Received: 14 April 2023 | Revised: 18 September 2023 | Accepted: 22 September 2023

DOI: 10.1002/nafm.10966

ARTICLE

# Fishery management strategies for Red Snapper in the southeastern U.S. Atlantic: A spatial population model to compare approaches

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**Funding information**
Office of Sustainable Fisheries' National Catch Share Program and Magnuson-Stevens Act Implementation

## Abstract

**Objective:** Red Snapper *Lutjanus campechanus* is an iconic species in the southeast U.S. Atlantic Ocean, sought by both commercial and recreational fleets. Five stock assessments over the past quarter century have shown Red Snapper to be experiencing overfishing. Highly restricted landings since 2010 have been insufficient to end overfishing because fishing effort is not species specific but rather applies generally to a complex of reef-associated species. Consequently, Red Snapper are discarded as bycatch when regulations prohibit their retention, and many of the discarded fish die from hook injury, barotrauma, or depredation.

**Methods:** Here we developed a spatial population model of Red Snapper and the multispecies fishery that captures them in the southeast U.S. Atlantic. We then simulated and compared 25 different management measures that fall broadly into the categories of gear modifications, discard mortality mitigation, size limits, spatial approaches, or temporal approaches. Criteria for comparison address the management goals of decreasing dead discards, rebuilding the age structure, and increasing landings and spawning biomass.

**Result:** We found that the most effective measures reduced fishing effort, either temporally or spatially, and that benefits could largely be obtained by focusing on the recreational fleet. Discard mortality mitigation (e.g., through use of descender devices) displayed a wide range in effectiveness depending on plausible levels of mortality reduction, but it addressed all management goals and in practice could be paired with other measures. A measure with restricted recreational effort combined with full retention of all fish caught showed the greatest potential to simultaneously rebuild the stock, increase landings, and eliminate dead discards.

**Conclusion:** To end overfishing of Red Snapper as required by law, resource managers should reconsider the policy of unrestricted effort of the private recreational fleet to this multispecies fishery. The benefits of restricted effort would include increased catch rates, larger landed fish, and fewer dead discards.

KEYWORDS

fishery management strategies, Red Snapper, simulation modeling, snapper–grouper fishery, spatial population model

## INTRODUCTION

Commonly, fishery managers are provided with scientific information in the form of stock assessments, and they must then choose among management measures to meet their goals of optimal yield supported by sustainable fish stocks. The measures available to managers vary widely, falling broadly into two main categories of input or output controls (Bellido et al. 2020). Input controls regulate fishing effort directly through measures such as access, spatial and temporal restrictions, or limits on gear. Output controls apply to the catch after fishing effort has taken place, regulating what is allowed to be landed through measures such as size limits, trip or bag limits, or quotas. For any given application, evaluating the benefits and risks of these various measures—prior to their selection—can help achieve management goals (Francis and Shotton 1997). Ideally, such evaluations would inform a management process in which the course of action, conditional on current scientific information, is agreed upon beforehand and then followed (Rosenberg 2003).

The primary benefit of a pre–agreed management strategy is to streamline decision making. Fishery management is a political process, and decisions are too often mired in serial delays, especially when the scientific advice would result in short-term catch or effort reductions (Shertzer and Prager 2007). For example, Cowan. et al. (2011) described how fishery managers were slow to sufficiently reduce catches of Red Snapper *Lutjanus campechanus* in the Gulf of Mexico, such that the substantial cuts eventually required could have been avoided if less severe management measures had been implemented earlier. Such delays may be built into the formal steps required to implement new regulations but can also result from litigation (Powers 2004) and protracted debate about the scientific advice (Rosenberg 2003). Because this advice always contains some level of uncertainty, a common and arguably rational response is that managers want more information before making a difficult decision. This rationale for delay can be more pronounced when the composition of management panels is dominated by stakeholders (Okey 2003). To inform those panels, regardless of composition, simulation-based analyses can provide an objective comparison of the benefits and risks of potential management measures (e.g., Cooke 1999; Mapstone et al. 2008; McQuaw et al. 2021; Bohaboy et al. 2022).

The Red Snapper fishery in the Atlantic Ocean along the southeastern United States is managed by the South Atlantic Fishery Management Council (SAFMC). The stock was first assessed in 1998 and estimated to be experiencing overfishing (Manooch et al. 1998). Since then, four more stock assessments—completed in 2008, 2010, 2017, and 2021—have estimated the stock to be overfished and experiencing overfishing (SouthEast Data, Assessment, and Review [SEDAR] 2021). Since 2010, the stock has been under a formal rebuilding plan, in which limits on fish landed per trip are low and the open season is extremely short. For example, in 2022 recreational anglers were allowed to retain one fish per person per day for only 2 days. However, despite strict regulations on landings since 2010, the rate of removals continues to exceed the SAFMC's threshold, primarily due to the magnitude of dead discards estimated from the recreational sector (SEDAR 2021). Thus, the SAFMC continues to consider various options to address overfishing.

Red Snapper are a reef-associated fish that are highly valued in the southeastern United States. Their life history is unusual in that they live long (maximum observed age exceeds 50 years) and mature young (majority of females mature by 2 years). They are captured primarily by hook-and-line gear as part of a multispecies fishery on reef-associated fishes, such as snappers (family Lutjanidae) and groupers (family Epinephelidae). Indeed, the SAFMC's Snapper Grouper Fishery Management Plan includes 55 species. Both recreational and commercial fleets target this complex of fishes, with the recreational sector being the dominant source of fishing mortality for many of these species, including Red Snapper (Shertzer et al. 2019). The multispecies nature of this fishery presents a management challenge in the sense that regulations have historically been established on a species-by-species basis, but fishing effort applies to the complex.

In general, this approach to management of using single-species output controls in a multispecies fishery can result in a substantial amount of discarded bycatch. When paired with nonnegligible release mortality (Davis 2002), discarded bycatch is a waste of natural resources that hinders conservation and results in foregone fishery yield (Harrington et al. 2005; Abbot and Wilen 2009). This has been the case for reef-associated fishes in the southeastern U.S. Atlantic (Rudershausen et al. 2007), including Red Snapper, where strict regulation of landings alone has been insufficient to end overfishing because of substantial discard mortality (SEDAR 2021).

**Impact statement**

We used a spatial population model of Red Snapper in the U.S. Atlantic to compare various fishery management strategies. The most effective strategy restricted fishing effort of the private recreational fleet, with benefits that included increased abundance, increased catch rates, larger landed fish, and fewer dead discards.

Here, we developed a spatial operating model of the Red Snapper population and fishery off the U.S. Atlantic coast. We used the model to simulate and compare various management measures, such as gear modification, size limits, and spatial or temporal approaches. Management goals for this stock of Red Snapper include decreasing dead discards, rebuilding the age structure, and increasing spawning biomass and sustainable landings. Thus, we used those criteria to compare the various management measures for their relative effectiveness. The model was not intended to provide tactical advice but rather strategic guidance about which measures are most likely to achieve management goals.

# METHODS

The Methods are structured as follows. We first describe the operating model in terms of population dynamics and fishery dynamics. Parameter values representing prevailing conditions (Table 1) were either taken directly from, or computed from, values or data sources used in the most recent stock assessment of Red Snapper (SEDAR 2021). We then describe 25 potential management measures and how they can be simulated by modifying relevant model parameters. Performance of the management measures was evaluated by first simulating the operating model under prevailing conditions and then simulating implementation of each measure. The final Methods subsection details those simulations and describes metrics used to compare performance of the management measures.

The simulations encompassed the jurisdictional boundaries of the SAFMC in federally managed Atlantic waters off the southeastern United States. For the operating model and management measures, this jurisdiction was divided into six distinct areas (Figure 1). Latitudinal breaks at 28°N and 32°N separated the jurisdiction into three regions: North Carolina and South Carolina (northern region), Georgia and north Florida (middle region), and south Florida (southern region). These regions were intended to capture, in general, spatial patterns that may occur in both Red Snapper abundance and fishing effort. Each of these regions was further divided into nearshore and offshore components, separated at the 35-m isobath, to represent depth-related patterns in Red Snapper age-specific habitat use, fishing effort, and discard mortality.

## Operating model—Population dynamics

We modeled abundance ($N$) in area $i$ at age $a$ in year $y$ by applying exponential decay:

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

**TABLE 1** Parameters used in the model. The column "Value" indicates the baseline values (prevailing conditions) that were obtained or derived from the most recent stock assessment (SEDAR 2021).

| Parameter | Description | Value |
|---|---|---|
| $\gamma_1$ | Weight-at-length coefficient | $1.65\times10^{-8}$ |
| $\gamma_2$ | Weight-at-length exponent | 2.99 |
| $K$ | Growth coefficient | 0.23 |
| $L_\infty$ | Asymptotic length | 911 |
| $a_0$ | Theoretical age at which length = 0 | −0.33 |
| $R_0$ | Unfished level of recruitment | $4.37\times10^{5}$ |
| $h$ | Steepness of recruitment function | 0.99 |
| $\varphi_0$ | Unfished spawners per recruit | 0.017 |
| $\beta_1^C$ | Fleet 1 catch selectivity slope | 0.015 |
| $\beta_2^C$ | Fleet 2 catch selectivity slope | 0.02 |
| $\alpha_1^C$ | Fleet 1 catch selectivity 50% location | 343 |
| $\alpha_2^C$ | Fleet 2 catch selectivity 50% location | 428 |
| $\beta_1^R$ | Fleet 1 retention slope | 10 |
| $\beta_2^R$ | Fleet 2 retention slope | 10 |
| $\alpha_1^R$ | Fleet 1 retention 50% location | 0 |
| $\alpha_2^R$ | Fleet 2 retention 50% location | 0 |
| $\rho_{1,i}$ | Fleet 1 asymptotic retention in all areas $i$ | 0.15 |
| $\rho_{2,i}$ | Fleet 2 asymptotic retention in all areas $i$ | 0.80 |
| $\delta_{1,i=1,2,3}$ | Fleet 1 discard mortality rate nearshore | 0.23 |
| $\delta_{1,i=4,5,6}$ | Fleet 1 discard mortality rate offshore | 0.25 |
| $\delta_{2,i=1,2,3}$ | Fleet 2 discard mortality rate nearshore | 0.32 |
| $\delta_{2,i=4,5,6}$ | Fleet 2 discard mortality rate offshore | 0.35 |
| $q_1$ | Fleet 1 catchability | 0.38 |
| $q_2$ | Fleet 2 catchability | 0.03 |

where $Z_{i,a,y}$ is the total mortality rate, which sums the natural mortality rate at age ($M_a$) and the fishing mortality rate at age ($F_{i,a,y}$). We modeled ages 1 through 50 years, which was a maximum age sufficient to ensure that fewer than 0.5% of individuals would survive to the oldest age under natural mortality alone.

Spawning potential was measured as the total annual mature female biomass ($B_y$) in the population, summed across areas: $B_y = \sum_i B_{i,y}$. Within each area, $B_{i,y}$ was computed as the product of the sex ratio (50:50, as in the stock assessment), maturity at age ($V_a$), weight at age ($W_a$), and abundance at age as follows:

$$B_{i,y} = \sum_a 0.5 V_a W_a N_{i,a,y} \tag{2}$$

The term $W_a$ followed a power function of length at age ($L_a$), $W_a = \gamma_1 L_a^{\gamma_2}$, where $\gamma_1$ and $\gamma_2$ are parameters (Figure 2). The term $L_a$ followed the standard von

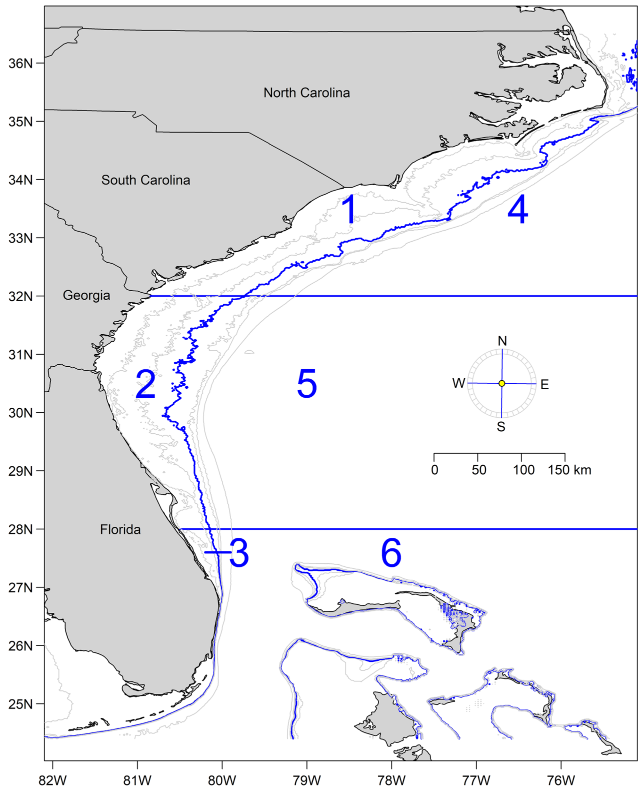


**FIGURE 1** Map of the study location depicting Atlantic Ocean waters off the southeastern United States. Numbers indicate the area designations used in the model, separated by the blue lines at 28°N and 32°N and the heavy blue curve at the 35-m isobath. Light-gray isobaths are drawn at 15, 25, 45, 55, and 200 m.

Bertalanffy function, $L_a = L_\infty\left(1 - e^{-K[a-a_0]}\right)$, in which $L_\infty$ is the maximum asymptotic length, $K$ is the growth coefficient, and $a_0$ is the theoretical age when length is zero (Figure 2; von Bertalanffy 1938).

Natural mortality decreased as fish grew larger, according to an inverse length relationship, $M'_a = e^{-\log(L_a)}$. This relationship has been shown to generally represent natural mortality in marine fishes (Lorenzen 2022; Lorenzen et al. 2022). We then scaled $M'_a$ to provide the same cumulative survival for ages 2+ as would be obtained under the hypothetical age-invariant natural mortality of 0.11/year as in the stock assessment (SEDAR 2021). This scaled vector was used in our model as $M_a$ (Figure 2).

Recruits of age-1 fish to the system ($R_y$) were computed using the Beverton–Holt spawner–recruit model,

$$R_y = \frac{0.8R_0hB_y}{0.2R_0\varphi_0(1-h)+B_y(h-0.2)}, \tag{3}$$

where $R_0$ is the unfished level of recruitment, $h$ is the steepness parameter controlling how quickly recruitment approaches $R_0$ as spawning biomass increases, and $\varphi_0$ is unfished spawners per recruit. For our Red Snapper case study, we set $h = 0.99$ to approximate a mean recruitment model in which recruitment is independent of spawning biomass as was used in the stock assessment (SEDAR 2021). Recruits to the system were distributed to nearshore areas ($i = \{1, 2, 3\}$) only, given the evidence that young Red Snapper (age-0 and age-1 fish) are primarily found in shallower water (Mitchell et al. 2014; Powers et al. 2018; Brodie et al. 2022).

We estimated relative abundance of each area ($p_i$) by applying the Vector Autoregressive Spatio-Temporal (VAST) package (Thorson and Barnett 2017; Thorson 2019) to fishery-independent data collected by the SouthEast Reef Fish Survey from 2011 to 2021. This survey samples reef-associated fishes throughout the management jurisdiction using paired Chevron traps and video gear (Bacheler et al. 2013). The model was simultaneously fit to video data for 21 species, including Red Snapper, at different locations and times by modeling response variables of presence or absence and catch rates (number of individuals observed per video frame) as a multivariate process using latent factors (for further details, see Cao et al., in review). The relative abundance of Red Snapper at each location on a 3- by 3-km grid was computed as the product of the predicted encounter probabilities and catch rates. Because depth contours vary by latitude, we assigned depths to each grid location in the VAST output using the *getNOAA.bathy* function in the marmap package of R (Pante and Simon-Bouhet 2013). Given latitude and depth of each location, we summed relative abundance estimates from VAST within each of the six areas, then divided by the sum across all areas to compute $p_i$, such that $\sum_i p_i = 1$ (Table 2).

We allocated recruits to nearshore areas in proportion to regional relative abundances. That is, the proportion of recruits allocated to the northern region was $p_1 + p_4$, the proportion allocated to the middle region was $p_2 + p_5$, and the proportion allocated to the southern region was $p_3 + p_6$ (Figure 1; Table 2),

$$N_{i,a=1,y} = \left(p_i + p_{i+3}\right)R_y \tag{4}$$

for nearshore areas $i = \{1, 2, 3\}$, and $N_{i,a=1,y} = 0$ for offshore areas $i = \{4, 5, 6\}$. We then modeled ontogenetic movement by assuming that a proportion of age-2 fish moved to the offshore areas, consistent with findings that age-0 and age-1 Red Snapper are found primarily in shallower water (<~35 m) but age-2 and older fish occur across all inhabited depths (Mitchell et al. 2014; Powers et al. 2018). For each region, we imputed the proportion of age-2 Red Snapper that moved from the nearshore area to the offshore area, such that equilibrium relative abundances in our model matched those provided by VAST (Table 2). These proportions were 0.25, 0.17, and 0.33 in the northern, middle, and southern regions, respectively.

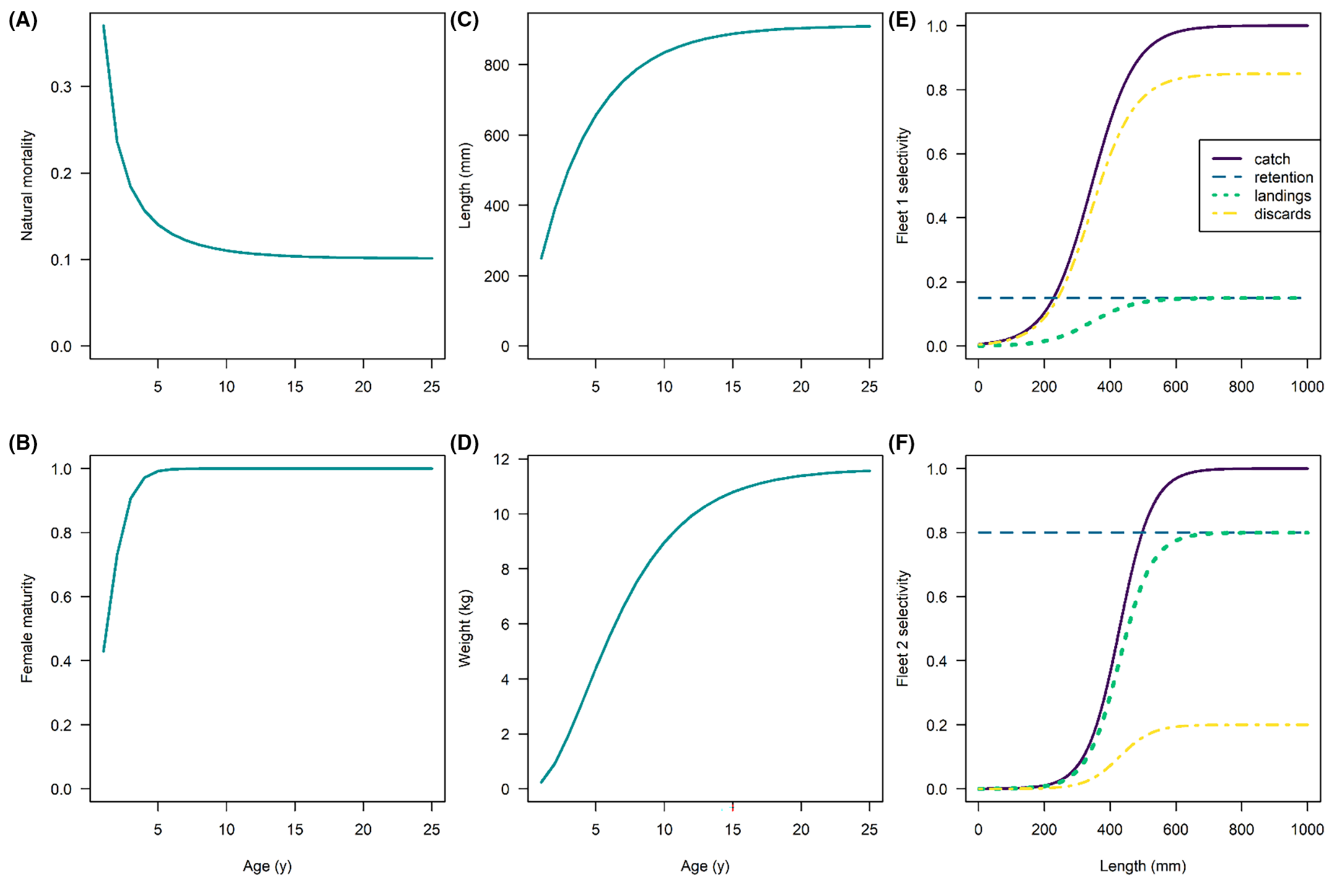


**FIGURE 2** Life history and selectivity ogives used as model inputs. Maximum age in the model is 50 years, although life history ogives are plotted here only through age 25 to better show values prior to saturation. Panels show (A) natural mortality rate (per year), (B) proportion of females mature at age, (C) length at age, (D) weight at age, (E) fleet 1 (recreational) selectivity functions, and (F) fleet 2 (commercial) selectivity functions. Selectivity functions shown represent prevailing conditions, which were altered for some management scenarios (as described in the text).

**TABLE 2** Relative abundance and fishing effort by area. Abundance values represent percentage of the total population within each area, and effort values are scaled to have a mean of 1.0 across areas. Values represent base levels, prior to implementation of any management scenarios. Fleet 1 represents the recreational sector and fleet 2 the commercial sector.

| Area | Region/depth | Abundance (%) | Fleet 1 effort | Fleet 2 effort |
|---|---|---|---|---|
| 1 | Northern/nearshore | 3 | 1.50 | 2.08 |
| 2 | Middle/nearshore | 77 | 2.12 | 0.88 |
| 3 | Southern/nearshore | 2 | 0.81 | 0.98 |
| 4 | Northern/offshore | 1 | 0.27 | 0.54 |
| 5 | Middle/offshore | 16 | 0.64 | 0.12 |
| 6 | Southern/offshore | 1 | 0.66 | 1.40 |

## Operating model—Fishery dynamics

The total fishing mortality rate comprised two fleets ($f$), each with a landings component and discarding component. We configured the first fleet ($f = 1$; fleet 1) based on recreational fishing in Atlantic waters off the southeastern United States and the second fleet ($f = 2$; fleet 2) based on commercial fishing. For simplicity, we described selectivity functions below as independent of time and area, although they readily generalize to become functions of either, and we detail all such generalizations in the section *Management scenarios*. For each fleet, we modeled selectivity at age of the catch ($S^C_{f,a}$) as a logistic function of length as follows:

$$S^C_{f,a} = 1 / \left[1 + e^{-\beta^C_f\left(\left[L_a - \alpha^C_f\right]\right)}\right], \tag{5}$$

where $\beta^C_f$ defines the slope and $\alpha^C_f$ defines the length at 50% selection. Selectivity was modeled as logistic to match the most recent stock assessment (SEDAR 2021), although here it is a function of length (instead of age) to simplify implementation of length-based management measures (e.g., size limits). The catch was apportioned between landings and discards according to a retention ogive ($S^R_{f,a}$) as follows:

$$S^R_{f,a} = \rho_{f,i} / \left[1 + e^{-\beta^R_f\left(L_a - \alpha^R_f\right)}\right], \tag{6}$$

where $\beta^R_f$ defines the slope, $\alpha^R_f$ defines the length at 50% retention, and $\rho_f$ defines the asymptotic retention that can range between 0 (all fish discarded) and 1 (maximum possible retention). With logistic catch selectivity, a value of $\rho_{f,i} < 1$ models the situation where discards include the larger fish in the population. This currently occurs for Red Snapper because although landing them is prohibited during closed seasons, fishing effort continues on the multispecies fishery, resulting in additional Red Snapper discards.

Given the catch selectivity and the retention function, selectivity of landings ($S^L_{f,a}$) is the product $S^L_{f,a} = S^C_{f,a} S^R_{f,a}$ and selectivity of discards is the product $S^D_{f,a} = S^C_{f,a}\left(1 - S^R_{f,a}\right)$. For each fleet, the age-specific fishing mortality rate is

$$F_{f,i,a,y} = \left(S^L_{f,a} + \delta_{f,i} S^D_{f,a}\right) \Phi_{f,i,y}, \tag{7}$$

where $\delta_{f,i}$ is the fleet- and area-specific discard mortality rate, and $\Phi_{f,i,y}$ is the total fishing mortality rate of fleet $f$ in area $i$ and year $y$. Given the fishing rates and natural mortality, we computed the total mortality (equation 1) in each area as $Z_{i,a,y} = M_a + \sum_f F_{f,i,a,y}$.

We computed $\Phi_{f,i,y} = q_{f,i,y} E_{f,i,y}$, in which $q_{f,i,y}$ is catchability and $E_{f,i,y}$ is fishing effort, two parameters of management focus. Because our interest is primarily in strategic planning and in the relative performance of various management approaches, we used a base level of effort for each fleet equal to 1 ($E_f = 1$, with arbitrary measurement units). Thus, any spatiotemporal variations in effort can be viewed as relative changes (e.g., a 50% reduction in effort would be effected by $E_f = 0.5$). Assuming $E_f = 1$, we set a base level of catchability by fleet ($q_f$) that generated the fleet-specific fishing rate estimated by the most recent stock assessment, averaged over the years 2010–2019, a period with stable management (SEDAR 2021). That base level could then be modified by area or year to model spatial or temporal variation in catchability imposed by management measures. Because of scaling to $E_f = 1$, the values of $q_f$ represent relative fishing rates between recreational and commercial fleets, not actual catchabilities on the water.

We set discard mortality rates (in units of deaths per released fish) in the nearshore areas (areas 1–3) equal to the values from the most recent stock assessment (SEDAR 2021): $\delta_{f=1,i=1,2,3} = 0.23$ for the recreational fleet and $\delta_{f=2,i=1,2,3} = 0.32$ for the commercial fleet. For the offshore areas (areas 4–6), we would expect discard mortality rates to be higher than in nearshore areas because of barotrauma effects associated with being captured from deeper waters (Davis 2002; Rudershausen et al. 2007). Indeed, for the recreational fleet, estimates of discard mortality in the deeper areas were slightly higher: $\delta_{f=1,i=4,5,6} = 0.25$ (Vecchio et al. 2020). Similar depth-specific estimates of discard mortality rate were not available for the commercial fleet; thus, we applied the ratio of offshore : nearshore discard mortality rates from the recreational fleet to the commercial fleet: $\delta_{f=2,i=4,5,6} = 0.32 \times \frac{0.25}{0.23} = 0.35$.

Given the rates of mortality, we computed landings ($L$) and dead discards ($D$) using the Baranov catch equation (Baranov 1918; Sharov 2021):

$$L_{f,i,a,y} = \frac{S^L_{f,a} \Phi_{f,i,y}}{Z_{i,a,y}} N_{i,a}\left(1 - e^{-Z_{i,a,y}}\right) \tag{8}$$

$$D_{f,i,a,y} = \frac{\delta_{f,i} S^D_{f,a} \Phi_{f,i,y}}{Z_{i,a,y}} N_{i,a}\left(1 - e^{-Z_{i,a,y}}\right) \tag{9}$$

Then, we summed over ages to compute landings and discards by fleet, area, and year, and we summed those values over areas to compute total annual landings and discards by fleet.

For the recreational fleet ($f = 1$), effort was apportioned into each of the three regions using data from the Marine Recreational Information Program (MRIP; https://www.fisheries.noaa.gov/insight/marine-recreational-information-program). The MRIP collects catch and effort information on recreational fishing activity in U.S. marine waters, measuring effort as angler trips. To define effort toward this multispecies fishery, we included any trip that caught one of the following reef-associated species: Black Sea Bass *Centropristis striata*, Blueline Tilefish *Caulolatilus microps*, Gag *Mycteroperca microlepis*, Gray Triggerfish *Balistes capriscus*, Greater Amberjack *Seriola dumerili*, Red Grouper *Epinephelus morio*, Red Porgy *Pagrus pagrus*, Red Snapper, Scamp *Mycteroperca phenax*, Snowy Grouper *Hyporthodus niveatus*, Tilefish *Lopholatilus chamaeleonticeps*, Vermilion Snapper *Rhomboplites aurorubens*, or Yellowtail Snapper *Ocyurus chrysurus*. These species were selected because they are dominant members (in terms of catch) of the SAFMC's Snapper Grouper Fishery Management Plan, such that effective effort toward the complex would have likely caught at least one species from the list. We examined MRIP effort data for 1982–2021 to identify the most recent time period within each region during which recreational fishing effort was relatively stable. By regressing regional effort on year using regression tree analysis (R package "tree", Ripley 2022), we found that the most recent periods of effort stability were 2006–2021 for the northern region, 2007–2021 for the middle region, and 2004–2021 for the southern region. Based on these results, we used the period 2007–2021 to calculate annual average effort within each region as effort was relatively stable for this time period for all three regions.

To apportion annual effort by region into nearshore and offshore areas, we used citizen science data collected as part of the MyFishCount program (https://www.myfishcount.com/). MyFishCount is a mobile application and website that allows recreational anglers to report information about their catch, such as location, depth, and species caught. We restricted the data set to the same suite of species listed above to estimate regional effort from MRIP data ($N = 334$ trips in the restricted data set). The ratios of nearshore to offshore effort were 85:15, 77:23, and 55:45 for the northern, middle, and southern regions, respectively, and we assumed those ratios were representative of the recreational fleet. These ratios were applied to regional effort to compute total recreational effort by area (Figure 1), which we rescaled to have a mean of 1.0 to represent relative values and for consistency with how catchability was defined (Table 2).

For the commercial fleet ($f = 2$), effort by area was estimated using logbook data reported by commercial anglers with snapper–grouper permits. These data were available for 1993–2021, but we restricted our analysis to years 2011–2021 based on a similar regression tree analysis as described for the recreational data. We also used the same suite of species to identify trips that targeted this multispecies, snapper–grouper complex. Because commercial trips vary widely in their duration, from hours to weeks, we used trip-hours as the measure of effort. Areas reported are from a spatial grid with resolution of 1° latitude by 1° longitude; we used the midpoint of the reported grid cell to assign depth fished of each trip. As with the recreational effort, we rescaled the commercial effort by area to have a mean of 1.0 (Table 2).

## Management scenarios

Using the operating model configured for Red Snapper, we simulated and compared 25 different management measures. The measures are either input controls that regulate effort or output controls that apply after fish are captured, and they can be categorized into five basic types: gear modification, discard mortality mitigation, size limit, temporal regulations, and spatial regulations (Table 3). Although in practice some of the measures could be combined, we treat most separately here to isolate their effects on achieving management goals.

In the context of rebuilding the Red Snapper stock, gear modifications represent an attempt to reduce fishing power. Possible examples would include bait specifications, hook type, or number of allowable hooks per line. Since the start of this fishery, fishing power has increased substantially, with improved boating equipment, electronics, and information-sharing technology (Cooke et al. 2021). Worldwide estimates of increases in fishing power range about 2–4% per year on average (Palomares and Pauly 2019). Against this backdrop, the intention of gear modifications would be to temper the rise in fishing power. Here, we implement such an effect on the recreational fleet by reducing its catchability by 10% ($q_{1,i,y} = 0.9q_1$) or 30% ($q_{1,i,y} = 0.7q_1$).

Reducing discard mortality is already a management priority, and the SAFMC requires that a descender device be onboard when targeting fish in the snapper–grouper complex. Descender devices can reduce discard mortality by mitigating the effects of barotrauma (Runde and Buckel 2018; Bohaboy et al. 2020; Runde et al. 2021; Stallings et al. 2023). Although anglers must have a descender device on board, there is no requirement to use it and a minority of anglers choose to do so or are even aware of the requirement (Curtis et al. 2019; Responsive Management 2022). Education programs may encourage their use and further reduce discard mortality of Red Snapper. Here we considered two different sets of estimates of reduced discard mortality, both assuming maximum possible usage (100%) of descending devices. The first set is based on the same data utilized in the stock assessment (Vecchio et al. 2020) but with the highest usage of descender devices: $\delta_{f=1,i=1,2,3} = 0.21$ and $\delta_{f=1,i=4,5,6} = 0.23$. The second set is based on the estimate of Runde et al. (2021), a study that took place at a depth of 37 m, and suggested a discard mortality rate for our offshore areas of $\delta_{f=1,i=4,5,6} = 0.13$; to compute a value for our nearshore areas, we applied the same ratio from the first set: $\delta_{f=1,i=1,2,3} = 0.13 \times (0.21/0.23) = 0.12$. For both sets of estimates, we assumed that they applied only to the recreational sector and that commercial fishing practices would continue with the status quo. Widespread, voluntary use of descender devices in the commercial sector seems unlikely, given the potential to slow onboard operations.

Minimum size limits are a common management tool of the SAFMC, although currently there is no such

**TABLE 3** Management measures considered and their effects on the population and fishery. The "Parameters" column indicates which model parameters were changed to simulate each management scenario, with baseline levels (see Table 1) in simulation years ($y$) 1 through 100 and modifications in years 101 through 200; unless fleet ($f$) or area ($i$) is specified, the modification applies to both fleets or all areas. Fleet 1 represents the recreational sector and fleet 2 the commercial sector. Values indicate percent changes from prevailing conditions that result from each scenario; values in the second row of each table cell (in italics) are from sensitivity runs with lower steepness (mean age and weights are not reported as they were unaffected by steepness). Metrics shown are total abundance ($N$), spawning biomass ($B$), mean age of the population ($\overline{a}$), landings of fleet 1 (L1), landings of fleet 2 (L2), dead discards from fleet 1 (D1), dead discards from fleet 2 (D2), mean weight of fish landed by fleet 1 (L1 $\overline{W}$), and mean weight of fish landed by fleet 2 (L2 $\overline{W}$).

| Scenario | Type | Control | Description | Parameters | N | B | $\overline{a}$ | L1 | L2 | D1 | D2 | L1 $\overline{W}$ | L2 $\overline{W}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Gear modification | Input | Reduce fleet 1 catchability by 10% | $q_{f=1,i,y}$ | 3.4 | 9.3 | 4.4 | −4.4 | 7.7 | −4.4 | 7.7 | 5.1 | 4.0 |
| | | | | | *7.2* | *13.3* | | *−0.9* | *11.7* | *−0.9* | *11.6* | | |
| 2 | Gear modification | Input | Reduce fleet 1 catchability by 30% | $q_{f=1,i,y}$ | 11.6 | 33.3 | 15.3 | −14.9 | 26.9 | −14.9 | 26.8 | 17.1 | 13.3 |
| | | | | | *23.6* | *47.7* | | *−5.7* | *40.5* | *−5.7* | *40.4* | | |
| 3 | Discard mortality mitigation | Output | Fleet 1 mortality based on Vecchio et al. (2020) | $\delta_{f=1,i,y}$ | 1.6 | 4.4 | 2.1 | 3.0 | 3.7 | −5.9 | 3.6 | 2.4 | 1.9 |
| | | | | | *3.4* | *6.2* | | *4.8* | *5.5* | *−4.2* | *5.5* | | |
| 4 | Discard mortality mitigation | Output | Fleet 1 mortality based on Runde et al. (2021) | $\delta_{f=1,i,y}$ | 10.4 | 29.6 | 13.7 | 19.1 | 23.7 | −37.9 | 23.6 | 15.3 | 11.9 |
| | | | | | *21.2* | *42.3* | | *30.8* | *35.9* | *−31.8* | *35.8* | | |
| 5 | Size limit | Output | 24-inch size limit | $\alpha_f^R, \beta_f^R$ | 8.3 | 17.7 | 5.6 | −59.5 | −46.2 | 29.0 | 281.2 | 105.9 | 65.1 |
| | | | | | *15.3* | *25.3* | | *−56.9* | *−42.7* | *37.3* | *305.9* | | |
| 6 | Temporal | Output | Asymptotic retention halved | $\rho_{f,i}$ | 6.7 | 18.9 | 8.8 | −43.8 | −42.0 | 22.4 | 247.8 | 10.3 | 8.4 |
| | | | | | *14.0* | *27.0* | | *−39.9* | *−38.0* | *30.7* | *271.5* | | |
| 7 | Temporal | Output | Asymptotic retention doubled | $\rho_{f,i}$ | −9.2 | −23.3 | −11.4 | 67.0 | −0.5 | −31.2 | −100.0 | −14.0 | −11.3 |
| | | | | | *−21.0* | *−33.3* | | *45.2* | *−13.4* | *−40.2* | *−100.0* | | |
| 8 | Temporal | Input | All effort reduced 25% | $E_{f,i,y}$ | 10.1 | 29.0 | 13.4 | −10.9 | −7.2 | −10.9 | −7.2 | 15.2 | 12.2 |
| | | | | | *20.8* | *41.5* | | *−2.3* | *1.8* | *−2.3* | *1.8* | | |
| 9 | Temporal | Input | All effort reduced 75% | $E_{f,i,y}$ | 50.3 | 168.7 | 71.1 | −50.3 | −43.3 | −50.4 | −43.4 | 68.1 | 53.1 |
| | | | | | *90.9* | *241.4* | | *−36.9* | *−28.0* | *−37.0* | *−28.1* | | |
| 10 | Temporal | Input | Fleet 1 effort reduced 25% | $E_{f=1,i,y}$ | 9.4 | 26.5 | 12.2 | −12.0 | 21.5 | −12.0 | 21.4 | 13.9 | 10.8 |
| | | | | | *19.2* | *37.9* | | *−4.0* | *32.5* | *−4.1* | *32.4* | | |
| 11 | Temporal | Input | Fleet 1 effort reduced 75% | $E_{f=1,i,y}$ | 43.6 | 142.3 | 60.3 | −53.8 | 106.4 | −53.9 | 105.9 | 59.5 | 45.5 |
| | | | | | *79.9* | *203.5* | | *−42.1* | *158.6* | *−42.2* | *157.9* | | |
| 12 | Temporal | Both | Fleet 1 effort reduced 25%, full retention | $E_{f=1,i,y}$, $\rho_{f=1,i}$ | −22.8 | −52.2 | −27.3 | 202.1 | −47.7 | −100.0 | −47.6 | −34.5 | −27.9 |
| | | | | | *−59.1* | *−74.7* | | *60.0* | *−72.3* | *−100.0* | *−72.2* | | |
| 13 | Temporal | Both | Fleet 1 effort reduced 75%, full retention | $E_{f=1,i,y}$, $\rho_{f=1,i}$ | 10.8 | 31.1 | 14.5 | 99.6 | 24.5 | −100.0 | 24.4 | 15.8 | 12.3 |
| | | | | | *22.1* | *44.5* | | *120.0* | *37.3* | *−100.0* | *37.1* | | |
| 14 | Spatial | Output | Close offshore areas to Red Snapper retention | $\rho_{f,i=4,5,6}$ | 2.5 | 9.3 | 6.8 | −9.7 | −8.9 | 4.0 | 49.2 | −7.2 | −4.4 |
| | | | | | *6.2* | *13.3* | | *−6.3* | *−5.6* | *7.8* | *54.6* | | |
| 15 | Spatial | Output | Close northern region to Red Snapper retention | $\rho_{f,i=1,4}$ | 0.7 | 2.2 | 1.3 | −3.0 | −10.2 | 1.6 | 59.1 | −0.3 | −1.3 |
| | | | | | *1.7* | *3.2* | | *−2.1* | *−9.3* | *2.6* | *60.6* | | |

**TABLE 3** (Continued)

| Scenario | Type | Control | Description | Parameters | N | B | $\overline{a}$ | L1 | L2 | D1 | D2 | L1 $\overline{W}$ | L2 $\overline{W}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 16 | Spatial | Output | Close middle region to Red Snapper retention | $\rho_{f,i=2,5}$ | 14.1 | 40.9 | 18.3 | −95.0 | −82.8 | 49.2 | 488.3 | 21.3 | 19.3 |
| | | | | | *28.4* | *58.5* | | *−94.4* | *−80.6* | *67.9* | *561.8* | | |
| 17 | Spatial | Output | Close southern region to Red Snapper retention | $\rho_{f,i=3,6}$ | 0.6 | 2.0 | 1.3 | −1.9 | −7.0 | 0.9 | 38.9 | −0.8 | −2.3 |
| | | | | | *1.4* | *2.9* | | *−1.1* | *−6.2* | *1.8* | *40.1* | | |
| 18 | Spatial | Input | Close offshore areas to effort, shifts nearshore | $E_{f,i,y}$ | 4.5 | 30.3 | 37.5 | 0.0 | −12.3 | −0.9 | −13.0 | −20.7 | −15.4 |
| | | | | | *14.9* | *43.3* | | *10.0* | *−3.5* | *9.1* | *−4.3* | | |
| 19 | Spatial | Input | Close northern region to effort | $E_{f,i=1,4,y}$ | 3.8 | 14.4 | 11.5 | −2.9 | −10.0 | −2.9 | −10.1 | −0.3 | −1.3 |
| | | | | | *9.4* | *20.6* | | *2.4* | *−5.1* | *2.4* | *−5.2* | | |
| 20 | Spatial | Input | Close middle region to effort | $E_{f,i=2,5,y}$ | 99.4 | 376.3 | 152.4 | −95.0 | −82.7 | −95.0 | −82.4 | 21.3 | 19.3 |
| | | | | | *167.2* | *538.4* | | *−93.3* | *−76.8* | *−93.3* | *−76.4* | | |
| 21 | Spatial | Input | Close southern region to effort | $E_{f,i=3,6,y}$ | 2.5 | 10.0 | 8.5 | −1.8 | −6.9 | −1.8 | −7.1 | −0.8 | −2.3 |
| | | | | | *6.6* | *14.4* | | *2.1* | *−3.2* | *2.0* | *−3.4* | | |
| 22 | Spatial | Input | Close offshore areas to Fleet 1 effort, shifts nearshore | $E_{f=1,i,y}$ | 3.6 | 26.2 | 30.4 | 0.5 | −10.9 | −0.3 | −10.3 | −20.1 | 1.1 |
| | | | | | *12.9* | *37.5* | | *9.5* | *−2.9* | *8.6* | *−2.2* | | |
| 23 | Spatial | Input | Close northern region to fleet 1 effort | $E_{f=1,i=1,4,y}$ | 2.2 | 7.5 | 5.0 | −2.9 | 12.2 | −2.9 | 12.1 | −0.3 | 11.3 |
| | | | | | *5.3* | *10.8* | | *0.0* | *15.5* | *0.0* | *15.5* | | |
| 24 | Spatial | Input | Close middle region to fleet 1 effort | $E_{f=1,i=2,5,y}$ | 79.8 | 289.2 | 118.4 | −95.0 | 194.8 | −95.0 | 193.5 | 21.3 | 73.7 |
| | | | | | *137.4* | *413.7* | | *−93.4* | *289.1* | *−93.4* | *287.5* | | |
| 25 | Spatial | Input | Close southern region to fleet 1 effort | $E_{f=1,i=3,6,y}$ | 1.6 | 6.1 | 4.6 | −1.8 | 5.6 | −1.9 | 5.7 | −0.8 | 6.6 |
| | | | | | *4.1* | *8.7* | | *0.6* | *8.2* | *0.6* | *8.3* | | |

regulation for Red Snapper. Here, we implement a 610-mm (~24-inch) minimum size limit. This was achieved in the model by modifying the retention functions for both fleets such that the slope parameter ($\beta_f^R = 0.1$) defines a curve with rapid ascent from zero to maximum retention at the size limit as specified by the location parameter ($\alpha_f^R = 610$).

Temporal regulations were implemented in several ways. First, we examined the effectiveness of season length for Red Snapper by adjusting the asymptotic retention parameters $\rho_{f,i}$ relative to their prevailing values (Table 1). In one such scenario, we adjusted the season length downward by halving the retention parameters, and in another, we simulated longer seasons by increasing the retention parameters. In this latter scenario, we doubled the retention parameter of the recreational sector and set it to 1 for the commercial sector (a doubling for the commercial sector would have exceeded 1, which is the maximum possible value for this parameter). For both simulations of season length, multispecies fishing effort outside the Red Snapper season continued. Thus, we considered an alternative set of temporal regulations in which all effort for the multispecies fishery (both inside and outside the Red Snapper season) was reduced. For these scenarios, we applied a 25% or a 75% reduction to the prevailing fishing effort (Table 2). For both levels of reduction, we applied them either to both sectors or to the recreational sector only. Two additional scenarios applied both levels of reduction to the recreational sector but simultaneously adjusted the retention function such that all Red Snapper caught would be retained ($\rho_{f,i} = 1$; zero discards). These latter two are the only scenarios that adjust more than one management approach simultaneously, with the objective being to eliminate the wasteful practice of discarded bycatch (Harrington et al. 2005).

Similar to temporal regulations, spatial regulations were applied either to Red Snapper only or to all fishing effort targeting the complex. Four scenarios prohibited Red Snapper landings from offshore areas, from the northern region, from the middle region, or from the southern region (Figure 1). These scenarios were modeled by setting retention $\rho_{f,i} = 0$ for all areas $i$ that corresponded to where landings were prohibited. Four additional scenarios prohibited year-round multispecies recreational and commercial fishing effort from offshore areas or from each of the three regions. For the scenario prohibiting offshore effort, we assumed that any offshore effort would be displaced to the nearshore areas of the same region. These four scenarios prohibiting effort were repeated but with the prohibition applied only to the recreational sector.

## Simulation details, sensitivity analyses, and performance metrics

Our primary interest was to compare expected outcomes of the various management scenarios, and thus we used deterministic simulations of the operating model to compute equilibrium values. We ran each simulation for 200 modeled years. The first 100 years applied the prevailing, base-level conditions (Table 1), and the second 100 years applied one of the 25 management scenarios by modifying relevant model parameters (Table 3). In each time block, 100 years was sufficient to reach equilibrium. We computed expected values of the prevailing conditions as the equilibria in year 100 and expected values of the management scenario as the equilibria in year 200. All analyses and simulations were conducted in R version 4.2.1 (R Core Team 2022).

The steepness parameter of the spawner–recruit function controls the level of density dependence in recruitment. Thus, steepness can be critical in determining the response of a population to management measures, and we consider that potential here through sensitivity analysis. Although the most recent stock assessment of Red Snapper did not detect a relationship between spawning biomass and recruitment (SEDAR 2021), it remains possible that such density dependence exists under the aphorism that "absence of evidence is not evidence of absence." To explore the effects of that possibility, we revised the operating model to have a steepness value of $h = 0.8$ and resimulated all 25 management scenarios. The value of 0.8 was the mean of a normal distribution estimated through meta-analysis of demersal marine fishes (Shertzer and Conn 2012).

To measure performance of each management scenario, we computed equilibrium values of abundance, spawning biomass, mean age of the population, landings (in numbers) of each fleet, dead discards (in numbers) of each fleet, mean weight (kg) of the landings of each fleet, and area-specific catch rates (landed fish in numbers per unit effort) of the recreational fleet. To compare management scenarios, we report the percent change in equilibrium values relative to the prevailing conditions. For example, a value of 100% would indicate that that metric doubled as a result of the management scenario.

# RESULTS

Of the 25 management scenarios explored, all but two resulted in increases of Red Snapper abundance, spawning biomass, and mean age of the population (Table 3). Reductions in the current rate of fishing, by whatever measure, allowed spawning biomass to increase more rapidly than abundance because the age structure shifted toward more older individuals as indicated by the increased mean age (example scenario in Figure 3). Whether landings, discards, or average weight of landings increased or decreased depended on details of the management measure (example scenario in Figure 4). For most scenarios, recreational catch rates increased modestly or else decreased. Five scenarios incurred increases that exceeded 50% (Figure 5).

Gear modifications that reduced catchability of fleet 1 helped rebuild the stock (Table 3). However, they reduced catch rates by about 6% (scenario 1) or 20% (scenario 2). In addition, gear modifications were inefficient in the sense that a 10% reduction in catchability only reduced landings and discards by less than 5% (scenario 1), and a 30% reduction in catchability only reduced landings and discards by less than 15% (scenario 2). Thus, to address the management goal of substantial reductions in dead discards, any gear modification would need to cause an even greater inefficiency in the fishery operation. For example, in simulations configured similarly to scenarios 1 and 2, catchability would need to be reduced by 72% to achieve at least a 50% reduction in the discard mortality from fleet 1.

Reductions in the discard mortality rate could also help rebuild the stock (Table 3). Such reductions, as might be achieved through increased use of descender devices, have potential to increase landings, increase catch rates, and reduce dead discards while promoting stock recovery. In our simulations, a modest effect of descender devices (scenario 3; ~8.5% decrease in discard mortality rate) reduced the fleet 1 dead discards by 5.9%, with a 3% increase in landings and catch rates. A more significant effect of descender devices (scenario 4; ~48% decrease in discard mortality rate) reduced the fleet 1 dead discards by 37.8%,

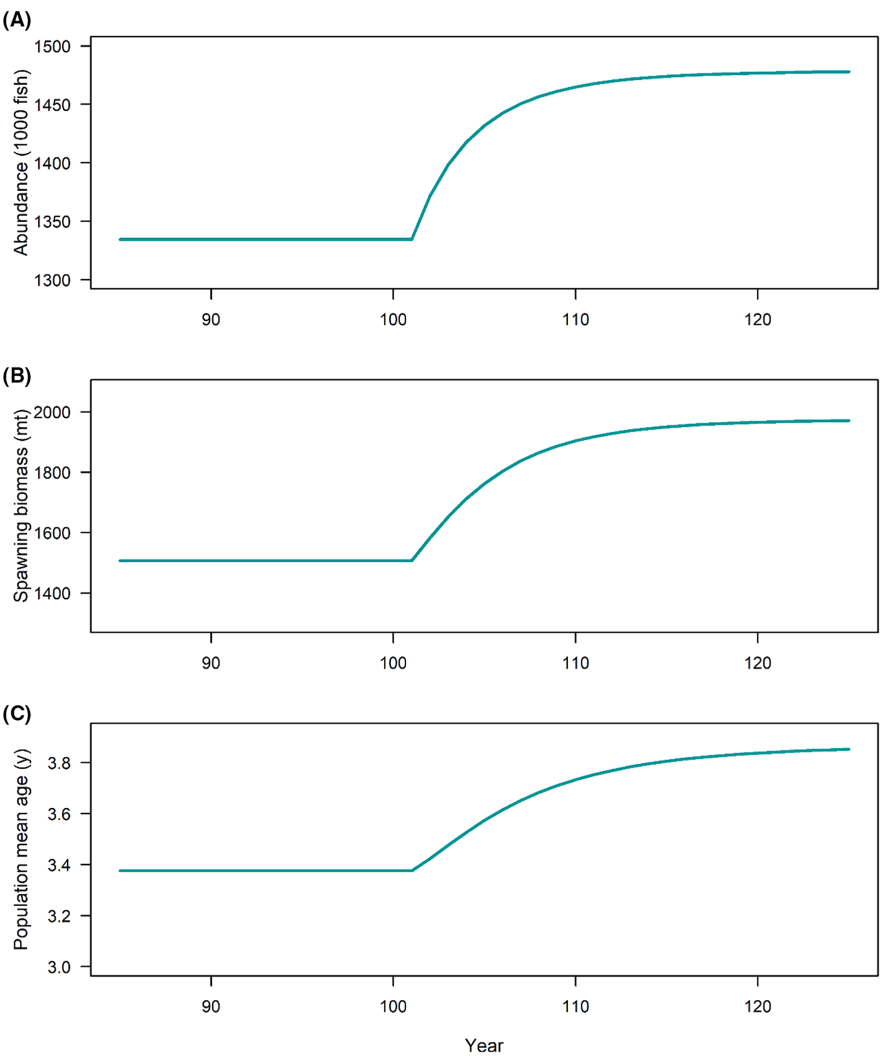


**FIGURE 3** Example effects of a management measure (scenario 13) on the population (A) abundance, (B) spawning biomass, and (C) mean age of the population. Management measures were implemented in year 101 of a 200-year simulation, and these plots focus on the time period surrounding implementation. The years prior to implementation show equilibria under prevailing conditions.

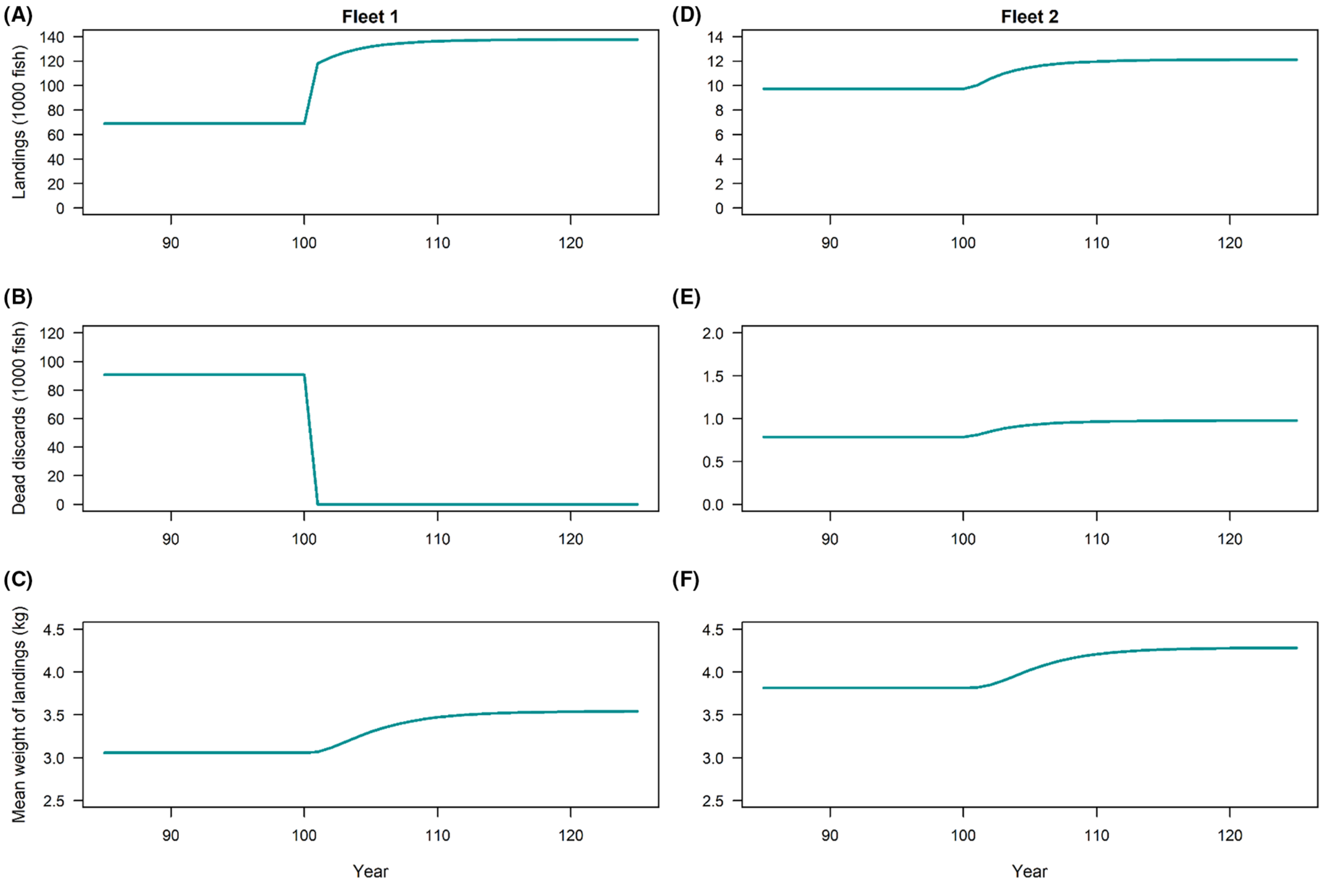


**FIGURE 4** Example effects of a management measure (scenario 13) on the fishery. The three panels in the left column show the effects on fleet 1 (recreational), and the three panels in the right column show the effects on fleet 2 (commercial). The first row of panels shows landings, the second row shows dead discards, and the third row shows mean weight of landings (note different *y*-axis scales for landings and discards). Management measures were implemented in year 101 of a 200-year simulation, and these plots focus on the time period surrounding implementation. The years prior to implementation show equilibria under prevailing conditions.

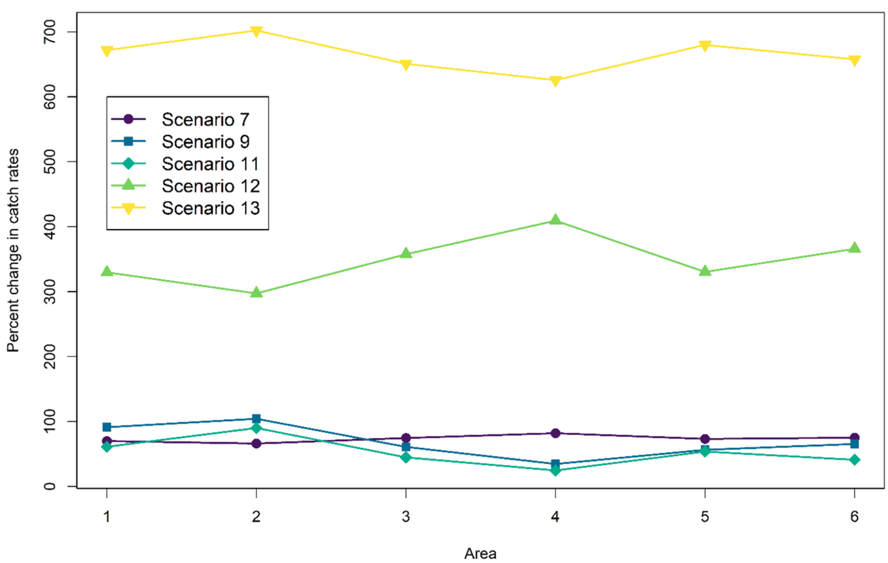


**FIGURE 5** Percent change in fleet 1 (recreational) catch rates by area for all management scenarios with percent changes greater than 50%. Catch rates were measured as the number of landed fish per unit effort.

with a 19.2% increase in landings and catch rates. These two scenarios may bracket the range in dead discard reduction achievable via use of descender devices. In both scenarios, both landings and discards of fleet 2 increased as abundance grew.

Although a size limit (scenario 5) could help rebuild the stock, it also resulted in fewer landings and more dead discards for both fleets (Table 3) and a ~50% reduction in recreational catch rates. The size limit did result in notable increases in the average weight of landings, but this of course was due to smaller fish being returned to the water dead or alive.

Temporal measures had mixed effects (Table 3). Not surprisingly, shortening the season to keep Red Snapper (scenario 6) and lengthening it (scenario 7) had opposite effects on the population and fishery, in which the shorter season enhanced rebuilding but converted otherwise landed fish to dead discards and reduced recreational catch rates by ~45%. The longer Red Snapper season increased recreational catch rates (Figure 5) but was one of only two scenarios in this study that lowered the stock abundance, to such a degree that fleet 2 landings decreased despite the increase in asymptotic retention. Reducing fishing effort on the snapper–grouper complex (scenarios 8–11) was among the most effective management strategies for rebuilding Red Snapper, increasing the average weight of landings and increasing recreational catch rates. In fact, most of these gains could be achieved by reducing the effort of fleet 1 only, which is evidenced by comparing results of scenario 8 to scenario 10 and scenario 9 to scenario 11. The 25% reductions in effort (scenarios 8 and 10) increased catch rates by about 15%, and the 75% reduction increased catch rates by 25–104%, depending on the area (Figure 5). Management measures that allowed full retention of all recreationally caught fish (scenarios 12 and 13) achieved the desired goal of eliminating dead discards from fleet 1 entirely, but whether the stock size would increase depended on the duration of the fishing season. The longer season with full retention (scenario 12) was the second of only two scenarios in this study that lowered stock abundance. However, the shorter season with full retention (scenario 13) not only enhanced rebuilding while eliminating fleet 1 discards, but also resulted in increased landings of both fleets as well as average weights of fish caught. These two scenarios showed the largest benefits to recreational catch rates of all management measures considered in this study, with increases near 350% for scenario 12 and 650% for scenario 13 (Figure 5).

Spatial measures all resulted in some amount of stock rebuilding and reduced landings (Table 3). These measures increased recreational catch rates by at most 1% and in several scenarios reduced them. For example, closing areas to Red Snapper landings (scenarios 14–17) reduced catch rates in those areas by 100% (by design), with little or no benefit to other areas. Closing offshore areas (scenarios 18 and 22) reduced recreational catch rates of inshore areas by ~15% as a result of effort shifting. Dead discards were increased by spatial measures that banned retention of Red Snapper in offshore areas (scenario 14) or one of the three regions (scenarios 15–17); however, dead discards were decreased by measures that closed all fishing effort in those same locations (scenarios 18–21). Most of the benefits to stock rebuilding could be obtained by restricting fleet 1 effort only, as evidenced by comparing scenarios that were otherwise the same (i.e., scenario 18 and 22, scenario 19 and 23, scenario 20 and 24, and scenario 21 and 25). When closing offshore areas to fleet 1 effort only (scenario 22), it may seem counterintuitive that fleet 2 landings declined when their effort was unaffected and overall abundance increased. This only makes sense in the context of a spatial model in which fleet 2 effort leans toward nearshore areas and the shift of fleet 1 effort from offshore to nearshore reduced the nearshore abundance (but not overall abundance) and consequently the fleet 2 nearshore landings. The spatial measures most effective for rebuilding were the three scenarios that restricted landings or effort in the middle region (scenarios 16, 20, 24), which was not surprising given that, under prevailing conditions, that region has both the most Red Snapper and highest recreational effort (Table 2). Still, even among those three scenarios, the population status differed considerably, while landings levels were identical. Closing the middle region to fishing effort eliminated nearly all dead discards of Red Snapper.

Sensitivity analyses with steepness reduced to $h = 0.8$ generally showed the same patterns for each management measure as did the model runs with $h = 0.99$ (Table 3). However, for each scenario, the population response in terms of abundance and spawning biomass was more exaggerated (positive or negative) than with higher steepness. This difference occurred because the density dependence incurred by lower steepness allowed recruitment to have a greater potential response to management over a wider range of spawning biomass levels. However, despite larger population responses, the equilibrium age structure (proportion at age) was the same as it was with higher steepness because age structure was determined by the mortality schedule, which was a function of the management measure (and not steepness). Given the same age structure, mean age of the population and mean weight of landings were independent of steepness (Table 3). Steepness did affect the response of landings and discards to management but primarily quantitatively (not qualitatively), with a few exceptions. In scenario 8 (all effort reduced 25%), lower steepness led to an increase (rather than decrease) in commercial landings and discards, which occurred because of the larger increase in abundance. Similarly, and for the same reason, in scenario 18 (close offshore areas to all effort), landings of both fleets increased; discards of both fleets still decreased but by a lower percentage. In scenarios 19 (close northern region to effort) and 21 (close southern region to effort), recreational landings increased (rather than decreased) because of increased catch per effort in the middle region.

## DISCUSSION

In this paper, we developed a spatial population model to compare fishery management strategies to rebuild the stock of Red Snapper in federal waters of the U.S. Atlantic. The spatial structure accommodated the ontogenetic movement of Red Snapper from nearshore to offshore areas, as well as area-specific abundance and fishing effort. Such structure can better account for the heterogeneity observed in the real system (Cadrin et al. 2023), and it allowed for the consideration of spatially explicit management measures. The 25 management measures that we evaluated varied considerably in their abilities to address the management objectives of (1) rebuilding the stock abundance, spawning biomass, and age structure, (2) increasing landings, and (3) decreasing dead discards. With regard to these objectives, we draw several main conclusions from our simulations. First, the measures most effective at rebuilding the stock are those that limit fishing effort, either throughout the year or in locations where Red Snapper are most abundant. Indeed, our simulations showed that most of the benefits could be achieved by limiting recreational effort alone, rather than effort from both recreational and commercial sectors, perhaps in part because a permitting system already restricts access of snapper–grouper commercial fishing. Second, input controls can reduce dead discards; output controls generally do not. The exception is the output control of discard mortality mitigation. Increased use of descender devices could reduce dead discards and thereby increase abundance and landings, but the effectiveness for Red Snapper management depends greatly on the degree of mitigation. Finally, a measure that limits recreational effort in the snapper–grouper fishery throughout the year (not just within the Red Snapper season) combined with full retention of all fish caught has potential to meet all management objectives considered here: rebuilding the stock, increasing landings, and reducing dead discards. It also resulted in the largest increases in recreational catch rates, and it may have potential to reduce or end overfishing for multiple species in the snapper–grouper complex.

We evaluated the management measures one at a time to isolate the effect of each strategy. Of course, in practice, multiple strategies could be applied simultaneously, with potentially cumulative benefits. For example, any management strategy that still results in discards should be accompanied with regulations or outreach programs designed to increase use of descender devices.

We see no downside to the increased use of descender devices; however, we acknowledge the wide variation in their estimated levels of mortality mitigation for Red Snapper in the Atlantic Ocean. At the time of the most recent stock assessment (SEDAR 2021), the most comprehensive study of this topic was that of Vecchio et al. (2020), implemented as our scenario 3. A more recent study by Runde et al. (2021) found a higher level of mitigation and was implemented as our scenario 4. The Runde et al. (2021) study was conducted at only a single location, but similarly high levels of mitigation have been found for Red Snapper in the Gulf of Mexico (Bohaboy et al. 2020). In addition, the level of mitigation could be affected by the relationship between depth and release mortality, given the expectation of increased barotrauma (and therefore more mitigation) when fish are caught in deeper water. The Vecchio et al. (2020) study showed a smaller effect of depth than has been described in the Gulf of Mexico (Campbell et al. 2014; Pulver 2017), which could be because Red Snapper in the Atlantic Ocean are generally caught in shallower water than in the Gulf of Mexico; more than 96% of 6999 fish caught in Vecchio et al. (2020) were shallower than 40 m. If so, release mortality of Red Snapper in the Atlantic Ocean may be more frequently caused by hook injury or depredation than by barotrauma. Nonetheless, dead discards from the recreational fleet have been identified as the primary driver of overfishing (SEDAR 2021), and discard mortality is likely to remain a wasteful use of Red Snapper and other species in the snapper–grouper complex for as long as private recreational effort remains unrestricted.

Results should be considered in light of several key model attributes. First, our operating model was deterministic and we did not attempt to capture effects of stochasticity in population or fishery dynamics. This was a calculated decision on our part to focus on expected (equilibrium) values, but we acknowledge that variance in model results could differ across the various management measures. Second, we assumed that new recruits to the population were distributed spatially according to the prevailing relative abundance by region as estimated from survey data (Cao et al., in review). In reality, the spatial distribution of recruits could fluctuate through time along with variability in oceanographic currents and perhaps with trends as the stock rebuilds (Karnauskas et al. 2022). Third, the spawner–recruit relationship assumed a steepness value of 0.99, which essentially models recruitment as a nearly constant value. Again, this was a calculated decision and was based on the most recent stock assessment of Red Snapper (SEDAR 2021), but if steepness were lower, the spawner–recruit relationship would have more curvature and we would expect to see a greater response to management measures (as shown by our sensitivity analyses with steepness of 0.8). Fourth, our two scenarios evaluating the effects of descender devices (scenarios 3 and 4) assumed 100% usage in the recreational fleet, which is idealistic (Responsive Management 2022), and thus those results should be interpreted as upper bounds on positive effects and lower bounds on negative effects. Finally, although we modeled multispecies fishing effort, our focus was on a single species, Red Snapper. For now, management by the SAFMC largely does occur on a species-by-species basis, but our model could be extended with additional stocks to support multispecies management of this multispecies fishery (Plagány et al. 2014). Such an extension would allow for the exploration of management trade-offs that might occur across species with different spatial patterns in abundance.

Our evaluation of management measures should not be confused with classical management strategy evaluation (MSE; Punt et al. 2016), although the goals are similar. A full MSE process would involve meetings with stakeholders and managers to develop fishery objectives, as well as modeling key uncertainties within a feedback loop between management implementation, the fishery, and the population. The management measures evaluated here are based on the experience of the authors with this fishery and on our interpretation of management objectives. Our analysis is better characterized as simulation modeling than as MSE, and our goal was to provide strategic, not tactical, management advice. In addition, our results could be useful for informing development of a full MSE in terms of identifying which types of management measures are most likely to succeed.

Simulation modeling of potential management measures has a rich history in fisheries (e.g., Johnson 1995; Cooke 1999; Mapstone et al. 2008; Butterworth et al. 2010; McQuaw et al. 2021). More than half a century ago, Paulik (1969) predicted that simulation models would become commonplace in the resource management agency of the future, and more recently, Bohaboy et al. (2022) recommended the increased use of simulation analyses to inform fishery managers. Much like our study, Bohaboy et al. (2022) simulated various management approaches for Red Snapper as part of a multisector fishery. Although their focus was on Red Snapper in the Gulf of Mexico and the simulation framework was quite different from ours, the two studies share some implications for management. Notably, most output controls common for recreational fisheries, such as size limits or landings quotas, are unlikely to achieve management goals. However, the mitigation of discard mortality, for example with descender devices, could provide tangible benefits, including fewer dead discards, increased spawning biomass, increased catch rates, and larger fish caught.

Red Snapper are part of a multispecies fishery, and the stock is experiencing overfishing, not because landings are too high but because of discard mortality from the private recreational fleet (SEDAR 2021). This occurs because fishing effort applies generally to the complex of species such that output controls (with the exception of descender devices) will increase dead discards, not reduce them (Abbot and Wilen 2009). Writing about groupers in this complex, Huntsman et al. (1999) stated that “only two options remain for reducing *F* sufficiently: areal closures and closed seasons.” Since then, recreational effort has increased along with the growing coastal human population (Thunberg and Milon 2002; Shertzer et al. 2019); more specifically, private recreational effort has increased by about 45% (National Marine Fisheries Service, Fisheries Statistics Division, personal communication). Simultaneously, fishing power (catchability) has grown as high-technology navigation and sonar systems become more affordable, more precise, and more informative in their fish identification and mapping capabilities. The rise of social media and recreational fishing organizations has allowed anglers to share information about fishing “hot spots,” even while on the water. Any efforts to reduce efficiency, for example through gear modifications, would occur against this backdrop of increasing fishing power. The cumulative effect for reef-associated fisheries in the southeastern USA is that increased recreational fishing effort with increased fishing power has been increasingly concentrated into relatively small, well-known, and well-advertised locations of hard-bottom habitat or artificial reefs.

In the southeastern U.S. Atlantic, commercial effort is constrained by limited entry permits, but the private recreational fleet is the dominant source of fishing mortality and remains open access (Shertzer et al. 2019). As described by Cox and Walters (2002), “In open-access fisheries, managers mainly react to the quality deterioration problem by trying to produce more fish and by using simple regulations such as bag and size limits. These tactics have never worked.” Indeed, open-access natural resources would seem inescapably destined for the tragedy of the commons (Hardin 1968). With recognition that marine resources are exhaustible and that recreational angling has high impact (Arlinghaus et al. 2019), especially in southeastern U.S. marine fisheries (Coleman et al. 2004; Shertzer et al. 2019), we reiterate here the decades-old call for restricted-effort fishery management, based on the principles of economics, ecology, and angling quality (Waters 1991; Cox and Walters 2002). In wildlife management, hunting effort is controlled through the use of open and closed seasons, lotteries, and tags. Similar systems would work for restricting private recreational fishing effort for the snapper–grouper complex in the southeastern USA (Johnston et al. 2007). In particular, open and closed seasons or lotteries appear particularly promising to control private recreational snapper–grouper effort, and either approach, if sufficiently structured, could be coupled with full retention of the catch to eliminate the wasteful practice of discarding. For-hire recreational fleets (headboats, charter boats) could be managed similar to commercial fleets with permits for limited entry to maintain their year-round business model and because their current level of effort is dwarfed by that of the private recreational fleet (Doerpinghaus et al. 2014).

## Management implications

For Red Snapper and other stocks in the snapper–grouper complex, use of descender devices not only reduces discard mortality, but can also address the objectives of increasing abundance, spawning biomass, and landings, and it can easily be paired with other management measures. However, overfishing in the snapper–grouper complex largely results from too much private recreational effort, which if left unrestricted will continue to increase as human (angling) populations grow in coastal areas of the southeastern USA. Thus far, resource managers have attempted to regulate the snapper–grouper fishery primarily using traditional approaches, such as species-specific size limits, trip limits, bag limits, and seasonal closures. For this multispecies fishery, such approaches have resulted in a commercial sector that is operating well below its economic potential (Liese and Crosson 2023) and have promoted the wasteful practice of discarding (Harrington et al. 2005), both of which are contrary to management objectives. Large area closures, although potentially effective for rebuilding the stock, are politically challenging to implement and could produce considerable localized economic costs as has been shown for the creation of marine protected areas (Sanchirico et al. 2002).

Our study demonstrates that restricting recreational fishing effort is effective at achieving management objectives for Red Snapper and would likely also benefit multiple species in the snapper–grouper complex that have shown declines in abundance and recruitment (Bacheler and Ballenger 2018; Bacheler et al. 2023; Wade et al. 2023). Other authors have discussed how restricting recreational effort can lead to increased catch rates and larger size of fish caught, both of which increase angler satisfaction (Cox and Walters 2002; Mapstone et al. 2008). Cox and Walters (2002) note that opposition to implementing restricted effort is common but also short-lived if improvements in trip quality become evident, as might occur with management measures that limit effort but allow for full retention (e.g., scenario 13). To effectively end overfishing of Red Snapper and other stocks in the snapper–grouper complex, as required by the Magnuson–Stevens Fishery Conservation and Management Act, policymakers should consider restricting effort of the private recreational fleet in this multispecies fishery.

### ACKNOWLEDGMENTS

We thank N. Klibansky, anonymous reviewers, the editor, and the associate editor for reviewing an earlier draft of this paper. Funding was provided by the National Oceanic and Atmospheric Administration. The scientific results and conclusions, as well as any views and opinions expressed herein, are those of the authors and do not necessarily reflect those of any government agency or institution.

### CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

### DATA AVAILABILITY STATEMENT

Data used for model development are available at https://sedarweb.org/assessments/sedar-73/.

### ETHICS STATEMENT

This work was completed in accordance with all institutional ethical guidelines.

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