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Length-based risk analysis of management options for the southern Florida USA multispecies coral reef fish fishery

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ABSTRACT

Exploitation impacts and management options for 15 coral reef fish species central to the commercial and recreational fisheries of the southern Florida USA coral reef ecosystem were evaluated using a length-based risk analysis (LBRA) framework. Population abundance-at-length composition data were obtained from several regional federal-state sampling programs. These and updated life history demographic data were integrated into a length-based numerical cohort model to generate LBRA fishery sustainability metrics from a probabilistic perspective. Three of five groupers, eight of eight snappers, and two of two grunts were below the 40% spawning potential ratio (SPR) stock sustainability minimum; ten of these stocks are at < 20% of their historical spawning biomass, some as low as 5%. Therefore, to ameliorate overfishing for the 13 stocks with sustainability risks ≥ 98%, fisheries management requires increased minimum sizes of first capture (L_c) and significant reductions in fishing mortality (F). To achieve sustainability and reduce sustainability risks area-time protections are also needed. While lack of data often limits the evaluation of management options, this paper establishes benchmarks from which data-limited approaches can move forward. In addition, the approach can be used to cross-check other data-rich analyses. A goal of this work is to effectively balance sustainability risks with fishery production to mitigate overfishing likelihoods and to increase the probability of sustainable fisheries.

1. Introduction

The economic and ecological importance of coral reef fishes makes their sustainability a key conservation concern (Ault et al., 2014; Cinner et al., 2020; Robinson et al., 2020; Strona et al., 2021). In southern Florida, commercial and recreational fisheries are worth about $6 billion per annum (Ault et al., 2005a; Ault, 2008) and are closely tied to healthy essential habitats that are part of the regional coral reef ecosystem (Coker et al., 2014; Woodhead et al., 2019). Threats to coral reef ecosystems are well known and include human impacts from global warming, such as bleaching and disease (Hoegh-Guldberg, 1999; Aronson and Precht, 2001; Carpenter et al., 2008; Brandt and McManus,

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2009; Eakin et al., 2019; Hughes et al., 2018; Hoegh-Guldberg et al., 2019; Precht et al., 2020). Local impacts related to development and pollution also occur (Hunter and Evans, 1995; Szmant, 2002; Silbiger et al., 2018). However, related to fisheries in south Florida, overfishing is by far the most serious threat to their sustainability – and also around the world (Pauly et al., 1998; Ault et al., 1998, 2005a, 2005b, 2014, 2018; Valentine and Heck, 2005; Cinner and McClanahan, 2006; McClanahan et al., 2008; Hardt, 2009; Sala et al., 2011; Nadon et al., 2015; Brown et al., 2020; Gough et al., 2020).

In the State of Florida, the coral reef ecosystem extends more than 500 km from the Dry Tortugas (Monroe County) to Stuart (Martin County) (Fig. 1). Coastal development and the associated human population grew substantially in this corridor over the past century, from a sparsely populated collection of small farming towns in the early 1920s, into a large, sprawling metropolis (Fig. 2). Today, about 43% of Florida's human population of 21.6 million people live in southern Florida, a 434% increase from the 1960s (Fig. 2**A–C**). Concomitant with this population increase was an extraordinary rise in fishing pressure. Growth since 1964 of the recreational fishing fleet was 410% while the commercial fishing fleet increase was only 45% (Fig. 2**D**). In addition to increasing fishing pressure due to larger numbers of fishers, the fishing power and complexity of both fleets greatly increased due to significant technical innovations such as global positioning systems, advanced fish-finding acoustics, state-of-the-art vessel designs, real-time weather data, communication networks, and even social media. Commercial fishers use highly diverse fishing gear that is deployed by large fleets mainly composed of small vessels. In addition, millions of recreational fishers land dozens of species across many widely distributed ports (Gallucci et al., 1996; Ault et al., 2005a; Amorim et al., 2020). A consequence of all this complexity is that it is extremely difficult, but not impossible, to assess and monitor the status of exploitation and trends of the coral reef fishery in south Florida.

More than two decades ago, Ault et al. (1998) developed a prototypical length-based assessment method for the multispecies coral reef fish fishery of the Florida Keys. They found that 70% of the snapper-grouper species were overfished. Their methods were subsequently applied in Florida (Ault et al., 2005b), Puerto Rico (Ault et al., 2008), the Hawaiian Islands (Nadon et al., 2015; Ault et al., 2018), and elsewhere. Since Ault et al. (1998), there have been substantial advances in dependent and independent fishery data, population demographics, mathematical and statistical methods, and computational power that have facilitated improved length-based risk analysis and sustainability status determination for data-limited fisheries. Recently, Ault et al. (2019) developed a length-based risk analysis (LBRA) estimation-simulation framework that used size–frequency distribution observations integrated over spatial-temporal considerations. The framework was also shaped by using fishing intensities from multiple data sources to guard against estimation bias. The LBRA incorporates probabilistic population demographic processes that characterize marine fishes, while at the same time facilitating the probabilistic representation of spawning and exploitable biomass relative to gear selectivity and fishing intensity over the entire length range at a given time. This framework directly addresses the uncertainty often found in fishery data to assess trends in tropical multispecies stock assessments. Demographic processes are also integrated into the assessments to evaluate sustainability metrics from a probabilistic perspective. In this way, methodological advances of assessing exploitation under uncertainty allow us to pose important questions. For example, what is the likelihood that a given stock is being fished sustainably and are fishery management strategies compatible with human demands for coral reef fishery resources?

Here we extend the LBRA estimation-simulation methods to evaluate the performance of different management options that use alternative harvest strategies for 15 exploited snapper-grouper species in the southern Florida coral reef ecosystem. Given that managers struggle to predict even one metric with certainty, we compared several key population metrics relative to currently used USA sustainability reference points. The comparisons produced probability distributions using

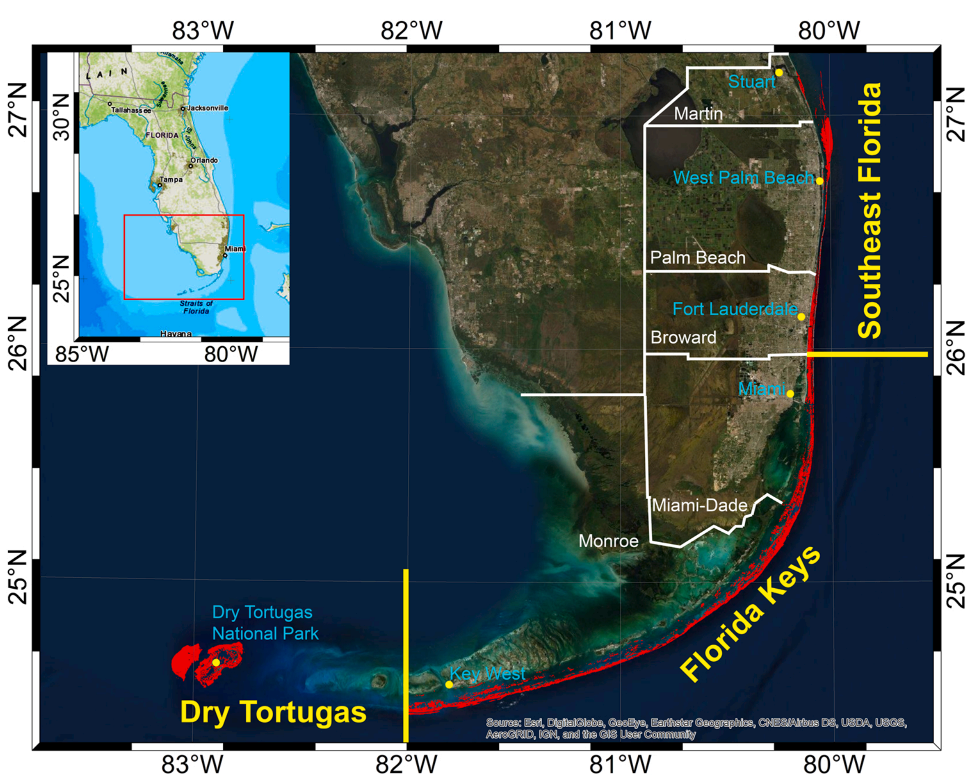

Fig. 1. The southern Florida coral reef ecosystem from the Dry Tortugas (Monroe County) in the southwest extending 500 km northeast across the counties of Miami-Dade, Broward, and Palm Beach to Stuart, Florida (Martin County). Red area alongshore is coral reef.

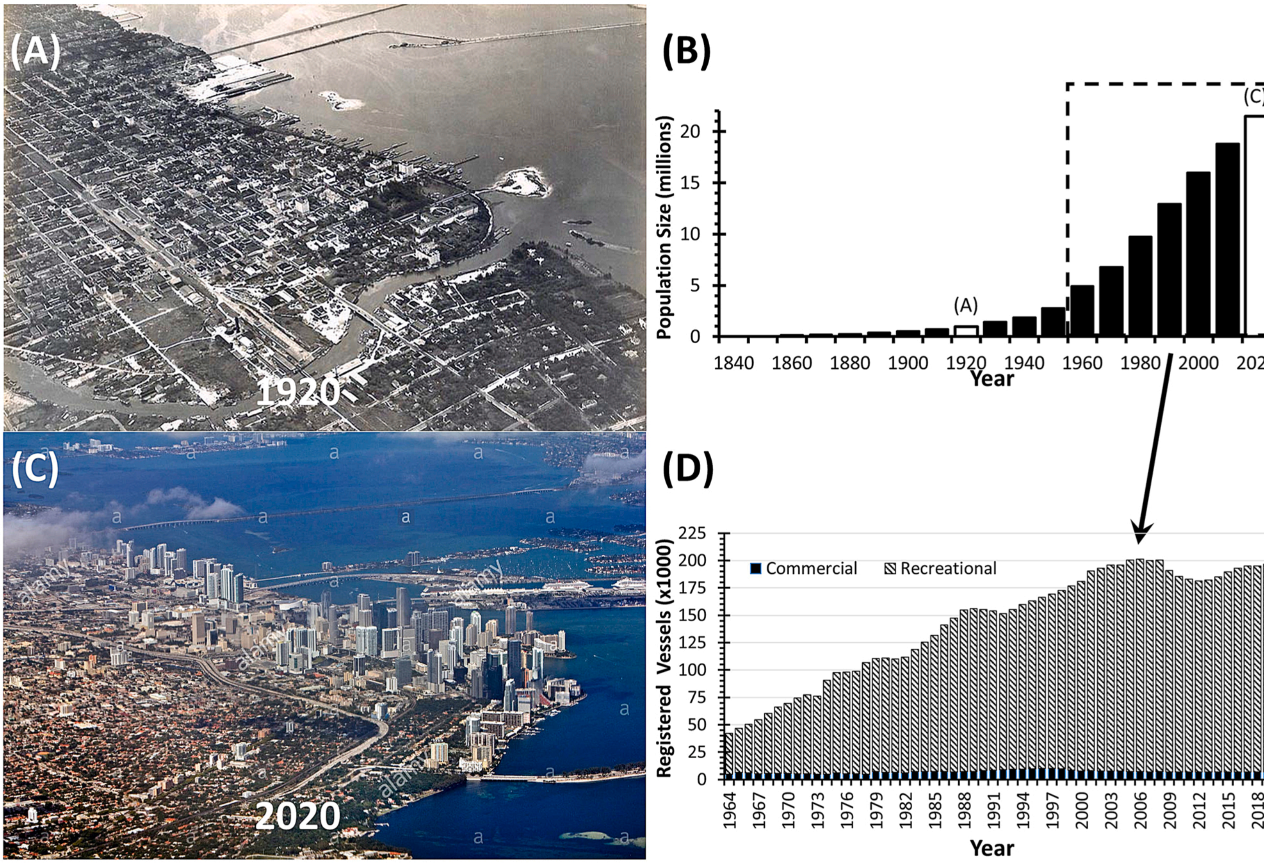

Fig. 2. (a) Southern Florida regional coastal development and fishery fleet growth: (A) downtown Miami circa 1920; (B) Florida's human population from 1840 to 2020 (Source: US Census), the open bars refer to panels (A) and (C); (C) downtown Miami 2020; and, (D) southern Florida registered commercial (black bars) and recreational (hatched bars) fishing vessels from 1964 to 2019 (Source: Florida Department of Motor Vehicles). Dashed rectangle in (B) corresponds to the time period shown in panel (D).

stochastic simulations of stock and fishery dynamics. The analyzes employed a precautionary approach to evaluate exploitation status under several specific management actions that could be implemented regionally. Application of the framework will help to secure sustainable reef fisheries in the face of growing human populations, exploitation pressures, and climate changes.

2. Methods and materials

2.1. LBRA overview

Average size ($\overline{L}(t)$) in the exploited phase of fishery stocks should directly reflect the total mortality $Z(t)$ resulting from changes in the observed population abundance at size distribution. The finite lifespan mathematical model for $\overline{L}(t)$ is a robust indicator of population size composition and specifically involves an estimate of variance which reflects uncertainty (Ault et al., 2014, 2019). To estimate the total mortality rate Z parameter, the "data limited" LBRA approach of Ault et al. (2019) requires a suite of species-specific life history demographic parameters: (i) natural mortality rate M estimated from lifespan survivorship to age a_λ; (ii) the von Bertalanffy length dependent on age growth function; (iii) the allometric weight-length relationship; (iv) the minimum length at first capture (L_c); (v) the largest observed size (L_λ); and, (vi) the length at which 50% of individuals attain sexual maturity (L_m). The LBRA method assumes knife-edged selection and constant recruitment.

Total mortality rate (Z) was described by the statistical properties of the average length estimated as normally distributed variable, consequently, a normal $N(\mu, \sigma^2)$ probability density function was parameterized by setting $\mu = \overline{L}(\mathrm{t})$ and $\sigma^2 = [SE(\overline{L}(\mathrm{t}))]^2$, and then these were used to generate random deviates of average length $\overline{L}(\mathrm{t})$. Random deviates of Z were computed from the average length deviates. In a similar manner, probability distributions of natural mortality rate M were computed from corresponding probability distributions for maximum age a_λ described in Ault et al. (2019). The above procedures were used to generate a pair of Z and M random deviates, from which a random deviate for fishing mortality F was computed ($F = Z - M$). This provided the input mortality rates for a single run of the numerical population model. In our applications, to achieve the asymptotic properties of the selected probability distributions, n = 10000 trial runs were carried out for a given species.

To conduct probabilistic sustainability analyses, a length-based cohort population simulation model tailored for data-limited situations by Ault et al. (2019) was used to incorporate stochastic mortality and growth for southern Florida coral reef fishes. In the numerical simulation model, variation in length-at-age around the von Bertalanffy growth function was modeled as normally distributed with a constant coefficient of variation of 7% (CV, standard deviation divided by the mean), following general characteristics observed in length-age growth studies for a wide variety of fish species (e.g., Then et al., 2015). The numerical simulation model tracks cohort numbers-at-size (both length and weight) over age and time. The length-based cohort population simulation model equations and parameters are found in Ault et al. (2019), and parameters used here are given in Table 1. For application in this study, the model time step Δt was monthly (12 equal periods for 1 year). Under data-limited unit of stock definitions, model assumptions were: (i) average annual constant recruitment ($h \cong 0.99$), apportioned evenly for each model time step; (ii) knife-edged length at sexual

Table 1
Parameter symbols, definitions and units for demographic relationships used in the LBRA estimation-simulation analysis of southern Florida coral reef fish populations.

Symbol	Definition	Units
a	Cohort age class ($a = 0, \ldots, a_\lambda$)	months
da	Variable of integration with respect to age	months
dL	Variable of integration with respect to length	mm FL
$Z(a,t)$	Total mortality rate at age a at time t	per year
$M(a,t)$	Natural mortality rate at age a at time t	per year
$F(a,t)$	Fishing mortality rate at age a at time t	per year
$N(a,t)$	Numbers (abundance) at age $a+\Delta a$ at time $t+\Delta t$	number of fish
$L(a,t)$	Length at age a and time t	mm FL
$W(a,t)$	Weight at age a and time t	kg
$B(a,t)$	Biomass at age a and time t	kg
L_∞	Asymptotic length	mm FL
K	Brody growth coefficient	per year
a_0	Age at which length equals zero	years
W_∞	Asymptotic weight	kg
α	Scalar coefficient of weight-length function	dimensionless
β	Power coefficient of weight-length function	dimensionless
a_λ	Maximum observed age (under exploitation)	years
L_m	Length at 50% maturity	mm FL
$\overline{L}$	Average length in the exploited phase	mm FL
L_c	Minimum length at first capture	mm FL
L_λ	Mean length at maximum age	mm FL
W_λ	Mean weight at maximum age	kg
$\overline{\overline{L}}$	Composite $\overline{L}$ from empirical sampling surveys	mm FL
$\widehat{M}$	Natural mortality rate estimated from $\widehat{a}_\lambda$	per year
$\widehat{F}$	Fishing mortality rate estimated from $\widehat{Z}$ and $\widehat{M}$	per year
$\widehat{\overline{L}}$	$\overline{L}$ estimated from numerical model	mm FL
F_{med}	Median of fishing mortality rate distribution	per year
f	Nominal fishing effort	numbers
$\theta(a)$	Sex ratio at age a	dimensionless
$\phi(L)$	Selectivity at length L	dimensionless
$\overline{N}(t)$	Average population abundance at time t	numbers
$\overline{B}_{EX}(L_c,t)$	Average exploitable population biomass	mt
$\overline{N}(L,t)$	Average abundance (numbers) at length at time t	numbers
$\overline{B}(L,t)$	Average population biomass at length at time t	mt
$Y_n(t)$	Yield in numbers at time t	numbers
$Y_w(t)$	Yield in weight at time t	mt
SSB	Spawning (mature) stock biomass	mt
SPR	Spawning potential ratio	dimensionless
F_{LRP}	Fishing mortality rate at limit reference 40% SPR	per year
F/F_{LRP}	Current F to reference F (Overfishing limit, OFL)	dimensionless
B/B_{LRP}	Current to reference spawning biomass	dimensionless

maturity L_m; and, (iii) knife-edged gear selectivity at length L_c. For all metrics, exploitable abundance $N_{EX}(t)$ at time t was calculated by integrating over the length and age ranges at a particular time interval Δt,

$$N_{EX}(t) = \int_{L_c}^{L_\lambda} \int_{a_c}^{a_\lambda} N(L|a,t)\phi(L)dadL \tag{1}$$

where $(L|a,t)$ was length conditioned on age a at time t (i.e., the distribution of lengths at a given age-time), and $\phi(L)$ was gear selectivity at length L at a given time. Note that L_λ refers to the statistical distribution of lengths at age a_λ, such that subscripts c in the integrand are the minimum length (L) or age (a) of capture, and subscripts λ in the integrand are the minimum and maximum L or a in the population.

The observation model component of the numerical cohort model generated estimates of the first-moment of size $\overline{L}(t)$ in the exploited phase of the stock given the demographic and fishery parameter estimation for comparison with resource monitoring data

$$\overline{L}(t) = \frac{\int_{L_c}^{L_\lambda} \int_{a_c}^{a_\lambda} F(t)N(L|a,t)\phi(L)L|a(t)dadL}{\int_{L_c}^{L_\lambda} \int_{a_c}^{a_\lambda} F(t)N(L|a,t)\phi(L)dadL}. \tag{2}$$

2.2. Extension of LBRA to evaluation of management options

To assess the effects of management options, comparisons of various population performance metrics at current and projected levels of fishing mortality $F(t)$ relative to management reference points were used. Six population metrics used in the assessments were: (1) fishing mortality rate (F); (2) exploitable population abundance (N_{EX}); (3) exploitable population biomass (B_{EX}); (4) yield in number (Y_n); (6) yield in weight (Y_w); and, (6) spawning stock biomass (SSB).

Yield in number Y_n at time t was calculated as usual by multiplying F at t by the average exploitable population abundance $\overline{N}_{EX}$ at t,

$$Y_n(t) = F(t)\overline{N}_{EX}(L_c,t) \tag{3}$$

Biomass *B(a,t)* at age a and time t was estimated as abundance in numbers $N(a,t)$ times weight $W(a,t)$. Thus, exploitable biomass $B_{EX}(t)$ at time t was calculated by integrating over the length and age ranges at a particular time interval Δt,

$$B_{EX}(t) = \int_{L_c}^{L_\lambda} \int_{a_c}^{a_\lambda} B(L|a,t)\phi(L)dadL = \int_{L_c}^{L_\lambda} \int_{a_c}^{a_\lambda} N(L|a,t)W(L|a,t)\phi(L)dadL, \tag{4}$$

Yield in weight Y_w at time t was calculated by multiplying F at t by the average exploitable population biomass $\overline{B}_{EX}$ at t,

$$Y_w(t) = F(t)\overline{B}_{EX}(L_c,t) \tag{5}$$

Similarly, spawning stock biomass (SSB) at a given level of fishing mortality at time t was obtained by integrating over the size-age range of sexually mature individuals in the population,

$$SSB(t) = \int_{L_m}^{L_\lambda} \int_{a_m}^{a_\lambda} B(L|a,t)dadL. \tag{6}$$

In the integrations of Eqs. (1)–(6), all lengths above knife-edged L_c or L_m were included.

Spawning potential ratio (SPR), a management benchmark that defines exploited stock reproductive capacity (c.f., Ault et al., 2014), was computed as the ratio of *SSB(t)* at an F at time t relative to that of an unexploited stock (i.e., $F = 0$),

$$SPR = \frac{SSB_{F(t)}}{SSB_{F=0}}. \tag{7}$$

Because of the recruitment assumption, relative spawning biomass and *SPR* are functionally equivalent.

Contemporary fishery management plans rely on harvest control rules (HCRs) that calculate annual catch limits and targets from stock assessment results under uncertainty and guide the adoption of management measures (Methot et al., 2014; Kvamsdal et al., 2016; Maunder et al., 2020). Basic to the design of HCRs is the establishment of limit and target reference points as fundamental concepts (Prager et al., 2003). A *limit reference point* (LRP) is maximum value of fishing mortality or minimum value of biomass that reflects the perceived last point of acceptable exploitation risk tolerance for a stock, and it should not be exceeded. Otherwise, it is considered that it might endanger the capacity of self-renewal of the stock (Cadima, 2003). Ault et al. (2019) developed a new procedure to establish limit reference points (LRPs) for population sustainability risk. This method employed three precautionary demographic principles: (1) setting $L_c = L_m$, which assured that exploitation was directed only toward mature adults and not juveniles, giving fish at least one chance to spawn in their lifetime on average; (2) setting $F = M$ as a proxy for F_{MSY}, that is, the fishing mortality rate that achieves maximum sustainable yield; and, (3) considering MSY as a hard limit to the associated exploitation rate F_{MSY} not to be exceeded (as opposed to a target). Their numerical cohort model was then used to calculate the SPR at F_{MSY} for a suite of Florida grouper (Epinephelidae) and snapper (Lutjanidae) species. For each family, the species' average SPR at MSY was rounded to the nearest 5% increment and defined as the "reference" %SPR point. The sustainability risk reference point for fishing mortality, F_{REF}, was defined as the F generating the reference % SPR. The average SPR at MSY for groupers was 39.8%, and the average for snappers was 38.4%. These were rounded to the nearest 5% increment, resulting in a reference SPR of 40% for both families. Here we specifically defined the LRP as 40% *SPR*, the minimum level of *SSB* needed to ensure sustainability; thus, the LRP fishing mortality rate F_{LRP} was set equivalent to the $F_{40\%SPR}$.

A *target reference point* (TRP), on the other hand, is a stock status indicator which reflects a desirable target for management and should embody desired biological or ecological benefits to be maintained in perpetuity with an accepted low probability of endangering the resource (Cope and Punt, 2009). MSY has most often been used in this sense, but is far from a conservative benchmark in the stable long-term management of fisheries (Pauly and Froese, 2021). MSY is typically obtained between 30% and 40% of unfished biomass, so MSY should be a limit, not a target, for fisheries management, i.e., $F_{TRP} < F_{LRP}$ and $B_{TRP} > B_{LRP}$ (e.g., Punt et al., 2014). A TRP based on a different target (e.g., $> F_{50\%SPR}$) may be attractive (Clark, 2002). Thus, target fishing pressure would be well below the MSY level and yield could be obtained with substantially lower effort and costs with substantially higher profits and benefits for the fishers. Such a TRP also has the obvious advantage of accounting for life history differences and finessing uncertainty about the stock-recruitment relationship. As such, the time-dependent variables $B(t)$ and $F(t)$ within the spatial stratifications of stocks and fishing intensities are best expressed as non-dimensional ratios, $F(t)/F_{LRP}$ (commonly termed the overfishing threshold, or here the "overfishing limit", $OFL = \frac{F}{F_{LRP}} = 1$), and $\frac{B}{B_{LRP}} = 1$, than in specific units of mass and time^{-1} (Prager et al., 2003). Here the OFL was defined as the yield in weight Y_w that corresponds to the stock's 40% SPR. The distribution of random deviates of F and the LRPs and TRPs were used to compute probability distributions for *OFL* and *SPR*. The proportion of the distribution of $F/F_{LRP} > 1.0$ was defined as the sustainability risk probability from the estimated fishing mortality rate. In the same manner, the proportion of the B/B_{LRP} stock reproductive capacity distribution < 1.0 was the estimated risk probability of *SPR* sustainability.

The "Kobe framework" was used to draw harmonized interpretation of results for the southern Florida reef fish fishery because it allows simultaneous consideration of the level of fishing mortality that achieves sustainable yields, while maintaining the appropriate spawning stock biomass to ensure yields into the indefinite future.

The numerical cohort model was configured to compute Y_w and *SPR* for the full range of feasible combinations of L_c and F, i.e., 'isopleth' surfaces, to facilitate placing current exploitation rates in context of fishery production and sustainability risks in our search for viable TRPs that produced sustainable benefits. These analyzes aided exploration of feasible future TRP options for species with currently high sustainability risk levels.

While there were a number of possible management interventions that could minimize sustainability risks and maximize stock production for the reef fish community, we used LBRA to test several management option scenarios: (1) **Business as Usual** (BAU): *laisse faire* management that imposes no changes in the 2012–2016 (F_{BAU}, L_c) conditions, and these conditions will define present and future resource sustainability status without intervention by decision-makers; (2) **Limit:** Increased minimum size limit (L_c) to a level equal to or greater than the defined LRP, here 40% SPR, by employing a "semi-eumetric fishing" policy to achieve minimum sustainability for a given species while the fishing mortality rate (F_{BAU}) remained at BAU levels; and, (3) **Target**: pivoted off the LRP with increased minimum size limits (L_c) to 40% SPR "eumetric" levels as in the **Limit** scenario, and also reduced fishing pressure (F) by 50% $(\text{i.e.}, F_{Target} = 0.5{*}F_{BAU})$ to achieve a TRP that provided conservative advice under conditions of uncertainty.

For each scenario, we evaluated expected distributional changes in both population *SPR* and yield-per-recruit (*YPR*). We also conducted prospective recovery simulations to determine the time required to return to the level of initial yields, and then ultimately the time to the new equilibrium when perceived full benefits of the management strategy were achieved.

2.3. Application to the southern Florida reef fish community

Applications were carried out for 15 southern Florida coral reef fish species following the statistical and mathematical methods given in Ault et al. (2019). Life history demographic parameters were obtained from Stevens et al. (2019). Length composition data were obtained from several NOAA Southeast Fisheries Science Center statistical sampling programs: (1) Trip Interview Program (TIP) of the commercial fishery is a dockside fleet intercept survey, (2) Marine Recreational Information Program (MRIP) of the recreational fishery is a dockside intercept survey of fishers on private/rental boats, small charter boats, and the shoreline (Dixon and Huntsman, 1992; Bohnsack et al., 1994; NMFS, 2017), and, (3) Reef Fish Visual Census (RVC) is a in situ fishery-independent diver survey (Smith et al., 2011). These multiple data sources were used to guard against estimation bias since the three statistical sampling programs independently collected information from reef habitats throughout southern Florida; however, no single survey, either fishery-independent or fishery-dependent, encompassed the full distribution of habitats for the entire reef fish complex. For example, the fishery-independent diving survey was restricted to depths < 33 m; and, commercial and recreational fishing was off-limits within a network of no-take marine reserves implemented between 1997 and 2007 (Smith et al., 2011; Ault et al., 2013). These data allowed estimation of the "average size" indicator variable $\overline{L}$, average length in the exploited phase of the population (i.e., the mean length of individuals $> L_c$, the minimum length of first capture regulated by the fishery), by data source.

To obtain representative population $\overline{L}$ values, composite $\overline{L}$ and SE($\overline{L}$) were estimated by taking the arithmetic means of $\overline{L}$ and var($\overline{L}$) from the three data sources following Ault et al. (2019). These computations were carried out separately for three regions (Dry Tortugas, Florida Keys, Southeast Florida; Fig. 1). The overall southern Florida $\overline{\overline{L}}$ and associated variance were computed as the average of the three regions weighted by the proportion of mapped reef habitat area (c.f., Smith et al., 2011). Ehrhardt and Ault's (1992) length-based finite lifespan model was used as the basis for total mortality rate estimation. L_λ was considered to be the mean length at age a_λ with varying length observations above and below that mean value; thus, the statistical distribution of all lengths conditioned on age $(i.e., L|a)$ above L_c were included in the empirical and statistical estimation of $\overline{L}$. Note that some applications (e.g., Chong et al., 2020) have treated L_λ as an absolute maximum boundary and excluded length observations greater than L_λ. This can lead to unpredictable results (Then et al., 2015).

In what follows, we provide evaluations of three scenario strategies (BAU, Limit, and Target) and then interpret these results for each priority species to provide a relatively comprehensive view of the risks to stock production (yield-per-recruit) and sustainability (spawning potential) for a range of life history demographics.

3. Results

3.1. Regional sustainability analyses

Parameters, symbols, definitions and units for 15 exploited southern Florida reef fish demographic relationships used in the LBRA population modeling are given in Tables 1 and 2. There was relatively good agreement between mean estimates of $\overline{L}$ for each species among the three independent data sources from 1979 to 2016, particularly in the later years (Fig. 3). Regional and southern Florida-wide estimates of average length for the 2012–2016 period (Table 3) were used to compute population total mortality rates $\widehat{Z}$ with the Ehrhardt-Ault (EA) algorithm. Mean total mortality rates Z estimates were used in the numerical model, subtracting off natural mortality $\widehat{M}$, to determine the fishing mortality rates $\widehat{F}$ for the 2012–2016 period. Grouper populations, in general, had extremely small sample sizes, indicating the rarity of these species in today's fishery.

There was a very strong sub-regional effect of fishing mortality and overexploitation as expressed by both the sub-regional $\overline{L}$ and the F/F_{LRP} overfishing limit. In general, exploitation effects were more pronounced in sub-regions with greater human population density, that is, Southeast Florida > Florida Keys >> Dry Tortugas (Fig. 4). The proportion of reef fish stocks having F/F_{LRP} ratios that exceeded sustainable levels was greatly skewed to unsustainable levels moving northward in the southern Florida region (Table 4).

3.2. Probabilistic sustainability analyses

The Ault et al. (2019) computational process for generating probabilistic mortality rates used the median of the distribution, F_{med}, as the expected value for F, since the distribution of F random deviates was generally asymmetrical. The numerical cohort model, utilizing the life history synthesis of Table 2, was used to compute the spawning potential ratio (SPR) at the limit reference point $F_{LRP} = F_{40\%SPR}$, or 40%SPR for each reef fish species, and the corresponding model-predicted exploitable population biomass at F_{LRP} was defined as B_{LRP}. To be considered minimally sustainable, the SPR for a given species must be ≥ 0.40 or $\geq$40% of the unfished SPR. Since the southern Florida reef fishery does not necessarily target a single species, considering the entirety of the multiple exploited stocks that comprise the fishery complex in a single framework facilitates a more holistic evaluation of the fishery management effects and preferable form for evaluation by evolving national (state and federal) and international fishery management standards.

LBRA evaluation of sustainability for 15 species of the exploited southern Florida reef fish community using the Kobe framework shows the current stock status and exploitation rate relative to target reference points such as B/B_{REF} and F/F_{REF}. Using median values, Fig. 5 presents all 15-reef fish species together in a Kobe strategy plot so that status of individual stocks can be visually compared to other members of the jointly exploited reef fish community. Results from the Kobe plot suggested a wide range of sustainability status for the 15 species. Considering that populations located in quadrant I are "sustainable", while those in quadrant III are overfished, Fig. 5 shows that 87% (3/5 groupers; 8/8 snappers; 2/2 grunts for 13/15 overall) of the 15 most economically-important species in the southern Florida reef fish community had SPRs below the minimum level for sustainability (Table 5).

In this regards, eleven of 15 of these species had sustainability risks of 100% (i.e., literally zero probability of a sustainable fishery) under the business as usual F, L_c regime. Two species, red grouper and white grunt, had $\geq 98.1\%$ and $\geq 99.9\%$ sustainability risks, respectively. These 13 species had a mean OFL of 3.57, range [1.75, 7.57]. Some species had up to 7 times the fishing mortality rate level that produces minimal sustainable populations. We concluded that ten of the 15 species are currently at < 20% of their historical abundance, thus the fleets have greatly over-exerted fishing intensity. The intense exploitation

Table 2
Life history demographic parameters (Stevens et al., 2019; Nadon and Ault, 2016) and management controls imposed for 15 exploited southern Florida coral reef-fish stocks used in the LBRA (length-based risk assessment). Symbols, definitions and units are given in Table 1.

	Demographic parameters											Management controls				
	a_λ	M	K	L_∞	a_0	α	β	W_∞	L_m	L_λ	W_λ	L_c	Bag Limit	Seasons[b]	ACL	Moratorium
GROUPERS																
Black	33	0.09078	0.1432	1299.53	-0.9028	8.74750E-06	3.0843	35.13	834	1289.41	34.30	600.0	X	Jan 1–Apr 30	X	
Goliath	37	0.08097	0.0937	2221.10	-0.6842	6.49002E-06	3.1510	227.66	1200	2156.00	207.31	450.0				≥ 1990
Nassau	22	0.13617	0.1000	932.00	-1.7000	4.16869E-06	3.2000	13.25	435	844.88	9.68	500.0				≥ 1990
Red	29	0.10330	0.1251	829.00	-1.2022	5.46000E-06	3.1800	10.43	292	810.05	9.69	500.0	X	Jan 1–Apr 30	X	
Red Hind	20	0.14979	0.1100	571.00	-3.1000	6.16595E-06	3.1422	2.83	215	526.01	2.19	[a]200.0	X	Jan 1–Apr 30	X	
SNAPPERS																
Cubera	55	0.05447	0.0500	1440.50	-3.3300	4.90000E-06	3.1600	46.89	536	1362.54	39.33	300.0	X			
Dog	29	0.10330	0.1052	878.00	-1.4900	2.15343E-05	2.9679	11.73	476	842.48	10.37	300.0	X			
Gray	28	0.10699	0.1700	676.19	-0.0250	7.22000E-06	3.1100	4.57	230	670.43	5.34	250.0	X			
Hogfish	23	0.13025	0.1058	848.99	-1.3290	9.50000E-05	2.7452	10.43	177	784.27	8.39	300.0	X	Nov 1–Apr 30	X	
Lane	17	0.17622	0.1700	449.00	-2.5900	5.92000E-05	2.8600	2.28	240	432.93	2.05	200.0				
Mutton	40	0.07489	0.1650	799.05	-1.2300	1.47710E-05	3.0275	9.06	323	798.16	9.03	400.0	X	May 1–Jun 30		
Schoolmaster	42	0.07133	0.1200	482.00	-2.7900	9.26000E-06	3.1100	2.05	200	479.77	2.02	250.0	X			
Yellowtail	23	0.13025	0.1330	489.35	-3.1320	6.14000E-05	2.7790	1.83	232	474.20	1.68	260.0	X		X	
GRUNTS																
White	18	0.16643	0.5200	280.95	-0.5800	8.48881E-05	2.7500	0.46	167	280.93	0.46	[a]200.0	X			
Bluestriped	23	0.13025	0.3200	314.00	-1.8000	9.30830E-06	3.1315	0.61	205	313.89	0.61	[a]210.0				

[a] No minimum size regulation for this species; and, L_c was inferred from fishery-dependent length frequency distributions.
[b] Annual period of closure.

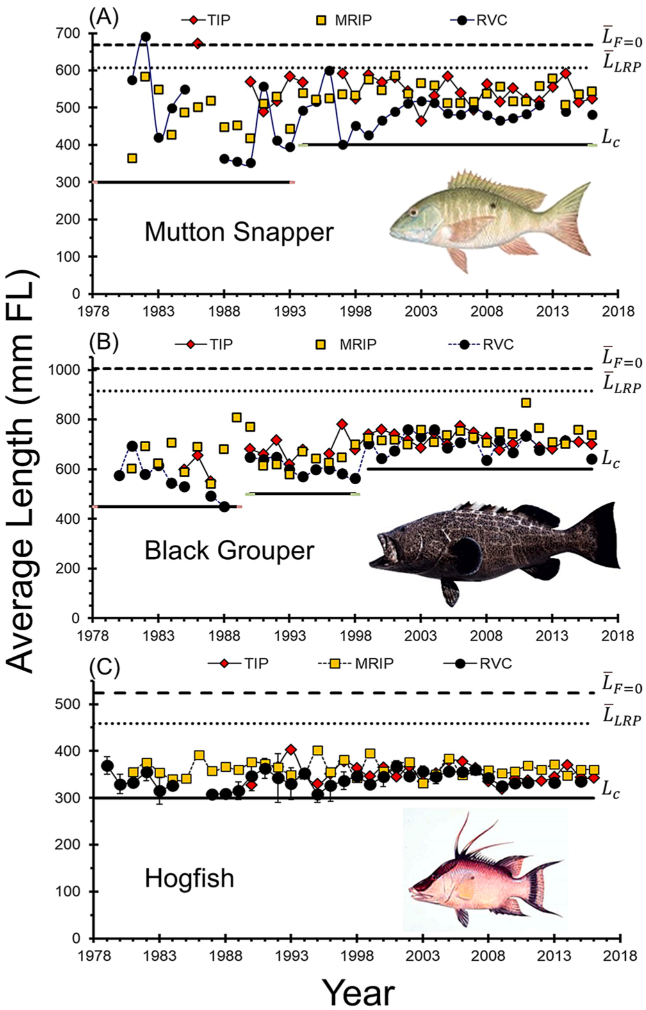

Fig. 3. Comparisons of $\bar{L}$ from 1979 to 2016 for three data sources for three southern Florida reef fishes: (A) mutton snapper; (B) black grouper; and, (C) hogfish. Solid horizontal is L_c, dotted horizontal line is $\bar{L}_{LRP}$, and dashed horizontal line unexploited $\bar{L}_{F=0}$.

levels in the southern Florida coral reef fish fishery greatly affects what management options are feasible to achieve sustainability goals. While fishery catchability affects the species differentially (Ault et al., 1998, 2005a), excessive fleet fishing power is systemic. The southern Florida coral reef fishery requires urgent management intervention.

3.3. Evaluation of fishery management interventions

Evidence of significant overfishing for the majority of the exploited and intensively managed southern Florida reef fish community provided impetus for exploring the effects of potential fishery management interventions via evaluation of reference points. To exemplify a range of reef fish community responses, we evaluated LRP and TRP intervention options for three priority species (i.e., mutton snapper, black grouper, and hogfish) by extending the probabilistic management options evaluation process to examine risks to population production and sustainability. Additionally, comparative summaries of fishery management intervention options results for all 15 principal species in the southern Florida reef fish community are given in Table 5. The three priority reef fish species, in particular, are among the most important to the valuable commercial and recreational reef fisheries, and are some of the most overfished species along the southern Florida reef tract.

3.3.1. Mutton snapper

An example for visualizing results is shown for mutton snapper in Figs. 6 and 7. Global surfaces dependent on L_c and F for "production" and "sustainability" options are given in Fig. 6**A–B**. Predicted potential yields, average size, average weight, exploited biomass, spawning biomass and reference points cut at the planes of BAU L_c and Limit L_c shown in Fig. 7**A–B** for a range of fishing mortality rates are extended to model-predicted population metrics (*i.e.*, $Y_w, B_{EX}, SSB, \bar{L}, \bar{W}$) scaled to their respective maximum values. In this representation, *SSB* and *SPR* are synonymous. For the BAU scenario, model-projected *SSB* (i.e., *SPR*) and yield-per-recruit over a range of fishing mortality rates are shown at the current regulated minimum size of $L_c = 400$ mm FL (Fig. 7**A**). The relationships among population metrics, fishing mortality rates, and LRPs show the inherent tradeoffs between fishery yield production and population sustainability.

Under the BAU scenario with "current" 2012–2016 conditions (i.e., $F_{BAU} = 0.2757, L_c = 400$ mm FL), the mutton snapper stock is at 12.6% SPR (Fig. 6**C–D**). First-order minimalist management involves investigation of two basic scenario strategies: (1) **Business as Usual (BAU) scenario** (i.e., *laisse faire* management that chooses to impose no changes in the 2012–2016 conditions); and a (2) **Limit scenario** (i.e., management raises L_c to at least the minimum size that produces 40% SPR, the LRP, Fig. 7**B**). Model-projected *SSB* (i.e., *SPR*) and yield-per-recruit over a range of fishing mortality rates are shown at the current regulated minimum size of $L_c = 400$ mm FL (Fig. 7**A**). The $F_{LRP} = 0.081$, meaning that the species is overfished and the fishing intensity is over-dimensioned, that is, the $\hat{F}_{BAU}$ is 3.4 times greater than the level considered minimally sustainable (i.e., $F_{40\%SPR}$, Fig. 7**A**). The average size in the exploited phase of the stock is $\bar{L} = 527.7$ mm FL. Under this scenario, the risk of overfishing is 100%. In the Limit scenario (Fig. 6**E–F**), the L_c was raised to 674.6 mm FL, which would increase SPR to 40%, the minimum threshold for sustainability. However, this strategy still results in ≥50% sustainability risks. After implementing policies to increase L_c, it would take the mutton snapper stock at least 9 years to recover to a point where the stock is considered minimally sustainable, and approximately 30 years to reach the new equilibrium (Fig. 10**A–B**). Under this scenario, the $\bar{L}$ of mutton snapper would increase to 714 mm FL and the risk of overfishing is 55.9%. When compared to the BAU scenario, SPR increases by 218%; $\bar{L}$ increases by 35%; $\bar{W}$ increases by 128%; and, fishery yield increases by 34.9% (Figs. 7**A–B,** 11**, and** Table 5). A risk averse management strategy would be one that invokes regulations that substantially reduced the fishing mortality rate. Under the Target scenario, L_c is raised to 674.6 mm FL, *and* fishing pressure is reduced by half to $F = 0.1378$. Under this scenario it would take mutton snapper 10 years to recover the stock to a point where it could be considered minimally sustainable (Fig. 10**A–B**) and the average size would increase to 728.4 mm FL while overfishing risks would decrease to 3.8% (Fig. 6**G–H**). When compared to the BAU scenario: SPR increased by 308.5%; $\bar{L}$ increased by 35.3%; $\bar{W}$ increased by 142.6%; and, annual fishery yields increased by 18.1%.

3.3.2. Black grouper

Under the BAU scenario with the current rate of fishing mortality ($F_{BAU} = 0.5432$) and minimum size of first capture ($L_c = 600$ mm FL), the black grouper stock is currently at 1.86% SPR. The $F_{LRP} = 0.0771$, meaning that the species is seriously overfished and the fishing pressure intensity is over-dimensioned by a factor of 7.05 times. The average size in the exploited phase of the stock is $\bar{L} = 728.9$ mm FL. Under this scenario, the overfishing risk is 100%. Under the Limit scenario (Fig. 8), the L_c was raised to 1100.97 mm FL, which would increase SPR to 40%, at which point black grouper would reach the minimum threshold for sustainability. After implementing policies to increase L_c, it would take the black grouper stock at least 10 years to recover to a point where the stock is considered minimally sustainable, and 22 years to reach the new equilibrium. Under the Limit scenario, black grouper $\bar{L}$ would increase

Table 3
Composite mean length ($\overline{L}$) and standard error (SE($\overline{L}$)) for 15 reef-fish species from three southern Florida regions for the period 2012–2016 estimated following Ault et al. (2019) from three statistical survey sources: RVC (reef fish visual census); TIP (commercial trip information program); and, MRIP (marine recreational information program). The southern Florida $\overline{\overline{L}}$ is the average of the three regions weighted by the proportion of mapped reef habitat area: Dry Tortugas, 0.250; Florida Keys, 0.543; southeast Florida, 0.207. L_c is the minimum size of first capture, CV($\overline{\overline{L}}$) is the coefficient of variation.

	L_c	Dry Tortugas		Florida Keys		Southeast Florida		Southern Florida		
		$\overline{L}$	SE($\overline{L}$)	$\overline{L}$	SE($\overline{L}$)	$\overline{L}$	SE($\overline{L}$)	$\overline{\overline{L}}$	SE($\overline{\overline{L}}$)	CV($\overline{\overline{L}}$)
GROUPERS										
Black	600	816.0	36.2	709.9	20.5	673.5	40.1	728.89	16.58	0.023
Goliath	450	1552.3	39.7	1363.6	31.2	1594.5	250.2	1458.46	55.29	0.038
Nassau	500	661.9	5.8	564.7	27.7			595.35	19.09	0.032
Red	500	592.0	10.1	582.1	16.4	566.0	25.3	581.24	10.61	0.018
Red Hind	200	377.2	21.5	358.6	28.6	310.6	33.6	353.33	17.86	0.051
SNAPPERS										
Cubera	300	732.0	40.0	551.1	134.3	480.6	0.3	581.73	73.68	0.127
Dog	300	608.2	21.5	426.4	41.5	344.9	7.2	455.00	23.22	0.051
Gray	250	329.3	15.1	320.2	6.1	315.2	20.0	321.43	6.51	0.020
Hogfish	300	387.1	20.3	348.2	7.7	354.1	15.7	359.15	7.34	0.020
Lane	200	257.3	21.7	255.8	11.1	240.1	12.4	252.91	8.50	0.034
Mutton	400	565.2	11.2	531.2	18.0	473.2	22.6	527.71	11.21	0.021
Schoolmaster	250	376.7	33.3	301.1	19.1	289.9	20.4	317.67	13.95	0.044
Yellowtail	260	308.8	3.7	309.8	3.8	304.9	21.0	308.54	4.90	0.016
GRUNTS										
White	200	242.6	6.0	236.8	3.2	238.3	8.5	238.53	2.88	0.012
Bluestriped	210	261.4	9.3	237.6	8.8	250.1	17.5	246.14	6.44	0.026

to 1136.6 mm FL and the overfishing risk is 54.3%. When compared to the BAU scenario, SPR increased by 2062.4%; $\overline{L}$ increased by 55.9%; $\overline{W}$ increased by 277.7%; and, fishery yield increased by 69.4% (Figs. 8E–F and 11**,** Table 5). Under the Target scenario, L_c is raised to 1100.97 mm FL, *and* fishing pressure intensity is reduced by half to $F = 0.2716$. It would take black grouper at least 10 years to recover the stock to a point where it could be considered minimally sustainable (Fig. 10**C–D**). Under this scenario, the average size of black grouper would increase to 1156.4 mm FL and decrease the overfishing risks to 3.8% (Fig. 8**G–H**). When compared to the BAU scenario: SPR increased by 2502.3%; $\overline{L}$ increases by 58.6%; $\overline{W}$ increases by 299.1%; and, annual fishery yields increase by 56.5%.

3.3.3. Hogfish

Under the BAU scenario with the current conditions ($F = 0.7459$, $L_c = 300$ mm FL), the hogfish stock is currently at 7.9% SPR. The $F_{LRP} = 0.122$, meaning that the species is seriously overfished and the fishing pressure intensity is over-dimensioned by a factor of 6.1 times. The average size in the exploited phase of the stock is $\overline{L} = 359.2$ mm FL. Under this scenario, the overfishing risk is 100%. For the Limit scenario (Fig. 9**E–F**), hogfish L_c was raised to 555.4 mm FL, which would increase SPR to 40%, at which point hogfish would reach the minimum threshold for sustainability. After implementing policies to increase L_c, it would take the hogfish stock at least 7 years to recover to a point where the stock is considered minimally sustainable, and approximately 15 years to reach the new equilibrium (Fig. 10**E–F**). Under the Limit scenario, the $\overline{L}$ of hogfish would increase to 585.9 mm FL and the overfishing risk is 65.2%. When compared to the BAU scenario, SPR increases by 408.5%; $\overline{L}$ increases by 63.1%; $\overline{W}$ increases by 271.0%; and, fishery yield increases by 71.7% (Figs. 9**E–F,** 11**, and** Table 5). Under the Target scenario, L_c is raised to 555.4 mm FL, *and* fishing pressure is reduced by half to $F = 0.3730$. Under this scenario, the average size of hogfish would increase to 605.2 mm FL, while the overfishing risks would decrease to 0.005% (Fig. 9**G–H**). When compared to the BAU scenario: SPR increased by 506.2%; $\overline{L}$ increased by 68.5%; $\overline{W}$ increased by 308.1%; and, annual fishery yields increased by 64.3%.

While the **Limit** scenario should achieve a base level of sustainability (e.g., $\geq$ 50% of SPR distribution values > 0.4), the risk of overfishing remained relatively large and risk prone. The **Target** scenario was profoundly risk averse because it also substantially reduced the fishing mortality rate. While in practice it may be difficult to reduce fishing pressure by exactly half, the intent of the Target scenarios was to illustrate the impacts of combining conventional fishery management techniques with other potential management actions that may effectively reduce the overall likelihoods or risks associated with overfishing.

4. Discussion

4.1. BAU strategy

The effects of different management options to balance fishery production and sustainability risks were explored for significantly overexploited coral reef fish species. For the business as usual (BAU) management regime, decision-metric combinations of F/F_{REF} and B/B_{REF} in the Kobe strategy format for the 15-species snapper-grouper reef fish complex showed that 13 of 15 species (86%) analyzed experienced high exploitation pressures (Fig. 5). LBRA analyses for three priority snapper-grouper species (i.e., mutton snapper, black grouper, and hogfish) showed that continuation of the BAU strategy results in 100% risk with respect to production (i.e., exceeding overfishing limits, Figs. 6, 8 and 9, **left panels**) and sustainability (i.e., below SPR requirements, Figs. 6, 8 and 9, **right panels**).

Comparison of exploitation effects by sub-regions of the southern Florida reef ecosystem indicated that more isolated coastal areas, further away from population centers, were least affected by fishing intensity, and vice versa (Fig. 4, Table 4), which indirectly demonstrated reducing fishing intensity as an effective management option. Increasing demands for reef fish resources services will continue to escalate proportional to threats from coastal development and human population growth (e.g., Fig. 2). Historic management strategies implemented in this fishery have not had their desired effects. Therefore, significant management changes are needed to ensure long-term sustainability of this ecologically and economically valuable reef fish fishery ecosystem.

Fishery management in the State of Florida was institutionalized in the mid-1980s with the formation of the Florida Marine Fisheries Commission to compliment the nascent regional Federal Fishery Management Councils that began in the 1970s, and to implement directed management actions in response to fishery declines and perceived threats. However, rulemaking for the southern Florida reef fishery did

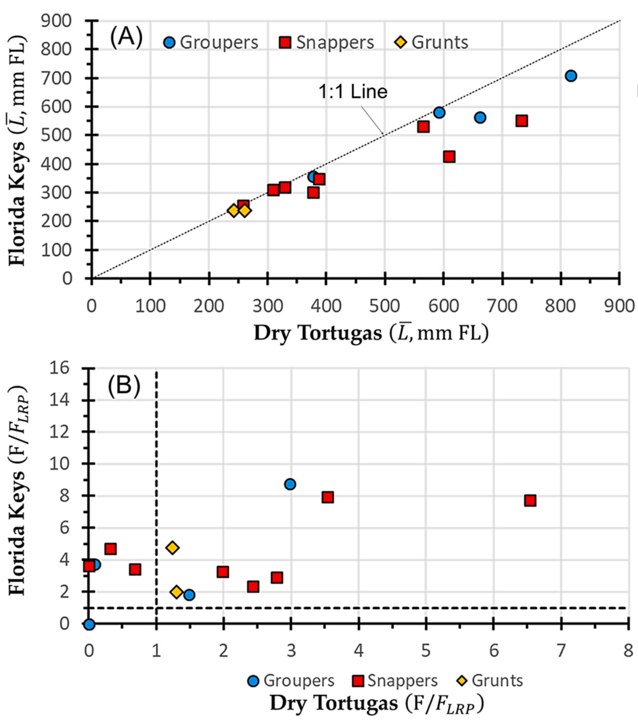

Fig. 4. Regional comparisons of $\overline{L}$ and F/F_{LRP} for (A–B) Florida Keys (Monroe and Miami-Dade Counties) and Dry Tortugas; and (C–D) Southeast Florida (Broward and Palm Beach Counties) and Dry Tortugas. Dashed diagonal line in panels A & C is 1:1; dashed horizontal and vertical lines in panels B & D are at minimal sustainability (i.e., $F = F_{LRP}$).

Table 4
Comparison of the distribution of F/F_{LRP} sustainability metric for 15 exploited reef fishes for the three subregions of the southern Florida coral reef ecosystem.

	Southern Florida Subregion		
F/F_{LRP}	Dry Tortugas	Florida Keys	Southeast Florida
0–1	6	2	1
$>1 - \leq 3$	7	4	5
>3	2	9	8
Not Present	0	0	1
Total	**15**	**15**	**15**

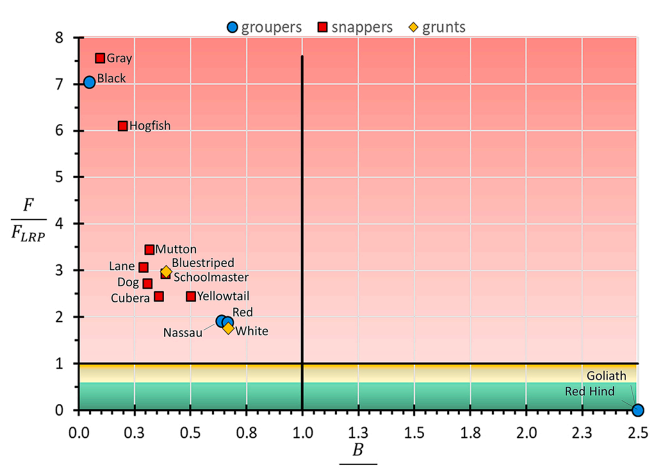

Fig. 5. Kobe strategy plot showing 2012–2016 median distribution values for 15 key southern Florida exploited coral reef fishes. Joint limit reference point is at median $F/F_{REF} = B/B_{LRP} = 1$. Overfishing in upper left quadrant III when $F/F_{LRP} > 1$ and $B/B_{LRP} < 1$. Target region for fishery sustainability is lower right quadrant I. Sustainability conditions: $\frac{F}{F_{LRP}} < 1$, $\frac{B}{B_{LRP}} > 1$, and SPR > 0.4.

not begin in earnest until the early 1990s, well after significant exploitation impacted fisheries (Ault et al., 1998). Regulations implemented since that time have included size limits, bag limits, gear restrictions, season limits, spatial closures, annual catch limits (ACLs), and even complete moratoria (Bohnsack and Ault, 1996; Bohnsack, 1998; Meester et al., 2004; Bohnsack et al., 2004; Ault et al., 2005a, 2013) (Table 2). Whatever the means utilized, results indicate that sustainability risks remain impossibly high at 100% (Figs. 6, 8 and 9, **Panels C–D**). While some management regulations are currently perceived as successful (e.g., yellowtail snapper, SEDAR 64, 2020), which have included a broad mix of regulations (Table 2), they are in fact not sufficient to mitigate rising risks to sustainability documented here (Fig. 5).

Importantly, despite the known risks and threats, few coral reef fish stocks have undergone assessments by NOAA's Southeast Data, Assessment, and Review (SEDAR, www.sedar.com) program. Notably, in 2018 the L_c's for mutton snapper and hogfish were increased by FWC to 450 and 400 mm FL from 400 and 300 mm FL, respectively. The necessity of the changes for mutton snapper and hogfish recommended in the 2017 SEDAR process were supported by the earlier findings of Ault et al., (2003, 2005b). However, these management changes in L_c, without a change in fishing intensity, could only be expected to increase SPR to about 15%, well below the 40%S SPR minimum for sustainability. Thus, while the 2018 L_c's lowered the risks, the changes in L_c were not sufficient to achieve sustainability.

It is clear that BAU reef fishery management regulations do not achieve sustainability goals, evidenced by the high likelihood of overfishing and extreme over-dimensionality (i.e., fishing power deployed per unit of area and time) of the recreational fleet. While the early days of the fishery were dominated by the commercial fleet, recreational fishing pressures that began in the 1940s are projected to double over the next 25 years. Under such fishing pressure growth, expectations of fish stock sustainability are not achievable. Therefore, drastic revisions to management strategies are needed to build long-term sustainable fisheries in southern Florida. The goal of the necessary sweeping changes in management strategies should be revised to maintain reef

Table 5
Comparative summary of LBRA results and implications for southern Florida reef fisheries resources sustainability. Symbols are defined in Table 1. BG is black grouper; GG is goliath grouper; NG is Nassau grouper; RG is red grouper; RH is red hind; CS is cubera snapper; DS is dog snapper; GS is gray snapper; HF is hogfish; LS is lane snapper; MS is mutton snapper; SM is schoolmaster snapper; YT is yellowtail snapper; WG is white grunt; BSG is bluestriped grunt.

	GROUPERS (5)					SNAPPERS (8)								GRUNTS (2)	
	BG	GG	NG	RG	RH	CS	DS	GS	HF	LS	MS	SM	YT	WG	BSG
Unexploited $\overline{L}$ (F = 0)	1,005.19	1,346.69	655.83	663.59	344.20	811.68	571.03	499.30	524.04	313.95	666.65	389.15	366.88	259.22	281.22
BAU															
L_c	600.0	450.0	500.0	500.0	200.0	300.0	300.0	250.0	300.0	200.0	400.0	250.0	260.0	200.0	210.0
$\overline{L}$	728.9	1,458.5	595.4	581.2	353.3	581.7	455.0	321.4	359.2	252.9	527.7	317.7	308.5	238.3	250.1
$\overline{W}$	6.16	68.92	3.09	3.46	0.72	4.17	1.98	0.56	1.02	0.47	2.84	0.60.	0.52	0.28	0.29
Y_W/R	3.9022	0.0000	1.0002	1.3454	0.0000	2.5034	0.9881	0.3690	0.6078	0.2920	1.7845	0.3611	0.2777	0.1486	0.1854
SPR	1.86%	100.00%	25.56%	26.57%	100.00%	12.29%	14.36%	3.82%	7.87%	11.48%	12.57%	15.46%	19.98%	26.72%	15.65%
F	0.5432	0.0000	0.3567	0.2780	0.0000	0.0978	0.1833	0.7373	0.7459	0.4537	0.2757	0.2201	0.3651	0.4059	0.4537
F_{LRP}	0.0771	0.0390	0.1580	0.1470	0.1160	0.0363	0.0756	0.0960	0.1220	0.1475	0.0810	0.0750	0.1491	0.2320	0.1580
OFL	7.05		1.92	1.89		2.69	2.42	7.63	6.11	3.08	3.40	2.93	2.45	1.75	2.87
Sustainability Risks	100.0%	0.0%	100%	98.1%	0.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	99.9%	100.0%
Limit															
L_c @40% SPR	1,100.97		583.08	594.01		868.01	572.39	554.05	555.43	313.52	674.64	382.70	339.97	241.77	275.91
SPR	40.0%		40.0%	40.0%		40.0%	40.0%	40.0%	40.0%	40.0%	40.0%	40.0%	40.0%	40.0%	40.0%
$\overline{W}$	23.28		4.28	4.92		16.0	5.00	2.83	3.79	1.04	6.47	1.26	0.85	0.36	0.47
Y_W/R	6.6106		1.0403	1.4397		4.5071	1.3252	0.8471	1.0434	0.3499	2.4070	0.4568	0.2992	0.1522	0.1981
Years to BAU Y_W/R	10			8				9	7		10	11	8		
Years to new equilibrium	23			19				18	15		37	28	19		
Sustainability Risks	54.3%							56.7%	65.2%		55.9%		50.1%		
Target															
Target F	0.2716		0.1783	0.1390		0.0489	0.0917	0.3687	0.3730	0.2269	0.1378	0.1100	0.1825	0.20297	0.22686
L_c @40% SPR	1,100.97		583.08	594.01		868.01	572.39	554.05	555.43	313.52	674.64	382.70	339.97	241.77	275.91
SPR	48.33%		53.68%	53.79%		56.56%	57.40%	47.11%	47.71%	53.40%	51.35%	52.41%	52.60%	54.84%	51.82%
$\overline{W}$	24.60		4.81	5.48		18.27	5.56	3.01	4.17	1.15	6.89	1.36	0.94	0.38	0.50
Y_W/R	6.1065		0.9072	1.2527		3.7383	1.0665	0.7998	0.9985	0.2999	2.1080	0.3973	0.2608	0.1236	0.1699
Sustainability Risks	3.8%							0.2%	0.005%		7.9%		0.7%		

fish sustainability that is currently threatened by dramatic increases in reef fish demand.

4.2. *Limit strategy*

To address how management options address systemic overfishing, we considered three management strategies: (1) **BAU**; (2) **Limit**; and, (3) **Target**. The Limit Strategy (LS) defined a precautionary reference point for fishing mortality of $F_{40\%SPR}$. This reference point is also known as the precautionary limit reference fishing mortality point F_{LRP} decision metric following Ault et al. (2019). The LS found that at F_{BAU}, simply increasing L_c to reach at least the *SPR*40% rarely achieved sustainability and constituted a minimalist strategy to reduce overfishing risks for any stock undergoing management in the southern Florida exploited reef fish community. Minimum size restrictions were previously used by fishery management for 12 of the 15 of the species examined here, but the size increases were not sufficient to affect minimal sustainability (Table 5). To some extent, minimum size limits, assuming knife-edged selection, act like a complete fishery closure until which time species transition in size from the old L_c to the new Limit L_c. The LS simulations presented here assumed zero catch-release mortality, an assumption which in practice will be violated because changes in minimum size will likely confer some mortality to incidentally caught fish $< L_c$. A possible solution to the catch-release mortality dilemma would be to determine whether or not particular gear characteristics or fishing areas could shift the L_c distribution upwards to minimize capture of reef fish $< L_c$. This could be achieved through cooperative research between fishers and scientists to identify hook types, hook sizes, and different baits that routinely catch fish larger than some desired L_c.

The probabilistic Limit scenarios that raised L_c to achieve the 40% SPR level for the species evaluated were only able to reduce sustainability risks to about 50% (Figs. 6, 8 and 9, **Panels E–F**). Thus, this suggests that relying on L_c is insufficient, though it does help increase SPR. Indeed, when pressed for action the LS is what BAU managers have implemented in southern Florida throughout a period of increasing fishing mortality. The new LS L_c was selected as a function of constant fishing mortality, and while Y_w is essentially asymptotic even as F increases, SPR could be improved by increasing L_c above the new LS L_c. Conceptually, assuming no dramatic negative trend in recruitment, the LS could produce essentially 40% SPR, even at infinite F. However, since such a condition is unrealistic, a second control option needs to be implemented that reduces F to lower sustainability risks. However, lowering F when future increases in the demand for reef fish services are unabated requires additional control options that significantly limit demand.

4.3. *Target strategy*

Meaningful reductions in sustainability risks for over-exploited fish in south Florida did occur based on changes tested in the Target Strategy (TS), where increased minimum size limits (L_c) were raised to 40% SPR "eumetric" levels as in the **Limit** strategy, and fishing pressure (F) was reduced by 50% (Figs. 6, 8 and 9, **Panels G–H**). The LS attained about 50% risk, which is right on the brink of being overfished. However, our analyzes did not specifically include catch-release mortality. The idea of our TS was to introduce a buffer in population size to ensure risks were well below 50%. Thus, in our case a TRP that adjusted L_c upwards to 40%SPR and lowering F to $0.5F_{BAU}$ effectively reduced the risks to almost zero. Perhaps that is overly conservative. However, a conservative approach to start could in practice be modified by managers, stakeholders and scientists based on an acceptable level of risk for each species under review.

To significantly mitigate sustainability risks a blend of significant size limit increases, combined with drastic fishing effort reductions, will be needed across the range of exploited reef fish species analyzed. To that end, extended analyses of L_c could be conducted based on man-

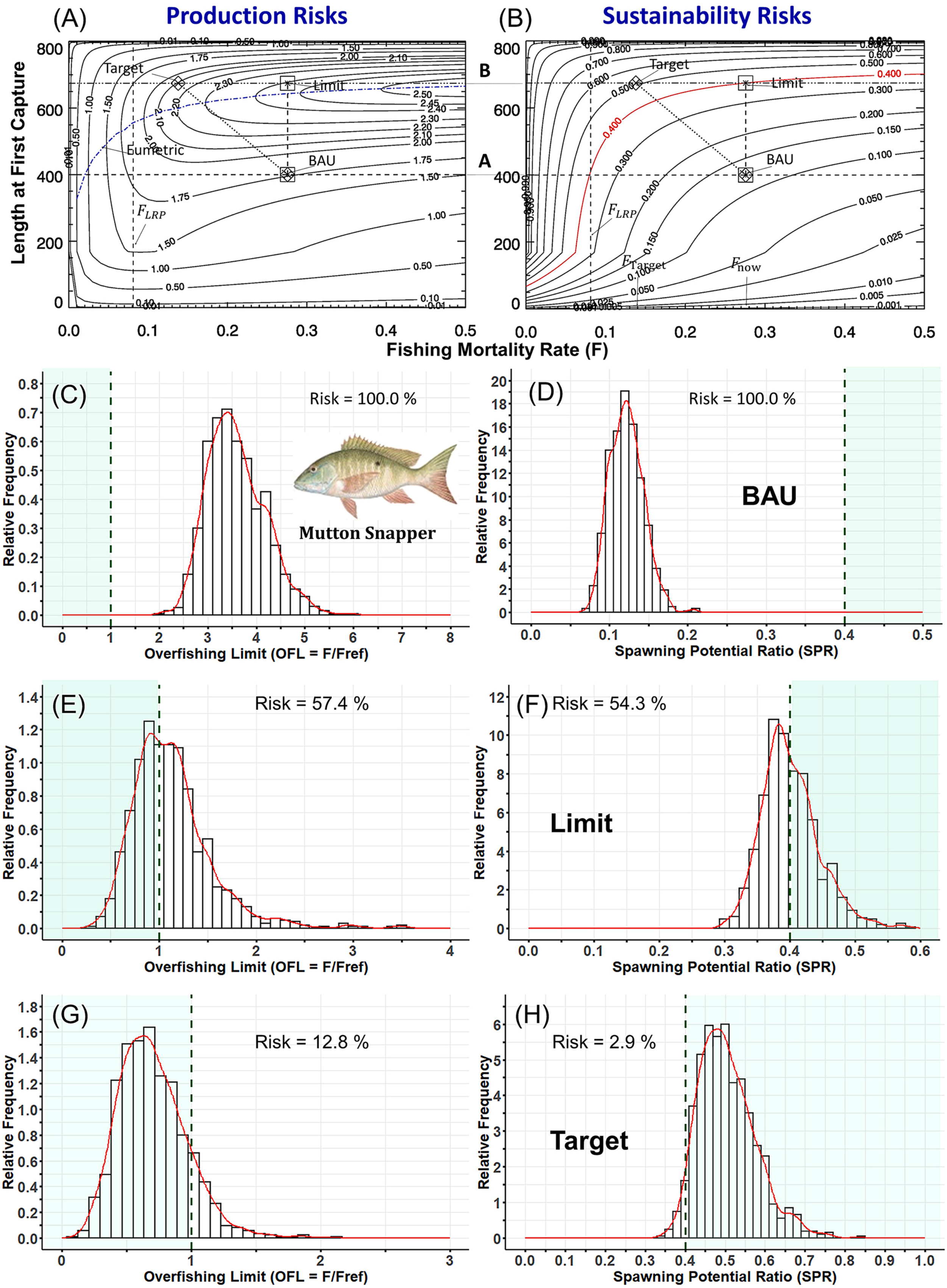

Fig. 6. Mutton snapper management options for: (A) **production** (Y_w/R); and, (B) **sustainability** (SPR) dependent on fishing mortality rate F and minimum size L_c. Horizontal dashed lines are L_c for business as usual (BAU, lower) and limit (upper) strategies. Dashed blue line is "eumetric", vertical dotted line is F_{LRP}. Sustainability risk distributions for: (C–D) business as usual (BAU); (E–F) limit; and, (G–H) target strategies. Green shaded areas denote sustainable population size. Dashed lines in (A–B) are the L_cs for BAU and Limit strategies cross-sections, respectively, shown in Fig. 7.

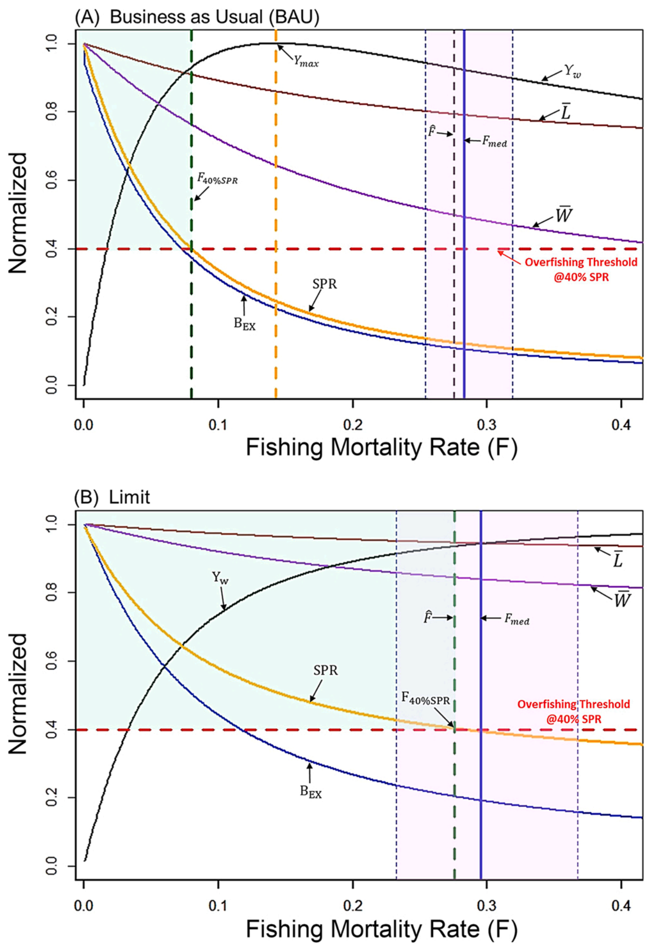

Fig. 7. Fundamental principles of evaluation of management options for two mutton snapper decision scenarios: (A) Business as usual (BAU, $L_c = 400$ mm FL); and, (B) Limit ($L_c = 674.64$ mm FL). Both panels show the relationships of relative changes in five population metrics scaled to their respective maximum values dependent on the fishing mortality rate *F*: (1) average length ($\overline{L}$, solid brown line) in the exploitable phase; (2) yield in weight, (Y_w, solid black line); (3) exploited biomass (B_{EX}, solid blue line); (4) spawning stock biomass (*SSB*; solid gold line); and, (5) average weight ($\overline{W}$, solid purple line). Horizontal red dashed line depicts the 40% SPR (i.e., SSB in this representation); vertical dark green dashed line is $F_{40\%SPR} = F_{LRP}$ fishing mortality LRP (limit reference point). Estimated $\widehat{F}$ distribution (light red), and median (F_{med}) value and respective 1st and 3rd quartiles. Green shaded areas denote sustainability.

agement controls such as limited entry, bag limits, closed seasons, and no-take marine reserves. However, the pros and cons of these options have important implications.

Limited entry attempts to control the level of fishing mortality (*F*) by reducing nominal fishing effort (*f*). For example, decreasing the number of boats, decreasing the number of days fished, or limiting the number of participants through the use of fishing licenses. Limited entry has not been used in the southern Florida recreational fishery, but there has been some minor implementation of limited entry for commercial reef fishers in some areas. Direct limits on nominal fishing effort are possible for small, concentrated industrial-commercial fleet fisheries. However, this type of effort control is not realistically applicable with the continuing exponential rise in the demand occurring in south Florida. Moreover, the limited entry system required to effectively control *F* in this fishery would be at the huge social cost of reducing the number of boats and people participating; however, implementing this option may be the only hope to maintain a sustainable fishery. While this is a scientifically sound and effective approach, there are significant societal and political considerations that operate outside the constraints of our models. Those "considerations" are outside the realm of science, in other words.

Recreational bag limits control the level of *F* by reducing how many fish a person is allowed to keep per day. This management option could alternatively be imposed as a per boat limitation. In south Florida, bag limits exist for most of the 15 species in the snapper-grouper complex. As a result, the localized depletion of reef fish spawning aggregations (Erisman et al., 2011) and schooling species (Le Pape et al., 2017) are prevented. However, when reef fish populations are already depleted, bag limits are not an effective regulation to reduce exploitation. For example, generally a bag limit of > 1 fish will have no impact when 99.9% of all anglers do not catch anything when the exploited phase resource is completely depleted (Ault et al., 2009), and of course, bag limits < 1 are impossible. Furthermore, in the situation of excessive nominal fishing effort, any potential reductions in fishing mortality due to reduced bag limits will quickly be erased by increases in nominal fishing effort because of the steady increase in people fishing the resources. Bag limits also suffer from catch-release mortality (e.g., hooking, handling, barotrauma, etc.), that is difficult, if not impossible, to quantify.

Seasonal closures are used for some species in south Florida, mainly implemented during peak season for spawning aggregations at predictable locations, helping to mitigate their localized depletion (Clarke et al., 2015). Closed seasons too have potential; however, the extraordinary level of nominal fishing effort creates a "derby fishing" environment during the open season. Southern Florida's warm winters makes the region a winter haven for sportfishing enthusiasts, thereby exacerbating seasonal recreational fishery demands and actually confining annual demand to a few months during the year (Friedman, 1988; Happel and Hogan, 2002; Smith and House, 2006). Such conditions significantly mar the use of closed seasons, except if implemented seasonally during peak periods of effort.

Annual catch limits (ACLs), mandated under the Magnuson-Stevens Fishery Management and Conservation Act, attempt to limit overfishing and control *F* by limiting total catch (Shertzer et al., 2008). There has been a movement to use ACLs for the reef fishery. This method essentially shuts down the entire fishery for some time within a year when quotas are exceeded, but this control method is only designed for management of a single species and accurate catch and effort reporting is notoriously problematic for recreational fisheries (Sullivan et al., 2006). Also, the catch-release mortality problem remains in full effect for all sizes and species vulnerable to whatever gear is used. This occurs because fishing still occurs on these species, even though they can't be kept for consumption.

Moratoria, or extended time-area closures, work like seasonal fishery closures, but over extended periods of time to protect over-exploited, threatened or endangered species that have reached critically low population abundance levels. Typically, reduced fishing mortality increases population growth rates (Escalle et al., 2016). Due to severe stock depletion, Nassau and Goliath grouper have been under complete moratoria since 1990. While Goliath grouper give subtle appearance to be recovering (Koenig et al., 2011), the same cannot be said for Nassau grouper whose stock has been declared threatened. Catch-release mortality and outright non-compliance continues to contribute to the Nassau grouper overfishing problem.

Placing a portion of the reef fish community under some type of spatial protection (e.g., areas where extractive fishing is prohibited) could be accomplished through a broadened spatial network of no-take marine reserves (NTMRs, areas where fishing is prohibited; Meester et al., 2004; Ault et al., 2005a, 2006, 2013). NTMRs shut down part of the fishery, but of course, people can fish elsewhere. In reserves, the catch-release mortality problem is avoided. NTMRs do not eliminate fishing effort from the fishery per se, but rather displaces it in space. In theory, they reduce current *F* to zero inside and increase current *F* outside, with the intent of a net reduction in average stock-wide *F*

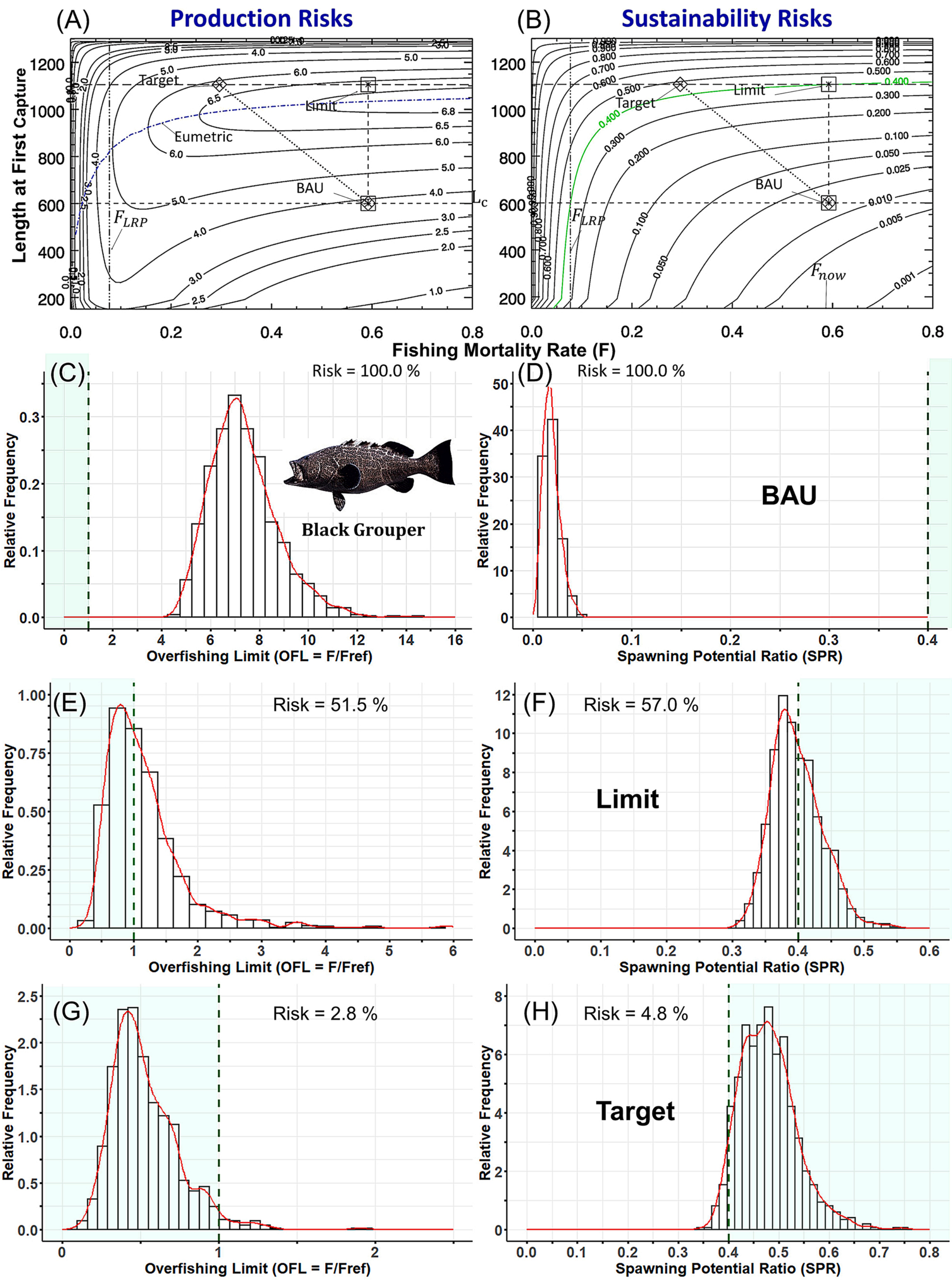

Fig. 8. Black grouper management options for: (A) **production** (Y_w/R); and, (B) **sustainability** (SPR) dependent on fishing mortality rate F and minimum size L_c. Horizontal dashed lines are L_c for business as usual (BAU, lower) and limit (upper) strategies. Dashed blue line is "eumetric", vertical dotted line is F_{LRP}. Sustainability risk distributions for: (C–D) business as usual (BAU); (E–F) limit; and, (G–H) target strategies. Green shaded areas denote sustainable population size.

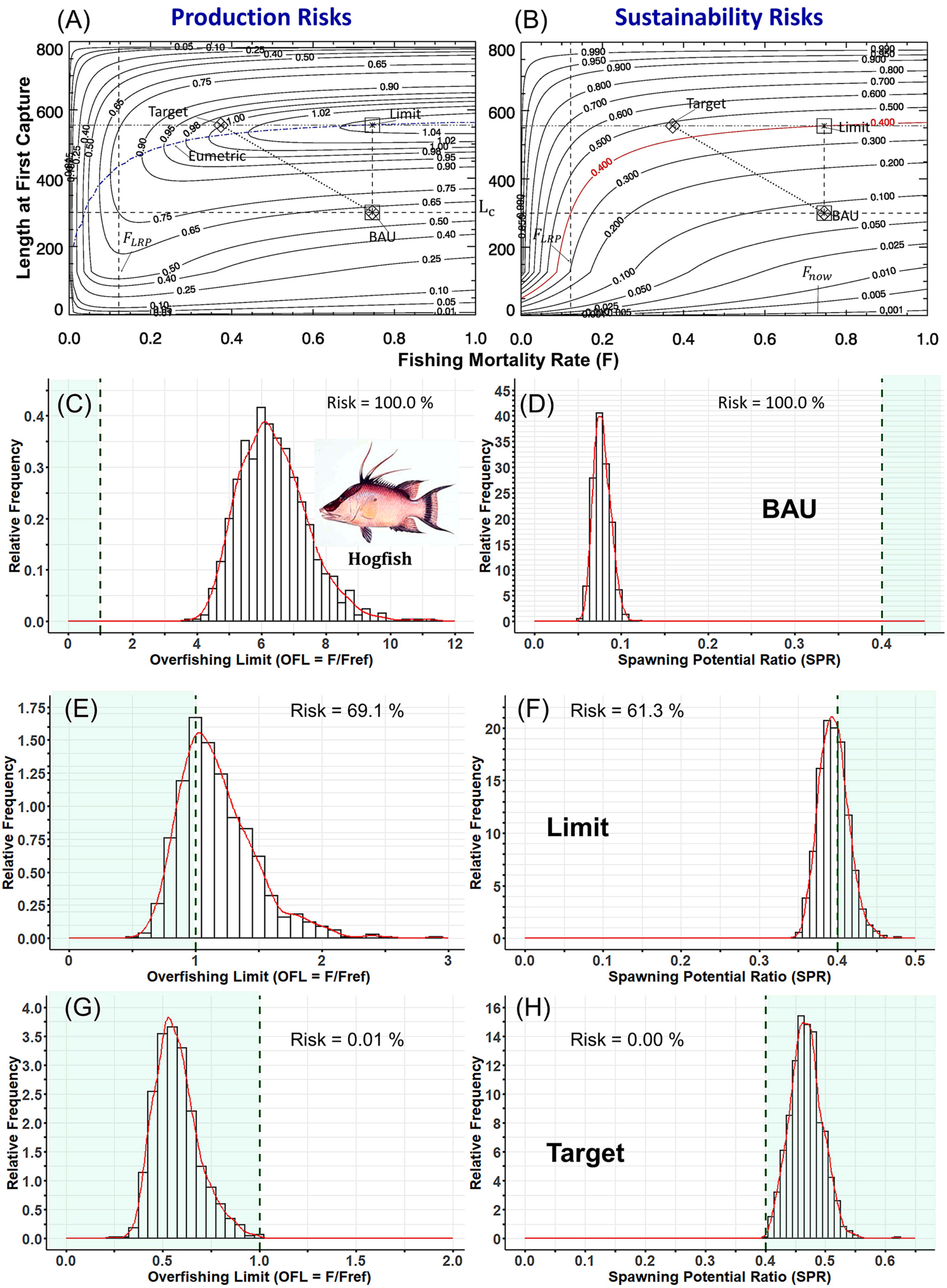

Fig. 9. Hogfish management options for: (A) **production** (Y_w/R); and, (B) **sustainability** (SPR) dependent on fishing mortality rate F and minimum size L_c. Horizontal dashed lines are L_c for business as usual (BAU, lower) and limit (upper) strategies. Dashed blue line is "eumetric", vertical dotted line is F_{LRP}. Sustainability risk distributions for: (C–D) business as usual (BAU); (E–F) limit; and, (G–H) target strategies. Green shaded areas denote sustainable population size.

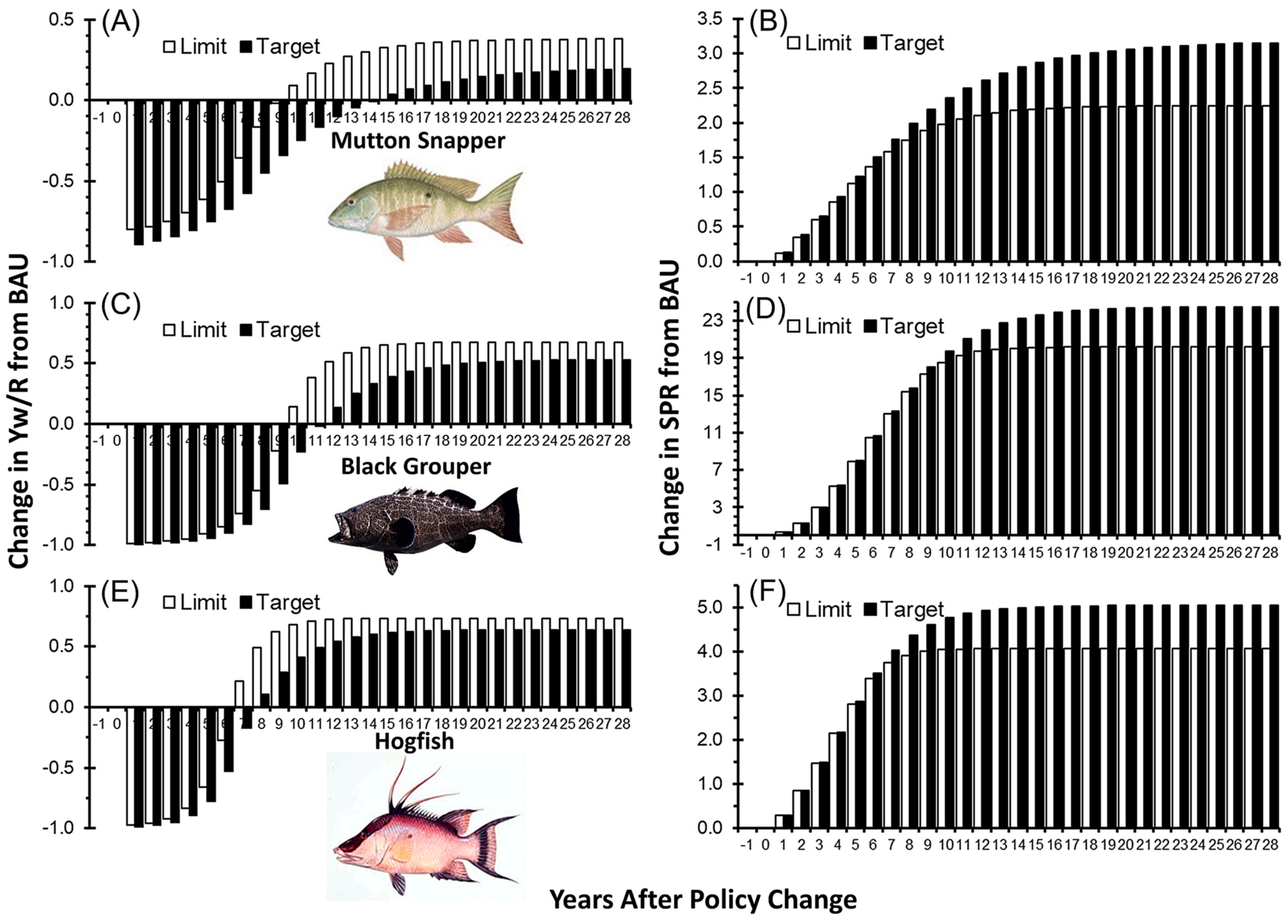

Fig. 10. Simulation analysis of resource recovery after policy implementation in year 0 for Limit and Target management strategy scenarios: (A–B) mutton snapper; (C–D) black grouper; (E–F) hogfish snapper. Left panels are change in yield in weight per recruit (Y_w/R) and right panels are change in SPR relative to BAU (Business as Usual) initial conditions.

(Meester et al., 2001). However, the assumption that fish migration out of the reserved areas may be significant could reduce the expected results.

Implemented as small Sanctuary Preservation Areas in the Florida Keys, and Ecological Reserves and Research Natural Areas in the Dry Tortugas, NTMRs have had positive impacts on Florida reef fisheries (Ault et al., 2006, 2013; Bartholomew et al., 2008), and elsewhere (Roberts et al., 2001; Gell and Roberts, 2003; Mumby and Harborne, 2010; Sala and Giakoumi, 2018; Sala et al., 2021). Regional analysis shows lower F in Tortugas (Ault et al., 2013), an area the furthest from human population centers, but also the location of the largest marine reserves. Although effective, not enough area in southern Florida has been protected to reduce stock-wide F to significantly mitigate sustainability risks.

4.4. Reflections on the analytical approach and its challenges

We used the LBRA data-limited methodology under various management options to explore harvest control rules (HCRs) based on reference points and to make probability statements about the (i) current stock status and (ii) current rate of fishing mortality relative to reference points (Methot et al., 2014; Maunder et al., 2020). HCR reference points are commonplace components of US Fishery Management Plans, but they of course greatly depend on precise knowledge of demographic and population processes (Maunder and Piner, 2014). Evaluating management options involves an assessment of demographic and population data, simulation analyses, and candidate HCRs to consider what options best meet management objectives.

In our study, we analyzed fifteen of the > 40 species of the exploited reef fish complex. We hedged on using unreliable demographic and stock dynamic process data that would have required us to make some unrealistic assumptions. While there may be a preferable method to estimate the true natural mortality rate (M), we noted that severely truncated population age-size structure(s) made it unlikely that fish survived to ages older than that observed in the fishery-dependent (FD) data. In addition, there were scant data available from the fishery for fish $< L_C$, making it difficult to estimate recruitment. Lack of contrasts in stock size that are necessary for robust stock-recruitment relationships were not available due to the relatively late start to the collection of data ($\geq$ 1979), which was after the period of significant and intense exploitation that greatly reduced stock size(s). As a consequence, we chose $h \approx 1$ for these analyzes, a value similar to what has been done for a range of species including Pacific bluefin tuna assessments (ISC, 2018).

Length composition data from FD and fishery-independent (FI) sampling programs were used to estimate $\overline{L}$ within fishing regions. In some circumstances there may be a clear choice for a single program that representatively samples the full spatial distribution of target species, but this was not the case for our study. The diver FI survey employed a sophisticated probability sampling design that stratified by environmental features (reef habitat type, depth) and spatial management zones (inside, outside marine reserves), but the survey sample frame was restricted to safe SCUBA diving depths of < 33 m (Smith et al., 2011). Commercial and recreational fishing covered the full depth range of the reef fish complex analyzed in this study (0–50 or 60 m), but the fleets were prohibited from fishing inside a network of marine reserves in the Florida Keys and Dry Tortugas regions. Taken together, the

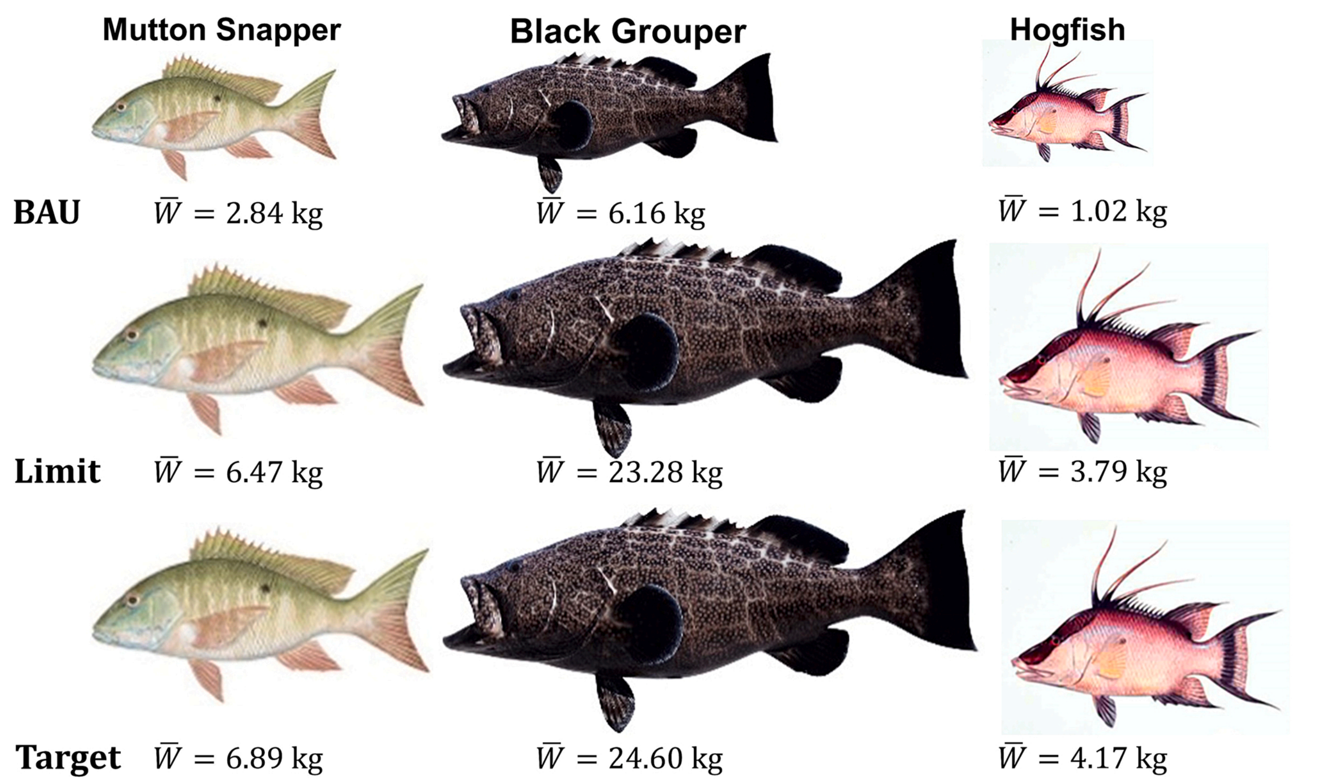

Fig. 11. Relative changes in average size (weight in kg) under the three management option scenarios for mutton snapper, black grouper, and hogfish.

FI and FD data sources encompassed the full space inhabited by the target species. It may have been preferable to estimate a single FD $\bar{L}$ before combining with the FI $\bar{L}$ via the standard practice of weighting the commercial and recreational length frequencies by their respective proportions of total landings. However, this presumes the availability of reliable catch data. As discussed in Ault et al. (2019), the data-limited LBRA approach is of high utility for cases where length composition data are available but catch-effort data are not. This is a common situation for tropical reef fisheries within U.S. jurisdiction (c.f., Ault et al., 2008), and southern Florida is no exception. The problem concerns recreational landings, which are computed by multiplying catch-per-person trip estimated from dockside intercept surveys with estimates of total fishing effort (person-trips) obtained from a separate telephone or mail survey. Depending on the survey methodology used (telephone or mail), estimates of total effort and subsequent total catch differ by 200–300%, and this discrepancy has not yet been resolved (NOAA MRIP online database). Hence our approach of considering the data sources as three separate sampling realizations for the same species, weighted equally to compute the composite $\bar{L}$.

Comparison of $\bar{L}$ between FD and FI data sources strongly suggested similar size selectivity of gears for the respective commercial and recreational fleets, as verified by the FI data (Fig. 3). That particular observation was not entirely surprising since, theoretically, if selection in the exploited phase of the stock were similar between the fishery-dependent (FD) and fishery-independent (FI) data sources, then $\bar{L}_{FD} = \bar{L}_{FI}$. The conclusions we reached were not particularly sensitive to the $\bar{L}$ mean and variance since $\bar{L}$ did not vary much between years and gears. Estimation of $\bar{L}$ was carried out over a 5-year period to alleviate sparse fishery-dependent length observations for less common species. In addition, the estimated magnitude of F puts $\bar{L}$ in an asymptotic region of the $\bar{L} = f(F, demographics)$ function. These factors strongly suggested that the knife-edged selection assumption was reasonable. Nonetheless, innovative methods to move FI surveys to depths greater than those of SCUBA divers are warranted, perhaps by using a strategic blend of 3D cameras and the dedicated effort of fishers distributed in a rigorous statistical survey design (e.g., Ault et al., 2018).

A constraint of the LBRA method is that it does not allow for the separation of commercial and recreational fleet effects in total mortality estimation. However, for data limited situations the LBRA does provide valuable insights into the stock status that can provide guidance for management decisions. For example, identification of the minimum sustainable population size (i.e., > 40% SPR), accepting the necessary trade-offs between what is taken out and what is left in the water. The simulation methods of our numerical model allowed insights into the transitional (non-equilibrium) stock dynamics (Fig. 10). These showed that stock recovery times varied relative to the species' life history demographics, and that substantial lead time must be built into expectations for stock recovery. In summary, the LBRA data-limited method provides a consistent basis for cross-checking outputs of contemporaneous high-parameter age-based statistical stock assessment models.

The length-based risk analysis (LBRA, Ault et al., 2019) model presented here uses life history parameters and a finite lifespan equation for mean and variance of lengths in the catch to estimate total mortality with zero bias at equilibrium (Ehrhardt and Ault, 1992), and a length-based simulation framework to validate estimates of size frequency distributions and calculate various reference points that quantify fishery sustainability risks. There are several other infinite lifespan length-based approaches that can also use this exact type of data, but they do things somewhat differently. The length-based spawning potential ratio (LBSPR, Hordyk et al., 2015) model accommodates incomplete knowledge of life history demographics by using "life-history invariants" to assess stock status by comparing the spawning potential as measured through the length composition data to that expected in an unfished stock. The length-based biomass (LBB, Froese et al., 2018) is a simple method for estimating relative biomass for the exploited size range from length-frequency data. Length-based pseudo-cohort analysis (LBPA, Canales et al., 2021) is an equilibrium alternative to LBSPR that uses multiple catch length-frequency distributions and penalized maximum likelihood. The length-based integrated mixed effects (LIME, Rudd and Thorson, 2018) model requires a single year of length composition data and biological data and relaxes equilibrium assumptions and attempts to account for time-varying recruitment and fishing mortality (though assumes constant selectivity), and derives population parameters associated with an age-structured model and length compositions. Some length-based methods (LBSPR) use simplified life history measures, LBB uses some demographics and simplified life history measures, while others (LIME) use life history more similar to LBRA to formally estimate total and fishing mortality. As a consequence, LBRA, LIME and LBPA provide absolute mortality estimates which facilitate estimation of stock abundance, biomass and the SPR decision metric. LBSPR and LBB provide only relative estimates.

4.5. Towards a community perspective

Coral reef fishery resources are part of complex food chains that integrate complicated population dynamics. As a result, in the absence of exploitation, population size/age structures of the reef fish community reflect a delicate evolutionary balance between reproductive and natural mortality processes that are driven by spatial and temporal reef ecosystem processes. Today's fishing intensity (i.e., the amount of nominal fishing effort per unit area) and demand from the fully utilized fishery resource far exceeds the capacity of natural surplus production. The delicate balances that previously characterized how reef ecosystems functioned pre-exploitation are therefore seriously disrupted. Perhaps the ineffectiveness of the many effort control measures implemented in southern Florida is because they are more suited for single species fishery situations rather than the multispecies complex that characterizes the reef ecosystem. A problem with traditional species-by-species management controls is that none actually prevent the capture of a particular species. Instead, fishing is allowed to continue across all the species in the reef fish community. In these circumstances, catch-release mortality becomes a paramount hindrance that limits the effectiveness of regulations such as bag limits, closures, ACLs and moratoria. Another practical problem limiting the effectiveness of these regulations is the assumption that fishers can correctly identify all the different species under regulation, or in fact, remember the regulations. While this might be a plausible assumption for commercial and recreational for-hire fishers, it is generally not true for many private vessel recreational fishers (Lyon et al., 2018), which currently dominate the southern Florida fishing community.

Notably, any increase in fishery L_c acts somewhat like an ecosystem-wide no-take marine reserve (NTMR) for fishes that are below the regulated L_c. However, the regulation is based on the assumption that those caught and released will survive inadvertent interactions with the fishery, and that these fishes will theoretically not be available to the fishery until transit through the time lag to attain the new L_c (e.g., Hastings and Botsford, 1999). This also assumes, of course, that the fishery can effectively target larger fish. On the other hand, decreases in F may be somewhat harder to implement given public reticence to direct restrictions on people's participation in the fishery. An advantage of NTMR spatial closures is that, if sufficient area is protected as a refuge from fishing, more fishing trips and longer fishing seasons may be possible.

In our analyses, we found that BAU policies have large risks, and Limit strategies, while reducing risks to about 50% are insufficient to sustain fishery stocks. As a result, size limits need to be considered at the community level across all species. The path forward that implements an appropriate Target strategy to rehabilitate reef fish populations along the southern Florida reef tract will include biological and political challenges. Ultimately, rehabilitating the southern Florida coral reef fishery requires management intervention by government agencies. In that regard, fishery managers must consider sophisticated information systems about reef fish populations to achieve sustainability of the entire community of exploited reef fishes, and sufficient numbers of reef fish to sustain both populations and the functioning of the entire reef community. There is also a great need for fishery users to understand and fully appreciate the ecological and economic value of these species. To ensure the persistence and sustainability of these important reef fisheries and the services they provide, a significant change in perspective is needed to foster their protection and management. The change in perspective must specifically address what are considered acceptable yields from the fishery resource. Given the long stock recovery times forecasted here, there is immediate need for management actions that combine L_c, F, area, and time because BAU reminds us that doing the same thing over and over and expecting a different result is sheer folly.

Disclaimer

The scientific results and conclusions, as well as any views or opinions expressed herein, are those of the authors and do not necessarily reflect those of NOAA or the Department of Commerce, or NPS and the Department of the Interior.

CRediT authorship contribution statement

JSA developed the concept of the paper; **JSA** and **SGS** created the initial draft and structure; All authors contributed to writing and editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Amorim, P., Sousa, P., Jardim, E., Azevedo, M., Menezes, G.M., 2020. Length-frequency data approaches to evaluate snapper and grouper fisheries in the Java Sea, Indonesia. Fish. Res. 229, 105576.
Aronson, R.B., Precht, W.F., 2001. White-band disease and the changing face of Caribbean coral reefs. In: Porter, J.W. (Ed.), The Ecology and Etiology of Newly Emerging Marine Diseases. Developments in Hydrobiology, 159. Springer, Doredrecht. https://doi.org/10.1007/978-94-017-3284-02.
Ault, J.S., Smith, S.G., Bohnsack, J.A., Luo, J., Stevens, M.H., Johnson, M.W., Bryan, D. R., DiNardo, G.T., 2019. Length-based risk analysis for assessing sustainability of data-limited tropical reef fisheries. ICES J. Mar. Sci. 76 (1), 165–180.
Ault, J.S., Smith, S.G., Richards, B.L., Yau, A.J., Langseth, B.J., O'Malley, J.M., Boggs, C. H., Seki, M.P., DiNardo, G.T., 2018. Towards fishery-independent biomass estimation for Hawaiian Islands deepwater snappers. Fish. Res. 208, 321–328.
Ault, J.S., Smith, S.G., Browder, J.A., Nuttle, W., Franklin, E.C., Luo, J., DiNardo, G.T., Bohnsack, J.A., 2014. Indicators for assessing the ecological and sustainability dynamics of southern Florida's coral reef and coastal fisheries. Ecol. Indic. 44, 164–172.
Ault, J.S., Smith, S.G., Bohnsack, J.A., Patterson, M., Feeley, M.W., McClellan, D.B., Ruttenberg, B.I., Hallac, D., Ziegler, T., Hunt, J., Kimball, D., Luo, J., Zurcher, N., Causey, B., 2013. Assessing coral reef fish changes and marine reserve dynamics in the Dry Tortugas, Florida USA. Fish. Res. 144, 28–37.
Ault, J.S., Smith, S.G., Tilmant, J.T., 2009. Are the coral reef finfish fisheries of south Florida sustainable? Proc. Int. Coral Reef. Symp. 11, 989–993.
Ault, J.S., Smith, S.G., Luo, J., Monaco, M.E., Appeldoorn, R.S., 2008. Length-based assessment of sustainability benchmarks for coral reef fishes in Puerto Rico. Environ. Conserv. 35 (3), 221–231.
Ault, J.S., Smith, S.G., Bohnsack, J.A., Luo, J., Harper, D.E., McClellan, D.B., 2006. Building sustainable fisheries in Florida's coral reef ecosystem: positive signs in the Dry Tortugas. Bull. Mar. Sci. 78 (3), 633–654.

Ault, J.S., Bohnsack, J.A., Smith, S.G., Luo, J., 2005a. Towards sustainable multispecies fisheries in the Florida USA coral reef ecosystem. Bull. Mar. Sci. 76 (2), 595–622.
Ault, J.S., Smith, S.G., Bohnsack, J.A., 2005b. Evaluation of average length as an estimator of exploitation status for the Florida coral reef fish community. ICES J. Mar. Sci. 62, 417–423.
Ault, J.S., Smith, S.G., Diaz, G.A., Franklin, E.C., 2003. Florida hogfish fishery stock assessment. Rep. Fla. Fish. Wildl. Conserv. Comm. 67 (+ Figures and Tables).
Ault, J.S., Bohnsack, J.A., Meester, G.A., 1998. A retrospective (1979–1996) multispecies assessment of coral reef fish stocks in the Florida Keys. Fish. Bull. 96 (3), 395–414.
Bartholomew, A., Bohnsack, J., Smith, S., Ault, J., Harper, D., McClellan, D., 2008. Influence of marine reserve size and boundary length on the initial response of exploited reef fishes in the Florida Keys National Marine Sanctuary, USA. Landsc. Ecol. 23, 55–65.
Bohnsack, J.A., Ault, J.S., 1996. Management strategies to conserve marine biodiversity. Oceanography 9 (1), 73–82.
Bohnsack, J.A., 1998. Application of marine reserves to reef fisheries management. Austral Ecol. 23 (3), 298–304.
Bohnsack, J.A., Ault, J.S., Causey, B., 2004. Why Have No-take Marine Protected Areas?, 42. American Fisheries Society Symposium, pp. 185–193.
Bohnsack, J.A., Harper, D.E., McClellan, D.B., 1994. Fisheries trends from Monroe County, Florida. Bull. Mar. Sci. 54 (3), 982–1018.
Brandt, M.E., McManus, J.W., 2009. Disease incidence is related to bleaching extent in reef-building corals. Ecology 90 (10), 2859–2867. https://doi.org/10.1890/08-0445.1.
Brown, C.J., Taylor, W., Wabnitz, C.C.C., Connolly, R.M., 2020. Dependency of Queensland and the Great Barrier Reef's tropical fisheries on reef-associated fish. Sci. Rep. 10, 17801.
Cadima, E.L., 2003. Fish stock assessment manual. FAO Fisheries Technical Paper 393. Rome, FAO. 161 p.
Canales, C.M., Punt, A.E., Mardones, M., 2021. Can a length-based pseudo-cohort analysis (LBPA) using multiple catch length frequencies provide insight into population status is data-poor situations? Fish. Res. 234 (2021), 105810.
Carpenter, K.E., Abrar, M., Aeby, G., Aronson, R.B., Banks, S., et al., 2008. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science 321 (5888), 560–563.
Chong, L., Mildenberger, T.K., Rudd, M.B., Taylor, M.H., Cope, J.M., Branch, T.A., Wolff, M., Moritz-Stabler, M., 2020. Performance evaluation of data-limited, length-based stock assessment methods. ICES J. Mar. Sci. 77 (1), 97–108.
Cinner, J.E., Zamborain-Mason, J., Gurney, G.G., Graham, N.A.J., MacNeil, M.A., et al., 2020. Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world. Science 368, 307–311.
Cinner, J.E., McClanahan, T.R., 2006. Socioeconomic factors that lead to overfishing in small-scale coral reef fisheries of Papua New Guinea. Environ. Conserv. 33 (1), 73–80.
Clark, W.G., 2002. $F_{35\%}$ revisited ten years later. N. Am. J. Fish. Manag. 22, 251–257.
Clarke, J., Bailey, D.M., Wright, P.J., 2015. Evaluating the effectiveness of a season spawning area by closure. ICES J. Mar. Sci. 72 (9), 2627–2637.
Coker, D.J., Wilson, S.K., Pratchett, M.S., 2014. Importance of live coral habitat for reef fishes. Rev. Fish Biol. Fish. 24, 89–126.
Cope, J.M., Punt, A.E., 2009. Length-based reference points for data-limited situations: applications and restrictions. Mar. Coast. Fish.: Dyn. Manag. Ecosyst. Sci. 1, 169–186.
Dixon, R.L., Huntsman, G.R., 1992. Estimating catches and fishing effort of the southeast United States headboat fleet, 1972–1982. U. S. Dept. Commerce, NMFS Technical Report, 23 p. and appendices.
Eakin, M., Sweatman, H., Brainard, R., 2019. The 2014–2017 global-scale coral bleaching event: insights and impacts. Coral Reefs 38 (4), 539–545. https://doi.org/10.1007/s00338-019-01844-2.
Ehrhardt, N.M., Ault, J.S., 1992. Analysis of two length-based mortality models applied to bounded catch length frequencies. Trans. Am. Fish. Soc. 121 (1), 115–122.
Erisman, B.E., Allen, L.G., Claisse, J.T., Pondella, D.J., Miller, E.F., Murray, J.H., 2011. The illusion of plenty: hyperstability masks collapses in two recreational fisheries that target fish spawning aggregations. Can. J. Fish. Aquat. Sci. 68 (1), 1705–1716.
Escalle, L., Gaertner, D., Chavance, P., Delgado de Molina, A., Ariz, J., Merigot, B., 2016. Consequences of fishing moratoria on catch and bycatch: the case of tropical tuna purse-seiners and whale and whale shark associated sets. Biodivers. Conserv. 25 (9), 1637–1659.
Friedman, S.K., 1988. Forecasting a seasonal population. Bus. Econ. 23 (3), 48.52.
Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A., Dimarchopoulou, D., Scarcella, G., Probst, W.N., Dureuil, M., Pauly, D., 2018. A new approach for estimating stock status from length frequency data. ICES J. Mar. Sci. 75 (6), 2004–2015.
Gallucci, V.F., Saila, S.B., Gustafson, D.J., Rothschild, B.J. (Eds.), 1996. Stock Assessment: Quantitative Methods and Applications for Small-scale Fisheries. CRC Press, Lewis Publishers, New York, p. 527.
Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol. Evol. 18 (9), 448–455.
Gough, C.L.A., Dewar, K.M., Godley, B.J., Zafindranosy, E., Broderick, A.C., 2020. Evidence of overfishing in small-scale fisheries in Madagascar. Front. Mar. Sci. 7, 317. https://doi.org/10.3389/fmars.2020.00317.
Happel, S.K., Hogan, T.D., 2002. Counting snowbirds: the importance of and the problems with estimating seasonal populations. Popul. Res. Policy Rev. 21 (3), 227–240.
Hardt, M.J., 2009. Lessons from the past: the collapse of Jamaican coral reefs. Fish Fish. 10 (2), 143–158.
Hastings, A., Botsford, L.W., 1999. Equivalence in yield from marine reserves and traditional fisheries management. Science 284, S65–S70.
Hoegh-Guldberg, O., 1999. Climate change, coral bleaching and the future of the world's coral reefs. Mar. Freshw. Res. 50 (8), 839–866.
Hoegh-Guldberg, O., Jacob, D., Taylor, M., Guillen Bolanos, T., Bindi, et al., 2019. The human imperative of stabilizing global climate change at 1.5 °C. Science 365, eaaw6974. https://doi.org/10.1126/science.aaw6974.
Hordyk, A., Ono, K., Sainsbury, K., Loneragan, N., Prince, J., 2015. Some explorations of the life history ratios to describe length composition, spawning-per-recruit, and the spawning potential ratio. ICES J. Mar. Sci. 72, 204–216.
Hughes, T.P., Kerry, J.T., Baird, A.H., et al., 2018. Global warming transforms coral reef assemblages. Nature 556, 492–496.
Hunter, C.L., Evans, C.W., 1995. Coral reefs in Kaneohe Bay, Hawaii: two centuries of western influence and two decades of data. Bull. Mar. Sci. 57 (2), 501–515.
ISC, 2018. Summary report of the Pacific bluefin tuna management strategy evaluation workshop. International Scientific Committee for Tuna and Tuna-like Species in the North Pacific Ocean. Yeosu, Korea. ISC/18/ANNEX/08.
Koenig, C.C., Coleman, F.C., Kingon, K.C., 2011. Pattern of recovery of the goliath grouper (*Epinephelus itajara*) population in the southeastern U.S. Bull. Mar. Sci. 87 (4), 891–911.
Kvamsdal, S.F., Eide, A., Ekerhovd, N.-A., Enberg, K., Gudmundsdottir, A., Hoel, A.H., Mills, K.E., Mueter, F.J., Ravn-Jonsen, L., Sandal, L.K., Stiansen, J.E., Vestergaard, N., 2016. Elem.: Sci. Anthr. https://doi.org/10.12952/journal.elementa.000114.
Le Pape, O., Bonhommeau, S., Nieblas, A.-E., Fromentin, J.-M., 2017. Overfishing causes frequent fish population collapses but rare extinctions. Proc. Natl. Acad. Sci. USA 114 (31), E6274.
Lyon, J.P., Tonkin, Z., Moloney, P.D., Todd, C., Nicol, S., 2018. Conservation implications of angler misidentification of an endangered fish. Aquat. Conserv. 28 (6), 1396–1402.
Maunder, M.N., Xu, H., Lennert-Cody, C., Valero, J.L., Aires-da-Silva, A., Minte-Vera, C., 2020. Implementing reference point-based fishery harvest control rules within a probabilistic framework that considers multiple hypothesis. In: Proceedings of the Inter-American Tropical Tuna Commission Scientific Advisory Committee 11th Meeting. Document SAC-11 INF-F Rev. 54 p.
Maunder, M.N., Piner, K.R., 2014. Contemporary fisheries stock assessment: many issues still remain. ICES J. Mar. Sci. 72 (1), 7–18.
McClanahan, T.R., Hicks, C.C., Darling, E.S., 2008. Malthusian overfishing and efforts to overcome it on Kenyan coral reefs. Ecol. Appl. 18 (6), 1516–1529.
Meester, G.A., Ault, J.S., Smith, S.G., Mehrotra, A., 2001. An integrated simulation modeling and operations research approach to spatial management decision making. Sarsia 86 (6), 543–558.
Meester, G.A., Mehrotra, A., Ault, J.S., Baker, E.K., 2004. Designing marine reserves for fishery management. Manag. Sci. 50 (8), 1031–1043.
Methot, R.D., Tromble, G.R., Lambert, D.M., Greene, K.E., 2014. Implementing a science-based system for preventing overfishing and guiding sustainable fisheries in the United States. ICES J. Mar. Sci. 71 (2), 183–194.
Mumby, P.J., Harborne, A.R., 2010. Marine reserves enhance the recovery of corals on Caribbean Reefs. PLoS One 5 (1), e8657.
Nadon, M.O., Ault, J.S., 2016. A stepwise stochastic simulation approach to estimate life history parameters for data poor fisheries. Can. J. Fish. Aquat. Sci. 73, 1–11.
Nadon, M.O., Ault, J.S., Williams, I.W., Smith, S.G., DiNardo, G.T., 2015. Length-based assessment of coral reef fish populations in the Main and Northwestern Hawaiian Islands. PLoS One 10 (8), e0133960. https://doi.org/10.1371/journal.pone.0133960.
NMFS, 2017. Marine Recreational Fisheries Program – Collaborating to Improve Recreational Fishing Estimates: Strategic Plan 2017–2022. U.S. Department of Commerce, NOAA National Marine Fisheries Service.
Pauly, D., Christiansen, V., Dalsgaard, J., Froese, R., Torres Jr., F., 1998. Fishing down marine food webs. Science 279, 860–863.
Pauly, D., Froese, R., 2021. MSY needs no epitaph – but it was abused. ICES J. Mar. Sci. 78, 2204–2210.
Prager, M.H., Porch, C.E., Shertzer, K.W., Caddy, J.F., 2003. Targets and limits for management of fisheries: a simple probability-based approach. North Am. J. Fish. Manag. 23, 349–361.
Precht, W.F., Aronson, R.B., Gardner, T.A., Gill, J.A., Hawkins, J.P., Hernández-Delgado, E.A., et al., 2020. The timing and causality of ecological shifts on Caribbean reefs. Adv. Mar. Biol. https://doi.org/10.1016/bs.amb.2020.08.008.
Punt, A.E., Smith, A.D.M., Smith, D.C., Tuck, G.N., Klaer, N., 2014. Selecting relative abundance proxies for B_{MSY} and B_{MSY}. ICES J. Mar. Sci. 71, 469–483.
Roberts, C.M., Bohnsack, J.A., Gell, F., Hawkins, J.P., Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. Science 294 (5548), 1920.
Robinson, J.P.W., Robinson, J., Gerry, C., Govinden, R., Freshwater, C., Graham, N.A.F., 2020. Diversification insulates fisher catch and revenue in heavily exploited tropical fisheries. Sci. Adv. 6 (8), eaaz0587.
Rudd, M.B., Thorson, J.T., 2018. Accounting for variable recruitment and fishing mortality in length-based stock assessments for data-limited fisheries. Can. J. Fish. Aquat. Sci. 75, 1019–1035.
Sala, E., Mayorga, J., Bradley, D., Cabrat, R.B., Atwood, T.B., Auber, A., Cheung, W., Costello, C., Ferretti, F., Friedlander, A.M., Gaines, S.D., Garilao, C., Godell, W., Halpern, B.S., Hinson, A., Kaschner, K., Kesner-Reyes, K., Leprieur, F., McGowan, J., Morgan, L.E., Mouillot, D., Palacios-Abrantes, J., Possigham, H.P., Rechberger, K.D., Worm, B., Lubchenco, J., 2021. Protecting the global ocean for biodiversity, food and climate. Nature. https://doi.org/10.1038/s41586-021-03371-z.

Sala, E., Giakoumi, S., 2018. No-take marine reserves are the most effective protected areas in the ocean. ICES J. Mar. Sci. 75 (3), 1166–1168. https://doi.org/10.1093/icesjms/fsx059.

Sala, E., Ballesteros, E., Starr, R.M., 2011. Rapid decline of Nassau grouper spawning aggregations in Belize: fishery management and conservation needs. Fisheries 26 (10), 23–30.

SEDAR 64, 2020. Southeastern US Yellowtail Snapper Final Stock Assessment Report. SEDAR (Southeast Data, Assessment and Review). North Charleston, SC. 386 p.

Shertzer, K.W., Prager, M.H., Williams, E.K., 2008. A probability-based approach to setting annual catch levels. Fish. Bull. 106 (3), 215–232.

Silbiger, N.J., Nelson, C.E., Remple, K., Sevilla, J.K., Quinlan, Z.A., Putnam, H.M., Fox, M.D., Donahue, M.J., 2018. Nutrient pollution disrupts key ecosystem function on coral reefs. Proc. R. Soc. B 285, 20172718. https://doi.org/10.1098/rspb.2017.2718.

Smith, S.G., Ault, J.S., Bohnsack, J.A., Harper, D.E., Luo, J., McClellan, D.B., 2011. Multispecies survey design for assessing reef-fish stocks, spatially-explicit management performance, and ecosystem condition. Fish. Res. 109 (1), 25–41.

Smith, S.K., House, M., 2006. Snowbirds, sunbirds, and stayers: seasonal migration of elderly adults in Florida. J. Gerontol. Ser. B 61 (5), S232–S239.

Stevens, M.H., Smith, S.G., Ault, J.S., 2019. Life history demographic parameter synthesis for Florida and Caribbean reef fishes. Fish Fish. 20, 1196–1217.

Strona, G., Beck, P.S.A., Cabeza, M., Fattorini, S., Guilhaumon, F., Micheli, F., Montano, S., Ovaskainen, O., Planes, S., Veech, J.A., Parravicini, 2021. Ecological dependencies make remote reef fish communities most vulnerable to coral loss. Nat. Commun. 12, 7282.

Sullivan, P.J., Breidt, F.J., Ditton, R.B., Knuth, B.A., Leaman, B., O'Connell, V., Parsons, G.R., Pollock, K.H., Smith, S.J., Stokes, S.L., 2006. Review of Recreational Fisheries Survey Methods. National Research Council. Ocean Studies Board. National Academies Press, Washington, DC.

Szmant, A.M., 2002. Nutrient enrichment on coral reefs. Is it a major cause of coral decline. Estuaries 25 (4b), 743–766.

Then, A.M., Hoenig, J.M., Gedamke, T., Ault, J.S., 2015. Comparison of two length-based estimators of total mortality: a simulation approach. Trans. Am. Fish. Soc. 144, 1206–1219.

Valentine, J.F., Heck, K.L., 2005. Perspective review of the impacts of overfishing on coral reef food web linkages. Coral Reefs 24, 209–213.

Woodhead, A.J., Hicks, C.C., Norstrom, A.V., Williams, G.J., Graham, N.A.J., 2019. Coral reef ecosystem services in the Anthropocene. Funct. Ecol. 33, 1023–1034.