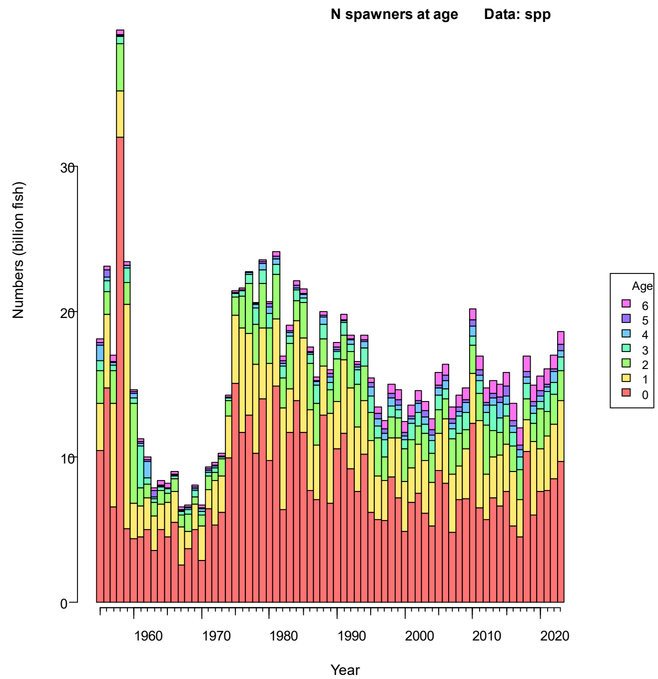
Above is an example plot of the fit to the northern adult index (NAD) for the base run (left) and the sensitivity run with a lower M (right). The additional fits to the indices looked as similar as this comparison.



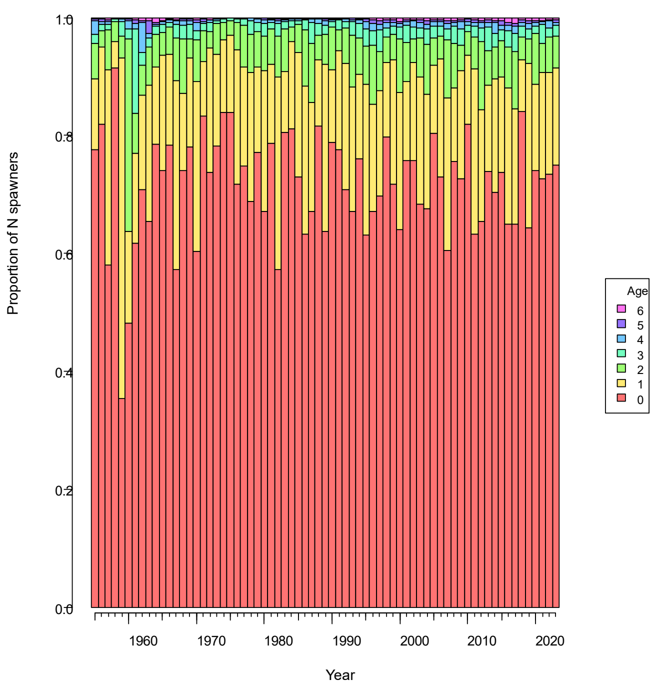
Above is an example plot of the fits to the commercial reduction age compositions for the southern area for the base run (left) and the sensitivity run with a lower M (right). The additional fits to the age and length compositions looked as similar as this comparison.



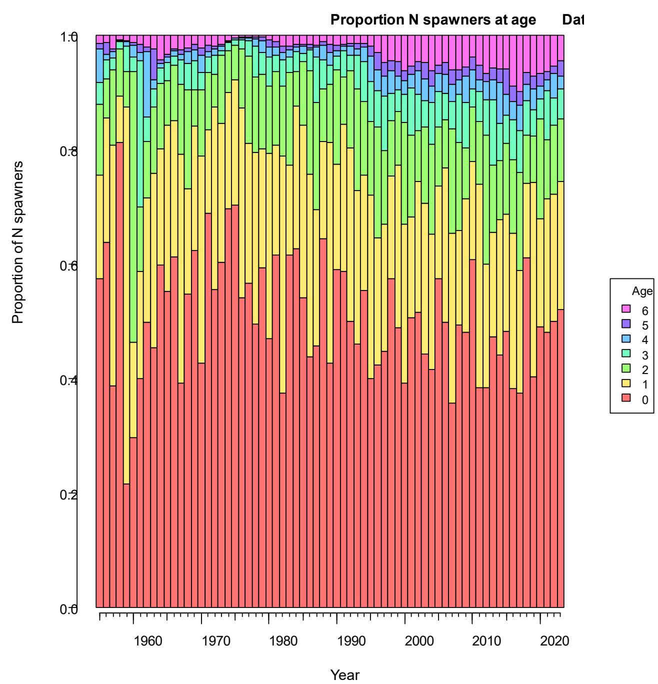
Numbers (billion fish)
Year
Age
6
5
4
3
2
1
0



N spawners at age Data: spp
Numbers (billion fish)
Year
Age
6
5
4
3
2
1
0



Proportion of N spawners
Year
Age
6
5
4
3
2
1
0



Proportion N spawners at age Dat
Proportion of N spawners
Year
Age
6
5
4
3
2
1
0

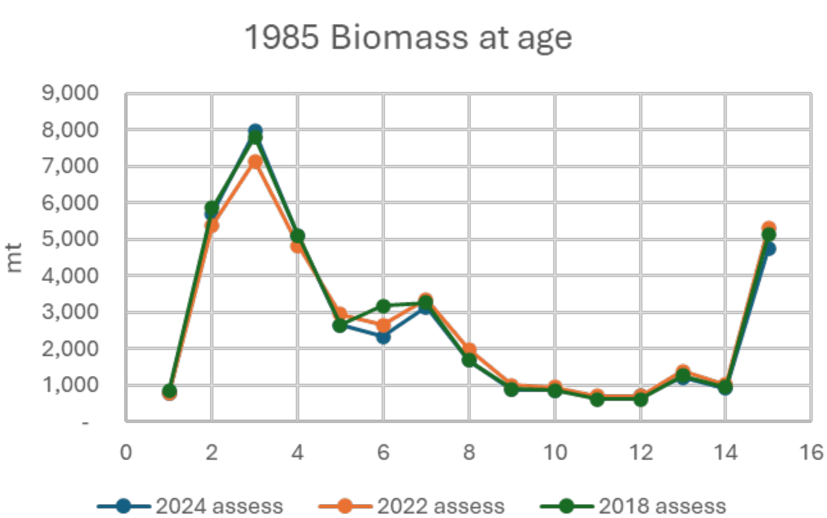
Above are the numbers at age and the proportion of numbers at age for the base run (left) and the sensitivity run with the lower value of M (right). These plots demonstrate the main difference between the two runs, which is the expected numbers and proportion of older aged individuals in the population.

Investigate change in biomass distribution across age stanzas of striped bass in Ecopath from 2020 ERP assessment to 2025 ERP assessment
Striped bass 1985 biomass at age did not change much between 2018 and 2024 stock assessments.

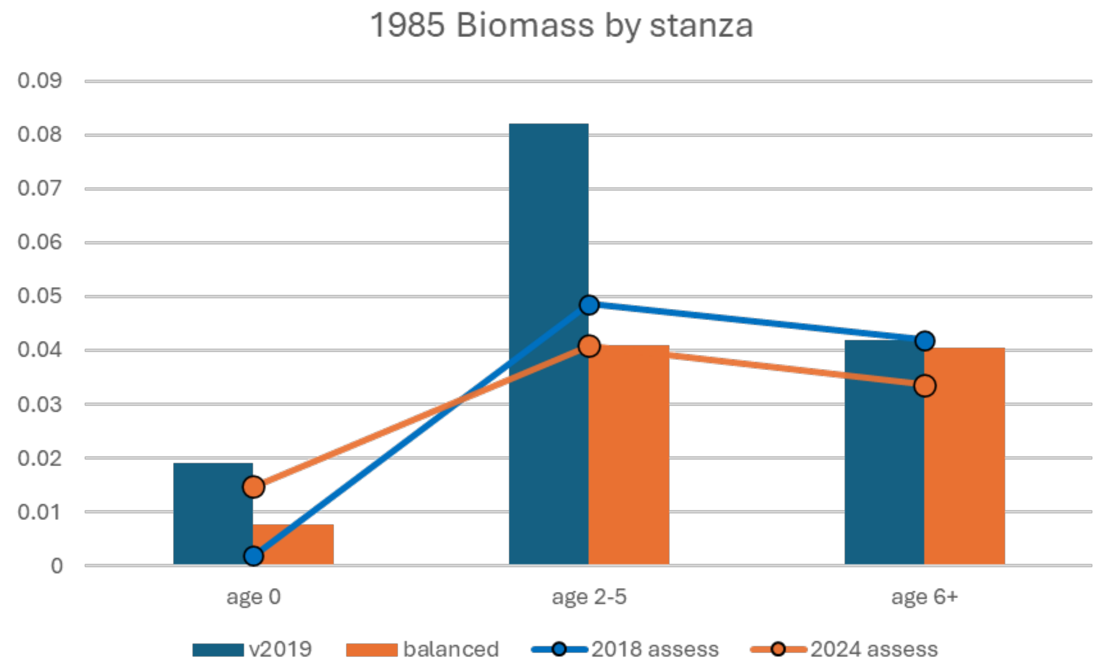
The differences are due to inconsistencies in how ages were assigned to stanzas
- ➔ v2019: age 1, 2-5, and 6+
- ➔ Current: age 1-2, 3-6, 7+

The current approach with age 3-6 as leading stanza matches the 1985 age distribution better than v2019.

This change carries through to all other data processing steps, making it more of a labeling mix-up. The NWACS-FULL model processed the input data the same way that the NWACS-MICE model did, so the stanza definitions are consistent across both NWACS models.



	1985 biomass	
age	2018 asses	2024 asses
1	846	770
2	5,865	5,702
3	7,807	7,969
4	5,104	5,088
5	2,655	2,645
6	3,181	2,336
7	3,260	3,151
8	1,692	1,693
9	898	872
10	858	871
11	621	691
12	611	711
13	1,275	1,207
14	956	915
15	5,134	4,734



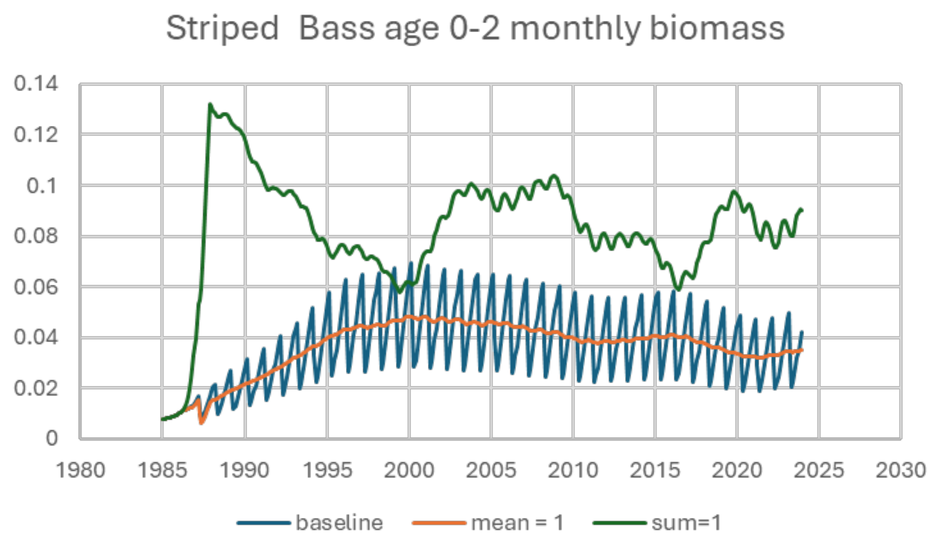
Investigate/document how egg production works in Ecosim
For multistanza groups, numbers at monthly ages $N_{a,t}$ and body weights $w_{a,t}$ are updated at each monthly timestep.

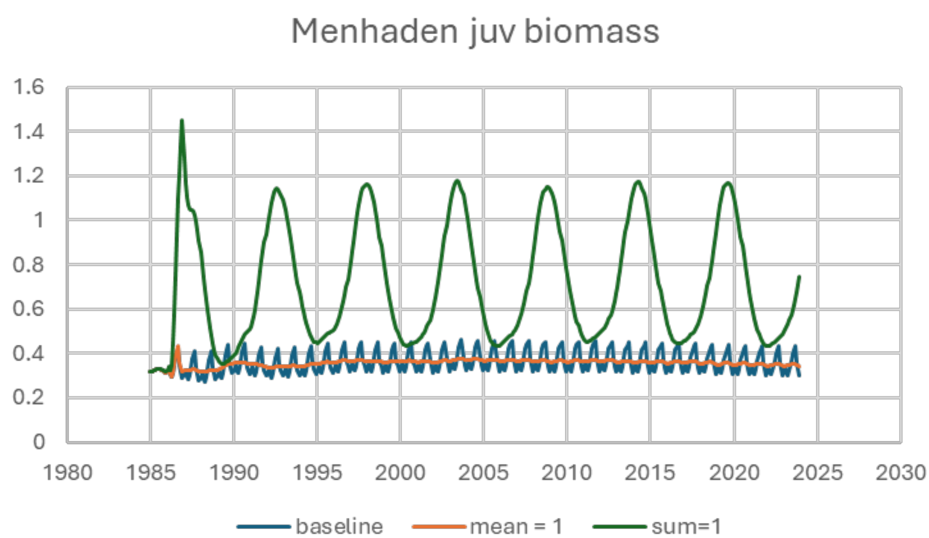
Initial numbers entering the first stanza each month are proportional to total egg production, and egg production is proportional to body weight minus a weight at maturity.

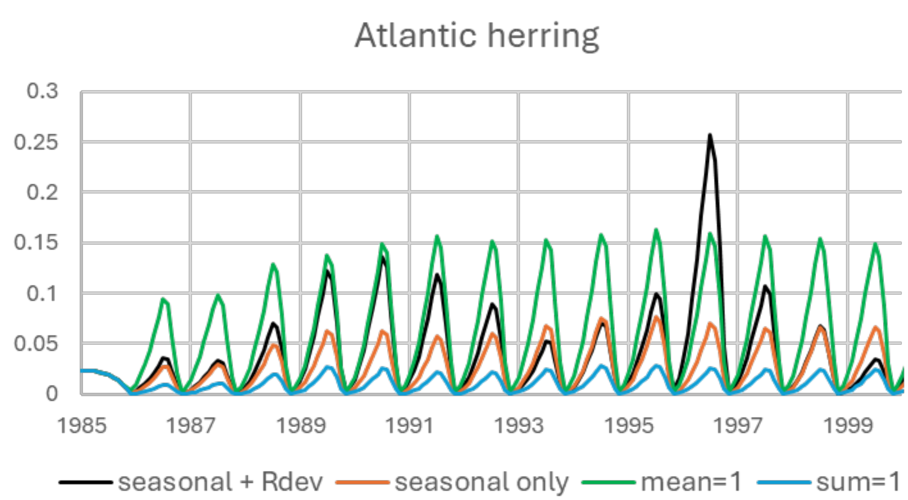
Forcing functions are applied as a multiplier on the number of eggs produced.
ft = 1 implies baseline egg production is realized during month t.
ft > 1 implies enhanced production during month t

$$E_t = f_t \cdot \sum_a N_{a,t}(w_{a,t} - w_{mat,t})$$

D. Chagaris tested the effects of scaling egg production to mean=1 and sum=1 and found odd behavior in seasonal egg production under those settings for striped bass and Atlantic menhaden, but Atlantic herring, which used both a constant seasonal and time-varying annual forcing function behaved as expected for long-term egg production.


Striped Bass age 0-2 monthly biomass
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
1980
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030
baseline
mean = 1
sum=1


Menhaden juv biomass
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
1980
1985
1990
1995
2000
2005
2010
2015
2020
2025
2030
baseline
mean = 1
sum=1


Atlantic herring
0.3
0.25
0.2
0.15
0.1
0.05
0
1985
1987
1989
1991
1993
1995
1997
1999
seasonal + Rdev
seasonal only
mean=1
sum=1

Document justification behind prey-switching parameter choices

Prey-switching explored in seasonal egg production models:

- ➔ Run 85 Pj = 0
- ➔ Run 86 Pj = 0.5

Diagnostics across Run 85 and 86 were similar for most species, but improved for striped bass in Run 86. Therefore, for the combined egg production and seasonal kij forcing run, Run 139 Pj = 0 for SB and 0.5 for others. This was inherited by runs 144 and 150 that included the Atlantic herring recruitment deviations.

Compare the effects on the ERPs of finer scale interpolation in striped bass and menhaden F rates separately for the trade-off plots

2-way interpolation, both species at 0.1

SB Fmult	diff from Ftarg=0.934	SB Fmult	AM Fmult	Bratio	diff from Bratio=1
0.863	0.071	0.926	0.516	1.032	0.032
0.884	0.050	0.926	0.589	1.020	0.020
0.905	0.029	0.926	0.663	1.008	0.008
0.926	0.008	0.926	**0.737**	0.996	0.004
0.947	0.013	0.926	0.811	0.982	0.018
0.968	0.034	0.926	0.884	0.968	0.032
0.989	0.055	0.926	0.958	0.954	0.046

2-way interpolation Striped Bass=0.01, Atlantic Menhaden=0.1

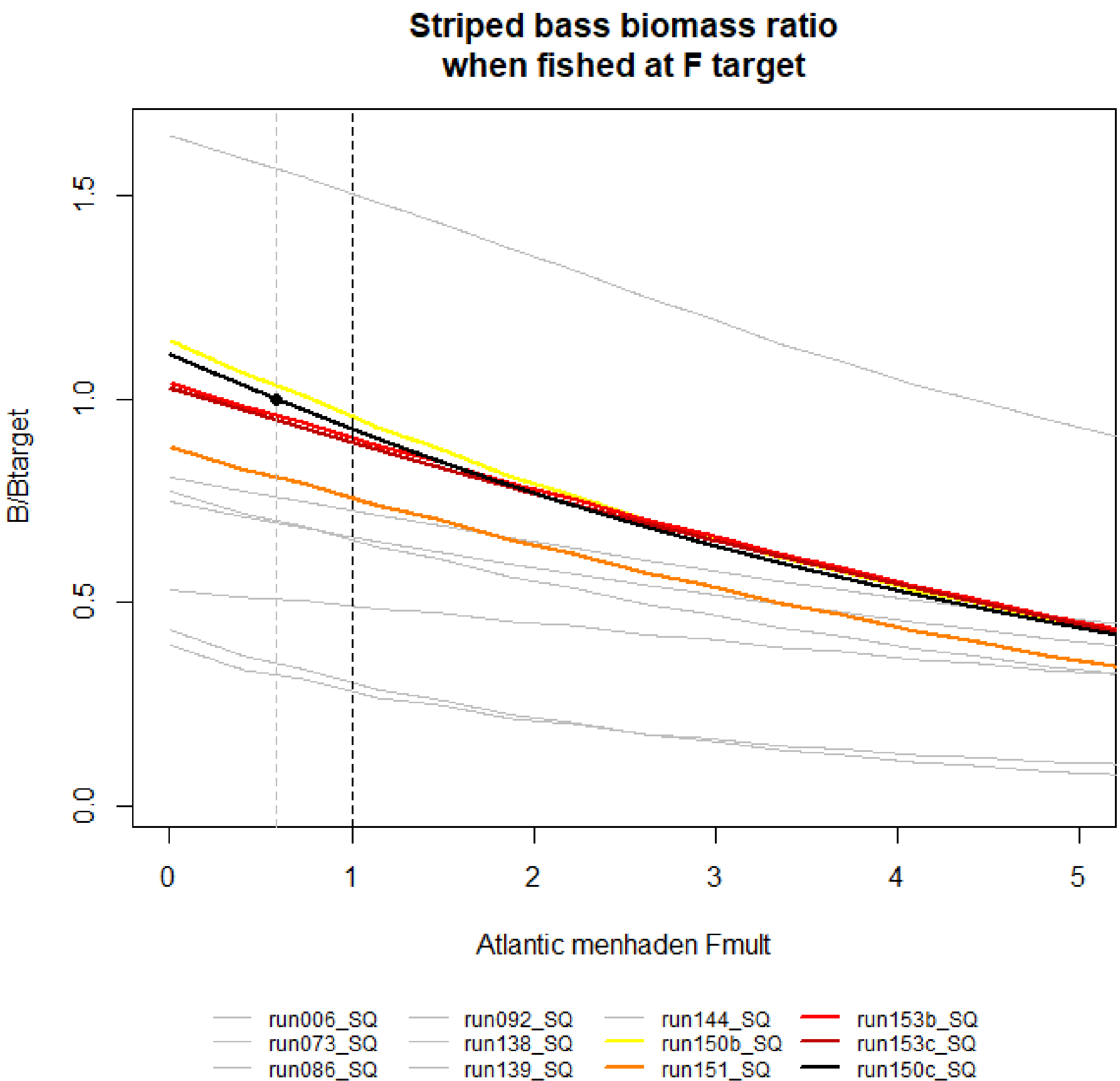
SB Fmult	diff from Ftarg=0.934	SB Fmult	AM Fmult	Bratio	diff from Bratio=1
0.9284	0.0056	0.9347	0.442	1.030	0.0305
0.9305	0.0035	0.9347	0.516	1.018	0.0185
0.9326	0.0014	0.9347	0.589	1.006	0.0064
0.9347	0.0007	0.9347	0.663	0.994	0.0056
0.9368	0.0028	0.9347	0.737	0.982	0.0176
0.9389	0.0049	0.9347	0.811	0.968	0.0316
0.9411	0.0071	0.9347	0.884	0.954	0.0456

1-way interpolation with Striped Bass at Ftarget

AM interpolated at 0.1				AM interpolated at 0.01			
SB Fmult	AM Fmult	Bratio	diff from Bratio=1	SB Fmult	AM Fmult	Bratio	diff from Bratio=1
0.934	0.368	1.041	0.0406	0.934	0.567	1.004	0.0039
0.934	0.442	1.027	0.027	0.934	0.575	1.003	0.0026
0.934	0.516	1.013	0.0134	0.934	0.582	1.001	0.0012
0.934	**0.589**	1	0.0001	0.934	**0.589**	1	0.0001
0.934	0.663	0.986	0.0135	0.934	0.597	0.999	0.0015
0.934	0.737	0.973	0.0268	0.934	0.604	0.997	0.0028
0.934	0.811	0.96	0.0399	0.934	0.612	0.996	0.0041

Interpolation was used to create smoothed surface plots, and then the ERPs were identified from the interpolated values. A 2-way coarse interpolation was initially used, creating imprecision when looking up the ERPs. These tables demonstrate the effects of different interpolations, and how the ERPs were identified where the Bratio is closest to 1. Ultimately, the best approach is to interpolate 1-way over the AM F rates in 100 increments per simulated F value, while using the projections with Striped bass fished at their Ftarget.

Plot trade-off curves only for plausible model runs



Tradeoff curves for a subset of model runs. Colored lines are models with configurations that are similar to base run, while the gray lines are from earlier model runs that were ultimately not considered as a potential base run.