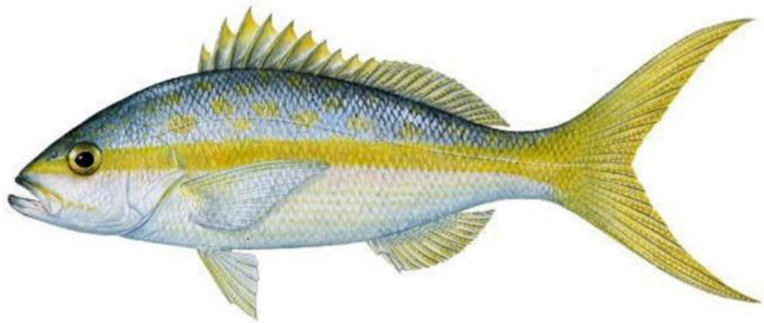

Interim Analysis for Southeastern U.S. Yellowtail Snapper

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Executive Summary

An interim analysis was conducted for Yellowtail Snapper following the Benchmark SEDAR 64 (S64) stock assessment (http://sedarweb.org/sedar-64). This analysis applied updated landings and discards data for each fleet (commercial, headboat, and MRIP [a combination of charter, private, and shore modes]) to the S64 base model from 2018 – 2020. Adjusted projections of spawning stock biomass, recruitment, and retained yield to inform the Acceptable Biological Catch (ABC) and the Annual Catch Limit (ACL) account for the updated landings and discards, as well as sector allocations in the South Atlantic.

Commercial landings decreased to 901.804 mt (1,988,139 lbs.) in 2018, increased again slightly in 2019 to 1,000.598 mt (2,205,944 lbs.), and then dropped in 2020 to 638.690 mt (1,408,072 lbs.) as a result of the COVID-19 pandemic. Headboat landings were 113,282 fish for 2018, 241,516 fish in 2019, and 169,626 fish in 2020. MRIP landings in 2018 were estimated to be 1,696,551 fish, then decreased to 805,637 fish in 2019, and increased again to 1,509,868 fish in 2020.

Commercial discards in 2018 were 29,956 fish and assumed in 2019 – 2020 to be 38,597 fish and 24,636 fish, respectively, using a 5-year average ratio. Headboat discards in 2018 were 46,598 fish, 62,499 fish in 2019, and 45,006 fish in 2020. MRIP discards were 2,760,814 fish in 2018, 1,601,356 fish in 2019, and 2,514,831 fish in 2020.

The interim analysis found that Yellowtail Snapper was not overfished nor undergoing overfishing in the terminal year 2020. The MFMT (defined as $F_{30\%SPR}$) was estimated to be 0.429 yr^{-1} and $F_{current}$ was estimated to be 0.292 yr^{-1}; therefore, the F ratio ($F_{current}$/MFMT) was equal to 0.68. The age-4 fishing mortality rates rose above the MFMT from 1993 – 1995, declined below the MFMT through 2001, and remained variable but stable through 2020. The $SSB_{F30\%SPR}$ for this interim analysis was estimated at 1,915.86 metric tons (4,223,743 pounds) and the MSST (defined as 0.75* $SSB_{F30\%SPR}$) was therefore defined as 1,436.90 metric tons (3,167,807 pounds). $SSB_{current}$ was estimated to be 2,810.33 metric tons (6,195,718 pounds); therefore, the SSB ratio ($SSB_{current}$/MSST) was equal to 1.47. The estimated spawning stock biomass remained above the MSST throughout the timeseries, but was below the target $SSB_{F30\%SPR}$ from 1995 – 2000. The trend in spawning stock biomass generally increased from 1996 – 2016 but then gradually declined through 2020.

Projected yield streams under constant fishing mortality rate scenarios equal to the MFMT (F = 0.429) or the rate associated with P* = 0.375 (F = 0.418; and therefore, above historical estimates) were initially high, but then decreased quickly to around 3.6 million lbs. by 2025 as the spawning stock biomass declined towards the size associated with $SSB_{F30\%SPR}$ and the OFL.

Constant catch scenarios based on average 3- and 5-year yields under with the constant fishing mortality rate equal to MFMT (4.298 and 4.071 million pounds, respectively) or the constant fishing mortality rate associated with P* = 0.375 (4.025 and 4.237 million pounds, respectively) scenario led to rapid declines in spawning stock biomass reaching at or below $SSB_{F30\%SPR}$ by 2025. Fishing mortality rates in these constant catch scenarios were estimated to exceed the MFMT after only a few years into the projections. Under the constant catch equal to the equilibrium yield at OFL (3.496 million pounds) scenario, spawning stock biomass declines to sizes similar to those in the early- and mid-2000s without approaching $SSB_{F30\%SPR}$. Fishing mortality rates for this scenario also did not approach the MFMT and were of similar magnitude to those of the past 20 years.

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1. Introduction

Interim analyses are designed to occur between regular stock assessments conducted through the Southeast Data Assessment and Review process (SEDAR) to provide the opportunity to adjust harvest recommendations based on current stock conditions. An interim analysis of southeastern U.S. Yellowtail Snapper was requested by both the South Atlantic Fishery Management Council (SAFMC) and Gulf of Mexico Fishery Management Council (GMFMC) after concerns were raised that management changes would require the use of projections beyond five years from the terminal year 2017 in the SEDAR 64 (S64) benchmark assessment (SEDAR 2020). Both Councils' Scientific and Statistical Committees (SSCs) discourage the use of projections beyond five years from the terminal data year in a stock assessment, due to increases in uncertainty in the projections beyond that point in time.

In this interim analysis, an Interim Base Model (IBM) was constructed from the S64 base model and run in Stock Synthesis (SS) version 3.30.15. Further descriptions of SS options, equations, and algorithms can be found in the SS user's manual (Methot et al. 2020), the NOAA Fisheries Toolbox website (http://nft.nefsc.noaa.gov/), and Methot and Wetzel (2013). The R statistical environment (R Core Team 2020) and the 'r4ss' package (Taylor et al. 2021) were utilized extensively to summarize model outputs, perform certain model diagnostics, and develop various graphics outputs.

Only commercial and recreational landings and discard data from 2018 – 2020 were requested to be updated in the IBM. Therefore, no indices nor any length and age composition data were updated. As was determined for the S64 benchmark assessment, only landings and discards data from Florida waters were considered as inputs into the IBM.

2. Data and Methods

2.1 Landings and Discard Data Sources

2.1.1 Commercial

Commercial landings of Yellowtail Snapper from 2018 – 2020 were obtained solely from Florida's Marine Fisheries Trip Ticket program. Since commercial landings were predominately from hook and line gear types, landings and discards were grouped among all gear types as they were in the S64 base model. Landings for 2018 were initially provided to the S64 analytical team during the S64 Data Workshop and were validated for this interim analysis. Landings decreased to 901.804 mt (1,988,139 lbs.) in 2018 after a timeseries high in 2017, but then increased again slightly in 2019 to 1,000.598 mt (2,205,944 lbs.; Figure 1, Tables 1-2). In 2020, commercial landings dropped to 638.690 mt (1,408,072 lbs.) as a result of the COVID-19 pandemic (Figure 1, Tables 1, 2, A1). Landings continued to be predominantly from the Florida Keys region (Table 1) and represented 96%, 95%, and 95% of the 2018 – 2020 landings, respectively. During this time, annual Yellowtail Snapper commercial landings from the south Atlantic (i.e., northeast, southeast, and Florida Keys regions) consistently comprised 99% of the total annual landings. Annual standard errors (in log-space) for commercial landings were assumed to equal 0.05 for years 2018 – 2020 (Table 2).

In the S64 benchmark assessment, commercial discards were inferred from NOAA's Coastal Fisheries Logbook Program (CFLP). The discards for 2018 were initially provided to the S64 analytical team during the S64 Data Workshop and validated to be 29,956 fish. Due to insufficient time and resources, the discards for 2019 – 2020 were assumed based on an average ratio of discards/landings calculated using 5 years of data from 2014 – 2018. The 5-year ratio was thus calculated to be 0.039 and applied to the landings data for 2019 – 2020. Discards for 2019 were calculated at 38,597 fish and at 24,636 fish for

2020 (Figure 3, Table 2). The coefficient of variation (CV) for commercial discards was calculated to be 3.58 (Table 2) and was a 5-year weighted average based on the discards and CVs from 2014 – 2018.

2.1.2 Headboat

Estimates of headboat landings and discards of Yellowtail Snapper from 2018 – 2020 were obtained from the Southeast Region Headboat Survey (SRHS) but were not characterized by region. Headboat landings continued to be a small component of recreational Yellowtail Snapper landings and were found to be 113,282 fish for 2018, 241,516 fish in 2019, and 169,626 fish in 2020 (Figure 2, Table 3). Headboat discards in 2018 were 46,598 fish, 62,499 fish in 2019, and 45,006 fish in 2020 (Figure 3, Table 3). The SRHS design prevents variance estimates from being developed, therefore the standard errors (in log-space) for years 2018 – 2020 were assumed equal to 0.25 for the landings and 0.5 for the discards (Table 3), as was configured in the S64 base model. Headboat landings in numbers were used as input to the IBM, however headboat landings in pounds are provided in Table A1.

2.1.3 MRIP

Estimates of recreational landings and discards of Yellowtail Snapper from 2018 – 2020 by anglers fishing from shore or using private, rental boats, or charterboats came from the Marine Recreational Information Program (MRIP). Landings and discards estimates were fully calibrated based on the Access Point Angler Intercept Survey (APAIS) and Fishing Effort Survey (FES). In 2018, MRIP landings were estimated to be 1,696,551 fish, decreased to 805,637 fish in 2019, then increased again to 1,509,868 fish in 2020 (Figure 2, Tables 4-5). MRIP landings from 2018 – 2020 were predominantly from the Florida Keys (Table 4) and southeast Florida regions and comprised 95%, 91%, and 97% of the annual landings, respectively. MRIP discards in 2018 were 2,760,814 fish, 1,601,356 fish in 2019, and 2,514,831 fish in 2020 (Figure 3, Table 5). The calculated annual CVs were converted to standard errors (in log-space) as $\sqrt{(log_e(1 + (CV)^2))}$ and are presented in Table 5. MRIP landings in numbers were used as input to the IBM, however MRIP landings in pounds are provided in Table A1.

After the completion of the S64 benchmark assessment, a concern arose over the atypically high MRIP landings estimate for 2017 in the southwest region of Florida (R. Rindone, Gulf of Mexico Fishery Management Council, pers. Communication). For this interim analysis, a term of reference (TOR) was included to evaluate any potential issues with this data and to determine whether special treatment would be appropriate. The percentages of annual total catch for Yellowtail Snapper from 2015 – 2016 and 2018 – 2019 in the southwest Florida region were on average 2.7% of the annual Florida-wide total catches, but in 2017 was 10.9%. In 2017, a total of 354 trips and 528 interviews caught Yellowtail Snapper in Florida where 33 trips (9.3%) and 50 interviews (9.4%) were sampled in southwest Florida. Of those 33 trips, more than half of the estimated total catch came from 5 trips, all of whose calculated catch weights (‘wp_catch’) were each >2,000 (range: 2919.11 – 4159.64). One trip claimed (i.e., type A data) 30 fish, another claimed 13 fish, one trip recorded 5 interviews each harvesting (i.e., type B1 data) 5 fish (i.e., 25 total fish), and another trip recorded 5 interviews harvesting a total of 23 fish and discarding (i.e., type B2 data) 3. Another trip with 2 interviews each claimed to have discarded 10 fish (i.e., 20 total fish). The CVs associated with the 2017 landings, releases, and total catch values for southwest Florida were 0.11, 0.02, and 0.08, respectively. These landings and releases data were elevated compared to the neighboring years as they were the result of a handful of trips that reported high catches and were associated with extremely high catch weights. It was the opinion of the analytical team not to alter the 2017 MRIP data in the IBM because these data are reflective of the inherent variability when estimating landings and discards in a large region mainly comprised of unfavorable Yellowtail Snapper habitat. In addition, these data had also been examined and approved for use during the S64 Data Workshop. Nonetheless, to evaluate the impact of the atypically high MRIP landings and discards in the southwest region in 2017, a sensitivity run was performed and explained in further detail below.

The 2020 MRIP data was also investigated for potential issues as there were concerns that reduced sampling coverage and higher than usual boating activity observed during the COVID-19 pandemic caused biases in the estimated landings and discards. In 2020, the data contained a total of 489 interviews (Table 6) where 295 (60.3%) were conducted by samplers with anglers who caught Yellowtail Snapper and 194 (39.7%) were imputed (i.e., 'imp_rec'==TRUE) from all APAIS data collected in 2018 and 2019 from the same strata as the 2020 data gap (R. Cody, NOAA, pers. communication). Original sample weights were also reduced by a factor of two to account for using two years of data (Cody 2021). These imputed interviews occurred mostly in the Florida Keys (128 imputed interviews) and southeast Florida (49 imputed interviews) regions. The landings and discards data for 2020 were consistent with recent years; therefore, it was determined that the reviewed methods of imputation were sufficient for Yellowtail Snapper and were used as inputs in the IBM.

2.2 Data Weighting

The Francis weights applied to the length and conditional length-at-age composition data were updated using the same method (TA1.8 in Francis 2011) as in the S64 base model. Given that no length or conditional length-at-age composition data were updated in this interim analysis, the IBM weights aligned closely with the S64 base model weights (Table 7).

2.3 Model Diagnostics

Model diagnostics of the IBM were performed in R using the 'r4ss' and 'ss3diags' (github.com/JABBAmodel/ss3diags) packages and largely follow the recommendations put forth in the Carvalho et al. (2021) 'cookbook' for integrated stock assessment models. While we briefly summarize each diagnostic here, further descriptions can be found in Carvalho et al. (2021) and references therein.

2.3.1 Convergence

Convergence of the IBM was initially assessed by determining that there were no parameters estimated at a bound, the final gradient was at most 0.0001, and that the Hessian matrix was positive definite.

High correlation among parameters was also assessed as it can lead to poor model stability along with flat likelihood response surfaces. While some parameters will always be correlated due to their structural nature (e.g., growth and stock-recruitment parameters), many highly correlated parameters may warrant reconsideration of modeling assumptions and parameterization. Therefore, correlation among parameters was examined and any correlations with an absolute value greater than 0.7 were reported. Parameters correlated due to their structural nature, were estimated in different phases of the IBM to reduce their direct influence on one another.

Once individual model convergence was established, a jitter analysis was performed on the parameter's starting values to gauge whether the IBM had converged on a global solution instead of a local minimum. For this analysis, initial values were jittered by up to 20% and 200 iterations were performed.

2.3.2 Goodness of fit

Fits to landings, discards, indices, and length and age compositions were evaluated via visual inspection of residuals. In addition, a non-parametric runs test (Wald and Wolfowitz, 1940) was performed on the indices and length composition data to test for randomness and the presence of temporal autocorrelation in residuals. Combined root mean square error (RMSE) values were also calculated for the indices and length composition data to evaluate goodness-of-fit.

2.3.3 Model Consistency

Consistency within the IBM was evaluated by identifying how the sources of information influence various model estimates. This was done first through a likelihood component profile on the equilibrium recruitment parameter, R0. This parameter, largely regarded as an ideal global scaling parameter, was sequentially fixed to plausible values ranging from 9.0 – 11.0 by 0.1 and the change in total and data-component likelihoods were examined.

An age-structured production model (ASPM) and an ASPM with estimated recruitment deviations (ASPMdev) were also developed in SS to investigate which processes were influencing the shape of the production function and whether composition data was influencing the variability in recruitment. For the ASPM, this was completed first by fixing all parameters to those values estimated by the IBM, except for the R0 parameter and the initial fishing mortality parameters. Next, the likelihood components (i.e., lambdas) for the length and age composition data were set to zero along with the recruitment deviations for both the early and main periods such that only the catch and indices of abundance were fit by the model. For the ASPMdev, the recruitment deviations of the ASPM were configured back to the values in the IBM and the bias-correction factor was re-adjusted following Methot and Taylor (2011). Trends in both spawning stock biomass and fishing mortality were compared between the IBM, the ASPM, and the ASPMdev.

The IBM was subject to a retrospective analysis which removed five successive years of data from the model (i.e., years 2016 – 2020). Iteratively removing data associated with the model's terminal year elucidates the effect of the final year on model results. If results of this analysis show a retrospective bias (consistent patterns of increasing or decreasing model estimates and related derived quantities with each retrospective peel), it can be an indication of model misspecification of temporal dynamics. It is preferable for estimates associated with each retrospective peel to be randomly distributed around the IBM results. Model performance was evaluated through visual inspection of retrospective patterns and the Mohn's Rho (ρ) metric (Mohn 1999, Hurtado-Ferro et al. 2015). Here, as in the S64 benchmark assessment, we use the 'rule of thumb' ρ values (-0.15 to 0.20) proposed by Hurtado-Ferro et al. (2015) for longer-lived species to characterize retrospective bias.

2.3.4 Prediction Skill

Having established model consistency and structural stability, the predictive skill of the IBM was evaluated to check whether the model's predictive capacity is consistent with the future reality. This was done in two ways. First, a retrospective forecast was performed by adding model-based hindcasts to each of the five-year peels of the retrospective analysis. Then, a forecast bias, which is an average relative error corresponding to the retrospective bias (i.e., Mohn's Rho (ρ) metric) was computed to gauge model performance and consistency when adding data.

The second method was through the hindcast cross-validation technique, which compares observations to their predicted future values, and was applied to both the indices and length composition data. Predictive skill was evaluated based on the mean absolute scaled error (MASE) which scales the mean absolute error of the forecasted value to the mean absolute error of the naïve in-sampled value and indicates whether the average model forecasts are better or worse than a random walk. For example, MASE scores >1 indicate average model forecasts are worse than a random walk (i.e., no predictive skill). However, a MASE score of 0.5 would indicate that the model forecasts twice as accurately as a naïve baseline prediction, thereby containing predictive skill.

2.4 Sensitivity Runs

The results of the IBM were first compared with those from the S64 base model to evaluate model consistency after the addition of available 2018 – 2020 landings and discard data.

To evaluate the impact of the elevated MRIP landings and discard data in 2017 for the southwest region of Florida, a sensitivity analysis was performed. The analytical team was requested to replace the 2017 southwest Florida landings and discard data (304.551 and 114.382 thousand fish, respectively) with the geometric mean of landings and discards from 2015, 2016 ,2018, and 2019 for the same region (44.750 and 33.064 thousand fish, respectively). Florida-wide landings for 2017 were therefore reduced from 1,550.296 thousand fish to 1,290.495 thousand fish, whereas Florida-wide discards were reduced from 2,274.822 to 2,191.679 thousand fish. These imputed values were used as sensitivity run inputs and the effects on model results were evaluated.

Lastly, after the S64 benchmark assessment was completed, the analytical team discovered that the MRIP CPUE was mischaracterized in the final stock assessment report as a 'total catch per trip' index but was in fact developed in units of 'total catch per angler'. While this characterization of effort was consistent with the SEDAR 27A (2012) benchmark assessment, it was the original intent of the S64 analytical team to update the index to units of 'total catch per trip' during the benchmark assessment process. For the sake of transparency and given the level of influence this index had on both the S64 base model and the IBM, a sensitivity run with an MRIP CPUE index correctly configured as total catch per trip was performed to evaluate any potential changes in stock abundance and to help inform discussions in future Yellowtail Snapper assessment processes.

2.5 MCMC Analysis

Monte Carlo Markov Chain (MCMC) is a method of generating posterior distributions of model parameters and was used in this analysis to estimate uncertainty in fishing mortality and spawning stock biomass. MCMC allows a probabilistic reporting of the uncertainty associated with the estimated values. Estimates of population values in the terminal year of the stock assessment are often the most uncertain. Assuming the MCMC posterior distributions provide reliable estimates of model uncertainty, the probability that the estimated terminal year value is above or below the overfished/overfishing reference points can be calculated. In this way, a level of risk associated with failing to reach the reference points can be quantitatively specified.

Two MCMC chains were produced. For each chain, a total of 10,000,000 iterations were performed but only one out of every 2,000 iterations was saved, resulting in 5,000 potential iterations used to generate estimates of uncertainty in fishing mortality and spawning stock biomass. Visual inspection of trace plots was used to adjust appropriate levels of burn-in and thinning as well as to address any autocorrelation in the iterations. Convergence of the two chains was assessed using Gelman and Rubin's (1992) potential reduction scale factor implemented in the 'coda' package (Plummer et al. 2006) in R.

2.6 Projections

Short- and long-term deterministic projections were conducted to estimate Yellowtail Snapper spawning stock biomass and yield (or fishing mortality rates) under a range of harvest scenarios. The method to project the assessment results was developed in the R statistical computing environment by assessment scientists at the NOAA Southeast Fishery Science Center. This method uses an iterative process to set fishing mortality rates each year to ensure that a given constant fishing mortality rate (or constant catch) scenario was achieved. In addition, fleet allocations are also specified and held constant for all years.

Commercial and recreational allocations are 52.56% and 47.44% in the south Atlantic, respectively, and are not currently specified in the Gulf. The south Atlantic fleet allocations are applied to all Florida landings since the region contributes most of the landings. Nearly 90% of Yellowtail Snapper commercial landings in Florida have occurred in Monroe County alone since 1962 (Table 3.2 in the SEDAR 64 SAR), and recreational landings in the south Atlantic comprise well over 95% of the total landings. Fleet allocations in the south Atlantic are very similar to the relative retained biomass by fleet within the IBM averaged over years 2018 – 2020 (50.8% commercial, 4.9% headboat, and 44.3% charter and private boat modes).

Projections were performed under several assumed conditions. Growth, stock-recruit parameters, as well as fleet selectivity and retention were kept constant as estimated by the IBM. Recruitment was projected beginning in 2021 by using the stock-recruitment parameters as estimated by the IBM and was projected to be 17.792 million fish. This is similar to the average estimated recruitment for the assessment timeseries 1992 – 2020 (17.790 million fish) and the average during the last three years in the timeseries 2018 – 2020 (17.098 million fish).

Several projection scenarios explored the effects of various fishing mortality and constant catch conditions. The first and second scenarios were run for 100 years (2021 – 2121) to reach equilibrium and investigated the effect of constant fishing mortality rates when fishing mortality rates were either held constant at $F_{30\%SPR}$ (MFMT) or the derived P* value (P* = 0.375). Only results from the short-term projections (2021 – 2031) are presented in this report. The third and fourth scenarios assumed a constant catch based on the retained landings estimated under the P* fishing mortality rate, averaged over the initial 3 years of the projection (2021 – 2023) or 5 years (2021 – 2025). Similarly, the fifth and sixth scenarios assumed a constant catch based on the retained landings estimated under $F_{30\%SPR}$ (MFMT), averaged over the initial 3 years of the projection (2021 – 2023) or 5 years (2021 – 2025). The TORs also state to evaluate the projected spawning stock biomass when catch is held constant at the equilibrium yield at F_{MSY}, however, Yellowtail Snapper in the southeastern U.S. is managed using $F_{30\%SPR}$ as an F_{MSY} proxy. Therefore, the final projection scenario evaluated spawning stock biomass and recruitment when catch is held constant at the equilibrium retained yield at $F_{30\%SPR.}$

3 Model Results

The landings data for the commercial, headboat, and MRIP fleets were fit well within the IBM (total negative log-likelihood = 5.904e-012) and the predicted landings for all three fleets during the interim years 2018 – 2020 matched the observed landings exactly. Discard data for the commercial, headboat, and MRIP fleets were also fit fairly well within the IBM (total negative log-likelihood = 146.444). Much like the S64 base model, the fits to the commercial discards by the IBM continued to be a little overestimated beginning in 2009 and continued through the interim years (Figure 4). Fits to the headboat discards (Figure 5) as well as the MRIP discards (Figure 6) through the interim years were also reasonable, as estimates were close to the observed values and within the 95% confidence intervals.

The predicted age-0 recruitment is summarized in Table 8 and Figure 7a-b. The IBM estimated age-0 recruitment as a mostly flat trend (~17.7 – 17.9 million fish) with large confidence intervals from 2017 – 2020 (Figure 7a) and, apart from 2019 (15.6 million fish), without any deviation (Figure 7b) from the estimated stock-recruitment relationship. This is due in large part to an absence of any indices for the IBM to fit to during the interim period and to the termination of the RVC juvenile index in 2016. Recruitment in 2019 was estimated to negatively deviate and may be in response to the large decline in MRIP landings in 2019 (Figure 2) and a declining trend in commercial landings beginning in 2018 after a timeseries high in 2017 (Figure 1), given their differences in selectivities.

The predicted total biomass and spawning stock biomass are summarized in Table 8 and Figures 8 – 9. The total biomass gradually decreased in trend during the interim period from a high (7,759 mt) in 2016 to 6,435 mt in 2020 (Table 8, Figure 8). The predicted spawning stock biomass also largely follows this trend and gradually declined from a high of 3,310 mt (7,296,388 lbs.) in 2016 to 2,730 mt (6,019,318 lbs.) in 2020 (Table 8, Figure 9).

The annual instantaneous fishing mortality rates on age-4 Yellowtail Snapper are presented in Table 9 and Figure 10. This age was designated in the S64 benchmark assessment based on the mid-point of the relative fleet-specific maximum selectivities, allows for a comparison of fishing mortality rates across time, and reduces the variability around this estimate caused by varying levels of fishing mortality on different ages over different years. Nonetheless, fleet-specific fishing mortality rates (i.e., instantaneous apical rates representing the fishing mortality level on the most vulnerable age class) are also presented in Table 10 and Figure 11. The annual fishing mortality rate on age-4 Yellowtail Snapper has been variable but stable for nearly the past two decades (mean age-4 F = 0.287 yr^{-1}) and this trend continued during the interim period (Figure 10). Fishing mortality in 2020 was estimated at 0.281 yr^{-1} (Table 9) and had an average of 0.305 yr^{-1} from 2017 – 2020. Apical fishing mortality rates by fleet during the interim period continued to be highest for the MRIP fleet despite an estimated decline in 2019 while rates for the commercial fleet remained steady with a low in 2020 as a likely result from the COVID-19 pandemic (Figure 11). The headboat fleet continued to exert the least amount of fishing mortality but saw an increase in 2019 given its highest reported landings since the early- and mid-1990s (Figure 11).

3.1 Model Diagnostics

3.1.1 Convergence

The IBM converged with a total objective function of 742.9. The model contained no parameters on the bounds, had a small final gradient <0.0001, and had a positive definite Hessian matrix. Highly correlated parameters were inspected, but all were found to be structurally correlated and therefore left as-is estimated in their different model phases.

The results of the jitter analysis suggested that the IBM had converged on a global solution but was a little sensitive to the initial parameter values. From the 200 jittered runs, 112 runs (56%) had a high gradient, and 58 runs (29%) did not have a positive definite Hessian matrix. No jittered runs were found to contain a total likelihood lower than the IBM. In an effort to adequately compare the results of the jittered analysis with those of the IBM, we filtered the jittered runs to include only those which both had a small final gradient and a positive definite Hessian matrix, indicating more plausible alternative model solutions. A total of 85 runs (42.5%) remained whose total likelihood continued to be equal to or greater than the IBM (Figure 12), therefore suggesting that the IBM had converged on a global solution.

3.1.2 Goodness of fit

The joint residual plots for both the indices (Figure 13a) and the mean length composition data (Figure 13b) indicated a good fit to the data as combined RMSE values were 0.191 and 0.035, respectively. Residual variability of the indices generally decreased over time as illustrated by the loess-smoother and the size of the boxplots. The interquartile ranges (box size) were greater in the early 1990s and may be attributed to the initial conflict between the MRIP CPUE (i.e., consecutive positive residuals) and Commercial CPUE (i.e., consecutive negative residuals) indices. Residuals and interquartile ranges of the mean length data were small and consistent across time, indicating general agreement with the fisheries and index data.

The residual series of the all the indices except for the Commercial CPUE passed the runs test (Figure 14a-d; Table 11). The Commercial CPUE had two years where the residuals were greater than three standard deviations and several years which were sequentially positive (2005 – 2011) or negative (1993 – 1998; Figure 14a), suggesting a potential misspecification which may need to be re-evaluated in the following benchmark assessment. When this index was removed from the S64 base model in the jack-knife analysis (SEDAR 2020), it had very little impact on the estimates of spawning stock biomass. In contrast, when the MRIP CPUE index (which passed the runs test here; Figure 14b) was removed, the scale of spawning stock biomass increased in recent years, further suggesting the IBM's reliance on this index.

All but one mean length residual series passed the runs test (Figure 15a-f; Table 11). The mean length residuals of the headboat fishery failed as most years exhibited non-random variation. Years 1992 – 1997 were sequentially positive while 2000 – 2011 were sequentially negative (Figure 15b). No length composition data were removed in the S64 base model, but further evaluation of the importance of this data series is recommended in the next benchmark assessment as the headboat fishery index was also rejected from use during the S64 Data Workshop (SEDAR 2020). The length composition data of the headboat fishery may also be conflicting with similar length composition data from the commercial (Figure 15a) and MRIP (Figure 15c) fleets along with the RVC adult (Figure 15d) and MRIP CPUE (Figure 15f) indices who all passed the runs test.

3.1.3 Model Consistency

The profile likelihood on the R0 parameter suggests that the parameter is largely influenced by the recruitment deviations component of the IBM (Figure 16). The age composition component also agreed with the minimum value found on the R0 profile, however, its data was found to be less informative as changes in gradient across the profile were smaller and more gradual. The discard and length composition data components were also less informative but were in conflict with the age composition and recruitment deviation components and favored a lower value on the R0 profile. The index data component appeared to be the least informative data source as values were mostly flat across the R0 profile.

The results from the ASPM indicate that for most of the timeseries there is enough information in both the catch and index data for the production function to largely drive the stock dynamics and for the model to be adequately informed about scale (Figure 17a). Fits to the Commercial CPUE (RMSE = 0.226; Figure 17b; Table 12) were slightly better than to the MRIP CPUE (RMSE = 0.238; Figure 17c; Table 12) and trends for the RVC juvenile index were expectedly flat (RMSE = 0.241; Figure 18a; Table 12) given the lack of any variability in recruitment in the ASPM. The RVC adult index had the worst fit (RMSE = 0.355; Table 12) of the four indices. When the recruitment deviations were included (i.e., in the ASPMdev), fits to all indices were improved (Table 12) and the estimated spawning stock biomass became very similar to that estimated by the IBM (Figure 17a), suggesting that the process error as captured by the variability of age-0 recruitment was needed to better fit the trends in the indices. The terminal 3 – 4 years of all models were also the most uncertain given the absence of any index or composition data and as evidence by the larger error bars surrounding the estimated recruitment (Figure 18b) and recruitment deviations (Figure 18c).

The ASPM estimated spawning stock biomass at a similar scale and trend to the ASPMdev and the IBM but began to deviate from the other models and decreased during the last 6 years of the timeseries (Figure 17a). Without variable recruitment, the ASPM was unable to fit to the recent estimated pulse in age-0 recruitment from 2012 – 2016 initially captured by the juvenile index (Figure 18a) and later observed in the MRIP CPUE index from 2013 – 2017 (Figure 17c) which, therefore, prevented an increase in population size.

The retrospective analysis showed no discernable patterns in estimates of spawning stock biomass or fishing mortality rates after removing successive terminal years (Figure 19a-b). All runs converged and no parameters were found on the bounds. The calculated values for Mohn's rho for SSB (ρ_M = -0.025; Table 13) and age-4 F (ρ_M = 0.033; Table 13) were well within the "acceptable" range for longer-lived species according to Hurtado-Ferro et al. (2015).

3.1.4 Prediction Skill

Retrospective forecasting showed that the one year forward projections were consistent with the overall estimated trend in the reference IBM (Figure 20). In addition, each retrospective peel and retrospective forecast was found to fall within the 95% confidence interval of the reference IBM. The forecast rho value for both SSB and age-4 F increased slightly to ρ_F = -0.037 and ρ_F = 0.047, respectively (Table 13), which suggested model stability with the historical data as well as consistency when subsequent data became available.

To have sufficient observations to gauge predictive capacity, a hindcast with cross-validation of the terminal eight years of data was performed (Figures 21 – 22). This resulted in five observations to measure the ability of IBM to predict the Commercial CPUE and the MRIP CPUE but only two observations for the RVC juvenile and adult indices due to the biennial timing of the survey. These limitations were also due to the absence of updated index and length composition data for the interim period (2018 – 2020). Regardless, we report the results of this diagnostic as it may be informative to management and also to compare to future Yellowtail Snapper assessments.

Both the RVC Adult and MRIP CPUE indices had MASE scores <1 which suggested the IBM contained reasonable prediction skill for these when compared to a naïve forecast (Figures 21b,d). The MRIP CPUE contained the lowest MASE score = 0.64, indicating the ability to predict is nearly twice as accurately as a naïve baseline prediction. The Commercial CPUE and RVC Juvenile indices, on the other hand, were not predicted well as the MASE scores were greater than one. The MASE score was the highest for the RVC Juvenile index (2.41). The model exhibited predictive capacity for all mean length data sources (MASE < 1), with the exception of the RVC Juvenile length data (MASE = 2.05; Figure 22a-f).

These results also corroborate the ASPM diagnostics which showed the importance of the indices in estimating the scale and trend of the stock's abundance. The model's ability to predict the MRIP CPUE index is reassuring as model scale was most sensitive to the inclusion of this index and was relatively invariable to the inclusion of the other indices (see 'jack-knife' analysis in SEDAR 2020).

3.2 Sensitivity Runs

The results of the comparison between the IBM and the S64 base model indicated that the addition of 2018 – 2020 commercial and recreational landings data to the IBM did not create additional conflicts when fitting the data. Moreover, historical estimates of stock abundance (Figure 23a), age-4 fishing mortality (Figure 23b), and age-0 recruitment (Figure 23c-d) were found consistent.

When the MRIP landings data for 2017 in the southwest region of Florida were altered, the impacts to the model were negligible and changes to stock abundance (Figure 24a), age-4 fishing mortality (Figure 24b), and age-0 recruitment (Figure 24c-d) were within the confidence intervals of the IBM. The 'MRIP 2017' sensitivity model estimated slightly lower age-4 fishing mortality and stock abundance in 2017 by slightly adjusting the 4 years of estimated recruitment (2013 – 2016) leading up to 2017. Estimates of stock abundance for years 2019 – 2020 also increased slightly as corresponding fishing mortality estimates remained slightly lower than in the IBM.

When the MRIP CPUE index was correctly configured to units of ‘total catch per trip’, the trend became moderately different (Figure 25a). Model fits to this index and corresponding estimates of stock abundance (Figure 25b) were higher for years 1994 – 1999, but lower for recent years 2014 – 2020. The trend for age-4 fishing mortality responded inversely (Figure 25c) and the lower estimates of spawning stock biomass correspondingly produced lower estimates of recruitment through the terminal year (Figure 25d).

3.3 MCMC Analysis

Of the 5,000 iterations from each chain, burn-in was set at 1,000 with a thinning rate of 2 to help eliminate starting point bias and some early serial correlation. Thus, a total of 2,000 iterations remained for each chain. The two chains were combined, and convergence was evaluated using trace plots (Figures 26 – 27) and the Gelman and Rubin’s (1992) potential scale reduction factor (PSRF) for selected model parameters (R0, SSB0, and steepness) and derived quantities (age-4 F in 2020, SSB in 2020, $F_{30\%SPR}$, SSB at $F_{30\%SPR}$, and the retained yield at $F_{30\%SPR}$). PSRF values for all selected parameters and stock status criteria were close to 1 and since none of the PSRF upper confidence intervals exceeded the ‘rule of thumb’ value of 1.1, it was concluded that the MCMC converged (Table 14).

Posterior distributions were produced for the derived quantities of $F_{30\%SPR}$ (Figure 28a), the retained yield associated with $F_{30\%SPR}$ (Figure 28b), spawning stock biomass at $F_{30\%SPR}$ (Figure 28c), and 75% of the spawning stock biomass at $F_{30\%SPR}$ (Figure 28d). Results of the IBM were found to fall within the interquartile range of the posterior distributions for all considered criteria (Figure 28a-d).

3.4 Stock Status Determination Criteria

A summary of the stock status determination criterion and their values according to the SAFMC and the GMFMC for this interim analysis are presented in Table 15.

The Maximum Fishing Mortality Threshold (MFMT; also referred to as the overfishing limit, OFL) for Yellowtail Snapper is defined as $F_{30\%SPR}$ and overfishing is occurring if the recent average of fishing mortality rates ($F_{current}$) exceeds the MFMT. $F_{current}$ is calculated as the geometric mean of age-4 Yellowtail Snapper fishing mortality rates for 2018 – 2020. The MFMT for this interim analysis was estimated to be 0.429 yr^{-1}, $F_{current}$ was estimated to be 0.292 yr^{-1}, and F_{2020} was estimated to be 0.281 yr^{-1}. Based on the results of the IBM, the southeastern U.S. Yellowtail Snapper stock continues to not be experiencing overfishing ($F_{current}$/MFMT = 0.68; Figure 29).

The minimum stock size threshold (MSST) for Yellowtail Snapper is defined as 75 percent of the spawning stock biomass associated with $F_{30\%SPR}$ (0.75* $SSB_{F30\%SPR}$). The stock is overfished if the recent average spawning stock biomass ($SSB_{current}$) is less than MSST. $SSB_{current}$ is calculated as the geometric mean of the spawning stock biomass for 2018 – 2020. The $SSB_{F30\%SPR}$ for this interim analysis was estimated at 1,915.86 mt (4,223,743 lbs.) and MSST was therefore defined as 1,436.90 mt (3,167,807 lbs.). $SSB_{current}$ was estimated to be 2,810.33 mt (6,195,718 lbs.) and SSB_{2020} was estimated to be 2,730.32 mt (6,019,318 lbs.). Based on the results of the IBM, the southeastern U.S. Yellowtail Snapper stock continues to not be overfished ($SSB_{current}$/MSST = 1.47; Figure 30).

The posterior distributions produced by the MCMC analysis were for the stock status determination criteria and benchmark reference points of $F_{30\%SPR}$ (MFMT), the retained yield associated with $F_{30\%SPR}$, $SSB_{F30\%SPR}$, and MSST and are presented in Figure 28a-d. Additional posterior distributions of the F ratio ($F_{current}$/MFMT) and SSB ratio ($SSB_{current}$/MSST) are presented in Figure 31a-b. The estimates for these reference points as derived by the IBM were near the median values and were within the interquartile ranges of the posterior distributions. The distribution of the F ratio is entirely below one, indicating a high probability that overfishing is not occurring, and the distribution for the SSB ratio is entirely above one, indicating a high probability that the stock is not overfished.

Maximum sustainable yield (MSY) for Yellowtail Snapper is defined as the retained yield associated with $F_{30\%SPR}$ and was estimated by the IBM at 1,587.08 mt (3,498,908 lbs.) with a standard deviation of 129.9 mt (286,380 lbs.). Optimum yield (OY) is defined as the Acceptable Biological Catch (ABC) value based on the SAFMC P* method and is the 37.5th quantile of the equilibrium distribution of the OFL (i.e., the MFMT). The MCMC distribution of the OFL has a median of 1,573.36 mt (3,468,661 lbs.), a standard deviation of 125.1 mt (275,798 lbs.), and a coefficient of variation of 0.08. The corresponding 37.5th quantile is 1,535.04 mt (3,384,180 lbs.) which is 97.6% of the OFL. The MCMC distribution of the equilibrium OFL and an approximate normal distribution based on the mean and standard error estimated by the IBM (1,545.68 mt; 3,407,637 lbs.) are very similar (Figure 32).

3.5 Projections

The $F_{30\%SPR}$ (MFMT) and the fishing mortality rate associated with P* = 0.375 (ABC) are of similar magnitudes (0.429 and 0.418, respectively) and are greater than historical fishing mortality rates after 1996. Associated fishing mortality rates, retained landings, spawning stock biomass, and recruitment values as estimated by the IBM for assessment years (1992 – 2020) and the forecasted years (2021 – 2031) are shown in Figures 33 – 36 and tabulated in Tables 16 – 18. Because of the similarity in the fishing mortality rates, the values are comparable between these two scenarios. Retained landings quickly increased in 2021 in response to the elevated fishing mortality rates, but then decreased quickly to around 3.6 million lbs. by 2025 (Figure 34; Table 16) as the spawning stock biomass declined towards the size associated with $SSB_{F30\%SPR}$ (Figure 35; Table 17) and the OFL. Projected recruitment followed the estimated stock-recruit relationship without additional variability and, therefore, remained fairly constant (Figure 36; Table 18). In addition to increased fishing mortality rates, declines in spawning stock and exploitable biomass also reflect the lower level of recruitment compared to the recent high recruitment period from 2012 – 2014.

Several constant catch scenarios were evaluated and the associated fishing mortality rates, retained landings, spawning stock biomass, and recruitment values are shown in Figures 37 – 40 and tabulated in Tables 19 – 21. These scenarios include the retained yield under $F_{30\%SPR}$ averaged over the initial 3 and 5 years of the projection (4.298 and 4.071 million pounds, respectively), retained yield under the level that corresponds to a P* value of 0.375 averaged over the initial 3 and 5 years of the projection (4.025 and 4.237 million pounds, respectively), and the equilibrium retained yield associated with $F_{30\%SPR}$ (3.496 million pounds). Retained yield and fishing mortality rates were the greatest under the 3- and 5-year average scenarios and were notably higher than the equilibrium yield at OFL. However, they also led to more drastic increases in fishing mortality rates and declines in spawning stock biomass.

Fishing mortality rates quickly surpass the MFMT (0.429) after 2022 under the 3-year average scenarios and after 2023 under the 5-year average scenarios (Figure 37; Table 19). In contrast, fishing mortality rates for the equilibrium yield at OFL scenario do not approach the MFMT and are of similar magnitude to those of the past 20 years. Spawning stock biomass under the 3- and 5-year average scenarios decline below $SSB_{F30\%SPR}$ after about 5 years and approach the MSST after about 10 years (Figure 39). Under the equilibrium yield at OFL scenario, spawning stock biomass declines to sizes similar to those in the early- and mid-2000s and doesn't approach $SSB_{F30\%SPR}$.

4 Discussion

This interim analysis 1) updates the S64 base model to the IBM with landings and discard data for commercial, headboat, and MRIP fleets for years 2018 – 2020, and 2) provides updated projections of yield and spawning stock biomass to inform Annual Catch Limit (ACL) values of southeastern U.S. Yellowtail Snapper based on several constant F and constant catch scenarios.

The results of the model diagnostics suggest the IBM may be suitable for use in the management of Yellowtail Snapper. The IBM demonstrated adequate fits to the various data components and the jitter analysis and low gradient (<0.0001) lend support that the IBM converged to a global solution. The IBM also exhibited model consistency as the removal of successive years of data showed no discernable retrospective patterns in estimates of fishing mortality rates and spawning stock biomass. The results of the R0 profiling, as well as the ASPM and ASPMdev, suggested that the estimates of absolute abundance and trend were consistent and primarily influenced by both the catch information and the variability in recruitment. Retrospective forecasting and the hindcast cross-validation techniques also suggested the IBM exhibited more predictive skill than a random-walk for most data sources with the exception of the Commercial CPUE index, the RVC Juvenile index, and the RVC Juvenile mean length data.

In addition to the sources of uncertainty which resided in the S64 base model, the primary sources of uncertainty added within the IBM were the lack of updated indices and length and age composition data. There were strong estimated recruitment classes from 2012 – 2014 (Figure 18a) which increased stock abundance and were seen moving through in the MRIP CPUE and Commercial CPUE indices (Figure 17b-c). Without any updated indices, the estimated recruitment variability for interim years was at or close to zero (Figure 7b); it's unclear whether any further strong recruitment has occurred after 2016 (i.e., the terminal year of the RVC Juvenile index) and whether the stock abundance is truly beginning to decline after displaying an increasing trend since 1997. While this declining trend was already being exhibited in the terminal years of both the MRIP CPUE and Commercial CPUE indices, there's uncertainty in whether this trend truly continues post-2017 and whether it truly begins to level off as seen in the 2020 terminal year. These data components are normally updated within a more comprehensive update assessment framework, and while updating these components would help to inform the population trend in recent years, it was beyond the scope of this interim assessment.

The variability of catch in the 2017 MRIP landings estimate for southwest Florida was found to be largely resulted from the weighting (i.e., 'wp_catch) of a handful of higher catch records sampled in a large area (Collier to Levy County) of non-primary Yellowtail Snapper habitat. High variability in the catch estimates of the MRIP data is not foreign to Yellowtail Snapper even in primary habitat (i.e., the Florida Keys region) as extremely high landings and discards were estimated for the earlier years of 1981 – 82, 1984, and 1989 – 1991. These estimates remained unaltered in the S64 base model and the IBM, are part of the earlier data period, and model sensitivity to them was assessed in the benchmark process via model start date. The results of the sensitivity run also showed the impact of reducing the 2017 data using a geometric mean was within the bounds of uncertainty produced by the IBM. Therefore, we did not deem special treatment was appropriate in this interim analysis process.

The IBM was found to be moderately sensitive to the reconfiguration of the MRIP CPUE index. This was not unexpected as the trends in stock abundance for both the S64 base model and the IBM are greatly informed by this index. More concerning, however, was the reduction in estimated scale from 2014 – 2017 corresponding to the differences in index trend and the indication that the stock may be approaching the $SSB_{F30\%SPR}$ and MFMT reference points (Figure 24b-c) much closer than estimated by the IBM. The declining trend across interim years, seen in the results of the IBM, was likewise influenced by the estimated lower recruitment and the absence of any potential index data to fit.

According to the IBM, the southeastern U.S. Yellowtail Snapper population is not overfished nor experiencing overfishing and the population is estimated around one-and-a-half times the minimum stock size threshold (MSST). The age-4 fishing mortality rates rose above the MFMT from 1993 – 1995, declined below the MFMT through 2001, and remained variable but stable through 2020. The estimated spawning stock biomass remained above the MSST throughout the timeseries, but was below the target $SSB_{F30\%SPR}$ from 1995 – 2000. The trend in spawning stock biomass generally increased from 1996 – 2016 but then began to gradually decline through 2020. Status designation of this stock has not changed since the S64 benchmark (terminal year 2017) nor since the first assessment (Muller et al. 2003).

The constant fishing mortality rate projections updated in this interim analysis differed only slightly from what was provided to both South Atlantic and Gulf SSCs in November 2020, as those projections included preliminary landings and discards data for 2018 – 2020. When future fishing mortality rates were equal to $F_{30\%SPR}$, the updated annual yield and spawning stock biomass streams projected for 2021 – 2025 were on average 115,000 lbs. lower and 26,000 lbs. smaller, respectively, than those provided in November 2020. Similarly, when future fishing mortality rates were held constant at the value associated with P* = 0.375, the updated yield and spawning stock biomass streams were on average 21,000 lbs. lower and 120,000 lbs. smaller, respectively, for the same period.

Constant catch projection scenarios are appealing for management as they may result in greater market stability and consistency in regulations. However, the 3- and 5-year scenarios were shown to lead to rapid declines in spawning stock biomass and fishing mortality rates which far exceed the MFMT after only a few years into the projections. When catch was held constant at the equilibrium yield at $F_{30\%SPR}$, the population was above and below the spawning stock and fishing mortality thresholds, respectively, through the end of the short-term projections (2021 – 2031). While the catch levels projected for the 3- and 5-year scenarios were similar to those estimated for the mid-1990s and from 2013 – 2018, they were also supported by increases in estimated recruitment, especially from 2012 – 2014. Therefore, these projections suggest that if recruitment is lower (i.e., closer to an historic average and an assumption in these projections), the catch levels for the 3- and 5- years scenarios may not be sustainable long-term to the population and would be largely dependent on continued increased levels of recruitment.

There were numerous caveats to these projection methods, including assumptions of recruitment in future years, the population scale in the terminal year (2020), as well as unchanging fleet selectivities, fleet allocations, growth, stock-recruit parameters, and fixed quantities in the IBM. Perhaps some of the most influential assumptions in short-term projections are that recruitment remains near the historic average and the MRIP CPUE index is representative of relative stock abundance (due to the sizeable effect the MRIP CPUE has on population scale in recent years and the trend in stock abundance).

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6 Figures

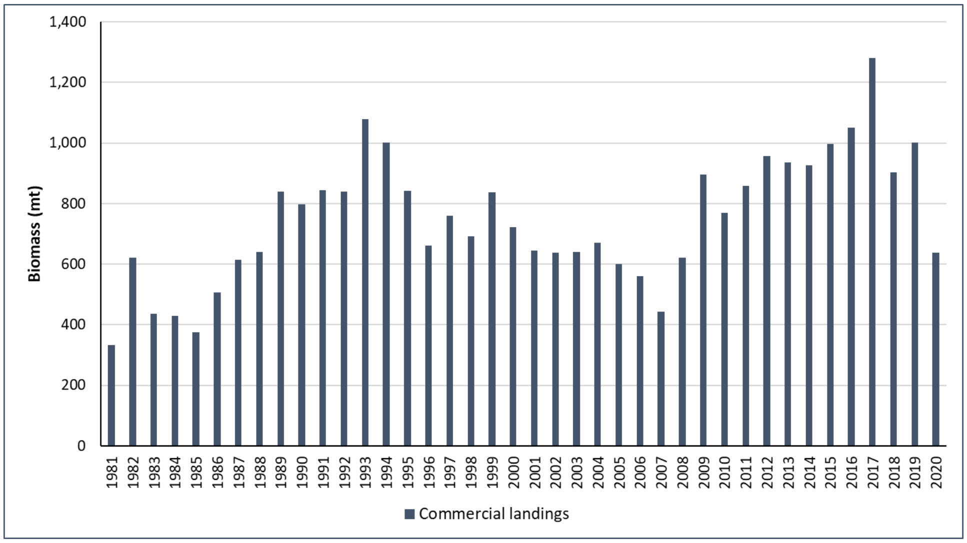

Figure 1. Southeastern U.S. Yellowtail Snapper commercial landings (metric tons) in Florida waters from 1981 – 2020.

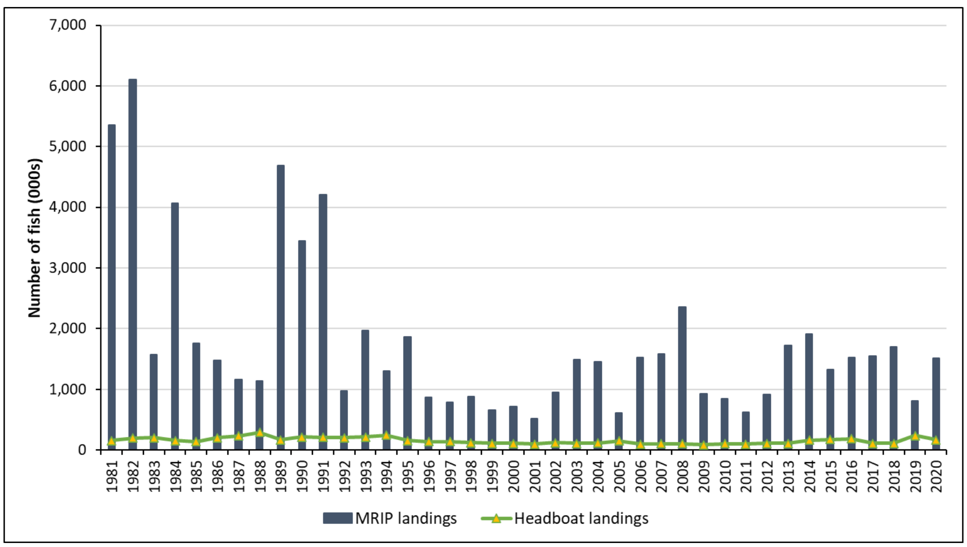

Figure 2. Southeastern U.S. Yellowtail Snapper recreational landings (thousands of fish) from the Marine Recreational Information Program (MRIP; navy bars) and the Southeast Region Headboat Survey (Headboat; green line with yellow triangles) in Florida waters from 1981 – 2020.

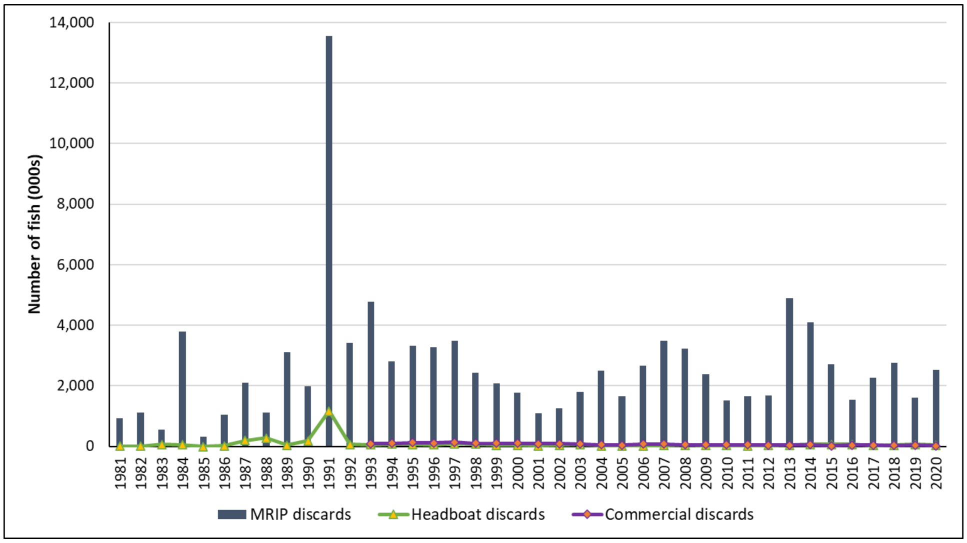

Figure 3. Southeastern U.S. Yellowtail Snapper commercial discards (thousands of fish; purple line with orange diamonds) and recreational discards (thousands of fish) from the Marine Recreational Information Program (MRIP; navy bars) and the Southeast Region Headboat Survey (Headboat; green line with yellow triangles) in Florida waters from 1981 – 2020.

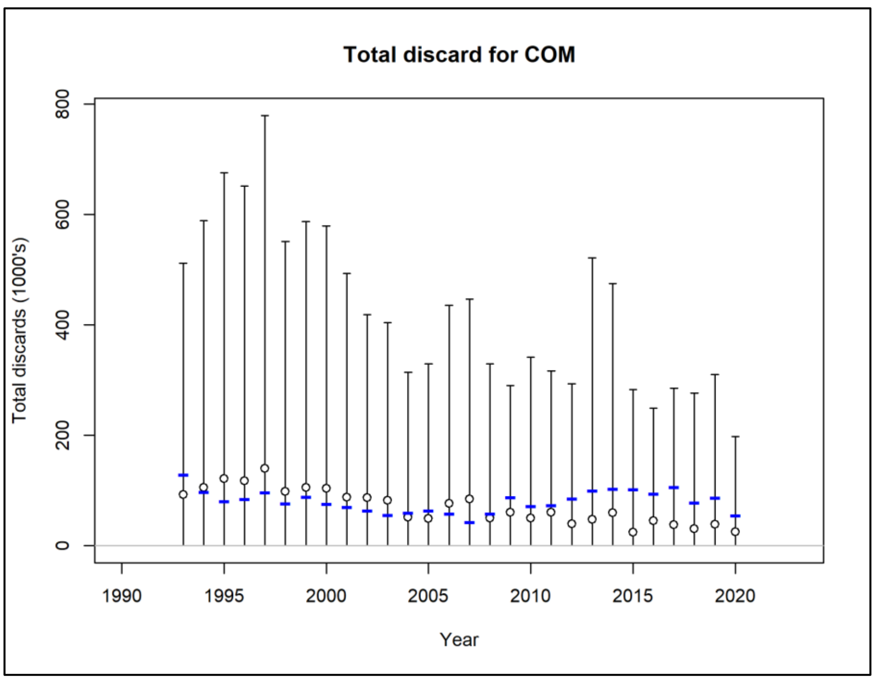

Figure 4. Southeastern U.S. Yellowtail Snapper observed (dots with 95% confidence intervals) and expected (blue dashes) discards (i.e., before applying the discard mortality rate for each fleet) by the commercial fleet in thousands of fish for the Interim Base Model.

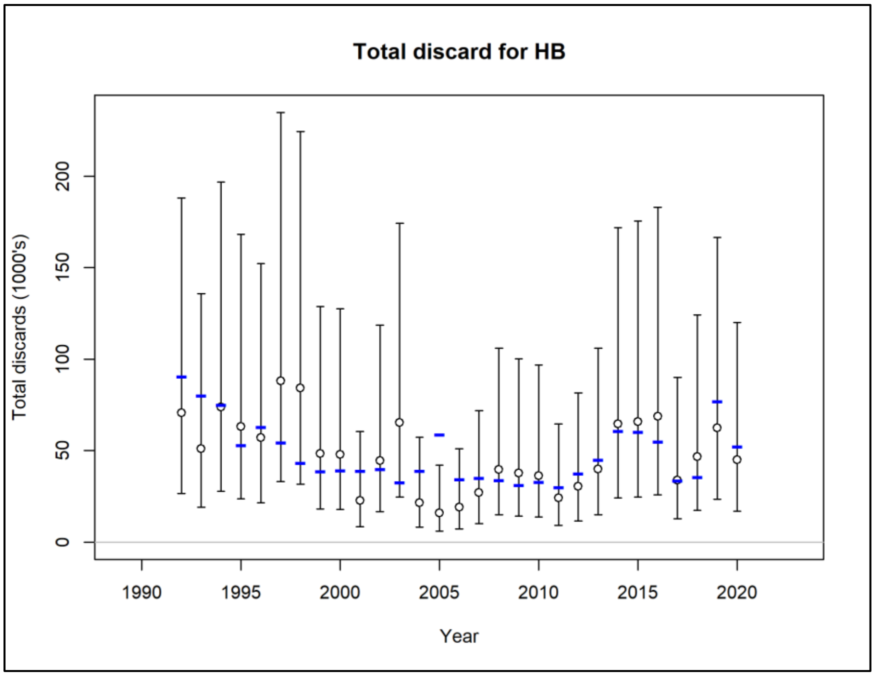

Figure 5. Southeastern U.S. Yellowtail Snapper observed (dots with 95% confidence intervals) and expected (blue dashes) discards (i.e., before applying the discard mortality rate for each fleet) by the headboat fleet in thousands of fish for the Interim Base Model.

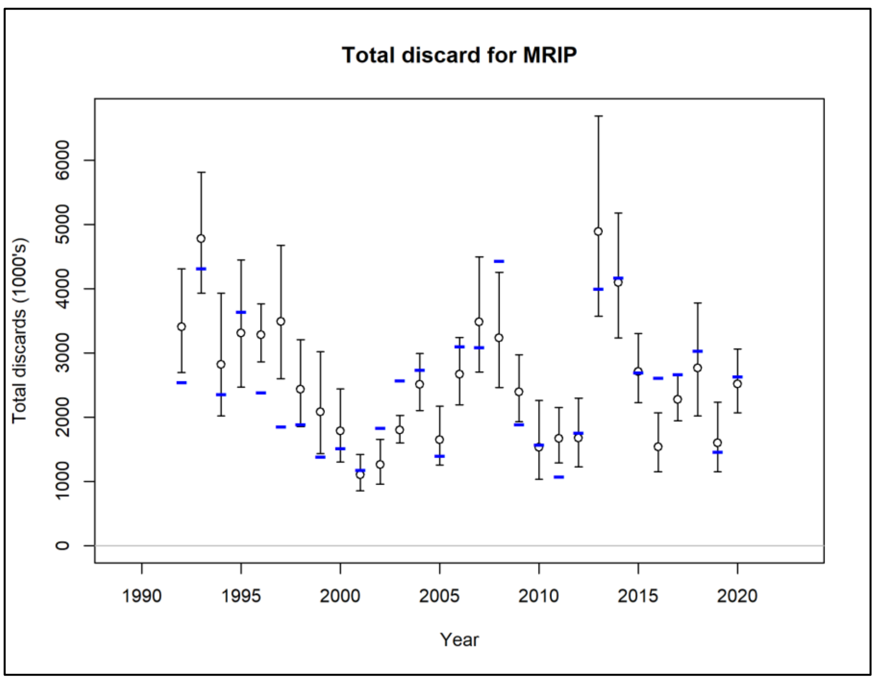

Figure 6. Southeastern U.S. Yellowtail Snapper observed (dots with 95% confidence intervals) and expected (blue dashes) discards (i.e., before applying the discard mortality rate for each fleet) by the MRIP fleet in thousands of fish for the Interim Base Model.

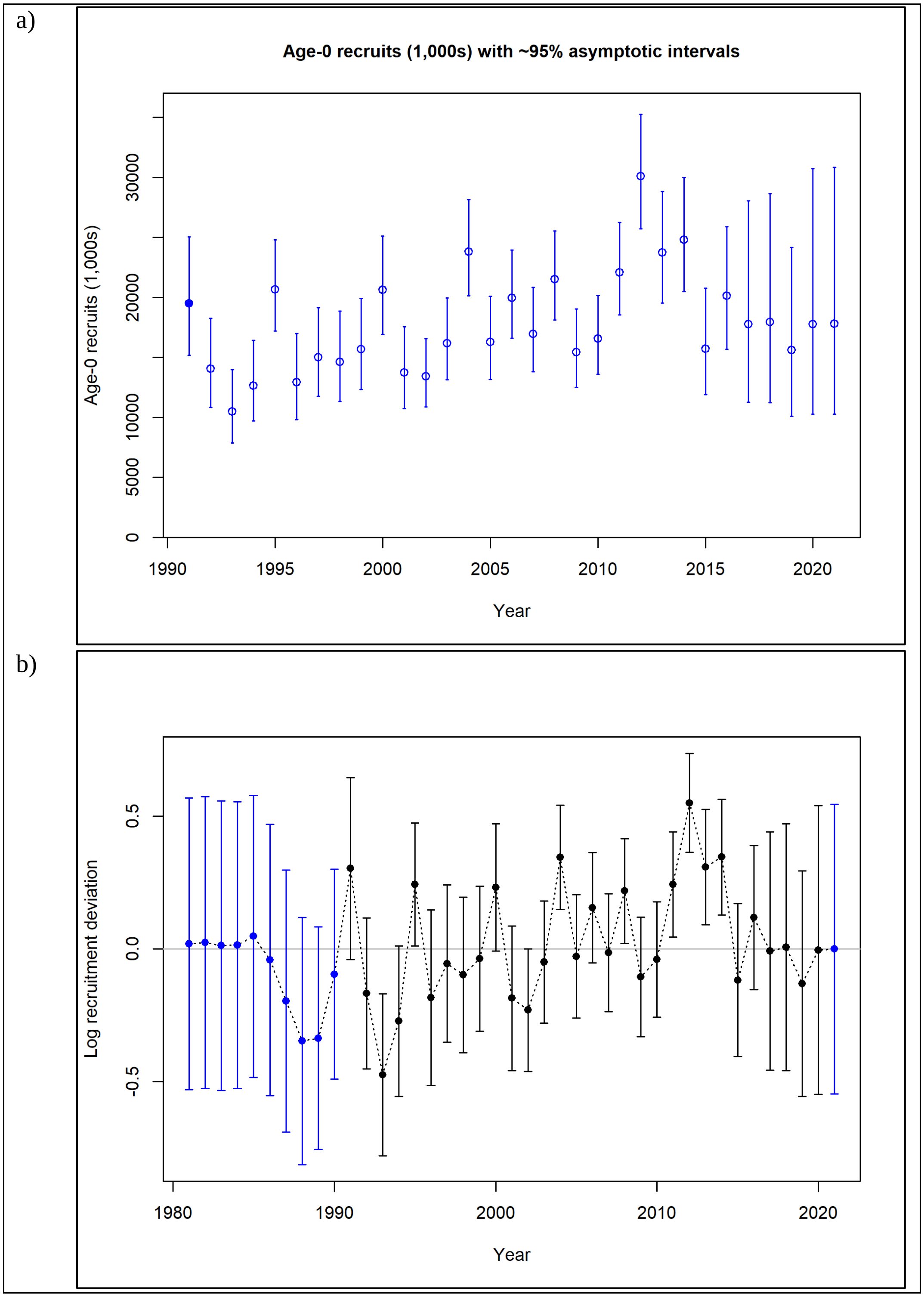

Figure 7. Southeastern U.S. Yellowtail Snapper a) estimated age-0 recruitment (blue/black dots) with 95% confidence intervals (blue/black lines) and b) log recruitment deviations (1981 – 2020). The blue dots and lines indicate when early recruitment deviations were estimated (1981 – 1990) while the black dots and lines indicate when the main recruitment deviations were estimated (1991 – 2020).

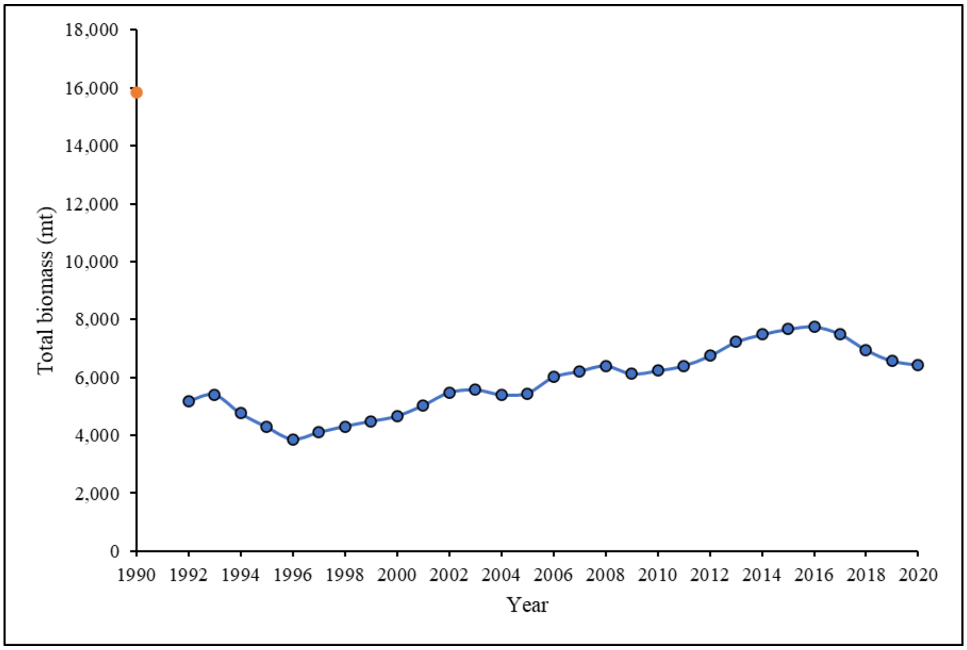

Figure 8. Estimates of total biomass (in metric tons) of southeastern U.S. Yellowtail Snapper (blue circles) from 1992 – 2020. The solid orange circle is the estimated unfished equilibrium biomass.

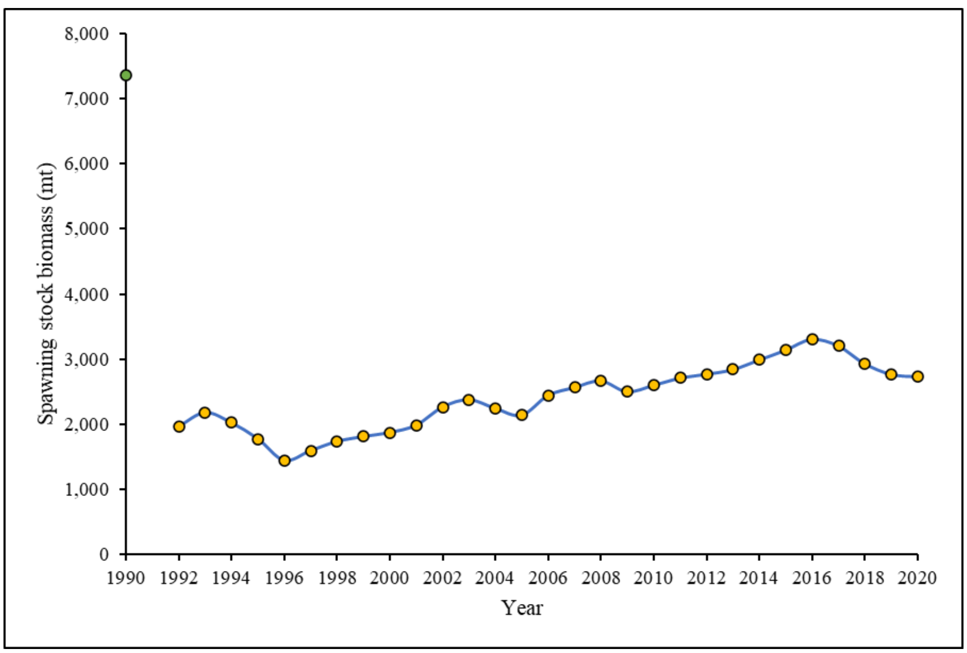

Figure 9. Estimates of spawning stock biomass (in metric tons) of southeastern U.S. Yellowtail Snapper (yellow circles) from 1992 – 2020. The solid green circle is the estimated unfished spawning stock biomass.

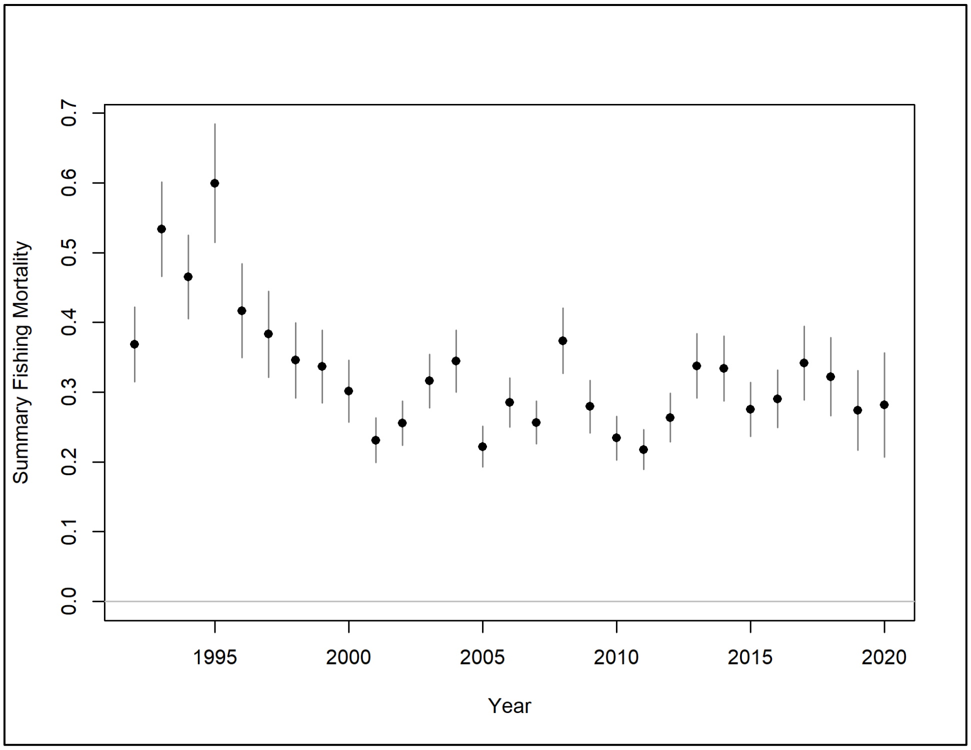

Figure 10. Annual instantaneous fishing mortality rates for age-4 southeastern U.S. Yellowtail Snapper with 95% confidence intervals for the Interim Base Model.

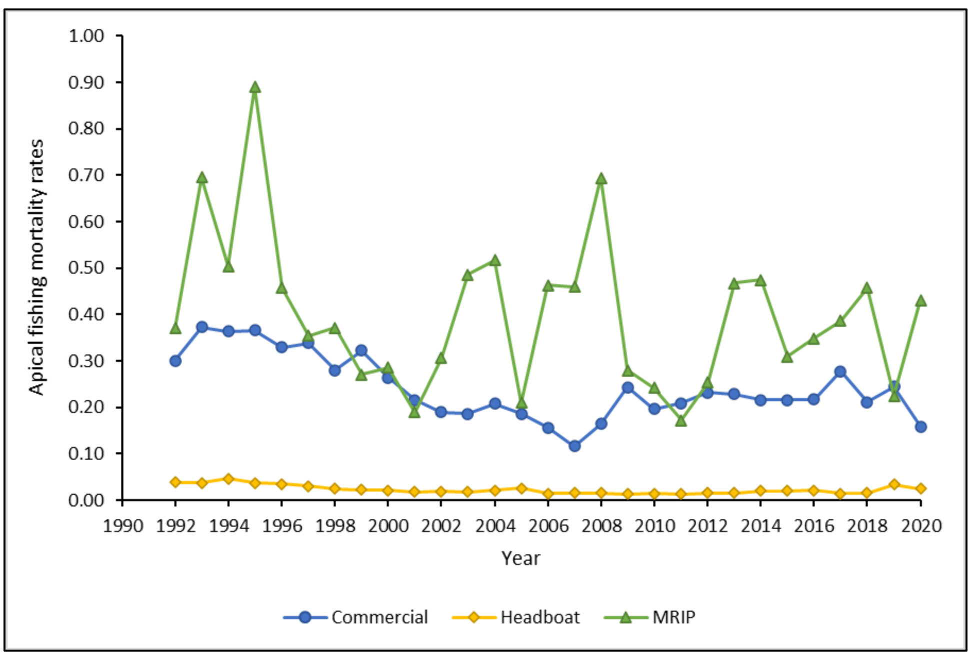

Figure 11. Annual fleet-specific instantaneous apical fishing mortality rates for southeastern U.S. Yellowtail Snapper for the Interim Base Model. This represents the instantaneous fishing mortality level on the most vulnerable age class for each fleet.

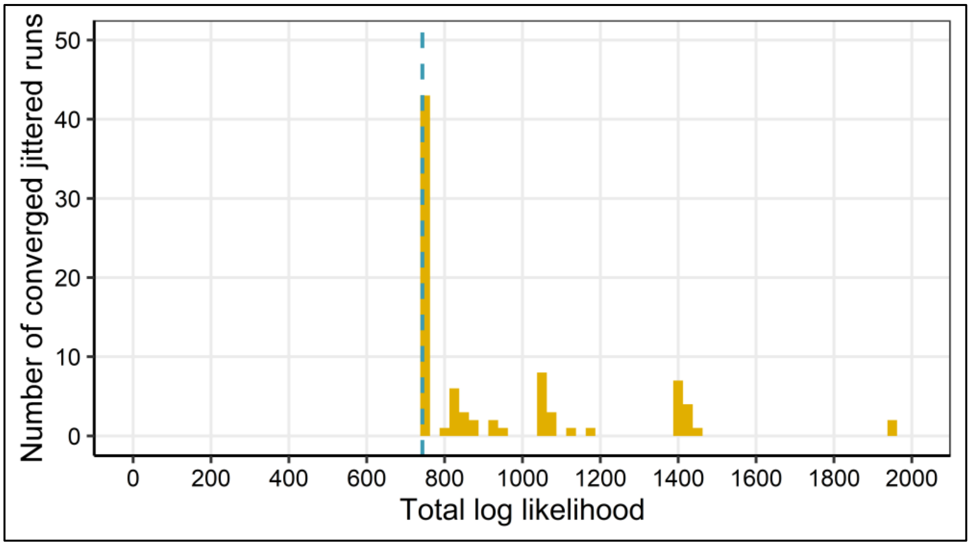

Figure 12. Total log-likelihood values from converged runs found by the jitter analysis (yellow bars) and the Interim Base Model (blue dashed line).

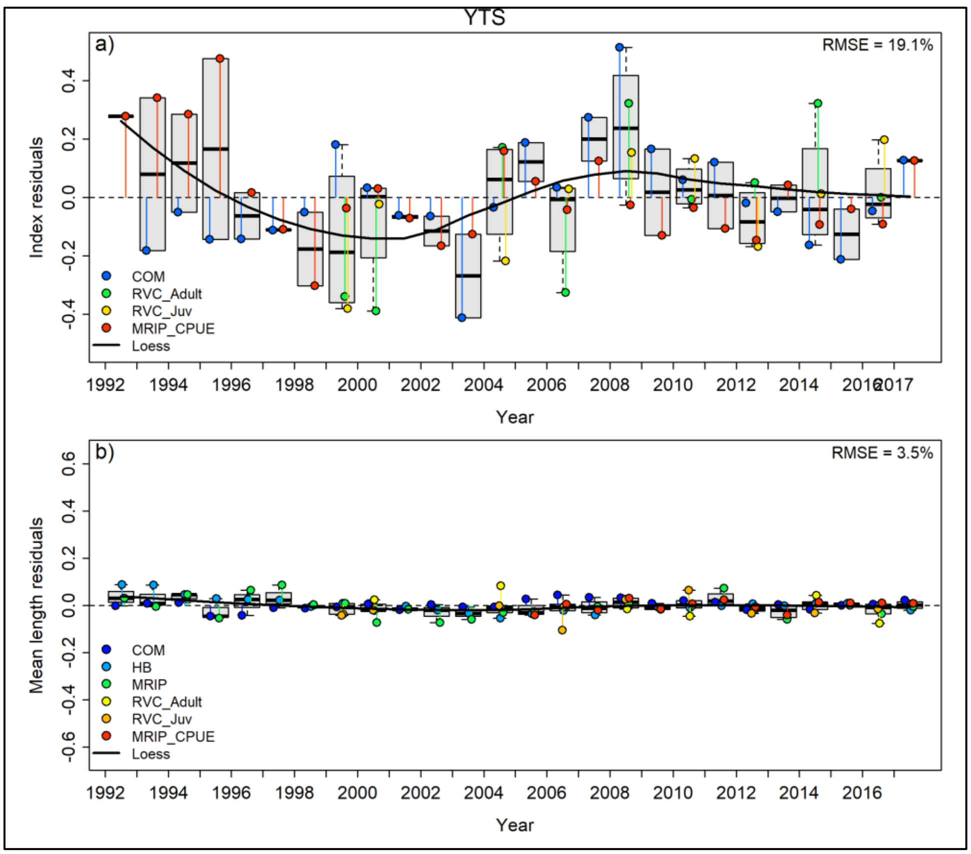

Figure 13. Joint residual plots for a) the indices of abundance and b) the annual mean length estimates of available fleets and indices from the Interim Base Model. Vertical lines with points show the residuals, boxplots show residual medians and quantiles, and solid black lines are a loess smoother. Root-mean squared errors (RMSE) are included in the upper right-hand corner of each plot.

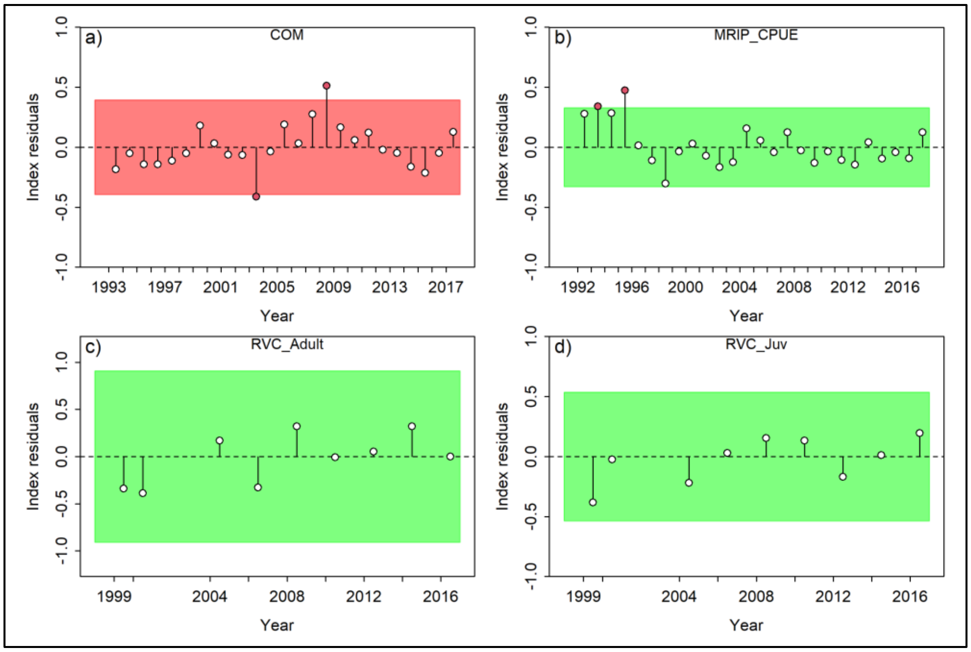

Figure 14. Runs tests results for the indices of abundance from the Interim Base Model. Green shading indicates no evidence ($p \geq 0.05$) and red shading evidence ($p < 0.05$) to reject the hypothesis of a randomly distributed time-series of residuals, respectively. Shaded regions span three residual standard deviations to either side from zero and red points outside of the shading indicate a violation of that 'three-sigma limit'.

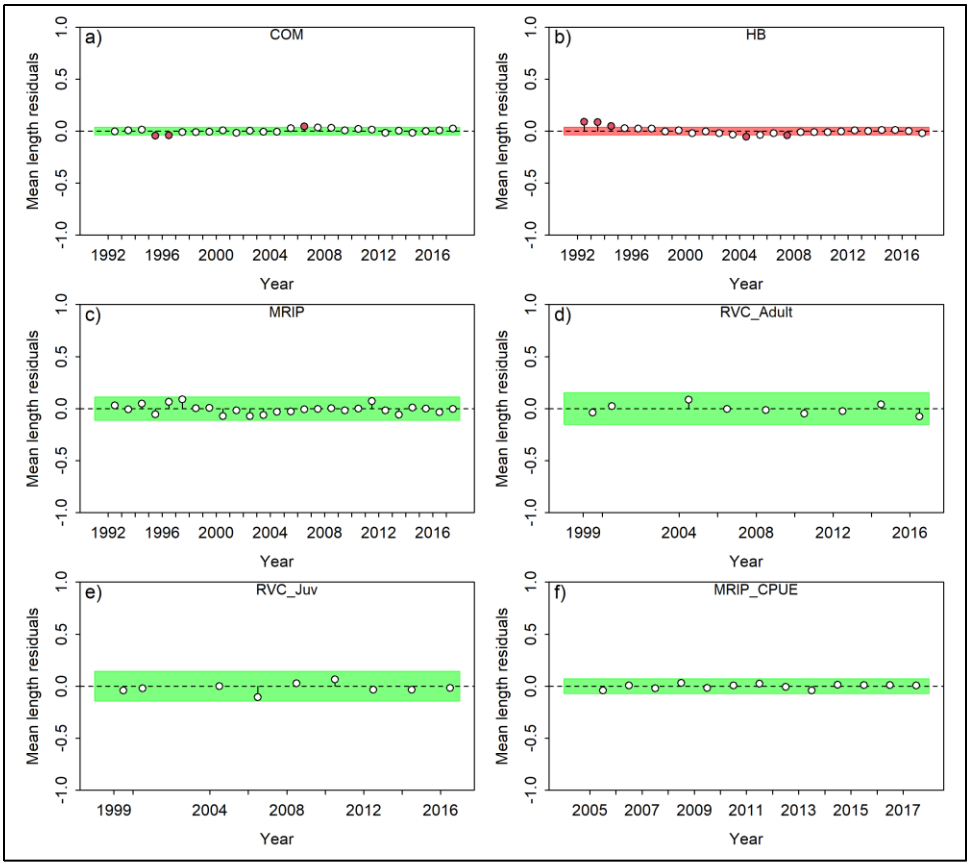

Figure 15. Runs tests results for the annual mean length estimates from the Interim Base Model. Green shading indicates no evidence ($p \geq 0.05$) and red shading evidence ($p < 0.05$) to reject the hypothesis of a randomly distributed time-series of residuals, respectively. Shaded regions span three residual standard deviations to either side from zero and red points outside of the shading indicate a violation of that 'three-sigma limit'.

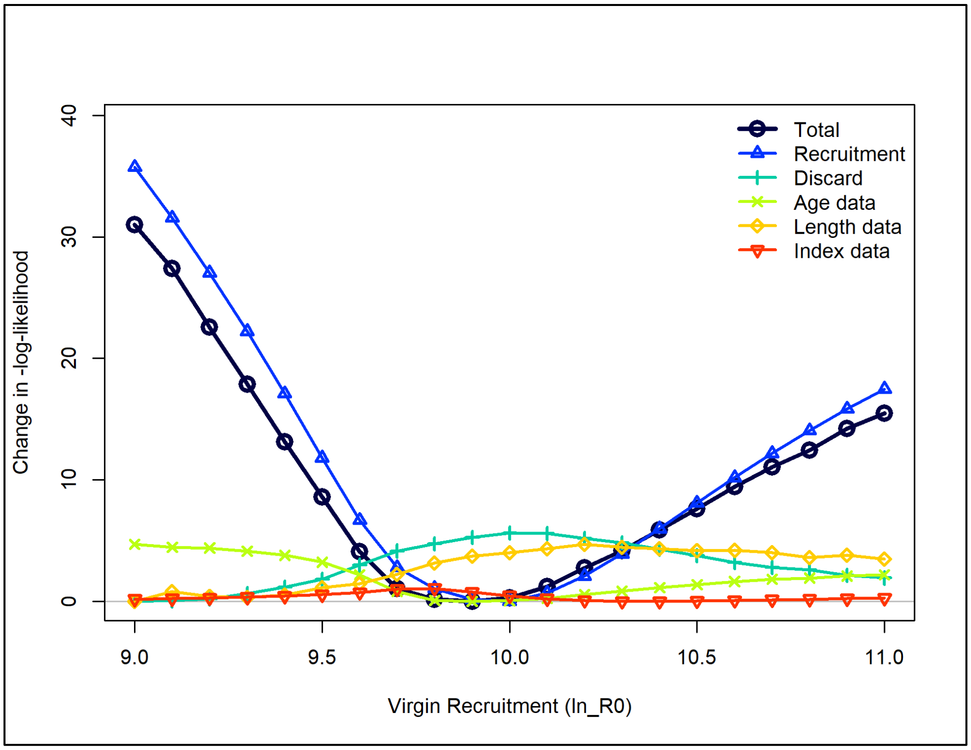

Figure 16. Log-likelihood profiles for the unfished (i.e., virgin) recruitment (R0) parameter for various data components in the Interim Base Model.

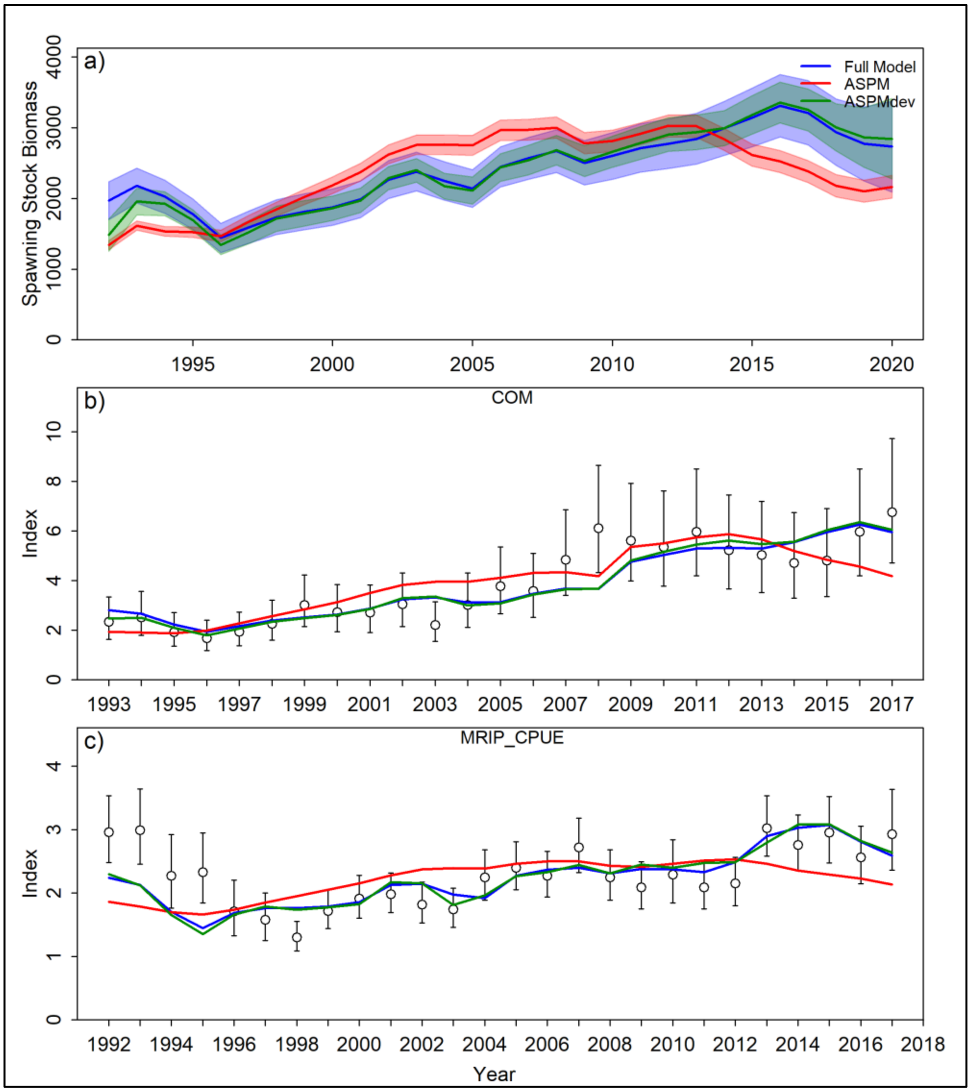

Figure 17. Results comparison between the Interim Base Model (Full Model), the deterministic Age-Structured-Production Model (ASPM), and the ASPM with recruitment deviations (ASPMdev) showing a) spawning stock biomass and b) – c) observed and predicted values for the commercial CPUE and MRIP CPUE indices.

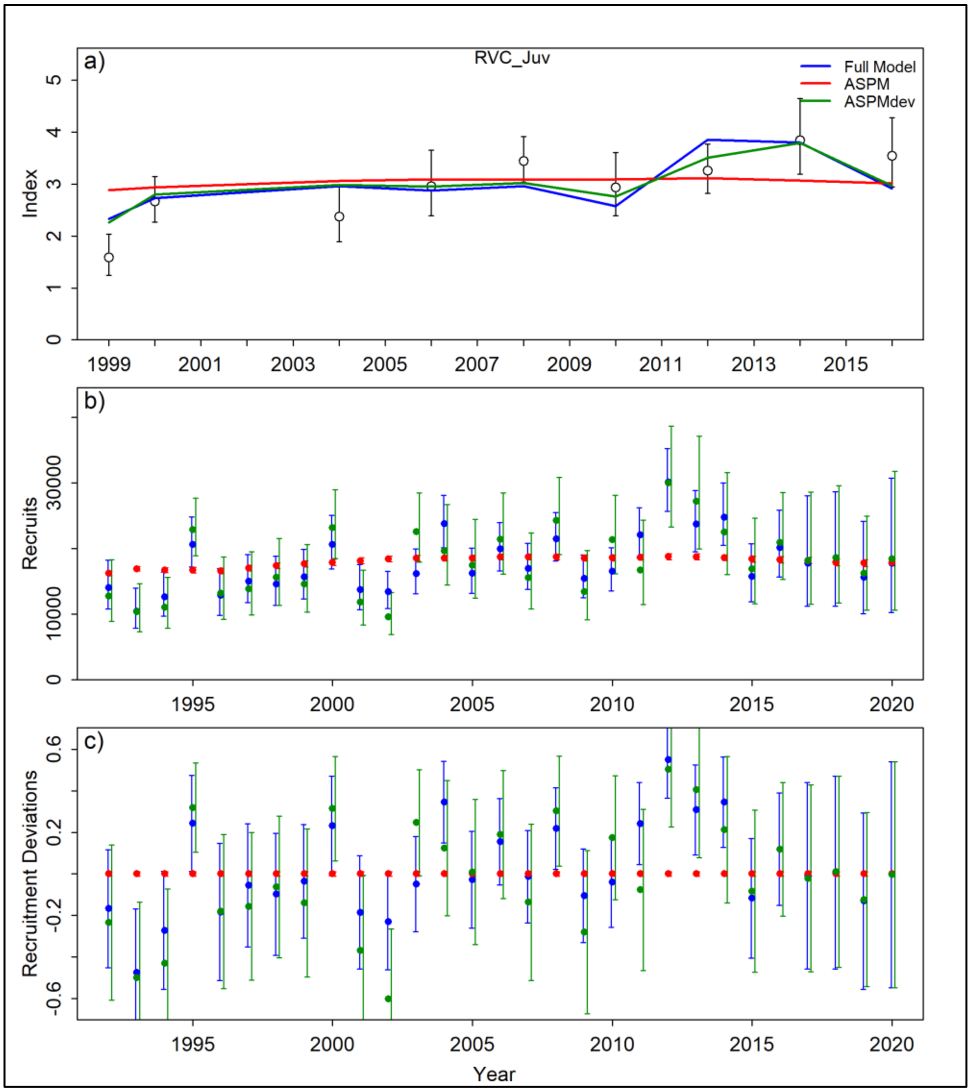

Figure 18. Results comparison between the Interim Base Model (Full Model), the deterministic Age-Structured-Production Model (ASPM), and the ASPM with recruitment deviations (ASPMdev) showing a) observed and predicted values for the RVC juvenile index, b) estimates of age-0 recruitment, and c) recruitment deviation estimates.

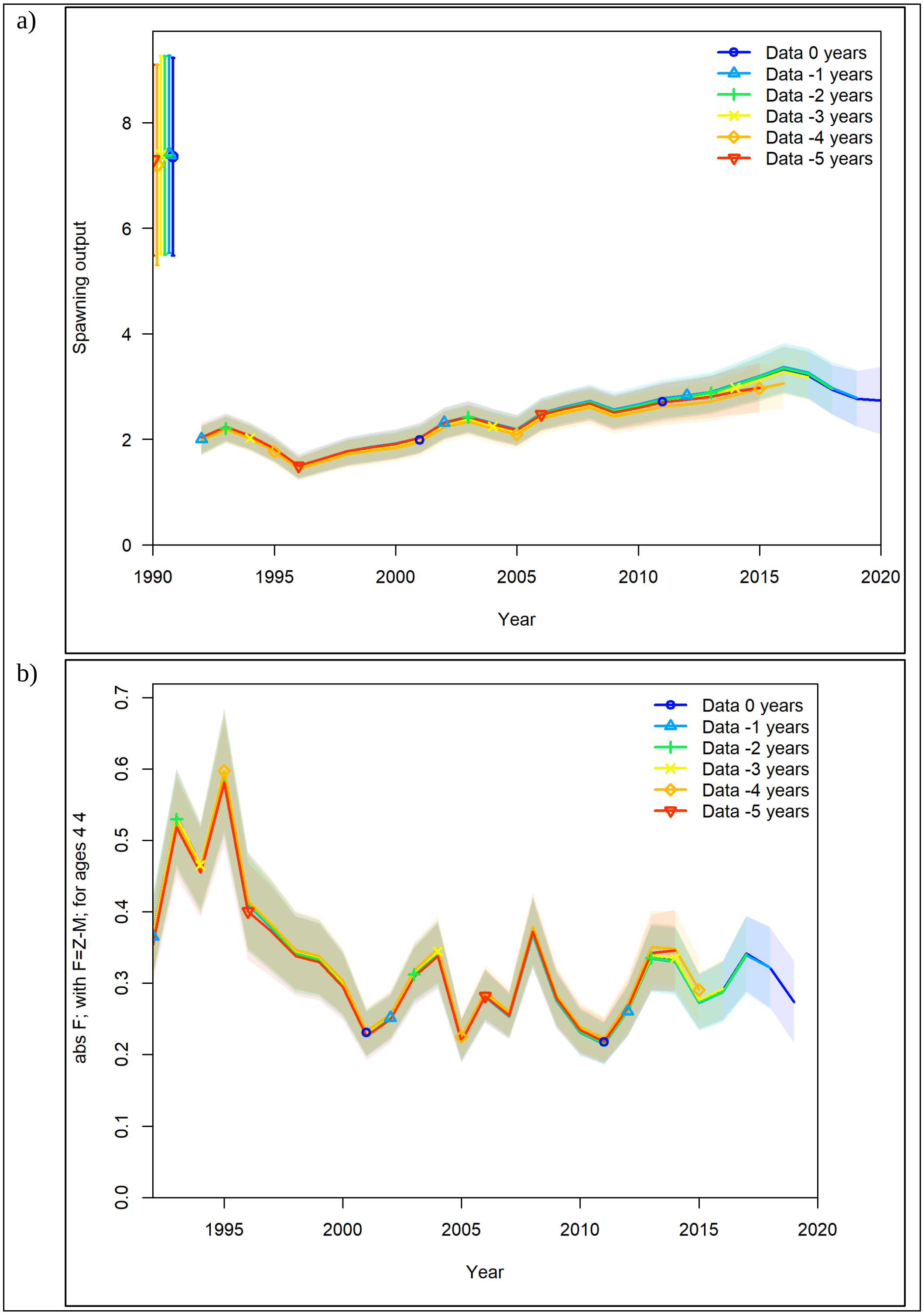

Figure 19. Results of a five-year retrospective analysis for a) spawning stock biomass and b) age-4 fishing mortality from the Interim Base Model.

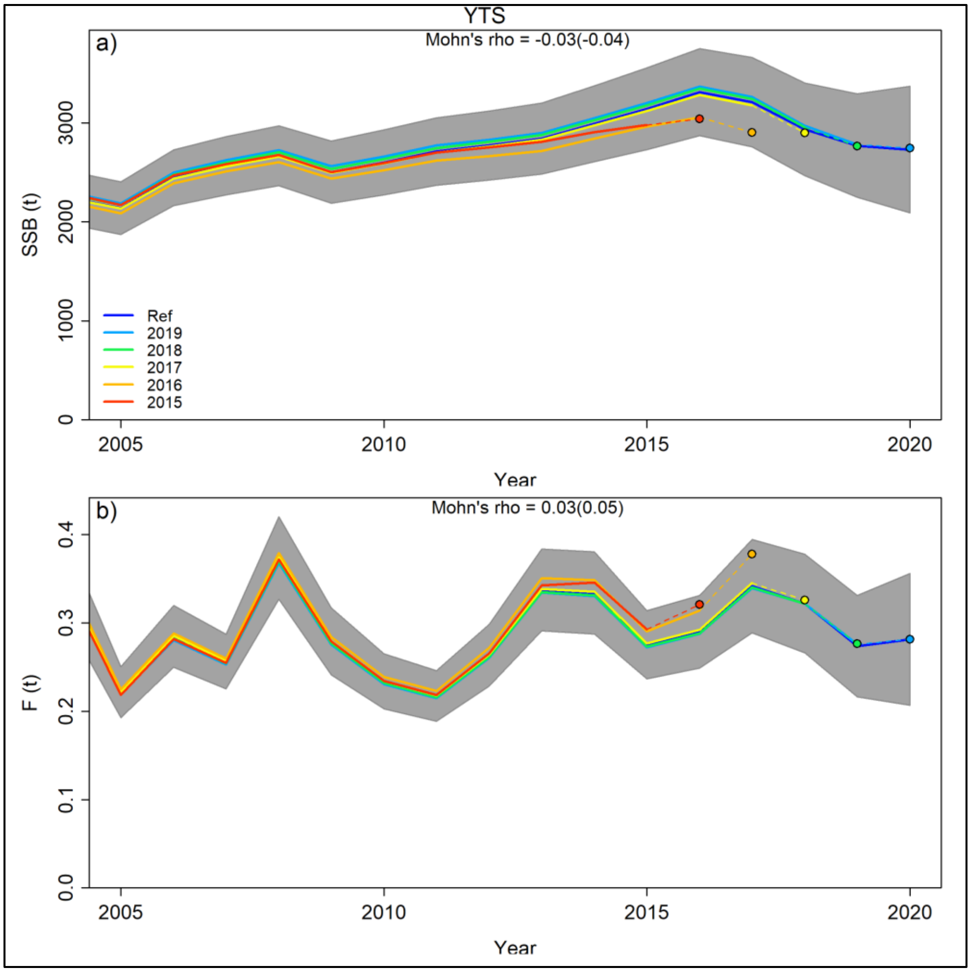

Figure 20. Retrospective forecast results of a) spawning stock biomass and b) age-4 fishing mortality conducted by re-fitting the Interim Base Model (Ref) after sequentially removing five years of observations. The Mohn's rho (ρ_M) statistic and corresponding forecast rho (ρ_F) values (in parenthesis) are provided at the top of each panel. One-year-ahead projections denoted by color-coded dashed lines with terminal points are shown for each peel. Grey shaded areas are the 95 % confidence intervals from the Interim Base Model.

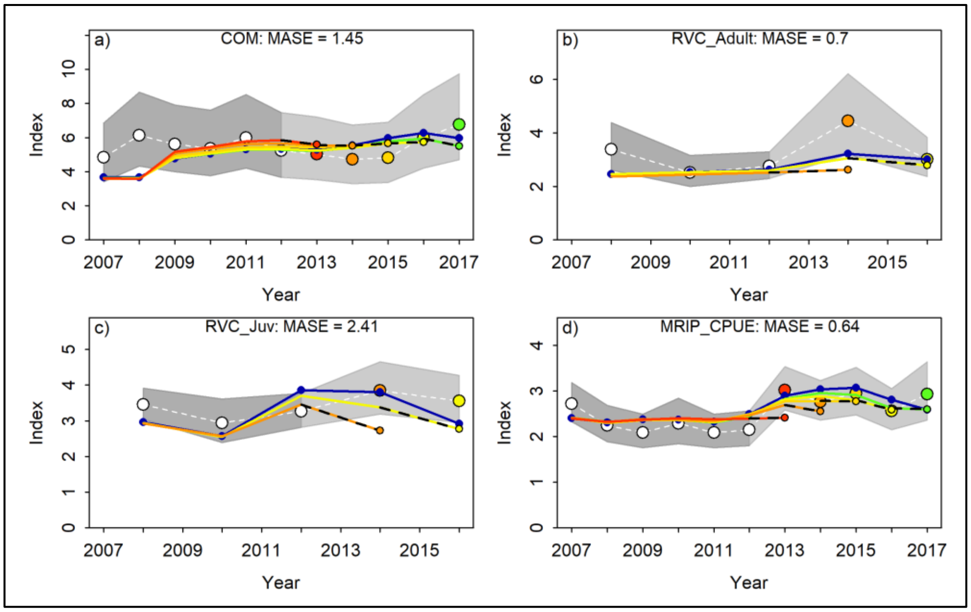

Figure 21. Hindcasting cross-validation results for the a) Commercial CPUE, b) RVC Adult, c) RVC Juvenile, and d) MRIP CPUE indices from the Interim Base Model showing observed (large white points connected with dashed line), fitted (solid lines), and one-year ahead forecast values (small terminal points). The color-coded solid circles are the observations used for cross-validation and the light-gray shaded area is the associated 95 % confidence intervals. The mean absolute scaled error (MASE) scores for each index and length composition series are provided at the top of each panel.

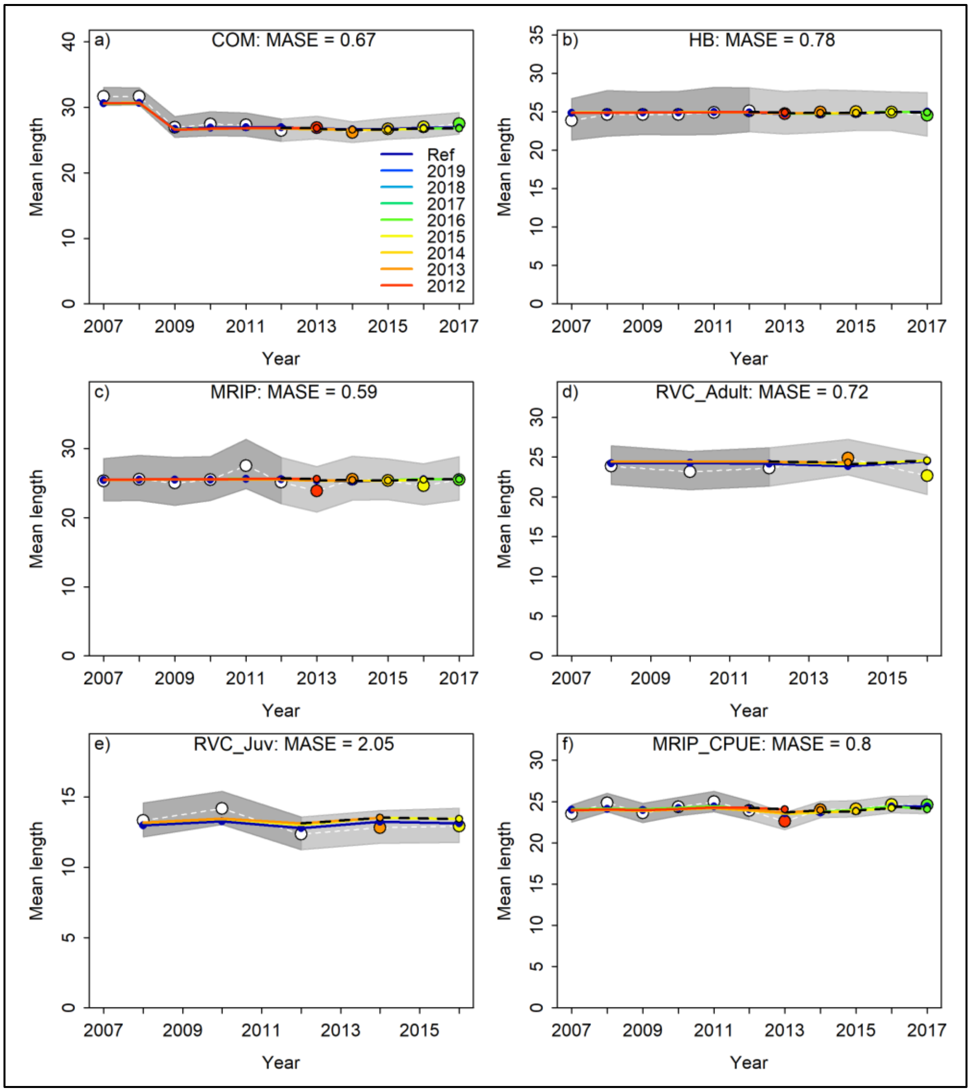

Figure 22. Hindcasting cross-validation results for a) Commercial, b) Headboat, c) MRIP, d) RVC Adult, e) RVC Juvenile, and f) MRIP CPUE annual mean length estimates from the Interim Base Model showing observed (large white points connected with dashed line), fitted (solid lines), and one-year ahead forecast values (small terminal points). The color-coded solid circles are the observations used for cross-validation and the light-gray shaded area is the associated 95 % confidence intervals. The mean absolute scaled error (MASE) scores for each index and length composition series are provided at the top of each panel.

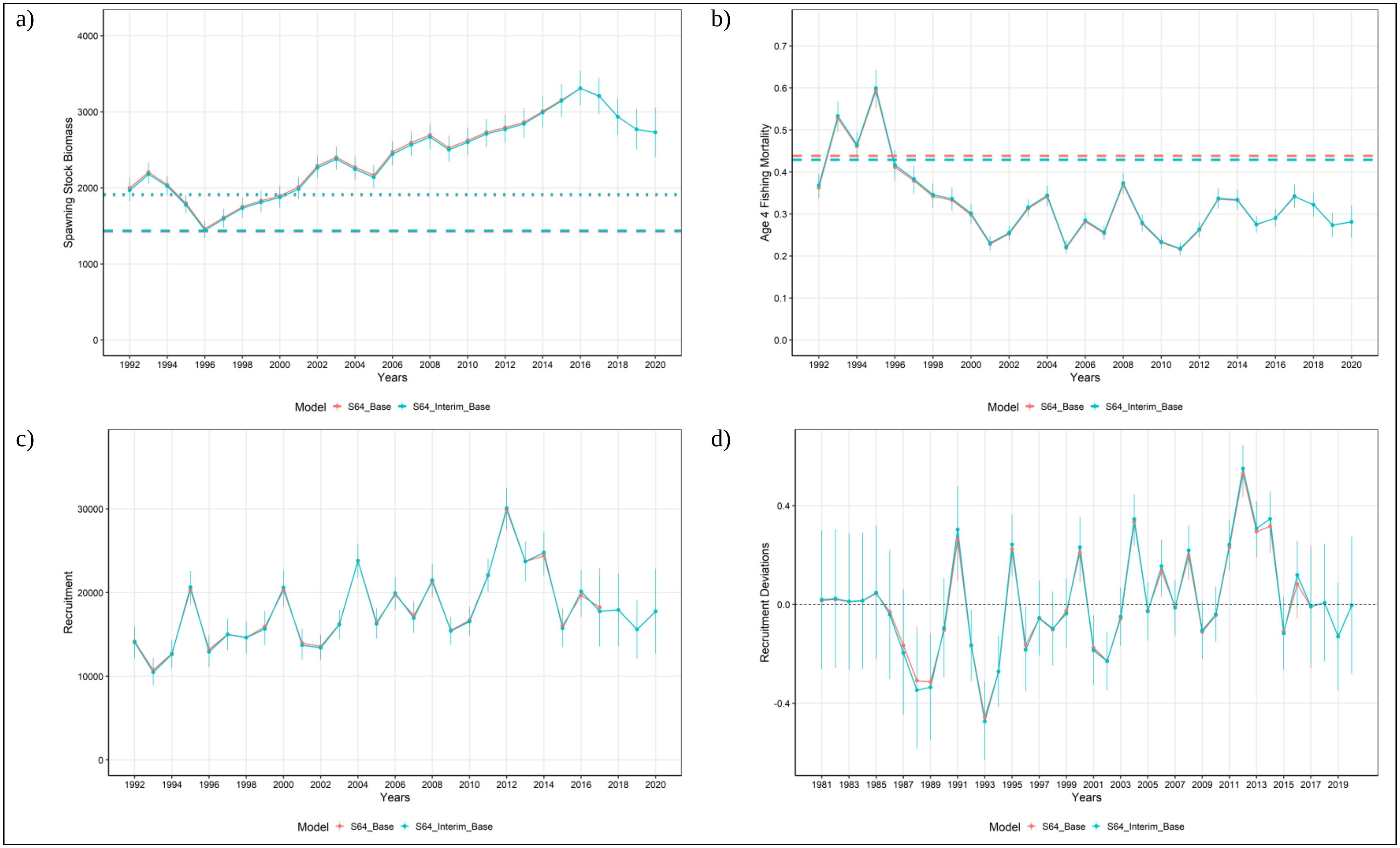

Figure 23. A comparison of the results between the Interim Base Model and the SEDAR 64 base model for a) spawning biomass shown relative to MSST (dashed line) and SSB at F30%SPR (dotted line), b) age-4 fishing mortality shown relative to MFMT (dashed line), c) age-0 recruitment, and d) estimated recruitment deviations.

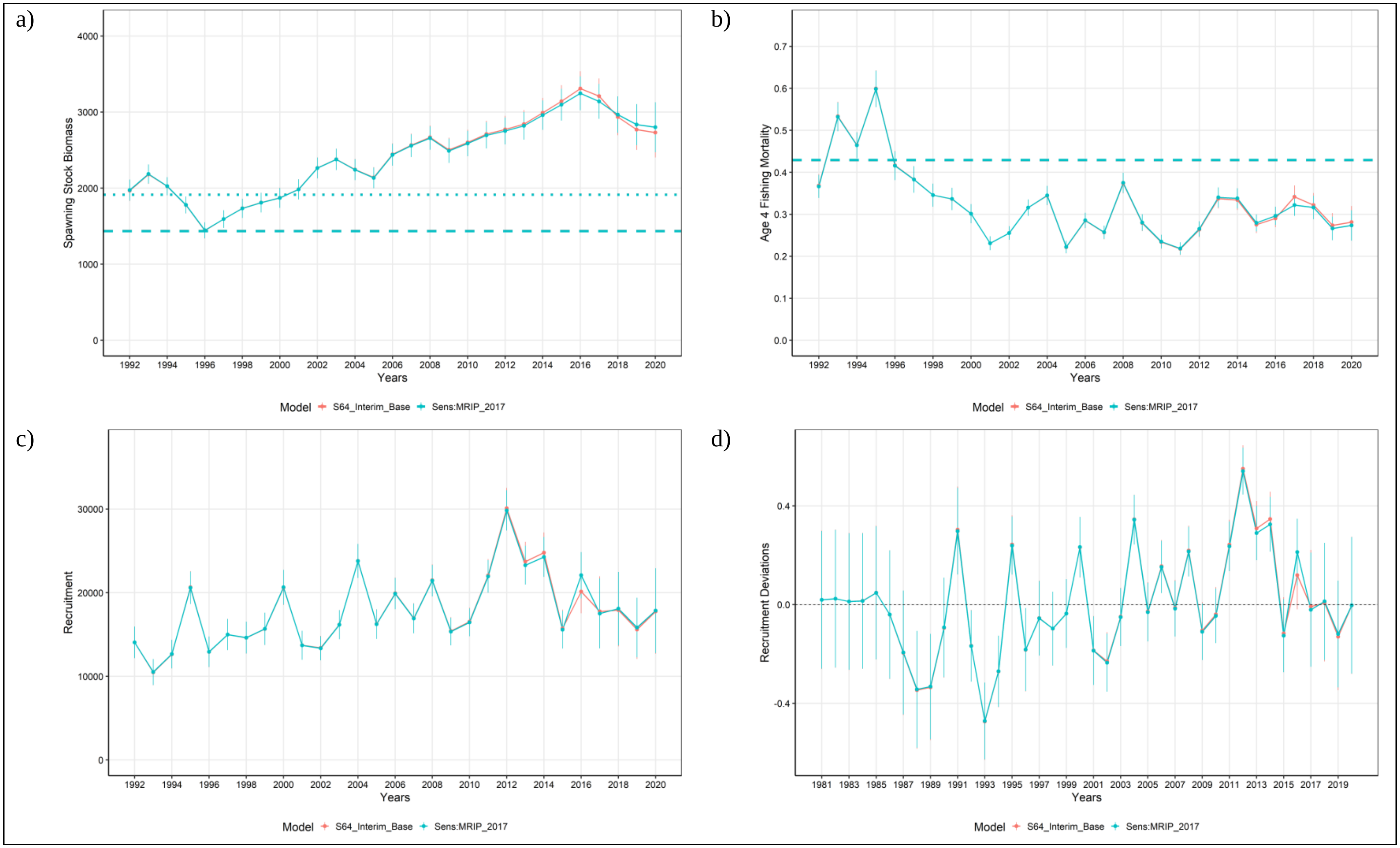

Figure 24. A comparison of the Interim Base Model to a sensitivity run which altered the 2017 MRIP data in the southwest region of Florida for a) spawning biomass shown relative to MSST (dashed line) and SSB at F30%SPR (dotted line), b) age-4 fishing mortality shown relative to MFMT (dashed line), c) age-0 recruitment, and d) estimated recruitment deviations.

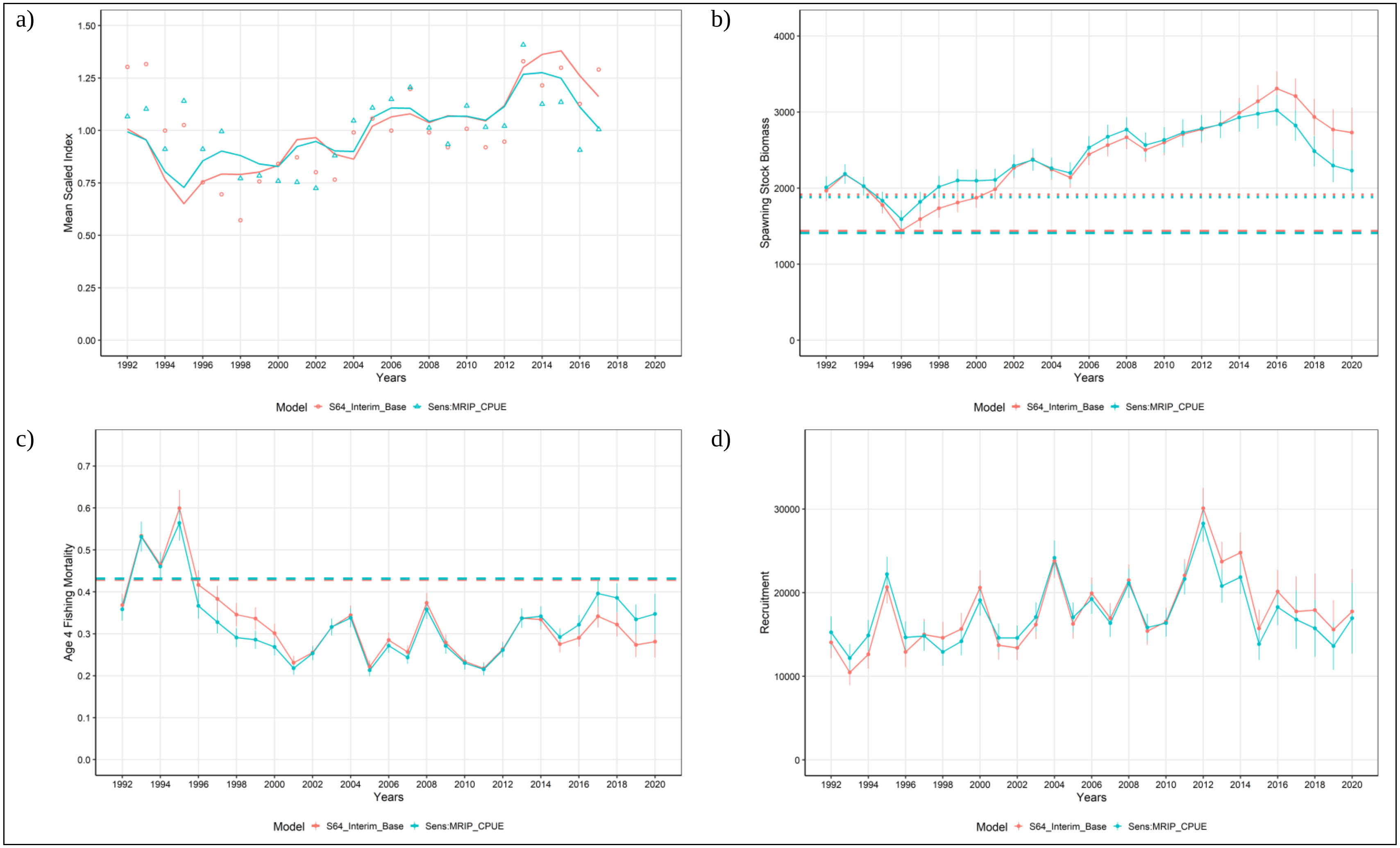

Figure 25. A comparison of the Interim Base Model to a sensitivity run which adjusted the 'total catch per trip' MRIP CPUE index with a) the two indices shown standardized to their means, b) spawning biomass shown relative to MSST (dashed line) and SSB at F30%SPR (dotted line), c) age-4 fishing mortality shown relative to MFMT (dashed line), and d) age-0 recruitment.

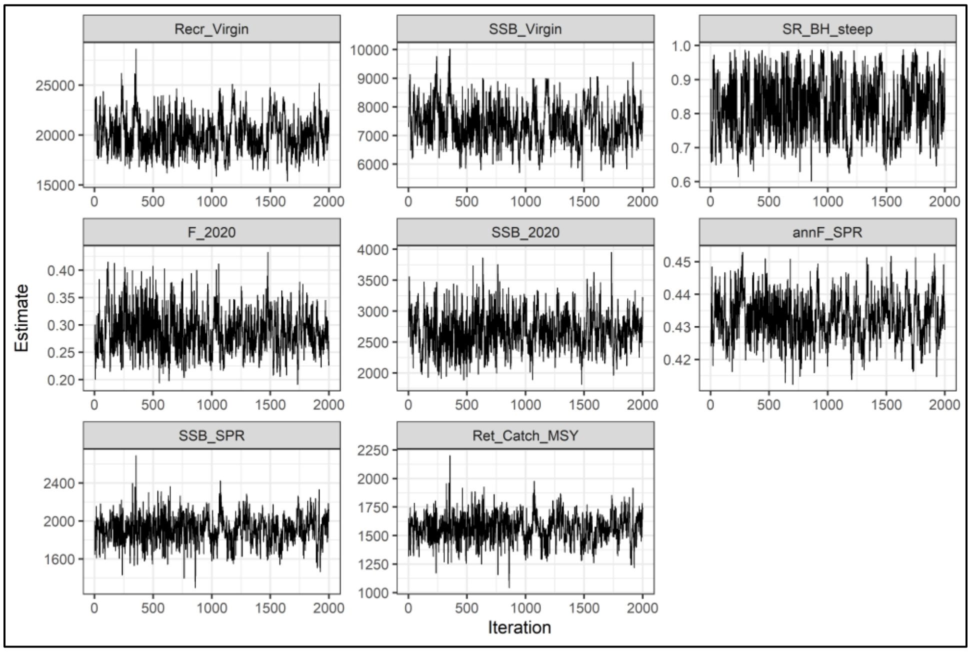

Figure 26. Trace plots of the first MCMC chain for selected parameters and derived quantities from the Interim Base Model.

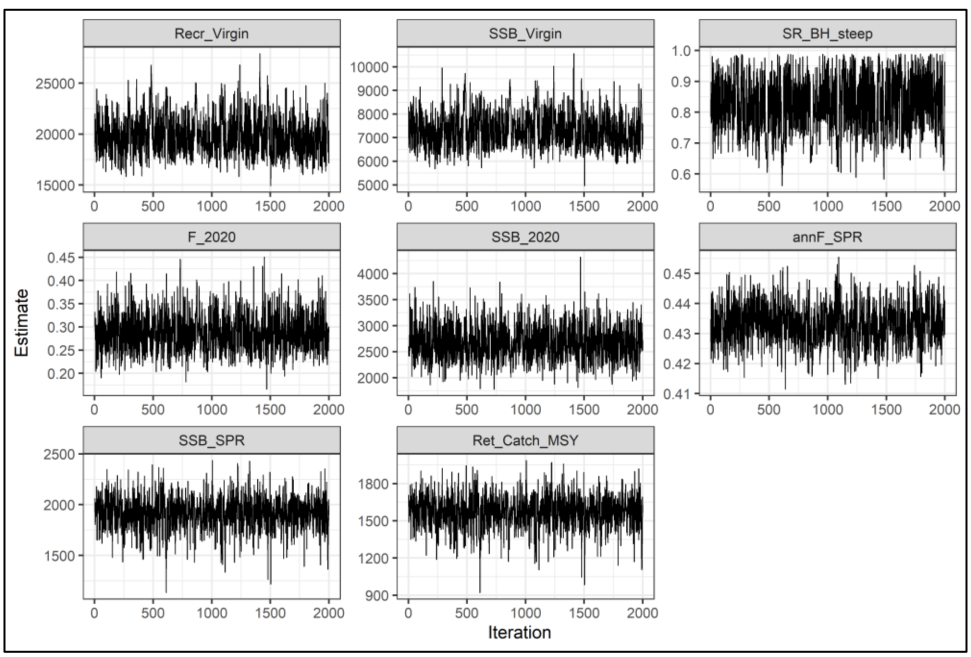

Figure 27. Trace plots of the second MCMC chain for selected parameters and derived quantities from the Interim Base Model.

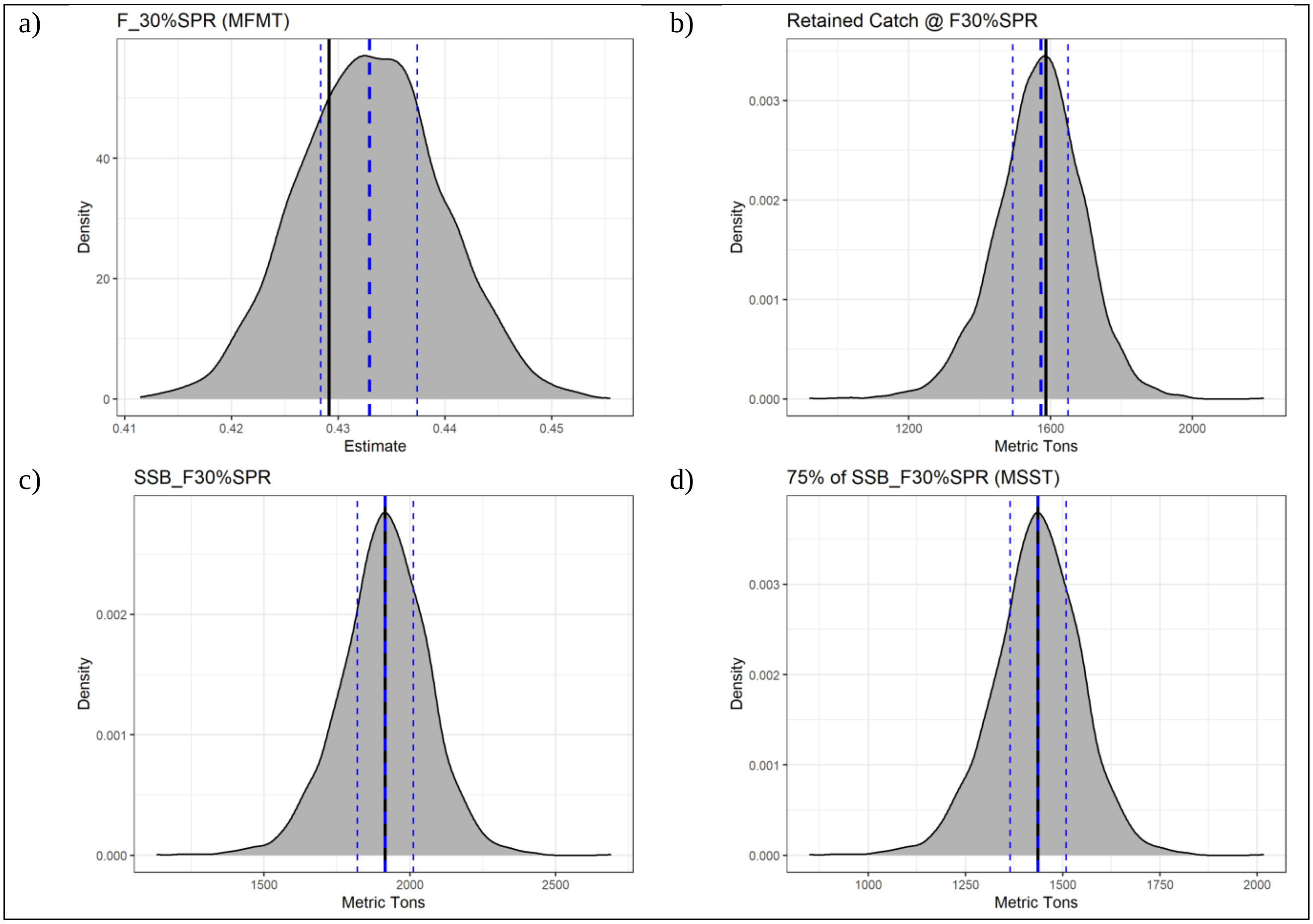

Figure 28. The posterior distribution for a) $F_{30\%SPR}$ (i.e., MFMT), and b) the retained catch associated with $F_{30\%SPR}$, c) SSB at $F_{30\%SPR}$, and d) 75% of SSB at $F_{30\%SPR}$ (i.e., MSST) from the combined two-chain MCMC. The blue dashed lines indicate the median and interquartile range while the solid black line is the estimate from the Interim Base Model.

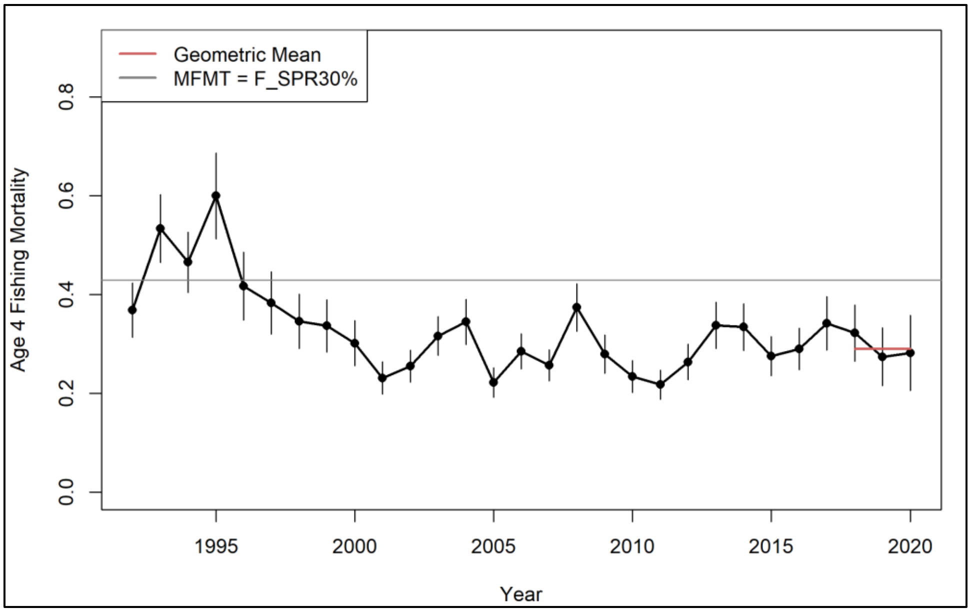

Figure 29. Annual estimates of age-4 fishing mortality relative to MFMT (grey solid line). The geometric mean of fishing mortality in the last three years ($F_{current}$) is shown in red. Vertical lines represent approximate symmetric 95% confidence intervals.

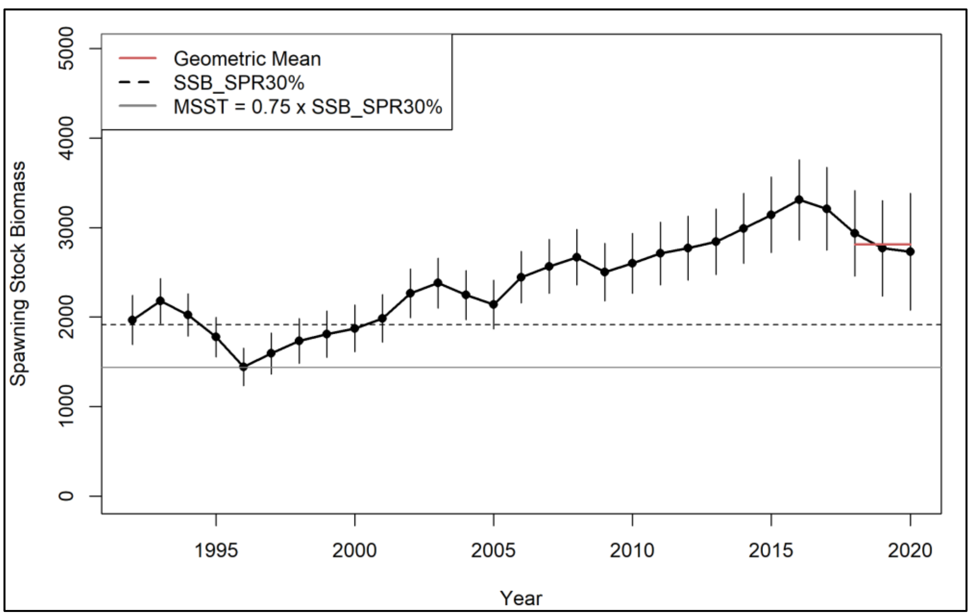

Figure 30. Annual estimates of spawning stock biomass (SSB) relative to MSST (grey solid line) and $SSB_{F30\%SPR}$ (black dashed line). The geometric mean of SSB in the last three years ($SSB_{current}$) is shown in red. Vertical lines represent approximate symmetric 95% confidence intervals.

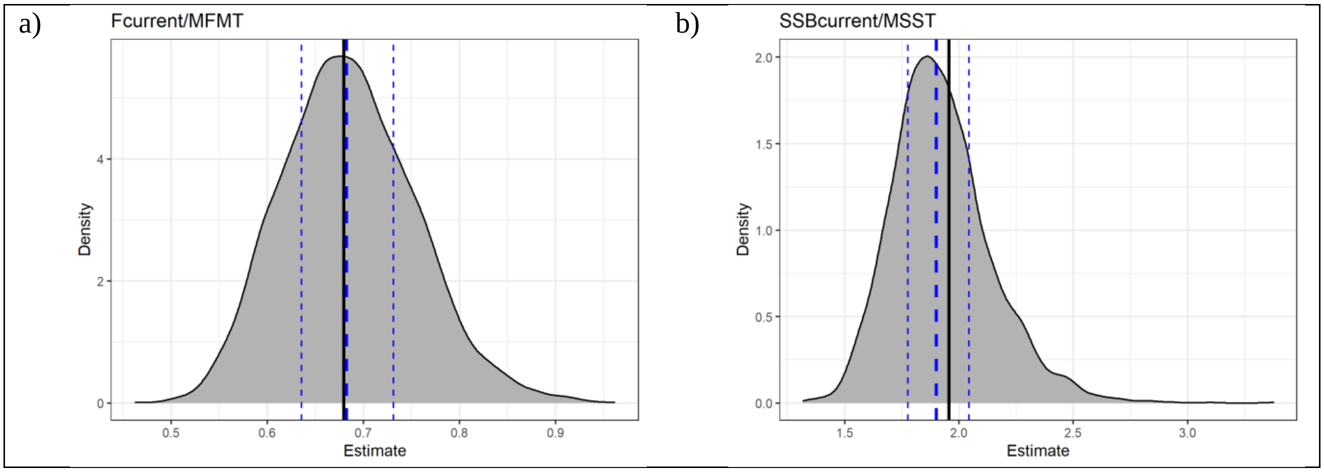

Figure 31. The posterior distribution for the F ratio ($F_{current}$/MFMT) and SSB ratio ($SSB_{current}$/MSST) values from the combined two-chain MCMC. The blue dashed lines indicate the median and interquartile range while the solid black line is the estimate from the Interim Base Model.

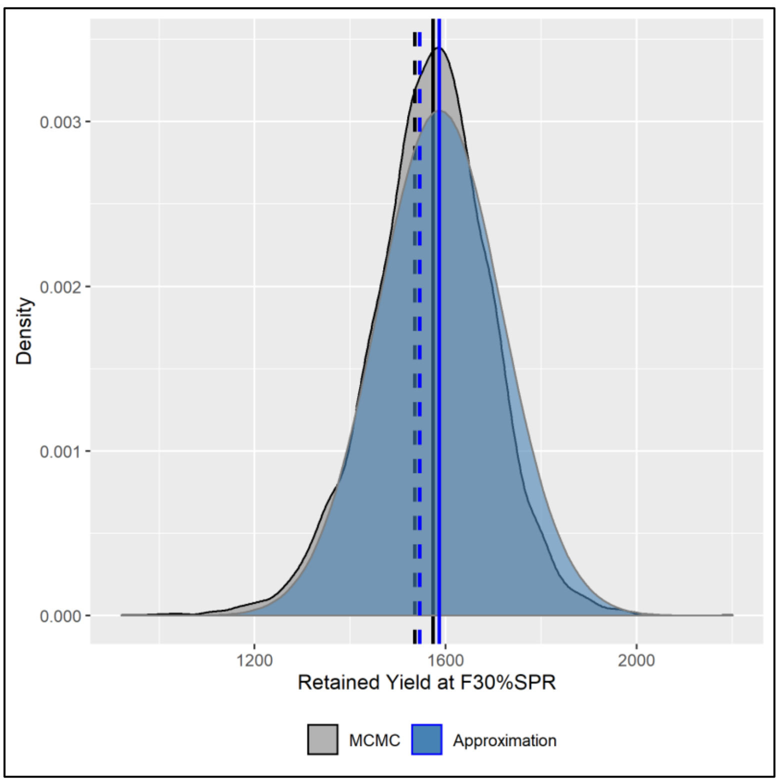

Figure 32. A comparison between the MCMC distribution of the equilibrium OFL (retained yield at $F_{30\%SPR}$; grey) and an approximate normal distribution of the OFL with a mean and standard error estimated by the IBM (blue). The medians and 37.5th quantiles are shown by the solid and dashed lines, respectively.

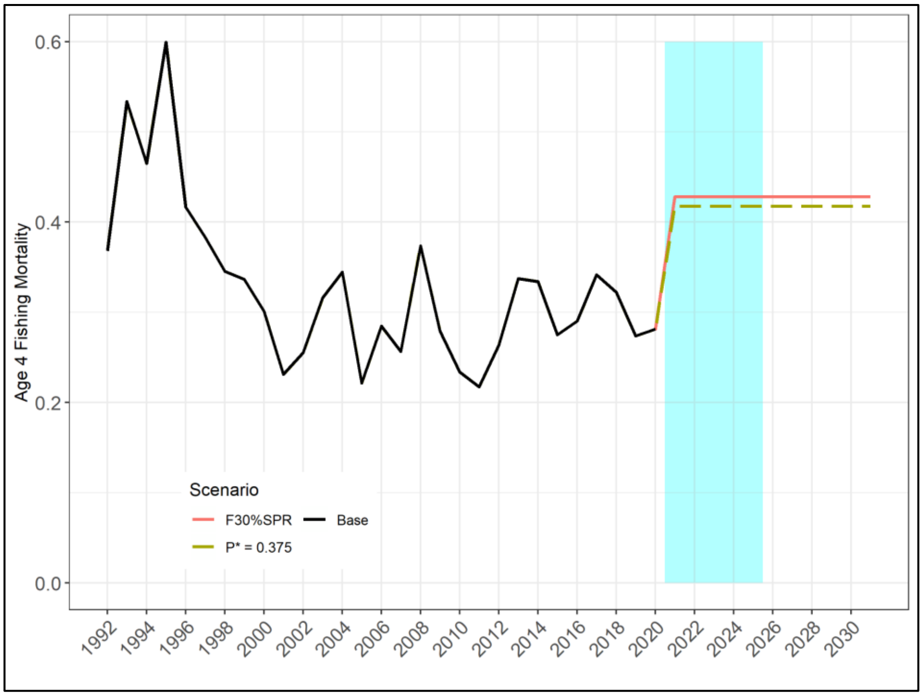

Figure 33. Age-4 Fishing mortality values under $F_{30\%SPR}$ (red line), the level that corresponds to a P* value of 0.375 (green line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

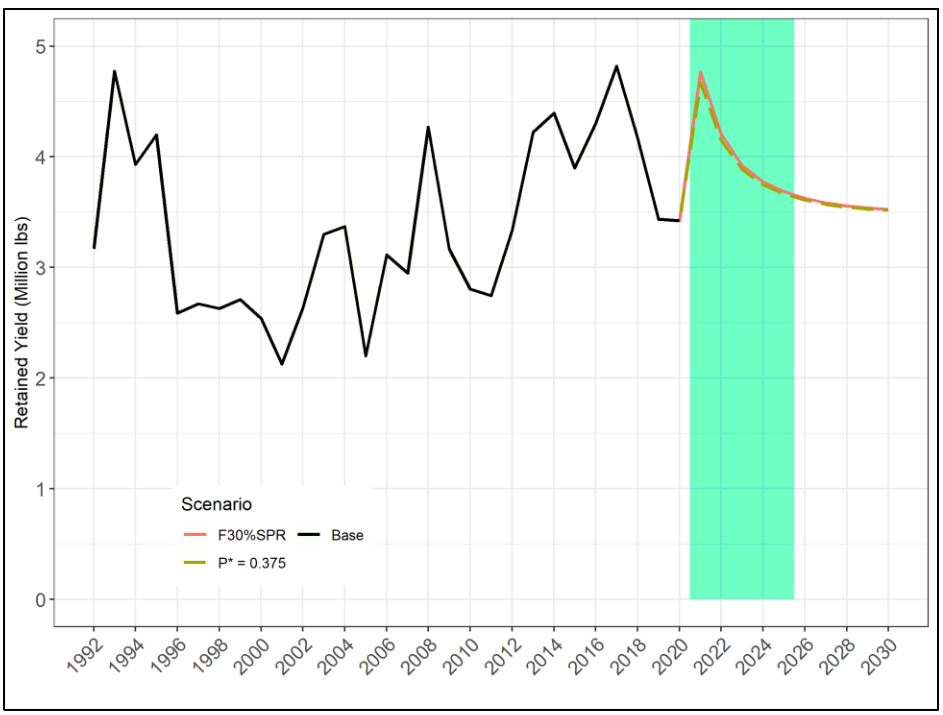

Figure 34. Retained yield (million pounds) associated with fishing at $F_{30\%SPR}$ (red line), the level that corresponds to a P* value of 0.375 (green line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

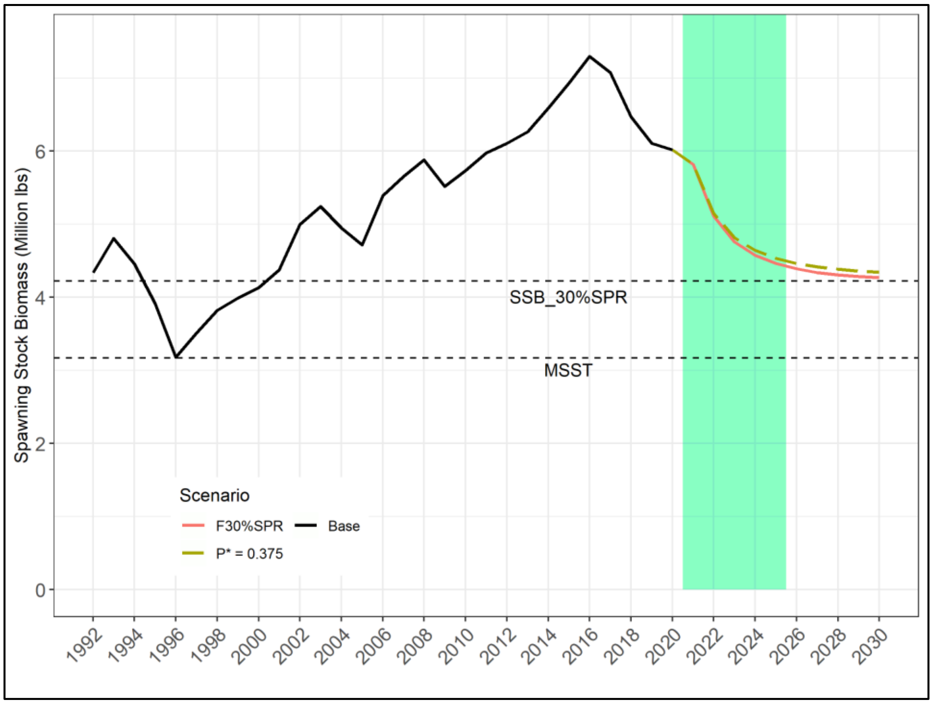

Figure 35. Spawning stock biomass (million pounds) associated with fishing at $F_{30\%SPR}$ (red line), the level that corresponds to a P* value of 0.375 (green line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

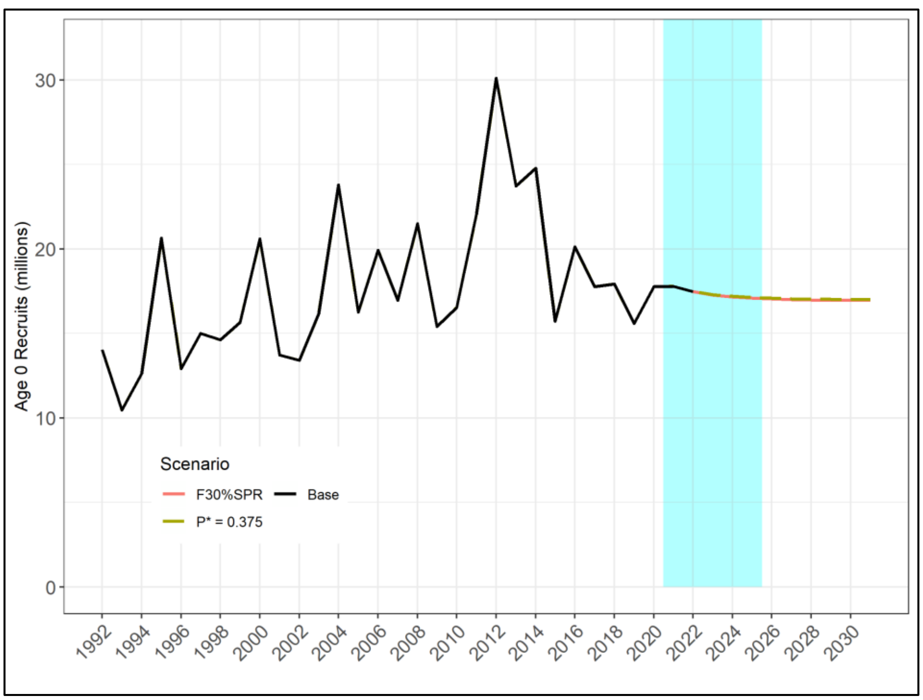

Figure 36. Age-0 recruits (in millions) associated with fishing at $F_{30\%SPR}$ (red line), the level that corresponds to a P* value of 0.375 (green line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

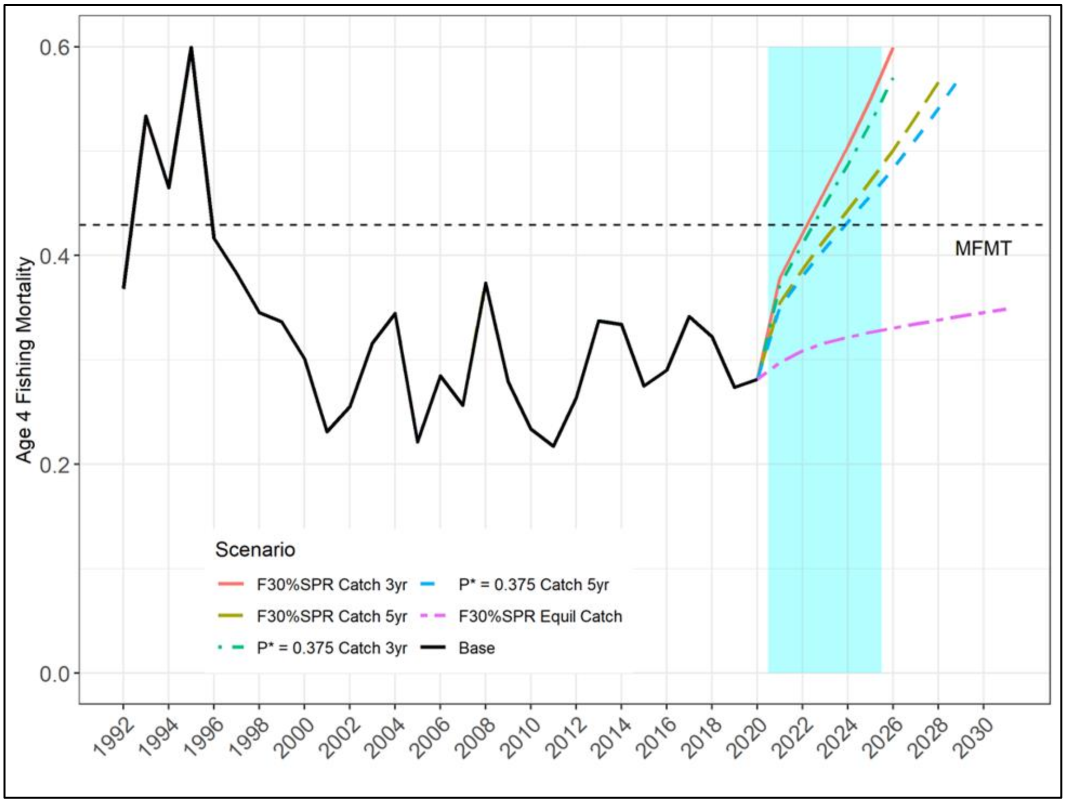

Figure 37. Age-4 fishing mortality values under several constant catch scenarios; retained yield under $F_{30\%SPR}$ averaged over 3 and 5 years (red solid line and green solid line, respectively), retained yield under the level that corresponds to a P* value of 0.375 averaged over 3 and 5 years (green dashed line and blue dashed lines, respectively), equilibrium retained yield associated with $F_{30\%SPR}$ (purple dashed line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

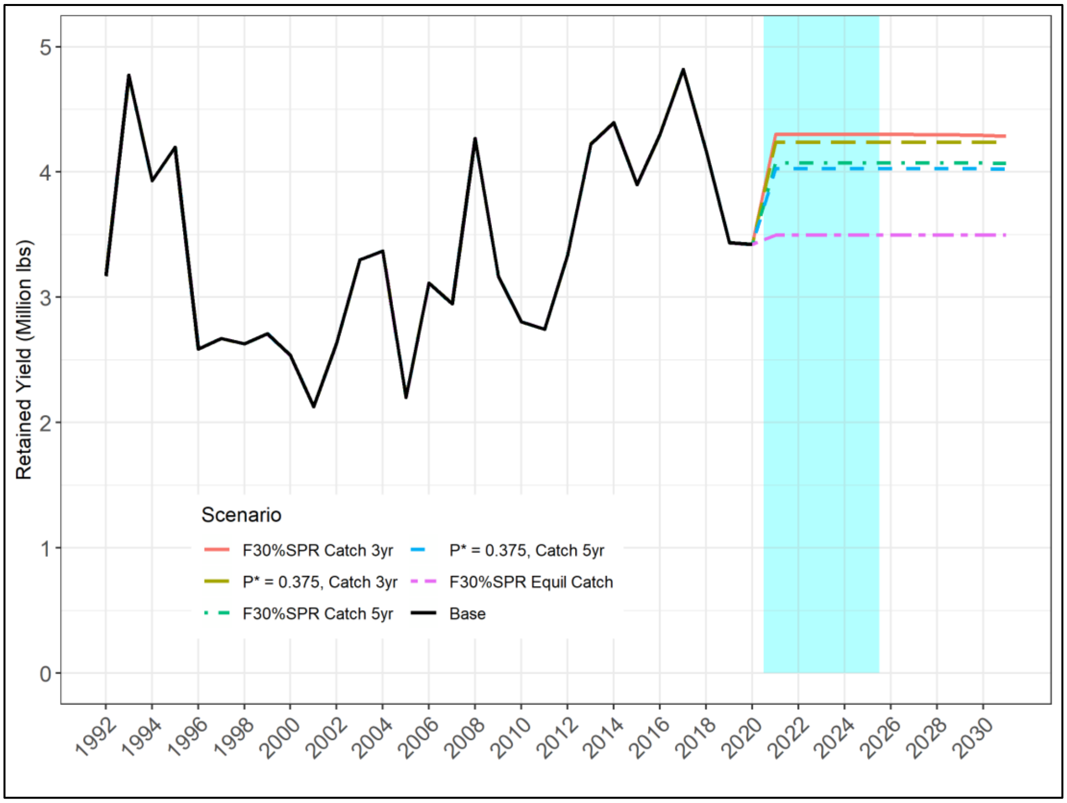

Figure 38. Retained yield (million pounds) under several constant catch scenarios; retained yield under $F_{30\%SPR}$ averaged over 3 and 5 years (red solid line and green solid line, respectively), retained yield under the level that corresponds to a P* value of 0.375 averaged over 3 and 5 years (green dashed line and blue dashed lines, respectively), equilibrium retained yield associated with $F_{30\%SPR}$ (purple dashed line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

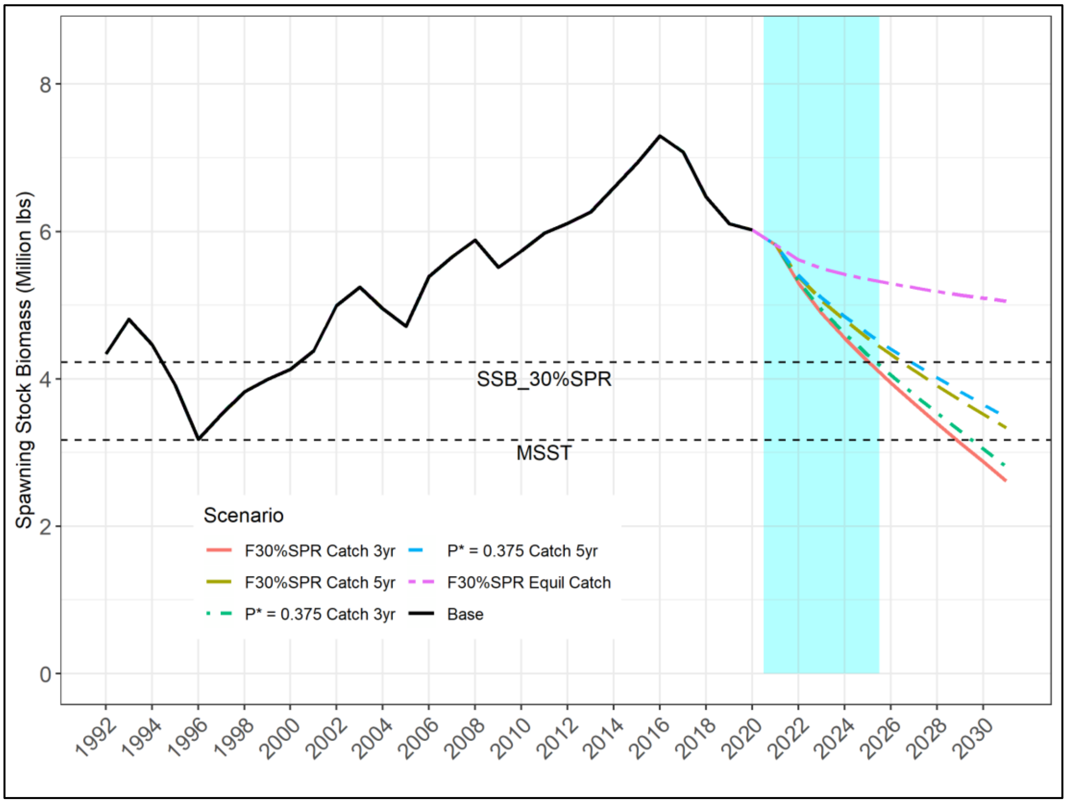

Figure 39. Spawning stock biomass (million pounds) under several constant catch scenarios; retained yield under $F_{30\%SPR}$ averaged over 3 and 5 years (red solid line and green solid line, respectively), retained yield under the level that corresponds to a P* value of 0.375 averaged over 3 and 5 years (green dashed line and blue dashed lines, respectively), equilibrium retained yield associated with $F_{30\%SPR}$ (purple dashed line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

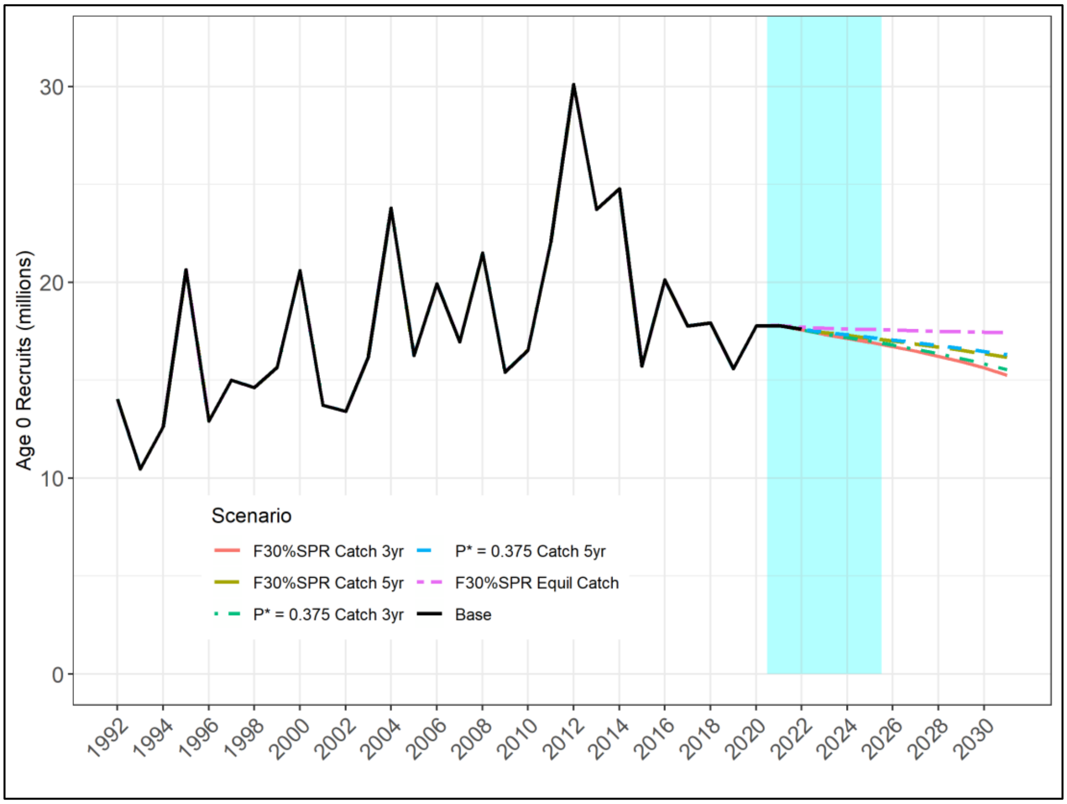

Figure 40. Age-0 recruits (millions) under several constant catch scenarios; retained yield under $F_{30\%SPR}$ averaged over 3 and 5 years (red solid line and green solid line, respectively), retained yield under the level that corresponds to a P* value of 0.375 averaged over 3 and 5 years (green dashed line and blue dashed lines, respectively), equilibrium retained yield associated with $F_{30\%SPR}$ (purple dashed line), and as estimated by the IBM (black solid line). The cyan region highlights the first five years of the projection (2021 – 2025).

7 Tables

Table 1. Commercial landings (pounds, metric tons) in Florida by region for years 2018 – 2020.

Landings (whole lbs.)

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2018	29	20,996	1,908,453	58,538	123	1,988,139
2019	41	21,669	2,098,050	85,988	196	2,205,944
2020	25	12,443	1,339,926	55,507	171	1,408,072

Landings (mt)

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2018	0.013	9.524	865.659	26.552	0.056	901.804
2019	0.019	9.829	951.659	39.004	0.089	1,000.598
2020	0.011	5.644	607.780	25.178	0.078	638.690

Table 2. Commercial landings (metric tons, pounds), their assumed standard error (in log-space), discards (thousands of fish), and their coefficient of variation in Florida for years 1981 – 2020.

Year	Landings (mt)	Landings (lbs.)	Standard Error (log-space)	Discards (000s)	Coefficient of Variation
1981	331.858	731,622	0.10		
1982	621.746	1,370,715	0.10		
1983	436.228	961,718	0.10		
1984	429.690	947,305	0.10		
1985	374.314	825,221	0.10		
1986	507.467	1,118,774	0.05		
1987	614.799	1,355,399	0.05		
1988	640.722	1,412,550	0.05		
1989	838.990	1,849,657	0.05		
1990	796.173	1,755,261	0.05		
1991	843.840	1,860,350	0.05		
1992	839.832	1,851,512	0.05		
1993	1,078.975	2,378,733	0.05	91.894	2.33
1994	1,000.400	2,205,506	0.05	104.953	2.35
1995	842.226	1,856,790	0.05	120.819	2.34
1996	661.835	1,459,097	0.05	117.016	2.33
1997	759.271	1,673,906	0.05	139.401	2.34
1998	691.470	1,524,431	0.05	97.937	2.36
1999	837.396	1,846,142	0.05	105.379	2.33
2000	721.992	1,591,720	0.05	103.543	2.34
2001	644.163	1,420,138	0.05	87.545	2.36
2002	638.447	1,407,536	0.05	86.703	1.95
2003	639.567	1,410,005	0.05	81.817	2.01
2004	671.289	1,479,939	0.05	51.467	2.60
2005	600.804	1,324,546	0.05	48.862	2.93
2006	561.040	1,236,882	0.05	75.741	2.42
2007	443.598	977,965	0.05	83.977	2.20
2008	621.421	1,369,999	0.05	49.966	2.85
2009	895.889	1,975,097	0.05	60.269	1.94
2010	768.364	1,693,953	0.05	49.540	3.00
2011	858.897	1,893,544	0.05	60.210	2.17
2012	955.851	2,107,291	0.05	39.464	3.28
2013	934.919	2,061,143	0.05	47.271	5.11
2014	926.807	2,043,260	0.05	59.156	3.58
2015	996.975	2,197,954	0.05	23.527	5.61
2016	1,050.023	2,314,905	0.05	44.739	2.33
2017	1,279.324	2,820,426	0.05	37.886	3.33
2018	901.804	1,988,139	0.05	29.956	4.19
2019	1,000.598	2,205,944	0.05	38.597	3.58
2020	638.690	1,408,072	0.05	24.636	3.58

Table 3. Headboat landings (thousands of fish), discards (thousands of fish), and their respective assumed standard errors (in log-space) in Florida for years 1981 – 2020.

Year	Landings (000s)	Standard Error (log-space)	Discards (000s)	Standard Error (log-space)
1981	159.972	0.25	9.865	0.50
1982	201.278	0.25	5.884	0.50
1983	205.315	0.25	71.705	0.50
1984	156.301	0.25	58.883	0.50
1985	137.632	0.25	1.785	0.50
1986	206.149	0.25	16.039	0.50
1987	235.527	0.25	194.371	0.50
1988	291.372	0.25	279.661	0.50
1989	166.437	0.25	38.546	0.50
1990	218.763	0.25	186.058	0.50
1991	212.789	0.25	1171.961	0.50
1992	205.367	0.25	70.613	0.50
1993	218.701	0.25	50.914	0.50
1994	243.158	0.25	73.847	0.50
1995	157.496	0.25	63.104	0.50
1996	137.599	0.25	57.175	0.50
1997	139.838	0.25	88.120	0.50
1998	120.526	0.25	84.235	0.50
1999	109.223	0.25	48.342	0.50
2000	109.300	0.25	47.851	0.50
2001	101.869	0.25	22.699	0.50
2002	121.012	0.25	44.506	0.50
2003	108.854	0.25	65.429	0.50
2004	118.422	0.25	21.535	0.50
2005	149.087	0.25	15.812	0.50
2006	98.974	0.25	19.154	0.50
2007	104.598	0.25	26.965	0.50
2008	103.362	0.25	39.757	0.50
2009	88.380	0.25	37.637	0.50
2010	102.174	0.25	36.335	0.50
2011	98.768	0.25	24.211	0.50
2012	110.815	0.25	30.564	0.50
2013	113.097	0.25	39.777	0.50
2014	163.990	0.25	64.492	0.50
2015	173.617	0.25	65.844	0.50
2016	184.576	0.25	68.637	0.50
2017	110.679	0.25	33.818	0.50
2018	113.282	0.25	46.598	0.50
2019	241.516	0.25	62.499	0.50
2020	169.626	0.25	45.006	0.50

Table 4. MRIP landings (thousands of fish) and releases (thousands of fish) in Florida by region for years 2017 – 2020.

Landings (000s)

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2017	0.000	304.551	839.815	400.493	5.437	1,550.296
2018	0.000	74.051	658.794	960.244	3.462	1,696.551
2019	0.000	76.392	478.745	250.499	0.000	805.637
2020	0.000	41.747	737.861	730.010	0.249	1,509.868

Live Releases (000s)

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2017	0.000	114.382	1,669.138	487.509	1.968	2,272.998
2018	0.456	50.630	1,513.459	1,151.028	45.240	2,760.814
2019	0.000	47.969	1,081.940	471.446	0.000	1,601.356
2020	0.000	96.067	1,982.903	433.940	1.921	2,514.831

Table 5. MRIP landings (thousands of fish), releases (thousands of fish), and their respective standard errors (in log-space) in Florida for years 1981 – 2020.

Year	Landings (000s)	Standard Error (log-space)	Discards (000s)	Standard Error (log-space)
1981	5,356.740	0.23	932.356	0.17
1982	6,098.713	0.22	1,120.300	0.23
1983	1,566.289	0.17	563.421	0.53
1984	4,067.863	0.41	3,787.895	0.37
1985	1,754.715	0.39	321.611	0.08
1986	1,475.112	0.39	1,050.654	0.28
1987	1,162.387	0.23	2,103.332	0.21
1988	1,137.940	0.15	1,116.803	0.27
1989	4,685.673	0.25	3,107.529	0.28
1990	3,440.760	0.41	1,980.252	0.14
1991	4,210.209	0.46	13,560.780	0.20
1992	969.581	0.20	3,406.179	0.12
1993	1,964.950	0.15	4,779.787	0.10
1994	1,301.688	0.14	2,815.507	0.17
1995	1,859.946	0.18	3,311.798	0.15
1996	871.358	0.17	3,282.277	0.07
1997	785.974	0.20	3,485.100	0.15
1998	878.573	0.24	2,435.771	0.14
1999	659.544	0.15	2,080.940	0.19
2000	722.441	0.30	1,781.311	0.16
2001	521.603	0.36	1,100.164	0.13
2002	951.985	0.14	1,259.174	0.14
2003	1,491.566	0.13	1,799.551	0.06
2004	1,459.769	0.34	2,505.699	0.09
2005	609.636	0.17	1,648.308	0.14
2006	1,527.089	0.21	2,664.445	0.10
2007	1,580.351	0.24	3,481.530	0.13
2008	2,351.513	0.26	3,235.121	0.14
2009	925.484	0.16	2,394.375	0.11
2010	849.533	0.13	1,526.499	0.20
2011	619.515	0.17	1,665.608	0.13
2012	910.906	0.28	1,675.632	0.16
2013	1,723.631	0.09	4,887.298	0.16
2014	1,906.725	0.09	4,092.275	0.12
2015	1,322.040	0.10	2,711.547	0.10
2016	1,524.592	0.10	1,539.521	0.15
2017	1,550.296	0.11	2,272.998	0.08
2018	1,696.551	0.13	2,760.814	0.16
2019	805.637	0.17	1,601.356	0.17
2020	1,509.868	0.19	2,514.831	0.10

Table 6. The number of MRIP angler interviews (non-imputed and imputed) which caught Yellowtail Snapper for private, charter, and shore modes combined by region for years 2017 – 2020.

Non-imputed interviews

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2017	0	50	337	137	4	528
2018	1	52	360	172	4	589
2019	0	26	230	114	0	370
2020	0	22	219	53	1	295

Including imputed interviews

Year	Northwest	Southwest	Keys	Southeast	Northeast	Total
2020	0	38	347	102	2	489

Table 7. Francis weights applied to length and conditional age-at-length data of the Interim Base Model (IBM) and compared with those applied in the SEDAR 64 Base Model (S64 Base).

		Francis Weights	
Data Type	Fleet/Index	IBM	S64 Base
Length Composition	Commercial	4.21	4.35
	Headboat	1.02	1.13
	MRIP	1.48	1.49
	RVC Adult	0.48	0.48
	RVC Juvenile	1.04	0.84
	MRIP CPUE	6.92	6.63
Conditional Age-at-length	Commercial	0.17	0.17
	Headboat	0.27	0.28
	MRIP	0.14	0.14
	FI Ages	0.10	0.14

Table 8. Predicted total biomass (metric tons, pounds), spawning stock biomass (SSB; metric tons, pounds), abundance (1000s of fish), age-0 recruits (1000s of fish), and depletion (SSB/SSB0) for southeastern U.S. Yellowtail Snapper from the Interim Base Model. Virgin is the estimated unfished condition while Initial is the estimated initial conditions of the stock before the model start year.

Year	Total Biomass (mt)	Total Biomass (lbs.)	SSB (mt)	SSB (lbs.)	Abundance (000s)	Age-0 Recruits (000s)	SSB/SSB0
Virgin	15,854	34,951,164	7,350	16,204,817	61,616	19,484	1.000
Initial	5,922	13,055,054	2,397	5,284,452	46,828	19,484	0.326
1992	5,184	11,429,720	1,967	4,337,325	42,102	14,050	0.268
1993	5,405	11,916,125	2,181	4,807,394	35,550	10,479	0.297
1994	4,760	10,494,829	2,023	4,460,982	32,461	12,631	0.275
1995	4,283	9,443,402	1,777	3,918,073	38,960	20,650	0.242
1996	3,861	8,512,214	1,443	3,181,619	34,426	12,914	0.196
1997	4,098	9,035,128	1,593	3,512,180	35,255	14,999	0.217
1998	4,309	9,499,641	1,735	3,824,178	35,302	14,616	0.236
1999	4,482	9,881,460	1,811	3,991,795	36,506	15,667	0.246
2000	4,662	10,277,387	1,874	4,131,370	42,212	20,609	0.255
2001	5,030	11,088,754	1,985	4,377,141	38,801	13,719	0.270
2002	5,477	12,075,123	2,265	4,994,324	37,224	13,408	0.308
2003	5,577	12,295,739	2,379	5,245,805	38,728	16,186	0.324
2004	5,408	11,922,497	2,245	4,950,430	46,602	23,797	0.305
2005	5,454	12,023,160	2,140	4,717,049	43,483	16,263	0.291
2006	6,011	13,251,816	2,445	5,389,811	46,621	19,932	0.333
2007	6,214	13,700,016	2,566	5,657,408	44,775	16,958	0.349
2008	6,405	14,120,481	2,668	5,882,169	48,504	21,498	0.363
2009	6,128	13,510,463	2,502	5,516,554	43,514	15,415	0.340
2010	6,233	13,741,154	2,601	5,733,158	42,809	16,543	0.354
2011	6,403	14,117,152	2,711	5,976,681	48,116	22,067	0.369
2012	6,747	14,874,108	2,771	6,109,575	59,303	30,103	0.377
2013	7,224	15,925,491	2,842	6,266,610	59,167	23,719	0.387
2014	7,491	16,513,927	2,991	6,593,622	59,902	24,786	0.407
2015	7,674	16,917,570	3,142	6,927,842	51,141	15,724	0.428
2016	7,759	17,104,721	3,310	7,296,388	51,110	20,133	0.450
2017	7,504	16,543,138	3,209	7,075,463	48,176	17,764	0.437
2018	6,949	15,320,478	2,935	6,470,053	46,222	17,929	0.399
2019	6,574	14,492,797	2,770	6,106,886	42,879	15,599	0.377
2020	6,435	14,185,826	2,730	6,019,318	43,588	17,765	0.371

Table 9. Estimates of annual instantaneous fishing mortality rates on age-4 southeastern U.S. Yellowtail Snapper combined across all fleets for the Interim Base Model.

Year	Age-4 F
1992	0.368
1993	0.534
1994	0.465
1995	0.600
1996	0.417
1997	0.383
1998	0.346
1999	0.336
2000	0.301
2001	0.231
2002	0.255
2003	0.316
2004	0.344
2005	0.222
2006	0.285
2007	0.256
2008	0.374
2009	0.279
2010	0.234
2011	0.217
2012	0.263
2013	0.337
2014	0.334
2015	0.275
2016	0.290
2017	0.342
2018	0.322
2019	0.274
2020	0.281

Table 10. Annual estimates of instantaneous apical fishing mortality rates by fleet for southeastern U.S. Yellowtail Snapper from the Interim Base Model. This represents the instantaneous fishing mortality level on the most vulnerable age class for each fleet.

Year	Commercial	Headboat	MRIP
1992	0.301	0.039	0.371
1993	0.372	0.037	0.696
1994	0.364	0.047	0.505
1995	0.366	0.038	0.890
1996	0.329	0.036	0.457
1997	0.339	0.030	0.354
1998	0.281	0.025	0.372
1999	0.322	0.022	0.271
2000	0.265	0.022	0.286
2001	0.217	0.018	0.189
2002	0.190	0.019	0.307
2003	0.186	0.018	0.487
2004	0.208	0.022	0.517
2005	0.186	0.026	0.210
2006	0.157	0.015	0.463
2007	0.117	0.015	0.460
2008	0.165	0.015	0.694
2009	0.243	0.014	0.280
2010	0.197	0.015	0.242
2011	0.210	0.014	0.173
2012	0.232	0.016	0.254
2013	0.229	0.015	0.468
2014	0.216	0.020	0.475
2015	0.216	0.020	0.310
2016	0.217	0.021	0.349
2017	0.278	0.014	0.386
2018	0.212	0.016	0.457
2019	0.244	0.035	0.226
2020	0.159	0.025	0.432

Table 11. Runs test applied to the residuals of indices and mean length composition data fit within the Interim Base Model. The p-value indicates whether there is evidence ($p < 0.05$) or no evidence ($p > 0.05$) to reject the hypothesis of randomly distributed residuals. Lower (Lower 3 SD) and upper (Upper 3 SD) three residual standard deviation values away from zero are presented along with the qualitative metric (passed/failed) which corresponds to the p-value.

Index	P-value	Lower 3 SD	Upper 3 SD	Metric
Commercial CPUE	0.001	-0.395	0.395	Failed
RVC Adult	0.656	-0.909	0.909	Passed
RVC Juvenile	0.148	-0.535	0.535	Passed
MRIP CPUE	0.135	-0.328	0.328	Passed

Mean length	P-value	Lower 3 SD	Upper 3 SD	Metric
Commercial	0.244	-0.038	0.038	Passed
Headboat	0.012	-0.037	0.037	Failed
MRIP	0.290	-0.114	0.114	Passed
RVC Adult	0.500	-0.152	0.152	Passed
RVC Juvenile	0.110	-0.143	0.143	Passed
MRIP CPUE	0.699	-0.073	0.073	Passed

Table 12. Index root mean square error (RMSE) values from the Interim Base Model (IBM), the age-structured production model (ASPM), and the ASPM with estimated recruitment deviations (ASPMdev).

Index	IBM	ASPM	ASPMdev
Commercial CPUE	0.1810	0.2259	0.1745
RVC Adult	0.2608	0.3547	0.2524
RVC Juvenile	0.1833	0.2410	0.1622
MRIP CPUE	0.1742	0.2377	0.1832

Table 13. Mohn's rho (ρ_M) and forecast Mohn's rho (ρ_F) values calculated from the retrospective and retrospective forecasting analyses, respectively, on estimates of spawning stock biomass (SSB) and age-4 fishing mortality rates (F) from the Interim Base Model. Analyses were performed across 5 successive years of removal from the terminal year 2020.

Quantity	Year Peel	Mohn's Rho (ρ_M)	Forecast Mohn's Rho (ρ_F)
SSB	2019	0.004	0.006
SSB	2018	0.007	-0.001
SSB	2017	-0.010	-0.013
SSB	2016	-0.077	-0.096
SSB	2015	-0.051	-0.082
SSB	Combined	-0.025	-0.037
F	2019	0.009	0.000
F	2018	-0.001	0.010
F	2017	0.010	0.012
F	2016	0.083	0.106
F	2015	0.063	0.105
F	Combined	0.033	0.047

Table 14. Gelman and Rubin's (1992) potential scale reduction factor (PSRF) values from the combined two MCMC chains for selected model parameters (R0, SSB0, and steepness) and derived quantities (age-4 F in 2020, SSB in 2020, $F_{30\%SPR}$, SSB at $F_{30\%SPR}$, and the retained yield at $F_{30\%SPR}$) of the Interim Base Model.

PSRF	PSRF Upper CI	Parameter or Derived Quantity
1.01	1.03	R0
1.00	1.00	SSB0
1.01	1.06	Steepness
1.01	1.02	SSB_{2020}
1.02	1.02	F_{2020}
1.00	1.00	$F_{30\%SPR}$
1.01	1.01	$SSB_{F30\%SPR}$
1.01	1.02	Retained yield at $F_{30\%SPR}$

Table 15. The stock status determination criterion for southeastern U.S. Yellowtail Snapper according to the South Atlantic Fishery Management Council (SAFMC) and the Gulf of Mexico Fishery Management Council (GMFMC). Note: values of MSST and OY are currently undefined for the GMFMC and they default to the definition provided below by the SAFMC. The IBM Value is derived from the IBM and the MCMC Value is the median value of the posterior distribution.

South Atlantic and Gulf of Mexico Fishery Management Councils

Criteria	Definition	IBM Value	MCMC Value (median)
$F_{30\%SPR}$	The fishing mortality rate associated with 30% SPR and the proxy used for F_{MSY}	0.429 yr^{-1}	0.433
MFMT (Maximum Fishing Mortality Threshold)	$F_{30\% SPR}$	0.429 yr^{-1}	0.433
$F_{current}$ (recent average fishing mortality rate on age-4 fish)	The geometric mean of F on age-4 fish for 2018 - 2020	0.292 yr^{-1}	0.295
$SSB_{F30\%SPR}$	The estimated spawning stock biomass associated with F at 30% SPR	1,915.86 mt (4,223,743 lbs.)	1,915.62 mt (4,223,214 lbs.)
MSST (Minimum Stock Size Threshold)	$0.75*SSB_{F30\%SPR}$	1,436.90 mt (3,167,807 lbs.)	1,436.72 mt (3,167,422 lbs.)
$SSB_{current}$ (recent average of SSB)	The geometric mean of SSB for 2018 - 2020	2,810.33 mt (6,195,718 lbs.)	2,733.63 mt (6,026,615 lbs.)
MSY (Maximum Sustainable Yield)	Yield at $F_{30\%SPR}$	1,587.08 mt (3,498,908 lbs.)	1,573.36 mt (3,468,661 lbs.)
OY (Optimum Yield)	ABC based on SAFMC P*	1,545.68 mt (3,407,637 lbs.)	1,535.04 mt (3,384,180 lbs.)

Table 16**.** Yellowtail Snapper projected landings in millions of pounds under $F_{30\%SPR}$ (MFMT) and the fishing mortality rate that corresponds to a P* value of 0.375 (ABC) from 2021 – 2031.

Year	$F_{30\%SPR}$	P* = 0.375
2021	4.766	4.671
2022	4.207	4.153
2023	3.922	3.887
2024	3.774	3.749
2025	3.684	3.665
2026	3.625	3.610
2027	3.584	3.572
2028	3.557	3.546
2029	3.538	3.528
2030	3.525	3.516
2031	3.516	3.507

Table 17. Yellowtail Snapper projected spawning stock biomass in millions of pounds under $F_{30\%SPR}$ (MFMT) and the fishing mortality rate that corresponds to a P* value of 0.375 (ABC), from 2021 – 2031.

Year	$F_{30\%SPR}$	P* = 0.375
2021	5.816	5.816
2022	5.114	5.151
2023	4.762	4.817
2024	4.577	4.642
2025	4.465	4.535
2026	4.391	4.465
2027	4.341	4.417
2028	4.307	4.384
2029	4.284	4.361
2030	4.268	4.346
2031	4.257	4.335

Table 18. Yellowtail Snapper projected age-0 recruitment in millions of fish under $F_{30\%SPR}$ (MFMT) and the fishing mortality rate that corresponds to a P* value of 0.375 (ABC), from 2021 – 2031.

Year	$F_{30\%SPR}$	P* = 0.375
2021	17.792	17.792
2022	17.467	17.487
2023	17.274	17.306
2024	17.163	17.203
2025	17.092	17.137
2026	17.043	17.091
2027	17.010	17.060
2028	16.986	17.038
2029	16.970	17.023
2030	16.959	17.013
2031	16.951	17.006

Table 19. Yellowtail Snapper projected annual age-4 fishing mortality rates under five constant catch projection scenarios, from 2021 – 2031.

Year	$F_{30\%SPR}$ Catch 3yr	$F_{30\%SPR}$ Catch 5yr	P* = 0.375, Catch 3yr	P* = 0.375, Catch 5yr	$F_{30\%SPR}$ Equil. Catch
2021	0.378	0.355	0.372	0.350	0.297
2022	0.421	0.387	0.412	0.380	0.309
2023	0.463	0.416	0.450	0.407	0.316
2024	0.504	0.443	0.487	0.432	0.322
2025	0.549	0.471	0.527	0.457	0.326
2026	0.600	0.501	0.570	0.484	0.331
2027	0.656	0.532	0.619	0.511	0.334
2028	0.722	0.566	0.674	0.541	0.338
2029	0.800	0.603	0.738	0.573	0.342
2030	0.894	0.643	0.813	0.607	0.345
2031	1.011	0.688	0.903	0.644	0.349

Table 20**.** Yellowtail Snapper projected spawning stock biomass (million pounds) under five constant catch projection scenarios, from 2021 – 2031.

Year	$F_{30\%SPR}$ Catch 3yr	$F_{30\%SPR}$ Catch 5yr	P* = 0.375, Catch 3yr	P* = 0.375, Catch 5yr	$F_{30\%SPR}$ Equil. Catch
2021	5.816	5.816	5.816	5.816	5.816
2022	5.298	5.388	5.322	5.406	5.615
2023	4.895	5.066	4.941	5.101	5.501
2024	4.552	4.798	4.618	4.847	5.421
2025	4.240	4.555	4.325	4.619	5.354
2026	3.947	4.329	4.051	4.406	5.294
2027	3.668	4.115	3.789	4.205	5.239
2028	3.398	3.911	3.537	4.014	5.187
2029	3.135	3.714	3.292	3.830	5.139
2030	2.875	3.523	3.051	3.652	5.094
2031	2.615	3.336	2.811	3.478	5.052

Table 21. Yellowtail Snapper projected age-0 recruitment in millions of fish under five constant catch projection scenarios, from 2021 – 2031.

Year	$F_{30\%SPR}$ Catch 3yr	$F_{30\%SPR}$ Catch 5yr	P* = 0.375, Catch 3yr	P* = 0.375, Catch 5yr	$F_{30\%SPR}$ Equil. Catch
2021	17.792	17.792	17.792	17.792	17.792
2022	17.560	17.603	17.571	17.611	17.706
2023	17.350	17.443	17.375	17.461	17.655
2024	17.147	17.295	17.188	17.323	17.618
2025	16.940	17.149	16.999	17.189	17.587
2026	16.720	17.002	16.801	17.053	17.558
2027	16.485	16.849	16.590	16.915	17.531
2028	16.228	16.691	16.365	16.772	17.505
2029	15.945	16.526	16.119	16.624	17.480
2030	15.627	16.351	15.848	16.470	17.457
2031	15.262	16.165	15.543	16.308	17.435

8 Appendix

Table A1. Yellowtail Snapper landings (pounds) for the Commercial, Headboat, and MRIP fleets in Florida for years 1992 – 2020.

Year	Commercial	Headboat	MRIP	Total
1992	1,851,512	258,950	1,400,647	3,511,109
1993	2,378,733	378,807	2,334,717	5,092,257
1994	2,205,506	269,870	1,656,934	4,132,310
1995	1,856,790	163,936	1,980,260	4,000,986
1996	1,459,097	140,935	1,087,893	2,687,925
1997	1,673,906	149,911	1,008,943	2,832,760
1998	1,524,431	122,895	1,061,541	2,708,867
1999	1,846,142	105,929	804,271	2,756,342
2000	1,591,720	97,521	729,810	2,419,051
2001	1,420,138	99,547	600,318	2,120,003
2002	1,407,536	110,936	870,838	2,389,310
2003	1,410,005	97,199	1,615,512	3,122,716
2004	1,479,939	104,071	1,668,828	3,252,838
2005	1,324,546	148,936	622,470	2,095,952
2006	1,236,882	85,399	1,701,112	3,023,393
2007	977,965	84,753	1,889,692	2,952,410
2008	1,369,999	94,070	2,697,920	4,161,989
2009	1,975,097	80,118	949,370	3,004,585
2010	1,693,953	89,739	978,430	2,762,122
2011	1,893,544	92,552	943,810	2,929,906
2012	2,107,291	121,417	972,774	3,201,482
2013	2,061,143	114,676	1,532,100	3,707,919
2014	2,043,260	177,331	1,998,309	4,218,900
2015	2,197,954	177,597	1,391,931	3,767,482
2016	2,314,905	188,058	1,522,151	4,025,114
2017	2,820,426	117,929	1,880,002	4,818,357
2018	1,988,139	104,935	1,521,940 *	3,615,014
2019	2,205,944	235,374	872,478 *	3,313,796
2020	1,408,072	147,282	1,433,681 *	2,989,035

Note: asterisks (*) denote interim years (2018 – 2020) where MRIP data were obtained via the publicly available portal. Landings in weight for years 1992 – 2017 were estimated by the SEFSC using the weight estimation methodology described in SEDAR 64-WP 12.