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SEDAR101-RD-01

March 2025

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Canadian Journal of
Fisheries and
Aquatic Sciences

Article

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Abstract

Coastal sharks are challenging to manage in the United States due to their slow life history, limited data availability, history of overexploitation, and competing stakeholder interests. Furthermore, species like the sandbar shark (*Carcharhinus plumbeus*) are subjected to international exploitation unmanaged by the US. We conducted a management strategy evaluation using Stock Synthesis on the sandbar shark to test the performance of various configurations of a threshold harvest control rule. In addition to uncertainties addressed in the operating model (OM), we built multiple implementation models to address uncertainties related to future levels of a partially unmanaged source of removals, the combined Mexican and US recreational (MexRec) fleet. We found that the presence of unregulated removals had the potential to significantly influence the success of the various management procedures (MPs) tested. Notably, if MexRec catches continue to increase with total stock abundance following historical trends, the rate of MexRec removals will be too large to allow the sandbar shark to recover across OMs. We present trade-offs between performance metrics across a range of 24 MPs and three implementation models.

Résumé

La gestion des requins côtiers aux États-Unis présente des défis en raison de la lenteur des cycles biologiques de ces poissons, la disponibilité limitée de données, un historique de surexploitation et les intérêts opposés de différentes parties prenantes. En outre, des espèces comme le requin gris (*Carcharhinus plumbeus*) font l'objet d'une exploitation internationale non gérée par les États-Unis. Nous avons réalisé une évaluation des stratégies de gestion en utilisant l'outil Stock Synthesis pour le requin gris afin de vérifier la performance de différentes configurations d'une règle de contrôle des prises seuils. Outre les sources d'incertitude intégrées au modèle opératoire (MO), nous avons constitué différents modèles d'application pour tenir compte de sources d'incertitude associées à l'intensité future d'une source de retrait partiellement non gérée, la flotte de pêche sportive combinée mexico-américaine (MexRec). Nous avons constaté que la présence de retraits non réglementés pourrait influencer significativement l'efficacité des différentes procédures de gestion (PG) testées. Notamment, si les prises de la MexRex continuent de croître alors que l'abondance totale du stock suit les tendances historiques, les différents MOs indiquent que le taux de retrait par la flotte MexRec sera trop important pour permettre le rétablissement du requin gris. Nous présentons les compromis entre différentes mesures de performance pour un éventail de 24 PG et trois modèles d'application. [Traduit par la Rédaction]

Introduction

Selected fishes that cross international boundaries are designated "highly migratory species" (HMS) by the US. These HMS are not as strictly bound to the Magnuson–Stevens Fishery Conservation and Management Act (MSA), which governs fishery management in the US (MSA 2007), to allow room for international collaboration and agreements. Management of international fisheries is particularly challenging, because several nations with conflicting management goals often need to collaborate to achieve their objectives or operate competitively as independent governing bodies (Munro 2009). Accordingly, the influence of external, unmanaged removals has rarely been explicitly considered on the efficacy of fisheries management (e.g., Van Beveren et al. 2020).

Domestic coastal sharks within the US Atlantic are currently managed under the National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Atlantic Highly Migratory Species Fishery Management Plan. Accordingly, many Atlantic coastal shark distributions span multiple countries and are consequently subjected to harvest by non-US countries. To date, no management procedure (MP) has been formally proposed or utilized for these sharks within the US (NMFS 2019).

Managing fisheries according to MPs is gaining traction worldwide (Punt et al. 2016; ICES 2019), as MP-based management is consistent with the Food and Agricultural Organization's (United Nations) precautionary approach (FAO 1996). MPs include a prespecified rule for adjusting management measures based on the status of a stock, commonly termed a harvest control rule (HCR; Restrepo et al. 1998; NMFS 2016). By conservatively reducing catch limits, MPs can account for scientific and management uncertainty and reduce the risk to the resource (MSA 2007; NMFS 2019). Accordingly, development of MPs is also increasing in the US (DeVore and Gilden 2019).

Management strategy evaluation (MSE) is the approach by which the performance of alternative MPs are evaluated through closed-loop simulation (Holland 2010; Punt 2010). The combination of an HCR, fishery-specific data-generating procedure, estimating model (EM; e.g., assessment model), and implementation procedure defines an MP. In addition to development of candidate MPs, MSE involves specification of management objectives, identification of major uncertainties within the fishery, development and conditioning of multiple operating models (OMs), and presentation of the tradeoffs among management objectives obtained from simulating the fishery under the various candidate MPs (Sainsbury et al. 2000; A'mar et al. 2006; Punt et al. 2016). An MSE is distinguished from a traditional risk analysis through the feedback loop that regularly applies the MP-derived catch back to the fishery in each time step (generally with associated implementation and management uncertainty). Including stakeholder input to clarify management objectives and foster buy-in to the management process is considered best practice within MSE (Punt et al. 2016; Goethel et al. 2019), though many pertinent questions can be investigated with MSEs with no direct stakeholder input. Consequently, the overwhelming majority of MSE simulations have been desk MSEs, defined as MSEs that do not directly include stakeholders (Punt et al. 2005; A'mar et al. 2006; Carruthers et al. 2016).

Coastal sharks are generally considered data-limited (Stevens 2000; Ellis et al. 2008; Cortés et al. 2015), highly susceptible to overexploitation (Musick et al. 2000; Stevens 2000), and challenging to assess and manage (Cortés 2011; Cortés et al. 2015). Specifically, sharks comprise intrinsically slow-growing populations (Musick et al. 2000; Stevens 2000; Cortés 2011) and undergo complex, sex-specific, and ontogenetically varying habitat use and migratory patterns (McCandless et al. 2007; Ellis et al. 2008; Grubbs 2010). Low economic fishery value has resulted in lower research prioritization of sharks (Stevens 2000; Ellis et al. 2008; Pilling et al. 2008), such that fundamental understanding of shark life history is still lacking for many species (Stevens 2000; Cortés et al. 2015). Particular areas of uncertainty for coastal sharks include estimates of natural mortality (Ellis et al. 2008; Cortés 2011), accurate age-determination protocols (Natanson et al. 2018; Natanson and Deacy 2019), and a generally understudied stock–recruitment (S–R) relationship (Taylor et al. 2013; Kai and Yokoi 2017). Furthermore, restricted spatiotemporal survey data (Grubbs 2010), unreliable stock structure and identification, uncertainty in the amount of unreported catch, poorly resolved discard statistics, and unknown postrelease mortality (Cortés 2011) pose challenges to assessment scientists. These data limitations coupled with the history of documented shark population declines due to unregulated overexploitation (e.g., Musick et al. 1993) have resulted in repeated calls for conservative and precautionary management measures (e.g., Musick et al. 2000; Dulvy et al. 2014).

Beyond challenges associated with assessing coastal shark stocks (Cortés 2011), the management of coastal sharks is itself contentious within the coastal and fishing communities (Carlson et al. 2019). Expected management objectives of coastal sharks strongly oppose one another, a problem exacerbated by the number of conflicting stakeholders and strong attitudes toward sharks (Castro 2016). In addition to fearful opinions of sharks and concern about their interactions with other threatened species (Carlson et al. 2019), fishers have overwhelmingly reported an overabundance of sharks and corresponding depredation, which directly impacts their catch and livelihood (Mitchell et al. 2018; Carlson et al. 2019; Tixier et al. 2020). These perspectives contrast with those of conservationists (Simpfendorfer et al. 2011; Castro 2016) and individuals within the shark tourism industry (Gallagher and Hammerschlag 2011; Cisneros-Montemayor et al. 2013). Commercial and recreational coastal shark fishers' goals may differ still (Punt et al. 2016; Gallagher et al. 2017) and contrast with the federal management guidelines (MSA 2007).

The purpose of this study is to examine potential management strategies for application to a large coastal shark species, the sandbar shark (*Carcharhinus plumbeus*). The southeastern US sandbar shark stock is harvested by both the US and Mexico. Using a desk MSE, we examined how various parameterizations of a US-based threshold HCR perform for the sandbar shark across uncertainties, including natural mortality, steepness, initial population size, form of the S–R relationship, and the level of future Mexican and US recreational (MexRec) harvest. Because the US cannot regulate Mexican catches, the future rates of Mexican harvest are a uniquely key uncertainty in this system. We are interested in understanding (1) how an MP would perform for coastal sharks more broadly and (2) how unmanaged, international removals would impact the expected performance of an MP. Accordingly, we developed three MSE implementation scenarios: (1) one to test the Conceptual MP performance, assuming all catches were controlled by the HCR, and two to test MP performance subject to unregulated (by the HCR) (2) high and (3) low Mexican removals. Performance metrics used to assess HCR performance reflected anticipated desires of shark-directed and nonshark-directed commercial and recreational fishers, conservationists, and ecotourism industries, as well as the limitations outlined by the MSA and subsequent reauthorizations (MSA 2007). This MSE is a first for the domestically managed Atlantic coastal sharks and has broad application to any stocks with an uncontrolled (by the MP) component to the catch.

Methods

Sandbar shark

Stock, fishery, and management

The focus of this study is sandbar shark management in the US. The sandbar shark is known to have a low intrinsic population growth rate, with a median age at maturity of 13 years (Baremore and Hale 2012), a reproductive cycle of 2 or 3 years (considered 2.5 years; Baremore and Hale 2012; SEDAR 2017), a maximum age of 31 years (SEDAR 2017), and comprises a single stock within the southeastern US and Gulf of Mexico (Heist et al. 1995). Sandbar sharks are preferred within the coastal shark fishery due to their larger sizes, proportionally large fins, and close proximity to land (Dulvy et al. 2014). Following an unmanaged expansion of the fishery in the 1980s, the southeastern US sandbar shark stock declined rapidly to overfished levels into the early 1990s. As a result of federal management implementations initiated throughout the mid-1990s, the stock has since begun to recover into the 2010s (Peterson et al. 2017; SEDAR 2017). Retention of sandbar sharks is prohibited in commercial and recreational fisheries, though a small research fishery is maintained. Currently, the sandbar shark is below its biomass threshold (i.e., overfished) and its current fishing mortality rate is less than the maximum threshold (i.e., is not experiencing overfishing; SEDAR 2017). However, uncertainty in stock status is high, as various sensitivity scenarios in the most recent stock assessment produced different depictions of stock status (SEDAR 2017).

The most recent stock assessment partitioned catch according to four fishing fleets: (1) the US commercial fleet in the Gulf of Mexico, (2) the US commercial fleet in the Atlantic Ocean, (3) the US recreational catches combined with landings from the Mexican fishery (MexRec fleet), and (4) dead discards attributed to the Gulf of Mexico menhaden purse seine fishery (SEDAR 2017). Catches are generally considered particularly uncertain for coastal sharks, largely because they were rarely identified to species level in the US historical time period and have never been reported by species in Mexico, the prohibitively high uncertainty around US recreational removal estimates, and the fact that all catch series were reconstructed prior to 1981. The MexRec fleet were initially combined because they were believed to have the same selectivity (E. Cortés, personal observation). Due to the consequent challenges associated with adequately separating the MexRec fleet, we relied on the peer-reviewed (Cortés et al. 2002; SEDAR 2006, 2011, 2017) combined fleet as representative of the best available information for the current analyses.

There is no HCR in place for coastal sharks in the US (NMFS 2019). Because the sandbar shark is currently overfished, a rebuilding plan is in place. A quota is recommended by defining the level of exploitation that would ensure the stock is not overfished with 70% probability by the end of the projection period. Annual commercial catch limits are then specified by first subtracting anticipated recreational catch and bycatch mortality (58 t for sandbar shark, which does not include Mexican catches) and then correcting for past over- or under-harvest (SEDAR 2017).

MSE protocol

An MSE was developed for the sandbar shark in the southeastern US using R (version 3.6.3; R Core Team 2020) and Stock Synthesis (version 3.30.15; Methot and Wetzel 2013). Stock Synthesis is a packaged tool for applying integrated, statistical catch-at-age assessments (Methot and Wetzel 2013) and has proven useful in MSE applications (Maunder 2014; Hicks et al. 2016; ISC 2019; Doering and Vaughan 2020; Sharma et al. 2020). We relied extensively on the R package "r4ss" (Taylor et al. 2021) for communication between R and Stock Synthesis and followed Maunder (2014) for using Stock Synthesis as the operational framework for an MSE (see Supplementary material for detailed protocol; R code and example Stock Synthesis control input files available at https://github.com/cassidydpeterson/SS_MSE).

Operating model (OM)

Stock synthesis OM development

The base OM was modified from the most recent Stock Synthesis assessment (SEDAR 2017) to include two sexes, four fishing fleets, two indices of abundance, and a low-fecundity stock–recruit (LFSR) relationship (Taylor et al. 2013; Figs. S1–S5). Though the recent assessment model contains 11 indices of abundance (SEDAR 2011), we only included two indices in the current simulation to reduce computing time and model complexity. The two indices included in the OM were chosen based on temporal and spatial coverage, selectivity, fit in the assessment model, and because the corresponding assessment results were very close to those of SEDAR (2017).

The Stock Synthesis model was then altered to reflect each OM scenario (Table 1) and conditioned on the available assessment data to ensure that each OM was consistent with the biology and exploitation history of the sandbar shark (e.g., Figs. S1–S5). Within the conditioning step, each OM was fitted following the most recent assessment model structure, apart from the requisite alteration for unique OM scenarios (e.g., the same life history parameters were fixed, etc.; see Supplementary materials for more details on OM specification). Note that the OM conditioning was part of the OM model development and did not occur within the simulation loop.

OM process error—parameter-generating process

Following OM conditioning, process error in the OM was generated using ADMB's Markov–chain Monte Carlo (MCMC) protocol (Monnahan et al. 2014) to generate alternative iterations or states of nature across which MP performance would be tested. The timeframe of the OM was then extended to include the full-simulation time horizon or projection period (years 2016–2115). MCMC was run across future years to generate recruitment and parameter deviations for the entire duration of the simulation. Additional complexity was built into the OM compared with the EM, inherently assuming that, in practice, the assessment model was simpler than the true underlying dynamics of the population. Process error was

Table 1. List of six operating models (OMs) with associated levels of relevant parameters.

	OM_Base	OM_BH	OM_Hih	OM_Loh	OM_lnR_0	OM_*M*_BH
M	*Current*	Current	Current	Current	Current	$\frac{1}{2}$ Current
h	*0.3*	0.3	0.4	0.25	0.3	0.3
R_0	*Current*	Current	Current	Current	2 × Current	Current
S–R	*LFSR*	BH	LFSR	LFSR	LFSR	BH
MSY	*531*	375	691	367	992	300
F_{MSY}	*0.1002*	0.0694	0.1230	0.0662	0.0967	0.0739
F_{MSY}/M	*0.802*	0.555	0.984	0.530	0.774	1.177
B_{MSY}	*642*	580	545	722	1292	1489
Year of recovery if $F = 0$	*2071*	2054	2042	>2115	2022	2024

Note: The base OM column (OM_BASE) is italicized. *M* is natural mortality, *h* is steepness, R_0 is the natural logarithm of virgin recruitment, and S–R is the form of the stock–recruitment relationship. The OM with $\frac{1}{2}$ Current produced a nonsensical yield–biomass curve when low-fecundity stock–recruit (LFSR) was specified; consequently, we chose to apply the Beveron–Holt (BH) S–R function to this OM scenario. Average *M* of ages 1+ is 0.125 for all OMs except OM_*M*_BH (where $M = 0.0627$). OMs are named after the parameter that was altered from the base OM, including the BH S–R relationship (OM_BH), high or low steepness levels (OM_Hih, OM_Loh), the magnitude of virgin recruitment (OM_lnR_0), and the natural mortality (OM_*M*_BH). "Current" denotes that the model assumed the estimated value from the most recent stock assessment, where current virgin recruitment = exp(6.27) and current age-specific $M = 0.160419$ for ages 0–5, 0.157755 for age 6, and 0.116805 for ages > 6 (SEDAR 2017). MSY is the maximum sustainable yield.

induced through time-varying recruitment deviations, selectivity, and catchability (*q*; Wilberg et al. 2010). Time-varying selectivity and catchability parameters were implemented through zero-reverting random walks (Methot et al. 2020) to ensure they would not stray into unrealistic values (Wilberg et al. 2010). Non-time-varying error was included in von Bertalanffy length-at-age, allometric weight-at-length, and S–R parameters (except steepness within the Beverton–Holt (BH) S–R relationship OMs). Recruitment autocorrelation was fixed in the OMs at the value estimated in the conditioning step.

To assist in the MCMC process (including reducing computing time and improving convergence), priors were placed on almost all estimated parameters (excluding the natural logarithm of virgin recruitment; Monnahan et al. 2019). We ensured priors were informative, particularly for parameters for which there were little data to inform parameter estimates (e.g., selectivity). Prior means were defined as the values estimated through conditioning each OM, and prior standard deviations were generally restricted to be an order of magnitude less than the respective prior mean. By necessarily constraining some priors, we ensured that future projections were viable.

OM observation error—data-generating process

Observation uncertainty, or uncertainty induced within the data-generating step, was included in historical observed catches, future catches, relative abundance indices, and length-composition observations. Data were generated using Stock Synthesis's parametric bootstrapping protocol. New data sets with variance properties consistent with the original data were created by calculating expected values for input data and then adding random samples from the probability distribution of the expected value for each input data type (Methot and Wetzel 2013; Methot et al. 2020). The OM assumed lognormal error in catch and abundance index observations and multinomial error in length compositions.

For each future year, we specified (1) catch as obtained from the HCR and implementation models, (2) standard error of catch, (3) effective sample size of length frequency observations, and (4) abundance index standard error. The bootstrap process subsequently constructed indices, length compositions, and applied observation error to commercial catches. These bootstrapped data were then used as observed data in the EM for the corresponding year. Within the simulation, future years of the OM were populated with expected values and bootstrapped values with observation uncertainty were input into the EM.

OM uncertainties

By configuring simulations to reflect various hypotheses about the structure and productivity of the underlying stock, it was possible to account for the plausible range of uncertainties in the population dynamics and assess the robustness of each MP to uncertainties in the system. Uncertainties explored included alternate levels of natural mortality, steepness, and overall magnitude of the resource, in addition to the form of the S–R relationship (Table 1). Multiple OMs were constructed to reflect each alternate level of the respective uncertainty. Given the computational demands of a full factorial design of each level of uncertainty, a "base" level of all parameters was chosen and each parameter was then allowed to vary in turn (Punt et al. 2016; Table 1).

Because the sandbar shark is exploited by both the US and Mexico, any MP employed by the US will not alter Mexican removals. The level of future Mexican removals consequently represents a major uncertainty in the system. As such, the magnitude of future MexRec removals was treated as an additional level of uncertainty realized through multiple implementation models.

Estimation model

The population was assessed by inputting the bootstrap-generated data into the EM, which was configured to replicate the stock assessment model used in practice to assess the sandbar shark (derived from SEDAR 2017). Where fea-

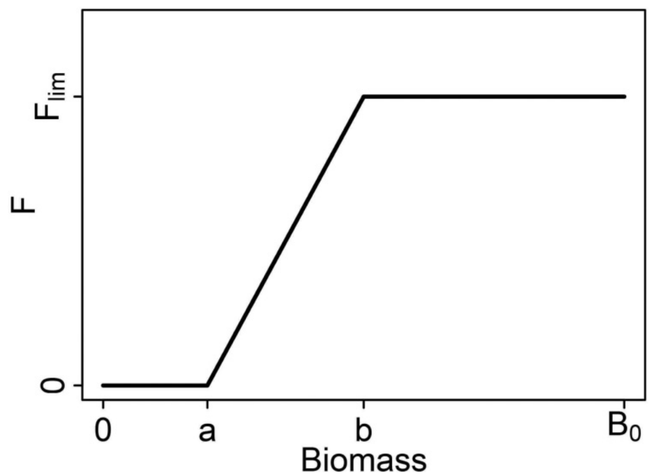


Fig. 1. Form of the threshold harvest control rule (HCR) examined in the current study, where F_{lim} is the maximum prescribed fishing mortality rate (F), a is the threshold biomass below which prescribed $F = 0$, and b is the threshold biomass below which prescribed F is reduced.

sible, the observations, available information, estimated parameters, and assumptions were kept consistent with those associated with the stock assessment model fitted in practice (SEDAR 2017). In the EM, selectivity and catchability were assumed to be time-invariant. Biological parameters were fixed and the stock was assumed to follow a BH S–R relationship. Therefore, the EM assumptions most closely approximate those from the BH_OM (with additional fixed and non-time-varying parameters; see Supplementary materials for additional details on EM specification).

Harvest control rule

The results of the EM were applied to the HCR to estimate a target catch. The HCR was built in R rather than using Stock Synthesis' forecast module. Threshold HCRs, or HCRs that have one or more breakpoints at which the control rule changes (Punt 2010), have generally been shown to be preferable due to precautionary reduction of allowable catch when stock size is low (Deroba and Bence 2008; Punt 2010; Kvamsdal et al. 2016). Consequently, the effects of various parameterizations of a threshold harvest rate HCR based on the following equation were explored:

$$F = \begin{cases} 0, & B < a \\ F_{\mathrm{lim}}\left(\dfrac{B-a}{b-a}\right), & a \le B \le b \\ F_{\mathrm{lim}}, & b < B \end{cases}$$

where F is fishing mortality, B is biomass, F_{lim} is the upper limit F, and a and b are parameters governing the reduction in prescribed F at reduced biomass levels (Fig. 1). A total of 24 unique parameterizations of the HCR were explored, as determined by a factorial expansion of six levels of F_{lim}, two levels of a, and two levels of b (Table 2) and guided by expert opinion

Table 2. Harvest control rule (HCR) parameterizations, where F_{lim} is the maximum prescribed fishing mortality rate (F), a is the threshold biomass below which prescribed $F = 0$, and b is the threshold biomass below which prescribed F is reduced.

	F_{lim}	a	b
HCR1	F_{MSY}	0	B_{MSY}
HCR2	F_{MSY}	0	$0.8B_{MSY}$
HCR3	F_{MSY}	$0.3B_0$	B_{MSY}
HCR4	F_{MSY}	$0.3B_0$	$0.8B_{MSY}$
HCR5	$F = M$	0	B_{MSY}
HCR6	$F = M$	0	$0.8B_{MSY}$
HCR7	$F = M$	$0.3B_0$	B_{MSY}
HCR8	$F = M$	$0.3B_0$	$0.8B_{MSY}$
HCR9	$0.8M$	0	B_{MSY}
HCR10	$0.8M$	0	$0.8B_{MSY}$
HCR11	$0.8M$	$0.3B_0$	B_{MSY}
HCR12	$0.8M$	$0.3B_0$	$0.8B_{MSY}$
HCR13	$0.6M$	0	B_{MSY}
HCR14	$0.6M$	0	$0.8B_{MSY}$
HCR15	$0.6M$	$0.3B_0$	B_{MSY}
HCR16	$0.6M$	$0.3B_0$	$0.8B_{MSY}$
HCR17	$0.4M$	0	B_{MSY}
HCR18	$0.4M$	0	$0.8B_{MSY}$
HCR19	$0.4M$	$0.3B_0$	B_{MSY}
HCR20	$0.4M$	$0.3B_0$	$0.8B_{MSY}$
HCR21	$0.2M$	0	B_{MSY}
HCR22	$0.2M$	0	$0.8B_{MSY}$
HCR23	$0.2M$	$0.3B_0$	B_{MSY}
HCR24	$0.2M$	$0.3B_0$	$0.8B_{MSY}$

Note: 30% of virgin biomass (B_0) was considered as the upper level for a following Clarke and Hoyle (2014) and Sainsbury (2008).

and the primary literature (Sainsbury 2008; Zhou et al. 2012; Clarke and Hoyle 2014; Cortés and Brooks 2018). Note that the HCR provides an F, which was then used to calculate a target catch. F was converted to catch by dividing the average pattern of fishing mortality-at-age ($\boldsymbol{F_a}$) from the years 1995–2015 by fishing mortality ($\boldsymbol{F_{prop}} = \boldsymbol{F_a}/F$). $\boldsymbol{F_{prop}}$ was then multiplied by the HCR-derived F and the vector of biomass-at-age ($\boldsymbol{B_a}$) to generate an estimated catch-at-age vector, which was summed to generate a target catch. $\boldsymbol{F_{prop}}$ served as a mechanism to appropriately include the relative catches of each fleet and their selectivity patterns in the target catch.

Implementation model

Overall implementation uncertainty was added following historical implementation uncertainty between observed catch and specified target catch from the years 2008 to 2019. Historically, observed catches have been biased low compared with specified target catch. Thus, the ratio of future observed catch to target catch was assumed to follow a lognormal distribution, and each year in the MSE projection randomly applied implementation uncertainty following this distribution. Based on observed data, empirical relationships were calculated between effective sample size of length composition data and either fishery catch for fishing fleets or population biomass for fishery-independent indices. Effective sample size for length compositions were projected following these empirically observed relationships (see Supplementary materials for additional information on empirical implementation model relationships).

Catch implementation

Following the stock assessment, catch was separated into four fleets in the OM: (1) Gulf of Mexico US commercial fleet, (2) South Atlantic US commercial fleet, (3) MexRec fleet, and (4) Gulf of Mexico menhaden purse seine fishery dead discards. In practice, catch limits for sandbar shark are set for the US commercial fisheries, not including the MexRec fleet or dead discards. The proportion of commercial catch in the Gulf of Mexico relative to the commercial catch in the Atlantic Ocean from the years 1995 to 2015 was modeled using a beta distribution. We assumed that commercial catch partitioning would follow this distribution into the future, and consequently, a randomly selected proportion of target catch was allocated to the Gulf of Mexico from the modeled distribution. In practice, the menhaden discard fleet is not included in the HCR-designated target catch. We assumed the menhaden discard fleet would continue to be linearly related to biomass following the historical relationship.

Because separation of the MexRec fleet was outside the scope of this study, it was retained as a single fleet in the current analyses. Furthermore, since Mexican catches are not managed by the US, a unique aspect of this MSE was predicting the trajectory of future Mexican removals. To address the uncertainty of future Mexican removals in the MexRec fleet, three implementation model scenarios were developed (two Expected implementation scenarios: HiMexRec and LoMexRec; and one Conceptual implementation scenario) to reflect various hypotheses of future MexRec landings (Fig. 2).

Expected implementation scenarios – The current management process is to designate a target catch, then subtract 58 t to obtain the US commercial catch limit, accounting for anticipated recreational removals and removals due to dead discarding. Accordingly, the expected implementation scenarios in the current study followed this process and the independence of the MexRec and menhaden discard fleets from the HCR-designated target US commercial catch was maintained.

Historically, MexRec removals increased with increasing biomass between 1995 and 2013, though in recent years (2008–2013), catches have remained low. The drivers of MexRec catches are conflated, such that high MexRec removals in the late 1990s may have been driven by high US recreational removals or by high Mexican removals. To book-end plausible Expected MP performance, two implementation models were constructed: (1) one in which MexRec removals will increase with biomass following the linear trend observed between 1995 and 2013 (HiMexRec scenario) and (2) one where MexRec landings remain low and vary around the mean removals observed between 2008 and 2013 (LoMexRec scenario; Fig. 3).

Conceptual implementation scenario – The Conceptual MP scenario examined how the MP would perform if all removals were managed by allowing MexRec catches to be subjected to the HCR, enabling determination of how these MPs would perform for a slow-growing coastal shark species more generally. In the Conceptual implementation model, HCR-designated target catch was not subjected to subtraction of the anticipated US recreational catches as in the Expected MP scenarios. Instead, half of the target catch was allocated toward the MexRec fishery, and the remaining half was split between the Gulf of Mexico and Atlantic Ocean using the beta distribution as described above.

Simulation specifics

In the current simulation, stock assessments occurred every 5 years. The target catch calculated in a given assessment year was applied as a constant catch in each year until the next assessment, with unique implementation uncertainty in each year. The time horizon of the simulation was 100 years, allowing sufficient time for the model to allow the overfished sandbar stock to recover, if possible. Each OM-HCR-implementation model scenario was run for 100 iterations.

This MSE tested the performance of 24 MPs across three future implementation scenarios on six unique OMs (Fig. 2). Only one data-generating model and one EM were created, such that each MP was defined by the data-generating model, the EM, and one of 24 HCRs. All factorial combinations of OM-MP-implementation model were explored in the current study.

Performance metrics

Performance metrics were identified based on best practices (e.g., Punt et al. 2016; Punt 2017), the goals of the current rebuilding plan (as referenced in SEDAR 2017), and a thought exercise wherein relevant stakeholder desires were considered given our understanding of the fishery. In SEDAR (2017), the rebuilding projection target was to rebuild the stock with 70% probability by the end of the 2070 rebuilding period. The performance metrics included the following: probability of stock recovery (where recovery was defined as $B \geq B_{MSY}$, where B is defined as spawning stock biomass and the subscript MSY indicates the corresponding value at maximum sustainable yield), average annual and total catch, mid-term (year = 2070, representing the end of the rebuilding period for sandbar shark) and end-year (year = 2115) estimation of stock status (B/B_{MSY} and F/F_{MSY}) and catch, probability of overfishing throughout the simulation horizon (POF; calculated by summing the number of years in which $F > F_{MSY}$ divided by the 100 years in the simulation horizon), average annual variability in catch (AAV $= \frac{\sum |C_t - C_{t-1}|}{\sum C_t}$, where C is catch at all times t within the simulation horizon), and annual average length within the stock. All performance metrics were calculated

Fig. 2. Description of management strategy evaluation (MSE) dynamics. The current MSE included six operating models (OMs), one data-generating model, one estimating model (EM), 24 harvest control rules (HCRs), and three implementation models. This sums to a total of 72 management procedures (MPs; one data-generating model × one EM × 24 HCRs × three implementation models = 72 MPs) that were applied to each of the six OMs. [Colour online.]

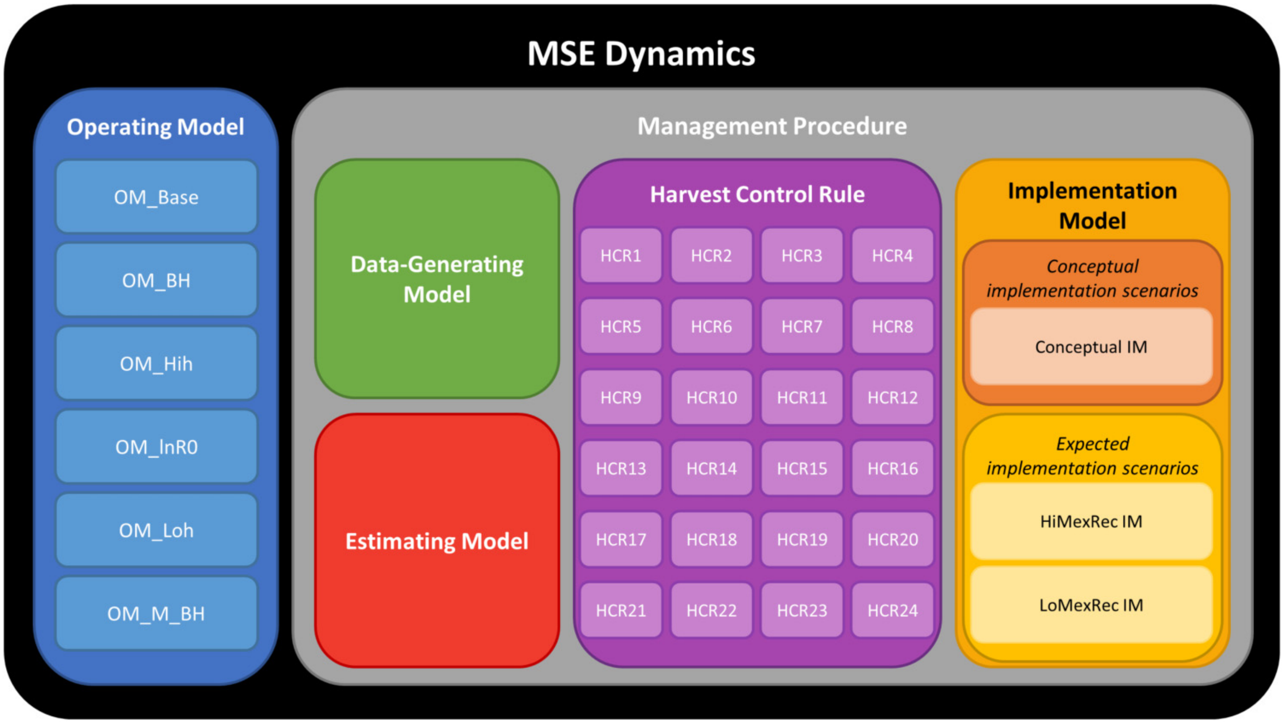
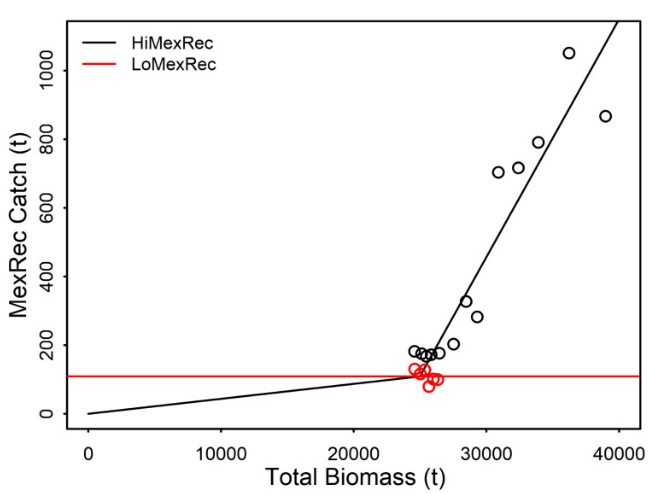


Fig. 3. Historical relationship (1995–2013) of observed Mexican and US Recreational (MexRec) catches and total sandbar shark stock biomass. Points plotted in black represent observations from the years 1995–2007, and red points were observed between the years 2008 and 2013. The superimposed lines demonstrate the alternate simulated relationships between MexRec catches with biomass, where the black line represents the "HiMexRec" implementation scenario while the red line represents the "LoMexRec" implementation scenario. [Colour online.]



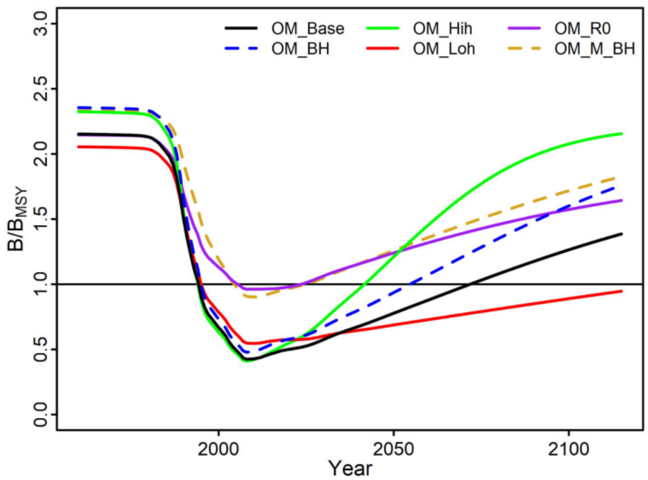
from the OM. Reference points (B_{MSY}, F_{MSY}) were estimated by Stock Synthesis within the OM conditioning for the year 2015. Note that for many nonshark fishers, coastal sharks are deemed a nuisance species (Carlson et al. 2019; C. Peterson, personal observation), as they are known to depredate other fisheries (Mitchell et al. 2018; Tixier et al. 2020). Consequently, we were also conscious of HCRs that resulted in very large biomass levels ($B > 1.5B_{MSY}$). Median performance metrics were presented following Butterworth and Punt's (1999) recommendation for *K*-selected species.

Results

OM parameterizations

In the absence of fishing (Fig. 4), the expected recovery of the stock would occur in the year 2071 in OM_Base, 2054 in OM_BH, 2042 in OM_Hih, sometime after the year 2115 in OM_Loh, 2022 in OM_lnR0, and 2024 in OM_*M*_BH (Table 1; see Supplementary material). The S–R relationship had implications for stock productivity and the shape of the biomass-yield curve (Figs. 4–5). Productivity was greater and MSY occurred at lower biomass levels under the BH S–R assumption compared with the LFSR assumption. A low steepness value of 0.25 was selected within OM_Loh, because a value of 0.2 resulted in a stock that was projected to decline in the absence of all fishing ($F = 0$ for all fleets). A BH S–R relationship

Fig. 4. Expected trajectories of relative spawning stock biomass (B/B_{MSY}) in the absence of fishing mortality in the simulated period (2016–2115) for each OM scenario. [Colour online.]



was assumed for the low natural mortality OM because the assumption of low natural mortality with an LFSR relationship resulted in a stock for which any fishing pressure would result in an overfished stock (i.e., $B_{MSY} \approx B_0$, where B_0 is virgin spawning stock biomass).

MP performance

MP performance varied based on both the implementation model and OM. Overall, the effect of the implementation model had a much greater impact on MP performance than HCR parameterization (Fig. 6). Nevertheless, the goal of an MSE is to develop MPs in the face of plausible uncertainties. Recall that management advice is generated from the EM, which does not necessarily match the simulated stock dynamics generated by the OM. Furthermore, note that results are presented relative to static reference points calculated for the year 2015 during the OM conditioning step, as the OM is not actively being fitted throughout the simulation. Consequently, changes in reference points in the projection period, such as those arising from shifts in fishing allocation, are not reflected in the results. (See Supplementary material for further details on MP performance with respect to individual OMs and implementation scenarios.)

HCR parameterization

MP performance across candidate HCRs reflects trade-offs in management objectives (Fig. 7). Across OMs, average age 1+ instantaneous natural mortality (0.0627 in OM_M_BH and 0.125 in all other OMs) was greater than F_{MSY}, except in the low M_BH scenario (see Table 1 for OM-specific F_{MSY}), suggesting that setting F_{lim} at a rate equal to M is likely too high. The ratio of F_{MSY}/M ranged from 1.177 in OM_M_BH to 0.530 in OM_Loh (Table 1). Consequently, when F_{lim} was high, US commercial catch increased, while terminal spawning stock biomass and probability of stock recovery declined. When F_{lim} was equal to $0.2M$, the resulting B_{2115} was much greater than B_{MSY} in the Conceptual implementation scenario, indicating that $F_{lim} = 0.2M$ was too low in those scenarios. When a was set equal to 30% of B_0, the probability of stock recovery improved over HCRs wherein $a = 0$, but cumulative commercial catch was much lower. When b was equal to 80% of B_{MSY}, the terminal spawning stock biomass and probability of recovery decreased slightly relative to when $b = B_{MSY}$, accompanied by a slight increase in cumulative US commercial catch (Fig. 8).

Effect of implementation scenarios

If MexRec catches increase with increasing stock size following historical exploitation patterns, then the rate of MexRec harvest may be too great to allow the sandbar shark stock to recover (18.0% recovery rate across all OMs and HCRs). In contrast, if MexRec catches remain small following recent years of low removals as the sandbar shark stock abundance increases, then the stock will have a much higher probability of recovery (63.7% recovery rate across all OMs and HCRs) by 2115. MP performance under the LoMexRec implementation scenario was closer to that of the Conceptual implementation scenario (Figs. 6 and 8). Exploration into the Conceptual performance of candidate MPs wherein all catches were controlled by the MP generally showed a more rapid and thorough recovery (63.2% recovery rate by 2115 across all OMs and HCRs) than when there was a source of uncontrolled and unaccounted for removals.

The current practice of subtracting 58 t from the target catch was sufficient to allow for stock recovery in most OMs before 2115. However, this constant deduction from the MSY was based on only US recreational catches and does not include Mexican removals. The average value of the combined MexRec removals between the years 2008 and 2013 was approximately 109 t, likely explaining the longer time-to-recovery compared with the Conceptual MP performance scenario (Fig. S14 vs Fig. S6).

The rebuilding deadline, as prescribed by the MSA, for the sandbar shark is 2070. The probability of stock recovery by the end of the rebuilding period varied by HCR and implementation model (Fig. 8). Averaging across OMs and HCRs for demonstration purposes, the Conceptual, LoMexRec, and HiMexRec scenarios had a 52.9%, 46.8%, and 21.4% probability of stock recovery by 2070, respectively.

Decision table

For the purposes of compiling results and displaying trade-offs, each OM was weighted equally, inherently assuming that the plausibility of each OM was equal (Tables S1–S3 and Fig. 8). When measured across OMs, the probability of recovery by 2070, or even 2115, rarely exceeded 70%. These resulting patterns in probability of recovery indicate that fishing mortality rates would need to be reduced to meet a 70% rebuilding target by the end of the simulated time horizon. The probability of recovery was further impacted by unmanaged removals, which also had the potential to notably reduce probability of recovery and increase POF (Fig. 8).

Fig. 5. Biomass-yield curves for the sandbar shark assessment when assuming an LFSR relationship (left) compared with assuming a BH stock–recruitment (S–R) relationship (right) where MSY is maximum sustainable yield, B_0 is virgin spawning stock biomass, and B_{2115} is spawning stock biomass at the year 2115. [Colour online.]

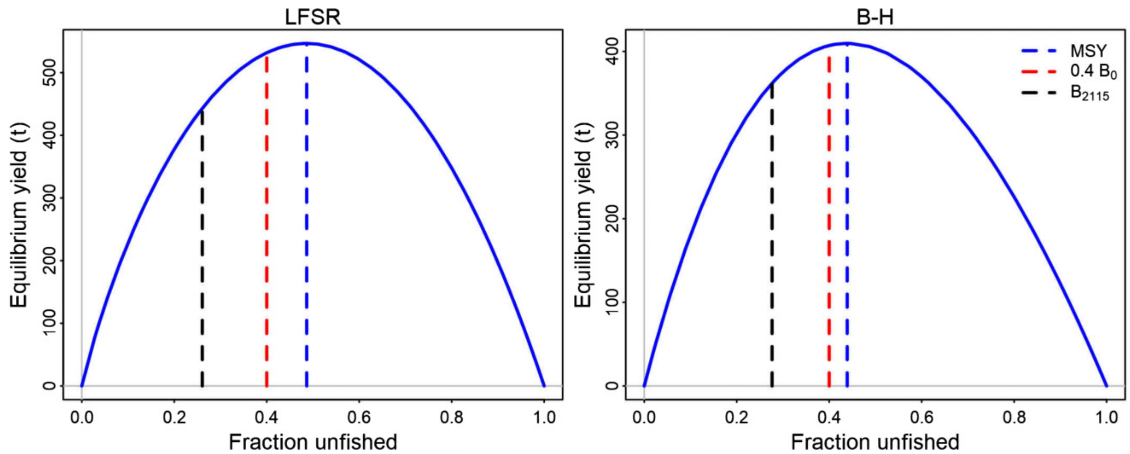


Fig. 6. Worm plots showing OM_Base relative spawning stock biomass trajectories (B/B_{MSY}) across each HCR, where F_{lim} is the maximum allowable fishing mortality rate, b is the threshold biomass level below which allowable fishing is reduced, and a is the limit biomass level below which allowable fishing mortality is set to zero. Results are presented across implementation scenarios (rows) and HCR parameterizations (F_{lim} values as columns), where F_{MSY} is the fishing mortality rate that would lead to biomass level that would produce maximum sustainable yield (B_{MSY}), and M is the natural mortality rate. Various configurations of a and b are colour-coded, where B_0 is virgin biomass. Each thin, transparent line represents one simulated iteration (100 iterations per OM × HCR × Implementation scenario). Thick, opaque lines represent median trajectories for each scenario. [Colour online.]

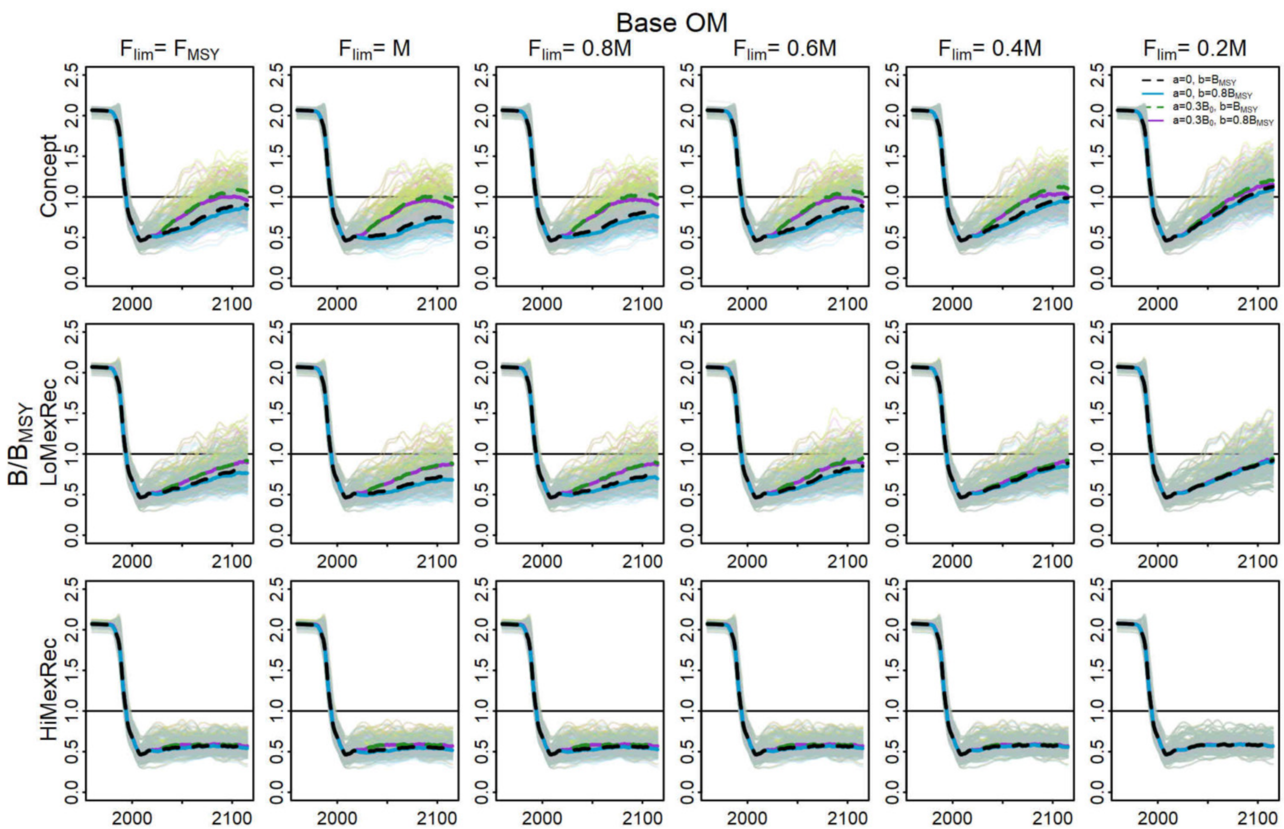
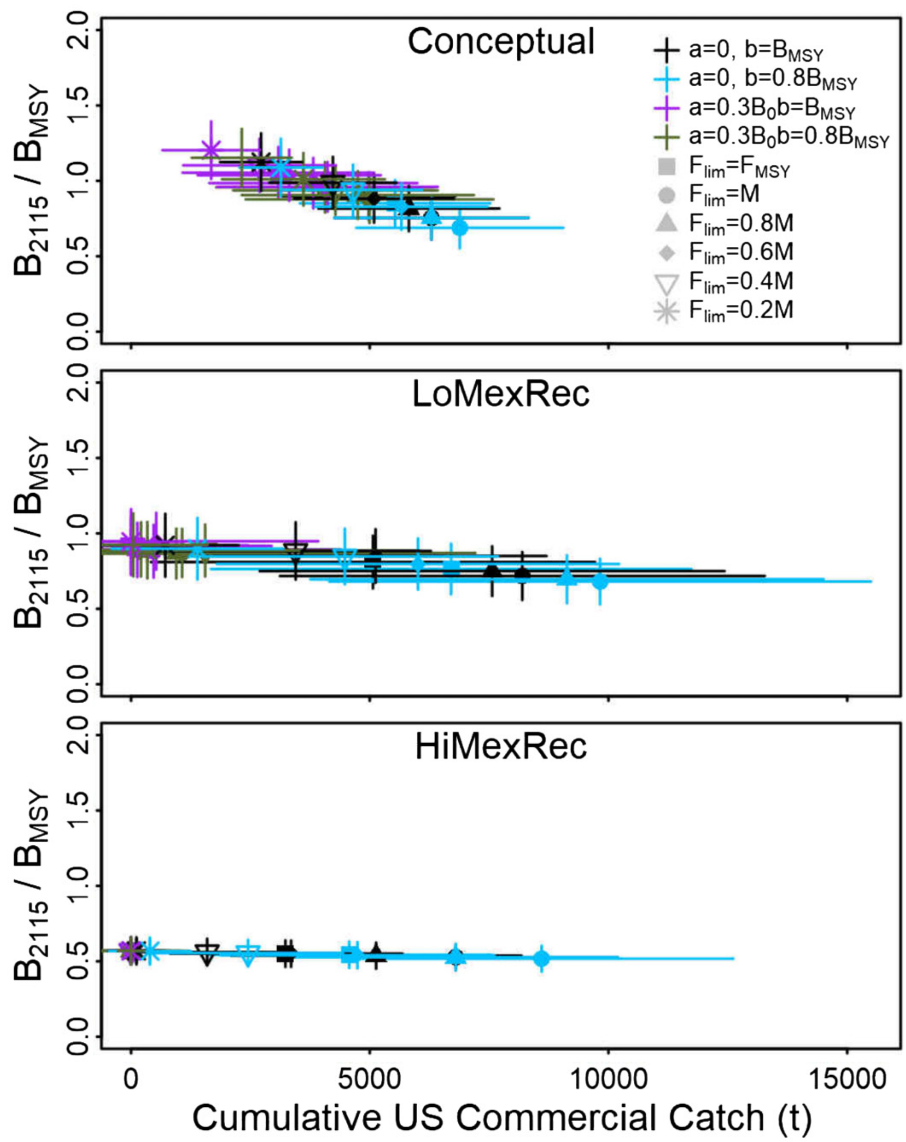


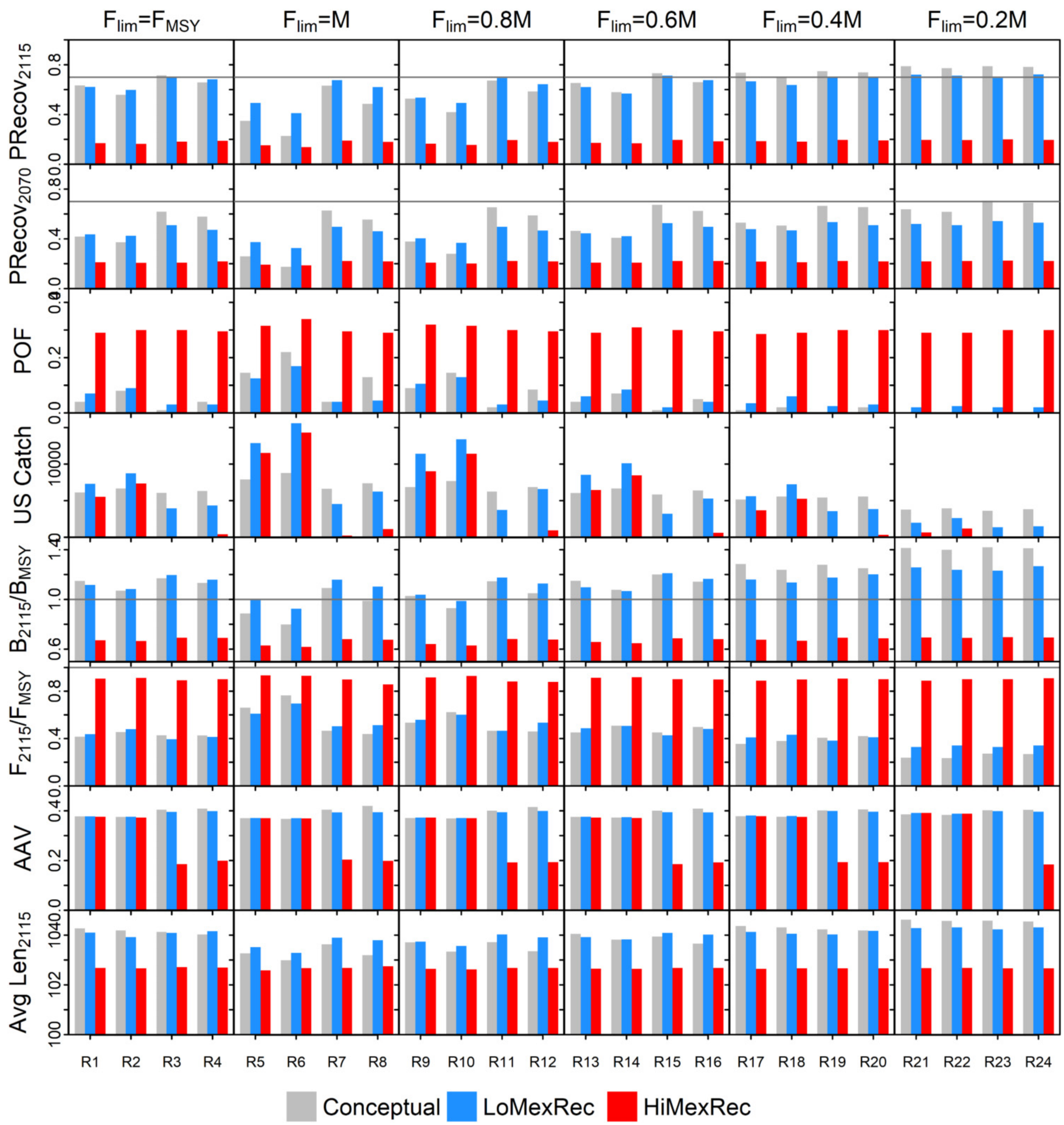
Fig. 7. Trade-off plots showing the relationship between terminal spawning stock biomass (B_{2115}/B_{MSY}) and cumulative US commercial catch throughout the entire simulation horizon of OM_Base across HCRs for each implementation scenario. HCRs are parameterized where F_{lim} is the maximum allowable fishing mortality rate, b is the threshold biomass level below which allowable fishing is reduced, and a is the limit biomass level below which allowable fishing mortality is set to zero. F_{MSY} is the fishing mortality rate that would lead to biomass level that would produce maximum sustainable yield (B_{MSY}), M is the natural mortality rate, and B_0 is virgin spawning stock biomass. [Colour online.]



While it is ultimately up to managers to determine acceptable risk level, few MPs tested resulted in acceptable recovery probabilities, defined as median probabilities of recovery greater than 50% by the end of the 2070 rebuilding period. Within the HiMexRec scenario, the median probability of recovery was less than 50% for all HCRs in the years 2070 and 2115 (Table S1). In the LoMexRec implementation scenario, the HCRs in which median recovery probabilities were acceptable were: HCRs where F_{lim} was equal to 0.2M, HCRs where F_{lim} was equal to 0.4M and a was set equal to 0.3B_0, and HCRs where F_{lim} was equal to F_{MSY} or 0.6M, a was 0.3B_0, and b was equal to B_{MSY} (Table S2). For the Conceptual implementation scenarios, all HCRs in which a was set to 0.3B_0 resulted in acceptable median recovery probabilities, although the HCRs where F_{lim} was equal to M did not maintain a median probability of recovery greater than 0.5 by 2115 when b was equal to 0.8B_{MSY} (Table S3). Predictably, cumulative US commercial catch was lower in scenarios where probability of recovery was higher (Tables S1–S3).

Notably, the median performance of each HCR across equally weighted OMs for each implementation scenario demonstrates the significance that the future unknown MexRec catches has on the future of the sandbar shark fishery, particularly with respect to the HiMexRec implementation scenario. Trade-offs inherent in fisheries management,

Fig. 8. Graphical decision table displaying HCR performance with respect to six management objectives, across three implementation models, and assuming equal weight for each OM. Performance metrics include probability of recovery by 2115 ($PRecov_{2115}$), probability of recovery by 2070 ($PRecov_{2070}$), probability of overfishing throughout the time horizon (POF), cumulative US commercial catch throughout the time horizon (US Catch), relative terminal spawning stock biomass (B_{2115}/B_{MSY}), relative terminal fishing mortality rate (F_{2115}/F_{MSY}), average annual variability in catch (AAV), and average length of females in the year 2115 (Avg. Len). HCRs (labeled R in the figure) are defined in Table 2, F_{lim} is the maximum allowable fishing mortality rate, F_{MSY} is the fishing mortality rate that would lead to biomass level that would produce maximum sustainable yield (B_{MSY}), and M is the natural mortality rate. [Colour online.]



such as the trade-off between increased US commercial catch and terminal relative spawning stock biomass, were clearly demonstrated for the sandbar shark, yet these trade-offs varied based on the magnitude of unmanaged removals from the population. Consider that compared with the Conceptual and the LoMexRec implementation scenarios, the HiMexRec scenario resulted in large increases in POF and decreases in the probability of recovery, without corresponding increases in cumulative US commercial catch (Fig. 8). MSE simulations further indicated that AAV and the annual average length of females in the stock would not be expected to change substantially with choice in HCR.

Discussion

We followed the Maunder (2014) approach to create a Stock Synthesis-based MSE simulation framework and applied it to the large coastal sandbar shark. The performance of variously parameterized HCRs demonstrates the management trade-off space for the sandbar shark. The best-performing threshold MPs generally displayed a ramp to zero fishing at low stock sizes and maximum fishing mortality rates less than F_{MSY} or 80% of M. The MPs were tested against a wide range of uncertainties, and a key uncertainty explored was the future rate of MexRec fishing, which was accounted for using multiple implementation scenarios. Notably, the future MexRec catches fundamentally determined whether recovery of the sandbar shark stock was achievable within the southeastern US. Comparison of the HiMexRec and LoMexRec implementation scenarios to the Conceptual implementation scenarios demonstrates the capacity for improved resource management when co-exploiting nations act cooperatively.

HCR parameterization

Unsurprisingly, we found that sustainable exploitation rates for the sandbar shark are low (Apostolaki et al. 2006), and in particular, the ratio of F_{MSY}:M across our OMs ranged from 0.530 to 1.177, with a mean value of 0.804 and a median value of 0.788. The exact optimal fishing rate relative to natural mortality was dependent on the OM (Table 1), and in practice, optimal F_{MSY}, and therefore F_{lim}, would further depend on the specifics of the fishery, including selectivity and allocation of fishing mortality (which notably changed in each simulated implementation scenario). These findings are comparable to estimates by Zhou et al. (2012), who defined an optimal F_{MSY}:M ratio of 0.41 for chondrichthyan fishes, and Cortés and Brooks (2018), who calculated a median ratio of 0.64 based on results of 33 shark stock assessments. Accordingly, F_{lim} had a larger effect on MP performance than the other HCR parameters. Given the ratio of F_{MSY}:M, fishing at a rate around 0.6M to 0.4M resulted in projections most comparable to those where F_{lim} was set equal to F_{MSY}. Fishing at a rate equal to the mean age 1+ natural mortality rate was too high across all OMs for the sandbar shark, whereas fishing at a rate of 0.2M was too low, resulting in forfeited catch after the stock recovered to B_{MSY}.

Where stock recovery is a primary management objective, threshold HCRs with a steep ramp and zero fishing at low stock sizes (e.g., with $a = 0.3B_0$ and $b = B_{MSY}$) may be good candidates for further evaluation as HCRs for Atlantic HMS. These HCRs decreased target catch to account for uncertainty in the observation and assessment of the fishery, and they appear rebuild the stock consistent with rebuilding plans as implemented under the MSA. The relatively small impact of the HCR parameter values, a and b, suggested that implementation of a precautionary MP was more important than defining optimal parameters of the HCR. Nevertheless, the choice in F_{lim}, a, and b demonstrated the trade-offs inherent in managing marine fisheries resources. Namely, when a was larger (i.e., more precautionary), the increase in B was countered by a substantial reduction in cumulative commercial catch. The effect of b was small, but larger b values resulted in lower cumulative catch and increased probability of recovery (Fig. 8). Median probabilities of stock recovery increased when a was $0.3B_0$ and were rarely acceptable ($\text{PRecov}_{2070} \geq 0.5$) when a was equal to 0.0, excepting the HiMexRec implementation scenarios, wherein median $\text{PRecov}_{2070} < 0.5$ for all HCR parameterizations (Fig. 8).

A key finding was that the success or failure of the MPs considered for the sandbar shark within the US was largely dependent on the rate of MexRec fishing. Comparably, Van Beveren et al. (2020) found that the presence and magnitude of unobserved catch had a much larger effect on the capacity of the transboundary northern mackerel stock to recover than the choice of HCR. This finding follows that of Thorpe and De Oliveira (2019), who noted that implementation of an HCR that reduced allowable fishing mortality at low stock sizes was more important than the exact specifications of the HCR.

Uncertainties in the system

This MSE included six OMs and three implementation models designed to address key uncertainties in the sandbar shark fishery. Accounting for uncertainties within an MSE is critical to evaluate whether each MP is robust to the reasonable uncertainties in the system (Butterworth and Punt 1999; Punt et al. 2016). The most significant sources of uncertainty for the sandbar shark were deemed to be future Mexican catches, the form and parameterization of the S–R relationship, and natural mortality.

Both natural mortality and the form and parameterization of the S–R relationship are uncertainties that should regularly be considered in an MSE (Deroba and Bence 2008; Punt et al. 2016), as HCR performance has been particularly sensitive to natural mortality in a variety of r- and K-selected life history strategists (Butterworth and Punt 1999). Furthermore, the S–R relationship has been known to be a significant source of uncertainty in elasmobranchs (Kai and Yokoi 2017; Kai and Fujinami 2018), along with natural mortality (Kai and Yokoi 2017). Punt et al. (2016) also recommended exploring uncertainty in the overall size of the resource, which we characterized through the magnitude of virgin recruitment.

Impact of the S–R relationship

A key uncertainty evaluated was the effect of assuming an LFSR (OM_Base) versus a BH S–R (OM_BH) relationship. The distinction is in the density-dependent compensatory response of the population following population reduction. While most stock assessment parameters were very similar between the OM_Base and OM_BH parameterized models (e.g., estimated F, depletion), derived MSY-based management reference points were different (Table 1). Estimated MSY, B_{MSY}, and F_{MSY} were lower in OM_BH than in OM_Base. Therefore, the OM_Base assumed the status of the stock was more pessimistic than the OM_BH stock status estimates (see Figs. 4–5).

We investigated the impact of assuming an LFSR relationship in the OM, while the EM assumed a BH S–R relationship on MP performance. If the sandbar shark stock follows an LFSR relationship and we assess the stock using a BH S–R relationship (e.g., OM_Base), then the EM will assume that B_{MSY} is lower than it really is, which could result in an overfished stock. On the other hand, though not tested in the current simulation, if the stock follows a BH S–R relationship and the EM assumes an LFSR relationship, then the stock could also be subjected to overfishing since MSY is larger for stocks that follow an LFSR relationship than those that follow a BH S–R relationship.

Form of implementation uncertainty

A unique aspect of this MSE was the necessity to account for uncertainty in future, unmanaged catches. This is a consideration that has not received much attention within the MSE literature (e.g., Van Beveren et al. 2020). We present an approach to incorporate uncertainty in future catches by building alternate implementation modules that envelop the expected range of future MexRec projections. The extent of future, relative to historical, uncertainty that should be incorporated into an MSE has been debated (e.g., Kolody et al. 2008; Butterworth 2008*a*; Butterworth 2008*b*). Although in cases such as the sandbar shark fishery, we agree with Kolody et al. (2008) that it would be negligent to exclude this critical source of uncertainty in our simulation.

MexRec catches were treated as a single fleet because of issues with species misidentification or lack of species-specific landing information, the uncertainty in recreational removals, and reconstructed catches in the early historical time period (Cortés 2011). Importantly, both fleets were assumed to exploit animals of similar sizes. Though the treatment of a single MexRec fleet was not ideal, we note that it ultimately did not impact the results of the current study. Given our current understanding of the fishery, the separated fleets would have been modeled with the same selectivity pattern and the same implementation scenarios would still be necessary to reflect our inability to predict future Mexican catches.

The ability to predict the future Mexican harvest of sandbar shark is presently lacking, so we explored the impacts of the two extreme cases of high- or low-projected MexRec catches on the sandbar shark stocks. The rate of increase in MexRec catches with stock biomass in the HiMexRec scenario is likely an upper bound, since the rate was based on both Mexican catches and US recreational catches, and harvest of sandbar shark has since been prohibited in the US recreational fishery. On the other hand, the rate of Mexican harvest in the LoMexRec scenario serves as a lower bound, since an increase in sandbar shark biomass will likely increase encounter rates of MexRec fishers, which could reasonably lead to increased catch-related mortality. By estimating plausible high and low MexRec catch scenarios, we are effectively creating an envelope around potential future states of nature. Ultimately, future MexRec removals will have a substantial impact on the ability of the sandbar shark stock to recover to B_{MSY}.

In the HiMexRec scenario, the sandbar shark fishery management objectives were maximized by deliberately overfishing the stock. Any foregone US commercial yield would merely be taken by the MexRec fleet. Consequently, there was no added benefit to reducing US catch in the short term, as it failed to result in long-term increases in yield or biomass. This scenario is akin to a pseudo-“prisoner’s dilemma” in which cooperation between two parties would yield in the most beneficial outcome overall, but each party assumes the other will not cooperate and instead acts in a self-interested manner wherein non-cooperation becomes the best individual strategy (Munro 2009). Although, MSA mandates prevent deliberate overfishing in the US (MSA 2007). In the LoMexRec scenario, recovery was achievable within a reasonable probability (e.g., 41%–72% depending on MP), but owing to additional removals that were not accounted for in the target catch determination, recovery time was greater than that within the Conceptual model when all major sources of fishery removals were managed.

The Conceptual MP performance served as a baseline for the sandbar shark, demonstrating the impact of additional, unmanaged catch on MP performance. The Conceptual MP also provides insight into how a threshold HCR would perform for other domestic coastal shark species, given a species of similar life history and fishery structure wherein all removals are managed by a single governing body. The improved management performance of the Conceptual MP further exemplifies what could be realized under a coordinated international management effort.

Conceptual versus expected implementation scenario performance

We illustrated the distinction between how intuitively an MP *should* perform a priori (Conceptual MP performance following the Conceptual implementation scenario) compared with how the MP is expected to perform in a given system (Expected MP performance following the Expected implementation scenarios). In this simulation, the Expected MP performance accounted for Mexican removals that were not subjected to the US’s MP (HiMexRec and LoMexRec implementation scenarios), while the Conceptual MP performance is the case in which all substantial fishery removals are subjected to management through the MP (Conceptual implementation scenario). In the Conceptual scenario, spawning stock biomass recovered until it plateaued at a level corresponding to the respective F_{lim}, accounting for natural differences between “true”, simulated dynamics and dynamics assumed in the EM for each OM (Fig. S6). However, in the HiMexRec and LoMexRec scenarios, recovery was unachievable or slower (Figs. 8 and Figs. S10 and S14), while US commercial catch and the length composition of the stock were potentially affected (Fig. 8). The impact of high MexRec fishing had the largest impact on the management objectives relative to the Conceptual scenario.

This research highlights the importance of considering relevant uncertainties that may affect the performance of an MP within a fishery of interest. Given the fishery-specific nature of an MP, it is generally understood that if the intent of the MSE is to adopt the MP, MSEs should be conducted on a stock-specific basis to ensure that the proposed MP can accommodate the specific life history and fishery of that stock (Butterworth and Punt 1999; Apostolaki et al. 2006; Kronlund et al. 2014; Forrest et al. 2018). The ultimate utility of MSE results is largely dependent on whether the OM is able to capture the true fishery and population dynamics and incorporate the full range of uncertainty (Butterworth and Punt 1999). However, in the absence of unlimited capacity to conduct many species-specific MSEs, implementation of a generic HCR simulation-tested through a generic (nonspecies-specific) desk MSE (e.g., Punt et al. 2016) will likely suffice for many stocks (e.g., 40–10 HCR; Punt and Donovan 2007). We conducted the Conceptual implementation scenario to serve as a generic MSE for other coastal shark species with similar life histories for which catches can be regulated.

Comparing Conceptual versus Expected MP performance suggests that failing to account for all unique aspects of the fishery (e.g., international removals) may substantially alter the MP performance in practice. For example, we emphasize the difference in MP performance between the Conceptual and HiMexRec Expected MP scenarios. We should not expect "generic" HCR performance (e.g., Conceptual MP scenario) within the US sandbar shark fishery. Further considerations in other systems may include significant ecosystem dynamics (e.g., red tide or climate change; Harford et al. 2018; Holsman et al. 2020), delays in data availability and fishery management implementation (e.g., Shertzer and Prager 2007), spatial or stock structure (e.g., Atlantic bluefin tuna; Carruthers and Butterworth 2018), among many others.

As in the sandbar shark fishery, the concept of multiple implementation models may be useful in additional unconventional circumstances. For example, consider fisheries where total and projected removals are unknown, including fisheries dominated by the recreational sector (Shertzer et al. 2019), bycatch species with high at-vessel or postrelease mortality (e.g., pelagic sharks; Bonfil 1994), or illegal, unreported, and unregulated (IUU) fishing (Stiles et al. 2013). Each of these concerns are particularly relevant for sharks managed within the US. The results of our study highlight the importance of fully considering how MP application will occur in the future within a given fishery.

Challenges managing coastal sharks

Despite encouraging preliminary indicators of stock recovery following unregulated overexploitation of coastal sharks in the 1970s and 1980s and subsequent precautionary management implementation in the 1990s (Peterson et al. 2017), assessments still show that a number of large coastal sharks are overfished and under rebuilding plans (SEDAR 2016, 2017). The fishery, along with the abundance of many coastal shark stocks, has seemingly not fully recovered (Carlson et al. 2012). Ultimately, the challenges of assessing coastal sharks are numerous and well documented (Musick et al. 2000; Stevens 2000; Cortés 2011).

Maintaining biomass at a level that supports removal of optimum yield is the objective that has been codified within US fisheries management legislation (MSA 2007), and in practice, optimum yield is generally considered equal to MSY for domestic coastal sharks. However, optimum yield is technically defined as MSY "as reduced by any social, economic, or ecological factor" (NMFS 2016). We further acknowledge that fishing activities can, in fact, be sustainable at levels other than MSY and B_{MSY}. As determined by the prioritization of management objectives for the sandbar shark, the optimal fishery configuration may be one in which the ideal biomass is not equal to B_{MSY}. Within such a contentious management framework, these topics may warrant additional consideration as fisheries management continues to evolve. We emphasize that it is not our goal as scientists to prescribe an optimal MP, as the best MP would be largely dependent on the personal ranking of management goals of each individual. Instead, we lay bare the inherent trade-offs between management objectives associated with each MP tested for the sandbar shark fishery in the US across system-wide uncertainties.

International fisheries management

This research additionally highlights the challenges and importance of cooperative management of migratory and transboundary stocks. International fisheries management is often subjected to the "tragedy of the commons", wherein the interests of competing nations likely do not support long-term sustainability goals (Munro 2009). This was demonstrated in our HiMexRec scenario, wherein overfishing the stock maximized US sandbar shark management objectives despite not achieving stock recovery. Likewise, McWhinnie (2009) demonstrated that fisheries shared by multiple nations are more likely to be overfished. These results are exacerbated when the target stock is slow-growing and (or) of high economic value (McWhinnie 2009).

International fisheries management is particularly challenged when participating nations are not a part of the management entity governing fisheries management of the stock (e.g., Koubrak and VanderZwaag 2020). These "free-riding" nations typically receive the benefits of sustainable and collaborative fisheries management without the requirement to abide by the regulations of the cooperative agreement (Munro 2009). Inevitably, the challenges and significance of collaborative international fisheries will only heighten in the face of climate change (e.g., Engler 2020; Koubrak and VanderZwaag 2020; Sumaila and VanderZwaag 2020), especially considering that changes in the fishery, such as those catalyzed by climate change, often stimulate disruption in cooperative management agreements (Munro 2009).

Conclusions

Execution of an MSE to characterize HCR performance on coastal Atlantic sharks has been repeatedly called for (Cortés et al. 2015; NMFS 2020). Management goals for Atlantic highly migratory species (HMS) include use of MSE to determine the legitimacy of various MPs and identification of barriers toward achievement of optimum yield for HMS (NMFS 2020). We conducted an MSE for a representative large coastal shark, which allowed us to identify trade-offs in management performance to the various HCR parameterizations tested for a large coastal shark, and identify unregulated removals as a potential barrier toward effective HMS management.

A key driver in the motivation to consider the Conceptual MP performance was the ability to apply the results of this MSE to other coastal shark species. Keeping in mind the caveats noted above, the results from this study may be useful for managing additional coastal shark species with similar life history, including those that are entirely distributed within US management boundaries or that are not harvested by other countries, until a stock-specific simulation may be undertaken. This study also highlighted that future MexRec fishing activities are a major uncertainty affecting the ability of the sandbar shark to recover. Utilization of multiple implementation models represented a way to explicitly account for uncertainty in future nonregulated removals. We believe these findings will prove useful in the future of Atlantic coastal shark management.

Acknowledgements

We acknowledge William & Mary Research Computing (https://www.wm.edu/it/rc) and the NMFS Beaufort Laboratory for providing computational resources and technical support, as well as Andrew Scheld, Nathan Vaughan, and two reviewers for comments on this manuscript. This paper represents Contribution No. 4085 of the Virginia Institute of Marine Science, William & Mary.

Article information

History dates

Received: 16 December 2021
Accepted: 24 March 2022
Accepted manuscript online: 25 March 2022
Version of record online: 16 August 2022

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Data availability

Code for this project is available on Github (https://github.com/cassidydpeterson/SS_MSE) and was archived with Zenodo (doi:10.5281/zenodo.6373778; https://zenodo.org/badge/latestdoi/238532004). Additional methods, results, tables, and figures, as well as a detailed MSE protocol are available in the Supplementary material.

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Michael J. Wilberg served as an Associate Editor at the time of manuscript review and acceptance; peer review and editorial decisions regarding this manuscript were handled by Sean Anderson.

Author contributions

CDP, MJW, EC, DLC, and RJL all contributed to the conceptualization and design of the study. CDP and RJL secured funding with assistance from EC. CDP performed coding and analysis for the study and wrote the original draft of the manuscript. MJW, EC, DLC, and RJL reviewed and edited all versions of the manuscript.

Competing interests

The authors declare there are no competing interests.

Funding information

CDP was funded in part by the NMFS Sea Grant Population and Ecosystem Dynamics Fellowship (NA17OAR4170242).

Supplementary material

Supplementary data are available with the article at https://doi.org/10.1139/cjfas-2021-0345.

References

A'mar, T.Z., Babcock, E.A., and McAllister, M.K. 2006. Contrasting deterministic and probabilistic ranking of catch quotas and spatially and size-regulated fisheries management. Can. J. Fish. Aquat. Sci. **63**(8):1777–1792. doi:10.1139/f06-078

Apostolaki, P., Babcock, E.A., and McAllister, M.K. 2006. Contrasting deterministic and probabilistic ranking of catch quotas and spatially and size-regulated fisheries management. Can. J. Fish. Aquat. Sci., **63**(8): 1777–1792. doi:10.1139/f06-078.

Baremore, I.E., and Hale, L.F. 2012. Reproduction of the sandbar shark in the western North Atlantic Ocean and Gulf of Mexico. Mar. Coast. Fish. **4**(1): 560–572. doi:10.1080/19425120.2012.700904.

Bonfil, R. 1994. World bycatches of sharks in high-seas fisheries: appraising the waste of a resource. *In* Bycatches in Fisheries and Their Impact on the Ecosystem. *Edited by* T.J. Pitcher and R.Y. Chuenpagdee. The Fisheries Center, The University of British Columbia, Vancouver, B.C., Canada. pp. 41–44.

Butterworth, D.S. 2008a Some lessons from implementing management procedures. *In* Fisheries for Global Welfare and Environment: Memorial Book of the 5th World Fisheries Congress 2008. *Edited by* K. Tsukamoto, T. Kawamura, T. Takeuchi, T.D. Beard, Jr. and M.J. Kaiser. TERRAPUB, Tokyo. 381–397.

Butterworth, D.S. 2008b. A commentary on: salvaged pearls: lessons learned from a floundering attempt to develop a management procedure for southern Bluefin Tuna. Fish. Res. **94**: 351–354. doi:10.1016/j.fishres.2008.09.034.

Butterworth, D.S., and Punt, A.E. 1999. Experiences in the evaluation and implementation of management procedures. ICES J. Mar. Sci. **56**: 985–998. doi:10.1006/jmsc.1999.0532.

Carlson, J.K., Hale, L.F., Morgan, A., and Burgess, G. 2012. Relative abundance and size of coastal sharks derived from commercial shark longline catch and effort data. J. Fish Biol. **80**: 1749–1764. doi:10.1111/j.1095-8649.2011.03193.x. PMID: 22497406.

Carlson, J.K., Heupel, M.R., Young, C.N., Cramp, J.E., and Simpfendorfer, C.A. 2019. Are we ready for elasmobranch conservation success? Environ. Conserv. **46**(4): 264–266. doi:10.1017/s0376892919000225.

Carruthers, T.R., and Butterworth, D. 2018. ABT-MSE: an R package for Atlantic bluefin tuna management strategy evaluation. Collect. Vol. Sci. Pap. ICCAT **74**(6): 3553–3559.

Carruthers, T.R., Kell, L.T., Butterworth, D.D.S., Maunder, M.N., Geromont, H.F. Walters, C., et al. 2016. Performance review of simple management procedures. ICES J. Mar. Sci. **73**(2): 464–482. doi:10.1093/icesjms/fsv212.

Castro, J.I. 2016. The origins and rise of shark biology in the 20th century. Mar. Fish. Rev. **78**(1-2): 14–33. doi:10.7755/MFR.78.1-2.2.

Cisneros-Montemayor, A.M., Barnes-Mauthe, M., Al-Abdulrazzak, D., Navarro-Holm, E., and Sumaila, U.R. 2013. Global economic value of shark ecotourism: implications for conservation. Oryx, **47**(3): 381–388. doi:10.1017/s0030605312001718.

Clarke, S., and Hoyle, S. 2014. Development of limit reference points for elasmobranchs. Western and Central Pacific Fisheries Commission.

Cortés, E. 2011. An Overview of Approaches used to Assess the Status of Shark Populations: Experiences from the USA and ICCAT in the Atlantic Ocean The 7th Session of the Working Party on Ecosystems and Bycatch. 11 pp. IOTC-2011-WPEB07-25

Cortés, E., and Brooks, E.N. 2018. Stock status and reference points for sharks using data-limited methods and life history. Fish Fish. **19**(6): 1110–1129. doi:10.1111/faf.12315.

Cortés, E., Brooks, L., and Scott, G. 2002. Stock Assessment of Large Coastal Sharks in the U.S. Atlantic and Gulf of Mexico. Sustainable Fisheries Devision Contribution SFD-01/02-152. NOAA Fisheries, Southeast Fisheries Science Center, Panama City.

Cortés, E., Brooks, E.N., and Shertzer, K.W. 2015. Risk assessment of cartilaginous fish populations. ICES J. Mar. Sci. **72**(3): 1057–1068. doi:10.1093/icesjms/fsu157.

Deroba, J.J., and Bence, J.R. 2008. A review of harvest policies: understanding relative performance of control rules. Fish. Res. **94**(3): 210–223. doi:10.1016/j.fishres.2008.01.003.

DeVore, J., and Gilden, J. Sixth national meeting of the regional fishery management councils' scientific and statistical committees. *In* National SSC Workshop. Pacific Fishery Management Council, Portland, OR, 2019. p. 68.

Doering, K., and Vaughan, N. 2020. SSMSE: management strategy evaluation (MSE) using stock synthesis (SS) [online]. NOAA Fisheries Integrated Toolbox, Available from http://nmfs-fish-tools.github.io/SSMSE/

Dulvy, N.K., Fowler, S.L., Musick, J.A., Cavanagh, R.D., Kyne, P.M. Harrison, L.R., et al. 2014. Extinction risk and conservation of the world's sharks and rays. Elife, **3**: e00590. doi:10.7554/eLife.00590. PMID: 24448405.

Ellis, J.R., Clarke, M.W., Cortés, E., Heessen, H.J.L., Apostolaki, P. Carlson, J.K., et al. 2008. Management of elasmobranch fisheries in the North Atlantic. *In* Advances in Fisheries Science: 50 years on from Beverton and Holt. *Edited by* A. Payne, J. Cotter and T. Potter. Blackwell Publishing, Oxford, UK. pp. 184–228.

Engler, C. 2020. Transboundary fisheries, climate change, and the ecosystem approach: taking stock of the international law and policy seascape. Ecol. Soc. **25**(4): 43. doi:10.5751/es-11988-250443. PMID: 32523609.

FAO. 1996. Precautionary approach to fisheries. Part 2: scientific papers. FAO, Lyskil, Sweden.

Forrest, R.E., Holt, K.R., and Kronlund, A.R. 2018. Performance of alternative harvest control rules for two pacific groundfish stocks with uncertain natural mortality: Bias, robustness and trade-offs. Fish. Res. **206**: 259–286. doi:10.1016/j.fishres.2018.04.007.

Gallagher, A.J., and Hammerschlag, N. 2011. Global shark currency: the distribution, frequency, and economic value of shark ecotourism. Curr. Issues Tour. **14**(8): 797–812. doi:10.1080/13683500.2011.585227.

Gallagher, A.J., Hammerschlag, N., Danylchuk, A.J., and Cooke, S.J. 2017. Shark recreational fisheries: status, challenges, and research needs. Ambio, **46**(4): 385–398. doi:10.1007/s13280-016-0856-8. PMID: 27995551.

Goethel, D.R., Lucey, S.M., Berger, A.M., Gaichas, S.K., Karp, M.A. Lynch, P.D., et al. 2019. Closing the feedback loop: on stakeholder participation in management strategy evaluation. Can. J. Fish. Aquat.Sci. **76**(10): 1895–1913. doi:10.1139/cjfas-2018-0162.

Grubbs, R.D. 2010. Ontogenetic shifts in movements and habitat use. *In* Sharks and Their Relatives II. Biodiversity, Adaptive Physiology, and Conservation. *Edited by* J.C. Carrier, J.A. Musick and M.R. Heithaus. CRC Press, Boca Raton, FL. pp. 319–350.

Harford, W.J., Grüss, A., Schirripa, M.J., Sagarese, S.R., Bryan, M., and Karnauskas, M. 2018. Handle with care: establishing catch limits for fish stocks experiencing episodic natural mortality events. Fisheries, **43**(10): 463–471. doi:10.1002/fsh.10131.

Heist, E.J., Graves, J.E., and Musick, J.A. 1995. Population genetics of the sandbar shark (*Carcharhinus plumbeus*) in the Gulf of Mexico and mid-atlantic bight. Copeia, **1995**(3): 555–562. doi:10.2307/1446752.

Hicks, A.C., Cox, S.P., Taylor, N., Taylor, I.G., Grandin, C. Ianelli, J.N., et al. 2016. Conservation and yield performance of harvest control rules for the transboundary Pacific hake fishery in US and Canadian waters. *In* An Introduction to Simulation-Based Methods. *Edited by* C.T.T. Edwards and D.J. Dankel. Routledge, New York. pp. 69–85.

Holland, D.S. 2010. Management Strategy Evaluation and Management Procedures: Tools for Rebuilding and Sustaining Fisheries OECD Food, Agriculture and Fisheries Working Papers, No. 25 OECD Publishing, Paris. doi:10.1787/5kmd77jhvkjf-en

Holsman, K.K., Haynie, A.C., Hollowed, A.B., Reum, J.C.P., Aydin, K. Hermann, A.J., et al. 2020. Ecosystem-based fisheries management forestalls climate-driven collapse. Nat. Commun. **11**: 4579. doi:10.1038/s41467-020-18300-3. PMID: 32917860.

ICES. 2019. Workshop on Guidelines for Management Strategy Evaluations (WKGMSE2), 21–23 January 2013. ICES HQ, Copenhagen, Denmark.

ISC. 2019. Report for the first North Pacific albacore management strategy evaluation. *In* 19th Meeting for the International Scientific Committee for Tuna and Tuna-Like Species in the North Pacific Ocean. Taipei, Taiwan, 11-15July, **2019**. p. 157.

Kai, M., and Yokoi, H. 2017. Evaluation of harvest strategies for pelagic sharks taking ecological characteristics into consideration: an example for North Pacific blue shark. Can. J. Fish. Aquat. Sci. **74**(6): 933–947. doi:10.1139/cjfas-2016-0170.

Kai, M., and Fujinami, Y. 2018. Stock–recruitment relationships in elasmobranchs: application to the North Pacific blue shark. Fish. Res. **200**: 104–115. doi:10.1016/j.fishres.2017.10.025.

Kolody, D., Polacheck, T., Basson, M., and Davies, C. 2008. Salvaged pearls: lessons learned from a floundering attempt to develop a management procedure for Southern Bluefin Tuna. Fish. Res. **94**(3): 339–350. doi:10.1016/j.fishres.2008.08.016.

Koubrak, O., and VanderZwaag, D.L. 2020. Are transboundary fisheries management arrangements in the Northwest Atlantic and North Pacific seaworthy in a changing ocean? Ecol. Soc. **25**(4): 42. doi:10.5751/es-11835-250442. PMID: 32523609.

Kronlund, A.R., Holt, K.R., Shelton, P.A., and Rice, J.C. 2014. Current approaches for the provision of scientific advice on the precautionary approach for Canadian fish stocks: harvest decision rules. DFO Canadian Science Advisory Secretariat Research Document **2013/080**: vi+: 29p.

Kvamsdal, S.F., Eide, A., Ekerhovd, N.-A., Enberg, K., Gudmundsdottir, A. Hoel, A.H., et al. 2016. Harvest control rules in modern fisheries management. Elementa, **4**: 000114. doi:10.12952/journal.elementa.000114.

Maunder, M.N. 2014. Management Strategy Evaluation (MSE) implementation in Stock Synthesis: Application to Pacific bluefin tuna. IATTC.

McCandless, C.T., Kohler, N.E., and Pratt, H.L. 2007. Shark Nursery Grounds of the Gulf of Mexico and the East Coast Waters of the United States. American Fisheries Society, Bethesda, MD.

McWhinnie, S.F. 2009. The tragedy of the commons in international fisheries: an empirical examination. J. Environ. Econ. Manag. **57**(3): 321–333. doi:10.1016/j.jeem.2008.07.008.

Methot, R.D., and Wetzel, C.R. 2013. Stock synthesis: a biological and statistical framework for fish stock assessment and fishery management. Fish. Res. **142**: 86–99. doi:10.1016/j.fishres.2012.10.012.

Methot, R.D., Wetzel, C.R., Taylor, I.G., and Doering, K. 2020. Stock Synthesis User Manual Version 3.30.15. NOAA Fisheries, Seattle, WA.

Mitchell, J.D., McLean, D.L., Collin, S.P., and Langlois, T.J. 2018. Shark depredation in commercial and recreational fisheries. Rev. Fish Biol. Fish. **28**(4): 715–748. doi:10.1007/s11160-018-9528-z.

Monnahan, C.C., Muradian, M.L., and Kuriyama, P.T. 2014. A Guide for Bayesian Analysis in AD Model Builder [online]. Available from https://repository.library.noaa.gov/view/noaa/42095

Monnahan, C.C., Branch, T.A., Thorson, J.T., Stewart, I.J., and Szuwalski, C.S. 2019. Overcoming long Bayesian run times in integrated fisheries stock assessments. ICES J. Mar. Sci. **76**(6): 1477–1488. doi:10.1093/icesjms/fsz059.

MSA. 2007. Magnuson-Stevens Fishery Conservation and Management Act. *Edited by* U.S.D.o.C.N.M.F. Service. U.S. Department of Commerce. Washington, D.C. p. 170.

Munro, G.R. 2009. Game theory and the development of resource management policy: the case of international fisheries. Environ. Dev. Econ. **14**(1): 7–27. doi:10.1017/s1355770×08004671.

Musick, J.A., Branstetter, S., and Covolcoresses, J.A. 1993. Trends in shark abundance from 1974 to 1991 for the Chesapeake bight region of the US mid-Atlantic coast. *In* Conservation Biology of Elasmobranchs. *Edited by* S. Branstetter. NOAA Technical Report. NMFS **115**, U.S. Department of Commerce, Washington, D.C. pp. 1–18.

Musick, J.A., Burgess, G., Cailliet, G.M., Camhi, M., and Fordham, S. 2000. Management of sharks and their relatives (Elasmobranchii). Fisheries, **25**(3): 9–13. doi:10.1577/1548-8446(2000)025%3c0009:MOSATR%3e2.0.CO;2.

Natanson, L.J., and Deacy, B.M. 2019. Using oxytetracycline validation for confirmation of changes in vertebral band-pair deposition rates with ontogeny in sandbar sharks (*Carcharhinus plumbeus*) in the western North Atlantic Ocean. Fish. Bull. **117**(1-2): 50–58. doi:10.7755/fb.117.1.6.

Natanson, L.J., Skomal, G.B., Hoffmann, S.L., Porter, M.E., Goldman, K.J., and Serra, D. 2018. Age and growth of sharks: do vertebral band pairs record age? Mar. Freshwater Res. **69**(9): 1440–1452. doi:10.1071/mf17279.

NMFS. 2016. Magnuson-Stevens provisions; national standard guidelines. Fed. Regist. **81**(201): 71858–71904.

NMFS. 2019. Scoping Document Amendment 14 to the 2006 consolidated Atlantic highly migratory species fishery management plan. NOAA Fisheries.

NMFS. 2020. Atlantic highly migratory species management-based research needs and priorities. Office of Sustainable Fisheries. NOAA Fisheries.

Peterson, C.D., Belcher, C.N., Bethea, D.M., Driggers, W.B., Frazier, B.S., and Latour, R.J. 2017. Preliminary recovery of coastal sharks in the south-east United States. Fish Fish. **18**: 845–859. doi:10.1111/faf.12210.

Pilling, G.M., Apostolaki, P., Failler, P., Floros, C., Large, P.A. Morales-Nin, B., et al. 2008. Assessment and management of data-poor fisheries. *In* Advances in Fisheries Science: 50 years on from Beverton and Holt. *Edited by* A. Payne, J. Cotter and T. Potter. Blackwell Publishing, Oxford, UK. pp. 280–305.

Punt, A.E. 2010. Harvest control rules and fisheries management. *In* Handbook of Marine Fisheries Conservation and Management. *Edited by* R.Q. Grafton, R. Hilborn, D. Squires, M. Tait and M.J. Williams. Oxford University Press, New York.

Punt, A.E. 2017. Strategic management decision-making in a complex world: quantifying, understanding, and using trade-offs. ICES J. Mar. Sci. **74**(2): 499–510. doi:10.1093/icesjms/fsv193.

Punt, A.E., and Donovan, G.P. 2007. Developing management procedures that are robust to uncertainty: lessons from the international whaling commission. ICES J. Mar. Sci. **64**: 603–612. doi:10.1093/icesjms/fsm035.

Punt, A.E., Pribac, F., Taylor, B.L., and Walker, T.I. 2005. Harvest strategy evaluation for school and Gummy Shark. J. Northwest Atl. Fish. Sci. **35**: 387–406. doi:10.2960/J.v35.m517.

Punt, A.E., Butterworth, D.S., de Moor, C.L., De Oliveira, J.A.A., and Haddon, M. 2016. Management strategy evaluation: best practices. Fish Fish. **17**: 303–334. doi:10.1111/faf.12104.

R Core Team. 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Restrepo, V.R., Thompson, G.G., Mace, P.M., Gabriel, W.L., Low, L.L. MacCall, A.D., et al. 1998. Technical Guideance on the Use of Precautionary Approaches to Implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act. NOAA Technical Memorandum NMFS-F/SPO-31. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Sainsbury, K. 2008. Best practice reference points for Australian fisheries. Report to the Australian Fisheries Management Authority.

Sainsbury, K.J., Punt, A.E., and Smith, A.D.M. 2000. Design of operational management strategies for achieving fishery ecosystem objectives. ICES J. Mar. Sci. **57**: 731–741. doi:10.1006/jmsc.2000.0737.

SEDAR. 2006. SEDAR 11 Stock Assessment Report Large Coastal Shark Complex, Blacktip and Sandbar Shark. NOAA/NMFS Highly Migratory Species Management Division, Silver Spring, MD.

SEDAR. 2011. SEDAR 21 HMS Sandbar Shark Stock Assessment Report. SEDAR, North Charleston, SC.

SEDAR. 2016. Update to SEDAR 21 HMS Dusky Shark Stock Assessment Report. SEDAR, North Charleston, SC.

SEDAR. 2017. SEDAR 54 HMS Sandbar Shark Stock Assessment Report. SEDAR, North Charleston, SC.

Sharma, R., Levontin, P., Kitakado, T., Kell, L., Mosqueira, I. Kimoto, A., et al. 2020. Operating model design in tuna regional fishery management organizations: current practice, issues and implications. Fish Fish. **21**(5): 940–961. doi:10.1111/faf.12480.

Shertzer, K.W., and Prager, M.H. 2007. Delay in fishery management: diminished yield, longer rebuilding, and increased probability of stock collapse. ICES J. Mar. Sci. **64**: 149–159. doi:10.1093/icesjms/fsl005.

Shertzer, K.W., Williams, E.H., Craig, J.K., Fitzpatrick, E.E., Klibansky, N., and Siegfried, K.I. 2019. Recreational sector is the dominant source of fishing mortality for oceanic fishes in the Southeast United States Atlantic Ocean. Fish. Manag. Ecol. **26**(6): 621–629. doi:10.1111/fme.12371.

Simpfendorfer, C.A., Heupel, M.R., White, W.T., and Dulvy, N.K. 2011. The importance of research and public opinion to conservation management of sharks and rays: a synthesis. Mar. Freshwater Res. **62**(6): 518–527. doi:10.1071/mf11086. PMID: 22319023.

Stevens, J. 2000. The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. ICES J. Mar. Sci. **57**(3): 476–494. doi:10.1006/jmsc.2000.0724.

Stiles, M.L., Kagan, A., Shaftel, E., and Lowell, B. 2013. Stolen Seafood: The Impact of Pirate Fishing on our Oceans. Oceana. Available from https://oceana.org/reports/stolen-seafood-impact-pirate-fishing-our-oceans/

Sumaila, U.R., and VanderZwaag, D.L. 2020. Canada and transboundary fisheries management in changing oceans: taking stock, future scenarios. Ecol. Soc. **25**(4):44. doi: 10.5751/es-12209-250444. PMID: 32523609

Taylor, I.G., Gertseva, V., Methot, R.D., and Maunder, M.N. 2013. A stock–recruitment relationship based on pre-recruit survival, illustrated with application to spiny dogfish shark. Fish. Res. **142**: 15–21. doi:10.1016/j.fishres.2012.04.018.

Taylor, I.G., Doering, K.L., Johnson, K.F., Wetzel, C.R., and Stewart, I.J. 2021. Beyond visualizing catch-at-age models: lessons learned from the r4ss package about software to support stock assessments. Fish. Res. **239**: 105924. doi:10.1016/j.fishres.2021.105924.

Thorpe, R.B., and De Oliveira, J.A.A. 2019. Comparing conceptual frameworks for a fish community MSY (FCMSY) using management strategy evaluation—an example from the North Sea. ICES J. Mar. Sci. **76**(4): 813–823. doi:10.1093/icesjms/fsz015.

Tixier, P., Lea, M.A., Hindell, M.A., Welsford, D., Mazé, C. Gourguet, S., et al. 2020. When large marine predators feed on fisheries catches: global patterns of the depredation conflict and directions for coexistence. Fish Fish. **22**(1): 31–53. doi:10.1111/faf.12504.

Van Beveren, E., Duplisea, D.E., Marentette, J.R., Smith, A., and Castonguay, M. 2020. An example of how catch uncertainty hinders effective stock management and rebuilding. Fish. Res. **224**: 105473. doi:10.1016/j.fishres.2019.105473.

Wilberg, M.J., Thorson, J.T., Linton, B.C., and Berkson, J. 2010. Incorporating time-varying catchability into population dynamic stock assessment models. Rev. Fish. Sci. **18**(1): 7–24. doi:10.1080/10641260903294647.

Zhou, S., Yin, S., Thorson, James T., Smith, Anthony D.M., and Fuller, M. 2012. Linking fishing mortality reference points to life history traits: an empirical study. Can. J. Fish. Aquat. Sci. **69**(8): 1292–1301. doi:10.1139/f2012-060.