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ARTICLE

Rapid approach for assessing an unregulated fishery using a series of data-limited tools

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Abstract

Objective: Fisheries provide countless benefits to human populations but face many threats ranging from climate change to overfishing. Despite these threats and an increase in fishing pressure globally, most stocks remain unassessed and data limited. An abundance of data-limited assessment methods exists, but each has different data requirements, caveats, and limitations. Furthermore, developing informative model priors can be difficult when little is known about the stock, and uncertain model parameters could create misleading results about stock status. Our research illustrates an approach for rapidly creating robust initial assessments of unregulated and data-limited fisheries without the need for additional data collection.

Methods: Our method uses stakeholder knowledge combined with a series of data-limited tools to identify an appropriate stock assessment method, conduct an assessment, and examine how model uncertainty influences the results. Our approach was applied to the unregulated and data-limited fishery for Crevalle Jack *Caranx hippos* in Florida.

Result: Results suggested a steady increase in exploitation and a decline in stock biomass over time, with the stock currently overfished and undergoing overfishing. These findings highlight a need for management action to prevent continued stock depletion.

Conclusion: Our approach can help to streamline the initial assessment and management process for unregulated and data-limited stocks and serves as an additional tool for combating the many threats facing global fisheries.

KEYWORDS

CMSY, Crevalle Jack, data limited, FishPath, local ecological knowledge, stock assessment

INTRODUCTION

Despite their importance, the status of many global fisheries remains unknown or poorly estimated due to a lack of sufficient data or institutional capacity required to conduct traditional stock assessments (Cope et al. 2023). The majority of global fisheries are lacking formal assessment, and studies have estimated that these unassessed fisheries may be in significantly worse condition than assessed fisheries (Costello et al. 2012; Blasco et al. 2020; Hilborn et al. 2020). Furthermore, due to increasing fishing pressure and constraints on fisheries management programs, the development of monitoring and assessment plans for all harvested fish species is an unattainable goal (Harford and Carruthers 2017; Sagarese et al. 2019). Although significant progress has been made toward improving fisheries data collection (e.g., Bryan et al. 2016; Amoroso et al. 2018; Rousseau et al. 2019), there will likely continue to be a need for alternative, data-limited approaches to stock assessment in the future (Sagarese et al. 2019). This is particularly true for areas like the southeastern United States, a highly biodiverse region where fisheries are dominated by the recreational sector (Shertzer et al. 2019) and where over 75% of stocks are considered data limited (i.e., lacking sufficient data to conduct traditional assessments; Berkson and Thorson 2015; Newman et al. 2015). There is an urgent need for (1) rapid assessment and management action that can keep pace with increasing fishing pressure and (2) methods that can identify unregulated and data-limited fisheries that are at risk of overexploitation and depletion (Sun et al. 2020).

Over the past few decades, numerous data-limited assessment methods have emerged to tackle this issue (Dowling et al. 2015). Rather than relying on traditional quantitative, model-based stock assessments, these methods estimate the status of fish stocks by using a range of approaches from expert judgment to multiple indicator models (Dowling et al. 2019). However, methods differ greatly in their data requirements, caveats, and context, making it difficult to determine which assessment method is the best choice for a particular fishery. Blanket application of generic models can lead to an inaccurate portrayal of fishery status and trends, thereby hindering effective management (Dowling et al. 2019). This is because using generic methods without first assessing whether they are suitable for the fishery of interest increases the likelihood of violating model assumptions and overlooking biases or other data quality issues. Fortunately, several decision support tools have been developed in recent years that aim to assist fisheries scientists, managers, and stakeholders in determining the appropriate methods for assessing and managing a given fishery (McDonald et al. 2018). One example is the FishPath tool (www.fishpath.org), a decision support tool that was developed in 2016 and helps to guide users through the selection of appropriate methods for monitoring, assessment, and management of data-limited fisheries (Dowling et al. 2016). The FishPath online assessment tool contains a repository of data requirements and assumptions for over 50 stock assessment methods, with a focus on data-limited options (Fitzgerald et al. 2018; Dichmont et al. 2021). Users first characterize their fishery via a series of multiple-choice questions concerning biological and life history attributes, fishery operational characteristics, data availability, socioeconomic factors, and governance context. The answers to these questions are then used to identify possible assessment and management options that are best suited to the fishery. Using a standardized tool like FishPath can provide consistency and objectivity to data-limited fisheries management and has the potential to become a key resource for the assessment and management of unregulated species (Fitzgerald et al. 2018).

Impact statement

This study presents a method for conducting rapid, low-cost fish stock assessments that was applied to the Crevalle Jack fishery in Florida. Application of this method to unregulated fish species can help managers better assess fish stocks and conserve important fisheries.

In addition to the development of numerous alternative approaches to traditional stock assessment, fisheries science is increasingly using stakeholder local ecological knowledge (LEK) to help identify conservation concerns (Silvano and Valbo-Jørgensen 2008; Gervasi et al. 2022b), estimate trends in stock status over time (Beaudreau and Levin 2014; Kroloff et al. 2019), improve fisheries models (Bélisle et al. 2018), and fill in critical knowledge gaps about species biology and ecology (Anadón et al. 2010). Local ecological knowledge is the in-depth knowledge of the local natural environment obtained by individuals or groups of people through personal observations, practical experience, and community dialog (Anadón et al. 2009). Research has shown that angler LEK can complement biological data and provide new insights (Silvano et al. 2008; Cardoso da Silva et al. 2020). For example, Figus et al. (2017) showed that both fishermen and scientists observed similar declines in the abundance and condition of Atlantic Cod *Gadus morhua* in the eastern Baltic Sea, Poland. In addition to this consensus, fisher LEK revealed a potential driver of the decline that was at odds with the findings of scientists, prompting additional avenues for research. There are several examples of angler LEK being used to directly inform fisheries management, including developing management options with a high probability of success and compliance (Heyman and Granados-Dieseldorff 2012), understanding causes of disagreement with existing management measures and differing stakeholder preferences (Hill et al. 2010; Figus and Criddle 2019), developing fishery surveillance indicators that can be used to continually monitor fisheries (Shephard et al. 2021), and providing estimates of model parameters used in stock assessments (Ainsworth and Pitcher 2005; Beaudreau and Levin 2014; Friedlander et al. 2015). Although these studies demonstrate clear benefits to incorporating angler LEK into fisheries management, there has been a lack of standardized protocols and methods for doing so (Hind 2015) and integration of LEK into biological assessments remains uncommon (Figus et al. 2017).

The goal of this study was to develop an approach for conducting rapid initial assessments (i.e., using only existing information sans additional data collection) of unregulated and data-limited fisheries that could be applied to the Crevalle Jack *Caranx hippos* fishery in Florida. The Crevalle Jack is a large marine species that is targeted by both commercial and recreational anglers, but the fishery in Florida is currently unregulated and data limited. Furthermore, research has suggested that the population may be in decline (Gervasi et al. 2022b). Our approach used angler LEK in conjunction with a series of data-limited assessment tools to assess the current status of the Florida Crevalle Jack stock, examine trends in stock status and exploitation over time, and develop initial management reference points, which are benchmarks that scientists and managers use to set targets or limits on fishing effort and to monitor the success of management strategies (Caddy and Mahon 1995). First, we used information gathered from LEK and other sources to fill out the FishPath assessment questionnaire and to choose a data-limited stock assessment method that was suited to the fishery of interest. Second, we conducted a stock assessment using the chosen method, with LEK informing unknown model parameters and filling in data gaps. Finally, simple sensitivity analyses were run to test how uncertain or unknown parameters (including those estimated by LEK) affected the estimates of stock status (Figure 1).

METHODS

Study species

The Crevalle Jack is a large pelagic fish species with a native range spanning the east coasts of North America and Central America (Smith-Vaniz and Carpenter 2007). In Florida, the Crevalle Jack is a popular sport fish species that is highly valued by recreational anglers for its strength and speed (Gervasi et al. 2022b). Crevalle Jack are also captured in commercial fisheries (mainly as bycatch) throughout the state but are unregulated and understudied. The Florida Fish and Wildlife Conservation Commission makes management decisions for all fisheries within state waters; such decisions include setting gear restrictions, size limits, and bag limits and instituting closed seasons. State waters extend 4.83 km (3 mi) from shore on the east coast of Florida and 16.09 km (10 mi) from shore on Florida's west coast (Figure 2). Several species (including the Crevalle Jack) are listed as unregulated species in the state, which means that they have no specific regulations regarding gear restrictions, size limits, bag limits, or closed seasons. Florida does, however, have a default limit of two fish or 45.36 kg (100 lb) per person per day (whichever is greater) for all unregulated species (Florida Fish and Wildlife Conservation Commission [FWC] 2021). Crevalle Jack are found in a variety of habitats, including offshore reefs (Smith-Vaniz and Carpenter 2007). Hence, they are also captured in federal waters within the U.S. Exclusive Economic Zone (Figure 2). Federal fisheries in the region are managed by the National Oceanic and Atmospheric Administration (NOAA) Fisheries, the South Atlantic Fishery Management Council, and the Gulf of Mexico Fishery Management Council (NOAA–Fisheries 2021). Crevalle Jack are not currently managed as a federal species, so there are no restrictions on their harvest in federal waters.

Due to the Crevalle Jack's unregulated status, limited research has been done in the region to assess the species' life history (e.g., stock boundaries and migration patterns) or fishery trends (e.g., trends in length and age composition; McBride and McKown 2000; Gervasi et al. 2022b; Jefferson et al. 2022). Therefore, the Crevalle Jack fishery in Florida can be considered data limited, a term that generally describes situations in which the data required to support a fully integrated stock assessment model (including catch time series, indices of abundance, length and age composition, and life history parameters) are missing (Cope et al. 2023). In the Florida Keys, recreational fishing guides have observed a concerning decline in Crevalle Jack catch rates, which is supported by available fisheries-dependent data (Gervasi et al. 2022b). This decline prompts a pressing need for assessment of the species in Florida and possible future management action. The Florida Crevalle Jack fishery is therefore an ideal candidate for applying the methodology outlined herein.

The FishPath tool

In this study, the FishPath assessment questionnaire was filled out by the lead author for Florida Crevalle Jack by using information compiled from various sources (Table 1; File S1 available in the Supplement separately online). When possible, published literature and existing fisheries-dependent data were used to answer the multiple-choice questions. However, some questions could not be answered without additional research or data collection. In these instances, LEK was used to fill in the knowledge gaps. Fortunately, LEK data concerning the Crevalle Jack population in south Florida were already available from interviews conducted in 2019 with expert recreational fishing guides in the Florida Keys (Gervasi et al. 2022b). Most of the remaining FishPath questions could be answered using this existing LEK data set. Interview methods are

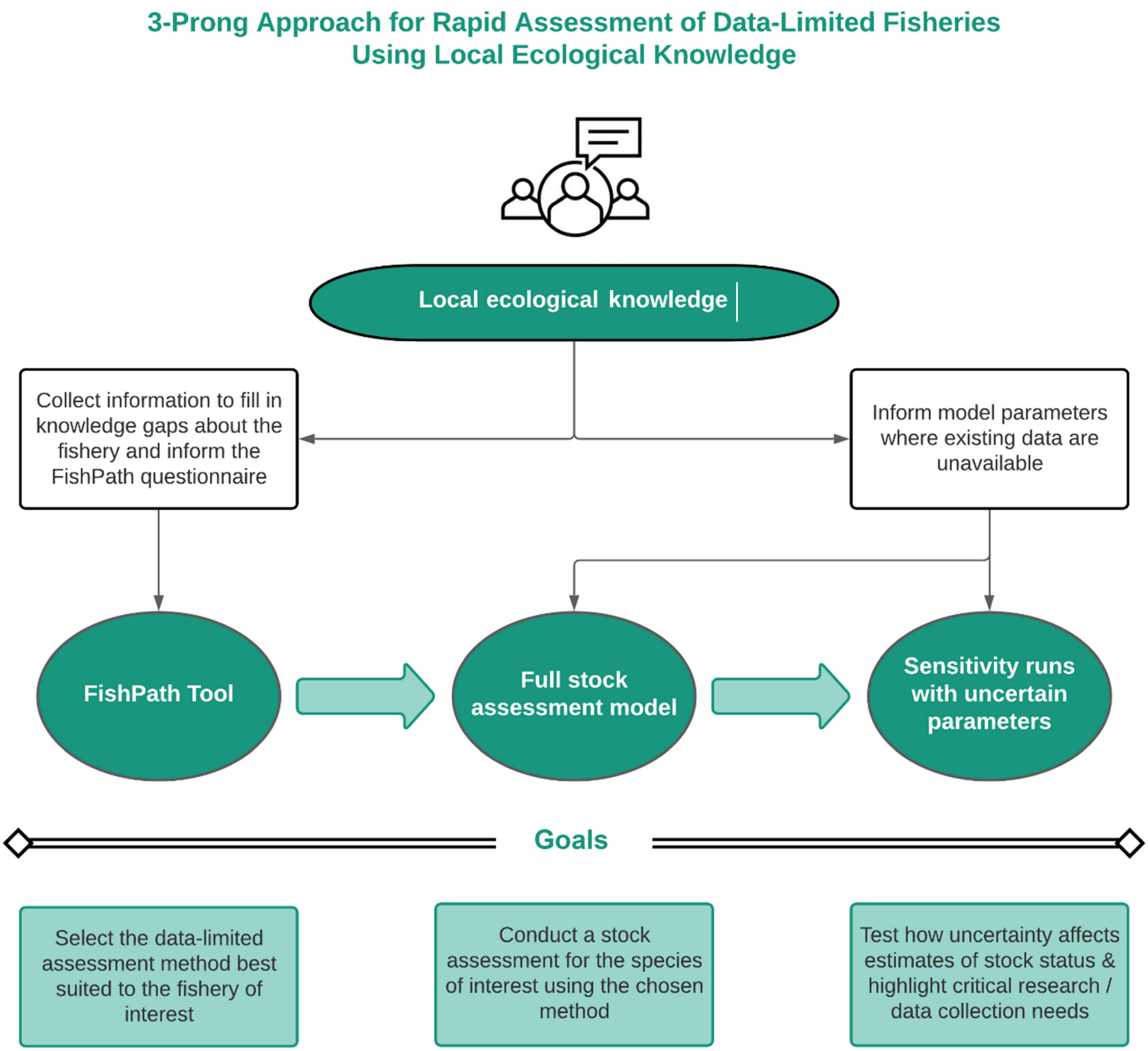

FIGURE 1 Framework presented in this paper for conducting rapid initial assessments of unregulated and data-limited fisheries using a three-prong approach, with angler local ecological knowledge (LEK) permeating each step. First, data from LEK and other sources are used to fill out the FishPath assessment questionnaire and choose a data-limited stock assessment method suited to the fishery of interest. Second, a stock assessment is conducted for the species of interest by using the chosen method, with LEK informing unknown model parameters. Finally, simple sensitivity analyses are run to test how uncertain or unknown parameters (including those estimated by LEK) affect estimates of stock status. Highly influential parameters highlight critical future research needs.

described in depth by Gervasi et al. (2022b). Briefly, key informant interviews were conducted with 18 veteran charter-for-hire captains in the Florida Keys, where a decline in Crevalle Jack catches had been observed. Captains were asked a series of open-ended questions to guide the conversations: (1) "What is your general background and experience fishing and guiding?"; (2) "What do you know about Crevalle Jack?"; (3) "Have you noticed any changes in Crevalle Jack fishing over time?"; and (4) "Is fishing for Crevalle Jack important to you?" More specific follow-up questions were asked as needed, with the goal of capturing perceptions about the Crevalle Jack fishery and stock status as well as gaining an understanding of how and why stakeholders interact with the species (Figure S1 available in the Supplement separately online). Common perceptions of the fish population and fishery among anglers were summarized and used in the current study to fill out the FishPath questionnaire. All protocols for human subject research were approved by Florida International University's (FIU) Institutional Review Board, and all participants gave consent before being interviewed. Any remaining FishPath questions were answered by consulting

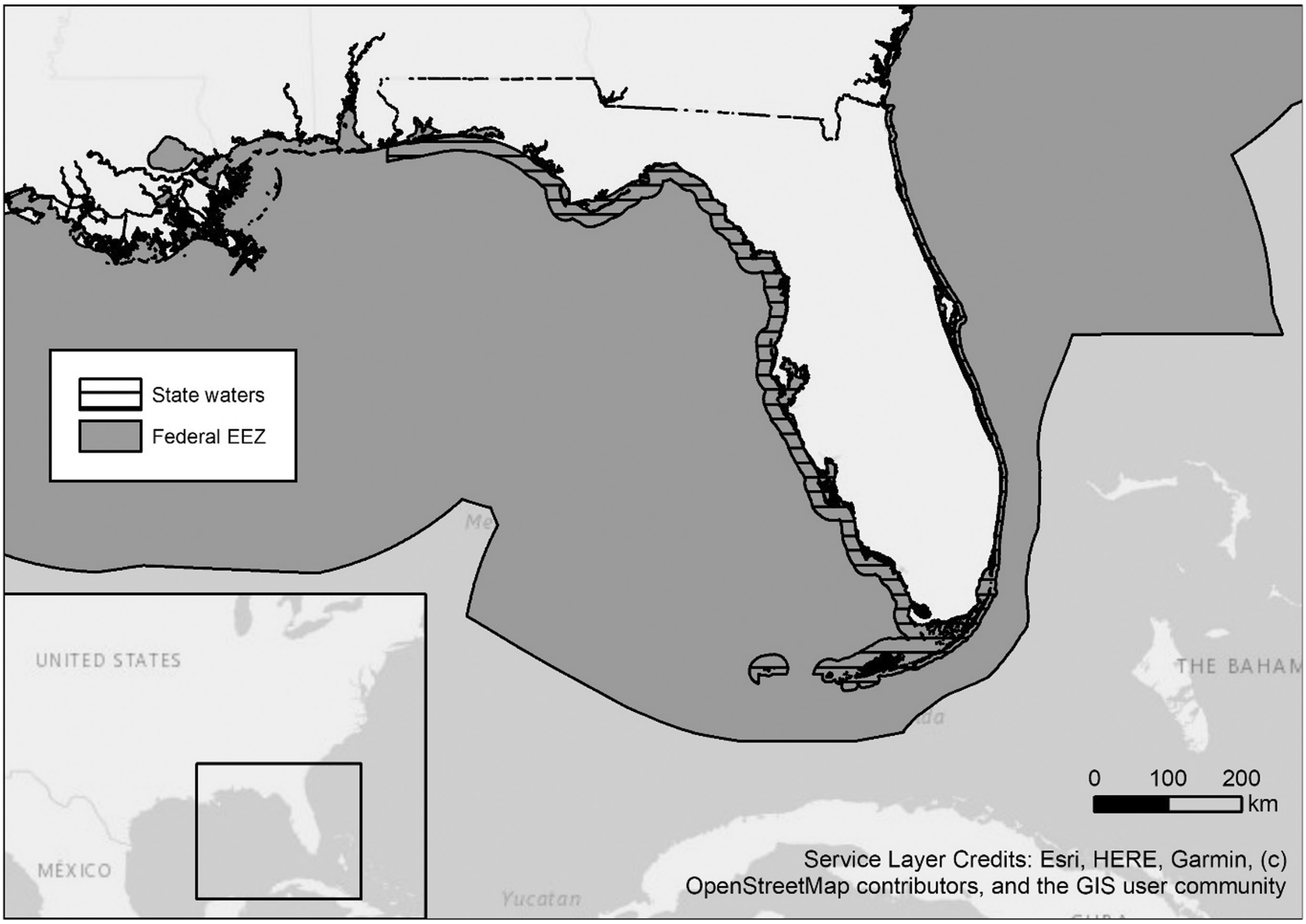

FIGURE 2 Map of the study area in Florida, highlighting state water boundaries (4.83 km [3 mi] from shore on the east coast; 16.09 km [10 mi] from shore on the west coast) and the U.S. federal Exclusive Economic Zone (EEZ). The inset map highlights the study area in the southeastern United States. (State boundary shapefile was downloaded from FWC-Fish and Wildlife Research Institute 2007; federal EEZ shapefile was downloaded from Flanders Marine Institute 2019).

with the previously interviewed anglers and asking them to address the specific question.

The FishPath assessment questionnaire includes five categories of questions concerning the biology and life history of the species, data availability, governance, management, and operational characteristics. All available fisheries-dependent and fisheries-independent surveys that operate in Florida and regularly encounter Crevalle Jack were compiled to answer questions about data availability. Previous literature on the Crevalle Jack was used to inform questions about biology and life history (Smith-Vaniz and Carpenter 2007; Caiafa et al. 2011; Alfaro-Martínez et al. 2016; Jefferson et al. 2022). Common perceptions of the Crevalle Jack fishery from LEK interviews were summarized to generate a LEK data set for use in the FishPath assessment and the resulting selected model. The LEK data included information on relative stock status and the nature of fishery operations, including targeting, species uses, and fishing areas (Table 1; File S1).

The FishPath assessment tool does not rank the possible assessment methods, but it does filter out any methods for which the minimum data requirements or criteria are not met based on the questionnaire responses (Dowling et al. 2016). The tool also displays "traffic light" caveats that highlight each possible assessment method's major assumptions and data requirements as they relate to the fishery of interest. Caveats that are red are important assumptions that might not be met according to the questionnaire responses, so those methods should be used with extreme caution. All options with one or more red caveats were eliminated. FishPath also ranks each method by assessment tier (i.e., model complexity), with tiers ranging from simple, extremely data-limited methods (pre-assessment) to robust methods that require additional data (high tier). Options were sorted by assessment tier, and the highest tier options that remained after the elimination of options with red caveats were considered the best options for assessment of the Florida Crevalle Jack fishery (File S2).

TABLE 1 Available data and information about Crevalle Jack that were compiled and used to fill out the FishPath assessment questionnaire. FL, fork length; FWC, Fish and Wildlife Conservation Commission; GOM, Gulf of Mexico; LEK, local ecological knowledge; MRIP, Marine Recreational Information Program; NOAA, National Oceanic and Atmospheric Administration.

Data category	Variable	Estimate	Source
Biological and life history attributes	Minimum size at maturity	~40 cm FL	Caiafa et al. 2011 (Colombia)
	Maximum size	~100 cm FL	Jefferson et al. 2022 (GOM)
	Longevity	20 years	Jefferson et al. 2022 (GOM)
	Size at 50% maturity	62.6 cm FL (males); 66.2 cm FL (females)	Caiafa et al. 2011 (Colombia)
	Length–weight	$\log_{10}(\text{weight}) = -16.47 + 2.79 \cdot \log_{10}(\text{FL})$	Jefferson et al. 2022 (GOM)
	Growth rate (females)	$l_{t(\mathrm{F})} = 903.04\left[1 - \mathrm{e}^{-0.39(t-0.73)}\right]$	Jefferson et al. 2022 (GOM)
	Growth rate (males)	$l_{t(\mathrm{M})} = 887.16\left[1 - \mathrm{e}^{-0.39(t-0.73)}\right]$	Jefferson et al. 2022 (GOM)
	Length–fecundity	Relationship not significant	Alfaro-Martínez et al. 2016 (Colombia)
	Natural mortality rate	1.12 year^{-1}	Caiafa et al. 2011 (Colombia)
	Fishing mortality rate	0.63 year^{-1}	Caiafa et al. 2011 (Colombia)
	Stock boundaries	Single Florida stock	Preliminary acoustic telemetry data (C. L. Gervasi, unpublished data)
	Habitat use	Variable	Smith-Vaniz and Carpenter 2007; fishing guide LEK (Gervasi et al. 2022b)
Available data	Commercial landings	Available for the entire state of Florida from 1950 to 2021	Florida FWC
	Commercial effort	Number of landings receipts can be a proxy	Florida FWC
	Commercial size composition	Size/age composition data are not available	N/A
	Recreational landings	Available for the entire state of Florida from 1981 to 2021	NOAA MRIP dockside survey
	Recreational effort	Estimated via household mail survey	NOAA Fishing Effort Survey
	Recreational size composition	Subset of size data available for landed fish	NOAA MRIP dockside survey
	Fisheries independent	No suitable surveys in the region	N/A
Fishery attributes and management	Stock status	High in the 1980s; gradual decline since	Fishing guide LEK (Gervasi et al. 2022b)
	Commercial gear used	Cast net, hook and line, gill net = 67%	Commercial data
	Recreational gear used	Hook and line = 99%	Recreational data
	Discards	>80% of recreational catch discarded annually	Recreational data
	Discard mortality rate	10%	Fishing guide LEK (guides were consulted specifically for this study)
	Targeting	Opportunistic; some targeting	Fishing guide LEK (Gervasi et al. 2022b)
	Management	Unregulated	Florida FWC

Model inputs

Model priors

All methods pertaining to model inputs were contingent upon the results of using the FishPath tool. We chose the highest tier type of assessment recommended: the CMSY–BSM model (CMSY = catch–maximum sustainable yield [MSY]; BSM = Bayesian state-space implementation of the Schaefer surplus production model; Froese et al. 2017, 2019). Details of this selection are presented in the Results (Assessment model selection). The CMSY–BSM model requires a time series of total fishery removals (hereafter referred to as "catch"); priors for resilience (defined as species productivity or resilience to fishing); priors for biomass B relative to carrying capacity k (i.e., B/k) at the beginning, middle, and end of the catch time series; and an optional biomass time series. For Crevalle Jack, prior estimates for the maximum intrinsic rate of population increase (r) were extracted from the "Estimates based on models" section of FishBase using a species resilience category of "medium" (Smith-Vaniz et al. 1990; Froese and Pauly 2021). A prior range for k was derived by the model from the maximum catch. The available time series of recreational fishing effort used to develop an index of Crevalle Jack abundance did not cover the entire time series of fishing effort. Therefore, fishing guide LEK was used to determine prior range categories for B/k at the beginning, middle, and end of the time series based on guide estimates of how population abundance has changed over time. The default B/k ranges corresponding to these categories from Froese et al. (2017) were used in the model (Table 2).

Catch time series

The following formula was used to create a time series of total fishery removals (catch time series) for Crevalle Jack in Florida from 1950 to 2021:

$$\text{Commercial landings}_t + \text{recreational landings}_t + \left(\text{recreational discards}_t \times \text{discard mortality}\right), \tag{1}$$

where t is year and "discard mortality" is an estimated discard mortality rate (Figure 3). Discard mortality occurs when fish are caught and released alive but die after release due to injuries suffered from the angling encounter or due to an increased susceptibility to predation (Rudershausen et al. 2007). The discard mortality rate is defined as the proportion of individuals that suffer from discard mortality and therefore can be considered a component of fishery removals. Previously interviewed fishing guides were asked to estimate a discard mortality rate. The average of their responses was used as the base discard mortality rate in this study. Preliminary acoustic telemetry research has revealed population connectivity of Crevalle Jack throughout the state of Florida (C. L. Gervasi, unpublished data). Hence, we assumed that state-level boundaries provided a reasonable approximation of the stock unit. All catch data were thus collected for the entire state. Commercial landings data were obtained from the National Marine Fisheries Service's Accumulated Landings System (NOAA 2021a). Landings data for Crevalle Jack were downloaded for all of Florida from the beginning of the time series (1950) to the last available year (2021). Crevalle Jack recreational landings (fish that were brought back to shore) and discards (fish that were caught and released either dead or alive) for the state of Florida were downloaded from the NOAA Fisheries' Marine Recreational Information Program (MRIP) online query tool for the period of record from 1981 to 2021 (NOAA 2021b). Additional details on how commercial and recreational landings data were obtained and how the discard mortality rate was calculated are available in File S3.

Abundance time series

No fisheries-independent surveys are operating in the region that regularly encounter adult Crevalle Jack, but relative abundance trends can be inferred from catch-per-unit-effort (CPUE) data (Maunder and Punt 2004). For the purposes of this study, we used the MRIP CPUE data subset for Florida to create an index of Crevalle Jack abundance for the entire state. Numerous factors besides stock abundance can influence fishery catch rates (e.g., spatial, temporal, and environmental variability), so we standardized the CPUE data for Crevalle Jack in Florida using generalized linear models (GLMs; Matthews 2014). Specifically, a delta-lognormal GLM approach (Lo et al. 1992) was applied, with the following categorical factors included in the model: year (1991–2021), season (spring: March, April, and May; summer: June, July, and August; fall: September, October, and November; winter: December, January, and February), fishing mode (shore, charter, or private), and day (weekday or weekend). Additional details on standardization methods can be found in Gervasi et al. (2022b). Filtered and cleaned MRIP data included 240,712 trips from 1991 to 2021 (31 years). Of these trips, Crevalle Jack were caught during 38,825 trips (16%). Based on model selection via backward stepwise regression and deviance tables, the final model for the proportion positive GLM included year and season as fixed factors, and the final model for the positive trip GLM included year, season, and fishing mode as fixed factors (Tables S1 and S2 available in the Supplement separately online).

Sensitivity analyses

To examine the sensitivity of our Crevalle Jack CMSY–BSM model to unknown or poorly estimated parameters, we conducted a series of sensitivity analyses and compared stock status as well as biomass and exploitation trends to those generated from our base model (Table 2). We explored seven different scenarios that tested the model sensitivity to potential uncertainty by varying the estimated discard mortality rate instead of using the LEK-derived value (analysis 1); ignoring the CPUE data and using only the CMSY model without the surplus production model

TABLE 2 Model inputs and parameters for the base Crevalle Jack stock assessment model and for the seven sensitivity analyses. *B*, biomass; CMSY, catch–maximum sustainable yield; CPUE, catch per unit effort; ENP, Everglades National Park; *k*, carrying capacity; LEK, local ecological knowledge; MRIP, Marine Recreational Information Program; PSE, proportional standard error; *r*, intrinsic population growth rate.

Variable	Base model	Sensitivity analysis 1	Sensitivity analysis 2	Sensitivity analysis 3	Sensitivity analysis 4	Sensitivity analysis 5	Sensitivity analysis 6	Sensitivity analysis 7
Description of sensitivity analysis	–	Varying the discard mortality rate	Excluding CPUE data (CMSY model only)	Excluding historical catch	Excluding high-PSE points in MRIP data	Using uninformed priors versus LEK priors	Using an alternative abundance data set	Accounting for effort creep
Catch start year	1950	1950	1950	**1981**	1950	1950	1950	1950
Catch end year	2021	2021	2021	2021	2021	2021	**2020**	2021
Abundance time series	MRIP	MRIP	**N/A**	MRIP	MRIP	MRIP	**ENP**	MRIP
Resilience category[a]	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium
Prior ranges for r[a]	0.35–0.80	0.35–0.80	0.35–0.80	0.35–0.80	0.35–0.80	0.35–0.80	0.35–0.80	0.35–0.80
Relative biomass category (beginning)[b]	Nearly unexploited	Nearly unexploited	Nearly unexploited	**Low depletion**	Nearly unexploited	**Uninformed**	Nearly unexploited	Nearly unexploited
Prior range for B/k (beginning)[b]	0.75–1.00	0.75–1.00	0.75–1.00	**0.4–0.8**	0.75–1.00	**0.01–1.00**	0.75–1.00	0.75–1.00
Relative biomass category (middle)[b]	Low depletion	Low depletion	Low depletion	**Low depletion**	Low depletion	**Uninformed**	Low depletion	Low depletion
Prior range for B/k (middle)[b]	0.4–0.8	0.4–0.8	0.4–0.8	0.4–0.8	0.4–0.8	**N/A**	0.4–0.8	0.4–0.8
Relative biomass category (end)[b]	Medium depletion	Medium depletion	Medium depletion	Medium depletion	Medium depletion	**Uninformed**	Medium depletion	Medium depletion
Prior range for B/k (end)[b]	0.2–0.6	0.2–0.6	0.2–0.6	0.2–0.6	0.2–0.6	**0.01–1.00**	0.2–0.6	0.2–0.6
Effort creep (%)	0	0	0	0	0	0	0	**2**
Discard mortality rate[b]	0.1	**Range from 0.0 to 0.5**	0.1	0.1	0.1	0.1	0.1	0.1

Note: Changes to model inputs and parameters from the base model for each sensitivity analysis are shown in bold.

[a]Derived from FishBase (Froese and Pauly 2021).

[b]Values were derived from angler LEK except in sensitivity analysis 1 (discard mortality not informed by LEK) and sensitivity analysis 5 (priors not informed by LEK).

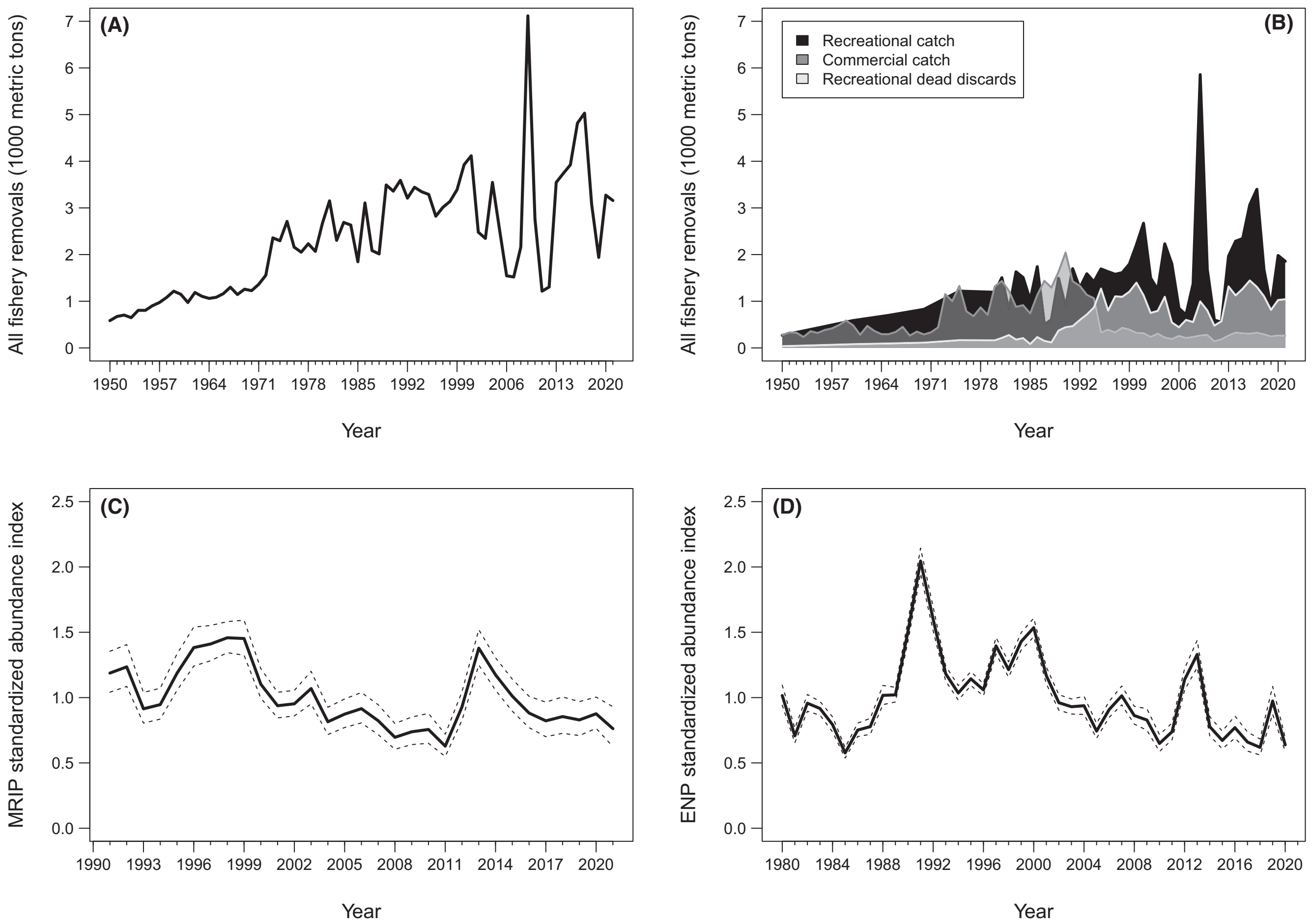

FIGURE 3 (A) Time series of Crevalle Jack catch (total fishery removals; thousands of metric tons) used in the initial CMSY–BSM model (defined in Methods) and (B) breakdown of fishery removals by fleet. In panels A and B, an estimated discard mortality rate of 10% was applied to the recreational live releases to obtain an estimate of recreational dead discards. (C) The Marine Recreational Information Program (MRIP) standardized abundance index, which was used in the base model and sensitivity analyses 1, 3, 4, 5, and 7 as the biomass time series, is shown. (D) The Everglades National Park (ENP) standardized abundance index, which was used in sensitivity analysis 6 as the biomass time series, is presented. Dashed lines in panels C and D are 95% confidence intervals.

(BSM; analysis 2); subsetting the catch time series to begin at 1981 (the beginning of the MRIP time series), thereby ignoring the estimated historic recreational catch and historic commercial catch (analysis 3); removing potential outlier data points from the MRIP time series with high proportional standard error (PSE; >50%) and replacing them with interpolated values based on adjacent years (analysis 4); using uninformed biomass priors instead of the LEK-derived priors (analysis 5); using an alternative standardized CPUE time series (developed by Gervasi et al. 2022b) based on the Everglades National Park (ENP) creel survey (Osborne et al. 2006), updated to include data from 2020, as an index of abundance (analysis 6); and, finally, incorporating effort creep (2% annual increase in catchability) into the model based on stock assessments for other fish species in the Gulf of Mexico (Thorson and Berkson 2010; analysis 7). All analyses were conducted in R version 4.2.3 (R Core Team 2023).

RESULTS

Assessment model selection

After eliminating FishPath assessment options with red caveat outputs and sorting by assessment tier (highest to lowest), two options with the high-tier designation remained (File S2). These options were production models (e.g., Schaefer, Fox, and Pella–Tomlinson models; Hilborn and Walters 1992) and the *qR* (catchability–recruits) method (McGarvey et al. 1997). Production models require a continuous time series of fishery removals, and the two major parameters in the models are *r* and *k*, which are used to estimate MSY. Production models additionally require at least one index of abundance. The *qR* method uses time series of catch by weight and in numbers, an estimate of natural mortality (*M*), and an average of weight at age to estimate biomass, catchability, exploitation rate, and yearly recruitment. We thoroughly reviewed both methods in the scientific literature to select the best option for assessment of the Crevalle Jack fishery. The qR method was eliminated as an option because catch by numbers is not recorded for Crevalle Jack in the commercial fishery and would have had to be estimated. Furthermore, M for Crevalle Jack in Florida is currently unknown and cannot be estimated with any certainty (available estimates are from areas outside the United States). The production model required fewer inputs of uncertain parameters.

We chose to apply the CMSY–BSM method created by Froese et al. (2017) and further updated by Froese et al. (2019). This method includes both the BSM and the CMSY model, which is similar to a production model but does not require a time series of abundance. The CMSY–BSM model was chosen because it estimates biomass, exploitation rate, MSY, and related fisheries reference points, with the only data requirements being catch and productivity (Froese et al. 2017). An extensive time series of fishery removals in Florida is available for Crevalle Jack, and productivity information is available from previous research. The CMSY model is an updated version of the catch–MSY method originally proposed by Martell and Froese (2013), which reviews of data-limited assessment methods have found to be a promising approach (International Council for the Exploration of the Sea 2014; Rosenberg et al. 2014). The predictions of the CMSY–BSM method have been validated against 48 simulated stocks and evaluated against 159 fully or partially assessed real stocks, and estimates of r, k, and MSY were not significantly different from the actual values for 90% of simulated stocks and 76% of real stocks (Froese et al. 2017). Furthermore, a detailed user manual and R code (last updated in 2019) are available for download (Froese et al. 2017), making the method easily accessible and reproducible. The updated version of the model (CMSY+ and BSM; Froese et al. 2019) was used in this study, but for simplicity we refer to it as the "CMSY–BSM model" hereafter.

Base model run

Our base stock assessment model run for Crevalle Jack revealed a gradual increase in exploitation and a corresponding gradual decline in stock size from 1950 to 2021 (Figure 4). Total catch was below MSY (3155 metric tons year^{-1}) from 1950 to 1988 and fluctuated around MSY for the remaining years, with catch being above MSY for 19 of the 33 years from 1989 to 2021. Several definitions for the terms "overfished" and "overfishing" exist in the literature, but generally a stock is considered "overfished" if B is below B_{MSY} by some degree and to be undergoing "overfishing" if fishing mortality F is above F_{MSY} by some degree (Froese and Proelss 2012, 2013; Langseth et al. 2019; Hilborn 2020). For the purposes of this study, we referred to the Crevalle Jack stock as overfished whenever model-estimated B was below B_{MSY} ($B/B_{\text{MSY}} < 1$) and as undergoing overfishing whenever model-estimated F was above F_{MSY} ($F/F_{\text{MSY}} > 1$). These definitions do not account for fluctuations around the thresholds due to inherent variability. A stock that is managed at MSY could be expected to fluctuate around B_{MSY}. However, the Crevalle Jack stock is unmanaged and stock size has continually declined, suggesting that the stock is not being sustainably harvested. According to the base model, F was above F_{MSY} for 14 of the 22 years since 2000, revealing that overfishing has been occurring regularly since 2000. Biomass was above B_{MSY} from 1950 to 2002 but was below B_{MSY} for every year from 2003 to 2011 and from 2017 to 2021, with B being the lowest in 2019. It appears that high levels of catch above MSY starting in 1989 led to overfishing beginning in 2000 and led to the stock becoming overfished starting in 2003. According to the assessment, the current status of the stock is overfished and undergoing overfishing: the estimated F_{2021}/F_{MSY} was 1.12, and the estimated B_{2021}/B_{MSY} was 0.88 (Table 3).

Model sensitivity runs

Our first set of sensitivity runs examined the impact of selecting various discard mortality rates for the recreational fishery by running a series of models with recreational discard mortality ranging from 0% to 50% in 10% increments (Table 2; Figure 5). Discard mortality rates above 50% were not considered because they were deemed highly unrealistic by fishing guides (who were asked specifically about discard mortality for this study). For all model runs, exploitation in 2021 (F_{2021}/F_{MSY}) was similar, ranging from 1.1 to 1.3. Stock size in 2021 (B_{2021}/B_{MSY}) generally increased with an increase in discard mortality, ranging from 0.81 at 0% mortality to 0.95 at 50% mortality (Table 3; Figure 5B). Regardless of the discard mortality rate used, the models revealed the same trend of gradually increasing exploitation and gradually decreasing stock size over time. The status of the stock in 2021 was overfished and undergoing overfishing regardless of the mortality rate used. The choice of discard mortality rate had little effect on the estimate of r in the model but greatly affected the estimate of k, with k increasing linearly as the discard mortality rate increased (Figure 5C,D). Since k directly affects the MSY estimate, the estimated MSY also increased linearly as the discard mortality rate increased. At a discard mortality rate of 0%, estimated MSY was 2320 metric tons; at a discard mortality rate of 50%, estimated MSY was 6790 metric tons (Table 3).

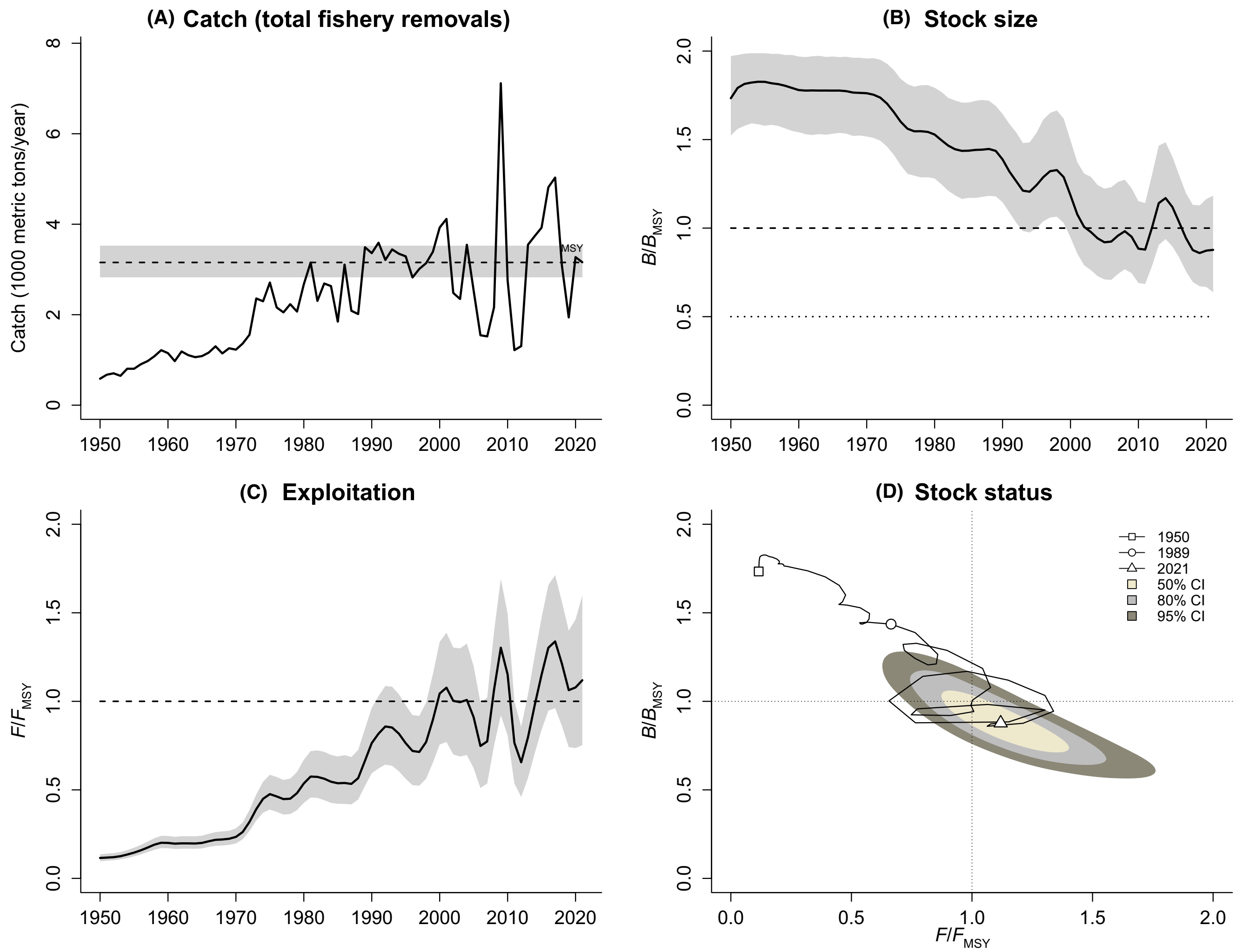

FIGURE 4 Summary of information relevant for management of Florida Crevalle Jack from the base CMSY–BSM model (defined in Methods): (A) catches (total fishery removals; thousands of metric tons per year) relative to maximum sustainable yield (MSY; dashed line); (B) development of predicted relative total biomass (B/B_{MSY}); (C) relative exploitation (fishing mortality F/F_{MSY}); and (D) trajectory of relative stock size (B/B_{MSY}) as a function of fishing pressure (F/F_{MSY}). Gray shading in panels A–C denotes 95% confidence limits for MSY, relative biomass, and relative exploitation, respectively. The oval shape around the assessment of the final year triangle indicates uncertainty (yellow = 50% confidence interval [CI]; gray = 80% CI; dark gray = 95% CI).

The remaining sensitivity analyses (2–7) examined the effects of excluding the abundance time series (using the CMSY model only), excluding historical catch data, excluding high-PSE data points from the MRIP data, using uniformed priors versus LEK priors, using an alternative abundance data set, and accounting for an effort creep of 2% per year (Table 2). Except for sensitivity analyses 5 (uninformed priors) and 6 (alternative abundance data set), each of the sensitivity model runs revealed the same pattern of gradually decreasing stock size over time (Figure 6A). Furthermore, estimated B_{2021} was below B_{MSY} for all models and estimated F_{2021} was above F_{MSY} for all models except in sensitivity analyses 2 and 5 (Table 3; Figure 6B). For all model runs, B was below B_{MSY} and F was above F_{MSY} for at least 4 years of the 71-year time series (Figures S3–S8).

Excluding CPUE data (analysis 2) and using uninformed biomass priors (analysis 5) led to the most optimistic depictions of current stock status. For analysis 2, excluding an index of abundance led to a trajectory of stock status over time similar to that from the base model, with stock size gradually declining while exploitation gradually increased over time (Figure S3). For analysis 5, starting and ending biomass priors (1950 and 2021) were set to a wide range (0.01–1.00), which told the model that we had no information about stock status at the beginning or the end of the time series (Froese et al. 2019). The intermediate biomass level was set to "NA," which allowed the model to estimate it from maximum or minimum catch according to some simple rules (Froese et al. 2017). This version of the model showed a trajectory of gradually increasing exploitation over time, which was the same as the

TABLE 3 Estimated management reference points from the base Crevalle Jack stock assessment model and from the seven sensitivity analyses. B_{2021}, biomass in 2021; B_{MSY}, biomass at maximum sustainable yield (MSY); CMSY, catch–MSY; CPUE, catch per unit effort; ENP, Everglades National Park; F_{2021}, fishing mortality in 2021; F_{MSY}, fishing mortality at MSY; k, carrying capacity; LEK, local ecological knowledge; MRIP, Marine Recreational Information Program; PSE, proportional standard error; r, intrinsic population growth rate.

Variable	Base model	Sensitivity analysis 1	Sensitivity analysis 2	Sensitivity analysis 3	Sensitivity analysis 4	Sensitivity analysis 5	Sensitivity analysis 6	Sensitivity analysis 7
Description of sensitivity analysis	–	Varying the discard mortality rate	Excluding CPUE data (CMSY model only)	Excluding historical catch	Excluding high-PSE points in MRIP data	Using uninformed priors versus LEK priors	Using an alternative abundance data set	Accounting for effort creep
r (year^{-1})	0.55	0.50–0.55	0.55	0.55	0.49	0.46	0.50	0.45
k (1000 metric tons)	22.9	17.8–54.2	22.9	23.4	24.3	29.0	25.1	27.8
MSY (1000 metric tons)	3.15	2.32–6.79	3.44	3.21	2.96	3.31	3.14	3.11
F_{MSY} (year^{-1})	0.28	0.25–0.28	0.31	0.28	0.24	0.23	0.25	1.81
B_{MSY} (1000 metric tons)	11.4	8.89–27.1	11.3	11.7	12.2	14.5	12.6	13.9
F_{2021}/F_{MSY}	1.12	1.10–1.13	0.93	1.10	1.32	0.95	1.04	1.81
B_{2021}/B_{MSY}	0.88	0.81–0.95	0.99	0.88	0.79	0.99	0.87	0.55
F_{2021} (year^{-1})	0.31	0.28–0.31	0.28	0.30	0.32	0.22	0.26	0.41

base model's trajectory, but it estimated that stock size in 1950 was below B_{MSY}, rapidly increased to high levels in the 1960s, and then gradually declined (Figure S6). The most important difference between these models and the base model was that excluding CPUE data and using uninformed priors painted a more optimistic picture of stock status, with the stock in 2021 still just below B_{MSY} but also below F_{MSY}, suggesting that overfishing is not currently occurring (Table 3; Figure 6B). Additionally, exploitation was above F_{MSY} for only a handful of years during the 71-year time series.

The most pessimistic models were sensitivity analyses 4 (excluding high-PSE data points) and 7 (including 2% effort creep). Two data points had PSEs above 50%: 1986 and 2009. The catch estimate for 1986 (3107 metric tons) was similar to the average of the time series. However, the catch estimate for 2009 (7116 metric tons) was anomalously high compared to the rest of the time series, whereas the mean catch before 2009 was 2078 metric tons (Figure 3B). Removing the 2009 data point and replacing it with an interpolated value brought the total catch for 2009 down to 2787 metric tons. The decrease in total catch for 2009 had a negative effect on r, with estimated r decreasing from 0.55 to 0.49. This resulted in a lower estimated MSY and F_{MSY} and a more pessimistic stock status, with the stock being more severely overfished and undergoing more severe overfishing in 2021 (Table 3; Figure 6). Based on this model, the Crevalle Jack stock was undergoing overfishing for 16 years of the 71-year time series (Figure S5). Effort creep is defined as some change in catchability or nominal effort in a fishery over time due to technological advancements (Palomares and Pauly 2019), such as major improvements in gear design, fish-finding devices, or vessel capabilities, all of which increase efficiency and therefore impact fishing mortality. The CMSY–BSM model allows the user to specify a linear annual increase in catchability, which results in a decrease in the CPUE index considered by the model. For this sensitivity analysis, a 2% linear increase in catchability was applied to the MRIP standardized abundance index based on previous stock assessments in the region (Thorson and Berkson 2010). Although effort creep did not impact the trajectory of stock status and exploitation over time, it led to a much steeper decline in stock status since the early 2000s than the base model (Figure S8). Furthermore, estimated B was below B_{MSY} for every year since 2003, with B/B_{MSY} almost as low as 0.5 in 2021, suggestive of a severely overfished stock.

Using an alternative abundance time series (analysis 6) had little effect on estimated management reference points. However, the ENP time series went back farther in time than the MRIP time series, and the trajectory of stock status over time differed slightly between the two

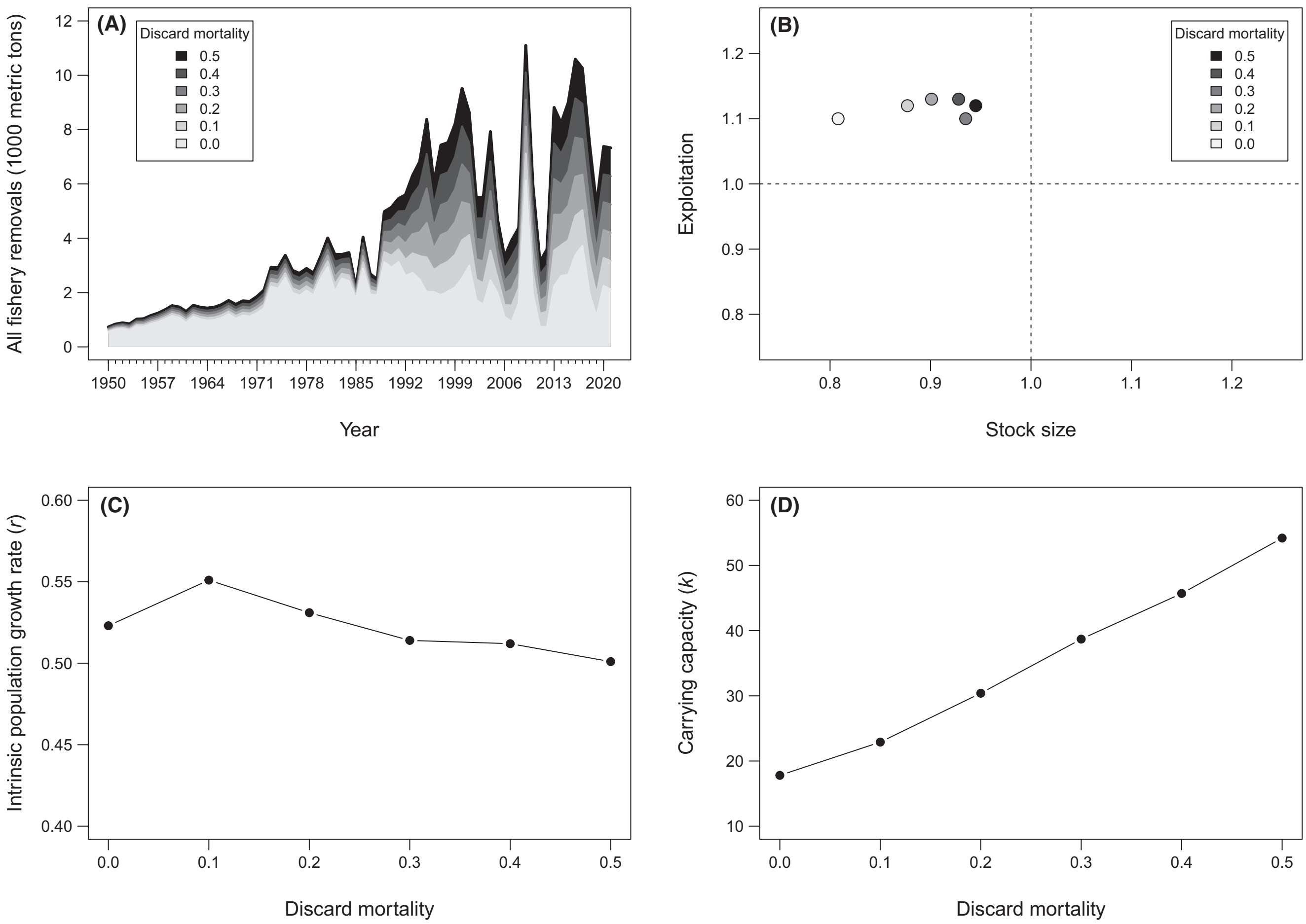

FIGURE 5 Results of the sensitivity analysis examining recreational discard mortality (sensitivity analysis 1): (A) time series of catch (thousands of metric tons) with discard mortality set at 0.0, 0.1, 0.2, 0.3, 0.4, or 0.5; (B) time series of exploitation (fishing mortality F/F_{MSY}, where MSY = maximum sustainable yield) on the *y*-axis and stock size (biomass B/B_{MSY}) on the *x*-axis in the final year (2021) for the range of discard mortality rates assessed (0.0–0.5); (C) effect of discard mortality rate on estimated intrinsic population growth rate (r); and (D) effect of discard mortality rate on estimated carrying capacity (k).

models. With the ENP time series as an index of abundance, stock size declined rapidly from 1970 to 1985 before increasing back to historic levels and then gradually declining from 1990 to 2020 in the same fashion as the base model (Figure S7). For the years in which the two abundance time series overlapped (1991–2020), model results were very similar.

DISCUSSION

In this paper, we demonstrated how a variety of data-limited tools, when used in combination, can aid in developing rapid yet robust assessments for data-limited, unregulated fisheries, thus providing a basis for initial management. Our approach took advantage of LEK to inform both model selection and analysis. We used LEK and other existing data sources to fill out the FishPath assessment questionnaire, which is a currently underutilized program that provides a transparent, standardized approach for selecting an appropriate stock assessment model. Local ecological knowledge was then used again to parameterize the chosen model when parameter estimates were unavailable from previous literature, which is the case for many data-limited fisheries. Finally, by identifying unknown and uncertain parameters and running sensitivity analyses to test their effects on estimates of stock status, we developed some clear goals and priorities for future research, which will help to ensure that funding and effort are invested in the greatest needs. The results of applying our framework to assessing stock status of the Crevalle Jack in Florida suggested that B has been below B_{MSY} for 14 of the past 19 years and that the stock is currently undergoing overfishing (with F slightly above F_{MSY}). Any increase in fishing pressure will likely lead to a continued decline in stock size. Fishing guides in the Florida Keys have observed a gradual decline in Crevalle Jack catch rates

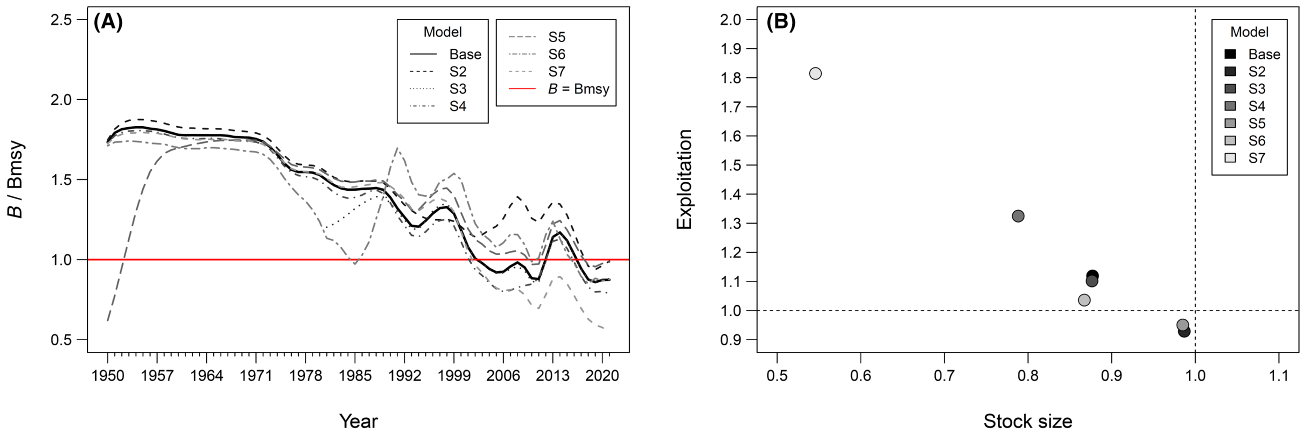

FIGURE 6 Results of the base model in comparison with sensitivity analyses 2–7 (S2–S7): (A) development of predicted relative total biomass (B/B_{MSY}, where MSY = maximum sustainable yield) for each model run; and (B) time series of exploitation (fishing mortality F/F_{MSY}) on the *y*-axis and stock size (B/B_{MSY}) on the *x*-axis in the final year (2021) for each model run.

beginning as early as 1985, with very low catch rates observed since the early 2000s (Gervasi et al. 2022b). Our stock assessment results align with the timing of this observation and further highlight the need to develop a management plan for this important fishery.

Crevalle Jack stock status and trends

Our base CMSY–BSM model revealed that the catch of Florida Crevalle Jack has been at or above MSY almost every year since 1989, with several years of overfishing occurring, and that the stock has been in an overfished state during almost every year since 2003. Stock size has been gradually declining over time, while recreational fishing effort appears to be continually increasing. Commercial landings were relatively low throughout the time series compared to recreational landings, and commercial landings dropped considerably in the mid-1990s (coincident with the commercial gill-net ban in Florida; Smith et al. 2003). The increasing recreational fishing effort is somewhat surprising, as fishing guides reported that the Crevalle Jack fishery in the Florida Keys is largely opportunistic and catch and release (Gervasi et al. 2022b). However, in the statewide MRIP data, recreational anglers report which species were primarily targeted on each fishing trip; out of all Florida trips, Crevalle Jack were reported as the 46th most targeted species out of 318 species listed as primary targets. Therefore, the Crevalle Jack is in the top 15% of recreationally targeted species throughout the state.

Studies have shown that recreational landings exceed commercial landings for many fisheries (Coleman et al. 2004; Radford et al. 2018; Lewin et al. 2019; Shertzer et al. 2019), and there is growing evidence that recreational fisheries can be responsible for declines in fish populations and can have other biological impacts (Lewin et al. 2006; Brownscombe et al. 2019). Even in predominantly catch-and-release fisheries, postrelease mortality and sublethal effects on physiology can have substantial impacts on fish populations (Rudershausen et al. 2007; Cooke et al. 2013). Worldwide, the number of recreational anglers (Kearney 2002; Pawson et al. 2008), the magnitude of recreational catches (Coleman et al. 2004; Felizola Freire et al. 2020), and the economic impact of recreational fishing (Arlinghaus et al. 2019) are increasing. Although recreational fisheries provide funding for conservation efforts and connect society with nature, thereby increasing public awareness and appreciation of conservation concerns (Griffiths et al. 2017; Arlinghaus et al. 2019; Brownscombe et al. 2019), these fisheries are prone to high uncertainty, which undermines sustainable management (Shertzer et al. 2019). Appropriate management action that balances the social and ecological dimensions of these fisheries is therefore vital.

In addition to increased fishing effort, other factors may have contributed to the decline and may continue to impact Crevalle Jack populations in the future. During LEK interviews, fishing guides were asked to speculate on potential reasons for the perceived decline in Crevalle Jack catches, and loss of prey was the most commonly mentioned reason (followed by recreational harvest; Gervasi et al. 2022b). Poor water quality, increased predators, and warmer winters were also potential factors mentioned by multiple guides. Research has shown that regional climate variability can lead to changes in the distribution and productivity of fish species (Brander 2007; Lotze et al. 2019). It is therefore possible that climate-induced shifts in prey or predator species have contributed to shifts in Crevalle Jack distributions in the region. Ecosystem-based management efforts in the South Atlantic and Gulf of Mexico could contribute to more holistic management of species such as the Crevalle Jack in the future (e.g., fisheries ecosystem plans; Levin et al. 2018).

The timing and trajectory of Crevalle Jack exploitation match the observations of recreational fishing guides in Florida, some of whom began noticing a decline in Crevalle Jack catch rates as early as 1985 (Gervasi et al. 2022b). Most guides, however, noticed the decline in the early to mid-2000s, which corresponds to the year when stock size began dipping below B_{MSY} (2003). Additionally, guides reported that the decline had been gradual, which again matches the model results (i.e., even for the analysis using uninformed priors, stock size declined gradually from 1970 to 2021). This agreement between fishing guide observations and model results provides confidence in the stock assessment and highlights the benefits of incorporating LEK into fisheries research. Consistency between LEK and other data sources has been observed in many studies (e.g., Poizat and Baran 1997; Aswani and Hamilton 2004; Zukowski et al. 2011; Rehage et al. 2019; Santos et al. 2019; Bourdouxhe et al. 2020), and the use of LEK in fisheries research and management has increased substantially over the years (Beaudreau and Levin 2014). A recent study by Shephard et al. (2021) showed that angler LEK matched stock assessment results for four recreational fisheries in Ireland, further demonstrating that LEK can provide valuable, robust information about fisheries stock status and trends.

Importantly, research and management efforts that rely on stakeholder input and collaboration are most successful in situations of mutual trust and respect, which can be difficult to build and maintain (Thornton and Scheer 2012). In our study, we solicited the aid of experienced recreational fishing guides to fill in knowledge gaps about the Crevalle Jack fishery and to help inform model priors. Studies have shown that fishing guides are ideal research partners, as they have substantial on-the-water experience and a vested interest in fisheries conservation (Kroloff et al. 2019; Adkins 2020; Gervasi et al. 2022a). To ensure continued trust and collaboration, fishing guides were informed of the results of this study and its potential management applications. As demonstrated by Gervasi et al. (2022a), it is important to involve anglers throughout the scientific research process and beyond to maintain trusted partnerships. For cases in which there is general distrust of science and management by key stakeholders, efforts to build trust and maintain relationships are vital to conducting LEK research and fisheries co-management (Thornton and Scheer 2012; Rubert-Nason et al. 2021).

Sensitivity analyses

Fisheries management is commonly based on setting target quotas or catch limits based on fisheries reference points from stock assessments (Newman et al. 2015). Uncertainty in model parameters that greatly affect the estimation of reference points can lead to target setting based on inaccurate estimations of stock status, thus increasing the risk for either overfishing or underutilizing the resource (Dankel et al. 2012; Cadrin et al. 2015; Privitera-Johnson and Punt 2020). Sensitivity analysis is a common approach used by stock assessment scientists to understand aspects of model uncertainty (Privitera-Johnson and Punt 2020). Compared to our initial CMSY–BSM model, none of the sensitivity analyses dramatically altered the overall pattern of exploitation and stock size over time or the estimated current stock status. In all models, exploitation increased over time, with harvest increasing to levels at or above MSY at some point during the time series. Stock size also generally decreased over time, with overfishing occurring in all models, although the number of years for which the stock was in an overfished state varied depending on the model. Exploitation in 2021 was high for all models, with F_{2021}/F_{MSY} ranging from 0.93 to 1.81 (Table 3). All models also showed that the stock in 2021 was overfished ($B_{2021}/B_{MSY} < 1$). This model consistency reveals high model precision and provides some additional confidence in our stock assessment results. However, there still may be unaccounted-for sources of uncertainty (i.e., “unknown unknowns”; Drouineau et al. 2016) that could affect model accuracy.

Despite the consistency in overall trends among model runs, estimated management reference points deviated from the initial model for some of the sensitivity analyses. Changing the discard mortality rate for our first analysis had the greatest effect on reference points, and k increased dramatically with an increase in the discard mortality rate. This change in k led to a substantial impact on estimated MSY and B_{MSY}, which are important values needed to determine fishery quotas. This analysis highlights the importance of estimating an accurate discard mortality rate for fisheries that are predominantly catch and release. When angling effort is high, catch-and-release fishing is often applied as a management solution for reducing angling impacts on important fisheries (Cooke and Schramm 2007). Although catch-and-release fishing can provide many benefits to fisheries when used appropriately (Arlinghaus et al. 2002, 2007), it can also have unintended and unaccounted-for consequences (Cooke et al. 2002; Cooke and Suski 2005). Several studies have shown that angling can have a multitude of physiological effects on fish, resulting in morbidity and mortality after release (Cooke et al. 2002; Campbell et al. 2010), and can increase vulnerability to predation (Holder et al. 2020). Accurately accounting for discard mortality in assessments of largely catch-and-release fisheries is therefore vital.

Of the remaining sensitivity analyses, using uninformed priors had the greatest effect on estimates of k

and B_{MSY}, resulting in a much more optimistic view of available biomass and stock status. According to this version of the model, the stock was marginally overfished but not undergoing overfishing in 2021. Failure to provide informed priors could therefore prevent management action from being taken, potentially leading to continued overfishing and even stock collapse. Previous research has shown that Bayesian methods (e.g., BSM) are highly sensitive to misspecified priors and that well-thought-out informative priors can considerably reduce uncertainty (Punt and Hilborn 1997). Expert anglers have been shown to provide accurate estimates of biomass trends in many studies (Beaudreau and Levin 2014; Shephard et al. 2021) and thus serve as a useful resource for developing informative priors. In fact, research has shown that synthesizing expert knowledge can be the most powerful approach for selecting informative model priors (Punt and Hilborn 1997). Previous studies the specifically employed the CMSY–BSM method have used expert knowledge to inform the relative biomass priors required by the model (Demirel et al. 2020). The results of this sensitivity analysis highlight the importance of the LEK component of our assessment framework (Figure 1).

Incorporating effort creep into the model also had a significant impact on estimated reference points and stock status. When accounting for a 2% linear increase in catchability, the model resulted in a much more pessimistic view of exploitation and stock status, with B_{2021} being critically low. Effort creep (also called “technology creep”) has been shown to significantly alter how fishing impacts fish stocks (Marchal et al. 2007; Scherrer and Galbraith 2020), but creep factors are typically only estimated to correct for the introduction of new technologies over short periods of time. Therefore, applying a blanket effort creep value to a long-term analysis is not ideal (Palomares and Pauly 2019). Unfortunately, there is a general lack of quantitative data on the speed and magnitude with which fishing power changes over time (Engelhard 2016). Future efforts to explicitly quantify changes in catchability due to advancements in fishing technology could greatly improve stock assessment models and inform better management.

Finally, excluding high-PSE data points from the MRIP data also led to a more pessimistic stock status than the base model due to the anomalously high MRIP catch estimate for 2009. If this was a true spike in catch reflective of a spike in abundance for that year, its cause is unknown. A strong recruitment event was a possible cause. It is well known that variability in juvenile recruitment rates due to environmental variability can lead to substantial temporal heterogeneity in population abundance (Shelton and Mangel 2011). However, fishing guides did not mention any particular spike in Crevalle Jack abundance in 2009 or anything else that would explain the spike. Additionally, the PSE of the MRIP estimate was above 50%, meaning that the estimate was very imprecise. It is therefore more likely that the catch estimate was based on a small sample size and is not a “true” reflection of total catch in that year. Studies have shown that data quantity significantly impacts stock assessment results (Chen et al. 2003). This sensitivity analysis further highlighted the importance of considering sample size and the precision of catch estimates when fisheries-dependent data are used to inform stock assessment.

Implications for management

Our sensitivity analyses revealed some uncertainty in the extent of overfishing that has occurred since 1950, but all models showed stock size trending in a negative direction, suggesting that management action is needed to halt the decline in stock size. The current exploitation rate is also at or slightly above MSY. Since the Crevalle Jack is currently an unregulated species in Florida (FWC 2021) and given that recreational fishing in the region is continually increasing (Hanson and Sauls 2011; Shertzer et al. 2019), it is likely that exploitation rates will continue to increase to unsustainable levels if the fishery remains unregulated. Importantly, with recreational fisheries the goal is not always to maximize yield. Fishing guides in the Florida Keys have observed that catch rates of Crevalle Jack have declined below a desirable level in recent years (Gervasi et al. 2022b). Thus, although F_{2021} was just above F_{MSY} and B_{2021} was only slightly below B_{MSY} for most of our model runs, management regulations that bring catch rates back up to desirable levels may be more beneficial to the guided fishery as an industry than managing for MSY. Further discussions with anglers as to what constitutes a desirable level of catch will help managers to set appropriate reference points. Because the Crevalle Jack is an unregulated species (i.e., there are no species-specific restrictions on harvest) in all U.S. Gulf and Atlantic states within the species' range, additional research into Crevalle Jack stock structure and stock status in other areas is also a critical next step.

Our suggested next steps for management include engaging in cooperative research and co-management (Kaplan and McCay 2004; Johnson and Van Densen 2007) and setting regulations on the Crevalle Jack recreational fishery that are acceptable to the stakeholders and that follow a precautionary approach. Beyond such steps, additional research can aid in reducing uncertainty and providing more concrete management recommendations. The results of our sensitivity analyses revealed the importance of estimating an accurate discard mortality rate since the vast majority of Crevalle Jack captured by recreational anglers in Florida are released. Tagging studies that assess how factors such as handling time, hooking location, depth, and predator abundance influence postrelease survival will aid in obtaining a better estimate of the survival rate (Jiang et al. 2007; Rudershausen et al. 2007; Flaherty-Walia et al. 2016). Accounting for effort creep was also shown to be incredibly important. Therefore, getting a better handle on how fishing technology and subsequent catchability of Crevalle Jack may have changed over time should be another research goal. This could potentially be accomplished via angler interviews and/or analysis of trends in the adoption and use of new fishing technologies in the region (e.g., Marchal et al. 2007). Finally, accurate delineation of stock boundaries is an important part of stock assessment (Ying et al. 2011; Berger et al. 2021). Preliminary acoustic telemetry research in Florida has revealed that Crevalle Jack make regular long-range movements throughout the state and that some individuals even cross state boundaries into other states within the Gulf of Mexico (C. L. Gervasi, unpublished data). These results suggest that to encompass the entire stock, the catch and abundance time series may need to be expanded to include data from other states. However, according to the MRIP data, approximately 95% of the Crevalle Jack captured by recreational anglers in Gulf of Mexico and South Atlantic waters are captured in Florida. Thus, even if individual fish migrate between state boundaries, fishing operations in other states are less likely to impact stock status since the majority of the Crevalle Jack fishery operates in Florida. For this reason, a stock unit extending beyond the state of Florida was not considered for the assessment conducted herein. However, as the acoustic telemetry data continue to reveal patterns of Crevalle Jack movements and migrations, the CMSY–BSM model could be re-run if necessary to account for changes in estimated stock boundaries. As new data about the species and the fishery are collected, the FishPath assessment questionnaire can also be updated and other data-limited assessment methods can be explored and compared. The three-prong assessment approach outlined herein first uses the FishPath tool to select an assessment method, then conducts an assessment using the chosen method, and finally runs sensitivity analyses for unknown or uncertain parameters. Local ecological knowledge permeates each step, rapidly filling in knowledge gaps that would otherwise take years of additional research and data collection to fill. This approach can easily be included as part of an adaptive management plan and can be applied to other unregulated species and in other regions.

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CONFLICT OF INTEREST STATEMENT

There is no conflict of interest declared in this article.

DATA AVAILABILITY STATEMENT

The data underlying this article will be shared upon reasonable request to the corresponding author.

ETHICS STATEMENT

All ethical guidelines were followed and no animals were handled in the development of this study.

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