Canadian Science Publishing

ARTICLE

Bottom-up conservation: using translational ecology to inform conservation priorities for a recreational fishery

Carissa L. Gervasi, Rolando O. Santos, Ryan J. Rezek, W. Ryan James, Ross E. Boucek, Christopher Bradshaw, Christopher Kavanagh, Jason Osborne, and Jennifer S. Rehage

Abstract: Translational ecology defines a collaborative effort among scientists and stakeholders to rapidly translate environmental problems into action. This approach can be applied in a fisheries management context when information needed to inform regulations is unavailable, yet conservation concerns exist. Our research uses a translational ecology framework to assess the stock status and develop research priorities for the crevalle jack (*Caranx hippos*) in the Florida Keys, USA, a currently unregulated species. Interview data that compiled expert fishing guide knowledge were used to develop hypotheses tested using existing fisheries-dependent datasets to check for agreement among sources and assess the consistency of observed patterns. Six hypotheses were developed concerning the status and trends of the crevalle jack population in the Florida Keys, and four of these hypotheses received clear support, with agreement between guide observations and one or more of the fisheries-dependent datasets. The results of our study outline an effective translational ecology approach for recreational fisheries management designed to rapidly recognize potential management needs as identified by fishing guides, which allows for actionable science and proactive management.

Résumé : L'écologie traductionnelle désigne les efforts de collaboration de chercheurs et de parties prenantes visant à traduire rapidement des problèmes environnementaux en gestes concrets. Cette approche peut être appliquée dans un contexte de gestion des pêches quand l'information nécessaire à l'élaboration de règlements n'est pas disponible, mais qu'il existe des situations préoccupantes. Nos travaux font appel à un cadre d'écologie traductionnelle pour évaluer l'état du stock et établir des priorités de recherche pour une espèce actuellement visée par aucune réglementation, la carangue crevalle (*Caranx hippos*), dans les Keys de la Floride (États-Unis). Des données d'interviews recensant les connaissances de guides de pêche chevronnés sont utilisées pour formuler des hypothèses testées à l'aide d'ensembles existants de données dépendantes de la pêche pour vérifier la concordance de différentes sources et évaluer la cohérence des motifs observés. Six hypothèses sont formulées concernant l'état et les tendances de la population de carangues crevalles dans les Keys de la Floride, et l'analyse appuie clairement quatre de ces hypothèses, pour lesquelles les observations des guides et un ou plusieurs des ensembles de données dépendantes de la pêche concordent. Les résultats de l'étude délimitent une approche efficace d'écologie traductionnelle pour la gestion des pêches sportives, conçue pour faire rapidement ressortir les besoins potentiels de gestion cernés par les guides de pêche et permettant du coup des applications pratiques de résultats scientifiques et une gestion proactive. [Traduit par la Rédaction]

Introduction

In today's world of anthropogenic global change and urgent ecological crises, clear pathways for actionable science are more important than ever (Chapin 2017). Translational ecology (TE) is a developing field aimed at addressing such urgent ecological issues that stems from the broader concepts of translational sociology (Callon 1986). TE describes collaborative efforts among scientists and stakeholders with the goal of rapidly translating environmental problems into action (Schlesinger 2010), and TE frameworks have successfully been used to address conservation issues within various complex social–ecological systems (Angeoletto et al. 2019; Chen and Jin 2019; Ward et al. 2020). For instance, Allison and Arnold (2018) highlight how TE has been used in the wind energy industry for decades to assess, avoid, and mitigate risks to wildlife. Yet, ecology remains predominantly a reactive field, with conservation practice occurring piecemeal and differing across ecological systems (Brooks et al. 2006; Sutherland et al. 2011; Cook et al. 2014; Crotty et al. 2019). Thus, the application of TE remains limited in key disciplines such as community ecology (Crotty et al. 2019) and environmental law (Adler 2020).

One field that would greatly benefit from the application of TE is recreational fisheries management. Recreational fishing is one of the most popular leisure activities worldwide, with five times more recreational than commercial anglers, generating US$190 billion in expenditures annually (The World Bank 2012; Arlinghaus et al. 2015, 2019). Today's recreational anglers can contribute to a large proportion of total fisheries landings in certain areas (Coleman et al. 2004; Arlinghaus et al. 2019; Felizola Freire et al. 2020). Thus, recreational fishing can negatively impact fish populations and their habitats through a myriad of direct and indirect means (Post et al. 2002;

Received 29 January 2021. Accepted 23 May 2021.

C.L. Gervasi, R.O. Santos, R.J. Rezek,* W.R. James, and J.S. Rehage. Institute of Environment, Florida International University, 11200 SW 8th St., Miami, FL 33199, USA.

R.E. Boucek. Bonefish & Tarpon Trust, 2937 SW 27th Ave., Suite 203, Miami, FL 33133, USA.

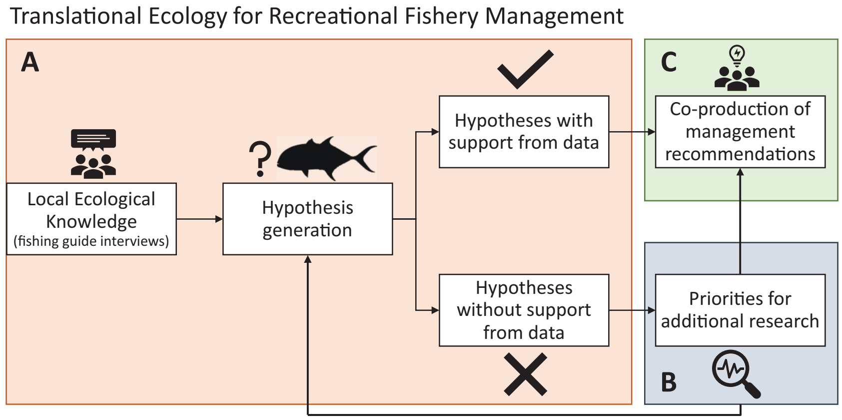
C. Bradshaw. Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, 100 8th Avenue SE, St. Petersburg, FL 33701, USA.

C. Kavanagh and J. Osborne. South Florida Natural Resources Center, Everglades National Park, 40001 State Road 9336, Homestead, FL 33034, USA.

Corresponding author: Carissa L. Gervasi (email: cgervasi@fiu.edu).

*Present address: Department of Marine Science, Coastal Carolina University, 100 Chanticleer Drive East, Conway, SC 29526, USA.

Fig. 1. Conceptual diagram of the translational ecology framework applied in this study to a recreational fishery: crevalle jack in the Florida Keys. Panel A outlines a rapid approach to developing hypotheses concerning fishery resources via fishing guide local ecological knowledge and using existing data to test the hypotheses. Hypotheses without clear support from existing data serve as priorities for additional research (panel B), which can provide missing support for existing hypotheses or lead to additional hypotheses. Finally, information from panels A and B are used to produce management recommendations supported by both fishery scientists and stakeholders (panel C). Results from panel A are presented in this paper, while panels B and C outline future directions. [Colour online.]



Cooke and Cowx 2004; Lewin et al. 2006; O'Toole et al. 2009), including the interaction of anthropogenic and climatic factors (Townhill et al. 2019). Further, due to the rapid growth in popularity and efficiency of recreational fishing, the stock assessment process is struggling to keep pace with evaluating fishery status, necessitating more rapid interim assessments and management actions. Translational ecology has the potential to help overcome many challenges currently hindering effective recreational fisheries management, such as assessment of data-poor fisheries (i.e., fisheries with insufficient information for estimating relative stock status and appropriate reference points), where most fishing effort is unreported, landings data are not available (i.e., for catch and release), and fisheries-independent monitoring is absent (Lester et al. 2003; Holder et al. 2020).

Our study aimed to apply a rapid, low-cost TE approach for assessing the stock status and developing research priorities for an unregulated and data-poor recreational fishery species (Fig. 1). Recreational fishing guide knowledge was used to generate six testable hypotheses concerning the stock status of the crevalle jack (*Caranx hippos*) in the Florida Keys. This species is highly valued by many recreational fishing guides and is an important predator in coastal environments (Kwei 1978; Saloman and Naughton 1984). Hypotheses were subsequently tested using fisheries-dependent time series (including commercial landings and recreational surveys). TE methods have been used to develop hypotheses concerning marine resources in the context of small-scale artisanal fisheries (Poizat and Baran 1997; Aswani and Hamilton 2004; Silvano and Valbo-Jørgensen 2008; Cardoso da Silva et al. 2020). However, to our knowledge, such a collaborative TE approach to hypothesis generation has yet to be applied to a large-scale recreational fishery. The research outlined herein entails the hypothesis generation and testing component (Panel A in Fig. 1) of our broader TE framework for crevalle jack conservation, which we are confident will serve as a model for the use of TE in the co-production of fisheries science with stakeholders, bridging the science to management gap.

Methods

Study species

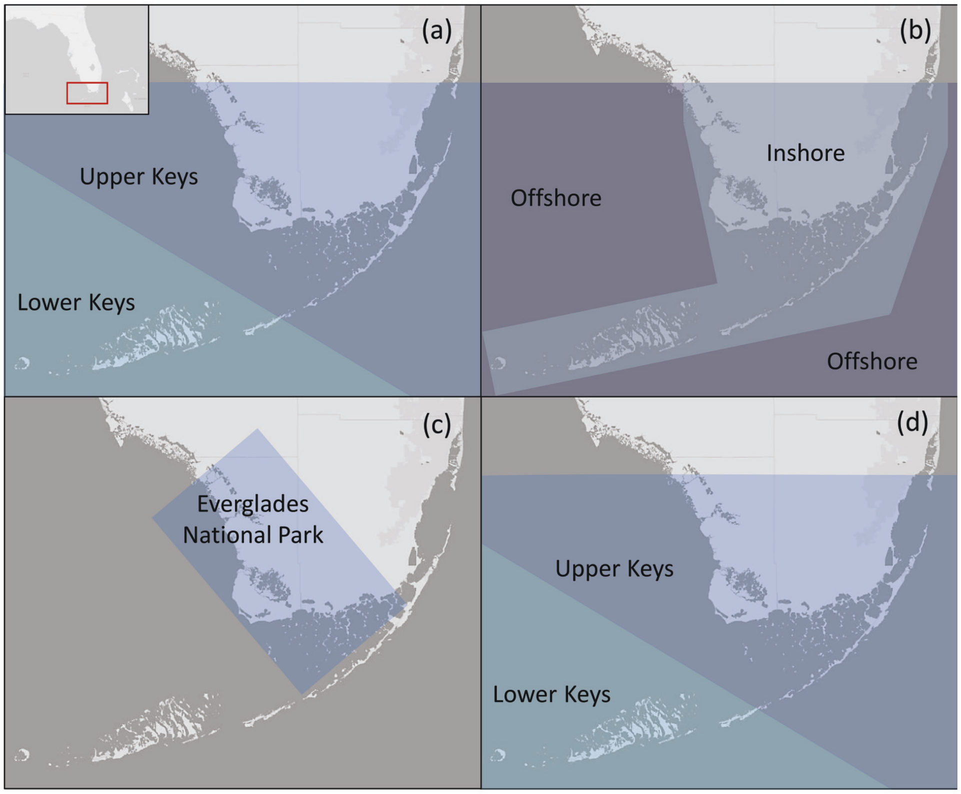
The crevalle jack was chosen as a study species for this research because it is a popular recreational fishery species throughout Florida but is currently unregulated and data-poor. In recent years, reports from recreational fishing guides in the Florida Keys suggest a decline in crevalle jack catches (Lower Keys Guides Association, personal communication, 2018), indicating management action may be warranted to conserve the population. Our translational ecology approach is ideally suited to studying the status and trends of such a species.

The crevalle jack is a large marine fish found throughout the tropical and temperate waters of the North Atlantic, from Nova Scotia to Uruguay in the west Atlantic and Portugal to Angola in the east Atlantic (Smith-Vaniz and Carpenter 2007). Crevalle jack grow rapidly, reaching about 200 mm FL within one year (Snelson 1992). Length at 50% maturity for crevalle jack in the Caribbean is 636 mm FL (Caiafa et al. 2011), and maximum size throughout the species range often meets or exceeds 22.7 kg (50 lbs.; 1 lb. $\sim$ 0.453 kg; Smith-Vaniz and Carpenter 2007). Crevalle jack spawn in tropical offshore waters in the spring or summer (Heyman and Kjerfve 2008), and larvae are distributed via ocean currents throughout the Gulf of Mexico and Atlantic coast (Berry 1959; Ditty et al. 2004). Juveniles are known to occur in inshore and estuarine habitats (Berry 1959; McBride and McKown 2000), while mature adults are found in various habitats, including coastal waters, canals, and offshore reefs (Smith-Vaniz and Carpenter 2007). A voracious carnivore, the crevalle jack is a major predator of small schooling fishes in coastal areas (Saloman and Naughton 1984), and it is caught in both commercial and recreational fisheries (Kwei 1978).

Though considered poor quality as a food fish, crevalle jack are valued by recreational anglers for their strength, speed, and voracity and are considered a "superb light tackle species" by the International Game Fish Association (IGFA 2006). Throughout Gulf of Mexico coastal waters, crevalle jack was the 10th most popular fishery species in 2017, with about 2.4 million lbs. (1089 metric tons) landed by recreational anglers and 423 000 lbs. (192 metric tons) landed by the commercial fishery (NMFS 2018). In Everglades National Park, where recreational fishing is a key economic activity (an estimated one in five Florida recreational anglers fish in the Everglades region; Fedler 2009), crevalle jack is the second most captured species behind only spotted seatrout (*Cynoscion nebulosus*), with 462 288 fish caught according to dockside angler surveys between 1980 and 2019 (NPS 2015). Recreational fishing guides often refer to crevalle jack as "trip savers" on guided trips when other targeted species are unobtainable.

Despite its importance to Florida's commercial and recreational fisheries and its role as a predator in coastal environments, the crevalle jack is currently an unregulated species, meaning there are no specific regulations regarding size limits, gear restrictions,

Fig. 2. Map of the study area. The state of Florida, USA, highlighting the South Florida region (inset map), approximate fishing range of the anglers interviewed, split into the Lower and Upper keys (*a*), extent of the MRIP survey subset to Monroe County, split into inshore and offshore (*b*), extent of the ENP creel survey (*c*), and extent of the commercial landings data subset to Monroe County, split into the Lower and Upper keys (*d*). Boundary lines are approximate and for illustrative purposes only. Commercial fishing was prohibited within ENP boundaries after 1985 (Osborne et al. 2006). Maps created using the Esri light gray canvas basemap (Esri 2011) with ArcGIS desktop (Esri 2020). [Colour online.]



bag limits, or closed seasons (FWC 2021*b*). In Florida, a default bag limit for recreational species of two fish or 100 lbs. per person per day (whichever is greater) applies to all unregulated species. Due to its unregulated status, little research has been done to assess crevalle jack life history, track abundance patterns, or determine mortality rates (McBride and Mckown 2000). Reports from Florida Keys fishing guides of a decline in crevalle jack catch rates may indicate a decline in abundance. If so, management efforts may need to be enacted to restore and ensure sustainable catch for the population in Florida (and potentially elsewhere in the US, where it remains unregulated in all 18 coastal states where the species occurs). Thus, our study has the potential to inform management throughout a large portion of the species range. However, though crevalle jack occur in the fishery throughout Florida and in other states, this study focused solely on the Florida Keys where our fishing guide collaborators observed a decline.

Hypothesis generation

Translational ecology comprises a diverse spectrum of approaches for tackling various research questions (Lawson et al. 2017). One such approach is local ecological knowledge (LEK), defined as the often place-based knowledge, beliefs, and practices concerning the natural environment that individuals or groups of people gain via observations, practical experience, or community dialogue (Anadón et al. 2009). To assess guide perceptions of crevalle jack population dynamics, key informant interviews with knowledgeable Florida Keys fishing guides were conducted between January and March 2019 (Fig. 2*a*). Key informant interviews have become a cornerstone technique for extracting local ecological knowledge and involve in-depth interviews with a nonrandom group of people who demonstrate expert knowledge about a particular topic gained via experience, participation, and (or) position (e.g., Heinen and Shrestha-Acharya 2011; Dongol and Heinen 2012). Key informant interviews have been used successfully to study bonefish (*Albula vulpes*) populations in South Florida for example, and the authors were able to identify periods of population decline, spatial patterns in decline, and the most likely factors contributing to the decline (Kroloff et al. 2019; Santos et al. 2019).

Saltwater recreational fishing guides in South Florida are typically individual, small business owners who have either a charter captain or charter boat license issued to them by the Florida Fish and Wildlife Conservation Commission (FWC). This license allows them to carry paying customers for the purpose of taking or possessing saltwater fish or organisms. In the Florida Keys, charter operations are a large component of the tourism industry. In 2012, Florida Keys fishing guides had a total economic impact of over US$111 million (Fedler 2013). Only guides who reported spending a minimum of 100 days·year^{-1} on Florida water and regularly guiding for at least 5 years were interviewed for this study. Interviewees were selected using the "snowball" method (i.e., word-of-mouth referrals by other guides and (or) guide associations; Atkinson and Flint 2001). By using a snowball sampling approach, we identified as many key informants as possible, focusing on charter captains with a lot of experience fishing in the Florida Keys. We interviewed everyone who was recommended to us as having expert knowledge of inshore fisheries, including crevalle jack, and was willing to be interviewed, for a total sample size of 18 guides. In key informant studies, saturation is typically used to determine appropriate

Table 1. Fisheries-dependent datasets used to test angler-derived hypotheses.

Dataset	Source	Years	Area
Marine Recreational Information Program (MRIP) survey	National Oceanic and Atmospheric Administration (NOAA)	1991–2019	Monroe County — inshore and offshore
Everglades National Park (ENP) creel survey	ENP	1980–2019	Inshore Monroe County within ENP
Commercial landings data	Florida Fish & Wildlife Conservation Commission (FWC)	1986–2019	Monroe County — Upper and Lower keys

Note: Source denotes the agency responsible for data collection and dissemination, Years are the years of data analyzed in this study, and Area is the region each dataset covers.

sample size. Hennink et al. (2017) examined 25 in-depth interviews and determined that code saturation was reached after nine interviews, where the range of thematic issues was identified. Similarly, Guest et al. (2006) examined 60 interviews and found that saturation occurred within 12 interviews. Muellmann et al. (2021) observed no change in results with an increase from 4–6 key informants to 12–15. Based on this literature we deemed 18 key informants a suitable sample size for our study.

From our interviews, we determined that guides typically operate within two broad regions of the Florida Keys, either from Marathon south to the Marquesas or the Florida Keys region north of Marathon and including Florida Bay and Biscayne Bay. So, we split interviewees into "Lower Keys" guides (Marathon and points south) and "Upper Keys" guides (north of Marathon). A semi-structured interview format was used with all key informants, and the interviews were recorded on an audio recording device with the guide's permission. Guides were asked four open-ended questions to direct 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? (4) Is fishing for crevalle jack important to you? More specific follow-up questions were asked if guides did not provide specific answers during the interview or if clarification was warranted (see online Supplementary material, Fig. S1[1]). This semi-structured interview format allowed us to gain as much information from guides as possible while ensuring specific questions were addressed (e.g., when did you start noticing a change in your crevalle jack catches?). Audio data were later transcribed, and common topics were compiled and developed into testable hypotheses about where crevalle jack are in decline, when the decline began, and what factors might be responsible for the decline. Topics were developed into hypotheses if more than 50% of guides interviewed agreed about a particular observation. This 50% cutoff was used because we aimed to develop one hypothesis per topic rather than employing a multiple hypothesis framework. All protocols for human subject research were approved by Florida International University's Institutional Review Board and all participants gave consent before being interviewed.

Fisheries-dependent datasets

Since the crevalle jack is an unregulated species and the species lacks a formal stock assessment process, abundance trends in South Florida are unknown. Furthermore, there are no existing fisheries-independent surveys in the region that frequently encounter crevalle jack, so our hypotheses were tested using existing fisheries-dependent datasets (Table 1). The three datasets used for hypothesis testing were (1) the NOAA Marine Recreational Information Program (MRIP) survey (NOAA 2021), (2) the Everglades National Park (ENP) creel survey (NPS 2015), and (3) the state of Florida commercial landings data (FWC 2021*a*). Each dataset provided independent information that was appropriate for addressing one or more of the LEK-derived hypotheses. For the MRIP and ENP surveys, crevalle jack catch-per-unit effort (CPUE) time series were standardized using generalized linear models (GLMs) to generate estimated annual indices of abundance that could be used to test hypotheses about changes in abundance over time. Commercial landings data and records of recreational landings were used to assess the extent of fishing harvest (i.e., a potential cause of decline) in the Florida Keys region, while sizes of landed fish from the MRIP dataset were used to assess changes in fish size. All datasets were subset to the Florida Keys region such that the data would be reflective of where the interviewed guides regularly fish and could be used to test the LEK-derived hypotheses (Fig. 2).

A Florida saltwater fishing license is required to land any saltwater species in Florida, in state or federal waters. Recreational licenses (including charter captain or charter boat licenses) do not allow the commercialization of catch. Harvest of more than 100 lbs. or two fish (whichever is greater) is considered commercial quantity and requires a commercial license (FWC 2021*b*). Recreational fisheries throughout the state of Florida are surveyed by the Marine Recreational Information Program (MRIP) conducted by NOAA Fisheries (formerly the Marine Recreational Fisheries Statistics Survey, MRFSS). This survey has monitored shore-based, private, and charter fishing modes since 1981, and recently underwent a substantial modification and peer review in 2018 following a three-year transition period (Papacostas and Foster 2018). MRIP data have been used to assess the status and trends and develop standardized indices of abundance to inform stock assessments for several fish species throughout the Western Atlantic and Gulf of Mexico, including sailfish (*Istiophorus platypterus*), vermilion snapper (*Rhomboplites aurorubens*), red grouper (*Epinephelus morio*), shortfin mako (*Isurus oxyrinchus*), and many others (Ortiz and Brown 2002; Babcock 2013; Rios 2015; Sagarese 2019).

Everglades National Park (ENP) was established in 1947 and voluntary dockside interviews have been conducted within park boundaries since 1958 (Davis and Thue 1979; Schmidt et al. 2002). Commercial fishing has been prohibited within the park since 1985 (Osborne et al. 2006), but recreational fishing is allowed and bag and size limits follow freshwater and saltwater recreational fishing regulations established by FWC. A sample of recreational anglers are interviewed by ENP personnel upon arrival post-fishing at either of two popular public ramps in the park. Recorded data include trip origin, area fished, number of anglers, hours fished, numbers of fish caught and released by species, etc. Creel survey data have been used to examine the impacts of coastal protected areas on recreational world records (Bohnsack 2011), to monitor the recovery of an endangered species (smalltooth sawfish (*Pristis pectinata*); Carlson et al. 2007), and to track trends in abundance for other data-poor species such as bonefish (Santos et al. 2017) and Atlantic goliath grouper (*Epinephelus itajara*; Cass-Calay and Schmidt 2009).

Commercial landings data for the state of Florida are collected by FWC. These fisheries include all species that are harvested for profit, including those sold for human consumption, aquariums, and medical use. Florida began a mandatory trip ticket program in 1984, and the first official year of landings is 1986. Commercial

[1]Supplementary data are available with the article at https://doi.org/10.1139/cjfas-2021-0024.

landings data have been used for several applications. These applications include the development of regulations to prevent overexploitation of shark species in Florida (Brown 1999), informing stock assessment of important species such as the Caribbean spiny lobster (*Panulirus argus*; SEDAR Stock Assessment Panel 2010) and red grouper (Wrege and Orhun 2019), and evaluating the sustainability of coral reef fisheries in the Florida Keys (McClenachan and Kittinger 2013).

Development of abundance indices

CPUE is often assumed to be proportional to stock abundance and is therefore commonly used as a relative abundance index when fisheries-independent data are unavailable (Maunder and Punt 2004). However, many factors can influence fisheries catch rates (e.g., spatial, temporal, or environmental variability). It is, therefore, necessary to standardize CPUE data to remove the influence of factors other than stock abundance before CPUE data can be used as an index of abundance (Maunder et al. 2006). Generalized linear models (GLMs; Nelder and Wedderburn 1972) are commonly used to standardize CPUE data (Maunder and Punt 2004; Venables and Dichmont 2004). The delta-lognormal GLM approach (Lo et al. 1992) has specifically been used to standardize CPUE for several species using the MRIP and ENP datasets (e.g., Carlson et al. 2007; Cass-Calay and Schmidt 2009; Cass-Calay 2012; Rios 2015; Sagarese 2019), and was thus used in this study. The delta-lognormal method combines separate GLM analyses on the positive trips (trips that captured the species of interest) and the proportion of positive trips (trips that captured the species of interest/total trips) to create a single index. Prior to model fitting, data exploration and filtering following the methods of Zuur et al. (2010) were conducted on all three datasets (MRIP, ENP, and commercial landings), including assessing the data for outliers, collinearity, zero-inflation, and balanced categorical covariates, and filtering the data as appropriate (see data filtering specifics below). The commercial landings dataset was deemed unsuitable for developing a standardized index of abundance because very few crevalle jack were captured on commercial trips in the Florida Keys (only 1% of trips captured crevalle jack). The commercial data were instead used to assess other hypotheses concerning fishing harvest (see hypothesis testing section below).

Data filtering

The MRIP data were subset to include only trips from Monroe County, which encompasses the Florida Keys (Fig. 2*b*), and several categorical variables were constructed from the data prior to analysis. These included Year (1991–2019), Month (1–12), Season (spring — March, April, May; summer — June, July, August; fall — September, October, November; and winter — December, January, February), Area fished (inshore — less than 10 miles (1 mile = 1.609 km) from shore, or offshore — greater than 10 miles from shore), and Fishing mode (shore, charter, or private). Gear type was also recorded in the data, but since 97% of Monroe County trips used hook and line gear, the data were subset to only include hook and line trips. Party code was defined as a single trip and catches for every individual within the party were summed, such that in our analyses we were examining the trip catch for every trip in the dataset. Since the party code was not recorded until 1991, the data were truncated so that only data from 1991 to 2019 were analyzed. Fishing effort for the MRIP data was defined as the number of people in the party who were interviewed multiplied by the reported hours fishing. Size information was available for a subset of landed crevalle jack, and fork lengths differed significantly between inshore (M = 378 mm, SD = 151 mm) and offshore (M = 590 mm, SD = 184 mm) trips. A Welch 2-sample t test between inshore and offshore fish showed a significant difference in fish size between the two groups ($p < 0.0001$), suggesting that these areas capture different size classes of the population, matching the presumed life history in South Florida consisting of inshore recruitment and juvenile habitat use and offshore habitat use and reproduction by larger adults (Smith-Vaniz and Carpenter 2007). Therefore, trips that occurred offshore were modeled separately from trips that occurred inshore, with offshore trips used to assess abundance trends of large adult and subadult crevalle jack. In contrast, inshore trips were used to assess abundance trends of smaller juvenile crevalle jack. Length at 50% maturity for crevalle jack in the Caribbean is 636 mm FL (Caiafa et al. 2011), suggesting that the majority of inshore and even some of the offshore crevalle jack were likely immature.

The ENP angler data were analyzed for the period 1980 to 2019 because in 1980 the survey was expanded to include routine surveys at both Flamingo and Everglades City boat ramps, where anglers continue to be interviewed presently (Schmidt et al. 2002; Osborne et al. 2006; Carlson and Osborne 2013). Categorical variables were also constructed from the ENP dataset and included Year (1980–2019), Month (1–12), Season (spring — March, April, May; summer — June, July, August; fall — September, October, November; and winter — December, January, February), and Area fished (6 fishing areas defined by Schmidt et al. 2002). The entire ENP region was analyzed since Upper Keys anglers typically fish throughout ENP coastal waters (Fig. 2*c*), so we did not subset the data further.

Accounting for catchability

The use of CPUE as an index of abundance assumes that catchability for the species of interest is constant; however, with multispecies fisheries, different fishing tactics employed to target focal fish species can influence catchability. Anglers do report the primary and secondary species targeted on each trip in the MRIP data, but crevalle jack were rarely reported as the main species targeted (less than 1% of trips). This matched the guide perspectives captured from interviews, which described crevalle jack were more a species of opportunity and not as targeted as other species. Since crevalle jack are often captured opportunistically, we wanted to ensure that we only included trips in our analysis that occurred in areas where crevalle jack were likely to be. Accounting for catchability in abundance models is regularly employed for species with low catch rates (Carlson et al. 2007). The MRIP data do not include specific fishing location information (e.g., habitat type), so we performed a hierarchical clustering analysis following the methods of Shertzer and Williams (2008) that clusters species based on how often they are captured together. We then assumed that any trip where a species in the crevalle jack cluster was captured was a trip that was also likely to capture crevalle jack (whether crevalle jack were captured or not). This allowed us to use the species composition of the catch as a proxy for habitat. To perform the clustering analysis, the data were formatted into a CPUE matrix with rows representing species and columns a combination of month, area, and mode of fishing, which are factors that may represent different fishing tactics and therefore influence catchability. Species were removed if they appeared in fewer than 1% of trips since rare species can distort inferred patterns (Koch 1987; Mueter and Norcross 2000). The data were transformed using a 4th root transformation, and a matrix of dissimilarities between species was computed with the Bray–Curtis measure of distance (Bray and Curtis 1957). Hierarchical cluster analysis was then used to partition species into groups. Trips were removed if they did not catch at least one of the species in the cluster with crevalle jack, and trips where no fish were caught were also excluded. Clustering and subsequent data filtering and model fitting were performed separately for offshore and inshore MRIP trips because smaller, inshore crevalle jack associated with different species than larger, offshore crevalle jack, and trends in abundance over time may differ between the groups. Since crevalle jack were the second most captured species in the

Table 2. List of hypotheses derived from LEK interviews, number (percent) of guides interviewed who agreed with each hypothesis, and proposed method for testing each hypothesis including the datasets used.

Hypothesis	Guide support	Analysis to test hypothesis
(1) Crevalle jack populations in the Florida Keys have declined	14 (78%)	Trends in standardized abundance over time (MRIP and ENP data)
(2) Populations have declined more in LK than UK	LK − 11 (92%) vs. UK − 3 (50%)	Analysis of regional trends in abundance indices (MRIP and ENP data)
(3) The decline started sometime after 2005	9 (64%)	Breakpoints in abundance indices (MRIP and ENP data)
(4) Fishing harvest in the Florida Keys is a contributor to the decline	12 (86%)	Commercial landings over time (commercial data) and recreational proportion harvested vs. released (MRIP and ENP data)
(5) Larger fish have declined more than smaller fish	10 (71%)	Comparison of abundance indices for offshore (larger fish) and inshore (smaller fish) trips (MRIP data)
(6) Crevalle jack are migratory and most abundant in the Florida Keys in winter	13 (72%)	Seasonal differences in abundance indices (MRIP and ENP data)

Note: LK, Lower Keys; UK, Upper Keys.

ENP dataset and the park encompasses a relatively small inshore only area, there was no need to employ a clustering analysis to account for catchability with the ENP data.

Statistical analyses

For both the MRIP and ENP data, a logit link function with a binomial error distribution was fitted to the proportion of positive trips (i.e., presence or absence of crevalle jack in a trip) following the delta-lognormal model approach. For the positive trips (i), total catch was defined as the sum of landed and released crevalle jack and the natural log of CPUE (lnCPUE_i) was used as the response variable, where

$$\text{(1)} \qquad \mathrm{lnCPUE}_i = \ln[(\text{total catch}_i)/(\text{number of anglers}_i \times \text{hours fished}_i)]$$

and a log link function with a lognormal error distribution was fitted to the data. A backward stepwise regression procedure was used to determine the set of fixed factors included in the final models among Year, Season, and Fishing mode for the MRIP data (offshore and inshore) and Year, Season, and Area fished for the ENP data. Deviance tables were constructed for each GLM to determine the percentage of total reduction in deviance ($\%\mathrm{rd_t}$) due to the addition of each factor:

$$\text{(2)} \qquad \%\mathrm{rd_t} = [(\mathrm{rd_f})/(\mathrm{rd_{nm}} - \mathrm{rd_{fm}})] \times 100$$

where $\mathrm{rd_f}$ is residual deviance attributed to the addition of a given factor, $\mathrm{rd_{nm}}$ is the residual deviance of the null model, and $\mathrm{rd_{fm}}$ is the residual deviance of the full model (Cass-Calay and Schmidt 2009). Factors were selected for inclusion in final models if the addition of the factor explained more than 5% of the deviance and the χ^2 test was significant ($p \le 0.05$). However, factors Year and Season were kept in final models even if they explained less than 5% of the deviance because our goals were to assess both annual and seasonal differences in crevalle jack abundance.

For each model, CUSUM plots were used to assess breakpoints in the time series. CUSUM control methods were developed by Page (1954) for industrial quality control applications and are designed to detect persistent changes in observed processes. Recently, the method has been used in ecological applications to assess underlying features of time series data and for environmental modeling (Mac Nally and Hart 1997; Manly and Mackenzie 2000; Keatley and Hudson 2012; Regier et al. 2019). The method entails a cumulative sum of the deviation of observations from a global mean, and the slope and direction of the line in a CUSUM plot enables identification of periods that are above average (positive slope), below average (negative slope), or are not changing (no slope) (Hawkins and Olwell 1998; Scandol 2003). A change from a positive slope to a negative slope (dome shape) would indicate a decline in the index over time. All statistical analyses were performed in R version 3.6 (R Core Team 2019).

Hypothesis testing

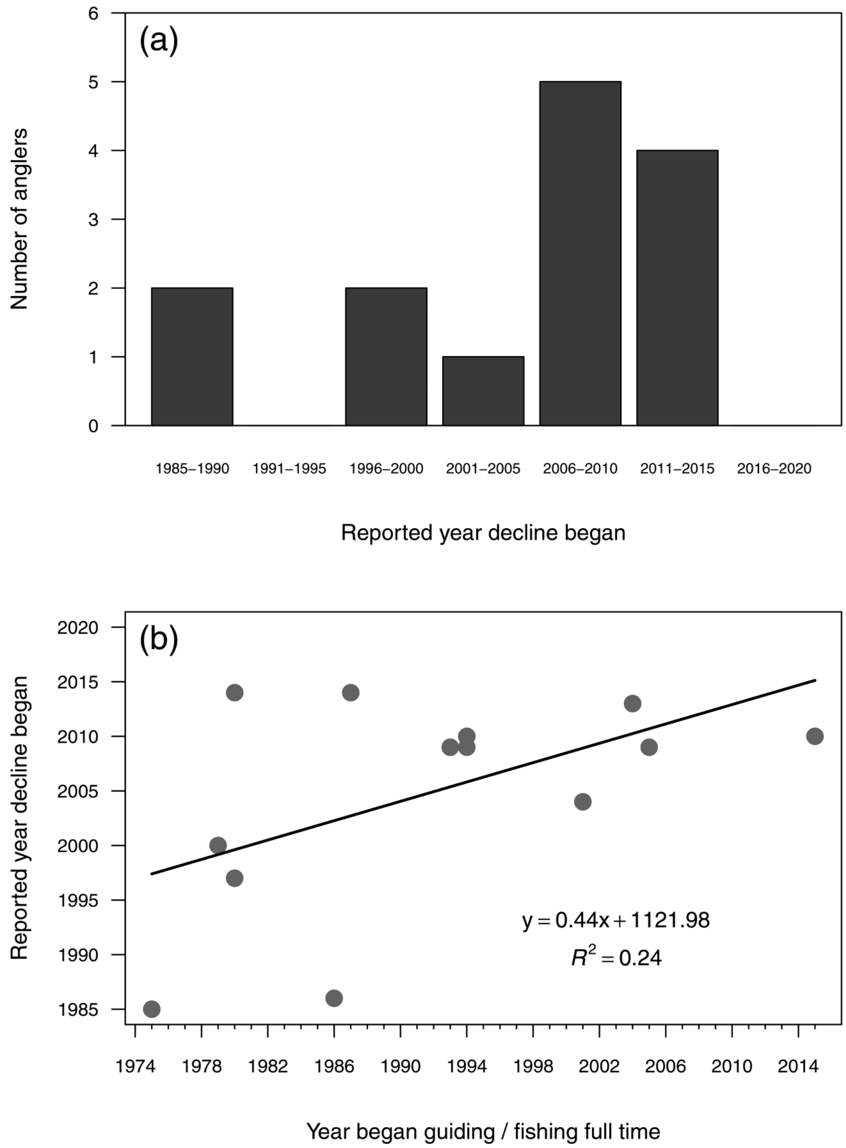
Each LEK-derived hypothesis (Table 2) was tested using one or more of the fisheries-dependent datasets (commercial, MRIP, and ENP). Comparisons between the LEK data and fisheries-dependent data were possible because expert interviewee duration of experience was on the same time scale as the examined catch data (both covering approximately the 1980–2019 period). Standardized indices of abundance derived from the MRIP and ENP datasets were used to assess whether crevalle jack populations in the Florida Keys have declined (Hypotheses 1, 2). Plots of cumulative sums of z-scored indices (CUSUM) were used to visually assess any breakpoints in the time series and determine when the decline began (Hypothesis 3). To assess whether fishing harvest (commercial and (or) recreational harvest) might have contributed to the crevalle jack decline (Hypothesis 4), total commercial landings over time for all gear types combined were assessed for both the Lower Florida Keys and the Upper Florida Keys for the period of record from 1986 to 2019 as a proxy of fishing mortality due to commercial harvest. As a proxy of fishing mortality due to recreational harvest, the proportion of crevalle jack landed (kept) by recreational anglers in ENP and MRIP data were calculated annually for 1980–2019 (ENP) and 1991–2019 (MRIP). We assessed whether the decline was specific to a certain size or age class of crevalle jack (Hypothesis 5) by creating separate indices of abundance for MRIP offshore and MRIP inshore trips, and the addition of Season as a fixed factor in our GLM models allowed us to assess seasonality of crevalle jack abundance in the Florida Keys (Hypothesis 6). Hypotheses that received support from multiple data sources were considered highly likely, while hypotheses with no support from long-term datasets were considered as priorities for additional research.

Results

Guide local ecological knowledge

Recreational fishing guide interviewees (n = 17, all male participants) had been full-time guides in the Florida Keys between 5 and 49 years, with an average of 26 years of experience guiding. One additional angler interviewed was not a guide but had been fishing in the Florida Keys for 14 years and was considered an expert angler. Of these anglers (hereafter referred to as guides), 12 operated in the Lower Florida Keys (from Marathon south), and 6 operated in the Upper Florida Keys (north of Marathon including Florida Bay and Biscayne Bay; Fig. 2*a*). All the guides interviewed at least occasionally fished for crevalle jack, and eight guides reported regularly targeting crevalle jack on guided trips. Five of these guides operated in the Lower Keys while three operated in the Upper Keys, so about half of the guides in each

Fig. 3. Timeframe in 5-year blocks when anglers first reported noticing a decline in crevalle jack catches in the Florida Keys (*a*) and reported year decline began compared to the year anglers began guiding or fishing full time (*b*). Black line in panel (*b*) denotes fitted linear regression.



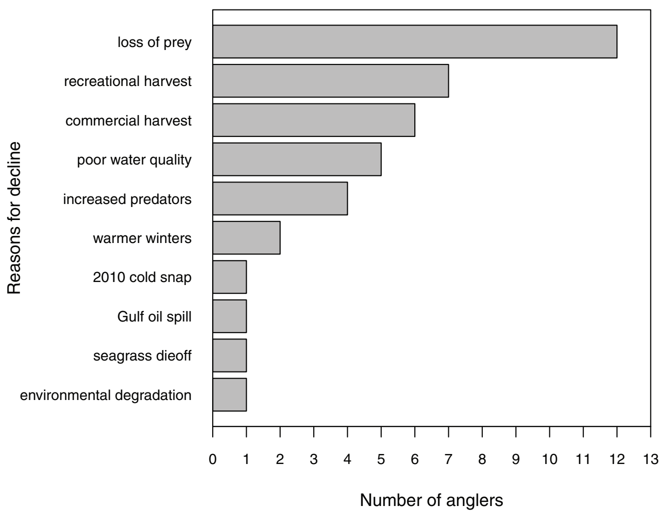
population regularly targeted crevalle jack. Of the 12 Lower Keys guides, 11 described a decline in crevalle jack catch over time while only three of the six Upper Keys guides noticed a decline. The remaining four guides (one from the Lower Keys and three from the Upper Keys) did not report any change in the crevalle jack population. Of the guides who did report a decline in catch, the average estimate for when the decline began was 2005, with a range from 1985 to 2014 (Fig. 3*a*). There was a relationship between years angling and estimated year the interviewees perceived the decline began, with more seasoned guides typically noting an earlier decline than newer guides (linear regression, $p < 0.05$; Fig. 3*b*).

Guides who reported a decline described between a 30%–100% decline in crevalle jack populations where they fish, with certain areas and size classes declining more so than others. For example, one guide estimated that catches had declined over 50% in offshore areas, but less than 50% in inshore areas. Another guide observed a 90%–100% decline in catches on the shallow water flats and about a 50% decline elsewhere. 10 of the 14 guides who reported a decline in crevalle jack catches also noticed a size decline, with fewer fish over 10 lbs. being observed. Most guides (13) agreed that crevalle jack in the Florida Keys are most abundant in the winter months and likely migrate out of the Keys region in spring or summer following changes in temperature and (or) migrating bait.

Guides who observed a decline in crevalle jack catches were asked to speculate on the reasons for the decline (Fig. 4). The most common explanation was a loss of prey, mentioned by 12 guides. Recreational and commercial harvest were the next most common explanations (7 and 6 guides, respectively), followed by poor water quality (5 guides) and increased predator abundance (4 guides). All the guides interviewed reported releasing at least 90% of the crevalle jack they captured while guiding or fishing recreationally. However, some reporting keeping a few a year to use as shark bait. When asked if they knew of anyone keeping crevalle jack for consumption or other purposes, most guides agreed that they rarely see the species brought into the docks. However, several guides mentioned that there might be populations of Florida residents who regularly capture crevalle jack for consumption.

All the guides expressed being pro-regulation of crevalle jack, and they all placed high quantitative and qualitative value on the

Fig. 4. Reasons for the decline in crevalle jack abundance as speculated by interviewed anglers. *X* axis denotes the number of anglers who mentioned each reason.



species. Several guides referred to crevalle jack as "trip savers", because when more commonly targeted species like Bonefish or Permit were nowhere to be found, guides could typically count on fishing a school of crevalle jack to keep their customers happy. One guide specifically stated: "I probably make more money putting smiles on people's faces because the fight is 2–3 times better than any species of fish we have inshore". Another guide referred to crevalle jack in South Florida as the "bread and butter of the flats fishing industry".

Hypothesis generation

Collected key informant data were used to develop six hypotheses (Table 2). Hypothesis 1 was that crevalle jack populations in the Florida Keys have declined, which received the support of 14 of the 18 guides interviewed. Hypothesis 2 was that crevalle jack populations have declined more so in the Lower Florida Keys than in the Upper Florida Keys. This hypothesis was generated based on the observation that while 11 guides operating in the Lower Keys observed a decline in crevalle jack catch, only three guides operating in the Upper Keys noticed any sort of decline. Hypothesis 3 concerned the timing of the decline. Nine guides estimated that the decline began sometime after 2005, but there was no clear consensus among guides, with two guides estimating the decline began as early as the late 1980s to early 1990s and three guides pinpointing the late 1990s to early 2000s (Fig. 3*a*). Hypothesis 4 was that fishing harvest (commercial and (or) recreational) at least in part contributed to the crevalle jack decline. Either commercial harvest, recreational harvest, or both were mentioned by 12 guides (Fig. 4). Hypothesis 5 suggests that the decline was size-selective, with larger fish having declined more than smaller fish. This hypothesis received support from 10 guides. The final hypothesis (Hypothesis 6) derived from fishing guide LEK was that crevalle jack are migratory with the highest abundances in the Florida Keys being observed in the winter months, an observation shared by 13 guides.

Hypothesis testing

One or more of the three fisheries-dependent datasets were used to test each of our six hypotheses. Filtered and cleaned MRIP data included 5687 inshore trips and 2330 offshore trips from 1991–2019 (29 years). Of these, crevalle jack were caught on 1122 inshore trips (20%) and 496 offshore trips (21%). Based on model selection via backward stepwise regression and deviance tables, the final inshore model for both the proportion positive and positive trip GLMs included Year, Fishing mode, and Season as fixed factors (Supplementary Tables S1, S2[1]), whereas the final offshore model for both the proportion positive and positive trip GLMs included Year and Season as fixed factors (Supplementary Tables S3, S4[1]). The filtered and cleaned ENP database consisted of 192 728 trips occurring from 1980–2019 (40 years), with 85 849 of those trips capturing crevalle jack (45%). After stepwise regression and deviance table selection, the final model was the same for both the proportion positive and positive trip GLMs and included Year, Area, and Season as fixed factors (Supplementary Tables S5, S6[1]). The cleaned commercial landings data consisted of over one million total trips in Monroe County from 1986–2019. Crevalle jack were landed on only 10 755 (∼1%) of those trips. Of the trips where crevalle jack were landed, 6487 (60%) occurred in the Lower Keys (Tortugas, Key West, and Marathon regions), and 4268 (40%) occurred in the Upper Keys (Everglades National Park and Miami regions; Levesque 2009). The most common gear type used on commercial trips where crevalle jack were landed was hook and line gear (48% of trips), followed by gillnets (10%), rod and reel (10%), and cast nets (6%). Most remaining trips were of unknown or other gear types (14%) with minor gear types comprising the remaining 12% of trips (including combinations of traps, nets, spears, etc.).

Hypothesis 1: crevalle jack populations in the Florida Keys have declined

Our first hypothesis was that crevalle jack populations in the Florida Keys have declined, which had the support of 78% of the guides interviewed. This hypothesis was tested using the MRIP and ENP standardized indices of abundance. Some evidence of a decline in CPUE over time was observed in the fisheries-dependent data, but only in certain regions. In the MRIP data, standardized CPUE for inshore trips (less than 10 miles from shore) remained relatively constant from 1991 to 2019, but standardized CPUE for offshore trips (greater than 10 miles from shore) declined steadily over the same period (Figs. 5*a*, 5*c*). A decline in the standardized abundance index was also apparent in ENP, though less pronounced than the MRIP offshore decline (Fig. 6*a*). For the MRIP offshore data, the maximum standardized abundance of the time series (in fish per unit effort) was 0.47 in 1991, while the minimum was 0.01 in 2017, which represents a 98% decline. For the ENP data, the maximum standardized abundance was 0.39 in 1991 and the minimum was 0.11 in 2018, a 72% decline. CUSUM plots for the z-scored MRIP offshore index and ENP index both show dome shapes indicative of declines in the time series starting around the early 1990s. Additionally, the negative slopes in the CUSUM plots reveal that abundance has been below the average of the time series almost every year since the early 2000s (Figs. 5*b*, 5*d*, 6*b*). The agreement among LEK, MRIP offshore, and ENP data lends support to this hypothesis.

Hypothesis 2: populations have declined more in the Lower Keys than the Upper Keys

Our second hypothesis was that populations have declined more so in the Lower Keys than the Upper Keys because most Lower Keys guides interviewed observed a decline (92%) while only 50% of Upper Keys guides observed a decline. We would expect that in regions where the population declined substantially, a higher proportion of guides would have noticed the decline, while in regions where the population decline was minimal, it would have been noticed by fewer guides. The MRIP dataset does not provide low enough resolution to assess differences in the Lower Keys vs. the Upper Keys, and we have no other information on crevalle jack abundance in the Lower Keys only. However, the ENP dataset mostly covers the Upper Keys region and is entirely inshore. A decline over time was apparent in the ENP data, though not as dramatic as the decline in the offshore MRIP data. Also, there is much more interannual variability in the ENP

Fig. 5. Nominal and standardized (modeled) crevalle jack CPUE (scaled to mean of 1) for MRIP inshore (*a*) and offshore (*c*) data, and CUSUM plots for *z*-scored MRIP inshore (*b*) and offshore (*d*) indices. Nominal CPUE is the average annual CPUE before standardization. Positive slopes on CUSUM plots denote periods where standardized CPUE was above the average of the time series while negative slopes denote periods where standardized CPUE was below average. Red dotted lines denote model 95% confidence intervals. Horizontal black dashed lines lie at the averages of each time series (scaled to 1). [Colour online.]

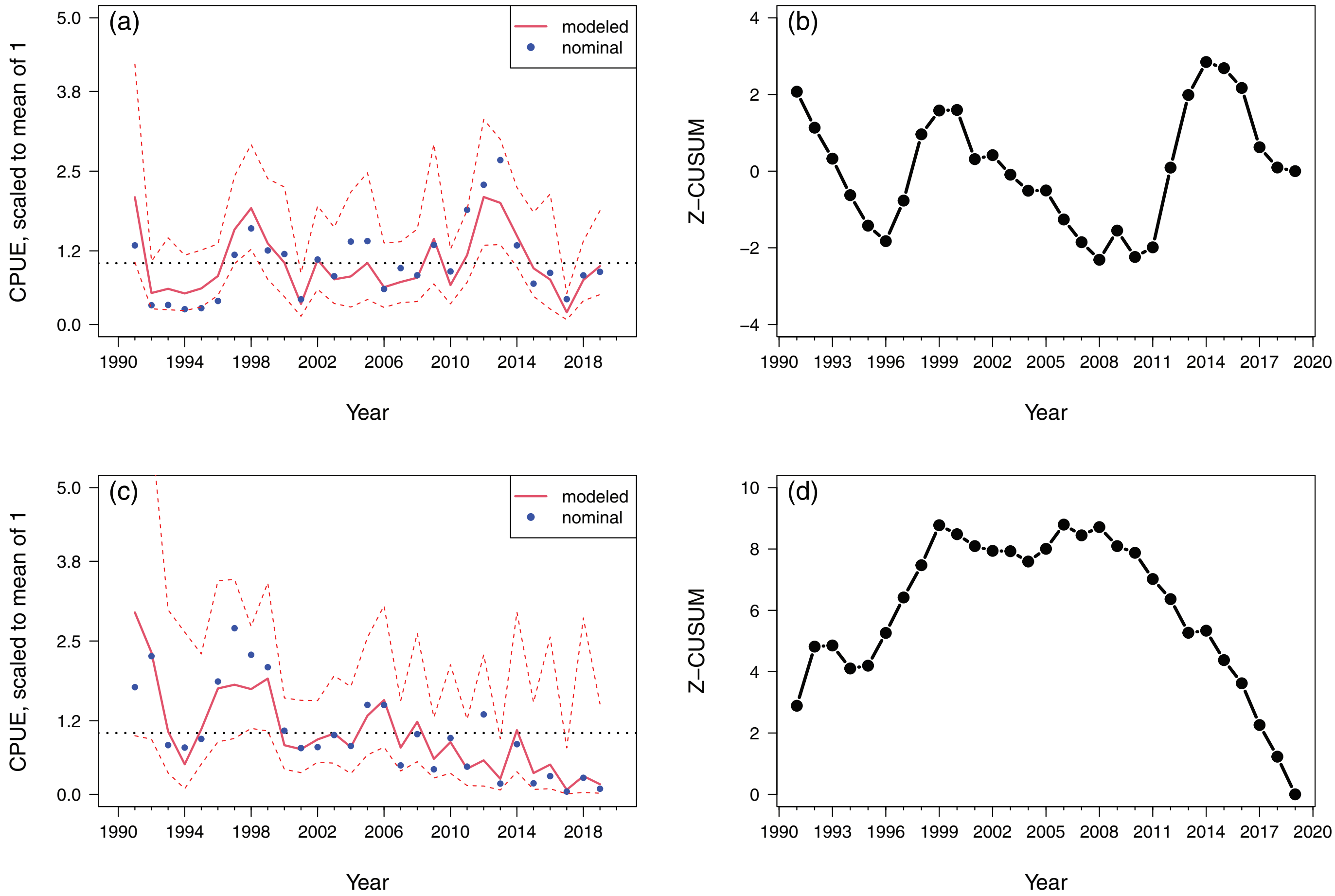


Fig. 6. Nominal and standardized (modeled) crevalle jack CPUE (scaled to mean of 1) for the ENP data (*a*) and CUSUM plot for the *z*-scored ENP index (*b*). Nominal CPUE is the average annual CPUE before standardization. Positive slopes on the CUSUM plot denote periods where standardized CPUE was above the average of the time series while negative slopes denote periods where standardized CPUE was below average. Red dotted lines denote model 95% confidence intervals. The horizontal black dashed line lies at the average of the time series (scaled to 1). [Colour online.]

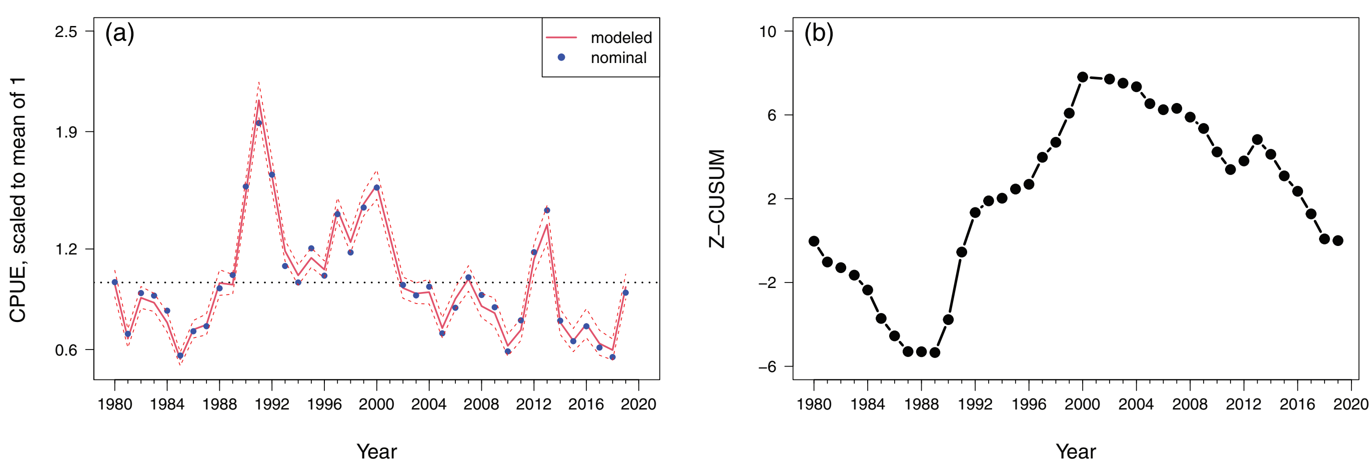
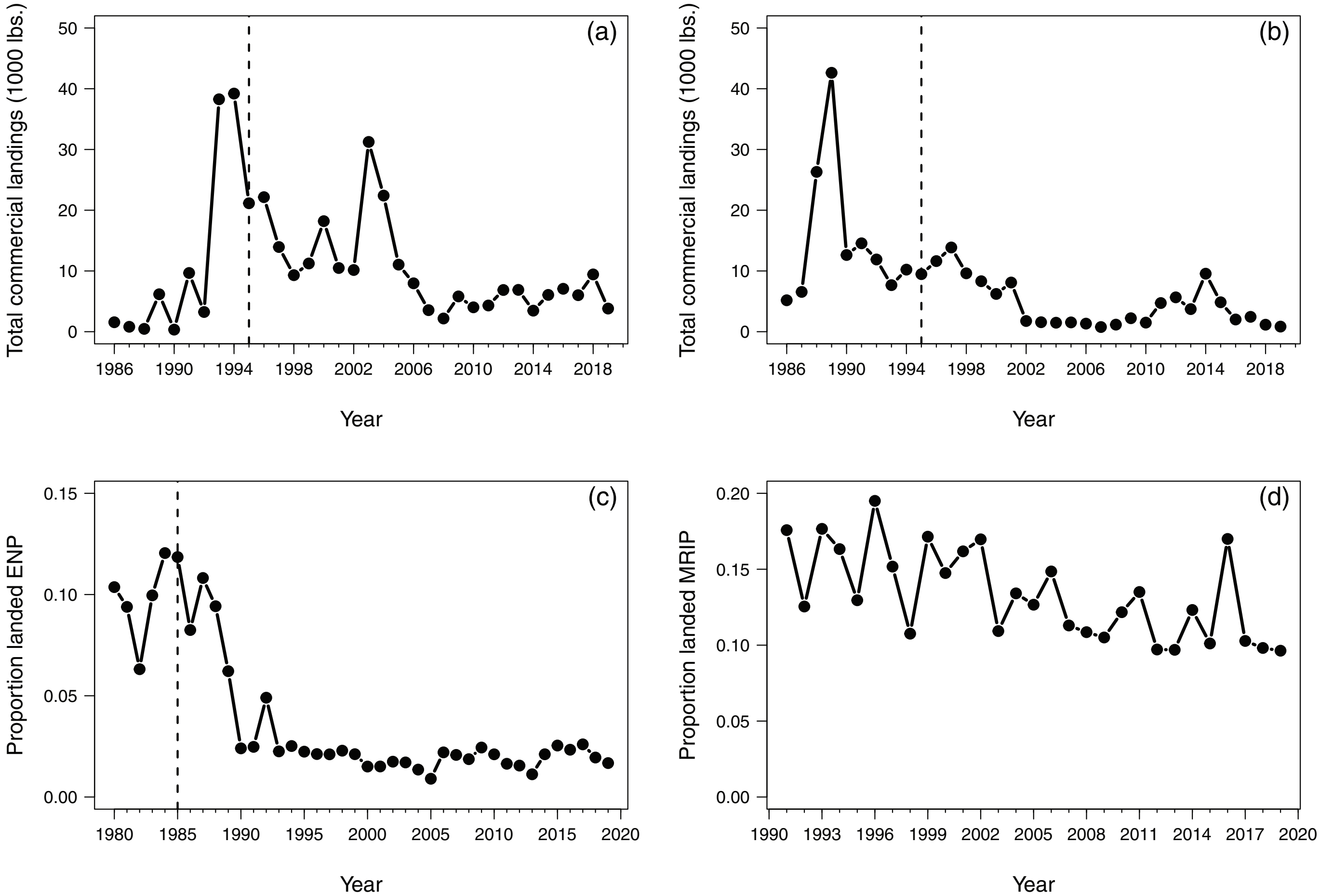


Fig. 7. Total annual commercial landings of crevalle jack by all gear types for the Upper Keys (*a*) and Lower Keys (*b*), proportion of crevalle jack landed by year in ENP data (*c*), and proportion of crevalle jack landed by year in MRIP data (*d*). Vertical dashed lines in panels (*a*) and (*b*) denote when commercial entanglement nets were banned in Florida (1995; Smith et al. 2003), and the dashed line in panel (*c*) denotes when commercial fishing was banned within Everglades National Park (1985; Osborne et al. 2006).



index, which could explain why some Upper Keys guides did not notice a decline. There was also no apparent decline in the MRIP inshore data. Generally, the Upper Keys guides reported mostly fishing inshore, while the Lower Keys guides reported fishing both inshore and offshore. Therefore, the discrepancy between the two guide groups may be more attributable to the habitats where they fish than the spatial domain where they fish. While fisheries-dependent data suggest declines in crevalle jack abundance in both regions, we do not have the spatial resolution in the data to assess differences in abundance between the Lower and Upper keys, so our support for this hypothesis is limited.

Hypothesis 3: the decline started sometime after 2005

Our third hypothesis was that the decline began sometime after 2005, which was when 64% of guides started noticing fewer encounters with crevalle jack and fewer individuals when they were encountered (Fig. 3*a*). One caveat of this hypothesis was that the interviewees who have been guiding the longest generally were the ones who reported the decline began the earliest (Fig. 3*b*), suggesting that recollection as to when the decline began depended on fishing experience. Our hypothesis as to when the decline began may therefore be different had we interviewed guides with more or less fishing experience. However, the guides who reported noticing the decline after 2005 had a wide range of experience, with two of them guiding since the 1980s. This lends support to our hypothesis and suggests that something might have happened around 2005 that caused anglers to observe a change in crevalle jack catches. Both the MRIP offshore and ENP datasets showed a decline in standardized abundance after a peak in 1991. Only two guides reported the decline beginning during this period. However, although no dramatic change in abundance on or after 2005 was apparent in the data, both MRIP offshore and ENP CUSUM plots showed that crevalle jack abundance has been below average almost every year since the early 2000s (i.e., negative slopes, Figs. 5*b*, 5*d*, 6*b*), which approximately aligns with when most guides noticed the decline (Fig. 3*a*). Both fisheries-dependent datasets revealed that there has been no sudden change in crevalle jack abundance at a particular point in time, but that the decline has instead been gradual. Guides also reported that the observed decline has been gradual and were unable to point to a particular event that prompted the decline.

Hypothesis 4: fishing harvest in the Florida Keys is a contributor to the decline

Our fourth hypothesis was that fishing harvest in the Florida Keys region above the capacity of fish stock recovery contributed to the decline, based on 86% of guides suggesting either commercial or recreational fishing pressure as likely contributors. This hypothesis did not receive support from fisheries-dependent data (commercial, MRIP, or ENP). Crevalle jack commercial landings in the Upper Keys were relatively high from the mid-1990s to mid-2000s, but then dropped considerably and have been below 10 000 lbs. annually since 2006 (Fig. 7*a*). In the Lower Keys, commercial landings were relatively high for a short period in the late 1980s but have since declined and been below 15 000 lbs. per year since 1990 (Fig. 7*b*). In 1995, commercial entanglement nets were banned in the state of Florida (Smith et al. 2003), which could explain some of the relatively high landings early in the time series. In ENP, the most crevalle jack landed by recreational anglers in a year was 1497 fish in 1984 (just prior to the ban on commercial fishing in ENP in 1985; Osborne et al. 2006), which accounted for 12% of the catch (the remaining 88% of captured crevalle jack were released). Since 1990, landed crevalle jack have only accounted for 5% or less of the total catch reported by recreational anglers (Fig. 7*c*). Similarly, landed crevalle jack have accounted for less than 20% of reported catch in the MRIP dataset since 1991 (Fig. 7*d*). Additionally, the proportion of landed crevalle jack in the MRIP data has declined slightly over time. Data showing low commercial landings and high release rates for recreationally captured fish do not support the suggestion by guides that fishing harvest in the Florida Keys has contributed to the decline in crevalle jack catch over time.

Hypothesis 5: larger fish have declined more than smaller fish

Our fifth hypothesis was that larger fish have declined more so than smaller fish (observed by 71% of the guides who reported a decline in crevalle jack catches). Since crevalle jack captured in the offshore MRIP dataset were significantly larger than crevalle jack captured inshore, inshore fish could be considered a smaller, younger population than offshore fish. This observation aligns with the life history of the species, since juveniles typically inhabit estuarine nursery habitats before presumably migrating to more offshore adult habitats (Smith-Vaniz and Carpenter 2007). Since a decline in CPUE over time was only observed for offshore trips, this lends evidence to the hypothesis that larger, older crevalle jack have declined in abundance more so than smaller, younger fish. However, a decline was also apparent in the ENP dataset, which consists of mostly inshore trips. Since the ENP data did not include size information for crevalle jack, it is unknown whether the decline observed in ENP was specific to larger fish, so this hypothesis may warrant additional research. Again, the MRIP offshore data showed a more dramatic decline over the time series than the ENP data, which lends some additional support to this hypothesis.

Hypothesis 6: crevalle jack are most abundant in the Keys in the winter

Our final hypothesis was that crevalle jack are migratory, with abundance in the Keys region being highest in the winter months, as observed by 72% of guides. This hypothesis was tested by including Season as a fixed factor in the MRIP and ENP standardization models. In the MRIP data, both the probability of encountering crevalle jack (proportion positive model) and the abundance of crevalle jack when caught (positive trips model) were significantly higher in the winter than in the summer inshore ($p < 0.001$; Supplementary Tables S1, S2[1]). Offshore, both the probability of encountering crevalle jack and abundance when caught were significantly higher in the winter than in spring or summer ($p < 0.01$; Supplementary Tables S3, S4[1]). In the ENP data, the probability of encountering crevalle jack was significantly higher in the winter than in the summer ($p < 0.0001$; Supplementary Table S5[1]), and the abundance of crevalle jack when caught was significantly higher in the winter than in the spring, summer, or fall ($p < 0.0001$; Supplementary Table S6[1]). These results support the hypothesis that crevalle jack are most abundant in the Florida Keys in the winter months and appear to display seasonal migration patterns.

Discussion

Translational ecology provides an intentional approach to collaborative, actionable science that can inform and improve decision-making for environmental conservation and management (Enquist et al. 2017). Although the terminology is recent, the concepts and ideas behind translational ecology are not new. In Michel Callon's seminal article on the "sociology of translation" published in 1986, he tells the story of scientists and scallop fishermen in France working together to find solutions to dwindling scallop populations (Callon 1986). Other examples of collaborative research that includes scientists, stakeholders, policymakers, and consumers working in multidisciplinary teams to solve real-world problems appear across numerous disciplines (e.g., Picou 2009; Wethington 2015; Eisenhauer et al. 2021). In fisheries, stakeholder involvement and recognition of fisheries as coupled social–ecological systems are key principles of Ecosystem-Based Management (Long et al. 2015). Many countries are actively working to shift environmental management to a more ecosystem-based approach, recognizing that environmental issues are often too complex and dynamic for conventional management to succeed (Long et al. 2015; O'Higgins et al. 2020). Translational approaches (including LEK and traditional ecological knowledge, or TEK) often provide vital insight when incorporated into Ecosystem-Based Management (Cinner and Aswani 2007; Ruiz-Mallén and Corbera 2013; Stori et al. 2019).

The use of TE in fisheries management is particularly well suited to partnerships with recreational fishing guides. In Florida, where saltwater recreational fishing is diverse and abundant, a US$9.2 billion industry, and a vital aspect of the state's culture (FWC 2018; NMFS 2018), recreational fishing guides can be ideal key informants and translational ecology partners (Santos et al. 2017; Kroloff et al. 2019). Due to frequent interactions with fishes and their environments and a vested interest in fisheries conservation, fishing guide knowledge can be used as a low-cost monitoring program. Developing state or federal fisheries-independent surveys for every unregulated species is not feasible given budgetary and personnel constraints, and even regularly analyzing existing data to ensure stocks are sustainably harvested is beyond the scope of most fishery management programs (Harford and Carruthers 2017; Sagarese et al. 2019). However, by working with the resource users–stakeholders to co-produce actionable science, scientists and managers can rapidly develop research priorities and effective management plans that can promote the sustainability of important fisheries resources.

The results of this study provide evidence of the utility of translational ecology to recreational fisheries management. Collaborating with experienced recreational fishing guides and accessing their local ecological knowledge via semi-structured interviews allowed us to rapidly generate six testable hypotheses concerning the status and trends of the crevalle jack fishery in the Florida Keys. By subsequently analyzing existing fisheries-dependent data, we provided multiple lines of evidence to support acceptance of four of these hypotheses, revealing that crevalle jack populations in the Florida Keys appear to be in decline, and that the decline has been gradual with below-average abundance since the early to mid-2000s. Large, adult fish mainly captured offshore appear to have declined the most, while a less dramatic decline was also observed in the Everglades National Park region. Crevalle jack also appeared to be seasonal residents in the Florida Keys, with the highest abundances observed in the winter months. Unsupported hypotheses helped identify several priorities for future research. These include determining the lifetime movement and migration patterns of the crevalle jack and examining possible factors that contributed to its decline, such as commercial and recreational fishing harvest in other areas throughout the population range.

Biases in LEK and fisheries-dependent data

Our study contributes to ongoing investigations into the utility of LEK and fisheries-dependent data for assessing fish population dynamics (Zukowski et al. 2011; Hind 2015; Aylesworth et al. 2017; Santos et al. 2019). Despite increasing acceptance of these types of data as reliable sources, they nevertheless suffer from several biases and limitations. Fisheries-dependent data are inherently affected by fishing dynamics and angler behavior, altering the relationship between CPUE and abundance (Maunder and Punt 2004). By standardizing the MRIP and ENP data prior to analysis, we accounted for various temporal and spatial dynamics that may influence catch. However, the choice of model used to standardize CPUE data may also influence the resulting abundance index. During preliminary analysis, we fit a series of additional models, including several generalized additive models (GAMs; Hastie and Tibshirani 1990) to the MRIP and ENP datasets. Regardless of the model used, the overall trends in abundance over time remained the same. Since the goal of our study was not to develop standardized indices of abundance for use in stock assessment, but instead to assess overall shifts in abundance over time, the relatively simpler delta lognormal models were chosen for easier interpretation, and for consistency with other studies that have analyzed these datasets (e.g., Cass-Calay and Schmidt 2003; Rios 2015; Sagarese 2019). Further, because the crevalle jack is a large, pelagic species with a distribution spanning the Western Atlantic, Caribbean, and the Gulf of Mexico regions, an index of abundance using data only compiled from the Florida Keys region would not be suitable for stock assessment nor management outside of the Florida Keys.

In this study, we observed a mismatch between the ENP data and the guide observations that could be attributed to limitations of the data. Although a decline in crevalle jack abundance was apparent in the ENP data, 50% of Upper Keys guides did not observe a decline. A limitation specific to recreational angler-reported data is that the accuracy of the data is dependent upon angler recall and willingness to report everything that was released. Since the majority of crevalle jack caught by recreational anglers in the Florida Keys are released, it is possible that the MRIP and ENP datasets are conservative estimates of the crevalle jack catch. It is also possible that the decline in standardized abundance over time represents a decline in angler willingness to report catches for this species or a bias in reporting, rather than representing a true decline in population abundance. Alternatively, the Upper Keys guides may have failed to perceive the decline because it was gradual and there was substantial interannual variability in abundance, as previously mentioned. Otherwise, our small sample size may have simply given us a biased view of the perceptions of Upper Keys guides.

A common criticism of LEK is that resource users are subject to the shifting baseline syndrome (SBS), a well-known phenomenon describing how human perception and memory are subject to how much experience an individual has with historic reference conditions (Pauly 1995). Numerous studies have shown that changing perceptions over time can lead to acceptance of degraded environmental conditions as normal, thus making target setting for species and habitat recovery difficult (Beaudreau and Levin 2014). For example, Barbosa-Filho et al. (2020) interviewed lane snapper (*Lutjanus synagris*) anglers in Brazil and found that older anglers were significantly more likely to report a decline in abundance over time than younger anglers. This shifting baseline contributed to disagreement among anglers as to whether specific management rules for lane snapper were necessary. SBS may explain why some guides we interviewed did not observe a decline in crevalle jack catches or why most guides did not notice the start of the decline (circa 1991 according to the MRIP offshore and ENP data) but noticed when abundance started dipping below average (early 2000s). This is supported by our observation that some of the anglers who have been guiding the longest suggested the decline began the earliest (Fig. 3*b*). Also, the three Upper Keys guides who did not observe a decline have only been guiding full time since the early 2000s. Since these anglers were not guiding during the period of relatively high abundance observed in the ENP data (1990–2000; Fig. 6*a*), it is possible they have not been fishing for crevalle jack long enough to notice the decline. Other LEK studies have found similar patterns, with older or more experienced anglers being more likely to report changes in abundance over time for a given species than younger or less experienced anglers (Beaudreau and Levin 2014; Frezza and Clem 2015).

Supported hypotheses

When rigorous fisheries-independent data are lacking, consistency in ecological patterns among multiple, independent yet imperfect datasets (e.g., fisheries-dependent data, LEK or expert opinion, gear selective data) can provide confidence in observed patterns and bolster the reliability and credibility of LEK-based research (Rehage et al. 2019). Several studies have demonstrated such consistencies between LEK and other data sources in a variety of applications (e.g., Poizat and Baran 1997; Aswani and Hamilton 2004; Zukowski et al. 2011; Santos et al. 2019; Bourdouxhe et al. 2020) and even shown that LEK data can provide better insights with less effort and lower costs than traditional data sources (e.g., fisheries-independent surveys; Aylesworth et al. 2017). In our study, four of our six LEK-derived hypotheses received clear support from fisheries-dependent data and provide critical information that can be used to develop management recommendations. The agreement among guides, MRIP data, and ENP data concerning the timing (below average abundance since the mid-2000s) and pattern of decline (gradual decrease mainly affecting large, adult offshore fish) provides confidence in these observed patterns and demonstrates the utility of our translational ecology approach. Although the exact cause of the crevalle jack decline remains unknown, declines in large, old fish are often indicative of overfishing and can have severe consequences for fish populations and ecosystems (Heino and Godø 2002). Disproportionate removal of large fish truncates age and size distributions, leaving only younger spawners that typically produce lower quality eggs and larvae than older spawners (Green 2008). Large, old female fish in particular contribute substantially to stock productivity and sustainability by ensuring reproductive success (Hixon et al. 2014). Removal of large fish from an ecosystem can also lead to cascading effects on other species and the environment, such as increasing abundance and altering behavior of prey fish (Baum and Worm 2009). Our analysis of fisheries-dependent data supported fishing guide concerns about the status and trends of crevalle jack in the Florida Keys, which suggests that implementing regulations to restore the population may be warranted. Since our results revealed that larger adults appear to be more at risk and crevalle jack make seasonal migrations into the Florida Keys, such management regulations could include a size or slot limit to protect large fish or a rotational closure. Slot limits have been particularly successful at preventing significant truncation of age or size structure while maintaining high fishery yields (Gwinn et al. 2015; Kasper et al. 2020). Rotational closures can be beneficial for preserving spatial heterogeneity in populations, especially when the closures protect vital spawning areas or seasons (Hsieh et al. 2010; Brownscombe et al. 2019*a*). If movement data reveals regular crevalle jack migrations to spawning grounds either in the Florida Keys or elsewhere, closure of these areas during the spawning season could aid in population recovery.

Unsupported hypotheses

While consistency among LEK and biological datasets provides confidence in observed ecological patterns, disagreement among data sources provides focal areas for future research that can elucidate patterns and mechanisms driving the discrepancies originally observed (Silvano and Valbo-Jørgensen 2008). Out of our six hypotheses, two had limited support from fisheries-dependent data and may serve as priorities for future research. These hypotheses concerned the spatial extent of the decline and the causes of the decline. The hypothesis that crevalle jack populations have declined more in the Lower Keys than the Upper Keys (Hypothesis 2) did not have much support from fisheries-dependent data, but we also lacked the spatial resolution to accurately assess this hypothesis (MRIP data encompassed the entire Keys region and ENP data only partially encompassed the Upper Keys). Furthermore, the extent of mixing between fish in the Lower and Upper keys is unknown, which makes pinpointing the extent of the decline difficult. Preliminary results of an acoustic telemetry study in South Florida showed several crevalle jack detected on both Upper Keys and Lower Keys receivers (C.L. Gervasi and J.S. Rehage, unpublished data), suggesting that there may be at least some mixing between the two regions. To fully determine the extent of the decline, research into the spatial distribution of crevalle jack (including seasonal and lifetime movement and migration patterns) will be necessary. Many marine fishes are known to exhibit complex movement patterns which have important implications for fisheries management (Zemeckis et al. 2017). For example, knowledge of fish movements is necessary for elucidating stock structure, and accurate stock identification is crucial for developing biologically relevant management unit boundaries (Pita et al. 2016; Cadrin 2020).

Fish movements can also affect their vulnerability to fishing pressure (Olsen et al. 2012), which could help explain the disagreement among datasets concerning Hypothesis 4. Most guides speculated that commercial and (or) recreational fishing harvest may have contributed to the crevalle jack decline, but fisheries-dependent data showed that commercial harvest of crevalle jack in the Florida Keys region is limited, and the majority of fish captured by recreational anglers are released. Furthermore, most recreational anglers use hook and line gear to capture crevalle jack (97% of MRIP trips and 100% of ENP trips that captured crevalle jack were hook and line). Although discard mortality rates have not been estimated for crevalle jack specifically, studies on other recreationally captured species in the region show that hook and line discard mortality is generally low (Flaherty-Walia et al. 2016). So, it seems unlikely that fishing pressure in the Florida Keys (either commercial or recreational) has contributed to the decline in crevalle jack abundance. We cannot, however, exclude the possibility of unreported bycatch discard mortality in the commercial fishery or higher harvest than reported in the recreational fishery. Gillnets were the second most common gear type used on commercial trips that landed crevalle jack. Dotson et al. (2009) examined bycatch of black crappie (*Pomoxis nigromaculatus*) in a Florida commercial gillnet fishery and found that bycatch mortality exceeded 30% in both years of the study. However, commercial gillnets were banned throughout most of Florida's coastal waters in 1995 (Smith et al. 2003), so bycatch mortality from this gear type would have only affected the earlier years of the time series. Nonetheless, investigating the extent of commercial bycatch mortality for crevalle jack may be needed before we can exclude the Monroe County commercial fishery as a contributor to the decline. As previously noted, crevalle jack are most abundant in the Keys region in the winter months, which agreed with guide observations and suggests that the species migrates elsewhere during the summer months. Preliminary acoustic telemetry results (C.L. Gervasi and J.S. Rehage, unpublished data) have revealed that crevalle jack make regular long-range movements into and out of the Florida Keys, moving northward along both the east and west coasts of Florida and occasionally migrating from one coast to another. It is therefore also possible that crevalle jack are being overfished somewhere else within the population range, like central Florida, where most of the crevalle jack commercial landings have been from over the past five years (C. Bradshaw, personal observation).

Other factors may also be contributing to the crevalle jack decline, such as loss of prey, poor water quality, or increased predator abundance, as several guides suggested. Loss of prey was the most common suggestion by guides, many of whom mentioned changes in ballyhoo (*Hemiramphus brasiliensis*) abundance as a possible factor. Pelagic schooling fishes are common prey for adult crevalle jack (Kwei 1978; Correia et al. 2017). Changes in the abundance or distribution of ballyhoo populations may therefore explain why larger, offshore crevalle jack appear to have declined more so than smaller, inshore crevalle jack. Additionally, regional climate variability has been linked to changes in the distribution and productivity of several fish species (Brander 2007), and continued temperature increases and declines in primary production due to climate change are anticipated to cause global decreases in marine animal biomass (Lotze et al. 2019). Research on bonefish in South Florida (another economically important recreational fishery species) has revealed that several factors likely contributed to a population decline that began in the 1950s (Frezza and Clem 2015; Santos et al. 2017), including habitat loss and modification, extreme weather events, and fishing mortality (Brownscombe et al. 2019*b*). In Florida Bay specifically, shifts in both recruit and adult survivorship of bonefish may have been caused by increased fishing effort, changes in abiotic factors, and (or) habitat changes (Klarenberg et al. 2019). Therefore, in addition to examining the spatial distribution of crevalle jack, another future research priority is to examine the varying factors operating at several spatial scales that affect crevalle jack populations.

Conclusions

In an era of global anthropogenic and climatic changes, new methods of environmental planning and management are necessary to ensure sustainable use and conservation of natural resources (Lipsman 2019). Combining multiple data sources and including stakeholders in science co-production under a translational ecology framework provides opportunities for rapid, proactive, and adaptive management (Chapin 2017; Zipkin and Saunders 2018). In our study, LEK-derived hypotheses supported by multiple data sources supplied key information that fisheries managers can use immediately to aid in the conservation of crevalle jack. These hypotheses suggest that (1) populations in the Florida Keys have been declining gradually with below-average abundance since the early 2000s, (2) large, old fish have declined more than small fish, and (3) crevalle jack are most abundant in the Keys region in the winter months. Additionally, our unsupported hypotheses led us to two main priorities for future research: (1) analysis of the seasonal and lifetime movement and migration patterns of Florida Keys crevalle jack and (2) examination into the anthropogenic and environmental dynamics and drivers occurring throughout the population range that may be responsible for the decline. These two hypotheses were most likely unsupported because of limitations in the available fisheries-dependent data, not because the guides we interviewed were incorrect in their observations. In fact, the general agreement among guides and the different data sources reveals that recreational fishing guide knowledge is an excellent source of information that has the potential to substantially improve fisheries management.

The results of our study outline an effective translational ecology approach that can be progressed by including fishing guides in future research efforts and in developing management recommendations via transdisciplinary research (Pohl 2008; Klenk 2018). Furthermore, our framework (Fig. 1) can be easily applied to other species and areas, as fishing guides typically encounter a wide variety of fishes as they tailor their charter trips to diverse clients. Most saltwater fish species native to Florida currently have no specific recreational regulations and are listed as unregulated species (FWC 2021*c*), including several species often targeted by recreational anglers. Formal stock assessments have not been conducted for most of these species and are not expected to be conducted in the future. Therefore, trends in abundance patterns are not being monitored, and the effects of fishing pressure on these populations remain unknown. Through collaborative efforts, there are a few instances where scientific results have successfully informed management changes for important recreational fisheries (e.g., Brownscombe et al. 2019*a*). However, knowledge-action gaps are still common (Cook et al. 2013) and there is ample room for increased collaboration among guides, angler associations, fisheries scientists, conservation groups, and managers. The translational ecology approach outlined herein provides an additional tool for the fishery scientist's toolbox that can help better develop conservation priorities and effective management.

Competing interests

The authors declare there are no competing interests.

Acknowledgements

The authors extend their sincere thanks to all the South Florida anglers and guides who participated in interviews and shared their experiences and passion for fishing. Special thanks go to Captains D. Kilpatrick and N. Linville, whose concern for crevalle jack initiated this project and who helped coordinate the interview process. The Lower Keys Guides Association (LKGA) and the Florida Keys Fishing Guides Association (FKFGA) assisted with identifying key informants for interviews, J. Heinen (FIU) aided in developing interview questions, and J. Kalafarski (National Park Service) provided the ENP data for the Gulf Coast. Much appreciation goes to A. Rios (NOAA Fisheries) and the entire FWC stock assessment group for their invaluable advice on analyzing the MRIP data, and to the two anonymous reviewers for comments that improved the manuscript. This work was reviewed and deemed exempt by Florida International University's Institutional Review Board (IRB Protocol exemption No. IRB-19-0157, 7 May 2019). Funding was provided by the Lower Keys Guides Association, private donation, the Everglades Foundation FIU ForEverglades Graduate Scholarship, and the International Women's Fishing Association (IWFA). This study was developed in collaboration with the Florida Coastal Everglades Long-Term Ecological Research program under National Science Foundation Grant No. DEB-1832229. This is contribution #1361 from the Institute of Environment at Florida International University.

References

Adler, R.W. 2020. Translational ecology and environmental law. Environ. Law, **50**: 703. doi:10.2139/ssrn.3451593.

Allison, T., and Arnold, A. 2018. Wind energy and wildlife: 20 years of translational ecology in action. Front. Ecol. Environ. **16**(6): 318–319. doi:10.1002/fee.1919.

Anadón, J., Giménez, A., Ballestar, R., and Pérez, I. 2009. Evaluation of local ecological knowledge as a method for collecting extensive data on animal abundance. Conserv. Biol. **23**(3): 617–625. doi:10.1111/j.1523-1739.2008.01145.x. PMID:19183211.

Angeoletto, F., da Silva Leandro, D., and Aguiar Paludetto, N. 2019. Ecologia Translacional: informações ambientais aplicadas ao Planejamento e Gestão de Rondonópolis, MT, Brasil. Terra Plural, **13**(3): 461–469. doi:10.5212/TerraPlural.v.13i3.0030.

Arlinghaus, R., Abbott, J.K., Fenichel, E.P., Carpenter, S.R., Hunt, L.M., Alós, J., et al. 2019. Governing the recreational dimension of global fisheries. Proc. Natl. Acad. Sci. U.S.A. **116**(12): 5209–5213. doi:10.1073/pnas.1902796116. PMID:30890655.

Arlinghaus, R., Tillner, R., and Bork, M. 2015. Explaining participation rates in recreational fishing across industrialised countries. Fish. Manage. Ecol. **22**(1): 45–55. doi:10.1111/fme.12075.

Aswani, S., and Hamilton, R.J. 2004. Integrating indigenous ecological knowledge and customary sea tenure with marine and social science for conservation of bumphead parrotfish (*Bolbometopon muricatum*) in the Roviana Lagoon, Solomon Islands. Environ. Conserv. **31**(1): 69–83. doi:10.1017/S037689290400116X.

Atkinson, R., and Flint, J. 2001. Accessing hidden and hard-to-reach populations: Snowball research strategies. Soc. Res. Update, **33**: 1–4. Available from http://sru.soc.surrey.ac.uk/SRU33.html.

Aylesworth, L., Phoonsawat, R., Suvanachai, P., and Vincent, A.C.J. 2017. Generating spatial data for marine conservation and management. Biodivers. Conserv. **26**(2): 383–399. doi:10.1007/s10531-016-1248-x.

Babcock, E.A. 2013. Updated index of abundance for shortfin mako sharks from the U.S. Marine Recreational Fisheries Statistics Survey. Collect. Vol. Sci. Pap. ICCAT, **69**(4): 1639–1646.

Barbosa-Filho, M.L.V., Souza, G.B.G., Lopes, S.F., Siciliano, S., Hauser Davis, R.A., and Mourão, J.S. 2020. Evidence of shifting baseline and fisher judgment on lane snapper (*Lutjanus synagris*) management in a Brazilian marine protected area. Ocean Coast. Manage. **183**: 105025. doi:10.1016/j.ocecoaman.2019.105025.

Baum, J.K., and Worm, B. 2009. Cascading top-down effects of changing oceanic predator abundances. J. Anim. Ecol. **78**(4): 699–714. doi:10.1111/j.1365-2656.2009.01531.x. PMID:19298616.

Beaudreau, A.H., and Levin, P.S. 2014. Advancing the use of local ecological knowledge for assessing data-poor species in coastal ecosystems. Ecol. Appl. **24**(2): 244–256. doi:10.1890/13-0817.1. PMID:24689138.

Berry, F.H. 1959. Young Jack Crevalles (*Caranx* species) off the southeastern Atlantic coast of the United States. Fishery Bulletin 152. Washington, D.C. Available from https://www.st.nmfs.noaa.gov/spo/FishBull/59-1/berry.pdf [accessed 28 January 2019].

Bohnsack, J.A. 2011. Impacts of Florida coastal protected areas on recreational world records for spotted seatrout, red drum, black drum, and common snook. BMS, **87**(4): 939–970. doi:10.5343/bms.2010.1072.

Bourdouxhe, A., Duflot, R., Radoux, J., and Dufrêne, M. 2020. Comparison of methods to model species habitat networks for decision-making in nature conservation: The case of the wildcat in southern Belgium. J. Nat. Conserv. **58**: 125901. doi:10.1016/j.jnc.2020.125901.

Brander, K.M. 2007. Global fish production and climate change. Proc. Natl. Acad. Sci. U.S.A. **104**(50): 19709–19714. doi:10.1073/pnas.0702059104. PMID:18077405.

Bray, J.R., and Curtis, J.T. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. **27**(4): 325–349. doi:10.2307/1942268.

Brooks, T.M., Mittermeier, R.A., Da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., et al. 2006. Global biodiversity conservation priorities. Science, **313**: 58–61. doi:10.1126/science.1127609. PMID:16825561.

Brown, S.T. 1999. Trends in the commercial and recreational shark fisheries in Florida, 1980–1992, with implications for management. N. Am. J. Fish. Manage. **19**: 28–41. doi:10.1577/1548-8675(1999)019<0028:TITCAR>2.0.CO;2.

Brownscombe, J.W., Adams, A.J., Young, N., Griffin, L.P., Holder, P.E., Hunt, J., et al. 2019*a*. Bridging the knowledge-action gap: A case of research rapidly impacting recreational fisheries policy. Mar. Pol. **104**: 210–215. doi:10.1016/j.marpol.2019.02.021.

Brownscombe, J.W., Danylchuk, A.J., Adams, A.J., Black, B., Boucek, R., Power, M., et al. 2019*b*. Bonefish in South Florida: status, threats and research needs. Environ. Biol. Fish. **102**: 329–348. doi:10.1007/s10641-018-0820-5.

Cadrin, S.X. 2020. Defining spatial structure for fishery stock assessment. Fish. Res. **221**: 105397. doi:10.1016/j.fishres.2019.105397.

Caiafa, H.I., Narváez, B.J., and Borrero, F.S. 2011. Algunos aspectos de la dinámica poblacional del jurel *Caranx hippos* (Pisces: Carangidae) en Bocas de Ceniza, Caribe colombiano. Rev. MVZ Cordoba, **16**(1): 2324–2335.

Callon, M. 1986. Éléments pour une sociologie de la traduction. L'Année Sociol. **36**: 169–208.

Cardoso da Silva, L. D M., Machado, I.C., dos Santos Tutui, S.L., and Tomás, A.R.G. 2020. Local ecological knowledge (LEK) concerning snook fishers on estuarine waters: Insights into scientific knowledge and fisheries management. Ocean Coast. Manage. **186**: 105088. doi:10.1016/j.ocecoaman.2019.105088.

Carlson, J., and Osborne, J. 2013. Standardized catch rates of bonnetheads from the Everglades National Park Creel Survey, 1978-2004. SEDAR34-WP-02. SEDAR, North Charleston, S.C.

Carlson, J.K., Osborne, J., and Schmidt, T.W. 2007. Monitoring the recovery of smalltooth sawfish, *Pristis pectinata*, using standardized relative indices of abundance. Biol. Conserv. **136**(2): 195–202. doi:10.1016/j.biocon.2006.11.013.

Cass-Calay, S.L. 2012. A standardized catch rate index for yellowfin tuna (*Thunnus albacares*) from the U.S. recreational fishery in the. Western North Atlantic Ocean. Collect. Vol. Sci. Pap. ICCAT, **68**(3): 953–966.

Cass-Calay, S.L., and Schmidt, T. 2003. Standardized catch rates of juvenile goliath grouper, *Epinephelus itajara*, from the Everglades National Park Creel Survey, 1973–1999. SEDAR6-RW-2. SEDAR, North Charleston, S.C.

Cass-Calay, S.L., and Schmidt, T.W. 2009. Monitoring changes in the catch rates and abundance of juvenile goliath grouper using the ENP creel survey, 1973–2006. Endang. Species Res. **7**(3): 183–193. doi:10.3354/esr00139.

Chapin, F.S. 2017. Now is the time for translational ecology. Front. Ecol. Environ. **15**(10): 539–539. doi:10.1002/fee.1737.

Chen, X., and Jin, H. 2019. A case study of enhancing sustainable intensification of Chinese Torreya Forest in Zhuji of China. Environ. Nat. Resour. Res. **9**(2): 53–60. doi:10.5539/enrr.v9n2p53.

Cinner, J.E., and Aswani, S. 2007. Integrating customary management into marine conservation. Biol. Conserv. **140**(3–4): 201–216. doi:10.1016/j.biocon.2007.08.008.

Coleman, F.C., Figueira, W.F., Ueland, J.S., and Crowder, L.B. 2004. The impact of United States recreational fisheries on marine fish populations. Science, **305**(5692): 1958–1960. doi:10.1126/science.1100397. PMID:15331771.

Cook, C.N., Mascia, M.B., Schwartz, M.W., Possingham, H.P., and Fuller, R.A. 2013. Achieving conservation science that bridges the knowledge–action boundary. Conserv. Biol. **27**(4): 669–678. doi:10.1111/cobi.12050. PMID:23574343.

Cook, C.N., Wintle, B.C., Aldrich, S.C., and Wintle, B.A. 2014. Using strategic foresight to assess conservation opportunity. Conserv. Biol. **28**(6): 1474–1483. doi:10.1111/cobi.12404. PMID:25381735.

Cooke, S.J., and Cowx, I.G. 2004. The role of recreational fishing in global fish crises. Bioscience, **54**(9): 857–859. doi:10.1641/0006-3568(2004)054[0857:TRORFI]2.0.CO;2.

Correia, E., Granadeiro, J.P., Regalla, A., Dias, E., Almeida, A., and Catry, P. 2017. Predatory pelagic fishes of the Bijagós Archipelago (Guinea-Bissau) show high overlap in diets dominated by sardinella. Afr. J. Mar. Sci. **39**(4): 389–396. doi:10.2989/1814232X.2017.1387605.

Crotty, S.M., Altieri, A.H., Bruno, J.F., Fischman, H., and Bertness, M.D. 2019. The foundation for building the conservation capacity of community ecology. Front. Mar. Sci. **6**: 1–8. doi:10.3389/fmars.2019.00238.

Davis, G., and Thue, E.B. 1979. Fishery data management handbook, Everglades National Park. Rep T-546. US National Park Service, Homestead, Fla.

Ditty, J.G., Shaw, R., and Cope, J.S. 2004. Distribution of carangid larvae (Teleostei: Carangidae) and concentrations of zooplankton in the northern Gulf of Mexico, with illustrations of early *Hemicaranx amblyrhynchus* and *Caranx* spp. larvae. Mar. Biol. **145**: 1001–1014. doi:10.1007/s00227-004-1381-z.

Dongol, Y., and Heinen, J.T. 2012. Pitfalls of CITES implementation in Nepal: a policy gap analysis. Environ. Manage. **50**: 181–190. doi:10.1007/s00267-012-9896-4.

Dotson, J.R., Allen, M.S., Johnson, W.E., and Benton, J. 2009. Impacts of commercial gill-net bycatch and recreational fishing on a Florida black crappie population. N. Am. J. Fish. Manage. **29**(5): 1454–1465. doi:10.1577/M08-137.1.

Eisenhauer, E., Williams, K.C., Margeson, K., Paczuski, S., Hano, M.C., and Mulvaney, K. 2021. Advancing translational research in environmental science: The role and impact of social sciences. Environ. Sci. Pol. **120**: 165–172. doi:10.1016/j.envsci.2021.03.010.

Enquist, C.A., Jackson, S.T., Garfin, G.M., Davis, F.W., Gerber, L.R., Littell, J.A., et al. 2017. Foundations of translational ecology. Front. Ecol. Environ. **15**(10): 541–550. doi:10.1002/fee.1733.

Esri. 2011. Light Gray Canvas. Sources: Esri, HERE, Garmin, INCREMENT P, © OpenStreetMap contributors, and the GIS user community. Available from https://www.arcgis.com/home/item.html?id=8b3d38c0819547faa83f7b7aca80bd76.

Esri. 2020. ArcGIS Desktop: Release 10. Environmental Systems Research Institute, Redlands, Calif.

Fedler, T. 2009. The economic impact of recreational fishing in the Everglades region. Available from https://www.bonefishtarpontrust.org/downloads/research-reports/stories/everglades-economics-report.pdf [accessed 8 April 2019].

Fedler, T. 2013. Economic impact of the Florida Keys flats fishery. Vero Beach, Fla. Available from https://lkga.org/wp-content/uploads/2017/05/btt-keys-economic-report.pdf [accessed 9 May 2021].

Felizola Freire, K.M., Belhabib, D., Espedido, J.C., Hood, L., Kleisner, K.M., Lam, V. W.L., et al. 2020. Estimating global catches of marine recreational fisheries. Front. Mar. Sci. **7**(12): 1–18. doi:10.3389/fmars.2020.00012. PMID:32802822.

Flaherty-Walia, K.E., Winner, B.L., Tyler-Jedlund, A.J., and Davis, J.P. 2016. Short-term discard mortality estimates for Gray Snapper in a West-Central Florida estuary and adjacent nearshore Gulf of Mexico waters. N. Am. J. Fish. Manage. **36**(2): 329–340. doi:10.1080/02755947.2015.1129003.

Frezza, P.E., and Clem, S.E. 2015. Using local fishers' knowledge to characterize historical trends in the Florida Bay bonefish population and fishery. Environ. Biol. Fishes, **98**: 2187–2202. doi:10.1007/s10641-015-0442-0.

FWC. 2018. The economic impacts of saltwater fishing in Florida. Florida Fish and Wildlife Conservation Commission. Available from https://myfwc.com/conservation/value/saltwater-fishing/ [accessed 4 January 2021].

FWC. 2021*a*. Commercial fisheries landings in Florida. Florida Fish and Wildlife Conservation Commission. Available from https://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/ [accessed 25 May 2021].

FWC. 2021*b*. Florida saltwater recreational fishing regulations. Florida Fish and Wildlife Conservation Commission. Available from http://www.eregulations.com/wp-content/uploads/2020/12/21FLSW_JAN-VLR.pdf [accessed 4 January 2021].

FWC. 2021*c*. "Unregulated" species. Florida Fish and Wildlife Conservation Commission. Available from https://myfwc.com/fishing/saltwater/recreational/unregulated [accessed 25 May 2021].

Green, B.S. 2008. Maternal effects in fish populations. Adv. Mar. Biol. **54**: 1–105. doi:10.1016/S0065-2881(08)00001-1. PMID:18929063.

Guest, G., Bunce, A., and Johnson, L. 2006. How many interviews are enough?: An experiment with data saturation and variability. Field Methods, **18**(1): 59–82. doi:10.1177/1525822X05279903.

Gwinn, D.C., Allen, M.S., Johnston, F.D., Brown, P., Todd, C.R., and Arlinghaus, R. 2015. Rethinking length-based fisheries regulations: The value of protecting old and large fish with harvest slots. Fish Fish. **16**(2): 259–281. doi:10.1111/faf.12053.

Harford, W.J., and Carruthers, T.R. 2017. Interim and long-term performance of static and adaptive management procedures. Fish. Res. **190**: 84–94. doi:10.1016/j.fishres.2017.02.003.

Hastie, T., and Tibshirani, R. 1990. Generalized additive models. CRC Press, London.

Hawkins, D.M., and Olwell, D.H. 1998. Cumulative sum charts and charting for quality improvement. *Edited by* P. Green and J.F. Lawless. Springer-Verlag, New York. doi:10.2307/2670188.

Heinen, J.T., and Shrestha-Acharya, R. 2011. The non-timber forest products sector in Nepal: emerging policy issues in plant conservation and utilization for sustainable development. J. Sustain. For. **30**(6): 543–563. doi:10.1080/10549811.2011.567376.

Heino, M., and Godø, O.R. 2002. Fisheries-induced selection pressures in the context of sustainable fisheries. Bull. Mar. Sci. **70**(2): 639–656. doi:10.3389/fmars.2019.00238.

Hennink, M.M., Kaiser, B.N., and Marconi, V.C. 2017. Code saturation versus meaning saturation: How many interviews are enough? Qual. Health Res. **27**(4): 591–608. doi:10.1177/1049732316665344. PMID:27670770.

Heyman, W.D., and Kjerfve, B. 2008. Characterization of transient multi-species reef fish spawning aggregations at Gladden Spit, Belize. Bull. Mar. Sci. **83**(3): 531–551.

Hind, E.J. 2015. A review of the past, the present, and the future of fishers' knowledge research: a challenge to established fisheries science. ICES J. Mar. Sci. **72**(2): 341–358. doi:10.1093/icesjms/fsu169.

Hixon, M.A., Johnson, D.W., and Sogard, S.M. 2014. BOFFFFs: on the importance of conserving old-growth age structure in fishery populations. ICES J. Mar. Sci. **71**(8): 2171–2185. doi:10.1093/icesjms/fst200.

Holder, P.E., Jeanson, A.L., Lennox, R.J., Brownscombe, J.W., Arlinghaus, R., Danylchuk, A.J., et al. 2020. Preparing for a changing future in recreational fisheries: 100 research questions for global consideration emerging from a horizon scan. Rev. Fish Biol. Fish. **30**(1): 137–151. doi:10.1007/s11160-020-09595-y.

Hsieh, C.H., Yamauchi, A., Nakazawa, T., and Wang, W.F. 2010. Fishing effects on age and spatial structures undermine population stability of fishes. Aquat. Sci. **72**(2): 165–178. doi:10.1007/s00027-009-0122-2.

IGFA. 2006. World record game fishes. 2006 ed. International Game Fish Association, Dania Beach, Fla.

Kasper, J.M., Brust, J., Caskenette, A., McNamee, J., Vokoun, J.C., and Schultz, E.T. 2020. Using harvest slot limits to promote stock recovery and broaden age structure in marine recreational fisheries: A case study. N. Am. J. Fish. Manage. **40**(6): 1451–1471. doi:10.1002/nafm.10511.

Keatley, M.R., and Hudson, I.L. 2012. Detecting change in an Australian flowering record: Comparisons of linear regression and cumulative sum analysis change point analysis. Austral Ecol. **37**(7): 825–835. doi:10.1111/j.1442-9993.2011.02344.x.

Klarenberg, G., Ahrens, R., Shaw, S., and Allen, M. 2019. Use of a dynamic population model to estimate mortality and recruitment trends for Bonefish in Florida bay. Environ. Biol. Fish. **102**(2): 349–363. doi:10.1007/s10641-018-0805-4.

Klenk, N. 2018. From network to meshwork: Becoming attuned to difference in transdisciplinary environmental research encounters. Environ. Sci. Pol. **89**: 315–321. doi:10.1016/j.envsci.2018.08.007.

Koch, C.F. 1987. Prediction of sample size effects on the measured temporal and geographic distribution patterns of species. Paleobiology, **13**(1): 100–107. doi:10.1017/S0094837300008617.

Kroloff, E.K.N., Heinen, J.T., Braddock, K.N., Rehage, J.S., and Santos, R.O. 2019. Understanding the decline of catch-and-release fishery with angler knowledge: a key informant approach applied to South Florida bonefish. Environ. Biol. Fishes, **102**: 319–328. doi:10.1007/s10641-018-0812-5.

Kwei, E.A. 1978. Food and spawning activity of *Caranx hippos* (L.) off the coast of Ghana. J. Nat. Hist. **12**(2): 195–215. doi:10.1080/00222937800770081.

Lawson, D.M., Hall, K.R., Yung, L., and Enquist, C.A. 2017. Building translational ecology communities of practice: insights from the field. Front. Ecol. Environ. **15**(10): 569–577. doi:10.1002/fee.1736.

Lester, N.P., Marshall, T.R., Armstrong, K., Dunlop, W.I., and Ritchie, B. 2003. A broad-scale approach to management of Ontario's recreational fisheries. N. Am. J. Fish. Manage. **23**(4): 1312–1328. doi:10.1577/M01-230AM.

Levesque, J.C. 2009. Establishment and closure of the directed Florida whiting (*Menticirrhus americanus*) fishery and implications for North Atlantic right whale (*Eubalaena glacialis*) conservation and management. Mar. Pol. **33**(2): 233–247. doi:10.1016/j.marpol.2008.07.003.

Lewin, W.C., Arlinghaus, R., and Mehner, T. 2006. Documented and potential biological impacts of recreational fishing: Insights for management and conservation. Rev. Fish. Sci. **14**(4): 305–367. doi:10.1080/10641260600886455.

Lipsman, J.E. 2019. Local knowledge and democracy in fisheries management: A case study of adaptation to the anthropocene in southeast Louisiana. Ecol. Soc. **24**(4): 20. doi:10.5751/ES-11100-240420.

Lo, N.C., Jacobson, L.D., and Squire, L. 1992. Indices of relative abundance from fish spotter data based on delta-lognormal models. Can. J. Fish. Aquat. Sci. **49**(12): 2515–2526. doi:10.1139/f92-278.

Long, R.D., Charles, A., and Stephenson, R.L. 2015. Key principles of marine ecosystem-based management. Mar. Pol. **57**: 53–60. doi:10.1016/j.marpol.2015.01.013.

Lotze, H.K., Tittensor, D.P., Bryndum-Buchholz, A., Eddy, T.D., Cheung, W.W.L., Galbraith, E.D., et al. 2019. Global ensemble projections reveal trophic amplification of ocean biomass declines with climate change. Proc. Natl. Acad. Sci. U.S.A. **116**(26): 12907–12912. doi:10.1073/pnas.1900194116. PMID:31186360.

Mac Nally, R., and Hart, B.T. 1997. Use of CUSUM methods for water-quality monitoring in storages. Environ. Sci. Technol. **31**(7): 2114–2119. doi:10.1021/es9609516.

Manly, B.F.J., and Mackenzie, D. 2000. A cumulative sum type of method for environmental monitoring. Environmetrics, **11**(2): 151–166. doi:10.1002/(SICI)1099-095X(200003/04)11:2<151::AID-ENV394>3.0.CO;2-B.

Maunder, M.N., and Punt, A.E. 2004. Standardizing catch and effort data: A review of recent approaches. Fish. Res. **70**: 141–159. doi:10.1016/j.fishres.2004.08.002.

Maunder, M.N., Sibert, J.R., Fonteneau, A., Hampton, J., Kleiber, P., and Harley, S.J. 2006. Interpreting catch per unit effort data to assess the status of individual stocks and communities. ICES J. Mar. Sci. **63**(8): 1373–1385. doi:10.1016/j.icesjms.2006.05.008.

McBride, R., and McKown, K. 2000. Consequences of dispersal of subtropically spawned crevalle jacks, *Caranx hippos*, to temperate estuaries. Fish. Bull. **98**: 528–538.

McClenachan, L., and Kittinger, J.N. 2013. Multicentury trends and the sustainability of coral reef fisheries in Hawai'i and Florida. Fish Fish. **14**(3): 239–255. doi:10.1111/j.1467-2979.2012.00465.x.

Muellmann, S., Brand, T., Jürgens, D., Gansefort, D., and Zeeb, H. 2021. How many key informants are enough? Analysing the validity of the community readiness assessment. BMC Res. Notes, **14**(1): 1–6. doi:10.1186/s13104-020-05413-7. PMID:33407799.

Mueter, F.J., and Norcross, B.L. 2000. Changes in species composition of the demersal fish community in nearshore waters of Kodiak Island, Alaska. Can. J. Fish. Aquat. Sci. **57**(6): 1169–1180. doi:10.1139/f00-051.

Nelder, J., and Wedderburn, R. 1972. Generalised linear models. J. R. Stat. Soc. **A135**: 370–384. doi:10.2307/2344614.

NMFS. 2018. Fisheries of the United States, 2017. NOAA Current Fishery Statistics No. 2017. National Marine Fisheries Service, US Department of Commerce, Silver Spring, Md. Available from https://www.fisheries.noaa.gov/resource/document/fisheries-united-states-2017-report.

NOAA. 2021. About the Marine Recreational Information Program. National Oceanic and Atmospheric Administration. Available from https://www.fisheries.noaa.gov/recreational-fishing-data/about-marine-recreational-information-program [accessed 25 May 2021].

NPS. 2015. Marine and estuarine resources management. National Park Service. Available from https://www.nps.gov/ever/learn/nature/marine.htm [accessed 25 May 2021].

O'Higgins, T., Lago, M., and DeWitt, T. (*Editors*). 2020. Ecosystem-based management, ecosystem services and aquatic biodiversity. Springer Nature, Switzerland. doi:10.1007/978-3-030-45843-0.

Olsen, E.M., Heupel, M.R., Simpfendorfer, C.A., and Moland, E. 2012. Harvest selection on Atlantic cod behavioral traits: Implications for spatial management. Ecol. Evol. **2**(7): 1549–1562. doi:10.1002/ece3.244. PMID:22957161.

Ortiz, M., and Brown, C.A. 2002. Standardized catch rates for sailfish (*Istiophorus platypterus*) from United States recreational fishery surveys in the Northwest Atlantic and Gulf of Mexico. Collect. Vol. Sci. Pap. ICCAT. **54**(3): 772–790.

Osborne, J., Schmidt, T.W., and Kalafarski, J. 2006. Year 2005 annual marine fisheries report Everglades National Park. Homestead, Fla. Available from https://www.nps.gov/ever/learn/nature/upload/2005EVERFisheriesSecure.pdf [accessed 30 July 2018].

O'Toole, A.C., Hanson, K.C., and Cooke, S.J. 2009. The effect of shoreline recreational angling activities on aquatic and riparian habitat within an Urban environment: Implications for conservation and management. Environ. Manage. **44**(2): 324–334. doi:10.1007/s00267-009-9299-3. PMID:19452206.

Page, A.E.S. 1954. Continuous inspection schemes. Biometrika, **41**: 100–115. doi:10.1093/biomet/41.1-2.100.

Papacostas, K., and Foster, J. 2018. National Marine Fisheries Service's Marine Recreational Information Program: Survey design and statistical methods for estimation of recreational fisheries catch and effort. Available from https://www.fisheries.noaa.gov/resource/document/survey-design-and-statistical-methods-estimation-recreational-fisheries-catch-and [accessed 25 May 2021].

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol. Evol. **10**(10): 430. doi:10.1016/S0169-5347(00)89171-5. PMID:21237093.

Picou, J. 2009. Disaster recovery as translational applied sociology: Transforming chronic community distress. Humboldt J. Soc. Relat. **32**(1): 123–157.

Pita, A., Casey, J., Hawkins, S.J., Villarreal, M.R., Gutiérrez, M.J., Cabral, H., et al. 2016. Conceptual and practical advances in fish stock delineation. Fish. Res. **173**: 185–193. doi:10.1016/j.fishres.2015.10.029.

Pohl, C. 2008. From science to policy through transdisciplinary research. Environ. Sci. Pol. **11**(1): 46–53. doi:10.1016/j.envsci.2007.06.001.

Poizat, G., and Baran, E. 1997. Fishermen's knowledge as background information in tropical fish ecology: A quantitative comparison with fish sampling results. Environ. Biol. Fishes, **50**(4): 435–449. doi:10.1023/A:1007317423165.

Post, J., Sullivan, M., Cox, S., Lester, N., Walters, C., Parkinson, E., et al. 2002. Canada's recreational fisheries: the invisible collapse? Fisheries, **27**(1): 6–17. doi:10.1577/1548-8446(2002)027<0006:CRF>2.0.CO;2.

R Core Team. 2019. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available from https://www.r-project.org/.

Regier, P., Briceño, H., and Boyer, J.N. 2019. Analyzing and comparing complex environmental time series using a cumulative sums approach. MethodsX, **6**: 779–787. doi:10.1016/j.mex.2019.03.014. PMID:31016141.

Rehage, J.S., Santos, R.O., Kroloff, E.K.N., Heinen, J.T., Lai, Q., Black, B.D., et al. 2019. How has the quality of bonefishing changed over the past 40 years? Using local ecological knowledge to quantitatively inform population declines in the South Florida flats fishery. Environ. Biol. Fishes, **102**(2): 285–298. doi:10.1007/s10641-018-0831-2.

Rios, A. 2015. Standardized catch rates of red grouper (*Epinephelus morio*) from the Gulf of Mexio Recreational Charterboat and Private Boat Fisheries (MRFSS) from 1986–2013. SEDAR42-AW-04. SEDAR, North Charleston, S.C.

Ruiz-Mallén, I., and Corbera, E. 2013. Community-based conservation and traditional ecological knowledge: Implications for social-ecological resilience. Ecol. Soc. **18**(4). doi:10.5751/ES-05867-180412.

Sagarese, S. 2019. Standardized catch rate indices for vermilion snapper (*Rhomboplites aurorubens*) during 1986–2017 by the U.S. Gulf of Mexico charterboat and private boat recreational fishery. SEDAR67-WP-09. SEDAR, North Charleston, S.C.

Sagarese, S.R., Harford, W.J., Walter, J.F., Bryan, M.D., Isely, J.J., Smith, M.W., et al. 2019. Lessons learned from data-limited evaluations of data-rich reef fish species in the Gulf of Mexico: implications for providing fisheries management advice for data-poor stocks. Can. J. Fish. Aquat. Sci. **76**(9): 1624–1639. doi:10.1139/cjfas-2017-0482.

Saloman, C., and Naughton, S. 1984. Food of crevalle jack (*Caranx hippos*) from Florida, Louisiana, and Texas. NOAA Technical Memorandum NMFS-SEFC-134. NOAA, Panama City, Fla.

Santos, R.O., Rehage, J.S., Adams, A.J., Black, B.D., Osborne, J., and Kroloff, E.K.N. 2017. Quantitative assessment of a data-limited recreational bonefish fishery using a time-series of fishing guides reports. PLoS ONE, **12**(9): e0184776. doi:10.1371/journal.pone.0184776. PMID:28892509.

Santos, R.O., Rehage, J.S., Kroloff, E.K.N., Heinen, J.E., and Adams, A.J. 2019. Combining data sources to elucidate spatial patterns in recreational catch and effort: fisheries-dependent data and local ecological knowledge applied to the South Florida bonefish fishery. Environ. Biol. Fishes, **102**: 299–317. doi:10.1007/s10641-018-0828-x.

Scandol, J.P. 2003. Use of cumulative sum (CUSUM) control charts of landed catch in the management of fisheries. Fish. Res. **64**(1): 19–36. doi:10.1016/S0165-7836(03)00104-8.

Schlesinger, W. 2010. Translational ecology. Science, **329**: 609–609. doi:10.1126/science.1195624. PMID:20688985.

Schmidt, T., Osborne, J., Kalafarski, J., and Greene, C., 2002. Year 2001 annual fisheries report, Everglades National Park. National Park Service, Homestead, Fla. Available from www.nps.gov/ever/naturescience/annualreports.htm.

SEDAR Stock Assessment Panel. 2010. Stock assessment of spiny lobster, *Panulirus argus*, in the Southeast United States. Update Assessment Workshop Report, Key West, Fla.

Shertzer, K., and Williams, E. 2008. Fish assemblages and indicator species: reef fishes off the southeastern United States. Fish. Bull. **106**: 257–269. Available from http://aquaticcommons.org/2089/1/shertzer2.pdf [accessed 8 October 2018].

Silvano, R., and Valbo-Jørgensen, J. 2008. Beyond fishermen's tales: Contributions of fishers' local ecological knowledge to fish ecology and fisheries management. Environ. Dev. Sustain. **10**(5): 657–675. doi:10.1007/s10668-008-9149-0.

Smith, S., Jacob, S., Jepson, M., and Israel, G. 2003. After the Florida net ban: The impacts on commercial fishing families. Soc. Nat. Resour. **16**(1): 39–59. doi:10.1080/08941920309174.

Smith-Vaniz, W., and Carpenter, K.E. 2007. Review of the Crevalle Jacks, *Caranx hippos* Complex (Teleostei: Carangidae), with a description of a new species from West Africa. Fish. Bull. **105**(2): 207–233.

Snelson, F.F., Jr. 1992. Biological studies of the crevalle jack, *Caranx hippos*, in Florida. University of Central Florida, report to the Florida Marine Research Institute, Department of Environmental Protection.

Stori, F.T., Peres, C.M., Turra, A., and Pressey, R.L. 2019. Traditional ecological knowledge supports ecosystem-based management in disturbed coastal marine social–ecological systems. Front. Mar. Sci. **6**: 1–22. doi:10.3389/fmars.2019.00571.

Sutherland, W.J., Fleishman, E., Mascia, M.B., Pretty, J., and Rudd, M.A. 2011. Methods for collaboratively identifying research priorities and emerging issues in science and policy. Methods Ecol. Evol. **2**(3): 238–247. doi:10.1111/j.2041-210X.2010.00083.x.

The World Bank. 2012. Hidden harvest: The global contribution of capture fisheries. Report No. 664- 69_GLB. Washington, D.C. Available from http://documents.worldbank.org/curated/en/2012/05/16275095/hidden-harvest-global-contribution-capture-fisheries#.

Townhill, B.L., Radford, Z., Pecl, G., Putten, I., Pinnegar, J.K., and Hyder, K. 2019. Marine recreational fishing and the implications of climate change. Fish Fish. **20**(5): 977–992. doi:10.1111/faf.12392.

Venables, W.N., and Dichmont, C.M. 2004. GLMs, GAMs and GLMMs: An overview of theory for applications in fisheries research. Fish. Res. **70**: 319–337. doi:10.1016/j.fishres.2004.08.011.

Ward, N.K., Torso, K.A., Arnold, C.A. (Tony), Mitchell, L., and Bakermans, M.H. 2020. Improving environmental decision-making through integrated governance, public engagement, and translational approaches. Bull. Ecol. Soc. Am. **101**(2): 1–8. doi:10.1002/bes2.1685.

Wethington, E. 2015. Translational Sociology. *In* Emerging trends in the social and behavioral sciences. *Edited by* R. Scott, S. Kosslyn, and M. Buchmann. John Wiley & Sons, Inc., Hoboken, N.J. doi:10.1002/9781118900772.etrds0365.

Wrege, B., and Orhun, M. 2019. Commercial landings of Red Grouper (*Epinephalus morio*) in the Gulf of Mexico. SEDAR61-WP-21. SEDAR, North Charleston, S.C.

Zemeckis, D.R., Liu, C., Cowles, G.W., Dean, M.J., Hoffman, W.S., Martins, D., and Cadrin, S.X. 2017. Seasonal movements and connectivity of an Atlantic cod (*Gadus morhua*) spawning component in the western Gulf of Maine. ICES J. Mar. Sci. **74**(6): 1780–1796. doi:10.1093/icesjms/fsw190.

Zipkin, E.F., and Saunders, S.P. 2018. Synthesizing multiple data types for biological conservation using integrated population models. Biol. Conserv. **217**: 240–250. doi:10.1016/j.biocon.2017.10.017.

Zukowski, S., Curtis, A., and Watts, R.J. 2011. Using fisher local ecological knowledge to improve management: the Murray crayfish in Australia. Fish. Res. **110**(1): 120–127. doi:10.1016/j.fishres.2011.03.020.

Zuur, A.F., Ieno, E.N., and Elphick, C.S. 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. **1**: 3–14. doi:10.1111/j.2041-210X.2009.00001.x.