



Modeling temporal closures in a multispecies recreational fishery reveals tradeoffs associated with species seasonality and angler effort dynamics

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ABSTRACT

Seasonal closures are commonly used to reduce fishing mortality in recreational and commercial fisheries, but they may be less effective when effort is merely displaced to the open season or in multispecies fisheries that allow for discarding to continue while other species are targeted. The latter is especially true for the valuable multispecies recreational reef fish fishery in the Gulf of Mexico, where discard mortality is high and it can be difficult to avoid catching one species while fishing for others. We evaluated the utility of complete bottom fishing closures (in addition to already mandated harvest closures) that would temporarily prohibit recreational reef fishing as a means to control effort, reduce the amount of dead discards, and improve stock status of multiple species. In this study we developed age-structured population models for six Gulf of Mexico reef fish species that dominate the recreational catch, with each model linked to a monthly effort dynamic model for the recreational fishery. The effect of closing any given month(s) varied across species and resulted in tradeoffs, such that some closures may result in positive effects on biomass of one species and negative effects on others. For example, a spring closure was predicted to have positive effects on Red Snapper *Lutjanus campechanus* spawning stock biomass but negative effects on Gag grouper *Mycteroperca microlepis* due to the contrasting patterns in harvest rates during those months. These tradeoffs were associated with seasonal availability patterns and the degree to which anglers might shift effort to the open season. The closure scenarios that were most likely to reduce dead discards without negatively impacting harvest, spawning biomass, or total effort occurred in late winter and early spring (March & April). In evaluating seasonal fishing closures, the gains in biomass and reductions in dead discards must be weighed against the socio-economic tradeoffs, in terms of lost effort-generated revenue at various spatial and temporal scales and angler dissatisfaction.

1. Introduction

Seasonal closures are a management strategy widely used to control and ultimately limit fishing-related mortality in marine and freshwater fisheries. There are two general types of seasonal closures in fisheries – “harvest closures” which restrict harvest by making it illegal to possess a species during the closed season and “fishing closures”, addressed here, that prohibit some or all gear types during the closed season. Often, seasonal harvest and fishing closures are intended to protect spawning or aggregating adults, thereby reducing catchability and fishing mortality (Russell et al., 2012). In many cases, seasonal closures are quite effective, easy to enforce, and generally accepted by managers for their simplicity. However, seasonal closures may be ineffective when fishers simply allocate more effort to the open season (Baum et al., 2003; GMFMC, 1999; O’keefe et al., 2013) or in mixed fisheries

where discarding continues during a species’ closed season while anglers target a different species. Highly restrictive seasonal closures can lead to periods of intensive effort or ‘derby’ fisheries resulting in unsafe fishing practices and market gluts, such as with Gulf of Mexico Red Snapper (Agar et al., 2014). Additionally, temporal closures may cause effort to shift entirely onto different species or species complexes thereby having unintended consequences on other fisheries (Rijnsdorp et al., 2012). There are also potential negative economic outcomes resulting from implementation of seasonal closures, especially if the closure results in a large overall reduction of fishing trips which in turn could have cumulative social and economic effects on fishing communities (ASMFC, 2005).

The multispecies recreational reef fishery in the Gulf of Mexico (GOM) is very popular with avid anglers and regulated under a combination of species-specific seasonal harvest closures, size limits, and

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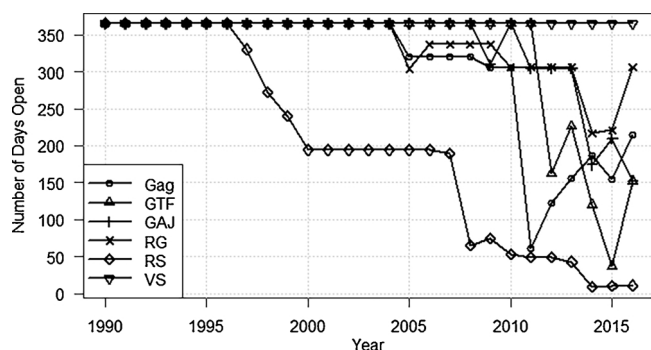


Fig. 1. Length of Gulf of Mexico recreational fishing seasons from 1990 to 2016 for Gag, Gray Triggerfish (GTF), Greater Amberjack (GAJ), Red Grouper (RG), Red Snapper (RS), and Vermilion Snapper (VS).

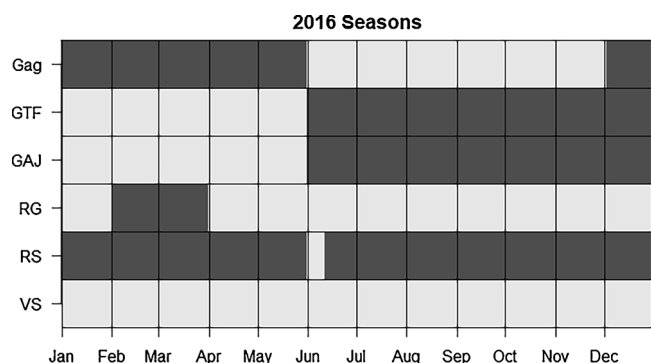


Fig. 2. Recreational fishing seasons in federal waters of the Gulf of Mexico for Gag, Gray Triggerfish (GTF), Greater Amberjack (GAJ), Red Grouper (RG), Red Snapper (RS), and Vermilion Snapper (VS) during 2016. Dark cells indicate the species is closed to harvest.

bag limits that were implemented after many stocks were determined to be overfished during the 1990s and 2000s. In 2017, of the nine GOM reef fish stocks for which status is known, only one was experiencing overfishing (Greater Amberjack, *Seriola dumerili*) and three were determined to be overfished (Gray Triggerfish *Balistes capriscus*, Greater Amberjack, and Red Snapper *Lutjanus campechanus*). Prior to 1996 there were no recreational harvest closures for reef fish in the GOM, but since then, the number of days open to fishing has varied over time for each species (Fig. 1). In 2016, the number of days open to fishing ranged from 11 for Red Snapper to 152 for Gray Triggerfish and Greater Amberjack, to 365 for Vermilion Snapper *Rhomboplites aurorubens* (Fig. 1). However, current recreational fisheries management mandates different harvest seasons for different species, such that there is little overlap in harvest seasons, and in 2016 there was not a month where all species were open or all species were closed to harvest (Fig. 2).

Recent management challenges in the GOM have been associated with effort reallocation around closed seasons. For example, the seasonal closure for commercial grouper fishing in the GOM failed to reduce harvest as expected because effort shifted to before and after the closed period (GMFMC, 1999). Additionally, recreational harvest of Gray Triggerfish and Greater Amberjack was underestimated in management projections by 21% and 30% respectively, due to unanticipated effort shifting, leading to quota overage and an early closure to the harvest season (GMFMC, 2016). Because reef fish species occupy the same habitats and are targeted with similar gear, it can be difficult to avoid catching one species while fishing for others such that the harvest closures do not limit fishing effort well. Further, discard mortality is generally high in these fisheries, owing to depth-related barotrauma, and in some cases the seasonal closures have caused an increase in dead discards. For species such as Gag Grouper *Mycteroperca microlepus*, a large portion of the total mortality is due to regulatory

discards (SEDAR, 2016), i.e. fish that are caught and then released because they are smaller than the minimum size limit or caught during the closed season while fishing for a different species. This has raised concerns by angler groups and managers alike, and prompted an evaluation of bottom fishing closures that would temporarily prohibit all recreational fishing with hook-and-line on the seafloor in reef habitats, potentially reducing the amount of dead discards in the fishery and improving stock status of multiple species.

In this study, we evaluated the potential outcomes of temporary closures to all ‘bottom fishing’ in the private recreational fleet of the GOM. Changes to single-species harvest seasons were not evaluated. When evaluating fishing closures in multispecies fisheries two important considerations must be made. First, species availability varies throughout the year and therefore the exact timing of a closure makes it possible that a single closure may have different effects on different species. Second, whether or not effort is actually reduced (or transferred to other fisheries) or redistributes from closed to open times, will determine if an improvement in overall fishing-related mortality can be achieved for a given species. These dynamics increase the need to develop robust evaluations of closures that account for dynamic fishing effort driven by angler behavior, as well as accounting for how different species would individually be affected. Our objectives were to a) determine which, if any, bottom fishing seasonal closures could lead to biomass gains over the long-term (20 years) for multiple species, b) identify the harvest efficiency and reductions in dead discards possible, and c) identify potential tradeoffs among species associated with effort displacement and the tradeoffs with overall changes in fishing effort.

2. Methods

To evaluate bottom fishing seasonal closures to the GOM private recreational fleet we developed age-structured population models for six reef fish species: Gag Grouper, Gray Triggerfish, Greater Amberjack, Red Grouper *Epinephelus morio*, Red Snapper, and Vermilion Snapper. Combined, these species made up approximately 85% of the total harvest by weight in the private recreational fishing sector outside state waters during 2016 (> 10 miles in West Florida, > 3 miles in other GOM states) (Personal communication from the National Marine Fisheries Service, Fisheries Statistics Division. [19 June 2018]). Throughout this paper, we refer to ‘harvest’, ‘landings’, and ‘retained’ for fish that were kept by the angler, ‘discarded’ or ‘released’ for fish that were caught and returned to the water, and ‘dead discards’ as those that died after release. Thus the ‘total catch’ is composed of both harvest and discards and the ‘total killed’ included both retained fish and dead discards. We estimated monthly harvest and discard rates using data from the National Marine Fisheries Service Marine Recreational Information Program (MRIP) in order to inform monthly patterns in catchability and therefore fishing mortality, attributed to both harvest and discards. The age-structured models were parameterized using the most recent stock assessments for each species, all conducted with the Stock Synthesis 3 framework (Methot and Wetzel, 2013), to maintain consistency with current stock status and accepted population parameters. In each population model, fishing mortality by the private recreational fleet was modeled with a monthly effort dynamic model that accounted for within year effort shifting and long-term effort response as a function of fish abundance. This simulation framework allowed us to implement closures of one or more months and estimate changes in population size, harvest, discards, and fishing effort. Details of the methodology are provided in the following sections.

2.1. Monthly catch-rate analysis

A main component of MRIP is the Access Point Angler Intercept Survey (APAIS) which consists of in-person interviews of anglers who have completed their fishing trip. The APAIS data, also referred to as ‘dockside intercepts’, are used to generate catch rates, among other

information, that when multiplied by fishing effort from a telephone survey provide estimates of the total number of fish harvested and discarded along with the number of trips taken (www.CountMyFish.noaa.gov). We compiled monthly catch, harvest, and discard rates (number of fish per angler hour) from the publicly available MRIP APAIS dockside intercept dataset from 2012–2015. The intercept data were analyzed using a generalized linear model (GLM), following the delta approach commonly used to generate indices of abundance for stock assessment (Lo et al., 1992; Maunder and Punt, 2004). In this approach, GLMs were fitted to presence/absence data with a binomial distribution and to numbers caught per angler per hour for positive trips only with a log normal distribution, and the index was taken as the product of the two submodels. Each data point consisted of (among other information) the number of fish caught, retained, and discarded along with the number of anglers, time spent fishing, area fished (state or federal waters), state in which the intercept was made, month, and year. Prior to running the GLM, any states or areas that did not account for at least 1% of positive occurrences were removed from the dataset. Independent variables considered were state, year, month, and area fished (all treated as factor variables) with the final models chosen using stepwise forward selection based on AIC criteria.

To generate the monthly index we first calculated the least-squares mean for each submodel and month as the average value across all other factor levels from a reference grid of predictions. Median monthly catch, harvest, and discard rates with confidence intervals were developed using Monte Carlo simulations. This was done by multiplying the standard error of the least-squares mean monthly estimate by 10,000 random normal deviates $X \sim N(\mu = 0, \sigma = 1)$ and adding them to the monthly least-squares mean value. When the log-normal and binomial least-squares means were correlated (Pearson's correlation p -value < 0.05) the correlation coefficient ρ was used to adjust the error deviate of the log-normal model as $\rho X_1 + X_2 \sqrt{(1 - \rho^2)}$, where X_1 and X_2 are random normal deviates from the binomial and log-normal models, respectively. For each Monte Carlo iteration, the binomial estimate was transformed from logit space to probability and multiplied by the exponentiated log-normal estimate to obtain a distribution of index values from which to calculate the catch rate statistics. A total of 36 GLMs were run (6 species \times 3 catch types \times 2 submodels) using the R statistical computing software package.

2.2. Age-structured projection models

A multi-species age-structured effort-dynamic simulation model was developed to estimate the effects of bottom fishing closure scenarios. An age-structured projection model was developed for each species and these were all driven by a common effort-dynamic model for a duration of 20 years. All species were last assessed under the Southeast Data Assessment and Review process using the Stock Synthesis 3 (SS3) assessment framework (Methot and Wetzel, 2013). The report files generated from each stock assessment model provided the necessary parameters to initialize the age-structured projection models (Table 1). The required parameters included age-specific vectors for size, weight, maturity or fecundity, sex ratios, natural mortality, numbers-at-age (beginning of terminal year), and fishery selectivities in terminal year. Additionally, the SS3 output files contained the estimated stock-recruit parameters of unfished recruitment (R_0), steepness (h), and unfished spawning stock biomass (SSB_0) used in the projections. Red Snapper was modeled separately for the eastern (Florida to Mississippi) and western (Texas and Louisiana) GOM, following the 2-area approach of the stock assessment that assigns 36% of recruits to the east and 64% to the west with no movement for older ages. For Red Snapper, SS3 fishing mortality rates, selectivities, and numbers-at-age were region-specific, with life history parameters shared among the two regions. The age-structured projection and effort-dynamic models are described in detail below and available as a single MS Excel file in Appendix A.

Instantaneous fleet-specific fishing mortality rates (F) were

estimated by the SS3 models in each year, along with selectivity (sel) and retention (ret) functions and a discard mortality rate ($Dmort$). For all fleets except the private recreational fleet, the fishing mortality rate remained constant throughout the duration of the simulation and the vulnerability-at-age to harvest (V) used in the projection model was taken as $V = sel(ret + (1-ret)Dmort)$, to account for processes of both harvesting and discarding fish across ages. For the private recreational fleet we accounted for retained catch and dead discards separately in our closure scenarios, and the age-specific vulnerability was separated into retention (V^{ret}) and dead discards (V^{disD}) to account for the smaller size structure of discarded fish, where $V^{disD} = sel(1-V^{ret})Dmort$ (Fig. 3). Fishing mortality was then partitioned into components of retention, F^{ret} , and dead discards, F^{disD} based on the proportion of fish killed by each process in the terminal year of the stock assessment where $F^{ret} = F(N^{ret}/N^{dead})$ and $F^{disD} = F(N^{disD}/N^{dead})$.

Population numbers-at-age were projected forward over the time period 2015–2035, in annual time steps, where the numbers-at-age in year-1 were taken as the terminal year abundance estimates from the stock assessments. For all other years, the number of age-0 recruits were calculated using the steepness formulation of the Beverton-Holt stock recruitment curve as implemented in SS3

$$N_{a=0,y+1} = \frac{4hR_0SSB_y}{SSB_0(1-h) + SSB_y(5h-1)} \quad (1)$$

The number of age 1+ fish were then modeled as

$$N_{a+1,y+1} = N_{a,y} \cdot \exp(-Z_{a,y}) \text{ where} \quad (2)$$

$$Z_{a,y} = M_a + (F_y^{ret} \cdot V_a^{ret}) + (F_y^{disD} \cdot V_a^{disD}) + \sum_{f=1}^n (F_f \cdot V_{f,a}) \quad (3)$$

where M is the age specific instantaneous natural mortality, F^{ret} and F^{disD} are the instantaneous fishing mortality rates in the private recreational fleet for retained catch and dead discards, respectively, and F_f is the fishing mortality from other fleets, f , each with their own vulnerability schedule $V_{f,a}$. In the case of Red Snapper, the numbers at age were modeled separately for each region with recruits calculated from total female SSB and distributed to the east and west at 36% and 64% respectively.

Year-specific fishing mortality rates applied in Eq. (3) for both retention and dead discards were calculated as the sum of monthly catchability multiplied by monthly effort,

$$F_y = \sum_m q_m \hat{E}_{m,y} \quad (4)$$

where q_m is a monthly catchability coefficient (one each for retention and discards) and $\hat{E}_{m,y}$ is fishing effort predicted at monthly time steps by the effort dynamic model described in Section 2.3. Baseline catchability coefficients, q , for the recreational fleet were calculated for each species by dividing the F^{ret} and F^{disD} by the total number of observed trips corresponding to the terminal year, i.e. $q = F/E$. For Red Snapper, baseline q was calculated separately for the recreational fleet in each region using observed region-specific effort, but scaled to monthly values using a common catch rate. The baseline q values were then multiplied by the mean-scaled median monthly harvest and discard rates from the MRIP GLMs to arrive at q_m for both retention and discards. This formulation allows for the F_y to change with monthly effort, and depending on the values of q for retention and discards in the months when fishing occurs, this could result in a net increase or decrease in F over the course of an entire year.

2.3. Effort dynamic model

As in previous studies that modeled effort dynamics in recreational fisheries (Allen et al., 2013; Arlinghaus et al., 2009; Cox et al., 2002; Post et al., 2003) we assumed that a major motivation driving anglers to fish is the abundance or biomass of the harvestable stock. For projecting

Table 1

Parameters used in the age-structured projection models. Units of length are centimeters, body weight in kilograms, and spawning biomass is expressed as number of eggs or metric tons. In most cases, the parameters were obtained directly from the latest stock synthesis (SS) assessment models. Those not obtained from SS but calculated for use in the projection models are indicated by asterisks (*). Spawning biomass and fishing mortality thresholds listed here are based on 30% of unfished spawning biomass, and may differ from the reference points adopted by management. In the Red Snapper SS model, a vector of fecundity-at-age was input instead of using a maturity-at-age relationship.

Parameter	Gag	Greater Amberjack	Gray Triggerfish	Red grouper	Red Snapper	Vermilion Snapper
SS Model Dimensions						
year-T	2015	2015	2013	2013	2014	2014
N-areas	1	1	1	1	2	1
N-fleets	6	4	5	5	14	4
N-sexes	2	1	1	1	1	1
Mortality, Growth, and Maturity						
Max. Age (A_{max})	31	10	10	20	20	14
Linf	132.21	143.6	58.97	82.72	85.64	34.4
k	0.11	0.21	0.14	0.12	0.19	0.33
Min Age for growth (A_{min})	1	0.5	0.5	1	0.75	0.5
Length at A_{min}	27.96	10.00	28.30	17.29	9.96	11.83
Length-Weight a	8.75E-06	7.05E-05	2.16E-10	5.99E-06	1.67E-05	2.19E-05
Length-Weight b	3.08	2.63	3.01	3.25	2.95	2.92
M at age-0	0.55	0.78	0.79	0.58	1.00	0.23
M at A_{max}	0.11	0.25	0.25	0.13	0.08	0.20
Age at 50% maturity	3.55	82.50	31.00	2.80	NA	14.09
Maturity slope	−2.83	−0.10	−0.07	−1.15	NA	−0.57
Hermaphroditic	protogyn.	no	no	no	no	no
Age at sex transition	10.745	NA	NA	NA	NA	NA
Stock-Recruit						
unfished recruitment: $\ln(R_0)$	8.52	7.93	9.76	10.39	12.04	10.19
steepness (h)	0.86	0.85	0.46	0.99	0.99	0.57
unfished SSB (SSB_0)	24,909	18,836	3.24E + 10	8,236,070	4.91E + 12	6.57E + 11
SSB_0 units	fem. mat. B	mat B	eggs	eggs	eggs	eggs
Private Recreation Fleet						
Apical F: retention (F^{ret})*	0.10	0.33	0.25	0.14	0.15	0.08
Apical F: discards (F^{disc})*	0.16	0.23	0.01	0.04	0.40	0.01
Discard mortality (D_{mort})	0.12	0.20	0.05	0.12	0.10	0.15
Proportion targeted*	0.39	0.05	0.01	0.12	0.41	0.01
Stock Status in year-T						
SSB_{year-T}	9,688	1,685	6.04E + 09	2.95E + 06	6.90E + 11	2.06E + 11
$SSB_{threshold} = 0.3 * SSB_0$	7,473	5,651	9.72E + 09	2.47E + 06	1.47E + 12	1.97E + 11
$SSB_{year-T} / SSB_{threshold}$	1.30	0.30	0.62	1.19	0.47	1.05
F_{year-T}	0.07	0.34	0.13	0.13	0.05	0.08
$F_{threshold} = F @ SSB_{threshold}$	0.19	0.20	0.17	0.21	0.05	0.57
$F_{year-T} / F_{threshold}$	0.37	1.73	0.76	0.60	0.87	0.13

inter-annual changes in fishing effort over the twenty-year simulation period, we assumed that the number of trips taken in a year, \hat{E}_y , was related to the size of the population vulnerable to harvest, defined as numbers-at-age times retention-at-age ($\sum_a N_a \cdot V_a^{ret}$) and summed over all species, and followed the logistic function

$$\hat{E}_y = 1 / (1 + e^{(NV_h - NV_{y-1}) / \sigma NV_h}) \cdot E_{max} \quad (5)$$

This assumes that recreational effort is driven by total reef fish abundance, and not related more or less strongly to any given species or other attributes of fishing (see Section 4.2). Here, the NV_h is the total vulnerable numbers of fish at which half of maximum effort is realized, NV_y is the total vulnerable numbers in year y , and E_{max} is the maximum allowable effort (i.e. the upper asymptote of the logistic curve) taken as the sum of maximum observed monthly efforts during 2004–2015. The sigma parameter controls the steepness of the logistics curve where a value near 1 results in a slow effort response and smaller values generate faster responses to changes in abundance. For our baseline scenario we assumed a moderate effort response ($\sigma = 0.9$), but due to uncertainty about the shape (or existence of) an effort response curve, we also evaluated scenarios under constant effort and a fast response with $\sigma = 0.3$ (Fig. 4). When effort dynamics were turned on, the NV_h parameter was calculated so that the response curve passed through the point of observed abundances and effort during the first year of simulation.

Predicted annual effort calculated in Eq. (5) was then distributed proportionally to months,

$$\hat{E}_{m,y} = \hat{E}_y \cdot \frac{E_m}{\sum E_m} \quad (6)$$

where E_m is the observed monthly effort in 2015 from the recreational survey datasets. To simulate a closure, each month was assigned a status, s ($1 = \text{open}$, $0 = \text{closed}$), and the number of affected trips, $A_{m,y}$, in each month and year was calculated as $A_{m,y} = \hat{E}_{m,y}(1-s_m)$. A proportion of the annual affected effort ($\sum A_{m,y}$) was then redistributed to open months based on the observed monthly effort proportions in 2015. The degree to which anglers might decide to reallocate affected effort to other months is represented as a proportion, λ . Low values of λ imply that effort is either lost altogether or transferred out of the reef fish fishery (to perhaps an un-modeled nearshore fishery) while high values of λ allow effort to redistribute to open months. This is a key uncertainty in the model, therefore we evaluated all scenarios over λ values from 0 to 1 in increments of 0.25 representing none to complete effort redistribution. These redistributed trips were then added to the unaffected trips in the open months to provide the realized monthly effort, $\hat{E}_{m,y}$, as

$$\hat{E}_{m,y} = \left(\lambda \sum_m A_{m,y} \right) \frac{E_m^{s=1}}{\sum_m E_m^{s=1}} + \hat{E}_{m,y} \cdot s_m \quad (7)$$

Lastly, the realized effort was multiplied by monthly catchability coefficients to obtain the monthly fishing mortality rates $F_{m,y}$ for both retention and dead discards, which were summed over each year and included in Eq. (3) of the age structured model. For Red Snapper, the

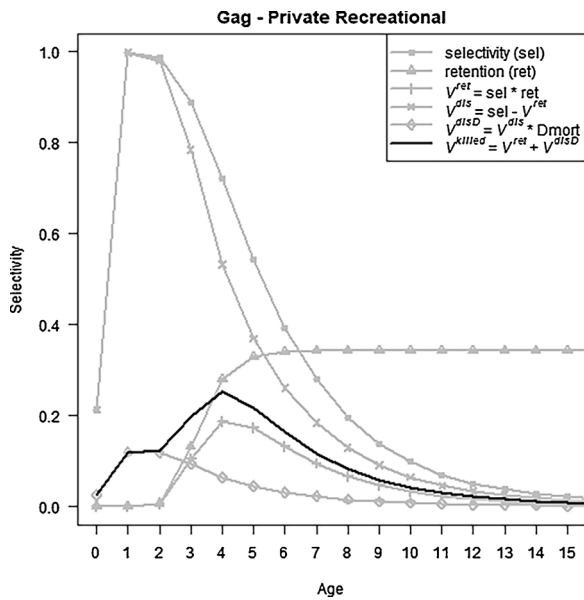


Fig. 3. Vulnerability-at-age of Gag being selected, retained, released and released dead by the private recreational fleet. Here, the vulnerability-at-age of being retained (V^{ret}) is the product of selectivity and retention. The vulnerability-at-age of fish being released is equal to selectivity minus retained catch, and the proportion that die after being released (V^{disD}) is the product of releases and a discard mortality rate. Finally, the vulnerability to being killed due to harvest and discards is the sum of V^{ret} and V^{disD} . All selectivity and retention parameters were taken from the terminal year of the stock assessment.

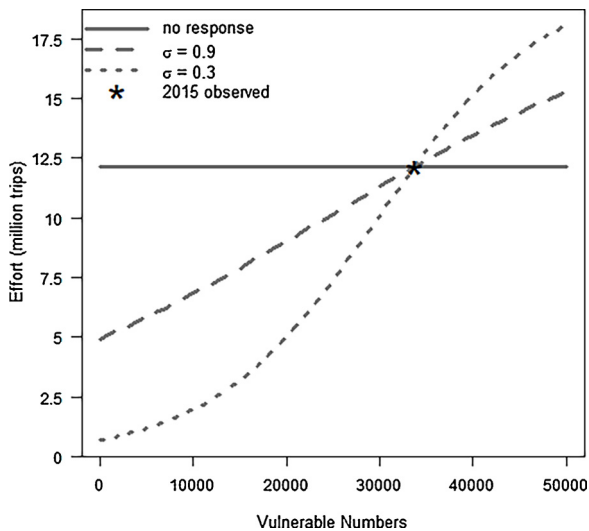


Fig. 4. Effort response shapes used to model long term changes in fishing effort as a logistic function of abundance of vulnerable sized fish. Three shapes were evaluated in the model including no response, the baseline scenario where $\sigma = 0.9$, and a fast effort response with $\sigma = 0.3$. Each curve was fit to pass through the vulnerable numbers and observed effort in 2015, based on the terminal year of each stock assessment and MRIP effort estimates.

realized monthly effort was partitioned to east and west regions based on 2015 observed monthly proportions of effort in each region.

2.4. Seasonal closure scenarios

The baseline (status quo) scenario consisted of no bottom fishing closures and a moderate effort response shape ($\sigma = 0.9$, Fig. 4). We evaluated monthly and quarterly closure scenarios, each over 5 levels of effort switching (λ) and 3 response shapes (σ) for a total of 240

scenarios ($240 = 16 \times 5 \times 3$). Importantly, each bottom fishing scenario is in addition to any species harvest closures already in place during 2012–2015, as reflected in the MRIP harvest and discard rates. An important motivation of this work was to explore whether bottom fishing seasonal closures could potentially reduce the amount of discards in the fishery. Therefore, we calculated the number of fish harvested (H) and discarded dead (D) based on the mean numbers-at-age in each year, $\bar{N}_{a,y} = N_{a,y}(1 - \exp^{-Z_{a,y}})/Z_{a,y}$, where

$$H_y = F_y^{ret} \sum_a \bar{N}_{a,y} \cdot V_a^{ret} \quad (8)$$

and

$$D_y = F_y^{disD} \sum_a \bar{N}_{a,y} \cdot V_a^{disD} \quad (9)$$

Harvest efficiency is expressed as the ratio of harvested fish to the total number of fish killed, $H/(H + D)$. Each scenario generated estimates of spawning stock biomass, harvest, discards, harvest efficiency, vulnerable numbers, and fishing effort for six reef fish species (summed across regions for Red Snapper). Outputs metrics were expressed as the percent change (\pm) in the terminal year (year-20) of the simulation relative to that in the terminal year of the status quo scenario. Lastly, we combined results across species by averaging the percent change, weighted by the proportion of dockside intercepts that indicated a targeted fishing trip for each species in the MRIP dataset (Table 1). This was intended to capture the relative importance of each species to private recreational anglers, and also serves to simplify interpretation of results.

3. Results

3.1. Recreational catch rates

Catch, harvest, and discard rates (fish per angler per hour) were variable across months for each species (Fig. 5). Of particular importance to this study were the contrasting patterns in catch rate, especially between groupers and snappers, and the high rate of discarding that occurred throughout the year for all species. Final equations and coefficients for all 36 GLM models are provided in Appendix A. Gag and Red Snapper, the two most often targeted species in the reef fish complex, exhibited near opposite patterns in catch rate. Gag catch rates were highest in November (0.241 fish/angler hour), remained high through March (0.178), and declined during the spring to a low of 0.054 in June. Conversely, Red Snapper catch rates were lowest from November (0.061) to February (0.085) and highest in May (0.421), June (0.648) and October (0.488). A similar pattern was observed for Red Grouper and Vermilion Snapper, with the former having higher catch rates during the winter spawning season (0.710–0.562) and the latter with peak catch rates during May (0.159). The monthly trend in catch rates for Greater Amberjack and Gray Triggerfish differed from those of the groupers and snappers. Greater Amberjack exhibited high catch rates in March (0.115) and August (0.122). Gray Triggerfish catch rates were variable throughout the year with a peak of 0.134 centered on September. The effect of regulatory closures was obvious in the catch rate analysis, leading to median harvest rates near zero for months with regulatory closures in place during 2012–2015 (Figs. 2 and 5). Uncertainty in the catch, harvest, and discard rate estimates was high for Greater Amberjack, Gray Triggerfish, and Vermilion Snapper (Fig. 5) because they were targeted less often and therefore had fewer positive occurrences in the APAIS intercept dataset.

3.2. Seasonal closures

In general, positive effects on SSB were predicted when closures were assigned to months with the highest catch rates, longer bottom fishing closures resulted in larger SSB increases compared to the status

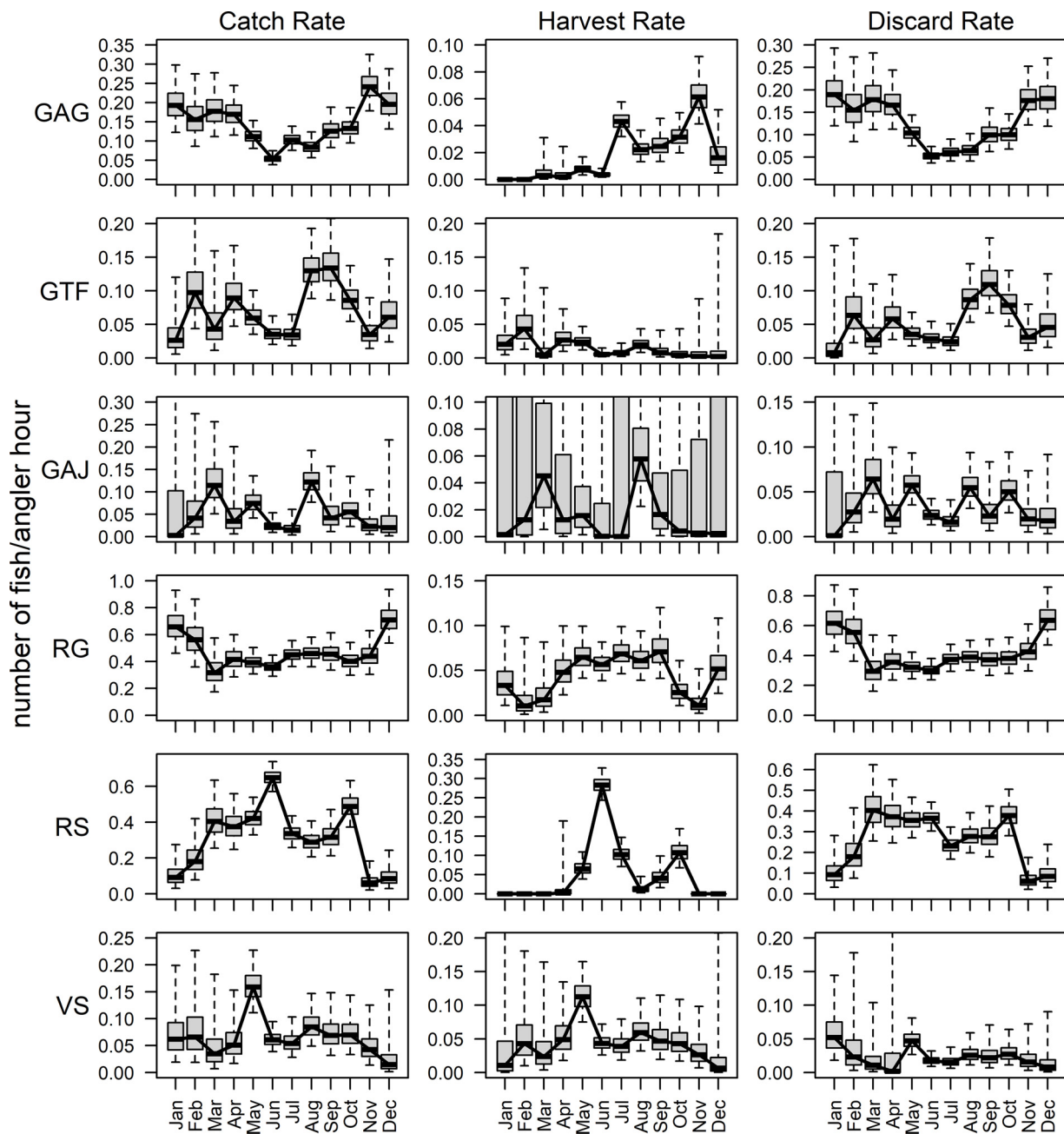


Fig. 5. Median monthly catch, harvest, and discard rates (fish per angler hour) from MRIP dockside intercepts 2012–2015 produced with a delta-lognormal GLM model. Each row is a species and each column is a catch rate type. The boxplots indicate the lower and upper 95% confidence intervals, the interquartile range, and median values (connected by line).

quo, and SSB was predicted to be lower at higher values of λ , which represents the proportion of affected trips shifting to open periods (Fig. 6). The model predicted SSB to remain within $\pm 5\%$ of status quo (across all λ) for most species in the monthly closure scenarios, with the exception of Gray Triggerfish in February (+6–8%) and April (+6–10%), Greater Amberjack in March (+21–30%) and August (+36–48%), and for Red Snapper in June (+7–8%) (Fig. 6). The only species predicted to experience a decline in SSB below -5% during the monthly scenarios was Greater Amberjack during June (-7–10%) and July (-9–12%) but only when λ was 0.75 or higher. A winter closure (Jan-Mar) was predicted to have no effect on SSB (SSB within $\pm 5\%$ across all λ) for all species except Gray Triggerfish (+6–13%) and Greater Amberjack (+13–38%), which exhibited a positive response in this scenario (Fig. 6). A spring closure (Apr-Jun) was predicted to cause increases in SSB greater than 5% across all λ for Red Snapper

(+6–10%) and Gray Triggerfish (+6–18%) but had mixed effects on Greater Amberjack (-13%–26%) and a negative effect on Gag (-7%) when λ was high (Fig. 6). The model estimated increases in SSB for almost all species and λ in the summer closure scenario (Jul-Sept) with largest increases predicted for Greater Amberjack (+25–67%), Gag (+3–12%), and Red Grouper (+4–10%) (Fig. 6). The fall closure (Oct-Dec) resulted in SSB increases for Gag (+7–13%), mixed effects on Greater Amberjack (-16–11%), moderate negative effects on Gray Triggerfish (-8% at $\lambda = 1$), and no effect for other species.

As expected, bottom fishing closures that occurred in months with high harvest rates (Fig. 5) led to large reductions in retained catch when compared to the status quo. For example, harvest was greatly reduced for Red Snapper in the June closure (-32 to 37%) and for Gag during the July (-15 to 22%) and November (-15 to 19%) closures (Fig. 7). Conversely, scenarios that restricted effort during current single-species

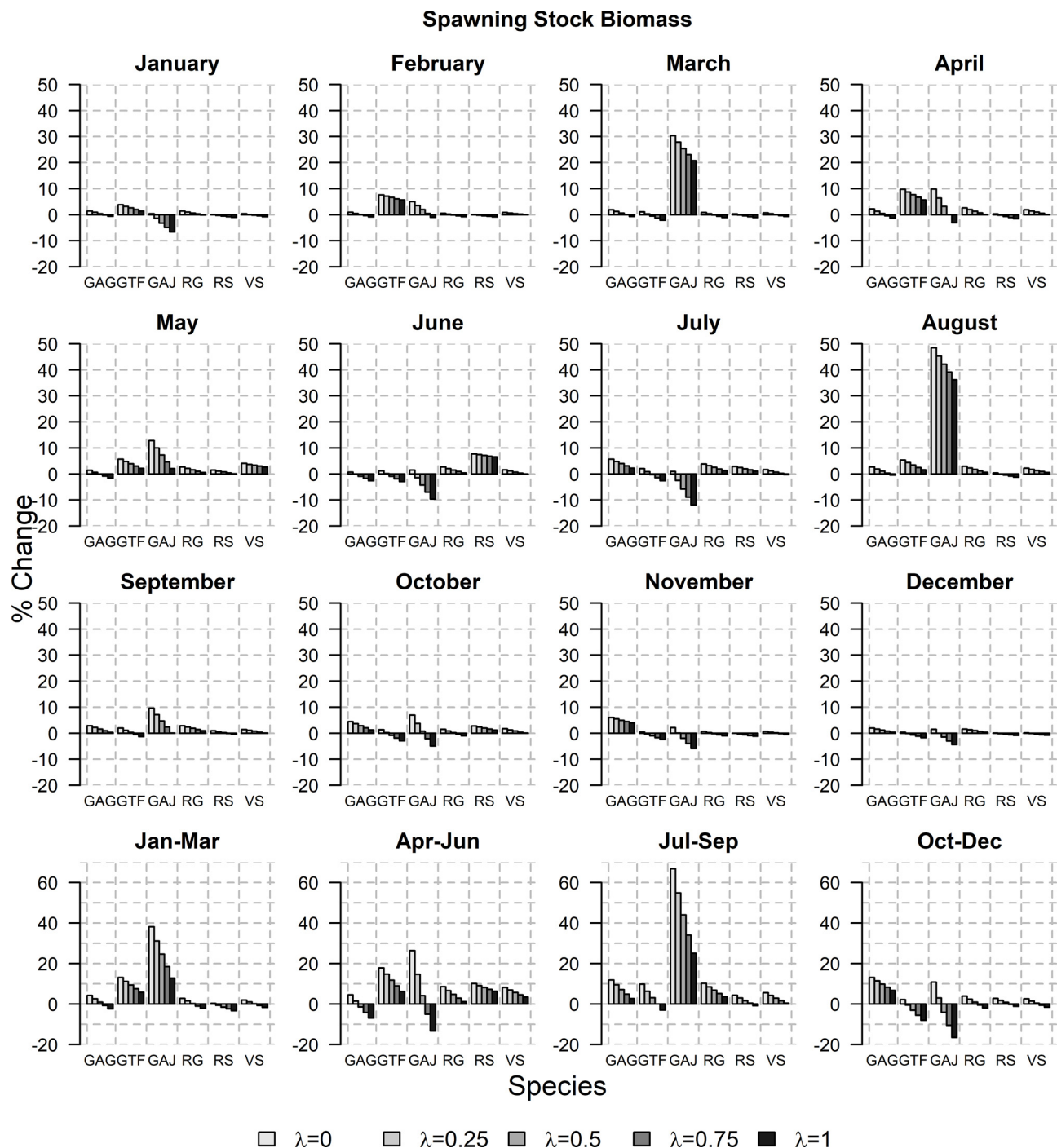


Fig. 6. Percent change in spawning stock biomass after 20 years of simulation compared to that predicted under status quo (no closures) for Gag, Gray Triggerfish (GTF), Greater Amberjack (GAJ), Red Grouper (RG), Red Snapper (RS), and Vermilion Snapper (VS) under monthly and quarterly bottom fishing closures with a moderate effort response shape ($\sigma = 0.9$). Scenarios were evaluated over a range of λ , which represents the proportion of affected trips that redistribute to other times of the year. Of these species, GTF, GAJ, and RS were in an overfished state at the beginning of the simulation.

seasonal harvest closures led to net increases in harvest, such as for Gag during bottom fishing closures from January to June which were predicted to cause an increase in harvest of at least 5% at high λ . This is because effort that would have occurred during a closed harvest season was affected by the bottom fishing restriction and displaced to months when the single-species harvest season is open. These effects were magnified in the quarterly closure scenarios (Fig. 7). Because discarding occurs year round, nearly all scenarios resulted in moderate to large reductions in dead discards (Fig. 8). Exceptions occurred when the bottom fishing closure overlapped with current species open seasons

and effort shifted to months when the harvest season was closed or discard rates were higher. For instance, discards increased by 5–7% for Gray Triggerfish in January and 7–11% for Vermilion Snapper in April.

Of particular interest is the likely effects of bottom fishing closures on harvest efficiency, defined as the proportion of total number of fish killed that is retained (Eqs. 8 & 9). In most cases, the monthly closures had little effect on harvest efficiency (Fig. 9) due to lack of strong reciprocal patterns in harvest and discard rates (Fig. 5). However, for Gag and Red Snapper a March or April bottom fishing closure resulted in improvements in harvest efficiency from about 6–9% due to effort

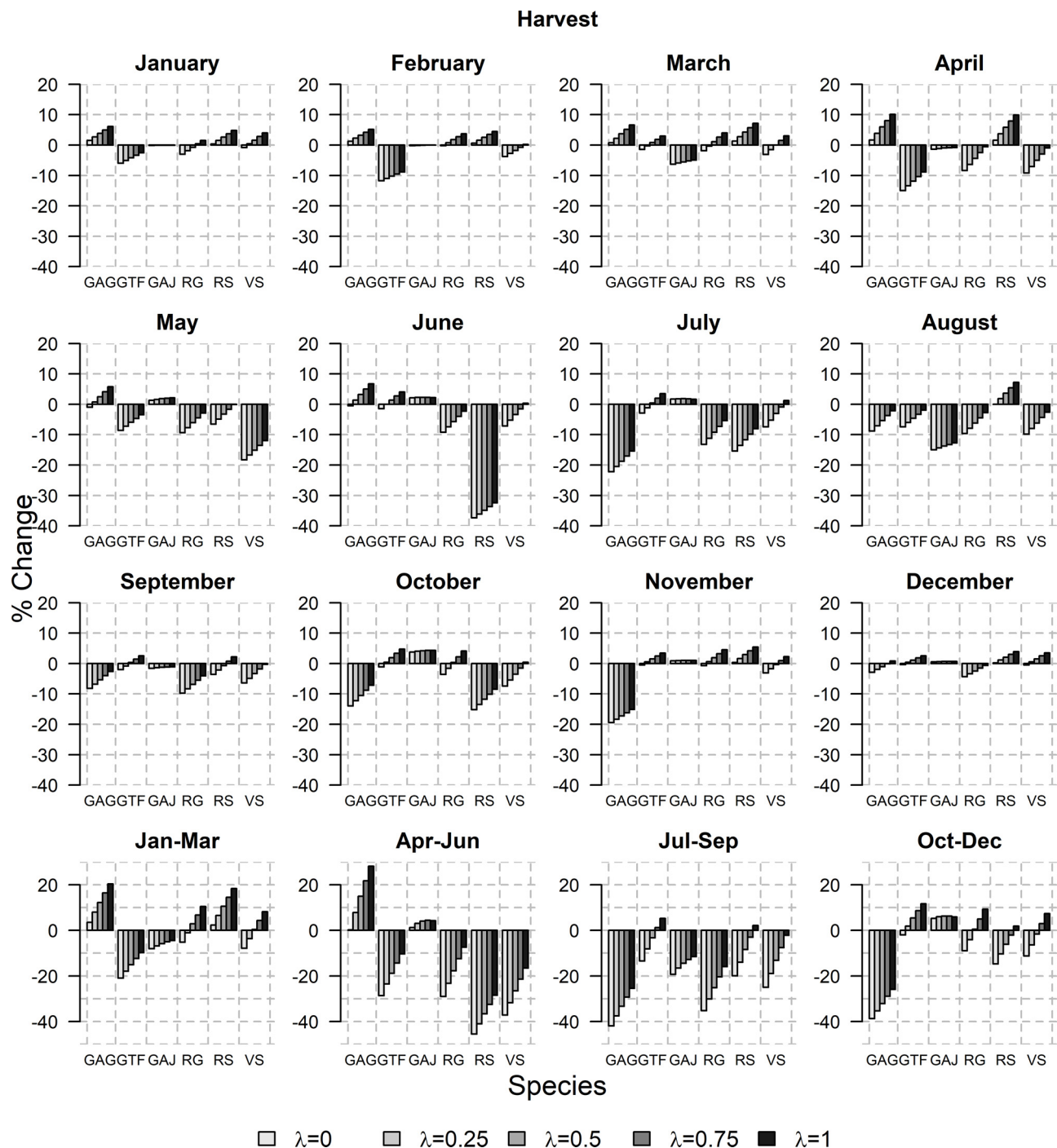


Fig. 7. Percent change in harvest after 20 years of simulation relative to the change predicted under status quo (no closures) for monthly and quarterly bottom fishing closures with a moderate effort response shape ($\sigma = 0.9$).

shifting from the species-specific harvest closure to months when harvest is permitted. In contrast, a June closure would have a strong negative effect on harvest efficiency for Red Snapper (-18%) because it restricted effort during the only month when harvest is permitted throughout the federal waters of the GOM. A similar effect is observed on Gag for the July (-12 to 13%) and November scenarios (-5 to 7%) and for Greater Amberjack in the August closure (-6 to 7%) (Fig. 9). The effects on harvest efficiency for Gag were larger in the quarterly fishing closures with improvements in the winter and spring closure ranging from 12 to 19% and declines in harvest efficiency of about 14–22% during a summer or fall closure (Fig. 9).

To summarize the effects of bottom fishing closures over all species

we averaged the percent change across species, weighted by the proportion of angler trips that targeted each (Table 1). The net effects on average SSB in the monthly closures were negligible (within $\pm 5\%$), with larger positive effects observed in the seasonal scenarios (Fig. 10). Closures occurring during the late winter and early spring (Mar–Apr) led to modest reductions of average dead discards of 4–13%, up to an 8% increase in harvest, and similar effects on average harvest efficiency (Fig. 10).

In the effort dynamic model, the number of trips taken in the terminal year is related to the accumulation of vulnerable fish during the projection period. Thus, closures that resulted in an increase in abundance such as in May and August allowed for a net increase in

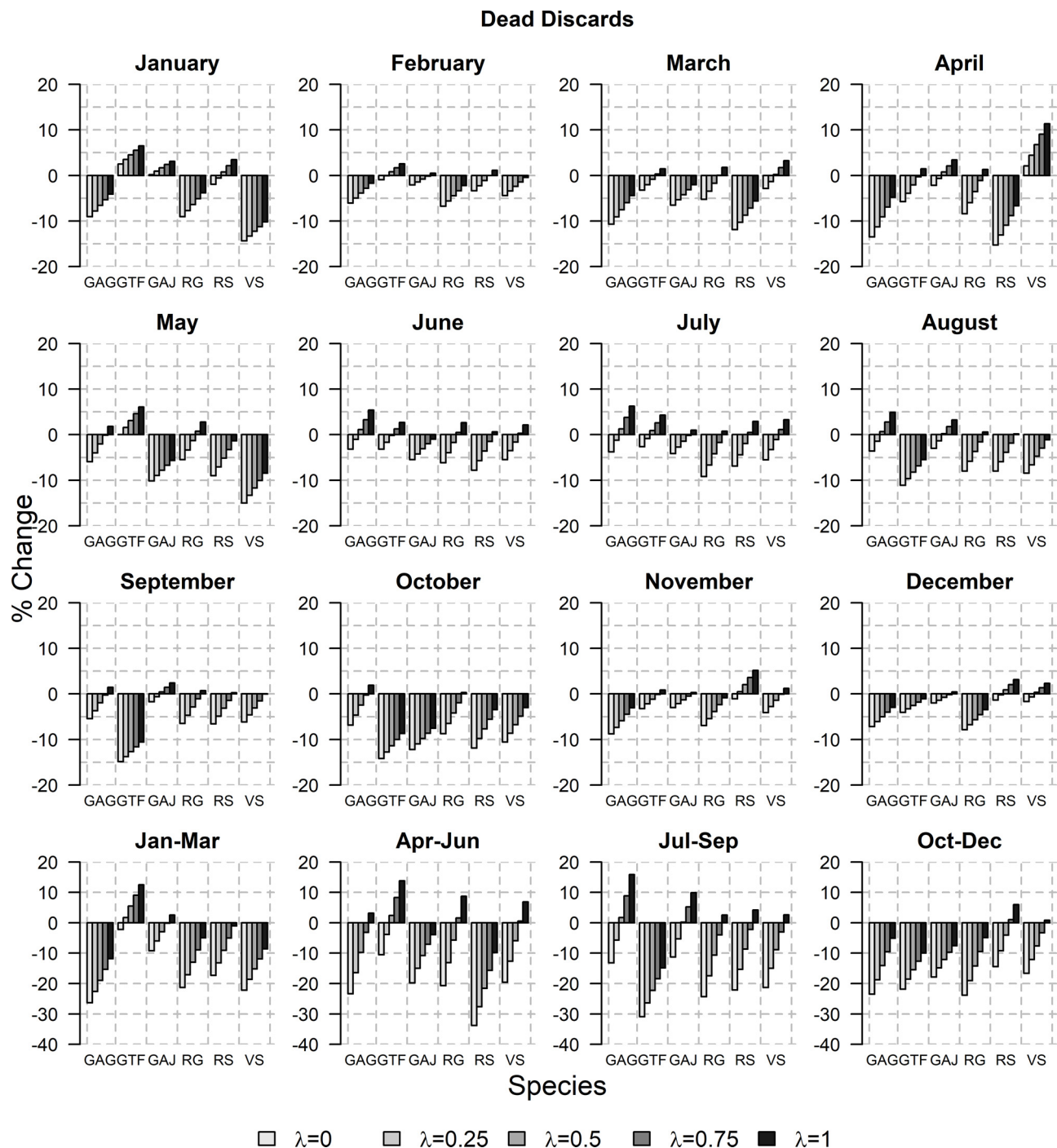


Fig. 8. Percent change in dead discards after 20 years of simulation relative to the change predicted under status quo (no closures) for monthly and quarterly bottom fishing closures with a moderate effort response shape ($\sigma = 0.9$).

fishing effort, but only when λ was high, (Fig. 11). The greatest losses in fishing effort were predicted for monthly fishing closures occurring from about April to September (-5 to 10%), but only when λ was less than 0.5 . The seasonal closures were predicted to cause large declines in effort (up to 27% in spring and summer closures), especially at low λ , which would likely result in strong socio-economic tradeoffs (discussed in Section 4.3). Lastly, the effects of alternative effort response shapes (Fig. 4) were as expected, with lower SSB biomass and higher harvest in scenarios with faster response dynamics (Fig. 12). With no effort response, SSB was always higher and harvest and discards were always lower because increases in stock size did not attract more fishing effort. When a fast effort response was assumed ($\sigma = 0.3$), effort increased

rapidly as the total number of vulnerable fish increased thereby mitigating any gains achieved by the bottom fishing closure.

4. Discussion

4.1. Seasonal closure scenarios

If the management objective for a bottom fishing closure is to reduce the amount of discards without negatively impacting harvest or spawning biomass, the most feasible scenarios would be closures occurring in late winter and early spring (March & April). During this time period, SSB either increased for all species or remained within $\pm 5\%$ of

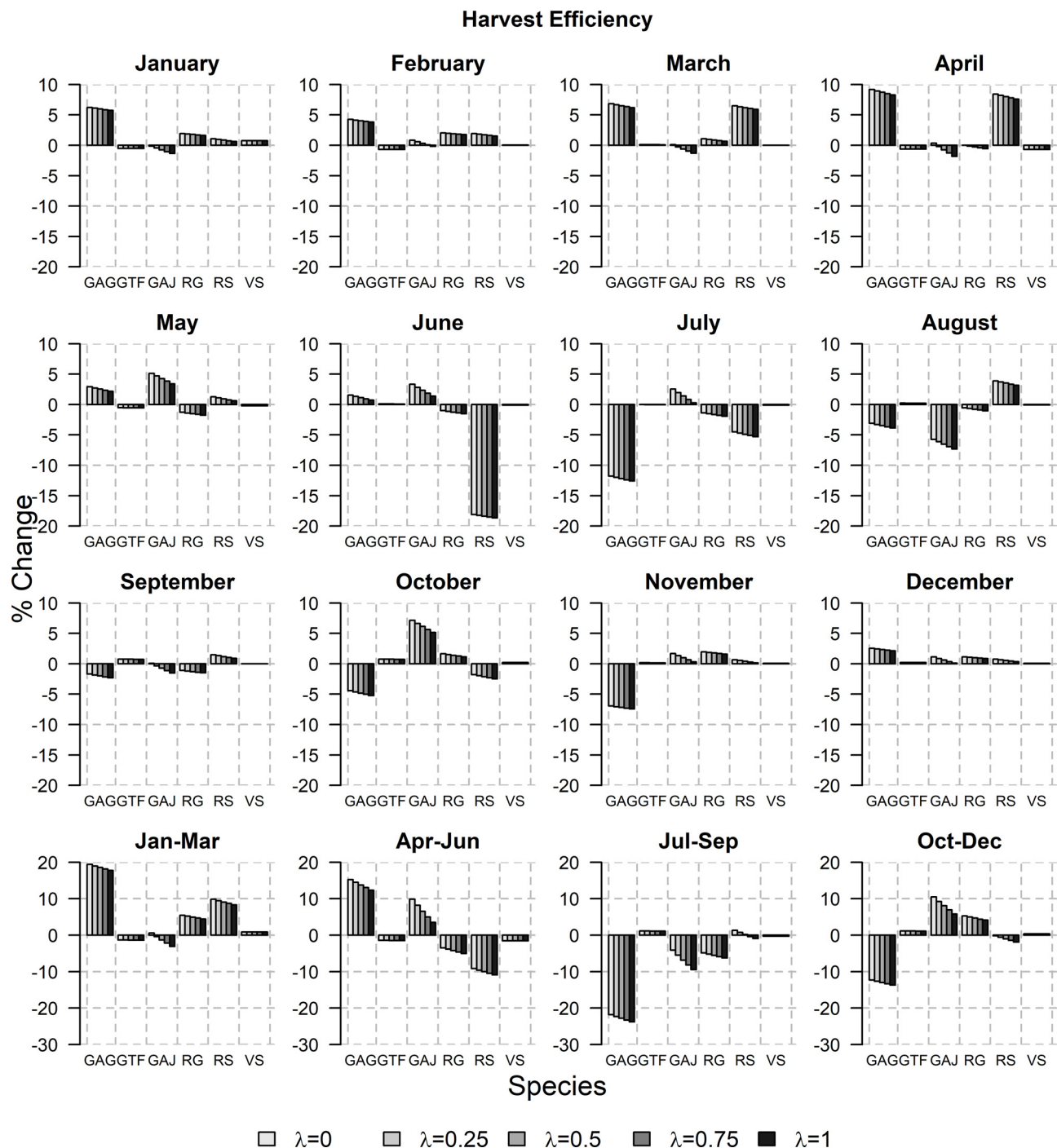


Fig. 9. Percent change in harvest efficiency after 20 years of simulation relative to the change predicted under status quo (no closures) for monthly and quarterly bottom fishing closures with a moderate effort response shape ($\sigma = 0.9$).

status quo, thus the tradeoffs across species were minimized. Additionally, the declines in dead discards exceeded lost harvest resulting in an improvement to harvest efficiency, at least for key species such as Gag and Red Snapper, because the harvest season was not open for those species and the discarding proportion was high during late winter and early spring. Also, the impacts on effort were less severe because most of the fishing takes place between April and July. On the other hand, closures occurring during January or December were predicted to have little effect and therefore would not contribute to meeting any management objectives. Any scenarios being considered must be weighed against the socio-economic tradeoffs, in terms of lost effort-generated revenue at various spatial and temporal scales and

angler dissatisfaction (see Section 4.3).

The effect of closing any given month(s) varied across species and resulted in tradeoffs, such that some closures may result in positive effects on biomass of one species and negative effects on others. These potential tradeoffs are associated with a) the seasonal availability patterns as indexed by the MRIP catch rates and b) the level of angler response (λ) that allows for shifting of effort onto month(s) where availability (or catchability) is high for some species but low for others. We view these results as the equilibrium or average expected outcomes of bottom fishing closures scenarios, absent any stochasticity due to recruitment variation and uncertainty in catch rates. If random variation had been included, we expect that the tradeoffs would be more or

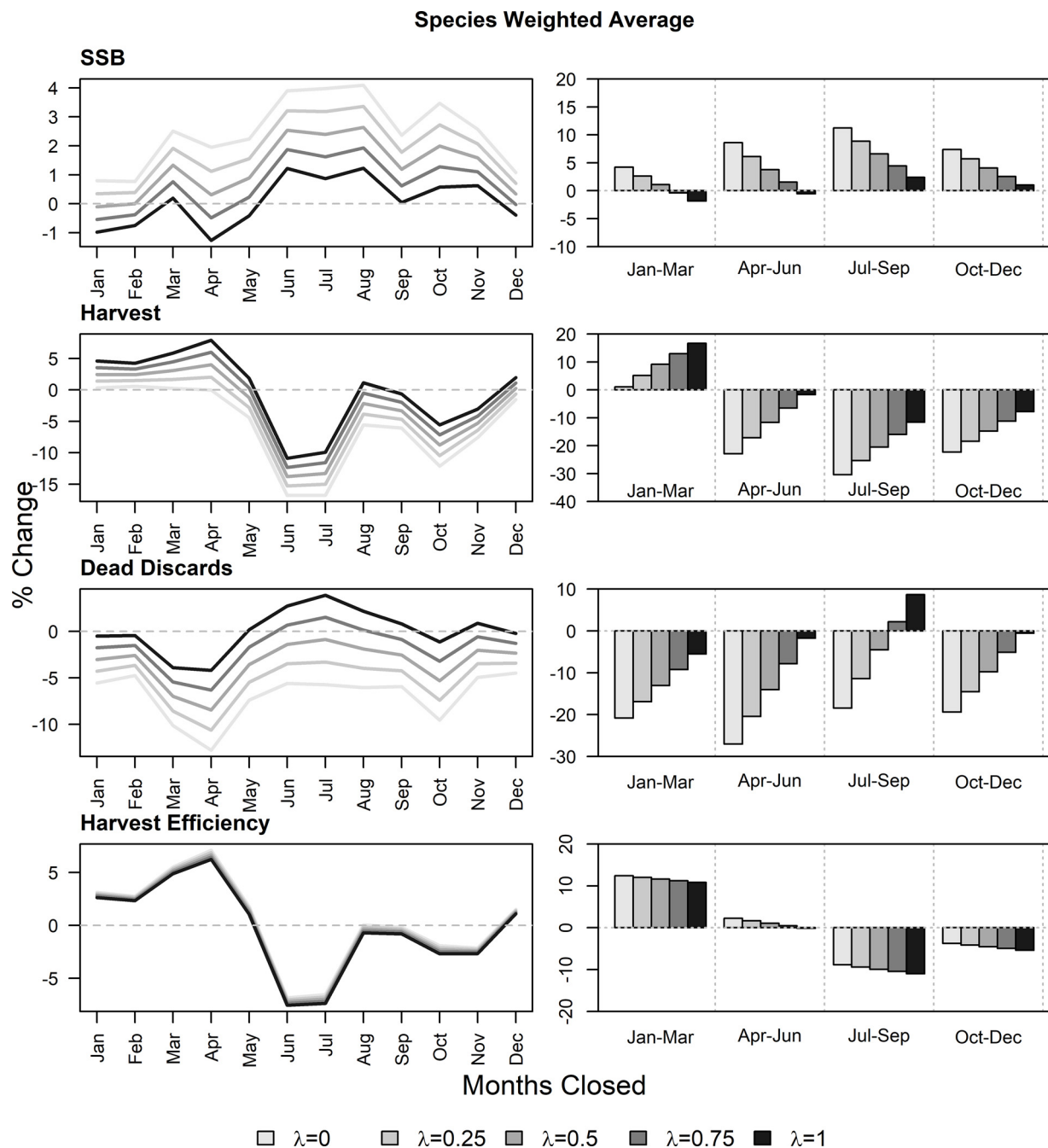


Fig. 10. Average percent change across species in spawning biomass (SSB), harvest, dead discards, and harvest efficiency after 20 years of simulation relative to the change predicted under status quo (no closures) for monthly and quarterly bottom fishing closures with a moderate effort response shape ($\sigma = 0.9$).

less severe around this mean value given the randomness of the inputs.

To a large extent, patterns in species availability are closely associated with spawning seasonality and the formation of spawning aggregations that make reef fish populations especially vulnerable to harvest (Coleman et al., 1996; Sadovy de Mitcheson and Erisman, 2012). Movement from home reefs to staging areas prior to spawning sites also make fish seasonally more available, such as the case with female Gag that form pre-spawning aggregations in shallow waters (~20 m) before spawning at deeper reefs (Koenig et al., 1996; Sedberry et al., 2006). The species considered in this study are known to exhibit divergent spawning seasonality and behavior, with peak spawning occurring during late winter for Gag, late winter to early spring for Red Grouper, spring for Greater Amberjack, and summer for Red Snapper, Vermilion Snapper, and Gray Triggerfish (Biggs et al., 2017). In all

cases, the peak in recreational landings overlaps with spawning seasons, except for when harvest is explicitly prohibited during spawning months, such as for Gag (Biggs et al., 2017). This supports our assumption that MRIP catch rates reflect seasonal availability, and as a result the timing of a bottom fishing closure is likely to have disproportionate impacts across species.

This work has two important limitations. First, the model only evaluated bottom fishing closures that are in addition to the species-specific harvest closures currently in place. For example, we did not simulate a scenario where bottom fishing is prohibited in June with Red Snapper open during a different month because the MRIP data used to adjust for monthly harvest and discard mortalities are not reflective of such a scenario. Additionally, the single-species harvest seasons are not adaptive to the changes in biomass over time in the bottom fishing

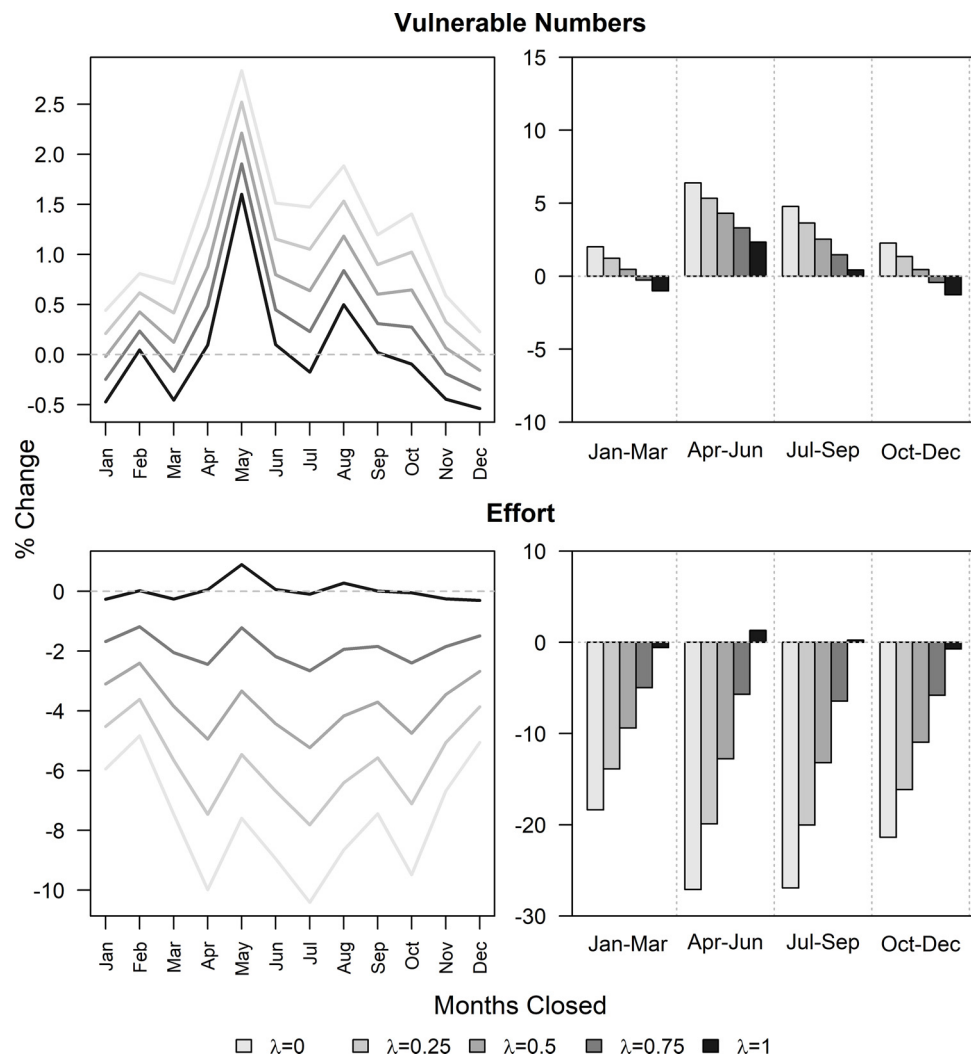


Fig. 11. Total number of vulnerable fish and effort in the final year of the simulation, expressed as a percentage of the total number and effort under status quo.

closure scenarios. To do so would require a complex structured decision making framework (Martin et al., 2009) that allows for alternative single-species regulations for each bottom fishing closure scenario and flexibility to adapt regulations to changes in biomass over time. However, this limitation should not detract from the utility of the study because it is more likely that managers would first adopt a more simple short-term (1–3 months) bottom fishing closure that does not require restructuring all the species-specific harvest seasons currently in place. A second, related limitation is that the efficacy of bottom fishing closures that we describe portends specifically to the status quo management regulations to which they are compared. Here we selected 2015 as our base year so that results would be applicable to these fisheries under current regulations. Had we selected a different base year and used data reflective of expired regulations and different stock statuses our results would have of course been different but perhaps less applicable to contemporary management. Specifically, the presence and duration of harvest seasons, combined with changes in discard rates due to bag and size limits would alter predictions for species such as Red Snapper that now have far more restrictive harvest regulations than 10–15 years ago. What this means is that our results can provide direct guidance towards the specific fishery considered while also demonstrating the merits of considering this novel management approach. They ought not to be construed as a sweeping, fishery-general comparison of harvest versus fishing closures, nor of the conditions under which one or the other would be expected to provide superior outcomes

of certain fishery metrics. However, results from our uncertainty analyses do suggest which specific fishery dynamics can be expected to have pronounced effects on the outcomes of bottom fishing closures.

4.2. Uncertainty in angler response dynamics

A key uncertainty in this model was the effort response parameter λ , which represented the degree to which trips affected by a bottom fishing closure will redistribute to other times of the year. A high λ has the potential to mitigate any gains in biomass made during the closed season and could exacerbate the species tradeoffs when effort shifts onto a month with high harvest or discard rates. Indeed, a temporary closure to the private recreational reef fish fishery in the GOM would effectively eliminating the option of fishing in certain locations for certain species at certain times of year. In light of this constraint, how will anglers behave? Will they redistribute their effort towards months where bottom fishing is open? Will they continue to fish during the closed months but target a different species complex, for example, pelagic or inshore species? Or, will they simply fish less altogether? Unfortunately, we know very little about how anglers might respond under these scenarios. Based on conversations with angler groups in the GOM (e.g. the Coastal Conservation Association, Sportfish Alliance), it is reasonable to assume that private recreational anglers invested in reef fish fishing (e.g. own offshore boats and gear) would likely experience a moderate to high degree of effort shifting in response to a bottom

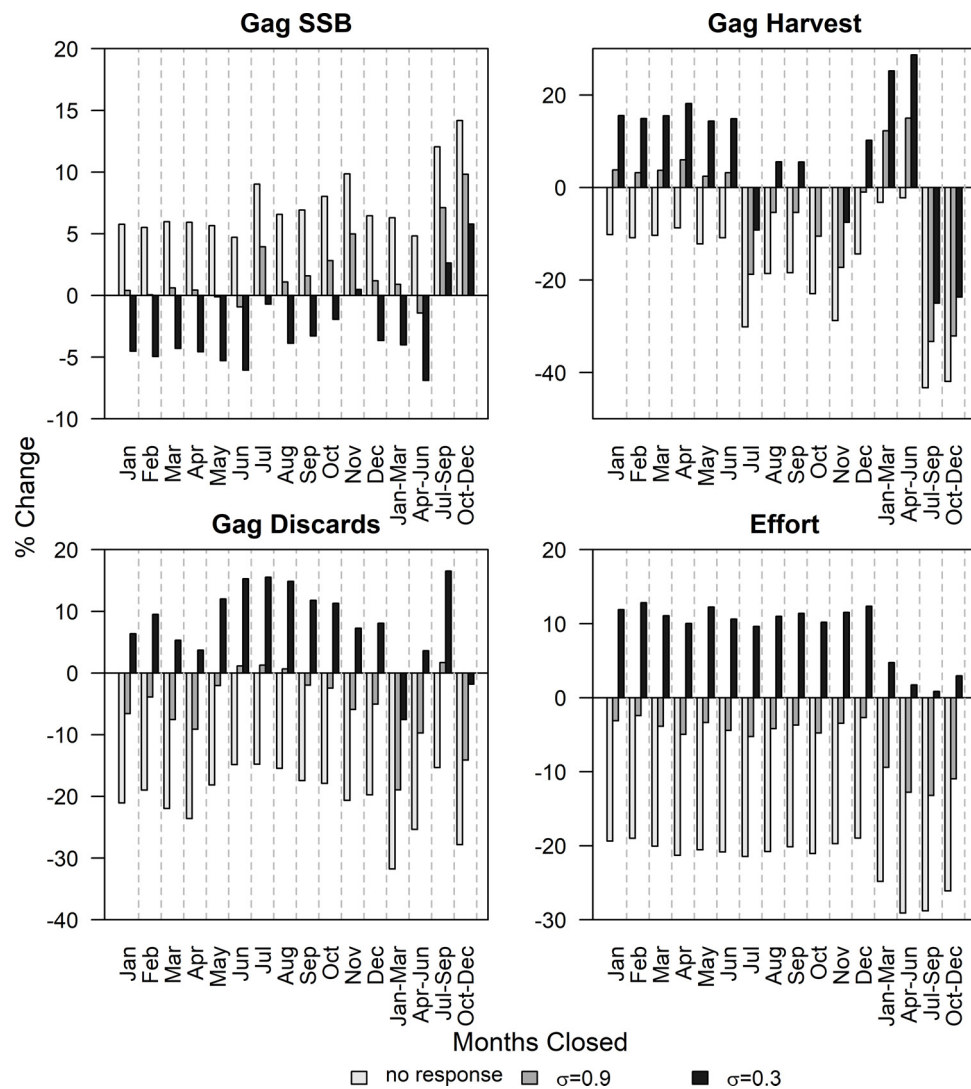


Fig. 12. Effects of the effort response shape on spawning stock biomass, harvest, discards, and effort after 20 years of simulation relative to the change predicted under status quo (no closures) for monthly and quarterly bottom fishing closures with a $\lambda = 0.5$.

fishing closure, therefore a λ of 0.75 might be our best guess at this parameter.

To address this uncertainty, a formal evaluation of angler response can be accomplished with “choice models” (Ben-Akiva and Lerman, 1985; Hunt, 2005; Hunt et al., 2013) that statistically analyze relationships between fishing decisions (when, where, and what to fish for) and attributes of fishing experiences such as expected catch rates, regulations, site-facilities, etc. (Hunt, 2005). A challenge here is that recreational bottom fishing closures have never been implemented in the GOM, so there are no empirical data to infer how anglers might respond. In such cases, discrete choice experiments (DCE) have proven useful for evaluating expected responses to yet-unimplemented management actions (Aas et al., 2000; Hunt, 2005; Beardmore et al., 2011). In absence of experimental or adaptive management approaches (to which managers have so far been adverse), these choice experiments are perhaps the best technique available to inform effort dynamic models in the context of bottom fishing closures evaluated in this study.

Another uncertainty associated with recreational angler behavior is the shape of the effort response curve and the utility metric to which it is related. In this analysis, the model demonstrated opposite results when a fast effort response curve ($\sigma = 0.3$) was assumed rather than a slow effort response ($\sigma = 0.9$) because any gains in biomass were negated by the addition of more effort in the fishery. Recreational

fisheries tend to be highly diverse, consisting of anglers with different motivations and constraints on fishing activities. When modeling effort dynamics, the total effort response is determined by the pattern of variation among participants. If that variation is high, as in recreational fisheries, the total fishing effort response tends to be flat (Walters and Martell, 2004), like that assumed in our baseline scenario. But what is more difficult to posit are the utility attributes that determine the aggregate amount of fishing effort that will occur at any given time and place. Here, we assumed that the utility metric consisted solely of the total number of vulnerable sized fish, but it is more likely that effort is driven more by one or two species or by other attributes of a fishing experience such as expected catch rate, congestion, facilities, and aesthetic value (Arlinghaus et al., 2008; Hunt, 2005). Development of multi-attribute utility metrics would likely improve our predictions about how anglers might respond to seasonal fishing closures.

4.3. Economic and social considerations

The socioeconomic effects of a restriction on reef bottom fishing rather than harvest would almost certainly be wide reaching and intense. A recreational bottom fishing closure would likely change when and where fishing trips are made, thereby altering the spatiotemporal patterns of revenue for coastal fishing communities. This would have an

overall effect on economic impact, defined as the total amounts and flows of money associated with an activity like fishing (Steinback, 1999). Economic impact is of interest to fisheries managers and local politicians who understand that recreational fisheries comprise an important amount of revenue to coastal communities (Adams et al., 2004). Economic impact is particularly relevant to consider for this work because impact is measured at spatiotemporal scales and is generally considered positively related to aggregate fishing effort, which will absolutely change in some way with broad recreational fishing closures. However, not all fishing trips are likely to generate the same amount or type of revenue to coastal communities. For example, anglers targeting reef fisheries may purchase more bait, ice, or gas than those taking inshore trips, or may be more or less likely to secure meals or lodging in coastal communities (Alvarez et al., 2014a, 2014b). Economic impact modeling tools, such as IMPLAN (Steinback, 1999; Chen et al., 2003) account for the cycling of fishing trip expenditures through a regional economy (Edwards, 1991). Economic impact analyses of reef/bottom fishing seasons is clearly needed to evaluate the economic tradeoffs associated with the scenarios evaluated in this study.

A temporary closure to the private recreational reef fish fishery in the GOM would also impose a strong limitation on angler choice - effectively eliminating the option of fishing in certain locations for certain species at certain times of year. Anglers may experience a loss of well-being and satisfaction from this constrained choice set, which may infringe upon anglers' sense of freedom and lead to dissatisfaction with management (Cox et al., 2002; Kearney, 2002), as is described by studies demonstrating anglers' strong resistance to marine protected areas (Voyer et al., 2014). Obviously, the eventual effects of regulations constraining angler choice need not necessarily be negative. If the regulation results in greater anticipated catch or harvest rates in the future, this might compensate for loss of utility/satisfaction. This study absolutely demonstrated potential gains in biomass from some scenarios that could be passed along to anglers through higher catch limits, increased bag limits, and smaller minimum sizes.

Declarations of interest

None.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.fishres.2018.10.018>.

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