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Fishing effort and the evolving nature of its efficiency

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Fishing effort and the evolving nature of its efficiency

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Editorial: Fishing effort and the evolving nature of its efficiency

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Editorial on the Research Topic
Fishing effort and the evolving nature of its efficiency

Introduction

The earth is currently facing a biological crisis with an unprecedented 1 million species currently at risk of going extinct (IPBES, 2019). Our improvements in technology have come at a cost, which have eroded natural systems faster than they can recuperate. By avoiding laws and chasing profits, overfishing remains the biggest threat to the oceans and the largest culprit to biodiversity loss, with 1/3 of fish stocks considered overfished, and the remaining 2/3rds fished at or near their maximum sustainable yield (Palomares et al., 2020).

Historically, fisheries have been thought of as being similar to agriculture, where the more effort you put in, the more harvest you should be able to extract. This was true until capacity and massive deployment of fossil fuels particularly after WWII outpaced regeneration times. Peak fish landings were reached in Europe in the late 1970s. The global peak catch was reached in the mid-1990s (Pauly and Zeller, 2016) and has since been declining (Figure 1). This assumption and the notion that investment in increasingly sophisticated location and catching technologies would generate higher yields has not been updated. The net result of this sort of 'arms race' impacts biodiversity (Palomares et al., 2020), and generates waning profitability, driven mostly by public subsidies, primarily in the case of industrial fisheries (Sumaila et al., 2019).

Biodiversity and functioning aquatic ecosystems act as a safety net, buffering us from changes affecting ecosystem function and the economies we have been able to build during long times of relative stability. In addition to excessive fishing, there are a range of threats to aquatic ecosystems, including climate change (such as global warming), in combination with ocean acidification and lowered levels of dissolved oxygen, river and ocean pollution, increasing dead zones as a result of overfertilization from runoff generated by industrial agriculture and untreated sewage, and massive influxes of invasive species (IPBES, 2019).

Unfortunately, the world's dominant measure of success is still based on gross domestic product (GDP), which counts any economic activity on the positive side of the balance sheet, even if it is destructive and it is not reflective of human welfare (Stiglitz, 2020). The underlying demand is continual growth of resource consuming economic activities, a feature clashing with the finite nature of resources, as reported by the Club of Rome 50 years ago

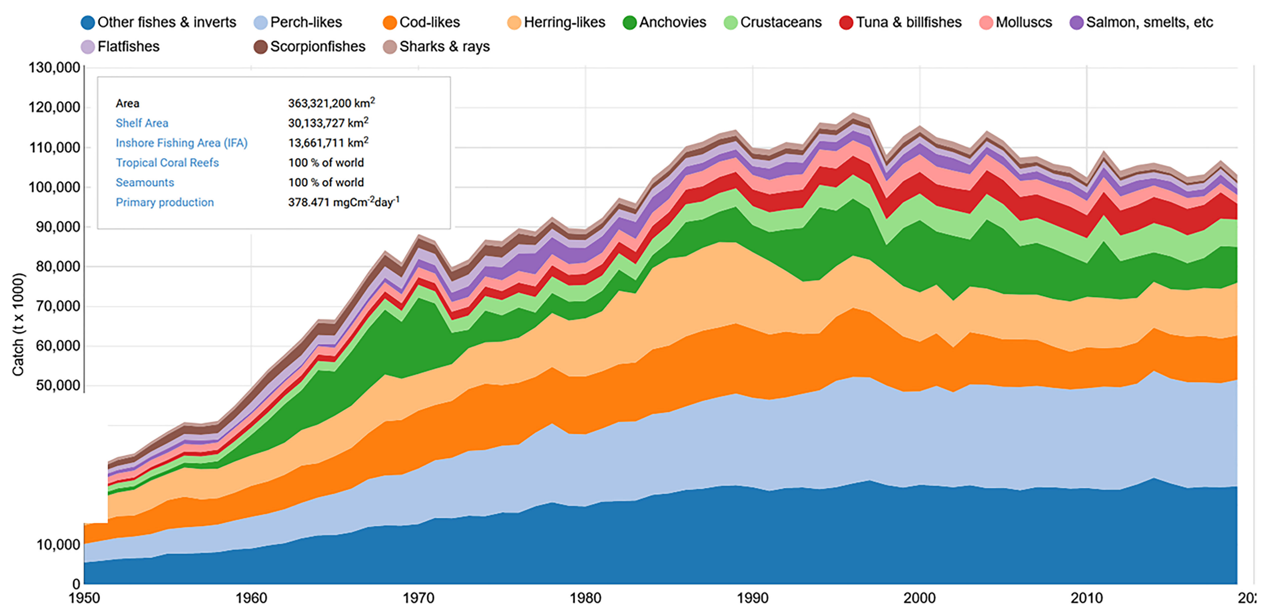

FIGURE 1
Catches (tonnes) by commercial groups in the Global Ocean (www.seaaroundus.org).

(Meadows et al., 1972) and reported contemporaneously as humanity exceeding the planetary boundaries (Steffen et al., 2015).

We are improving our understanding of those planetary boundaries and exploring the mix of measures that might be adopted and effective in recovering systems. Clearly, emerging technologies have a role to play, but the way we think about them, develop, and deploy them need to be grounded in a much greater understanding of human behaviour, greater awareness of risk and an explicit drive to prevent any unintended consequences (Jasanoff, 2007). In times of multiple and overlapping crises and large implementation deficits of existing agreements, laws, and rules, connecting local institutions and mobilizing place-adapted capacities to these larger issues is important to operationalize general rules and insights (Nauen, 2021).

This Frontiers Research Topic (RT) “Fishing Effort and the Evolving Nature of its Efficiency” explores the evolution of fishing effort, notably in coastal artisanal fisheries presenting time series analyses of fishing effort and their impact on coastal habitats or on the functioning of those ecosystems. Overall, 10 articles were authored by 54 researchers from eight different countries, presenting critical insights from across the globe—from Seychelles to Peru, and from the South Atlantic to the Mediterranean. Here we review the contributions from these studies and identify remaining challenges. Based on findings from the ten papers, we provide an overview of the lessons-learned and key recommendations for sustainable management measures. Hence, we have organized the content in two thematic areas: (i) trends in fishing effort from across the globe, and ii) fishing impacts on different ecosystems considering interactions with climate change.

Trends in fishing effort

Vianna et al. investigated long term catch and effort trends and reported that total catches of the Republic of the Marshall Islands (RMI) were 27% higher than the data officially reported by FAO. The fisheries in RMI are mainly artisanal, and there has been a gradual shift from predominantly non-commercial to commercial small-scale fisheries in the past decades. From 1950 to 1990 total catch and effort was found to be stable, however in the late 2000s, the continued increase in fishing effort corresponded with a gradual reduction in total catch, which also affects the overall production and the economics. Since the RMI is one of the archipelago areas in the Pacific Ocean, climate change impacts on marine and terrestrial life is increasing, additional adaptive coastal fisheries management measures need to be explored.

From the other side of the globe, Demirel et al. investigated the stock status of 54 commercial fish stocks from the eastern Mediterranean and Black Sea. The region was classified as “data-poor”, with a low number of assessed stocks in comparison to the reported biodiversity and the number of exploited species (FAO, 2022; Froese et al., 2018). In their study, the catch-based assessment algorithm CMSY (Froese et al., 2017) was used to obtain fisheries reference points and future scenarios to rebuild fisheries. Of the 54 stocks, 94 percent was found to be overexploited. Recovery times were estimated at 15 years for 60 percent of the stocks if fishing pressure is reduced by 50 percent. Regardless of the assumptions using this assessment approach, under a business-asusual fishing scenario, all stocks are likely to be impacted.

Christ et al. examined the historical development of catch and effort in the Republic of Seychelles as a case study on current and future options of resource sustainability for island countries. The fishing industry is crucial for food supply and employment provision in many island countries. The paper highlights that the reconstructed catch was 1.5 times higher than that officially reported by FAO and that fishing effort greatly increased from 1950 to 2017. However, the trend of artisanal catch per unit of effort (CPUE) was declining over time, suggesting a reduction in relative abundance of fish populations within the Seychelles' EEZ or targeted fishing areas.

De la Puente et al. focused on reconstruction of long-term fishing effort in Peruvian smallscale fisheries from 1950 to 2018. The paper analyzed changes in the fleet's fishing efficiency and economic performance and revealed that fishing effort increased at much faster rates than catches, particularly in the period since 2006. Peru is one of the world's leading fishing countries, smallscale fisheries are an important contributor to national employment, food security and gross domestic product. De la Puente et al. reported that the expanding fishing effort has become unsustainable and uneconomical, with fisheries essentially 'growing into poverty'. A key aspect of the study was exploring the social, legal and economic drivers fostering fleet growth, and the importance of a bottom-up governance approach for the well-being of small-scale fishers.

Zeller et al. reconstructed long term fishing effort of small-scale fisheries in Mozambique. They found fishing effort increased by nearly 60 times, while CPUE in the small scale fleet showed a strong decline. The increase in fishing effort is driven by motorization and growth in vessel numbers. The continuing increase in the fishing capacity of small-scale fisheries in the absence of effective and restrictive management actions may exacerbate overexploitation.

Farquhar et al. investigated the impact of weather conditions on the duration of active fishing hours for small scale fishers in Madagascar. The study combined fishers' knowledge on meteorological conditions and long-term remotely sensed data to analyse the impact of weather trends on fishing effort for over 40 years. The increase in adverse weather conditions led to a decreasing fishing trend in effort with a loss of 21.7 available fishing hours per year. This study demonstrated the impact of changing weather conditions on fishers' well-being and food security, with ongoing impacts expected under climate change.

Fishing impacts

White et al. highlighted concerns on changes in fishery resources due to climate and current fishing practices leading to declining abundance of reef fishes. The changes in fishery resources may adversely affect the well-being of small-scale fisheries. Their investigations into Tokelau's reconstructed catch and CPUE over 50 years indicate that the domestic catch was greatly underreported. Since 2010, extractions are estimated to be nearly four times larger than the data reported by FAO. The lesson-learned by Tokelau's case is that high uncertainty and reporting problems of the fishing resources contributes to ineffective management practices in which changing catch composition may alter local people's every-day habits and traditional practices in the short term.

Kripa et al. examined fishing impacts on the Indian oil sardine *Sardinella longiceps* in the Arabian Sea. It constitutes the fifth largest sardine fishery in the world. During the 2010s the fishery experienced a dramatic collapse in relation to an increase in fishing effort due to changes in gear size and engine capacity, prior to El Nino Southern Oscillation in 2015. The paper highlights the negative climate impact on the maturation process of Sardinella caused by a mismatch of its larval development and phytoplankton bloom periods. The collapse of this fishery affected the livelihood of thousands of small-scale fishers. This paper illustrates the importance of a more responsive fisheries administration with timely restrictions on fishing effort and protection of spawning stocks (e.g., through a closure) in order to minimize the combined impacts of excessive fishing effort and environmental change.

Ferra et al. focused on one of the novel technologies, namely using the mandatory automatic identification system (AIS) originally intended to avoid collisions at sea, to check its efficiency for quantifying unobserved trawling activity in the Mediterranean Sea. Although this technology can be useful in marine spatial planning, it has limitations, namely transmission gaps (e.g., periods of weak signal detection), interference with other signals or deliberately switching off the system, to conceal fishing activities. The paper illustrates that determining duration and distance of transmission gaps in bottom trawlers from their harbor enabled quantifying unobserved fishing and its effects on overall trawling pressure. The authors conclude that their results may help to revise the estimation of fishing effort from AIS data to understand actual fishing.

Jalali et al. focused on the spatio-temporal dynamics of recreational fishing in Australia. As one of the lesser researched parts of fisheries, the authors emphasize that recreational fishing is a popular pastime and multi-billion-dollar industry in Australia. It plays an important economic role in some regions of the continent. Data was derived from boat-based creel surveys during a 10-year period from 2010 to 2019, and used their survey data as an input for geospatial modeling. Their study highlights the importance of spatially explicit approaches to inform fisheries management for minimizing the impact on coastal communities but maximizing benefits for local businesses.

Conclusions

The contributions of this Research Topic illustrate the local articulations of a general trend of increasing fishing effort in a wide range of fisheries. The papers follow the global trend of fully or overexploited, even collapsing resources (Pauly and Zeller, 2016). The corollary of decreasing income and increasing insecurity is another common observation. Several studies also indicate breakpoints or shifts in ecological systems as species specific or environmental forcing such as climate change adds pressure. High uncertainty and reporting problems of the catches contribute to the breakdown of management practices in which changing catch composition may alter local people's every day-habits and traditional practices in the short term (White et al.). Under a business-as-usual scenario, many stocks are on the brink of collapse with uncertain prospects of recovery (Demirel et al.). Declining catches over time suggest a decline in relative abundance of fish populations (Christ et al.; Zeller et al.). Increased fishing effort is a counterproductive response, resulting in small-scale fishers growing into poverty in some circumstances (De la Puente et al.). More responsive fisheries administration with timely restrictions on fishing effort and protection of spawning stocks is essential (Kripa et al., 2018), particularly in conjunction with trust-building measures of bottom-up governance. New approaches, such as analyzing AIS data of fishing fleets, can contribute to reducing data gaps (Ferra et al.; Jalali et al.) and could support increased management measures against IUU fishing and excessive build-up of capacity. Climate change impacts on marine and terrestrial life is becoming more visible and concerning, hence coastal fisheries management needs to consider a range of adaptive measures (Vianna et al.). Fishers' well-being, safety and food security do not only depend on classical fisheries management but also needs climate change adaptation strategies (Farquhar et al.). Fishing in the absence of management controls, combined with other human-induced effects (e.g., climate change, pollution), diminishes the resilience of marine ecosystems. The continued benefits that humans derive from the seas require tangible counter measures that help rebuild depleted resources and re-establish prosperous small-scale fisheries. The efforts around the world to improve data quality, to understand changes from past to present, and to develop management recommendations with that information are a useful contribution. It is essential to share such research insights broadly beyond the scientific community. When scientists, managers, artisanal fishers, and other citizens bring their different experiences and analyses to discussions and interactions, there is a greater chance of positive change for aquatic resources and communities. That change must not only support resource recovery but needs to be equitable in the distribution of costs and benefits perceived as fair by stakeholders. In a few fisheries encouraging experiences are progressing. The key elements of these case studies are that they are established based on sound, robust science that feeds into practicable and adaptive management systems. These examples need to be more widely adapted to the local context for many systems.

Author contributions

All authors listed, have made substantial, direct and intellectual contribution to the work, and approved it for publication.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Overfishing and Climate Drives Changes in Biology and Recruitment of the Indian Oil Sardine *Sardinella longiceps* in Southeastern Arabian Sea

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The recent fluctuations in abundance of the Indian oil sardine *Sardinella longiceps*, a tropical small pelagic clupeid fish, was investigated in the light of overfishing and variations in its habitat ecology in southeastern Arabian Sea. In 2012, its landings peaked to an all-time record making it the fifth largest sardine fishery in the world, and within 3 years the catches were reduced to nearly a tenth of that level. This study examined the fishery dependant factors such as effort, catch rates and expansion of fishing area; the biological variations in fish size, maturity and recruitment; and tried to relate this to the environmental variations in the sardine habitat and food availability. The 2012 mega harvest was a result of a 2-time increase in gear size and engine capacity of fishing crafts and a 3.7-time increase in fishing effort. The female maturation process was strongly influenced primarily by rainfall and then by upwelling and the resulting influx of cold nutrient-rich water in the habitat from April much before the start of the monsoon in June. After 2013, the weak monsoons and the 2015 El Nino Southern Oscillation resulted in a warmer (by an average of 1.1°C) period which negatively impacted the maturation process. The abundance of jellyfishes which are larval and young fish predators in the habitat negatively affected recruitment after 2013. The mismatch in timing of phytoplankton productivity and sardine larvae in the habitat also affected the recruitment success. These environmental divergences coupled with the excessive capture (beyond maximum sustainable yields) of spawning stock and juveniles from 2010 has resulted in this biological catastrophe which has affected the livelihood of thousands of small-scale fishers. A more responsive fisheries administration with timely restriction on fishing effort and protection of spawning stocks by way of fishery closure would have helped minimize the impacts.

Keywords: Indian oil sardine, overfishing, biology and recruitment, environmental drivers, food availability, timing mismatch

INTRODUCTION

The Indian oil sardine, *Sardinella longiceps*, is a small pelagic clupeid fish with distribution limited to tropical waters of northern and western Indian Ocean: Gulf of Aden, Gulf of Oman, but apparently not Red Sea or the Persian Gulf, and eastward to India, including the Andaman Islands. This forage species comes under the group 'herring, anchovies and sardine, (HAS) which contributes to 18.6% (15.3 million tons) of the global marine capture fishery production (FAO, 2016). Among the five nations which harvest oil sardine (0.572 million tons) from their coastal waters, India's average production is the highest with 65% (376,189 t) followed by Oman, 63,439 t, Yemen, 58,839 t, Iran, 45,800 t, and Pakistan, 27,372 t (FAO, 2016).

Globally, the sardine and anchovy fisheries have shown wide fluctuations and the reasons for these have been extensively investigated (Lluch-Belda et al., 1992a,b; Schwartzlose et al., 1999). Typically collapses are characterized by a reduction in catch of less than 10% of the maximum and by a long recovery time after reaching a biomass minimum (Mullon et al., 2005; Worm et al., 2009; Petitgas et al., 2010).

During the period 1950–2014, India has been the major contributor of oil sardine catch with 66–96% (average 80%) of the global oil sardine catch. Among the Indian maritime states, Kerala has always been the major producer of oil sardine (Hornell, 1937; Nair and Chidambaram, 1951; Pillai et al., 2003) as it is one of the most productive upwelling zones (Banse, 1959) along the southeastern Arabian Sea. With average annual landing of 0.22 million tons during 2001 to 2010, the Kerala State's landing exceeded the catch from other nations like Oman, Yemen, Iran and Pakistan (FAO, 2016).

The Indian oil sardine supports the main non-motorized and motorized fisheries sectors of Kerala which are operated along 590 km coastline and spread across 222 villages. About 145,396 fishermen are actively involved in this small-scale coastal fisheries (CMFRI, 2012). The importance of the sardine fishery can be judged from the fact that this species has singly contributed on an average to 37% of the total marine fish landings of Kerala (0.58 million tons) during the period 2001–2010. At the national level the total oil sardine catch from all states contributed to 15% of the all India marine fish production of 3.7 million tons during 2011–2015.

This fishery has declined and collapsed several times during 19th and 20th centuries and several fishery based reasons have been cited along with effects of environmental factors especially the southwest monsoon and coastal upwelling (Raja, 1969; Longhurst and Wooster, 1990; Madhupratap et al., 1994; Jayaprakash, 2002; Krishnakumar et al., 2008; Xu and Boyce, 2009). However, reasons for fluctuations in oil sardine fishery linking the fishery dependent and habitat changes on the biology of sardine and the resultant density changes has not been attempted so far.

In the present century, oil sardine catches of Kerala did not decline drastically till 2012. The recent sardine crisis started in 2013, with the total landing declining to 0.21 million tons from an all-time peak of 0.39 million tons in 2012, indicating a decline of 46%. It further declined to 0.15 million tons in 2014, and continued to decline in 2015 when only 0.068 million tons were landed indicating a decline of 82% from that of 2012.

The biology of the fish has been a subject of detailed investigations. Studies have been made on fecundity, spawning seasons, size at maturity, gonadal development, sex ratio and diets (Jayaprakash and Pillai, 2000). It has been observed that there is differential growth between the early and late recruits arising from the same spawning stock in a year and that the fish attains 9.5–11.0 and 11.0–12.5 cm in the 2nd and 3rd month itself (Raja, 1973; Yohannan et al., 1998). Oil sardine grows rapidly, matures early and a few continue to survive beyond 18 months (Longhurst and Wooster, 1990). Radhakrishnan (1965) observed that the minimum size at first maturity is 12.0–13.9 cm. Majority of investigations indicate that the size at first maturity is 15.0 cm and that the spawning period is from May to August coinciding with the southwest monsoon (Jayaprakash and Pillai, 2000). It has also been observed that the fecundity varies with the age of the fish and size of the ovary and generally ranges between 37,000 to 80,000 (Jayaprakash and Pillai, 2000).

The reason for this decline was investigated by analyzing the fishery related and fishery independent factors including the environmental and biological variations and the biotic pressures on the oil sardine population along Kerala. This study investigated the change in pattern of fishing oil sardines and the resultant impacts. Further, we studied the environmental parameters controlling maturity, spawning and recruitment of sardines and analyzed the influence of major ocean atmospheric processes such as El Niño.

The investigation used various sets of real-time and satellite based datasets for the period 2010–2015 which were collected as a part of investigations of projects supported by the Ministry of Agriculture and the Ministry of Earth Sciences, India.

MATERIALS AND METHODS

Details of datasets pertaining to sardine fishery, ocean-atmospheric parameters of the fishing grounds, fish biology and phytoplankton densities are given in **Table 1** and **Supplementary Table S1**. The biological details of the fish from Kochi and Kozhikode, two major oil sardine landing centers (**Figure 1**) were used in the analysis. Hydrological sampling was done off Kochi (**Figure 1**). Oil sardines are distributed (**Figure 1**) as small and large shoals up to about 50–100 m depth, but are most common up to 30 m depth (Pillai et al., 2003).

Oil Sardine Landing Data

Oil sardines are fished by different craft-gear combinations like, fishing boats with outboard engines using ring seines (OBRS), mechanized units using ring seines (MRS), outboard units using gill nets (OBGN), mechanized trawlers (MTN), outboard units using boat seines (OBBS), other motorized units (MOTRS). Outboard units are those traditional crafts fitted with outboard engines, while mechanized units have inboard engines. Estimation of catch and effort were made by using a sampling method, the multistage stratified random sampling design (Srinath et al., 2005). The method involves recording of

TABLE 1 | Type and source of data used in the analysis.

S. No.	Parameter	Source
Fishery parameters		
(1)	Monthly gear-wise oil sardine catch and effort 2010–2015	CMFRI- NMFDC database
(2)	Depth-wise oil sardine fishing locations off Kerala	Derived from NMFDC enumerator data sheets
(3)	Catch rates – catch per hour of fishing	NMFDC database
Ocean-atmospheric parameters		
(1)	Surface and bottom Salinity, Temperature, DO, Chlorophyll a at 5, 10, 20, and 30 m depth stations off Kochi	Data collected for this study
(2)	Rainfall data	IMD
(3)	Multivariate ENSO Index (MEI)	NOAA
(4)	Local Thermal Anomaly	Derived from SST data taken from NOAA
Biological parameters		
(1)	Monthly Length frequencies, 2010–2015 Kochi/Kozhikode	Data collected for this study
(2)	Monthly maturity stages 2011–2015	Data collected for this study
(3)	Gonadosomatic index (GSI) 2013–2015	Data collected for this study
Plankton parameters		
(1)	Phytoplankton monthly densities 2010–2015	Data collected for this study
(2)	Jellyfish biomass from trawl survey	Data collected for this study

NMFDC, National Marine Fisheries Data Centre of Central Marine Fisheries Research Institute, Kochi, India; IMD, India Meteorological Department; NOAA, National Oceanic and Atmospheric Administration, United States.

catch and effort at each landing center by experienced observers for a 24 h period on a sampling day. Normally, 16–18 days in a month are selected at random for observation. The daily catch and effort from at least 10% of vessels were multiplied with total number of vessels on the observation day to arrive at daily estimates for 187 landing centers. The fishing effort is expressed by the number of unit operations by a craft-gear combination (unit), the fishing hours expended by the unit during the month, the man-hours expended by the units during the month. This daily estimate from different sampling days in a month were pooled and multiplied with number of fishing days to arrive at monthly estimates of catch and effort.

Catch data from 1960 to 2015 and catch per unit effort (CPUE) of sardines in various gears for the period 2007 to 2015 were used for the analysis. CPUE was used as an index of abundance in the absence of biomass data. Details on the gear used and the engine horsepower were collected through direct enquiry at the landing centers and fishing villages. Apart from this, the annual sardine catch data for the period 1960–2015 were used to categorize the stock into abundant, less abundant, declining and collapsed based on catch-based rapid stock status method (Mohamed et al., 2010) which is modified version of the method developed by Froese and Kesner-Reyes (2002).

Oil Sardine Biology

Weekly random samples of sardines (N = 100) were collected from the ring seine catches from the landing at Kochi and Kozhikode for biological parameters during 2010–2015. Sardines were measured (N = 31,200) to the nearest mm using a scale and weighed in an electronic balance (nearest 0.1 g) after blotting the water from the surface of the fish. The length frequency data, were grouped into three viz., less than 10 cm, 10.1–14 cm, and above 14 cm, assuming that sardine less than 10 cm are immature juveniles, 10–14 are maturing sardines and above 14 cm are adult mature sardines based on previous reports on the size at first maturity (Jayaprakash and Pillai, 2000; Nair et al., 2016). The length data for each month were pooled from both the centers and the percentage contribution for each group was estimated and then monthly sardine length frequencies were weighted to the catch. These were used to derive monthly mean, minimum and maximum lengths.

Maturity was estimated by the macro and microscopic observation of 6104 male and female gonads. The female gonads were staged based on a modified ICES scale proposed by Raja (1969) and Zaki et al. (2012) as immature (stage 1), developing virgin (stage IIa), maturing (stages III and IV), mature or gravid (stages V and VI), partially spent and spent (stages VIIa and b) and spent resting (stage IIb) (**Table 2**). For analysis the data were grouped into mature (Stages III to IV); partially spent (VIIa); spent (VIIb) and spent resting (IIb). The gonadosomatic index (GSI) for females was calculated using the formula: GSI = (weight of gonad × 100)/weight of fish) every month and mean values were plotted.

Hydrography and Climatology of Oil Sardine Habitat

Water samples from sardine fishing area off Kochi were collected from four stations (surface and bottom at 5, 10, 20, and 30 m depth zones) at monthly intervals (**Figure 1**). This transect was considered as representative of the sardine fishing area off Kerala coast. Sea water temperature and salinity at different depths were recorded using a CTD probe (YSI Incorporated, United States). The water samples were fixed and analyzed as per standard methods for dissolved oxygen, chlorophyll-a (CHL) and nitrate (Strickland and Parsons, 1968; APHA, 1998). From the same stations phytoplankton (PHY) samples were collected using a 20 μm Indian Ocean Standard net with digital flow meter (KC Denmark, Denmark), fixed in buffered formalin and cells were counted in the laboratory after identifying the major species.

To analyze the variations of parameters that are closely related to upwelling viz., temperature, salinity (PSU), dissolved oxygen (DOX) and nitrate content, two dimensional surfaces of these parameters were generated with time (month) on x-axis and depth on y-axis for the observation station. The observations from 30 m depth station were about 25 km from the shore. Since in 2013–2014 the sardine fishing grounds were abound with coelenterate medusa, experimental trawling was done at these stations and the Hydrozoan and Scyphozoan medusa collected were identified and biomass estimated based on the swept area method (Pauly, 1983).

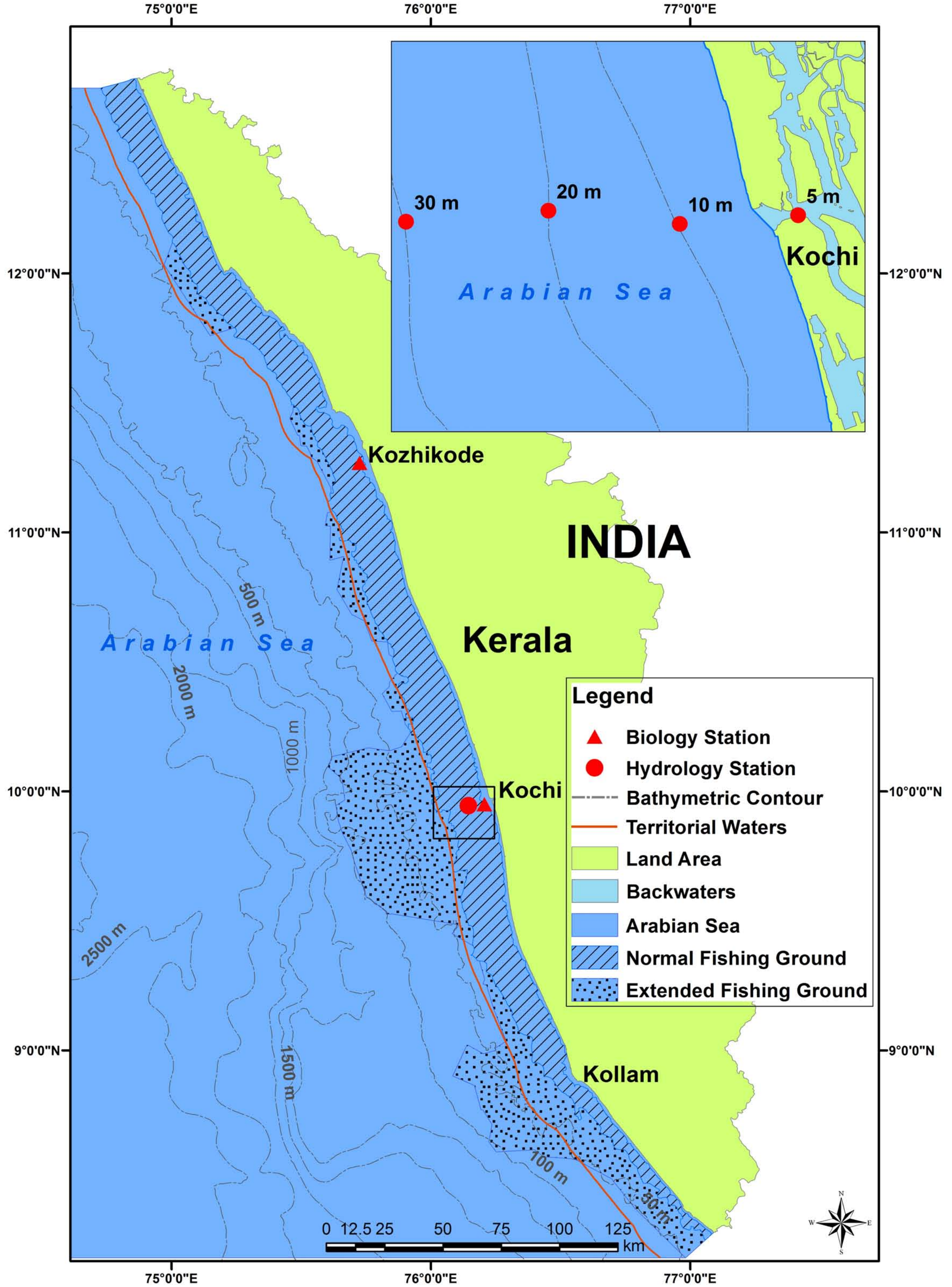

FIGURE 1 | Map showing southeastern Arabian Sea off Kerala State with bathymetric contours. The hydrology and biology sampling stations are indicated. Inset shows detail of hydrology sampling stations. Shaded areas indicate the traditional oil sardine fishing grounds and the stippled area indicates the extended fishing grounds.

TABLE 2 | Maturity staging of female *Sardinella longiceps* by macroscopic analysis used in the study following Raja (1966).

Stage	Macroscopic appearance of ovary
(I) Immature	Ovaries soft, cylindrical, pink and completely transparent. Extend to about half of body cavity
(IIa) Developing virgin	Ovaries soft, cylindrical, pink and extend more than half of body cavity. Ova not visible to naked eye.
(IIb) Spent resting	Dark red or brownish red ovaries, surface wrinkled. Oviducts wide but shorter, extend more than half of body cavity.
(III) Maturing	Turgid, opaque and yellow with granular appearance. Oviduct very much reduced. Extend to more than half of body cavity. Ova visible to naked eye.
(IV) Maturing	Compact, vascular, bright yellow ovaries. Oviduct not distinct. Occupy more than three-quarters of body cavity.
(V) Mature or gravid	Orange yellow and fully vascular. Extend more than the length of the body cavity.
(VI) Running	Appear as cream colored cellophane bag filled with boiled sago. At a slight prick, gelatinous mass of transparent ova flows out. Extend more than the length of the body cavity.
(VIIa) Partially Spent	Dark red, either throughout or at the posterior half. A bit flaccid, shrunken with wrinkle on the tunica. Occupy about three-quarters of body cavity.
(VIIb) Spent	Elongated, honey- colored, bloodshot, flabby, limb, flattened and gelatinous with wrinkles on surface. Wide oviduct now discernible. Occupy slightly more than half of body cavity.

Monthly rainfall data (RNF) for Kerala state were obtained from the website of Indian Meteorological Department. Upwelling regions are known to have cooler waters and the difference in temperature between two locations along the same latitude has been considered as an indicator of upwelling (Jayaram et al., 2010). Based on this principle, Local Thermal Anomalies (LTA) have been used extensively to study the variation in upwelling all along the Indian west coast (Smitha et al., 2008; Shah et al., 2015). Using the equation LTA = $T_{\text{off}} - T_{\text{coast}}$ where T_{off} is the temperature of offshore grid (Longitude: 75, 76; Latitude: 10, 11) and T_{coast} is that of a grid (Longitude: 72, 73: Latitude: 10, 11) nearer to the coast. Positive LTA values suggest coastal upwelling processes.

Multivariate ENSO Index (MEI) was estimated using six major observed variables over the tropical Pacific which were obtained from the NOAA web site[1] (Wolter and Timlin, 1993). MEI gives the variations in El Niño/Southern Oscillation (ENSO) which is considered as the most important coupled ocean-atmosphere phenomenon causing global climate variability on inter-annual time scales.

Statistical Analysis

In order to relate oil sardine abundance and life history events with physical, chemical and climatological parameters, the data on fishery, ecology and biological characteristics were subjected to Pearson's correlation and statistically significant correlations were identified. Principal component analysis (PCA) was applied to define the relationships between catch rate (catch per hour [CPH] in MRS and OBRS), recruitment (less than 10 cm fish, LT10), mature fish (greater than 14 cm fish, GT14) and environmental variables. All biological parameters were fourth root transformed and scaled before the analysis. The Pearson correlation matrices were created utilizing R (R Core Team, 2014) 'Performance Analytics' package and PCA was done using 'factoextra' package. Rows in the data frame with null values in any of the fields were discarded while carrying out the PCA.

To assess relative contribution of different parameters (RNF, MEI, LTA, PSU, DOX, CHL, PHY, GT14, and LT10) to CPUE, two multiple linear regression equations were fitted with CPHOBRS and CPHMRS as response variables and RNF, MEI, LTA, PSU, DOX, CHL, PHY, GT14, and LT10 as regressors. The relative importance of regressors, i.e., individual regressor's contribution to the multiple regression model, were assessed using the Lindeman, Merenda and Gold method applied in the 'relaimpo' package in R (Lindeman et al., 1980; Ulrike, 2006).

RESULTS

Fishery Dependent Factors

During the period 1961–2015, the sardine catch fluctuated between a low of 1554 tons in 1994 to an all-time peak of 0.39 million tons in 2012 (**Figure 2**). As per the rapid stock status method, the collapsed status was reached only once (1994); however, the stock status was depleted during 1986 and almost reached the depleted status twice in 1963 and 2015 (**Figure 2**). One significant observation was the abundant status of the stock during 2010–2012. In comparison to 2010, the landings increased by 24.2 and 54.1% in 2011 and 2012, respectively. The catch was significantly ($P < 0.001$) and negatively related to MEI, phytoplankton density ($P < 0.01$), chlorophyll ($P < 0.05$), and rainfall ($P < 0.05$) (**Table 3**).

Increase in Effort and Catch Per Unit Effort

The oil sardines were exploited by a number of crafts including mechanized, motorized and non-motorized. The two principal gears exploiting oil sardines were MRS and OBRS and their percentage contribution to the total catch was inversely proportional (**Figure 3**). Mechanized pelagic trawl nets also caught considerable oil sardine during peak abundance period of 2012 and 2013. The percentage of oil sardine in non-mechanized decreased from 16% in 2007 to 6% in 2015.

The effort in terms of number of MRS units increased from 20,152 in 2007 to 74,416 units in 2012 – 3.7 times increase over that of 2007 and the catch per hour of these units increased from an average of 494 kg h^{-1} in 2007 to 1378 kg h^{-1}in 2012 and then declined to 303 kg h^{-1} in 2015 (**Figure 4**). The increase in catch during 2012 was primarily due to the increase in effort by these units. Though the OBRS unit operations increased during 2008, their numbers decreased subsequently. However, their unit operations increased marginally during 2012. The catch rate of OBRS units was high, 548 kg h^{-1} in 2007 which decreased to 341 kg h^{-1} in 2008. From 2009 onwards, the catch rate showed an increasing trend and reached 670 kg h^{-1} during 2011 and thereafter decreased steadily to 159 kg h^{-1} in 2015. During the

[1]https://www.esrl.noaa.gov/psd/enso/mei/

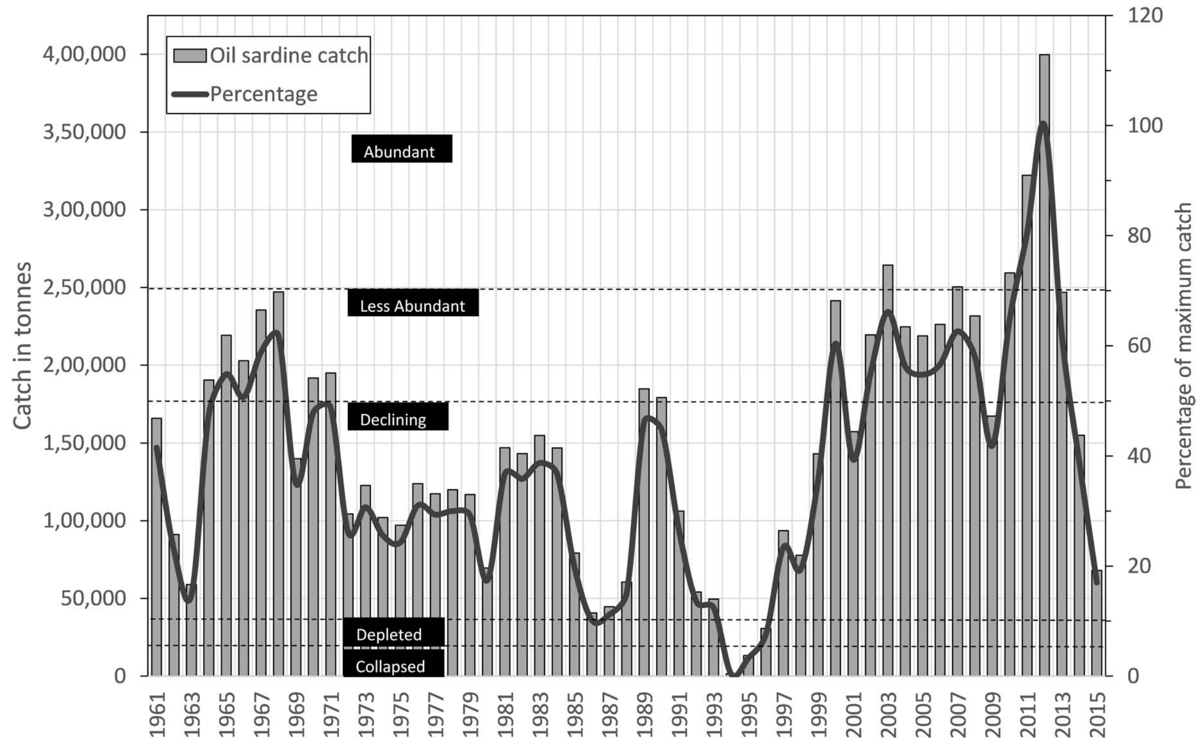

FIGURE 2 | Time series of estimated oil sardine catch (bars) from Kerala State with percentage of historical maximum (line) and classification of stock-status based on the percentage.

TABLE 3 | Pearson correlation matrix (*r*-values and significance) between oil sardine catch, catch rates, size groups, maturity stages and environmental factors.

	RNF	MEI	LTA	PSU	DOX	CHL	PHY	CPH MRS	CPH OBRS	<10	10–14	>14	SPR	MAT	PSP
MEI	0.21														
LTA	0.22	−0.16													
PSU	−0.48***	0.11	0.06												
DOX	−0.49 ***	−0.08	−0.23	0.27*											
CHL	0.16	0.33**	−0.05	0.01	−0.30*										
PHY	−0.03	−0.03	0.09	0.13	0.05	−0.07									
CAT	−0.29*	−0.39***	0.02	0.06	0.23	−0.29*	−0.32**								
CPH MRS	−0.42***	−0.34**	−0.12	0.12	0.24*	−0.31**	−0.31**								
CPH OBRS	−0.42***	−0.52***	−0.09	0.1	0.42***	−0.34**	−0.32**	0.56***							
<10	0.29*	−0.01	0.2	−0.24*	−0.34**	−0.1	0.13	−0.21	−0.30**						
10–14	−0.46 ***	−0.11	−0.17	0.26*	0.35**	−0.11	0.09	0.19	0.2	−0.09					
>14	0.35**	0.1	0.1	−0.18	−0.23*	0.13	−0.13	−0.11	−0.09	−0.23	−0.95***				
SPR	−0.09	0.04	−0.21	−0.14	−0.06	0.13	0.09	−0.05	−0.05	0.19	−0.1	0.04			
MAT	0.41***	0.15	0.11	−0.26*	−0.30*	0.13	−0.2	−0.1	−0.2	−0.1	−0.45***	0.47***	−0.23*		
PSP	0.70***	0.2	0.24*	−0.19	−0.44***	0.11	0.09	−0.27*	−0.38**	0.39***	−0.43***	0.30*	−0.24*	0.29*	
SPE	0.45***	−0.11	0.15	−0.29*	−0.16	−0.06	0.21	−0.1	−0.14	−0.05	−0.31**	0.31**	−0.1	0.13	0.47***

*Significance in bold: ***$p < .001$, **$p < .01$, *$p < .05$; negative sign denotes negative correlation.*
RNF, rainfall; MEI, multivariate ENSO index; LTA, local temperature anomaly; PSU, salinity; DOX, dissolved oxygen; CHL, chlorophyll a; PHY, phytoplankton density; CAT, catch; CPH MRS, catch per hour mechanized ring seine; CPH OBRS, catch per hour outboard ring seine; <10 – sardines of less than 10 cm TL; 10–14, sardines between 10 and 14 cm TL; >14, sardines of more than 14 cm TL; SPR, spent recovering; MAT, mature; PSP, partially spent; SPE, spent.

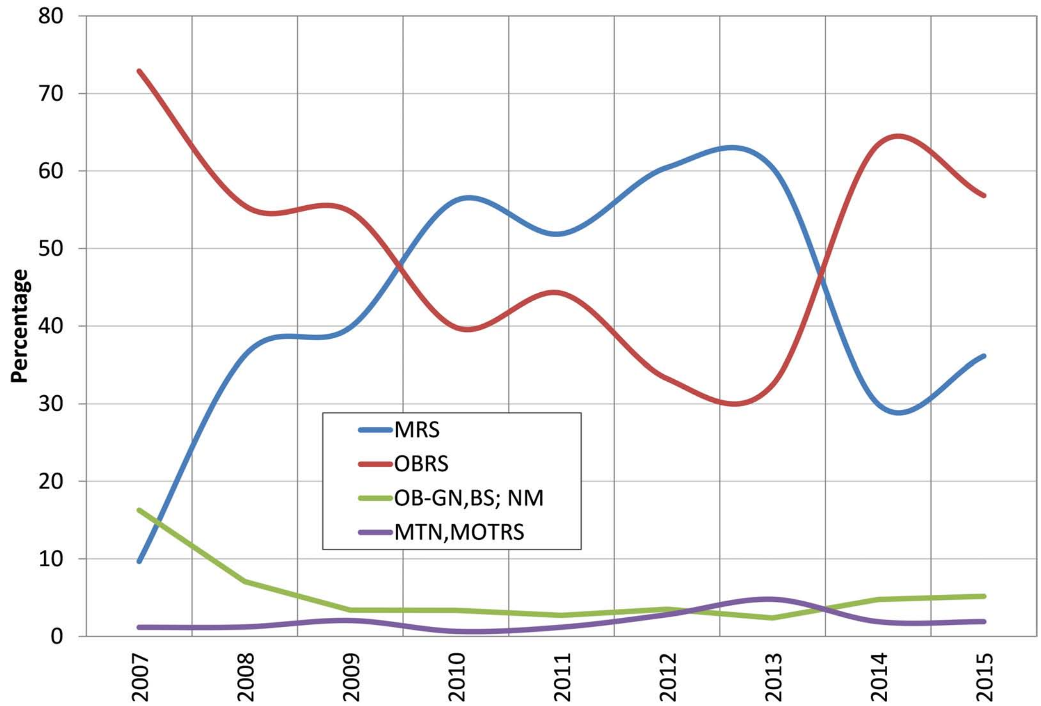

FIGURE 3 | Percent contribution of different gears to oil sardine catches during 2007–2015. MRS, mechanized ring seines; OBRS, outboard ring seines; OB-GN, outboard gillnetters; BS, beach seines; NM, non-mechanized; MTN, mechanized trawl nets; MOTRS, mechanized other gears.

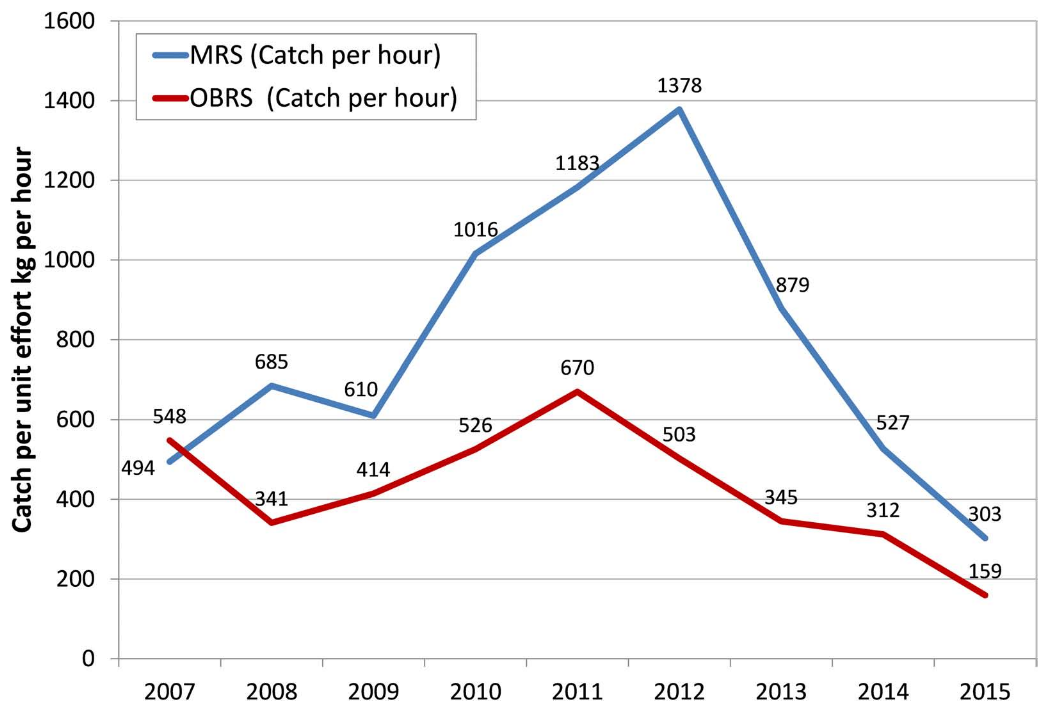

FIGURE 4 | Catch rate (in kg per hour) of oil sardines in the two principal gears MRS and OBRS.

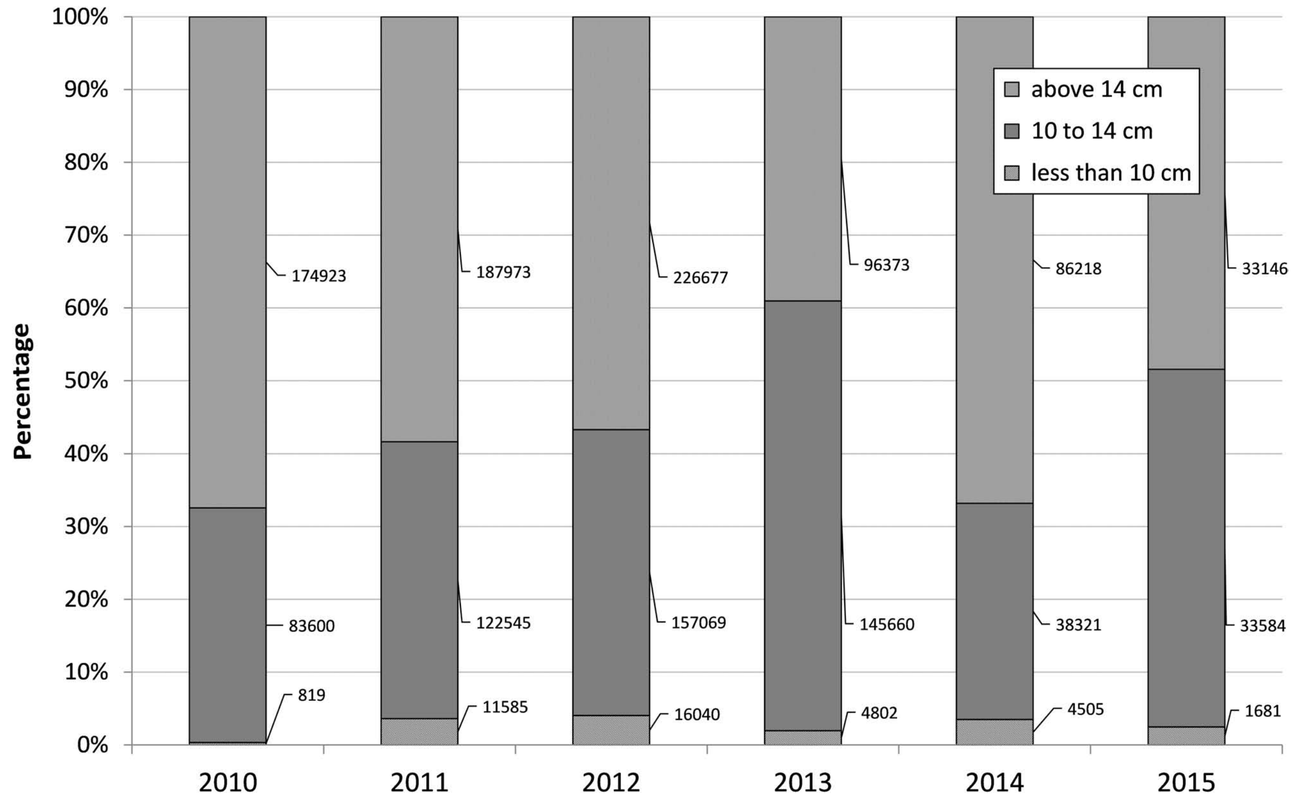

FIGURE 5 | Percent contribution of the three size groups of oil sardine to annual catch during 2010–2015. Figures indicate size group wise tonnage.

period 2013–2015, the effort (not shown) as well as the catch rate of both MRS and OBRS units decreased considerably.

The catch rates in MRS and OBRS were negatively related to rainfall ($P < 0.001$), MEI ($P < 0.001$); chlorophyll a ($P < 0.01$) and phytoplankton density ($P < 0.01$). The catch rates in both MRS and OBRS were positively correlated with dissolved oxygen ($P < 0.05$; $P < 0.00$, respectively) (**Table 3**).

Fishing Beyond Conventional Fishing Grounds

Until 2013, the main fishing area for sardines was between the 5–30 m depth zones **Figure 1**. However, the MRS with high powered engines have fished in areas beyond the conventional sardine fishing zones mainly during March, May, and June which is the period when sardines mature and become ripe for spawning. Such fishing activities were mainly from the two main fishing ports of Kochi and Kollam.

Biological Variations

Length Based Analysis

The monthly length frequency analysis of the sardine catch indicated there was excessive harvesting of juveniles (less than 10 cm) size group during the period 2011–2013 (**Figure 5**). About 16,040 tons of juveniles (less than 10 cm) forming 4% of the total catch were harvested in 2012 and about 4802 tons in 2013. The 10–14 cm size group which is 0-year class forms a major component of sardine catches. The 1 year class (>14 cm) formed the maximum percentage in 2010, 2011, 2012, and 2014 (**Figure 5**).

Juvenile sardine (<10 months) and 1+ year group sardine catch were significantly and positively correlated with rainfall ($P < 0.05$) and negatively correlated with salinity ($P < 0.001$) and dissolved oxygen ($P < 0.01$). While the 10–14 cm TL group catches were negatively correlated to rainfall ($P < 0.001$) and positively correlated to salinity ($P < 0.05$) and dissolved oxygen ($P < 0.01$). The 10–14 and above 14 cm groups were inversely correlated ($P < 0.001$) (**Table 2**).

The monthly mean, maximum and minimum lengths during 2010–2015 are shown in **Figure 6**. The mean lengths fluctuated between 10.0 and 17.5 cm. The decrease in the mean lengths coincided with peak recruitment in those months. The variations in minimum lengths indicated that peak recruitment took place only once a year invariably during July to September. In 2011, two peak recruitments were noticed, one in February and another in July. The maximum lengths varied between 12 and 22 cm but were mostly above 15 cm.

Sardine Reproduction – Maturity, Spawning and Recruitment

Sardines were found to mature and spawn from May/June to August/September in most years although there were wide inter-annual variations in initiation of maturation and peak spawning activity. The percentage of different maturity stages in the sardine

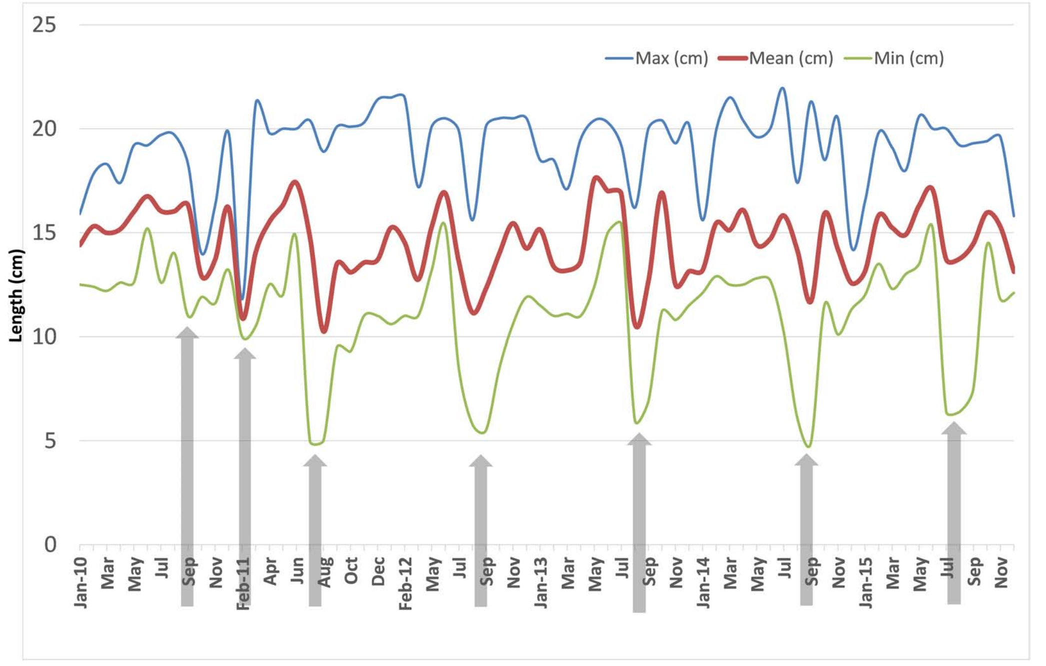

FIGURE 6 | Monthly mean, maximum and minimum size of oil sardine in fishery catches from January 2010 to December 2015. Arrowheads indicate period of peak recruitment as judged from minimum sizes.

population is given in **Figures 7a–f**. The GSI during 2013 to 2015 is presented in **Figure 8**.

In years in which the fishery was a success (2010–2013), the female maturation process was broad starting from May and strong (more than 50% of population). In a departure from this pattern, in 2014 and 2015, the maturation process was narrow and erratic with two pulses in 2015.

In 2014, though there was maturation from May onwards, the percentage mature did not exceed 40% of the population. Spawning may have taken place during 3rd week (17th and 18th of June) as inferred from the high GSI values which fell to 3.1 by end of June. However, within a month, the GSI increased and remained high throughout July as indicated by more number of partially spent sardines during the spawning period. Though recruitment was observed in July, peak was observed to be slightly delayed, shifting to August and September as inferred from the low mean lengths (**Figure 6**).

During 2015, though the gonads matured by May/June, the gonadal maturation was not complete as indicated by the low GSI values (**Figure 8**). In 2013 and 2014, the GSI values were very high indicating good maturation, but in 2015, the highest GSI value, 6.12 was observed during end of June (**Figure 8**). This low GSI decreased further to below 1, indicating that maturation was poor and incomplete. Another significant observation was the large percentage of spent resting sardines in 2015. Completely spent females did not contribute to a significant percentage.

The mature, partially spent and spent females were positively correlated ($P < 0.001$) to rainfall (**Table 2**). They were also negatively correlated to 10–14 cm TL size groups. The PSP were positively correlated to LTA ($P < 0.001$) and negatively correlated to dissolved oxygen. The PSP were also negatively correlated to catch rates of MRS ($P < 0.05$) and OBRS ($P < 0.01$).

Environmental Variations and Impacts of Ocean-Atmospheric Phenomena on Sardine Habitat

Variations in Hydrography

The major sardine habitat in the coastal waters with depth from 5 to 30 m is characterized by wide environmental fluctuations. This region is influenced by upwelling and also by the river runoff due to monsoon. The variations in temperature, salinity, dissolved oxygen, and chlorophyll-a content of the sardine habitat are shown in **Figure 9**.

The upwelling strength was highly variable in all years with strong and sustained upwelling as indicated by low temperature and low dissolved oxygen values observed during 2010–2012 (**Figures 9**, **10**). The temperature difference between the surface and bottom waters (deltaT) was high during 2012 and there seemed to be fairly cool conditions for an extended period which is reflected in the estimated LTA values. The duration and intensity of upwelling was also high during 2012 compared to that of subsequent years. During the subsequent period (2013–2015), the upwelling strength was greatly reduced.

In September, 2013 the surface waters were low saline while the bottom waters were high saline, which would have led to salinity stratification. In 2014, upwelling was weak throughout the year. The *in situ* observations show that the temperature of bottom water were below 25°C during July–August. By June, the dissolved oxygen values decreased in the 10–30 m area and the

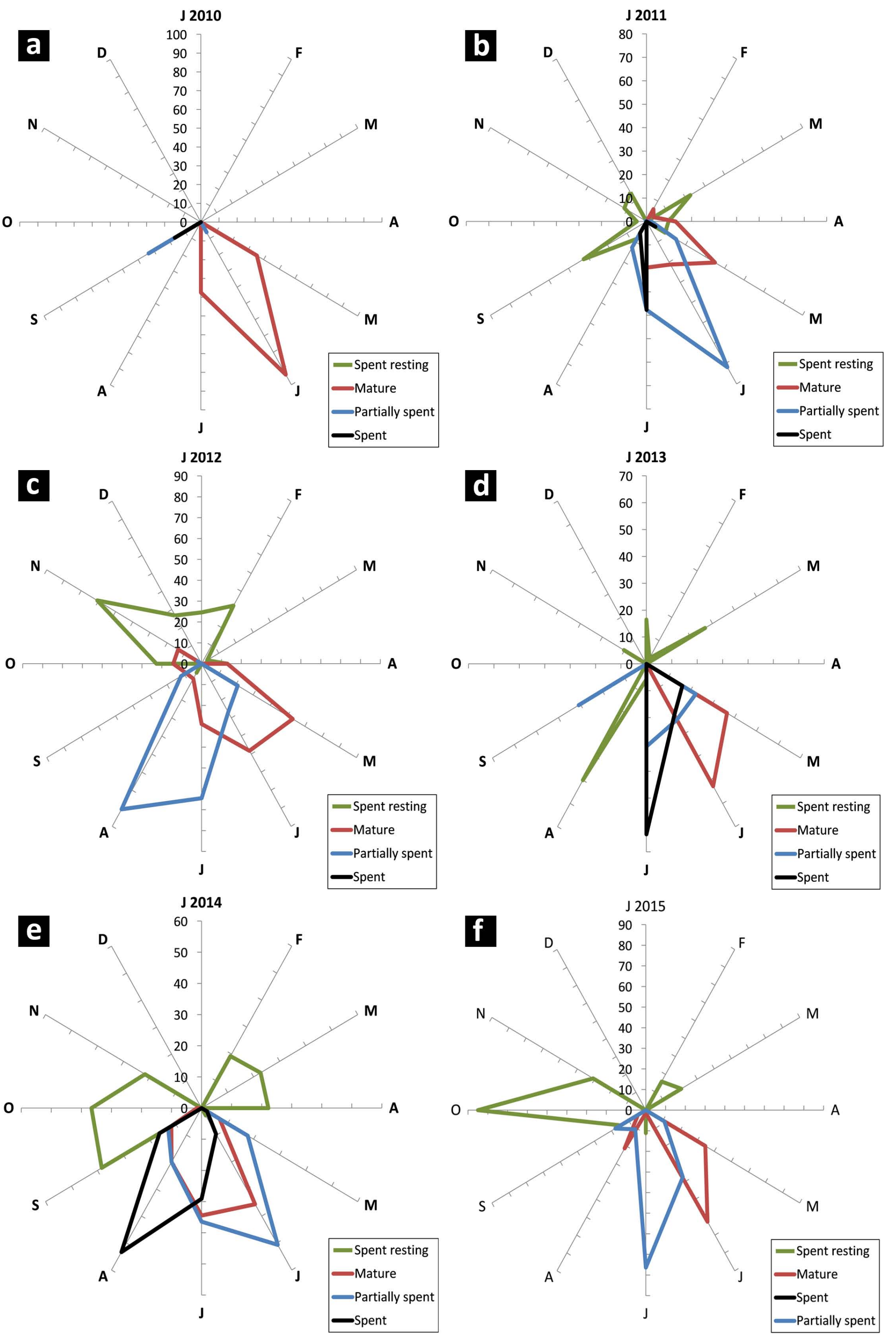

FIGURE 7 | Radar chart of monthly female maturity stages (in percent) of oil sardine from 2010 to 2015 **(a–f)**. Maturity stages important for indicating peak breeding and spawning are shown. Due to overlap, some stages are not visible.

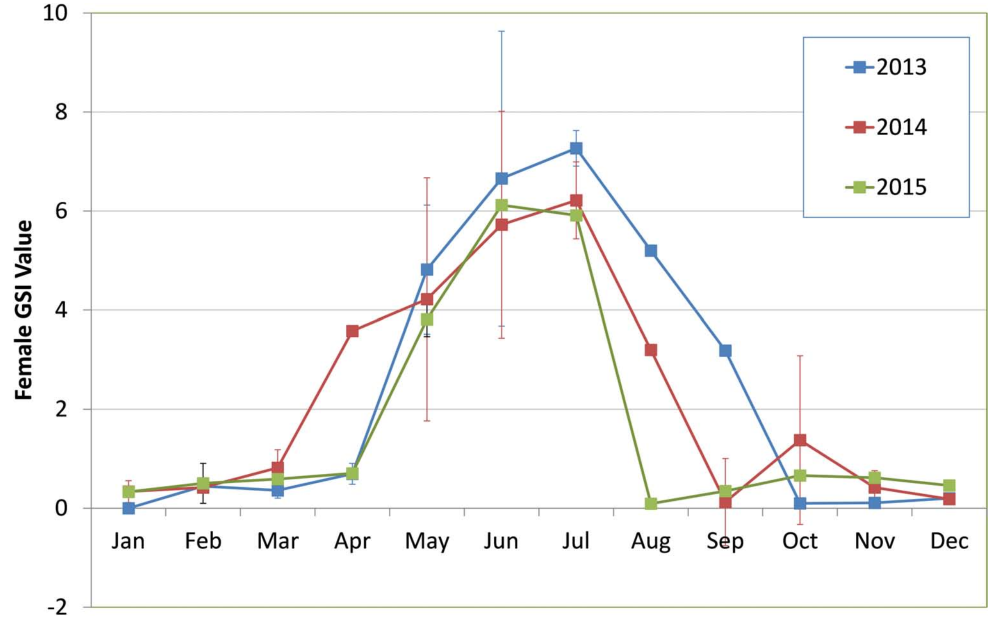

FIGURE 8 | Monthly trend in mean gonadosomatic index (GSI) values of female oil sardines during 2013–2015. Vertical lines indicate standard deviation from the mean.

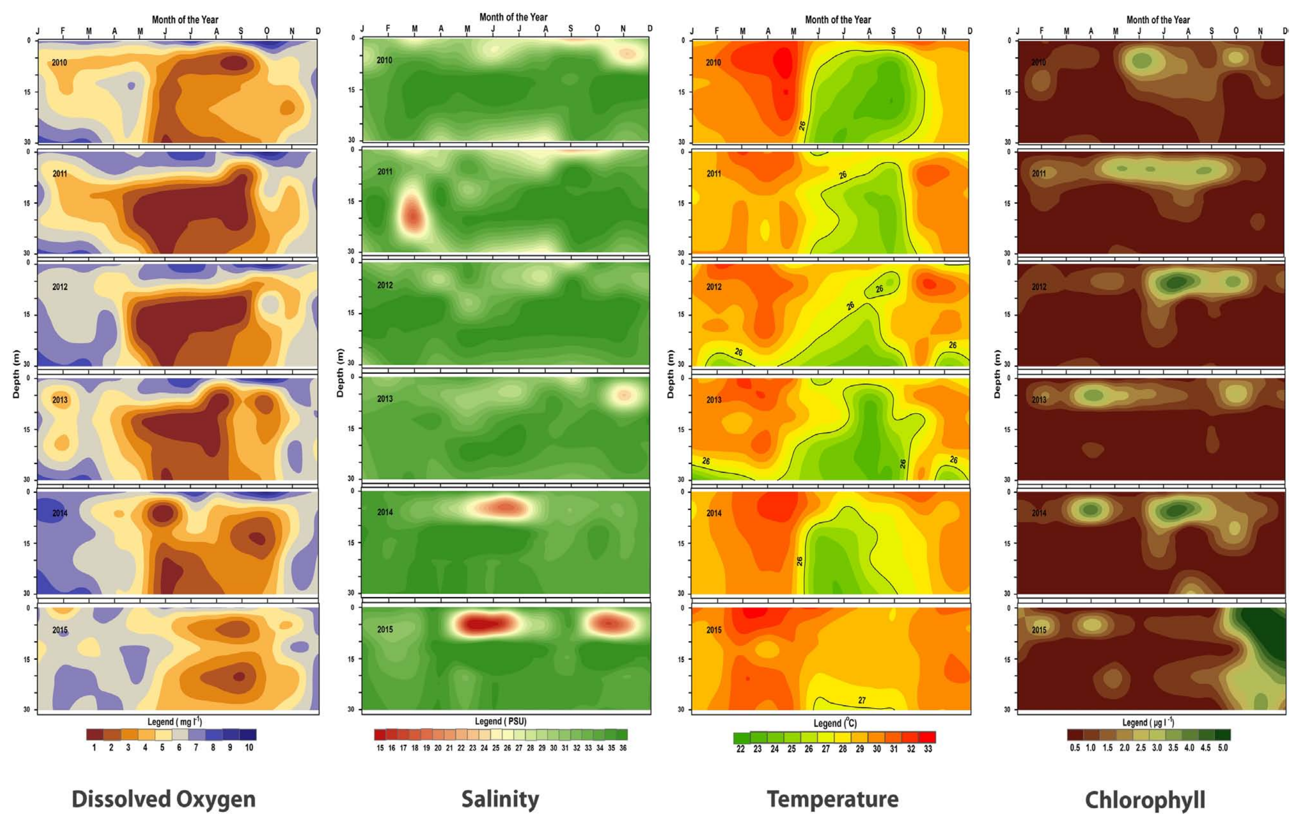

FIGURE 9 | Contour plots of monthly values of dissolved oxygen, salinity, temperature and chlorophyll-a at different depths (0–30 m) during 2010–2015 in the oil sardine fishing grounds off Kochi, southeastern Arabian Sea.

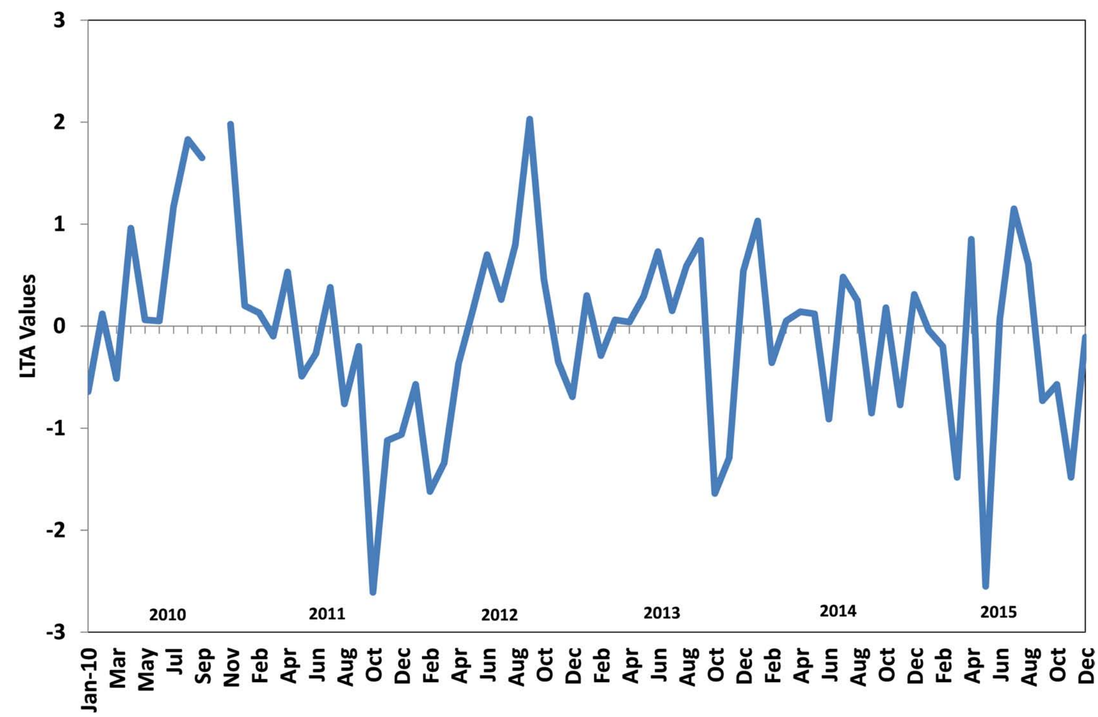

FIGURE 10 | Monthly Local Thermal Anomaly (LTA) values from 2010 to 2015 in the oil sardine fishing grounds off Kochi, southeastern Arabian Sea.

salinity remained high indicating limited upwelling in the shelf area. However, at 5 m depth the salinity had dropped due to rainfall runoff.

Upwelling was generally weak in 2015 and not observed in 5, 10, and 20 m depth zones. At 30 m depth slight lowering of temperature (26.6°C) was observed in July and as per LTA data there was upwelling in July. However, the extent and the intensity was low. Low oxygen conditions were observed but were not severe as compared to previous years (**Figure 9**). The temperature during this period was above 27°C from the surface to 20 m depth all through the year. Moreover, in 2015, ΔT was low (max = 4.4°C) indicating poor upwelling. The average temperature in the sardine fishing grounds during 2015 were nearly 1.1°C higher than that observed during the period 2010–2014.

Variations in Monsoon

The variation in monsoon and departures from normal during June to September was found to be different during all years particularly in 2013 and 2014. The rainfall during June and July of 2013 was 60 and 14% more than the normal. Monsoon was deficient during June/July of 2014 but 74 and 22% more during August and September.

The total rainfall showed a negative deviation from normal during 2012 (−24%) and 2015 (−26%) and was above normal in 2013 (+26%) and near normal in 2011 and 2014. When the deviation trend in monthly rainfall during monsoon (June to September) was plotted it showed wide fluctuations (−50% to +74%). The widest deviation from normal was observed in July, August, and June rainfall. After 2012, the deviation in September rainfall was moderate and was mostly positive.

Changes in El Niño/Southern Oscillation – ENSO

ENSO is the most important coupled ocean-atmosphere phenomenon that causes global climate variability on inter-annual time scales. The multivariate ENSO index (MEI) plotted from 2010 to 2015 showed that the index which was positive during the early part of 2010 declined and remained negative from June–July 2010 to April–May 2012 (**Figure 11**). Subsequently the MEI fluctuated without much deviation until February–March 2014, and then increased to high levels in August–September 2015.

Phytoplankton Densities

The phytoplankton cell densities in sardine fishing areas fluctuated between 1.57×10^4 cells l^{-1} in April 2013 and 3.1×10^6 cells l^{-1} in March 2014. The average values were high in 2011 and 2012, declined in April 2013 and then reached very high densities from June 2013 to December 2014. These high values were primarily due to phytoplankton blooms of diatoms species such as *Skeletonema, Chaetoceros, Bacteriastrum,* and *Fragilaria*, besides harmful dinoflagellate blooms of *Trichodesmium* (April 2014) and *Notiluca* (September 2013). In 2010, 2011, and 2012 there were phytoplankton density peaks during monsoon.

Larval Predators - Jellyfishes

During June 2013 and Aug–September 2014, in the inshore surface and column waters of the main sardine fishing area from 5 to 30 m depth zone, jellyfish blooms were observed.

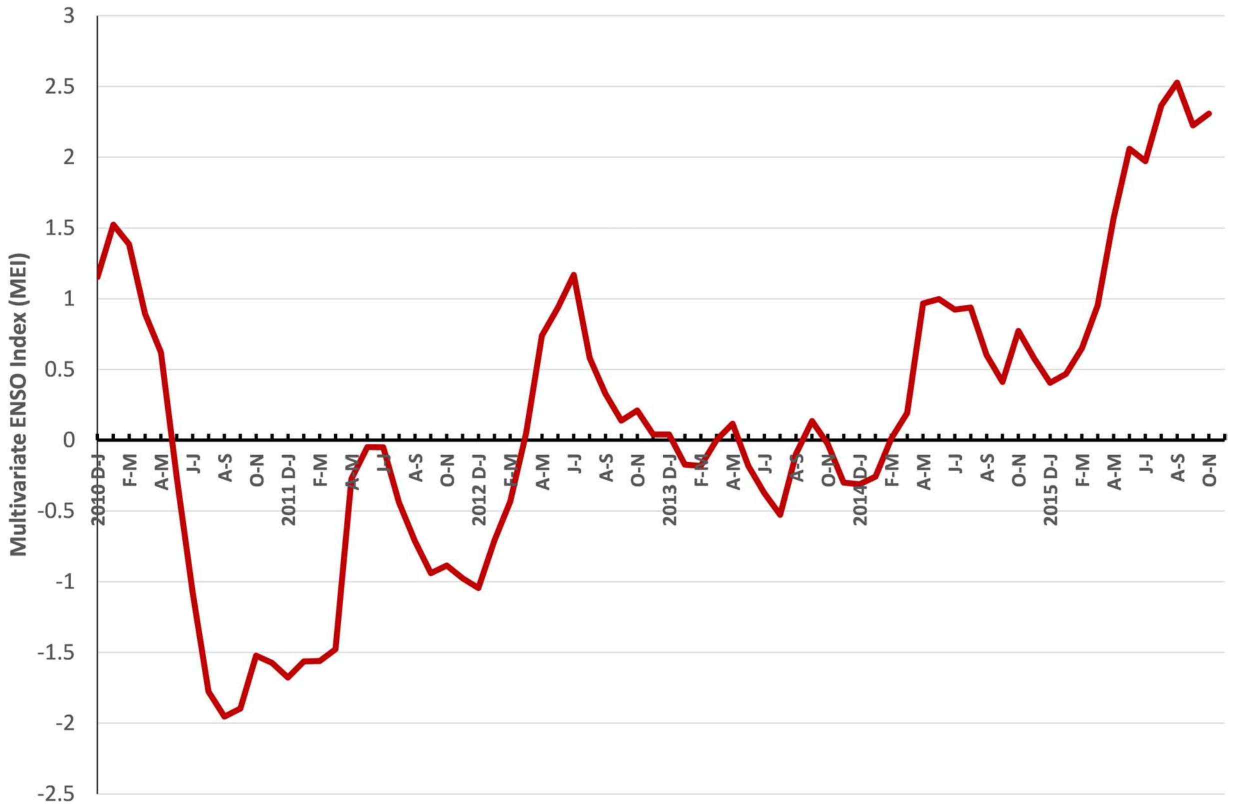

FIGURE 11 | Bimonthly multivariate ENSO index (MEI) values from 2010 to 2015.

The Hydrozoan jellyfish (*Aequorea pensilis*), Scyphozoan jellyfish (*Lychnorhiza malayensis*) and crown jellyfish (*Netrostoma coerulescens*) were observed in the sardine habitat at biomass of 1483, 918, and 4625 kg.nm^{-2} during June 2013, August and September 2014, respectively. Predation by these macro-plankters would also have affected oil sardine recruitment.

Principal Component Analysis and Relative Importance Model

Principal component analysis results (**Figure 12**) indicated that the abundance of sardines in both MRS and OBRS showed similar relations. Sardine abundance, recruitment and female maturity were more influenced by rainfall, upwelling (LTA), chlorophyll, MEI and salinity. The influence of dissolved oxygen and phytoplankton density were relatively low. The first 3 axes explains the majority (72.8%) of the variance in the data as indicated by the Eigenvalues (**Table 4**).

The assessment of relative contributions of different parameters/regressors to the fitted multiple linear regression model indicated that the CPUE in OBRS was influenced by MEI, PHY, LT10, DOX, and LTA in decreasing order (**Table 5**). Other parameters showed minimal influence. In MRS, the major regressors which influenced the CPUE were MEI, PHY, RNF, LTA, and DOX in decreasing order (**Table 5**). The proportion of variance explained by the model for CPHOBRS was 59.7% and for CPHMRS was 47.9%.

DISCUSSION

Fishery Dependent Factors

Change in Gear Length/Depth and Engine Power

An assessment of the ring seine fishery of Kerala using time series data on catch and effort during the period 1984–2004 through surplus production model had indicated that the fishing was close to the state of equilibrium in 2007 (Balan and Sathianandan, 2007). However, even after 2008, more units were operated and more intense fishing was carried out. The MSY estimated was 0.23 million tons (Balan and Sathianandan, 2007) and the landings exceeded these figures during 2010 and 2011 before peaking to 0.39 million tons in 2012. Earlier MSY estimates of oil sardine for the entire west coast were much lower indicating the fluctuating nature of the sardine biomass (Sekharan, 1974; Annigeri et al., 1992).

In the present investigation, overfishing is identified as one of the reasons for oil sardine fishery decline and the modifications made in the gear, increase in effort and area of operation have contributed to overfishing. In India, since last one and half centuries, sardines have been fished from coastal waters mainly using seines (ICAR, 1971; Pillai et al., 2003) but recently these have been enlarged in size.

In the recent past, the gear dimensions were increased. The length and width of these seines increased from about 42 m × 5.2 m in 1960s to 620 m × 100 m in 1990 and to

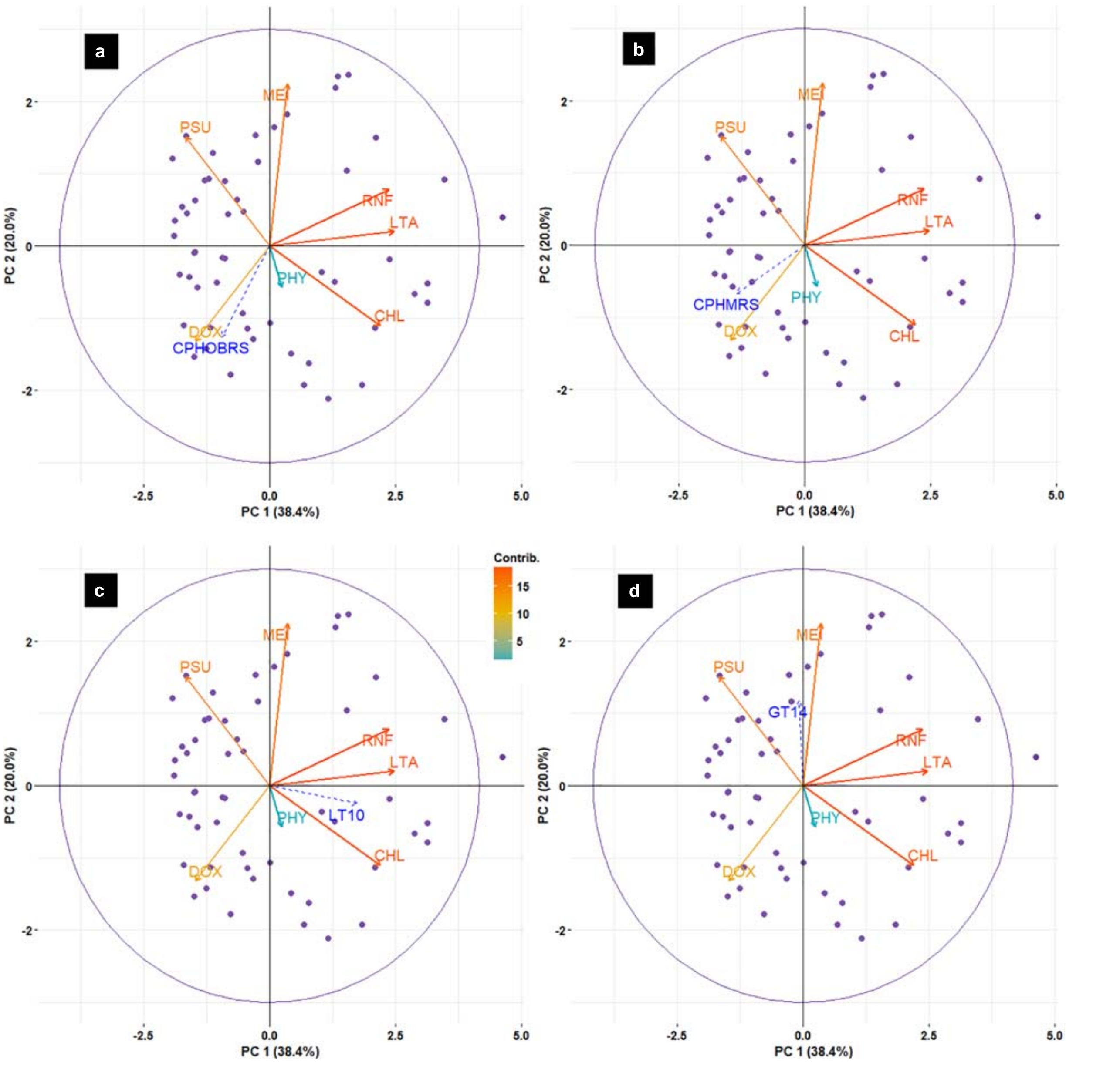

FIGURE 12 | Biplot of principal component analysis (PCA) for environmental data and oil sardine abundance [CPHOBRS **(a)** and CPHMRS **(b)**], recruitment [LT10 **(c)**] and maturity [GT14 **(d)**] in southeastern Arabian Sea.

about 900 m × 90 m in 2004 (Edwin et al., 2010). The engine capacities were increased and the number of units also exceeded the number recommended by fisheries institutes (Edwin et al., 2010; Bhoopendranath and Hameed, 2012). Our observations indicate that the engine capacity of the motorized country crafts were more than doubled to 65 hp while the inboard engine crafts were nearly doubled to 190 hp. The increased catch during 2012 can be attributed to the increased efficiency in fishing gear leading to increased fishing effort and extension of fishing grounds.

The increase in gear length would have led to increase in efficiency and more volume of the shoal would have been trapped by the encircling gear. The size of sardine shoal has been estimated to range from 2 to 20 m in length and depth (Balan, 1961) and their speed as 5 km hr^{-1}. Smaller shoals are also known to make rippling sounds and join to form larger shoals and larger gears can completely encircle the shoal.

Increase in Effort and Catch Per Unit Effort

During the period 2013–2015, the effort as well as the CPUE of both MRS and OBRS units decreased considerably which forced the units to abstain from fishing in 2015. In fact, the increase in effort during 2012 should have resulted in low CPUE if the sardine biomass was low. However, the CPUE was not affected in 2011 and 2012 mainly because there was high abundance of

TABLE 4 | Eigenvalues and the explained variance and cumulative variances of the principal components limited to 5 axes for oil sardine environmental data in southeastern Arabian Sea.

	Axis 1	Axis 2	Axis 3	Axis 4	Axis 5
Eigenvalue	2.69	1.40	1.02	0.75	0.54
% of variance explained	38.36	19.99	14.51	10.76	7.68
Cumulative % of variance explained	38.36	58.36	72.87	83.63	91.31

TABLE 5 | Percentage of variance in CPUE (CPHOBRS and CPHMRS) explained by the parameters rainfall (RNF), multivariate ENSO index (MEI), local thermal anomaly (LTA), salinity (PSU), dissolved oxygen (DOX), chlorophyll a (CHL), phytoplankton (PHY), greater than 14 mm fish (GT14) and less than 10 mm fish (LT10) and the percent contribution of each variable to the fitted regression model.

Response variable	CPHOBRS	CPHMRS
	Relative importance (%)	
RNF	6.15	13.53
MEI	33.12	28.85
LTA	7.80	10.07
PSU	1.25	4.54
DOX	9.46	9.84
CHL	3.94	4.04
PHY	22.39	24.13
GT14	0.89	1.33
LT10	15.01	3.66
Total	100.00	100.00
Proportion of variance explained by the model (%)	59.74	47.86

sardines in the coastal waters. The reason maybe the favorable environmental conditions that led to successful recruitment which led to high biomass. The sardine biomass declined in 2013, 2014, and 2015 which is indicated by the low CPUE in spite of reduction in effort and this is clearly evident in the catch per hour obtained during the period by MRS and OBRS units. Similar decline in catch rates have been recorded in forage fishes and it has been observed that such population collapses shared a set of common and unique characteristics such as high fishing pressure for several years before collapse and a sharp drop in natural population productivity (Essington et al., 2015).

Fishing Beyond Conventional Fishing Grounds

Before modernization, the sardine fishing was restricted to about 16 km from the shore (Nair and Chidambaram, 1951). Subsequently, fishing was expanded to about 25 m depth zone along the coastline. In all the documents on sardine fishery during the 19th century to this century, it has been indicated that sardines are not available in near shore areas after March and they are presumed to migrate to inshore waters during June (ICAR, 1971). The present study indicates that sardines are available in areas beyond 30 m depth during the pre-monsoon season and exploitation of this hitherto unexploited portion of the stock would have affected stock regeneration.

Biological Variations

Length Frequency

During 2012, excessive exploitation of juvenile were observed and this would have affected the sardine population in the subsequent years. In recent years there was a decline in smaller size groups and the modal length increased indicating low recruitment which can also be due to unfavorable environmental conditions. Studies (Chidambaram, 1950) on the length frequency of oil sardine during 1936 to 1943, coinciding with high catch in 1930s followed by collapse of fishery in 1943, indicated that when immature sardines were caught in large quantities, it resulted in proportionate reduction in the number of spawners in the fishery in succeeding years. Thus it is inferred that the excessive capture of juveniles affected the stock to a considerable extent.

The fisheries-induced changes in age structure may impact the dynamics of the stocks in various ways (Rouyer et al., 2011) and in the present situation the exploitation of juveniles caused an imbalance in the age structure to a population that has already been impacted by the low productivity and high abiotic stress in the preferred habitat. All size classes of oil sardine were significantly influenced by environmental variables such as rainfall, salinity and dissolved oxygen.

Sardine Reproduction – Maturity, Spawning and Recruitment

The successful recruitment during 2012 can be attributed to the good maturation and successful spawning for a prolonged period (May–November). This can also be related to the comparatively low temperature in the sardine habitat during this period. Similar instances have been observed in Monterey Bay sardine *Sardinops caerulea* a temperature dependent species which forms a fishery on the west coast of North America. Upwelling and cold water have been correlated with higher catches and catch per unit of effort (CPUE) for the Monterey Bay sardine, whereas warmer water produced lower catch and CPUE, especially during an El Nino event (Lluch-Belda et al., 1986). It was evident that the favorable environment led to prolonged spawning and this was reflected in the year classes. Conversely, when warm waters spread the coastal areas as during an El Nino event, the Californian sardines restrict their movement and the catch rates decline. Several commonalities with the Californian sardines and Japanese sardines (Kawasaki and Omori, 1988) exist along the southwest coast of India. Earlier fluctuations in oil sardine fishery have been related to the surface temperature, specific gravity of seawater, availability of food, spawning and survival of the young ones (Nair and Chidambaram, 1951).

Our evidences show that in the oil sardine all reproductive activities were heightened during the comparatively cooler period and were very poor in 2015. Based on this, we conclude that Indian oil sardines are affected by El Nino as indicated by high and positive MEI values and the high negative correlation of catch rates with MEI. Though this is a tropical habitat, within the low range of temperature fluctuations, oil sardines prefer the cooler period than the warmer period. This preference for lower temperature may be the reason for sardine catches in deeper waters during March when the surface waters are comparatively warm. This can also be the reason for the disappearance of oil sardine from coastal surface waters during the period. They become partly demersal during the warm period and maybe that is the reason for their availability in mid-water trawls during the summer months.

The oil sardine fishery along the southeastern Arabian Sea is mainly by the 0-year class and this makes the fishery vulnerable to factors which control recruitment. The oil sardine fishery along Oman coast also depends on the current year's recruitment (Al-Jufaili et al., 2006; Zaki et al., 2012). After a detailed analysis of the length frequency of sardine landed along Kerala coast during the period 1962–1966 it was concluded that the juvenile broods born during June to August support a good fishery while those born later in the year September/October fail to establish themselves due to weaker number (Raja, 1969). In 2013 and 2014 the oil sardine spawning was erratic and was extended to periods beyond August. The resultant broods would have been weak due to the dinoflagellate blooms in 2013 and the early chlorophyll maxima in 2014. The recruitment success was not strong as that of 2010, 2011, and 2012 when spawning and recruitment were early.

From the evidences gathered in the present study it can be is concluded that maturation in April/May, spawning in May/June/July and recruitment by July/August would support a good population of 10–14 cm size group which would lead to good spawning population in the ensuing year; provided, food is not limiting and there are no environmental stressors. Recruitment was mostly during July/August in almost all years. However, during 2013 and 2014, more recruitment was during August/September extending even up to October. Because of delayed spawning the food availability for the recruits was affected. This supports the match-mismatch hypothesis proposed by Cushing (1990) and later expanded by Ji et al. (2010), Asch (2015), and Checkley et al. (2017).

The environmental stressors especially salinity stratification, hypoxic conditions, predator pressures and the competition for food during the period with other ichthyoplankton would have led to larval mortality and/or low rates of larval survival. The maturation process was strongly and positively influenced by rainfall and spawning by LTA. Low recruitment led to decline of the fishery during 2013–2015. In 2015, the impacts of poor maturation during May and occurrence of large percentage of spent resting oil sardines (due to gonad atrophy) indicate poor spawning strength. The 2015 maturation process was also weak and much narrower as indicated by low GSI values. In earlier studies, spent resting stages were not observed during a larger part of the year (Longhurst and Wooster, 1990). This also would have led to poor recruitment. Availability of appropriate food is a perquisite for gonad maturation (Hunter and Leong, 1981). During 2015, phytoplankton density was low and the chlorophyll maxima occurred very late in the year which are indicative of low food availability. Moreover, the poor upwelling would have prevented nutrient enrichment. Changes in sardine reproductive traits in the northern Atlantic and the western Mediterranean have been found to be related to environmentally driven changes in food availability (Silva et al., 2006).

Environmental Variations and Impacts of Ocean-Atmospheric Phenomena on Sardine Habitat

The major coastal and ocean atmospheric events controlling the Malabar upwelling region which is the major habitat of Indian oil sardine have been described in detail (Madhupratap et al., 1994). A diagrammatic calendar on the biology of oil sardine and the major environmental events controlling these changes have also been presented (Longhurst and Wooster, 1990). Though no correlation was found between rainfall and landings, it was agreed that the major period of oil sardine abundance occurred only during periods when monsoon onset was trending toward earlier rather than later dates (Longhurst and Wooster, 1990). In the present study also rainfall was found to be positively correlated to juvenile abundance and maturity stages and PCA confirmed maximum influence of rainfall. However, high rainfall, very high positive departure from normal monsoon (during JJAS) was also found to negatively affect recruitment. Here, the theory of optimum environmental window becomes significant (Cury and Roy, 1989).

Longhurst and Wooster (1990) have argued that events taking place during the early part of the year, especially March/April, have more influence on success of sardine fishery. We also agree with this observation since we have found that the GSI starts increasing from these months clearly indicating the relationship between, nutrient enrichment from upwelling followed by diatom bloom leading to maturation. Lack of food during this period can lead to gonad atrophy (Raja, 1964) or even poor maturation as was observed during 2015. Thus, we also argue that the events during the pre-monsoon period is important. A good monsoon by itself cannot guarantee successful recruitment, if good gonad maturation has not taken place during pre-monsoon period. Hence we conclude that initiation of upwelling during pre-monsoon (for maturation); followed by normal monsoon (for spawning); which increases the food availability in the near shore waters that is essential for growth of juveniles (for recruitment) are key to the overall success of the oil sardine fishery.

Low Oxygen

One of the reasons which affected oil sardine recruitment in 2013 is the low oxygen during August in the inshore waters. Such oxygen deficient upwelled waters in the Arabian Sea have been observed earlier also (Muraleedharan and Prasannakumar, 1996; Gupta et al., 2016). It has been indicated by Longhurst and Wooster (1990) that if such low oxygen water spreads the shelf before the spawners enter the coastal waters, then they may be prevented from reaching the near shore waters and this can lead to a poor fishery. Such an instance has been observed in 1994 (Kripa et al., 2015) when very low oxygen values were reported in the near shore waters. In this investigation, oxygen deficient waters were found to affect the early life stages of the sardine and the spawners. It has been observed earlier that low oxygen waters is not a near bottom feature, rather it rises to within 10 m of surface off Kochi toward the end of monsoon and this favors the oil sardine fisheries as indicated by the positive correlation of oxygen with catch rates. However, the low oxygen condition during intense upwelling toward end of monsoon can affect recruitment of sardine as in 2014 and 2015. Both juveniles and adult sardines are affected by low oxygen values as indicated by the correlation analysis and the relative importance model.

Phytoplankton Variation - Low Food Availability

The oil sardine has been identified mainly as a filter feeder, and its main food consists of diatoms, dinoflagellates and small zooplankters (Remya et al., 2013). However, it has been observed that the intensity of grazing is high even in 10–20 m depth and that very rich feeding grounds exist at these depths. A common characteristic of forage fish collapse is the high fishing pressure for several years before collapse combined with a sharp drop in natural productivity (Essington et al., 2015). In the present study these two factors are clearly evident. Remarkably, phytoplankton densities were inversely correlated with oil sardine catch and catch rate and the relative importance model showed that food was a major factor affecting abundance. This may be due to high levels of grazing during periods of high abundance and catches. A similar inverse relationship between phytoplankton abundance and pelagic fish biomass in the Mediterranean Sea has been observed (Patti et al., 2011).

Upwelling

The LTA index which indicates the strength of the upwelling was positively correlated with maturity of oil sardine particularly partially spent females. Processes related to upwelling along the Kerala coast is usually initiated by early April much before the monsoon (Longhurst and Wooster, 1990). This correlation gives a clue that the initiation of maturation in oil sardine is trigged by the initiation of upwelling. However, this needs to be investigated in greater detail.

Upwelling is a major oceanographic phenomenon happening along the west coast of India with high inter-annual variations in its intensity. Most pelagic fishes take advantage of the upwelling and the resulting higher primary productivity during the monsoon season by timing their breeding and spawning to coincide with this period (Madhupratap et al., 1994). The upwelling index along Kerala coast showed an increasing trend from 1998 to 2007 which resulted in higher catches of small pelagics (Manjusha et al., 2013). The LTA values in the present study were generally positive up to 2012, and thereafter, showed a declining trend. Furthermore, climate change is expected to affect sardine and anchovy populations through ocean warming, change in nutrient supply and changes in food webs (Checkley et al., 2017). Therefore, the environmental events and their timing is of great importance in the maturation process, spawning and recruitment success of oil sardines in the southeastern Arabian Sea.

Impact of El Niño/Southern Oscillation - ENSO

Globally, 2015 has been considered as a warm year with high temperature and low food in the oceans (Diamond and Schreck, 2016). The average seawater temperature in sardine habitat was 29.8°C during 2015, which is nearly 1.1°C higher than the average observed (28.6°C) for the last 5 years. Positive SST anomalies exceeding 0.6°C dominated in the tropical Indian Ocean. There was a substantial warming in the tropical Indian Ocean, partially due to influences of the 2015 El Niño. The mean SST in the tropical Indian Ocean increased by 0.13–0.2°C in 2015, becoming the warmest year since 1950 (Xue et al., 2016). Explicably, the oil sardine catch and catch rates were negatively correlated with MEI and the relative importance model showed that MEI was the most important parameter explaining the variance (33%), thus indicating that the El Niño had negatively affected the sardine stock and its fisheries. In the round herring (*Etrumeus teres*) fishery in Ecuador, landings were related to the MEI in a non-linear analysis by up to 80% (Ormaza-González et al., 2016). Among the different factors affecting spawning and recruitment of anchoveta populations, El Niño has been reported as the most striking (Checkley et al., 2009).

Biotic Pressures

Intense blooms of four species of jellyfishes were observed in the sardine habitat especially in the shallower regions during the peak recruitment period. It is presumed that this would have contributed partly to the recruitment failure since they predate on zooplankton and fish larvae and are known to affect recruitment in fish populations (Lynam et al., 2005). They occupied the same niches preferred by young oil sardines. When they form blooms, jellyfishes can disrupt pelagic ecosystems (Mills, 2001). In the recent years, jellyfishes have been considered to prey on the young stages of small pelagics and compete for their food (Roux et al., 2013).

The oil sardine along the coast of Oman has also shown inter-annual changes and has exhibited a declining trend from 2001 to 2011. The major reasons cited are rising sea temperature, thermal stratification of the water column and the trophic pressure imposed on sardine populations by large pelagic predators (Piontkovski et al., 2011).

Sardine Fisheries Management and Conclusions

The Indian oil sardine plays a major role in coastal economy of Kerala and is nicknamed as "*family provider*." During 1895 the demand from the oil and fertilizer industries prompted fishermen to use small meshed nets along the Malabar (part of Kerala) Coast which led to harvesting of fishes less than 10cm. However, it is documented that elderly fishermen who had the foresight to see the danger behind use of these nets requested the local village authorities to ban use of these nets which they believed would destroy the stock of their preferred fish (Nair and Chidambaram, 1951). Maybe this move by fishermen about 120 years back is the first record of fishermen themselves coming forward with a request to impose ban on small meshed nets for protection of sardine fishery resources in the Indian sub-continent.

In 1943, when the sardine fishery declined and collapsed, the then Government of Madras, which had the control of fisheries of Kerala State restricted fishing of oil sardine in Malabar which was later extended to another 2 years from 1945. This rule prohibited use of small meshed nets for immature sardine all through the year and also controlled juvenile fishing by prohibiting landing of oil sardine below 15 cm. The legislation lapsed in 1947 due to practical difficulties encountered in enforcement.

In 2014, when the sardine fishery reached the declined status, the scientists recommended to the Government to ban fishing and landing of oil sardines below 10 cm (Mohamed et al., 2014). In the same year Government of Kerala promulgated the ban on juvenile fishing. The fishermen also understood the need for protecting juveniles and abstained from juvenile fishery. In similar instances, in other sardine fisheries, short term regulatory mechanism have been introduced to protect the stock. It has been indicated that in 1985 when more juveniles of Monterrey Bay sardine were present in the surface waters the fishery was declared as closed to prevent growth overfishing (Lluch-Belda et al., 1986).

The age and size structure of exploited fish stocks is one of the criteria for Good Environmental Status of commercial fish (Brunel and Piet, 2013). The stock collapse of sardine and anchovies in California was largely unavoidable regardless of exploitation level (Lindegren et al., 2013). Although lower catch ratios would not eliminate the probability of collapse, reducing exploitation would markedly affect the rate of decline i.e., delay the stock collapse and accelerate its subsequent recovery.

The MSY for oil sardines in Kerala has been estimated as 250,000 tons (Balan and Sathianandan, 2007) and from 2010 till 2013 (4 years), the catch has consistently exceeded this target. This over exploitation combined with unfavorable environmental conditions, have put the stock under pressure leading to decline in abundance. Based on historic catch records it has been estimated that oil sardine takes an average 8 years to recover after historic depletions (Mohamed and Veena, 2016). It has been inferred that the Indian oil sardine makes short inshore migrations for spawning along the Kerala coast. Just before spawning the adult sardines move toward the coast and then after spawning, the juveniles are more abundant in the near shore waters. The dependence of these life stages on each of these coastal niches can play significant role in the stock recovery process and the physical as well as biotic changes within the habitats can affect their survival. In order to make more meaningful spatial management of these stocks, better understanding of the stock migration is necessary.

A large number of fishers are dependent on the Indian oil sardine for their livelihood, and the current stock depletion has affected their economic well-being. Evidence gathered in the present study indicate that the key factor influencing oil sardine biomass in the Arabian Sea off Kerala is MEI and the resultant changes in food availability, rainfall, upwelling and dissolved oxygen. The unbridled increase in fishing effort and capacity is also partly responsible for the present crisis. Maintaining more abundant populations is a way to increase the species's capacity to adapt to environmental change (Sumaila et al., 2011). Although the stock productivity and recruitment are affected by various ocean-atmosphere phenomena listed above, in order to give a reasonable stability to the stock population density, it is necessary that the fisheries administration of the Kerala State urgently execute effort control and introduce a fishery closure during the peak breeding and/or spawning period.

DATA AVAILABILITY

The raw data supporting the conclusions of this manuscript have been uploaded as **Supplementary Information**.

AUTHOR CONTRIBUTIONS

VK and KM contributed to conception and design of the study. VK organized the database. SP, TA, and SK performed the statistical analysis. VK and KM wrote the first draft of the manuscript. KK, RJ, DP, PA, PN, AD, KA, JB, ND, AS, and PV collected and analyzed the samples. All authors contributed to manuscript revision, read and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00443/full#supplementary-material

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Future of Fishing for a Vulnerable Atoll: Trends in Catch and Catch-Per-Unit-Effort in Tokelau's Domestic Marine Fisheries 1950–2016

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Tokelau is among the most vulnerable countries to climate change from both an environmental and economic perspective, whilst being highly dependent on marine resources for dietary nutrition. Industrial as well as small-scale fisheries are present in Tokelau's waters, with Tokelau itself only participating in small-scale fisheries. Industrial fisheries consist exclusively of foreign distant-water tuna fleets. This study aims to reconstruct and investigate the trends in the domestic small-scale marine fisheries catches, fishing effort, and catch-per-unit-effort (CPUE) from 1950 to 2016. We used kWdays as our metric of fishing effort or fishing capacity, estimated using length, motorization and type of fishing vessels. Total fishing effort was approximately 11,900 kWdays in 1950 and increased rapidly after the 1980s with the introduction of larger motorized vessels. Despite evolving fishing effort, catches taken in subsistence fisheries have been relatively consistent at approximately 370 $t \cdot year^{-1}$, resulting in a reduction of subsistence CPUE from 32.4 $kg \cdot kWdays^{-1}$ in 1950 to 2.6 $kg \cdot kWdays^{-1}$ in 2016. This trend is opposite to that of the artisanal fishery, where CPUE increased since the start of this fishery in 2003, from 1.7 $kg \cdot kWdays^{-1}$ to 2.6 $kg \cdot kWdays^{-1}$ in 2016. Tokelau's domestic catch is greatly underreported, with reconstructed domestic catch since 2010 being nearly four times larger than the data reported by the Food and Agriculture Organization (FAO) of the United Nations on behalf of Tokelau. The abundance of reef fishes are predicted to decrease while the abundance of pelagic fishes is expected to increase within Tokelau's waters due to climate change, likely further altering future fishing practices. The present CPUE analysis, combined with the forecasted effects of climate change, suggests that the domestic fisheries in Tokelau may be on an unsustainable path, highlighting food security concerns, despite the potential for growth in offshore fisheries.

Keywords: catch reconstruction, small-scale fisheries, Pacific Islands, climate change, CPUE (catch-per-unit-effort)

INTRODUCTION

The Pacific Island countries and territories (PICTs) are considered some of the most vulnerable countries to the projected impacts of climatic change (Cheung et al., 2013; Valmonte-Santos et al., 2016). Concurrently, PICTs are among the most marine resource dependent countries on earth (Bell et al., 2013). PICTs are generally characterized by small, relatively isolated island groups, with challenging topographies and largely infertile soils (Valmonte-Santos et al., 2016). Despite human dietary shifts, with a progression from traditional local food sources to imported and processed food (Charlton et al., 2016), local fisheries continue to play a vital role in PICT tradition and food security (Bell et al., 2009; Zeller et al., 2015; Valmonte-Santos et al., 2016). Among the PICT countries, atoll states that are characterized by low-lying islands are thought to be the most susceptible to changing climates (Woodroffe, 2008), with limited arable land, water supplies and space for habitation (Storlazzi et al., 2015). Both sea level rise and increasingly severe storm surges driven by climate change are predicted to reduce atoll island land area, therefore reducing the space for land-based food production and possible habitation (Storlazzi et al., 2015, 2018). Due to the reliance on marine resources in PICTs, there is a high potential for a resource crisis in the near future as climate change increasingly alters marine ecosystems.

The Pacific Island region includes Melanesian, Polynesian, and Micronesian island countries and territories (Charlton et al., 2016). Polynesian islands are volcanic areas or low-lying coral atolls resulting in resource poor land, and Tokelau, an overseas territory of New Zealand, is the smallest in the South Pacific (Chand et al., 2003; Barnett, 2010; Charlton et al., 2016). Tokelau consists entirely out of low-lying atolls and has the smallest population of the PICTs, with just 1500 inhabitants in 2016 (Barnett, 2010; Statistics New Zealand, 2016). The small inhabitable land mass of atolls limits their capacity to support a growing population, and the emigration of a portion of each family is these days recognized as a cultural norm (Hooper and Huntsman, 1973).

Unsurprisingly, Tokelau has a high traditional dependence on local fisheries (Toloa et al., 1991). A survey by Chapman et al. (2005) revealed almost all Tokelauan households participate in fishing, indicating the socio-economic value and importance of domestic fisheries to Tokelauan society. Despite this, Gillett (2009, 2016) noted that Tokelau's per capita consumption of fish is expected to be much lower than that of its neighbor states' Kiribati and Tuvalu. This is largely due to Tokelau's comparative affluence when compared to its neighboring states, with considerable financial support from New Zealand enabling islanders to import an estimated 99.4 kg·person^{-1}·year^{-1} of alternative protein/food sources. Nonetheless, local demersal fisheries are expected to decrease in Tokelau under various climate change scenarios, although it's coral reefs are still expected to supply ~410 kg·person^{-1}·year^{-1} of fish until 2020 (Bell et al., 2011b, 2015). Although Tokelau has relatively high food imports compared to other nearby islands, local reef fishes remain a vital component in the diet and tradition (Gillett, 2016).

Unlike demersal reef fishes, tunas and nearshore pelagic fishes are projected to increase in biomass within Tokelau's waters under modeled climate change scenarios (Bell et al., 2013; Dueri et al., 2016). Currently, the Exclusive Economic Zones (EEZ) of PICTs provide over 30% of the world's tuna catches (Chand et al., 2003). License fees for foreign distant-water fishing vessels have seen an increase of 400% in recent years, creating considerable economic gains for these countries (Bell et al., 2015). However, prospective gains are limited considering Tokelau's low human development index (HDI) of 0.75 and ongoing reliance upon industrial fisheries access fees for national income (Lam et al., 2016). Tokelau is a non-self-governing territory and relies heavily upon New Zealand for financial support (Carpenter, 2015). With a remote location and small land area, Tokelau has limited capacity to expand its economic base, including fish processing plants or fish sales, with its primary asset being foreign industrial fishing access fees within the EEZ (Carpenter, 2015). In the 2014/15 financial year, revenue from industrial fishing access agreements was just over 50% of government revenue (Gillett, 2016). Given Tokelau's current dependence on foreign aid and its desire for increased self-governance (Carpenter, 2015), understanding Tokelau's dependence on fisheries for economic and subsistence purposes is particularly important. Therefore, the present study reconstructed and investigated the trends in the domestic small-scale marine fisheries catches, fishing effort, and catch-per-unit-effort (CPUE) in Tokelau from 1950 to 2016. Our work is part of a publically available global fishing database[1] and builds on and updates a previous technical catch data reconstruction for Tokelau by Zylich et al. (2011).

MATERIALS AND METHODS

Exclusive Economic Zone (EEZ)

Tokelau is comprised of three atolls (Fakaofo, Nukunonu, and Atafu) separated by relatively shallow oceanic waters (van Pel, 1958), located between 8°–10° S and 171°–173° W (**Figure 1**). Nukunonu has approximately 5.5 km^2 land area and 109 km^2 lagoon coverage, making it the largest of the three atolls. Fakaofo is located to the southeast and is the second largest, with 3 km^2 of land area and 59 km^2 of lagoon coverage (Ono and Addison, 2009), followed by Atafu to the northwest with 2.5 km^2 of land and 19 km^2 of lagoon. The small population of Tokelau is split fairly evenly between the three atolls, and mainly concentrated in one area on each atoll (van Pel, 1958). The EEZ was declared in 1980, and according to the *Sea Around Us* spatial database (Pauly and Zeller, 2015) has a total area of slightly over 319,000 km^2 with a shelf area (to 200 m depth) and an Inshore Fishing Area (Chuenpagdee et al., 2006) of 279 km^2. According to the 2006 Millennium Coral Reef Mapping Project (Andréfouët et al., 2006), Tokelau has a total coral reef area of 204 km^2 and supports a coastal fish production of 610 t·year^{-1} (Bell et al., 2015).

[1] www.seaaroundus.org

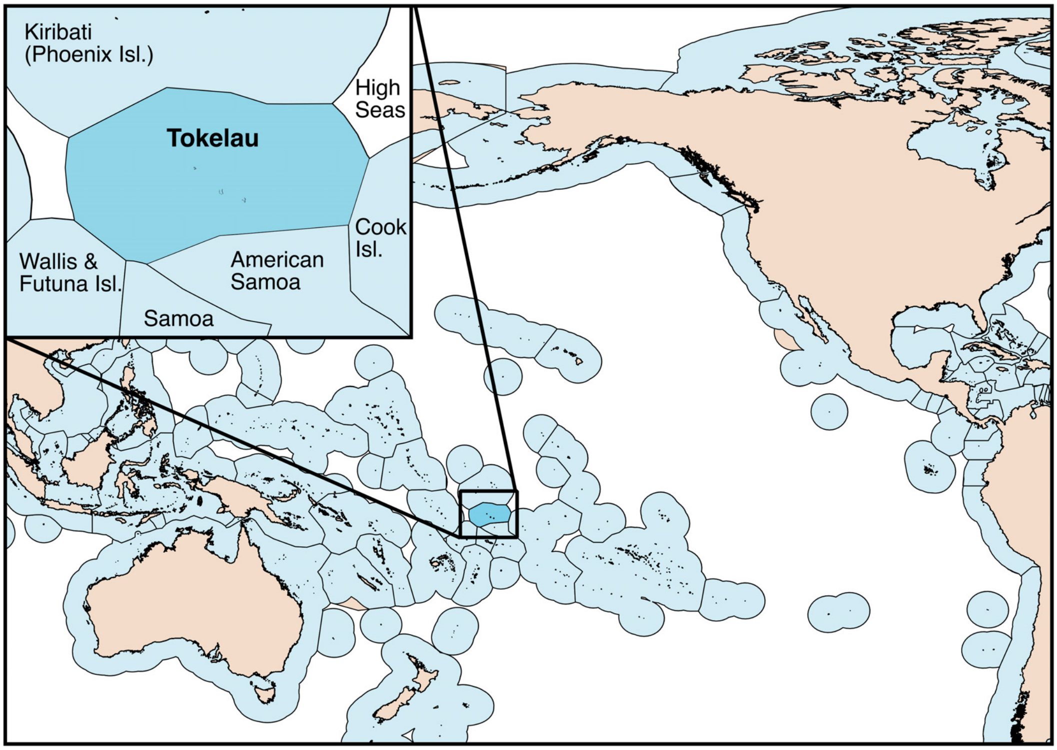

FIGURE 1 | The Exclusive Economic Zone (EEZ) of Tokelau, an overseas territory of New Zealand, and the neighboring EEZs of Kiribati (Phoenix Islands), Wallis and Futuna Islands (French overseas territory), Samoa, American Samoa (United States overseas territory), and the Cook Islands.

Catch Reconstruction

We estimated total domestic catch for Tokelau using a variety of sources and known catch amounts. For years where the estimated total catch was greater than the amount reported by the Food and Agriculture Organization of the United Nations (FAO) on behalf of Tokelau (FAO, 2018), we considered this portion as unreported catch. Several reports point to the historical absence of commercial fisheries operating domestically in Tokelau. Therefore, we assumed all domestic fishing in Tokelau in the early years was small-scale subsistence fishing, i.e., for self- and family-consumption, barter or community use (Zeller et al., 2016). Subsistence catches in Tokelau include fish and invertebrates that are landed for personal/family consumption, as gifts to send overseas, and for trade/sharing within the community. Zylich et al. (2011) derived a taxonomic breakdown for domestic catches based on van Pel (1958), Gillett and Toloa (1987), and Passfield (1998). Primary species caught domestically include flyingfish (*Cypselurus* spp.), parrotfish (Scaridae), jacks/trevallies (Carangidae), and Bigeye scad (*Selar crumenophthalmus*).

Artisanal fishing is defined as small-scale commercial fishing, and includes fish caught for sale at local markets or export (Zeller et al., 2016). As there are strong societal obligations to fulfill family and community subsistence needs before selling catch, no fishers are operating solely for commercial benefit (Vunisea, 2004). The first profit from fishing was reported in the 2006 census, where 7 households (2% of households) reported that they had made income from selling excess catch (Statistics New Zealand, 2007). In the previous census in 2001, no profits from fishing were reported (Statistics New Zealand, 2002). Based on these data, we assumed artisanal fishing began in 2003. By 2014, an estimated 10% of small-scale fishers' catch was sold for profit (Gillett, 2016). No discards are estimated for either subsistence or artisanal fisheries, as they are conservatively considered to produce minimal waste, assuming all catch is utilized for a purpose such as consumption or as bait. Baitfish was calculated by the amount of bait required for small-scale catch of near-shore tuna and other pelagic species (*Thunnus* spp., *Istiophorus platypterus*, *Sphyraena barracuda*, etc.) and Requiem sharks (Carcharhinidae) (Zylich et al., 2011). Since Tokelauan fishing primarily uses traditional methods utilizing less bait, a 32:1 ratio by Gillett (2011) was modified to 50% bait usage (64:1) and applied to the total small-scale tuna/pelagic target catch each year. Baitfish consists of an even proportion of species including but not limited to flying fish and bigeye scad throughout the entire time period.

As subsistence fishing is driven by dietary demand, we estimated yearly demand using a population time series and per capita consumption rates. For the early time period (1950–1980s), one trading ship brought Tokelau basic items such as flour, sugar, rice, kerosene, and tobacco three to four times per year (van Pel, 1958). It is assumed due to Tokelau's remote location that they were not receiving canned meats or fish at this time (Passfield, 1998). The preliminary catch reconstruction by Zylich et al. (2011) estimated a per capita seafood consumption rate of 255 kg·person^{-1}·year^{-1} based on the assumption that total animal protein was supplied by fresh seafood. Gillett (2009) estimated the 2007 total catch for Tokelau to be 250 tonnes based on estimates from several independent studies completed between 1977 and 1998, including Hooper (1985) and Passfield (1998). Using Gillett's estimate of 250 tonnes in 2007 and updated population data, we estimate a fresh seafood consumption/catch rate of 171.8 kg·person^{-1}·year^{-1} since 2007. As the availability of alternative protein sources has increased over the years due to increased food imports, and demand for fresh seafood has declined, we linearly interpolated between 255 kg·person^{-1}·year^{-1} in 1950 and 171.8 kg·person^{-1}·year^{-1} in 2007 (held constant after 2007) to create a complete time series of fresh seafood demand in Tokelau.

Prior to 1980, there were no known seafood exports from Tokelau. However, by 1998 an estimated 5.4 tonnes of catch were exported from Fakaofo, and approximately 16.2 tonnes exported for all of Tokelau (Passfield, 1998). Exports have been steadily increasing since, and an estimated 125 tonnes of catch were exported in 2007 due to increased transport between Tokelau and other PICTs, such as Samoa (Gillett, 2009). Between 2005 and 2014, fish exports were estimated to have doubled, increasing to 192 tonnes in 2014 (Gillett, 2016). Although the majority of exports from Tokelau are considered to originate from subsistence fisheries, consisting of gifts of frozen and dried fish for friends and family (Hooper, 1985; Passfield, 1998; Gillett, 2009), artisanal exports have also been increasing in proportion in recent years (Gillett, 2016).

Fishing Effort (Fishing Capacity)

Fishing effort was estimated as fishing capacity of the small-scale fishing fleet using the method developed by Greer (2014), rather than as a temporal measure of fishing effort only (e.g., trip duration) or gear specific effort (e.g., hook-hours or net kilometers). Fishing capacity was chosen in order to remain consistent with global fishing effort database developments by the *Sea Around Us*, to allow standardized comparisons between a wide range of fishing fleets, gears and fishing approaches around the world. A literature search was performed to identify information about Tokelau's small-scale fishing fleets from 1950 to 2016, including the number, size, motorization and type of fishing vessels, the number of fishing days and the gears utilized. Prior to the first vessel census in 2003 (Chapman et al., 2005), we estimated the number of vessels for 1950 to 2003 by applying the rate of household vessel ownership in 2003 (85%)(Chapman et al., 2005) to historical population data (Bertram and Watters, 1984; Statistics New Zealand, 1992, 1998, 2002). Vessel surveys were subsequently conducted in 2006, 2011 and 2016 during the national censuses (Statistics New Zealand, 2007, 2012, 2017). These surveys were used as anchor points and linear interpolation was used to create a continuous vessel time series from 2003 to 2016. We assumed a constant household fishing participation rate of 99% across the study period (Chapman et al., 2005). For the households participating in fishing without a vessel, we assumed half of the household members were engaging in shore- or reef-based fishing activities. The manpower of these shore-based fishers was converted to a vessel equivalency using a ratio of 5 shore fishers to one unmotorized canoe, the largest crew size of a traditional Tokelauan fishing group, for a conservative estimate of fishing effort (Hooper and Tinielu, 2012).

Vessels were assigned a length based on available information. Tokelauan fishers traditionally used 7–10 m outrigger canoes built from wooden planks (Chapman, 2004). These were used exclusively until the importation of 3–5 m motorized aluminum dinghies beginning in the 1980s, and which were wide-spread by the 1990s (Watt and Chapman, 1998; Chapman, 2004). By the first vessel survey in 2003, aluminum dinghies were the most common fishing vessel, comprising more than 80% of the fishing fleet (Chapman et al., 2005).

Fishing effort (kWdays) was then estimated by the product of the number of vessels, vessel engine power and days fished. Engine power (kW) was inferred from the length of motorized fishing vessels using the equation:

$$\mathrm{kW} = 0.436^{*}\mathrm{L}^{2.021}$$

where L is vessel length in meters (Anticamara et al., 2011). Aluminum dinghies were assigned a power of 9.11 kW, assuming vessels were an average length of 4.5 meters. Unmotorized vessels (i.e., wooden canoes) were assigned an engine equivalent value of 0.37 kW, assuming that one fisher can produce 74.57 watts of work over an eight-hour work day (Avallone et al., 2007). We assumed that fishers spent 121 days per year fishing based on an average household estimate of 14 person-hours per week fishing (Passfield, 1998).

Finally, a fishing sector was assigned to the estimated fishing effort. We assumed artisanal fishing began in 2003 as no profits from fishing were reported in 2001, with the first reported profit from fishing in the subsequent census in 2006 (Statistics New Zealand, 2002, 2007). By 2014, an estimated 10% of small-scale fishers' catch was sold for profit (Gillett, 2016), i.e., was deemed artisanal. We linearly interpolated between 0 in 2002 and 10% in 2014 to assign a proportion of fishing effort to the artisanal sector for 2003 to 2013. Subsequently, we assumed that the proportion of artisanal fishing effort has remained constant.

Catch Per Unit Effort (CPUE)

As catch volumes and effort alone often do not provide sufficient information on the status of a country's marine resources, we derived catch per unit effort (CPUE) to better describe the development of fishing in Tokelau. We recognize that CPUE data needs to be treated with caution, and can be problematic as an indicator of stock status or abundance (Maunder et al., 2006; Harley et al., 2011), but in the absence of better data on abundance or biomass of exploited domestic fisheries stocks in Tokelau, we use CPUE here as a minimal indicator. A CPUE time-series was computed from 1950 to 2016 for both the artisanal and subsistence sectors. In order to derive the CPUE per sector each year, we divided the reconstructed catch time-series for each fishing sector by the calculated standardized effort (capacity) for the same sector for 1950 to 2016.

RESULTS

Domestic Catches

The reconstructed domestic small-scale catches by Tokelau from 1950 to 2016 summed to 25,400 tonnes, and averaged between 300 and 450 t·year^{-1} (**Figure 2A**). Subsistence catches increased from around 380 tonnes in 1950 to a peak of 458 tonnes in 1960 (**Figure 2A**), when the population on the islands reached a maximum of approximately 1900 people. This was followed by a gradual decline in subsistence catches due to declining human population trends and increasing alternative protein imports, such as canned meats. Thus, subsistence demand decreased after 1960 to an all-time low of 295 tonnes in 1996, before increasing again due to increasing exports to 424 tonnes by 2016 (**Figure 2A**). Artisanal catches gradually increase from around 3 tonnes in 2003 to 47 tonnes in 2016 (**Figure 2A**).

The domestic small-scale catches as reconstructed here for Tokelau (**Figure 2A**) are dominated by reef and reef-associated species (*Cypselurus* spp., Scaridae, and *Selar crumenophthalmus*) (**Figure 2C**), and domestic catches are very small compared to the industrial distant-water fleet catches of large pelagic species (mainly tuna and billfishes) taken within the EEZ of Tokelau (**Figure 2B**). However, while these foreign fleet-dominated fisheries provide substantial foreign exchange earnings for Tokelau, they do not contribute directly to food security and locally consumed seafood.

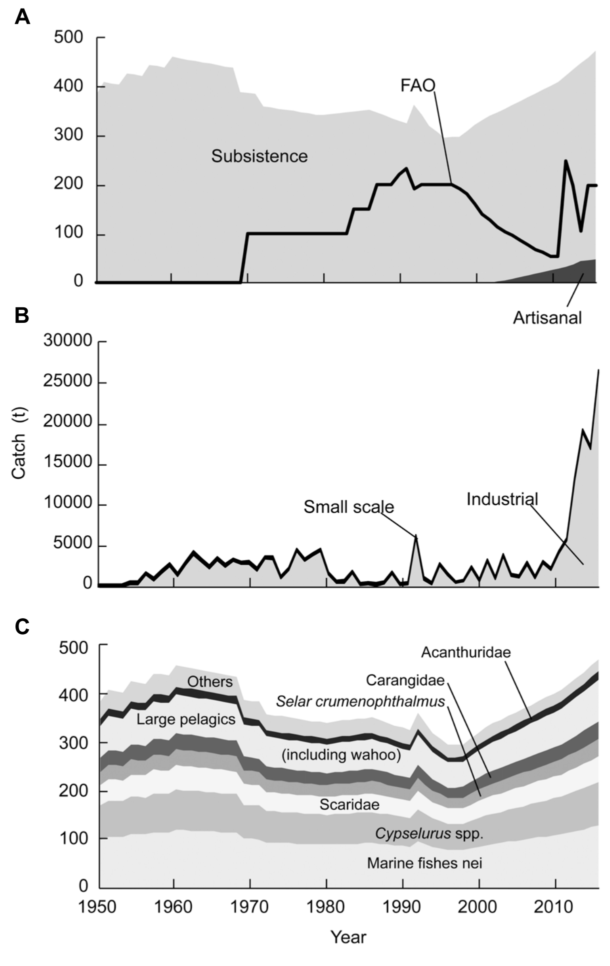

FIGURE 2 | Fisheries catches in Tokelau's EEZ, showing **(A)** domestic artisanal (small-scale commercial) and subsistence (small-scale non-commercial) catches for 1950–2016 as reconstructed here. The data reported to FAO by New Zealand on behalf of Tokelau are overlaid as a solid line; **(B)** total catches in the EEZ, consisting of both industrial distant-water fleet catches of large pelagics by foreign fleets (Le Manach et al., 2016) and domestic small-scale catches (as presented in **A**); and **(C)** taxonomic composition derived for Tokelau's small-scale (subsistence and artisanal) fisheries.

Fishing Effort (Fishing Capacity)

Total fishing effort estimated as fishing vessel capacity has grown from 11,900 kWdays in 1950 to 180,600 kWdays in 2016 (**Figure 3A**). Despite a gradual decrease in the total number of vessels from a maximum of 273 vessels in 1966 to 212 in 2016 (**Figure 3B**), fishing effort in terms of vessel capacity has

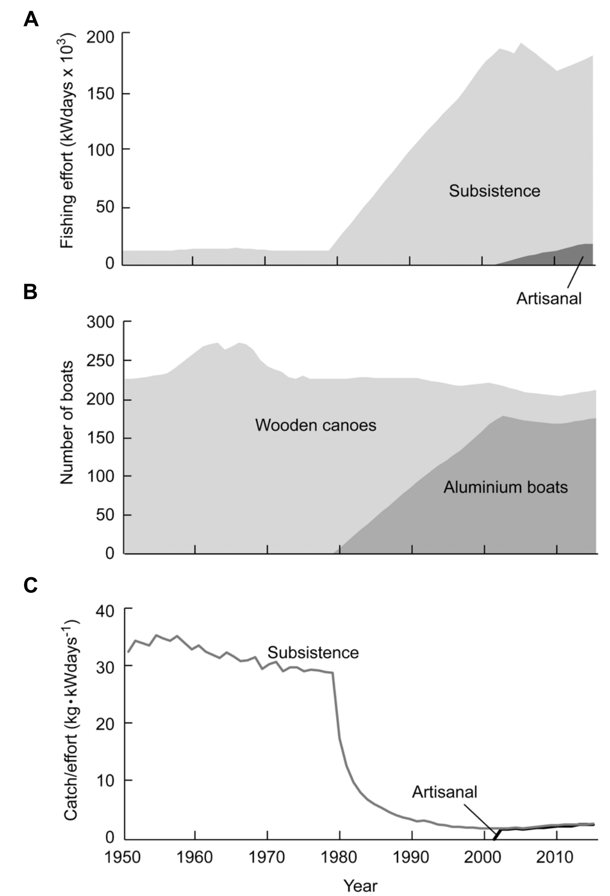

FIGURE 3 | Fishing effort for the domestic, small-scale fisheries in Tokelau as **(A)** fishing capacity (kWdays) by fishing sector; **(B)** total number of vessels by vessel type from 1950 to 2016; and **(C)** catch-per-unit-effort (CPUE) of Tokelau's domestic subsistence and artisanal fisheries from 1950 and 2003, respectively, until 2016.

increased rapidly since 1980. This effort (capacity) increase is entirely due to the introduction and wide-spread change of vessel type from canoes to motorized aluminum dinghies since 1980. Artisanal fishing began in 2003, with 1,565 kWdays of effort exerted, and has grown to 18,000 kWdays in recent years (**Figure 3A**).

Catch Per Unit Effort (CPUE)

The subsistence CPUE declined steadily from a peak of around 35 kg·kWdays^{-1} in the mid 1950s to approximately 29 kg·kWdays^{-1} in the late 1970s (**Figure 3C**). From 1970 to the mid-1990s the CPUE for the subsistence fisheries declined strongly, driven exclusively by the increased fishing capacity with the introduction of motorized aluminum vessels, followed by a stabilization at around 2.2 kg·kWdays^{-1} since 2000.

The artisanal CPUE increased from 1.7 kg·kWdays^{-1} at the start of the artisanal fishery in 2003 to a peak of 2.6 kg·kWdays^{-1} in 2015, while in 2016 the CPUE dropped slightly to 2.1 kg·kWdays^{-1} (**Figure 3C**).

DISCUSSION

Fresh fish has been a staple in the diet of the Tokelauan people for a long time (Passfield, 1998). This tradition is reflected in the reconstructed subsistence catch, which mirrors the changes in human population size on the islands, especially in earlier years (see **Figure 2A**). In more recent years, subsistence catches declined due to the declining human population and increases in alternative protein imports, with a minimum subsistence catch of 295t in 1996 (Gillett, 2016). Subsequently, increasing availability of motorized aluminum fishing boats in the late 1990s, allowing more rapid and easier movement between local fishing areas and islands led again to a rise in subsistence catches, generally driven by non-commercial exports to friends and family out-of country (Gillett, 2009). As we did not account for alternative factors (e.g., extreme weather events) potentially leading to inter-annual variations in catches, the fluctuations in the subsistence catch from 1950 to 2016 are due to human population changes, exports (gifts) and imports (alternative proteins).

Despite the ubiquitous participation in small-scale fishing in Tokelau, present day *per capita* fish consumption rates are not as high as in many neighboring atoll countries due to the availability of easy imports of non-fish proteins from New Zealand (Gillett, 2016). In the past, all or nearly all members of the community participated in the catching of small fishes and invertebrates on the reefs (often led by women), or poled for larger pelagic fishes on the outer edges of the reefs and in near-reef oceanic waters (Chapman, 1987; Toloa et al., 1991; Ono and Addison, 2009). With almost 100% of Tokelau's population actively engaged in fishing (Gillett, 2016; Weissbach, 2017), marine resources are arguably the most important socio-economic entity in Tokelau, and have been fundamental to the traditional Tokelau lifestyle (Toloa et al., 1991).

The introduction of new fishing technologies such as goggles, nets and spear guns has considerably increased the pressures on existing resources, with the most notable being the introduction of motorized aluminum boats in the 1980s. New technology has allowed for intensified pressure on and faster access to reef dwelling fishes, but has also allowed fishers to fish in near-reef oceanic waters targeting more pelagic fishes (Chapman, 2004). This intensification in fishing capacity is clearly evidenced here in the strong increase in estimated fishing effort (**Figure 3A**) despite little changes in the number of vessels (**Figure 3B**). Though capacity is an imperfect proxy for fishing effort, as it does not capture the true extent of fishing activities (Bell et al., 2017), we made the assumption of proportionality in absence of detailed information on fishing activities performed by small vessels with low mechanization. Tokelau's fisheries are highly integrated into society and driven by subsistence need. Despite this importance to the wellbeing of Tokelauan society, very little information is available about their fisheries. We encourage investigations by local and regional experts to develop and disseminate more knowledge and data from these data-poor fisheries.

This strong increase in fishing effort measure (in terms of fishing capacity) has led to strongly declining catch per unit effort rates (CPUE) during the same time period (**Figure 3C**). The changes in CPUE suggest that despite the new technologies the catch rates may not have improved but have rather declined (King, 1991; Toloa et al., 1991). It is important to note that although artisanal CPUE displayed an initially increasing trend, both the subsistence and artisanal CPUE show a period of rapid change, followed by what appears to be a settling of the catch rates. Single assessments sometimes skew measures of fish stock health, particularly CPUE studies as they often do not account for spatial information such spatial expansion in the area fished (Walters, 2003; Maunder et al., 2006). Domestic fishing in Tokelau may have slightly expanded beyond the atoll reef boundaries with the increased availability of motorized vessels, increasingly targeting more pelagic species. However, with the largest vessels being ∼5 m open aluminum dinghies, fishing range is most likely still relatively restricted spatially, considering the full scale of the EEZ. Thus, due to the localized nature of Tokelau's domestic small-scale fisheries, we suggest that the potential growth in fished areas due to technological effort developments has not significantly impacted the CPUE measure (**Figure 3C**).

In the late 1980s to early 1990s, the average South Pacific CPUE for large pelagic species (e.g., *Katsuwonus pelamis* and *Thunnus albacares*) in the troll line fisheries was 5 kg·line^{-1}·h^{-1}, almost half that of Tokelau's 9 kg·line^{-1}·h^{-1} (Dalzell et al., 1996). Tokelau had the largest troll line CPUE rate range in the study and was assessed before the commencement of artisanal fishing and expansion of industrial fishing within the EEZ. Immediate neighbor American Samoa had an equivalent CPUE of 3 kg·line^{-1}·h^{-1} (Dalzell et al., 1996), declining to 1 kg·line^{-1}·h^{-1} by the early 2000s (Craig et al., 2008). The average CPUE for beach seines for Tokelau in the 1980s and 1990s was 14 kg·set^{-1}, similar to Cook Islands 13 kg·set^{-1} but low compared to the 131 kg·set^{-1} in Kiribati (Dalzell et al., 1996). Gillett and Tauati (2018) estimated annual small-scale tuna catches of 41 kg·person^{-1}·year^{-1}, matching our 2016 estimates. Tuna (*Thunnus albacares* and *T. obesus*) per capita catch rates have been increasing throughout the time series, peaking with 41 kg·person^{-1}·year^{-1} in 2016.

The most notable change in the general domestic fisheries system in Tokelau over the last 60+ years was the commencement of small-scale commercial (i.e., artisanal) fisheries. Artisanal fishing was non-existent prior to the early 2000s, mainly due to Tokelau's community structure, traditional community sharing culture, small population and isolation. Commercial small-scale fisheries have slowly been growing, in part due to new technological adaptations such as motorized aluminum dinghies and modernized fishing gear (Watt and Chapman, 1998; Chapman et al., 2005) as well as increased market access (Gillett, 2016). The expansion in fishing for profit in many Pacific Islands has created a divide in traditional fisheries management practices (King, 1991; Gillett and Tauati, 2018). In the past, each atoll in Tokelau self-governed their communities and the surrounding reefs (Toloa et al., 1991). Self-governing controlled by elders included everything from releasing live small fish, preventing harmful fishing methods and total fishing bans (lafu) in certain areas (Toloa et al., 1991). However, when fishing for profit becomes a prominent component of income for some members of a community, fishing bans and other restrictions can lead to strong opposition, often resulting in less effective management (Voyer et al., 2014; Gillett and Tauati, 2018). As artisanal fishing grows, Tokelau will be faced with the need to take more official and formal management steps such as licensing and restrictions on the taxonomic composition of catch (targeting), as well as gear use and place and time of permitted fishing (Walters, 2000; Allison and Ellis, 2001).

Over the same time period when these changes to domestic fisheries occurred, foreign fishing for large pelagics has also increased in Tokelau's EEZ waters (**Figure 2B**). This has the potential for future conflict between domestic and foreign tuna fishing, risking clashes between domestic food demand and foreign exchange earnings for Tokelau. With the ever-rising global demand for seafood, in particular tuna (Jacquet and Pauly, 2007), controlling and restricting the expansion of industrial fisheries within the Tokelau EEZ is vital. Tokelauans do not have the facilities to monitor large offshore fishing vessels, thus as a key stake-holder in the future of Tokelau's fisheries, New Zealand has the incentive for promoting sustainable fishing practices (Carpenter, 2015; Gillett, 2016). As a member of the Pacific Islands Forum Fisheries Agency (FFA) Tokelau receives support for ecosystem based management to avoid oceanic mismanagement (FFA, 2017), a main driver of the collapse of fisheries (Jacquet and Pauly, 2007).

Tokelau is particularly vulnerable to the effects of climate change due to its low atoll elevation, small landmass, geographic remoteness, food and water insecurity, as well as its dependence on financial support and food imports from New Zealand (Hastings, 2009; Lam et al., 2016; Valmonte-Santos et al., 2016). With the predicted climate variability, weather extremes, sea level rise and ocean warming, Tokelau can expect significant impacts on its natural ecosystems, economy and way of life (Diamond et al., 2012; Storlazzi et al., 2018). As Tokelau is a low-lying atoll, the most serious environmental impacts will almost certainly present as severe weather events, coastal erosion, land inundation and degradation, sea level rise driven flooding, larger shore-waves, and fresh water contamination (Connell, 2016; Barnett, 2017; Storlazzi et al., 2018). Even if Tokelau remains habitable in the long term, these effects are estimated to devastate the already extremely limited agriculture potential, as well as damage infrastructure, such as boats and docks, and decrease coastal fish stocks (Barnett, 2010; Storlazzi et al., 2015, 2018).

Given that domestic fishing in Tokelau is currently heavily reef based or reef-associated, a main concern of climate change is the accelerated rate of ocean warming with combined effects leading to an overall decline in reef fishes (Cheung et al., 2013; Barnett, 2017; Cheng et al., 2017). Overall, the effects of likely increasing fishing pressures from both the small- and large-scale fisheries, combined with climate change will likely have a negative effect on the marine resources of Tokelau, although Tokelau's EEZ is thought to benefit from climate change driven increases in some pelagic species availability due to a rise in El Niño systems (Bell et al., 2011a). Emigration is already common in Tokelau, mainly for economic reasons, and currently assists in maintaining a semblance of sustainability (Hooper and Huntsman, 1973). Despite the desire for independence, Tokelau's reliance on New Zealand for alternative protein imports, financial assistance, and emigration will likely grow as climate change challenges local terrestrial and marine resources. Although the potential surge in pelagic fishes can lead to more foreign exchange earnings (Bell et al., 2013), domestic catches will most likely remain low. Realistically, these economic gains provide no direct benefit to the traditional Tokelauan lifestyle and do not directly increase local food security. The increasing dependence on imports will decrease traditional practices, and likely the health of the Tokelauan people, as activity declines and consumption of canned foods increases (Valmonte-Santos et al., 2016).

AUTHOR CONTRIBUTIONS

RW and ARC: catch reconstruction and synthesis, drafted the manuscript. AC: effort reconstruction and synthesis, drafted the manuscript. MP, DP, and DZ: expertise and guidance of reconstruction and data management, drafted and edited the manuscript.

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ORIGINAL RESEARCH
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Long-Term Fishing Catch and Effort Trends in the Republic of the Marshall Islands, With Emphasis on the Small-Scale Sectors

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Small-scale fishing has been an important element of the livelihood and food security in Pacific island countries throughout history; however, such catches have been under-reported in the official fisheries data. Here, we reconstruct the total domestic catches and fishing effort of the Republic of the Marshall Islands (RMI) by fishing sectors for 1950–2017. Reconstructed total catches were estimated to be 27% higher than the data officially reported by the Food and Agriculture Organization (FAO) of the United Nations on behalf of the RMI. Catches of the truly domestic, but export-oriented, industrial tuna sector accounted for 84% of the total catch, dominating catches since the early 2000s. The subsistence component contributed 74% of total small-scale catches, of which 92% was deemed unreported. The remaining 26% of small-scale catches were artisanal, i.e., small-scale commercial, in nature, of which 45% was deemed unreported. Trends suggested steady growth in small-scale catches from 1,100 t·year^{-1} in the early 1950s to a relatively stable level of 4,500 t·year^{-1} since the 1990s. However, over the 2009–2017 period, there was a gradual reduction of 2% per year in subsistence fishing, which was paralleled by a concomitant increase in artisanal catches of 3% per year. This gradual shift from predominantly non-commercial to commercial small-scale fisheries may be related to efforts to commercialize small-scale fisheries in the past decades. Small-scale fishing effort increased approximately 13-fold from the early 1980s to the late 2000s, stabilizing at around 401,000 kWdays since then, while catch-per-unit-of-effort (CPUE) displayed an inverse pattern, declining eightfold between the 1980s and 1990s, and stabilizing around 15 kg·kWdays^{-1} in recent decades. These findings may assist sustainable coastal fisheries management in the RMI, which is particularly important given the increasing impacts of climate change on local stocks.

Keywords: catch reconstruction, artisanal fisheries, subsistence fisheries, fishing effort, fisheries catch data, catch-per-unit-of-effort, Pacific islands, shark catches

INTRODUCTION

For millennia, small-scale fisheries have been playing a prominent role in food security and livelihoods for Pacific island countries (Johannes, 1978). Despite the importance of these fisheries, their contribution to national catches has been largely underestimated in the official catch statistics in most Pacific island countries and territories (Zeller et al., 2015). This under-representation exemplifies the historical marginalization of small-scale sectors (Pauly, 2006), which is an issue that, thankfully is slowly being addressed (FAO, 2015; Pauly and Charles, 2015). The lack of comprehensive and representative data on small-scale fisheries is often attributed to the challenges in monitoring artisanal and subsistence catches, as these are generally landed at a large number of spatially dispersed landings sites that are hard to monitor. The poor accounting is also often justified by the incorrect assumption that catches from these sectors are not substantial in volume or economic importance (Zeller et al., 2006, 2015) and thus not a high government priority. However, reconstructions of the small-scale catches showed that these components account for approximately 70% of the total domestic catches in Pacific small-island countries, and are crucial for the food security and livelihood of local populations (Zeller et al., 2015). In the past, traditional societal rules provided some degree of management and control of fishing effort though spatial and temporal fishing restrictions as well as rotation of fishing grounds (Johannes, 1978). However, the transition to centralized government systems in many of these countries, and the advent of increased commercialization of small-scale fisheries has weakened these traditional control mechanisms, often without the implementation of adequate new approaches to effective management (Beger et al., 2008).

The seafood provided by small-scale fisheries constitutes a major food security resource and is critical for good nutrition in most coastal developing countries (Golden et al., 2016). In the Republic of the Marshall Islands (RMI, **Figure 1**), fishing is an important activity (Beger et al., 2008). Similar to other Pacific islands, annual consumption of fish in the RMI is disproportionally high compared to other coastal countries (Bell et al., 2009; Kronen et al., 2010). Recognizing the importance of the fisheries sector, the Marshall Islands government made strategic investments in the 1990s to develop the fishing industry (both small- and large-scale), aiming to improve the rural economy and meet the domestic demand for seafood in the main population centers on Majuro and Ebeye Islands (Smith, 1992; Gillett, 2010). Following the decrease in copra “dried coconut meat” exports, a former stable export, the government also focused on increasing foreign exchange earnings from fisheries resources by signing foreign fishing access agreements with several countries, including South Korea, Japan, and Taiwan, for offshore tuna fishing in the RMI Exclusive Economic Zone (EEZ) waters (Beger et al., 2008). These agreements accounted for the majority of foreign revenues generated by the large-scale fishing industry, and with further development of foreign industrial fishing, the contribution of this sector to the national economy increased from 7 to 27% of the gross domestic product (GDP) between the 1990s and the late 2000s (FAO/FishCode, 2005; FAO, 2009).

Fisheries in the RMI exploit both coastal, reef-associated stocks and offshore pelagic stocks. Coastal catches are attributed exclusively to the truly domestic small-scale fisheries (Beger et al., 2008), which include both the subsistence sector, i.e., non-commercial catches primarily for home and family consumption; and the artisanal sector, i.e., small-scale catches primarily destined for commercial sale (Zeller et al., 2016). Subsistence fishing is a very common activity in the RMI, particularly on the outer atolls where the availability of jobs and associated cash income for goods and services is more restricted (Beger et al., 2008; FAO, 2009; Gillett, 2016). Artisanal fisheries are important for supplying seafood to the main urban centers, and artisanal operations are more common on the main atolls of Majuro, Kwajalein, and Arno (**Figure 1**). The contribution of coastal fisheries to basic nutrition and food security is substantial in the RMI, with fish caught locally by the small-scale fisheries (both artisanal and subsistence) contributing to an overall consumption rate of around 82 kg per person per year (Gillett, 2016).

The offshore large-scale, industrial fisheries targeting pelagic species are conducted by both a domestic-flagged, locally-based fleet and larger foreign fleets from several countries that fish under different agreements (MIMRA, 2017). These industrial fleets operating in the EEZ of the RMI target mainly tunas, including skipjack (*Katsuwonus pelamis*), albacore (*Thunnus alalunga*), bigeye (*Thunnus obesus*), and yellowfin tuna (*Thunnus albacares*) as well as other large pelagics such as billfishes. These fisheries also targeted pelagic sharks (Bromhead et al., 2012); however, this activity became illegal with the declaration of a shark-fishing ban in 2011 (Cramp et al., 2018).

The domestic industrial fleet started consistent operations relatively recently, as the first report of annual domestic catches of approximately 7,600 t occurred in 2000. Since then, the Marshallese-flagged industrial fleet has grown considerably, increasing from four long-liners and five purse-seiners in 2007 to 31 long-liners and 10 purse-seiners in 2017 (WCPFC, 2018b). However, part of this fleet (long-liners in particular) consists of chartered vessels operated by a foreign-owned (Hong Kong Chinese) joint-venture fishing company based in Majuro (MIMRA, 2017; WCPFC, 2017). True beneficial ownership of these fishing operations, including unknown levels of revenues, may still be largely foreign owned.

The foreign industrial fleets operating in the RMI EEZ include vessels flagged to various Asian countries, the United States, and other Pacific island countries. In 2017, 257 foreign-flagged vessels were licensed to fish in the RMI (MIMRA, 2017). These fleets consist mainly of purse-seine and long-line vessels, although a Japanese pole-and-line fishing fleet also operates in the country. Joint-venture fishing companies (Chinese and Taiwanese) also operate a Marshall Islands-based foreign fleet, with the largest of these ventures operating 24 vessels from China and the Federates States of Micronesia (MIMRA, 2017). The fishing access fees paid by foreign countries to fish in the EEZ waters of many Pacific island countries provide important foreign exchange earnings for the host countries in whose waters they operate. The access fees

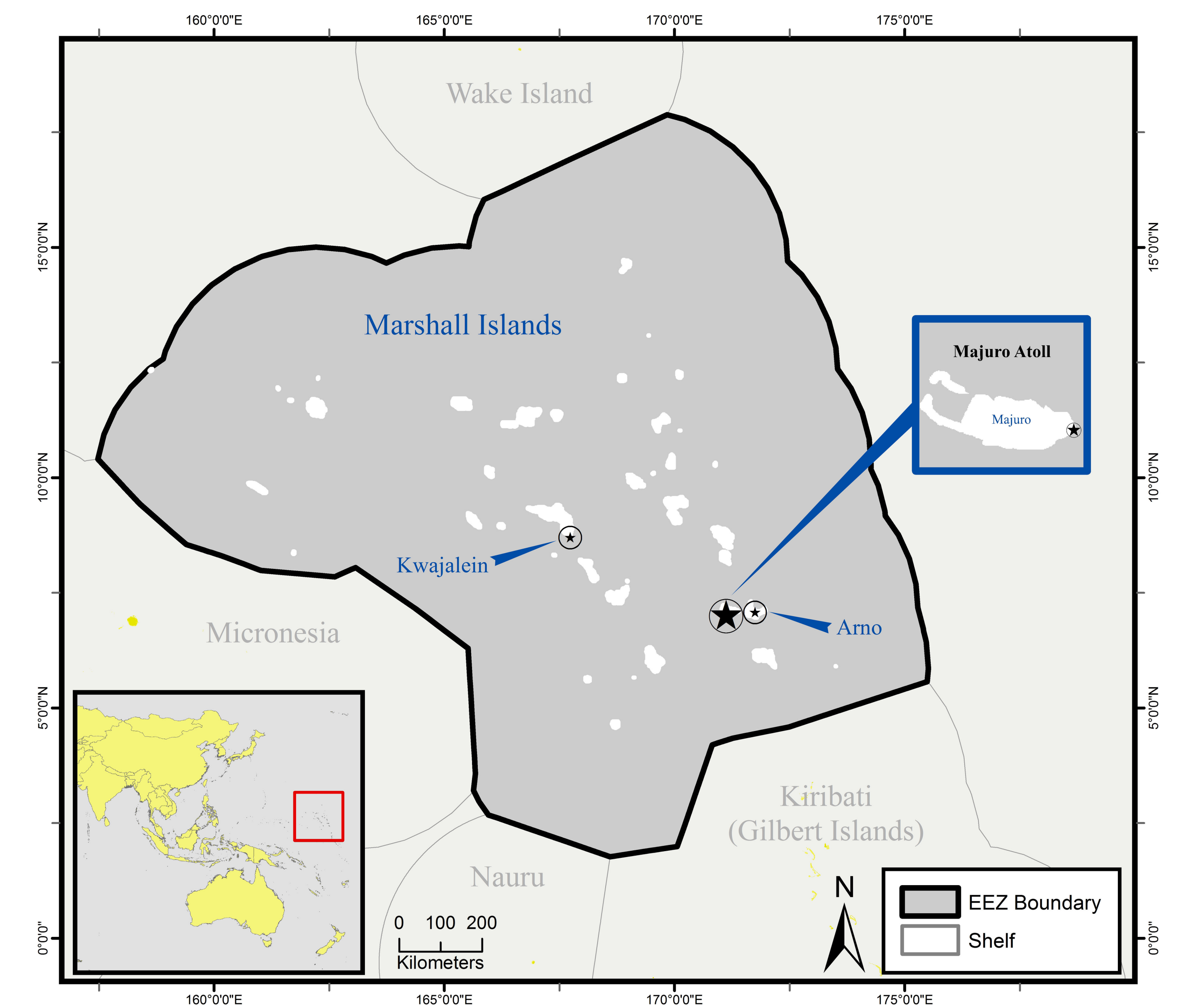

FIGURE 1 | The Exclusive Economic Zone of the Republic of the Marshall Islands, and the most populated atolls.

paid by foreign countries to the RMI generated approximately US$34.1 million in 2017, which were generated through fishing rights, vessel day scheme revenue, license fees, transshipment fees, fishing violation fines, boat chartering fees, and fishery observer fees (MIMRA, 2017).

Fisheries of coral reef species for the global aquarium trade also occur in the RMI, with operations occurring from Majuro and Ebeye. In 2014, exports were estimated to generate approximately US$50,000 (Gillett, 2016); with approximately 103,500 individual fish and 15,200 individual invertebrates exported in 2017 (MIMRA, 2017). The industry exports over 50 different species of fish from a variety of families, including Pomacanthidae, Chaetodontidae, Acanthuridae, Labridae, Serranidae, Pomacentridae, Balistidae, Cirrhitidae, Gobiidae, and Blenniidae (Dalzell et al., 1996), with the most common species being the flame angel, *Centropyge loriculus* (Gillett, 2011). As catches for the aquarium trade are not for human consumption, and constitute relatively small amounts in terms of tonnage, these were not addressed here.

The importance of the fishing sectors for the economic and food security well-being of the RMI is clear. However, the national government faces a challenge balancing both the economic benefits of the offshore industrial tuna fishery and the crucial food security benefits of sustainable use of marine resources for the local population (FAO, 2009). In this context, better understanding the historical patterns and trends over time in total catches by each fishing sector, but especially by the under-represented small-scale sectors, is crucial to enable informed and effective management decisions (Pauly, 1997). Therefore, the objective of our study was to derive a time series of the total catches taken by the domestic fisheries, i.e., large- and small-scale domestic fisheries in the RMI from

1950 to 2017, with particular emphasis on the small-scale sectors. We applied a structured catch reconstruction approach based on the principles described in Zeller et al. (2016), which uses publicly available sources of secondary data and information to complement the officially reported catch statistics, as reported to the global community by the Food and Agriculture Organization (FAO) of the United Nations on behalf of the RMI. We further combined the reconstructed catches with reconstructed estimates of fishing capacity and effort to present time series estimates of catch-per-unit-of-effort (CPUE) by fishing sector for 1950–2017.

MATERIALS AND METHODS

A preliminarily reconstruction of the marine fisheries catches of the Marshall Islands (**Figure 1**) was conducted by Haas et al. (2014) for the 1950–2010 period. Here, we build on the original reconstruction to account for most recent data and new information that has become available and update the time series to the most recent year (data year 2017) of data officially reported by the FAO on behalf of the RMI. We obtained the reported baseline catch data from the FAO[1], which reports fisheries landings on behalf of each country (Garibaldi, 2012). Using secondary data and information from Gillett (2016), and following the reconstruction principles in Zeller et al. (2016), we estimated domestic demand for locally-sourced seafood, and compared this to the non-exported portion of FAO landings, which are deemed to remain in-country for domestic consumption. We considered the difference between these two components to represent the unreported catch of small-scale fisheries in the RMI (see **Table 1** for a summary of the reconstructions).

We classified the catches of pelagic species generally captured by the Marshallese-flagged industrial tuna fisheries and reported by the Western and Central Pacific Fisheries Commission (WCPFC) as the reported component of the domestic industrial fisheries, consisting mostly of tunas, billfishes, and pelagic sharks. Catches taken by foreign fleets fishing in the RMI EEZ were not included in the domestic catches presented here, but were addressed globally in Coulter et al. (2020).

Small-Scale Sectors

Small-scale catches were attributed to one of two sectors: artisanal or subsistence, although we recognize that there is often overlap between these two sectors (Zeller et al., 2016). Within the *Sea Around Us* global databases, small-scale fisheries are spatially limited to the *Inshore Fishing Area*, which is defined as the waters within 50 km from inhabited shores or waters up to 200 m depth, whichever comes first (Chuenpagdee et al., 2006; Chuenpagdee and Pauly, 2008; Zeller et al., 2016). Artisanal fisheries catches are defined as primarily for commercial sale, while subsistence fisheries have self- and family-consumption or local barter as primary purpose, rather than commercial sale (Zeller et al., 2007, 2016; Gillett, 2011).

[1] www.fao.org/fishery/statistics/collections/en

TABLE 1 | Anchor points and sources used to estimate time series of catches by sector in the Republic of Marshall Islands in 1950–2017.

Sector	Year	Catch (t)	Source
Artisanal	1950	0	Assumption
	1946–2008	–	Linear interpolation
	2009	1,230	Derived from Gillett (2016)
	2010–2013	–	Linear interpolation
	2014	1,500	Derived from Gillett (2016)
	2015–2017	–	Linear extrapolation
Subsistence*	1950	1,017	Assumption
	1951–2008	–	Linear interpolation
	2009	3,275	Derived from Gillett (2016)
	2010–2013		Linear interpolation
	2014	3,000	Derived from Gillett (2016)
	2015–2017	–	Linear extrapolation
Recreational**	1958	0	Assumption
	1959–1987	–	Linear interpolation
	1988	4.9	Anon (1988)
	1989–1997	–	Linear interpolation
	1998	6	Whitelaw (2003)
	1999–2017	–	Linear extrapolation
Industrial***	1950–2017	–	FAO catch data

**Subsistence catches estimated based on per capita population fish consumption (see methods). **Recreational catches estimated based on the reported number of tourists visiting the Republic of Marshall Islands (see methods). ***The retained bycatch was estimated and added to account for the unreported component of the catch (see methods).*

Artisanal Catches

As core data anchor points (Zeller et al., 2016) for artisanal catches, we used estimates of the artisanal catch component as a proportion of the total coastal catch, which were available for 2007 (Gillett, 2009) and 2014 (Gillett, 2016). To derive the artisanal catch component for each year between 2007 (25%) and 2014 (33%), we interpolated the proportion of artisanal to total coastal catch between these anchor points. To estimate the total volume of artisanal catch in each year, we applied the derived artisanal proportions to the independent estimates of total coastal catch of 4,510 t in 2009 and 4,500 t in 2014 (Gillett, 2016). We used these estimated values to interpolate the artisanal catch values for the years between 2009 (1,230 t) and 2014 (1,500 t). We assumed that artisanal fishing (i.e., small-scale commercial) began, or re-started after WWII with an assumed 0 metric tons of artisanal catch in 1945. We then interpolated the artisanal catch values linearly between 0 metric tons in 1945 and the artisanal catch anchor point in 2009. For 2015–2017, we linearly extrapolated the 2009–2014 time series of artisanal catch to 2017. We recognize that this increases the uncertainty around these estimates, and these data points will require revision once future data and information becomes available.

Prior to 2004, we considered the unreported component of the reconstructed artisanal catch to be the difference between the total reconstructed artisanal catch (as estimated above) and the official catches reported to FAO as the non-specific category "marine fishes nei." After 2004, tonnages for "marine fishes nei" reported to FAO outweighed the total reconstructed artisanal catches, thus for years after 2004, all reconstructed artisanal catches were deemed reported. Surplus "marine fishes nei" data reported to FAO after 2004 were accounted for under the reconstructed subsistence fishing catches (see below).

Separately to the reconstruction of general artisanal catches, we also reconstructed catches of trochus (sea snail) and sea cucumber, as these taxa were not typically consumed locally and were targeted for export (Gillett, 2016). Data reported to FAO by the RMI list trochus catches between 1987 and 2004. We thus conservatively assumed that the trochus target fishery started in 1987, and for 1987–2004, we considered total trochus catch to be equal to the amount reported to the FAO. For the 2005–2017 years, when trochus was no longer reported to FAO but the fishery was known to continue (Gillett, 2016), we approximated the catches by interpolating between the value of 0.25 t reported to the FAO in 2004 and the estimate of 9 t reported for 2014 in Gillett (2016). For 2015–2017, we held the 2014 amount constant.

The sea cucumber fishery is also an export fishery that developed in the RMI in response to demand by Asian markets in the 1990s. In order to reconstruct total sea cucumber catches, we used export data for dried, processed sea cucumber exports reported by the FAO commodity trade and production data (FAO, 2019), which present data for 1996, 1997, 2003, 2007, and 2011–2016. A national report suggested that commercial fishing of sea cucumber also occurred between 2002 and 2005 (MIMRA, 2014). Based on this information, we assumed sea cucumber fisheries to have started in 1996, when exports were first reported, but stop after 1997 until 2002, in 2003 exports restarted and continued until 2017. Based on the time series, we converted the exported amount of frozen, dried, and salted sea cucumbers (FAO, 2019) to wet weight on the basis of an average of 90% weight lost in processing (MIMRA, 2014).

Subsistence Catches

We obtained subsistence catch anchor points of 3,275 t for 2009 and of 3,000 t for 2014 from Gillett (2016), by subtracting the artisanal catch estimates from the total coastal fisheries catch estimates for these years. To derive a complete time series of subsistence catch estimates, we converted these point estimates of subsistence catch into *per capita* subsistence catch rates for 2009 (62.6 kg per person) and 2014 (56.7 kg per person) by using the total population data for the RMI in these years (World Bank 2019[2]). These *per capita* subsistence catch rates were then interpolated between 2009 and 2014, and extrapolated linearly to 2017, using the 2009–2014 time series trend. The increase in access to alternative food sources in urban centers of Pacific island countries in recent decades has been responsible for a progressive decrease in use of locally-sourced seafood (Thow et al., 2011). Due to the absence of information for earlier periods, we conservatively assumed an annual *per capita* subsistence catch rate of 78.2 kg per person for 1950, i.e., 25% higher than in 2009. Thus, we assumed that the RMI population relied more on subsistence fishing in the earlier decades than in more recent years. We then interpolated the catch rate linearly between 1950 and 2009. To derive the total subsistence catch, we multiplied the derived annual *per capita* subsistence catch rates by the corresponding human population of the RMI for 1950–2017. Population data were obtained from World Bank data for 2011–2017 and from census information by the RMI Office of Planning and Statistics[3] from 1950 and from 1960–2010. We interpolated the population between 1950 and 1960 to complete the time series.

Given the nature of the data reported by the RMI to the FAO, for 1950–2004, all subsistence catch was deemed unreported, as the reconstructed artisanal catch (see section "Artisanal Catches") outweighed the reported catch of coastal taxa for these early years. Furthermore, data collection and reporting in the earlier years was more likely to only address commercial fisheries. After 2004, the reported component of subsistence catch was assumed to equal the remaining FAO catch data for "marine fishes nei" attributed to the small-scale sector, after the reported artisanal catch was accounted for (see section "Artisanal Catches"). Unreported subsistence catch was assumed to be the difference between the assumed reported subsistence catch and the total estimated subsistence catch for 2004–2017.

Recreational Catches

Recreational fishing in the RMI is an activity conducted mainly by international tourists. To reconstruct catches for this sector, we assumed recreational fishing to be proportional to the number of tourists visiting the country annually. Data on annual visitors were obtained from the Economic Policy, Planning, and Statistic Office of the RMI[4]. We assumed recreational tourism fishing to have only started with the phasing out and subsequent ceasing of nuclear testing by the United States in the RMI in 1958. Thus, we interpolated from an assumed zero fishing-tourists in 1958 to the first year of the reported total number of tourists time series (1995–2017[4]) to obtain a complete time series estimate for tourists. We used the recreational catch estimates of 4.9 metric tons in 1988 and 6 metric tons in 1998 (Anon, 1988; Whitelaw, 2003) as catch data anchor points and derived recreational catch rates per tourist for 1988 and 1998. We interpolated the derived recreational catch rates for the years between 1988 and 1998, and held the rates constant before 1988 and after 1998, before applying them to the time series of tourists for 1958–2017. While our anchor points report demersal fish catches to some extent, these studies were mainly focused on pelagic fish (Anon, 1988; Whitelaw, 2003); therefore, our estimates of recreational catches are likely conservative and underestimate recreational catches of demersal species.

Taxonomic Composition

We used information from Dalzell et al. (1996) and Gillett (2011) to derive a taxonomic composition of the estimated combined (artisanal plus subsistence) small-scale catches. This breakdown was applied to the reported catches labeled as "marine fishes nei" in the FAO data, and to the unreported subsistence and artisanal catches estimated here.

[2] https://data.worldbank.org/country/marshall-islands?view=chart

[3] https://rmi.prism.spc.int/index.php

[4] https://rmi.prism.spc.int/

Recent improvements in the taxonomic classification of invertebrates in the data reported by the FAO on behalf of the RMI were taken into account in the reconstruction, which allowed for a more accurate taxonomic breakdown of the catches of trochus (Trochidae), spiny lobsters (Palinuridae), and mud crabs (Portunidae), reported as artisanal catches. The catches of sea cucumbers were disaggregated to species level based on the taxonomic breakdown described in MIMRA (2017).

Large-Scale Sector

The industrial fleet in the RMI targets mainly large tuna species; however, other large pelagic species such as sharks and billfishes are also captured, either as incidental bycatch or targeted catch (Bromhead et al., 2012). Often, because billfishes are valuable and exportable, they are thought to be reported in the data supplied to the FAO, while many other species are not. We attributed all reported catches of tunas and other large pelagic species such as billfishes and pelagic sharks to the large-scale sector within the RMI EEZ by flag state of the fishing entity. To estimate the amount of unreported retained bycatch taken by the domestic industrial fleets in the RMI, we combined the total catches of the four major tuna species, skipjack (*K. pelamis*), albacore (*T. alalunga*), bigeye (*T. obesus*), and yellowfin tuna (*T. albacares*) reported by the Marshallese-flagged fleet and allocated the total by gear type, maintaining the percentages of the catch composition from the total catch. Based on information in Gillett (2011), we attributed 75% of the reported catch of the four species combined to the purse-seine fleet and 25% to the long-line fleet. Using anchor points for under-reporting by gear type from Gillett (2009), we increased all purse-seine catches by 5%, and all long-line catches by 30%, and treated these catches as unreported, retained bycatch.

The taxonomic composition of the estimated unreported bycatch was derived using data in MIMRA (2009), which details the non-target species caught in the locally-based offshore fleet (**Table 2**). The top 15 non-target species by weight for the year 2008 were identified by the percentage of the non-tuna catch (MIMRA, 2009), and these percentages were applied to the unreported catch time series. We recognize that this does not take into account potential inter-annual variability in composition or consistent trends over time of unreported catches, or the contributions of minor taxa to unreported catches.

Because the catches of foreign fleets are either landed in foreign ports (mainly in Asia or the United States) or are transshipped in Majuro (FAO, 2009) for direct export shipping, these are assumed to be accounted for in the FAO fisheries landing data reported by these foreign flag countries, and are therefore not included in this study. However, a recent study reported substantial under-reporting of industrial tuna fisheries due to transshipment activities in the Western and Central Pacific (Anon, 2019), which highlights the need for a comprehensive reconstruction of unreported catches of global tuna fisheries (Coulter et al., 2020).

Discards

The large-scale tuna fisheries in the Pacific Ocean were considered to have an overall average discard rate of 10.8% for 1950–2010 (Schiller, 2014; Coulter et al., 2020). We estimated the total volume of catches that were likely discarded by the domestic industrial fisheries by applying this discard rate to the reconstructed landed catch by this sector. We reconstructed the taxonomic composition of discards by applying the taxonomic composition of discards by the entire tuna fishery in the Pacific region to our estimates (Schiller, 2014). We recognize that this does not take into account inter-annual variability or time series trends in the discarding behaviors or patterns. Particularly for sharks, our estimates of discards represent a conservative estimate, as the implementation of the prohibition of retention of sharks in 2011 may have influenced the discard patterns.

TABLE 2 | Non-tuna catch composition of pelagic species in the large-scale sector of the Republic of the Marshall Islands. Based on MIMRA (2009).

Common name	Taxon name	% Non-tuna catch
Blue marlin	*Makaira mazara*	7.9
Black marlin	*Istiompax indica*	0.3
Striped marlin	*Kajikia audax*	4.2
Swordfish	*Xiphias gladius*	1.0
Other billfishes	Istiophoridae	1.3
Blue shark	*Prionace glauca*	10.5
Mako shark	*Isurus oxyrinchus*	1.1
Oceanic whitetip shark	*Carcharhinus longimanus*	5.0
Silky shark	*Carcharhinus falciformis*	21.9
Other sharks/rays	Elasmobranchii	36.9
Rainbow runner	*Elegatis bipinnulata*	1.7
Wahoo	*Acanthocybium solandri*	3.5
Common dolphinfish	*Coryphaena hippurus*	2.7
Triggerfishes	Balistidae	0.3
Opah	*Lampris guttatus*	1.7

Percent composition is with reference to non-target industrial catch.

Fishing Effort and CPUE

For RMI fishing effort, we used fishing effort data from the global database of reconstructed fishing effort developed by the *Sea Around Us* and available online[5]. The effort data presented in this database were reconstructed according to the methodology described in Greer (2014). Briefly, this methodology utilizes the number and average engine power of vessels ($kW \cdot boat^{-1}$) and estimates of the number of days fished by each fleet/gear sector to estimate fishing effort/capacity by each fishing sector, as a measure of energy spent fishing in a determined period of time (kWdays). Effort estimates for small-scale fisheries accounted for the fact that some fisheries may not utilize vessels, i.e., fishing from shore and reef gleaning. These estimates also accounted for the increase in use of motorized vessels, reported to occur from the 1980s onward (Chapman, 2004). The increase in vessel motorization in the small-scale fisheries in each year was estimated based on country data on the proportion of motorized small-scale vessels in key regions of the country in 2007 (Pinca et al., 2009). These proportions were interpolated between 1980, when motorization of vessels started, and the 2007 anchor point. We

[5] www.seaaroundus.org/data/#/fishing-entity-effort/132

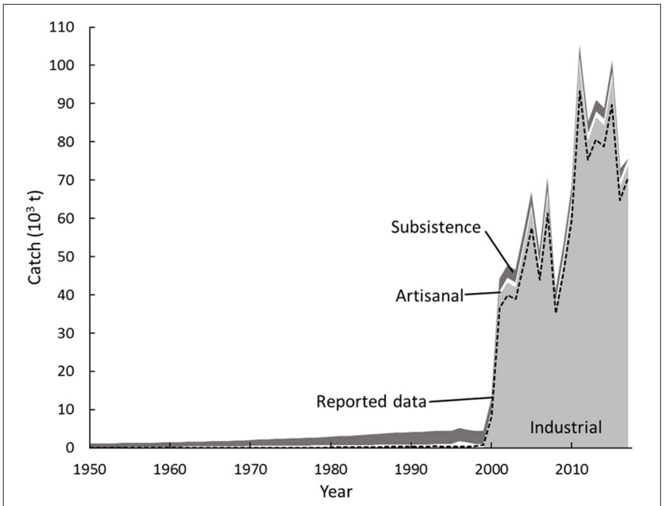

FIGURE 2 | Total reconstructed catch by fishing sector for the Republic of the Marshall Islands (RMI) for the time period 1950–2017, with data reported by FAO on behalf of the RMI overlaid as a dashed line.

then linearly extrapolated the time series to 2017. Information and data on the fishing fleets, as well as changes over time were sourced from the scientific and gray literature (Greer, 2014) and were used as anchor points to reconstruct a time series of likely effort for each fishing sector in the RMI between 1950 and 2017.

We derived a time series of CPUE by combining our reconstructed catches with the reconstructed effort data by fishing sector over the entire time period of the study.

RESULTS

Total reconstructed catches for the domestic fisheries of the RMI were estimated at slightly over 1.3 million t between 1950 and 2017, which is 27% higher than the 1.037 million t reported by the FAO on behalf of the RMI for the same time period (**Figure 2**). Overall, the domestic industrial sector targeting large pelagic species, which only became important after 2000 (**Figure 2**), dominated total catches and accounted for over 84% of total reconstructed catches, while small-scale sectors accounted for around 16% (**Figure 2**). Following the beginning of domestic industrial fishing operations in the early 2000s, the catches in this fishery initially increased steeply, followed by a period of catches around 49,000 t·year^{-1} between 2001 and 2009. After 2010, catches increased to around 82,000 t·year^{-1} (**Figure 2**). Overall, these fisheries discarded a total of 119,000 t between 2000 and 2017, with an average of around 6,600 t of fish discarded per year (**Figure 3A**).

The domestic industrial fishing effort began in the early 1990s with intermittent attempts to establish a long-line domestic fishery. However, consistent fishing effort started in 2000 reaching about 2.9 million kWdays in 2005 (**Figure 3B**). After 2006, fishing effort increased at a rate of 13% year^{-1}, peaking in 2011 with a total effort estimated at just over 5.4 million kWdays,

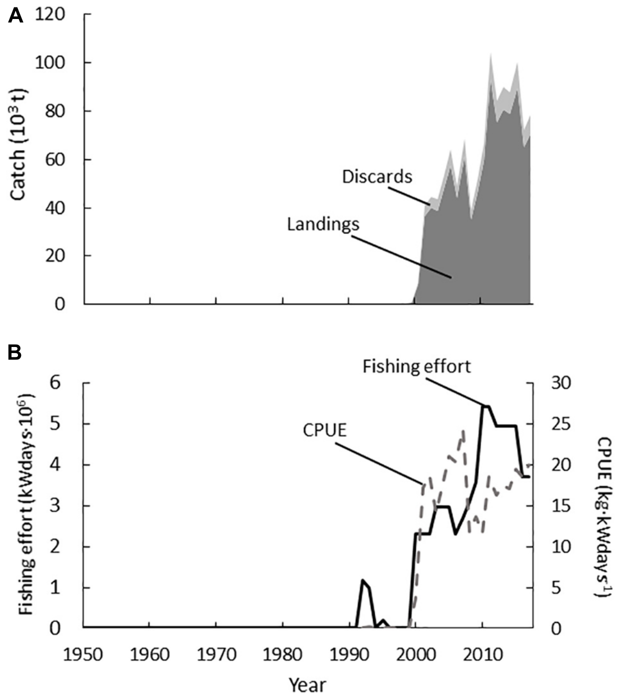

FIGURE 3 | Reconstructed catch and effort for the industrial fishing sector for the Republic of the Marshall Islands for the time period 1950–2017. **(A)** Catch separated by landed catch (landings) and discarded catch. **(B)** Fishing effort and catch-per-unit-of-effort (CPUE).

before dropping slightly to 3.7 million kWdays by 2017. Due to the patterns of catch and fishing effort over time, the mean CPUE of the domestic industrial fishery from 2001 to 2017 was 17.5 kg·kWdays^{-1} (**Figure 3B**).

Due to the predominance of the export-oriented industrial fisheries for large pelagics, fishes from the family Scombridae comprised the largest portion (77%) of the total reconstructed catch (**Figure 4A**), with skipjack tuna (*K. pelamis*) accounting for 57% of the total. Sharks also represented a large component of the industrial catch, with a reconstructed total of just under 83,000 t over the entire time period, and around 6,000 t·year^{-1} in recent years. Silky (*Carcharhinus falciformis*) and blue sharks (*Prionacea glauca*) were the two species that contributed the most to catches of sharks, accounting for approximately 29 and 14% of the volume of sharks caught, equaling around 1,740 and 840 t·year^{-1} in recent years, respectively. These catches were largely due to bycatch in the domestic industrial fishery. Major discards consisted of both target and non-target species, including blue sharks (37%), skipjack tuna (19%), silky sharks (5%), and yellowfin tuna (4%), while other fish and other sharks accounted for 36 and 5%, respectively.

When considering the small-scale sectors only, the reconstruction suggested that actual total small-scale catches were around 3.9 times higher than the assumed small-scale catches reported by the FAO on behalf of the RMI (**Figure 5**). The small-scale artisanal and subsistence sectors accounted for 4 and 12% of the total reconstructed catches, respectively

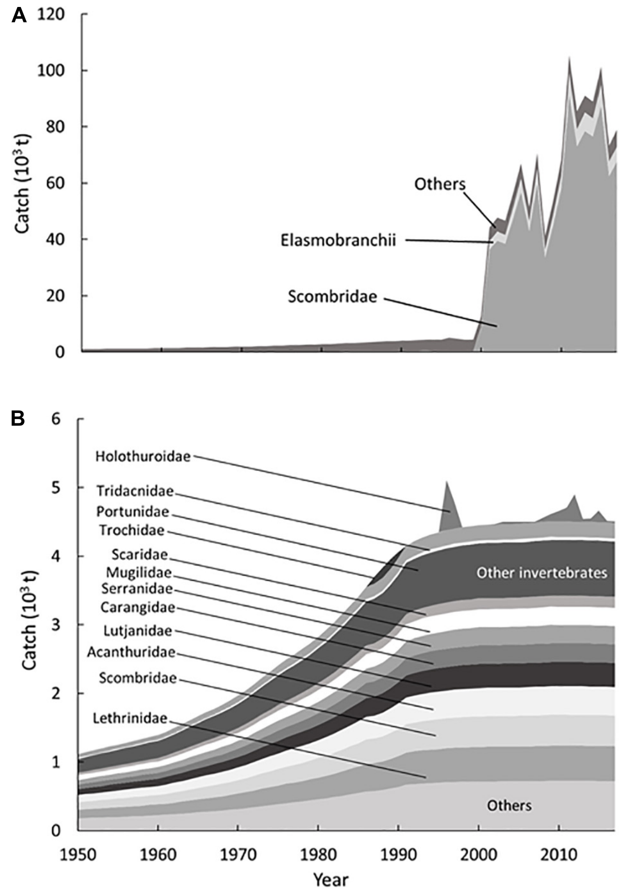

FIGURE 4 | Taxonomic composition of: **(A)** the total reconstructed catches for the Republic of the Marshall Islands for the time period 1950–2017 by major taxonomic categories ("Others" represents 54 additional taxonomic categories, largely from the small-scale sectors) and **(B)** the small-scale sectors only (artisanal, subsistence and recreational). "Others" comprise 18 additional taxa (including 14 families) with minor contributions to the catch.

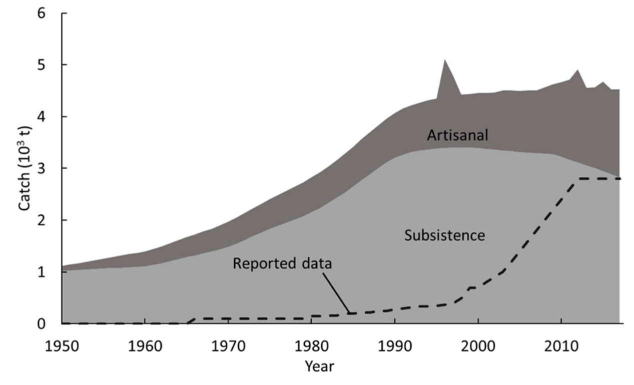

FIGURE 5 | Reconstructed catches for the small-scale sectors of the Republic of the Marshall Islands (RMI) for the time period 1950–2017. Data reported by the FAO on behalf of RMI, overlaid as a dashed line, represent assumed reported landings excluding the reported large pelagic tuna, billfish, and shark catches, which were targeted specifically by the large-scale industrial sector. Due to the low values, recreational catches are not visible in the graph.

(**Figure 5**). Looking at the small-scale sector individually, the artisanal and subsistence sectors accounted for 26 and 74% of the catches over the entire period, respectively. Estimated unreported artisanal catches accounted for over 45% of artisanal catches, while unreported subsistence catches accounted for substantially more, at around 92% of subsistence catches. Catches by the recreational fishing sector accounted for at least 233 t (0.1% of small-scale catches) over the time period, with an average catch of approximately 4 metric tons per year. Our reconstructed recreational catch estimate is likely conservative, particularly for demersal reef species, as catches of this group were likely underrepresented by the anchor points available.

Overall, small-scale catches increased steadily from around 1,100 t·year^{-1} in 1950 to the early 1990s, after which catches seemed to level off at around 4,500 t·year^{-1} (**Figure 5**). A few smaller peaks of up to 5,100 t·year^{-1} occurred in 1996, 2011, and 2014 due to the artisanal boom-and-bust sea cucumber (Holothuroidea) fisheries (**Figures 4B**, **5**). Artisanal sector catches increased steadily over time, reaching approximately 1,700 t·year^{-1} in 2017. Catches of the subsistence sector increased from 1,000 t·year^{-1} in 1950 to 3,200 t·year^{-1} in 1990, and remained relatively constant at just over 3,300 t·year^{-1} between 1990 and 2008. From 2009 onward, reconstructed catches by this sector declined by 2% per year, on average. This apparent decline by the subsistence sector was compensated by an average increase of 3% per year in catches by the artisanal fishing sector (**Figure 5**).

Catches of the small-scale sector consisted of approximately 75% finfishes and 25% invertebrates (**Figure 4B**). The predominant fish species in the small-scale catches were the humpback red snapper (*Lutjanus gibbus*) and humpnose big-eye bream (*Monotaxis grandoculis*), each accounting for around 5% of small-scale catches in recent years. Skipjack (*K. pelamis*) and yellowfin tuna (*T. albacares*) were also important components of the small-scale fisheries, and combined accounted for around 8% of the catches over the last 10 years. For invertebrates, elongate giant clam (*Tridacna maxima*), bear paw clam (*Hippopus hippopus*), and scaly or flute giant clam (*Tridacna squamosa*) together accounted for around 35% of the invertebrate catch (**Figure 4B**).

The overall fishing effort of the small-scale sectors was relatively low until the end of the 1970s, at around 14,300 kWdays. Due to the introduction of motorization and growth in the number of boats (**Figure 6A**), small-scale fishing effort increased from just under 31,000 kWdays in the early 1980s to around 490,000 kWdays in 2017 (**Figure 6B**). Throughout this time period, the fishing effort per vessel increased almost three times more for the artisanal sector (14–196 kWdays·vessel^{-1}) when compared to the subsistence sector (7–42 kWdays·vessel^{-1}, **Figure 6A**). Due to this pattern of fishing effort over time, the CPUE of the small-scale sector remained constant at around 122 kg·kWdays^{-1} between 1950 and 1980. This period was followed by a sharp decline of eightfold in the CPUE, with stabilization of the mean CPUE around 15 kg·kWdays^{-1}

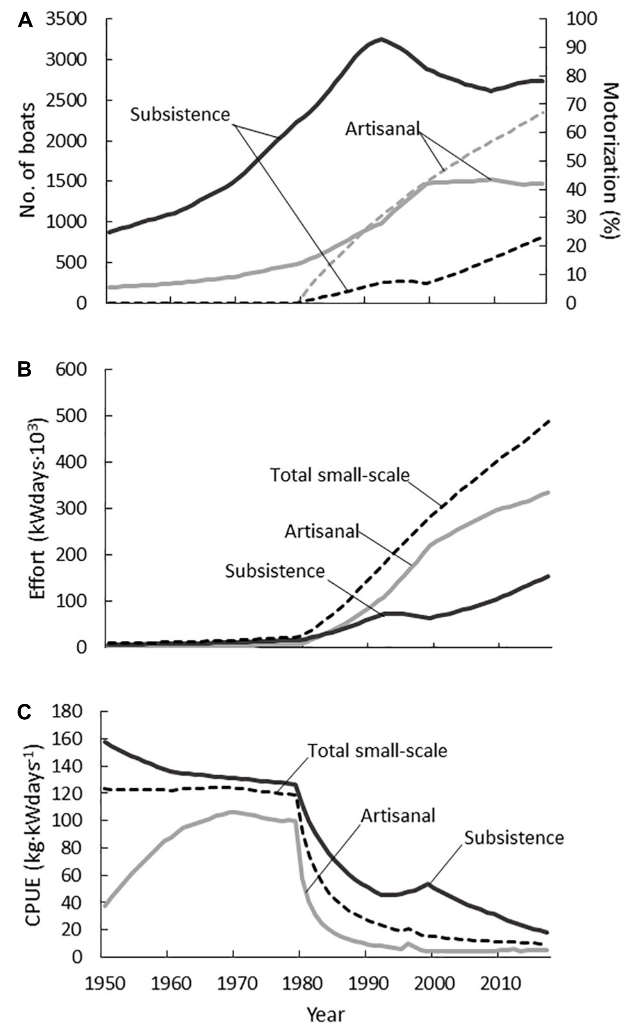

FIGURE 6 | Fishing effort for the small-scale fisheries sectors in the Republic of the Marshall Islands for 1950–2017 as reconstructed based on Greer (2014). **(A)** Number of boats (solid lines) and percentage of motorized boats (dashed lines). **(B)** Fishing effort. **(C)** Catch-per-unit-of-effort (CPUE).

from the 1990s to 2017 (**Figure 6C**). In this period, the mean CPUE of artisanal and subsistence fishing were around 6 and 38 kg·kWdays^{-1}, respectively.

DISCUSSION

The reconstructed total domestic catches for the RMI were estimated to be 27% higher than the landings reported by the FAO on behalf of the country. However, analysis of the reconstructed catches of the small-scale coastal fisheries sectors alone revealed that reconstructed catches for these sectors were 3.9 times the data reported by FAO. The differences were largely due to the under-representation of small-scale artisanal and particularly small-scale subsistence sectors in the data collection, estimation, and reporting system, which is a common issue for small-scale fisheries in many countries (Zeller et al., 2015). To address this data gap for the RMI, our reconstruction focused on improving the estimates of total domestic small-scale catches back to 1950, with a particular focus on separating artisanal and subsistence fisheries, as these are the most important sectors for the food security of the local population (Charlton et al., 2016).

Subsistence fisheries accounted for 74% of the small-scale catches since the 1950s, highlighting the exceptional food security importance of this sector for the local population (FAO, 2015; Pauly and Charles, 2015; Zeller et al., 2015). Despite representing the largest percentage of total small-scale catches, the subsistence fishery appears to be the most underrepresented sector in the official statistics, with unreported subsistence catches seemingly accounting for 68% of total small-scale catches. The small-scale fisheries sectors are the major source of fresh seafood for the population of the RMI, and are considered by the government as strategic for the development of the domestic economy, particularly outside the main population centers (Gillett, 2010). While this recognition is important, the magnitude of the unreported catches revealed here suggests that the actual degree of socio-economic, and especially food security, importance of these two small-scale sectors to the country likely continues to be under-estimated. This highlights the necessity of ongoing improvements in the monitoring and estimation of data for small-scale fisheries in the RMI, which is also the case elsewhere throughout the Pacific (Zeller et al., 2015).

Trends in small-scale fisheries in the RMI have changed over time, with a progressive increase in catches from 1950 to the early 1990s, followed by stabilization of total small-scale catches at around 4,500 t·year^{-1} (**Figure 5**). This can be partially explained by a stabilization in the domestic demand for fish as a consequence of a gradual reduction in the *per capita* consumption of fresh seafood, which itself was partially offset by the positive human population growth. Several studies have reported shifts in the diet of the population of many Pacific island countries, including the RMI, with reduction of consumption of local products such as fresh fish, and replacement with imported industrialized food such as canned meats (Cortes et al., 2001; Gittelsohn et al., 2003; Ahlgren et al., 2014; Charlton et al., 2016). This shift has been more pronounced in the main population centers and has been associated with the increase in nutritional health issues and chronic diseases in the RMI (Ahlgren et al., 2014).

Despite the relative stability of small-scale fisheries catches over the last two decades, trends for the specific sectors differ. Since the late 2000s, declining non-commercial subsistence catches have been partially compensated by a steady increase in catches by the commercially focused artisanal sector. In the 1990s, the government established a program to purchase fish from the outer atolls for sale in the major urban centers (Gillett, 2016; MIMRA, 2017). This increased commercialization of part of the small-scale catches from the outer atolls could explain the observed sectoral trend differences. This substitution suggests a gradual increase in the importance of a cash-based economy among small-scale fishers and increased commercialization of seafood otherwise caught for subsistence purposes. Despite accounting for a smaller proportion of the total small-scale catches since 2010, the subsistence sector still accounts for approximately 65% of total small-scale catches in the RMI. This contribution is comparable to other small-island countries and territories in the Pacific, where in 2010 the subsistence sector accounted for 69%, on average, to total domestic catches when export-oriented industrial tuna fisheries were not considered (Zeller et al., 2015). This contribution is likely to be much higher in the outer atolls of the RMI, where most households are involved in subsistence fisheries and the supply of imported food is more limited than in the urban centers (FAO, 2009).

The effective fishing effort (fishing power or capacity) for the small-scale sectors increased 13-fold between 1980 and 2010, mainly due to the wide-spread introduction of motorization and new boats in the fisheries (Chapman, 2004). The effect of motorization was more evident in the artisanal fleet, which despite having a smaller number of boats when compared to the subsistence sector, was the main driver of the increase in fishing effort by the small-scale fisheries in the RMI (**Figure 6**). A similar pattern of sharp increase in fishing effort by artisanal fisheries due to the introduction of motorized vessels is also evident in other developing countries, including small-island countries in the region (e.g., White et al., 2018).

Our reconstruction suggests that the increase in fishing capacity resulted in a strong decline in CPUE of the small-scale fisheries from 120 kg·kWdays^{-1} in 1980 to 15 kg·kWdays^{-1} by the 2010s, before stabilizing. This decline in CPUE, driven by the increased fishing power of motorized vessels, suggests that the underlying stocks may have experienced a substantial decline in abundance. The CPUE of the subsistence sector remained consistently higher when compared to the artisanal sector through the entire time series; however, there was a decreasing trend in the subsistence CPUE since 1990. This may suggest localized overexploitation of near-shore habitats as these represent the main fishing grounds, which may be easily accessed by subsistence boats with lower levels of motorization. Comparatively, the mean CPUE of the subsistence sector of the RMI in the 2010s was approximately 20 times higher than in countries in West Africa, where fish stocks have been heavily over-exploited by industrial and artisanal fisheries (Belhabib et al., 2018). Within the Pacific, the subsistence CPUE in the RMI was approximately 12 times higher when compared to Tokelau (White et al., 2018), suggesting that the catches of this sector may still be proportionally high for the region.

Projections of expected national demand for fish protein suggest that the RMI has the potential to supply its own demand with enough local fish to assure good nutrition of the population (Bell et al., 2009). Similar projections have been made globally, but require careful control and restriction of export-oriented distant water fishing fleets that do not contribute to domestic food security in developing countries (Hicks et al., 2019; Pauly, 2019).

The predicted impacts of climate change on small-scale fisheries, including loss in productivity and resilience, are expected to be particularly accentuated in tropical areas (Cheung et al., 2009, 2010; Pauly and Cheung, 2018), and especially so in small-island countries such as the RMI (Hanich et al., 2018). Effective and careful management of the coastal fisheries and protection of habitats to enhance resilience of stocks may partially alleviate or delay the detrimental impacts of climate change (Burden and Fujita, 2019; Rogers et al., 2019). As such, monitoring and careful estimation of comprehensive catch and effort data for each fishing sector in the RMI is crucial to provide appropriate data baselines to monitor and evaluate changes in stocks. Specifically, collection and estimation of comprehensive and detailed catch data at species level over time will allow stock assessments to be undertaken, given recently developed, continuously improved, and freely available data-limited assessment methods (Froese et al., 2017; Palomares and Froese, 2017; Palomares et al., 2018).

The domestic, locally based industrial fishing sector was responsible for the largest portion of total fisheries catches reported by FAO on behalf of the RMI. These catches consist mainly of tuna and other large pelagic species, and account for the majority of the country's seafood exports (Gillett, 2016). Catches of skipjack tuna, which account for most of the industrial catches in the Marshall Islands, are highly affected by inter-annual variation of environmental variables, and result in oscillation of the annual domestic industrial catches (Gillett, 2011). However, the official contribution of fisheries to the national GDP has increased in the last few years, mainly as a consequence of increased revenues from the fishing access fees paid by foreign-flagged industrial vessels, but also due to an increase in fish export earnings from the RMI-flagged industrial fisheries (Gillett, 2016; MIMRA, 2017). Questions around the true and full socio-economic benefits to RMI of re-flagging and locally based joint-venture operations with majority foreign beneficial ownership, versus foreign access fees remain to be examined in detail.

In 2011, the RMI declared a complete ban on commercial shark fishing, retention, and trade of sharks and shark parts within its EEZ, stipulating that sharks caught by industrial fleets, whether foreign or domestic, have to be released (Ward-Paige, 2017). Our reconstructed estimates showed that the amount of sharks caught annually before the implementation of the shark fishing ban was noteworthy, with around 4,800 t in 2010. The official data reported by FAO on behalf of the RMI did not document cartilaginous fishes (sharks, skates, and rays) prior to 2009, after which a specific category for this taxon was included. However, frozen sharks and shark fins were documented frequently in national catch (MIMRA, 2009) and export data for the RMI (Gillett and Lightfoot, 2002; Muller, 2006; Gillett, 2009). Since the introduction of the reporting category in the FAO data after 2009, small but regular shark catches have been reported on behalf of the RMI, with the highest landings reported to FAO in 2017 (231 t), clearly indicating that despite the official shark fishing ban, landings of sharks by the industrial fisheries still occur to some extent.

The effectiveness of the fishing ban regulation in reducing shark mortality is largely unknown. However, reported mortality rates of oceanic whitetip (*Carcharhinus longimanus*, IUCN status: Vulnerable) and silky sharks (*Carcharhinus falciformis*, IUCN status: Vulnerable) caught by the industrial fisheries in the RMI in 2017 and 2018 were 46–60 and 71–93%, respectively (WCPFC, 2017, 2018a). Post-release mortality of sharks in offshore fisheries is known to be high (Campana et al., 2009), further contributing to a high mortality rate of sharks interacting with these fisheries. Moreover, evidence from satellite telemetry suggests that illegal fishing is also inflicting high mortality rates on reef shark species, i.e., gray reef sharks (*Carcharhinus amblyrhynchos*, IUCN status: Near Threatened), in coastal waters with transshipment at sea and landing at distant ports (Bradley et al., 2019). Therefore, it is highly likely that, despite the shark fishing ban, the industrial sector is still inflicting considerable direct and indirect fishing mortality on the shark populations across different habitats of the RMI.

In 2017, an amendment to the shark fishing ban weakened the initial regulation, allowing possession of sharks and shark parts obtained by the industrial fisheries outside RMI waters (Republic of the Marshall Islands, 2017; Cramp et al., 2018). The RMI has an observer program for monitoring of the large-scale commercial fleet, which largely focuses on the monitoring of purse-seiners (MIMRA, 2017). This program has been progressively improved over the years (MIMRA, 2017; WCPFC, 2017), with increasing monitoring of the long-line fleet; however, onboard coverage of this fleet remains limited (WCPFC, 2018a). The weakening of legislation, especially without having a full observer program in place for long-line fleets, which impose the largest mortality on sharks (Bromhead et al., 2012), effectively minimizes any limited control of the origin of the sharks and parts retained by this fleet, potentially stimulating illegal catch of sharks within RMI waters. Thus, it is important that the observer program expands to cover all fishing vessels and all fishing trips within the RMI EEZ. Such complete or near-complete observer coverage and associated, independent accountability, transparency, and traceability needs to be treated as a normal cost of doing business by industrial fleets (Zeller et al., 2011).

We reconstructed the total domestic catch of the RMI by complementing the officially reported baseline catch data as presented by the FAO on behalf of the RMI with conservative estimates of unreported catch components from secondary data and information sources (Zeller et al., 2016). Thus, the uncertainty of the reconstructed data is directly related to the underlying secondary data sources and associated assumptions. Despite the likely higher uncertainty around the reconstructed catch estimates compared to the undocumented and unreported uncertainty around the official catch records (Pauly and Zeller, 2016, 2017), the catch data presented here provide for a more comprehensive account of the likely total domestic catch history in the RMI. Given the crucial food security importance of the small-scale fisheries to the country, our study highlights the need for improved monitoring and estimation of the small-scale fisheries, which would allow for more representative data reporting nationally and internationally, as well as provide a better data foundation for management decisions.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

GV analyzed the data and drafted the manuscript. EH, BD, RW, and LH analyzed the data and edited the manuscript. DZ was an expertise and provided guidance in reconstruction and data management, and drafted and edited the manuscript.

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First Large-Scale Eastern Mediterranean and Black Sea Stock Assessment Reveals a Dramatic Decline

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The Mediterranean Sea is classified as a "data-poor" region in fisheries due to its low number of assessed stocks given its biodiversity and number of exploited species. In this study, the CMSY method was applied to assess the status and exploitation levels of 54 commercial fish and invertebrate stocks belonging to 34 species fished by Turkish fleets in the Eastern Mediterranean (Levantine) and Black Seas, by using catch data and resilience indices. Most of these marine taxa currently lack formal stock assessments. The CMSY method uses a surplus production model (SPM), based on official catch statistics and an abundance index derived from scientific surveys. The SPM estimates maximum sustainable yield (MSY), fishing mortality (F), biomass (B), fishing mortality to achieve sustainable catches (F_{msy}), and the biomass to support sustainable catches (B_{msy}). Our results show the estimated biomass values for 94% of the stocks were lower than the required amount to support sustainable fisheries (B_{msy}). Of the 54 stocks, 85% of them can be deemed as overfished; two stocks were not subject to overfishing (*Sardina pilchardus* and *Trachurus mediterraneus* in the Marmara Sea) while only one stock (*Sprattus sprattus* in the Black Sea) is healthy and capable of producing MSY. Annual values of the stock status indicators, F/F_{msy} and B/B_{msy}, had opposing trends in all regions, suggesting higher stock biomasses could only be achieved if fishing mortality is drastically reduced. Recovery times and levels were then explored under four varying F/F_{msy} scenarios. Under the best-case scenario (i.e., $F = 0.5F_{msy}$), over 60% of the stocks could be rebuilt by 2032. By contrast, if normal fishing practices continue as usual, all stocks will soon be depleted [if not already] ($F = 0.95F_{msy}$), whose recover may be impossible at later dates. The results of this study are supported by previous regional assessments confirming the overexploitation of Turkish fisheries is driving the near-total collapse of these marine wild fisheries. Hence, the need to urgently rebuild Turkey's marine fisheries ought to be prioritized to ensure their future viability.

Keywords: stock status, data-poor stocks, CMSY, recovery, Turkey, fisheries management

INTRODUCTION

Fisheries can effectively be managed if there is sufficient scientific understanding of the current biomasses and trends of fish stocks. However, because most stocks lack robust assessments, practical advice for them, and hence management capabilities, are impaired. The main element of management frameworks in the adoption of adequate multi-annual management plans (MAPs) and fishery standards according to the Marine Stewardship Council (MSC) is stock assessment (Aranda et al., 2019). However, stock assessments are understood to be costly and so they are not usually set as a priority for less-developed nations (Cope and Punt, 2009). Thus, for the majority of stocks, only time-series of landings data exists and some biological information on harvested species, such as their fecundity, length-at-maturity, growth rates (Froese and Pauly, 2019); this dearth of knowledge has compromised the ability to manage them.

The proportion of stocks that are routinely and regularly assessed is greater in the Northeast Atlantic (Cardinale et al., 2013) than in the Mediterranean and Black Seas (Lleonart, 2015; Stergiou et al., 2016). This disparity is mainly due to the multi-species nature of fisheries in the Mediterranean (Lleonart and Maynou, 2003), and the more advanced fisheries management capacities of Northern European countries. Consequently, the Mediterranean and Black Seas are generally classified as "data-poor regions," stemming from a combination of very few empirically assessed stocks and the little landings and biological data available (Pilling et al., 2008). However, the situation has improved slightly since 2010, as more stocks began to undergo assessments (Colloca et al., 2013), but the quality of these data must still be critically improved upon, especially for the "extremely data-poor" Eastern Mediterranean (Tsikliras et al., 2015).

Data-poor regions require a different approach to stock assessment and fisheries management that deviates from classical age-based models, by incorporating novel techniques designed to produce fisheries reference points when biological data are limited and catch/biomass time series are short or incomplete (Froese et al., 2017). The "Surplus Production Model" (SPM) demands the least amount of input data compared to other models for estimating Maximum Sustainable Yield [MSY] (Winker et al., 2018). Although SPM has its limitations, such as not incorporating the size or age structure of a stock's population (Wang et al., 2014), the model does provide key stock status indicators (i.e., current biomass and fishing mortality) and related fisheries reference points (Bmsy and Fmsy) enabling the formulation of species' stock assessments in data-limited regions (Punt and Szuwalski, 2012; Froese et al., 2017). Recent improvements to the SPM such as its implementation of Bayesian inferential statistics (McAllister et al., 2001) now provide a realistic estimation of stock biomass relative to carrying capacity (B/k). Establishing fisheries reference points and stock status indicators (F/Fmsy and B/Bmsy) can promote effective management toward achieving sustainability. The main objective of both the EU Common Fisheries Policy (CFP) and the General Fisheries Commission for the Mediterranean (GFCM) is to manage fish stocks so they attain MSY by 2020, at the latest, to deliver the highest possible long-term catches. Additionally, the EU's "Marine Strategy Framework Directive" (MSFD, 2008) main criteria for assigning "Good Environmental Status" (GES) for fisheries is that fishing mortality (F) does not exceed the level (Fmsy) producing MSY, Spawning Stock Biomass (SSB) is consistent with the level associated with MSY, and the proportion of adult fish is high enough to ensure stock renewability. Since Turkey is now improving its capacities and initiatives in line with this directive, stock sustainability is indeed a vital short-term goal.

Background on the Development and Changes in the Turkish Fisheries

Turkey is one of the strongest fishing countries in the Mediterranean (**Figure 1A**), mostly owing to its anchovy catch from the Black Sea (**Figure 1B**). The main controls currently implemented as management measures in Turkey include the following (Resmî Gazete, 2016, Sayı: 29800 #2016/35): (i) Temporal industrial fishing bans in the Black and Marmara Seas from April 15 to September 1, and a spatial trawling ban in the Marmara Sea and the Bosphorus and Dardanelles Straits (Turkish Strait System-TSS); (ii) Minimum Landing Size (MLS) regulations for most commercial taxa; (iii) Vessel licensing restrictions; (iv) Minimum mesh size regulations; and (v) Protected species (i.e., dolphins [Delphinidae], monk seals [Phocidae], sponges [Porifera], sturgeons [Acipenseridae], grouper [Epinephelinae], and several sharks and rays [Elasmobranchii]). Additionally, catch quotas were previously implemented for the striped Venus clam (*Chamelea gallina*) in the Black Sea and for the bluefin tuna (*Thunnus thynnus*) in the Mediterranean. The bluefin tuna quota is still operational, along with a recent regulation for swordfish *Xiphias gladius*, both of which are mandated by the International Commission for the Conservation of Atlantic Tunas (ICCAT).

The concept of "Fishing down marine food webs" (Pauly et al., 1998) is clearly exemplified in Turkey. There, fishers first depleted the most profitable stocks before moving on to the next lesser- valued stocks, until left with smaller and less economically valuable taxa. Prior to the development of industrial fisheries in the 1970s, fishers could catch as much swordfish, bluefin tuna, Atlantic mackerel, Atlantic chub mackerel, horse mackerels, and bluefish as they could carry (Ulman and Pauly, 2016). However, in 1980, the Turkish economy was transformed, from a semi-state controlled one to a free market economy. This new economy sought rapid development, and, to encourage this, provided bank credit, subsidies and custom duty exemptions to fishers to quickly develop this sector, which propelled it toward unplanned and uncontrolled growth fueling too much fishing effort (Knudsen et al., 2010). The large marine pelagic predators were first removed from the Black and Marmara Seas in the early 1970s (Oğuz, 2017) due to their lucrative value. This was followed by the collapse of demersals in the 1980s namely stingrays, elasmobranchs, flatfish, turbot, and red mullet (Knudsen et al., 2010) when bottom trawling developed in the 1980s, backed by state funding (Can, 2013). Following the collapse of most demersal populations in both seas, bottom trawling became non-operational in the mid to late 2000s due to the ensuing

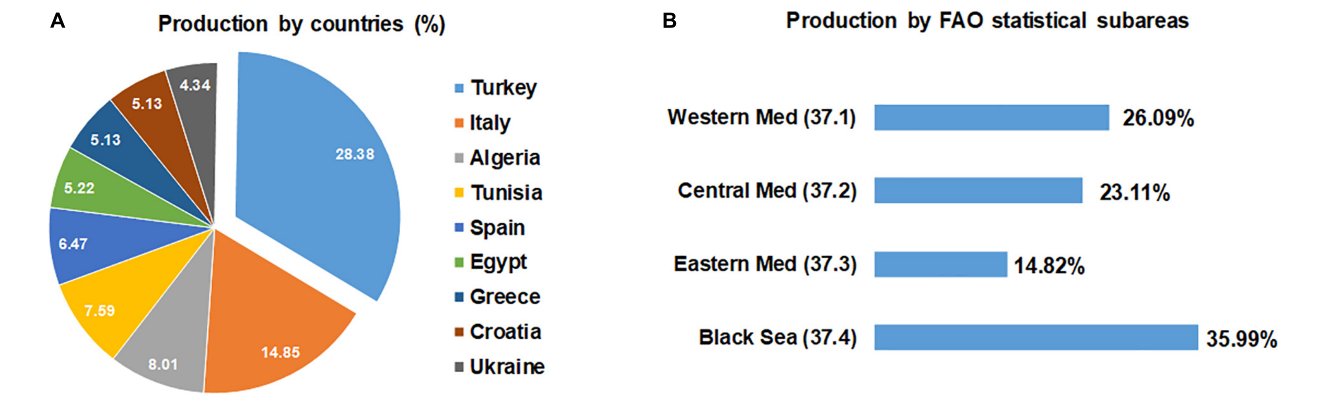

FIGURE 1 | (A) Top ranked fishing nations in the Mediterranean and Black Seas and **(B)** portion of catches taken from each subregion in 2017 (Food and Agriculture Organization [FAO], 2018).

demersal fish scarcity from depleted stocks, so many boats switched their main harvesting methods to mid-water trawling for pelagic species.

These drastic declines of elasmobranchs and other large predatory fish should alone be a priority concern for management, because large predators are essential for exerting healthy top–down control ecosystem functioning (Myers and Worm, 2003; Steneck, 2012). The benthic food web in the Black Sea littoral area has been heavily disrupted by the considerable loss of its largest predators (i.e., cartilaginous fish; *Squalus acanthias*, *Raja clavata*, *Scophthalmus maximus*), due both to overfishing and high discards; with these top–down controls gone, a noticeable response (Frid et al., 1999) was a greater abundance of crustaceans (especially *Liocarcinus depurator*) in the absence of their demersal predators (Zengin et al., 2014). A benthic non-indigenous species (NIS) sea snail native to the Pacific Northwest, the Rapa whelk *Rapana venosa*, first appeared in the Black Sea in 1946, spreading throughout the basin over the next decade—due to its high abundance and demand in other countries, Turkey has been exporting this species into Asia since the 1980s, in sizeable amounts (Turkish Statistical Institute [TUIK], 2018). It has since become a dominant predatory species in the marine food web (Zengin et al., 2014), in both the Black and Marmara Seas. Other Black Sea top predators include three cetacean species: the Black Sea harbor porpoise (*Phocoena phocoena relicta*), bottlenose dolphin (*Tursiops truncatus*), and short-beaked common dolphin (*Delphinus delphis*), but unfortunately, turbot fisheries using trammel netting commonly results in cetacean by-catch (Birkun and Krivokhizhin, 2011; Tonay, 2016). For this reason, their populations are likely declining, especially that of *P. phocoena relicta*, the dominant Western Black Sea cetacean (Birkun and Frantzis, 2008).

The Levantine Sea also suffered major predator losses, but not as severely as in the Black and Marmara Seas probably due to its larger size and openness. The marine biodiversity of the Eastern Mediterranean has become more tropical since the opening of the Suez Canal in 1869, and its more recent enlargement in 2015, which serves as a migration corridor for NIS of Indo-Pacific origin (European Environment Agency [EEA], 2012). A NIS is considered invasive when it causes either ecological or economical damage or poses a threat to human health. A few invasive species are problematic for fishers, but none more so than the silver-cheeked toadfish *Lagocephalus sceleratus*, an extremely toxic pufferfish species that has already threatened human life in the Eastern Mediterranean (Kosker et al., 2016) via its neurotoxin tetrodotoxin (TTX), which is 1000 times more poisonous to humans than cyanide (Lago et al., 2015). This pufferfish has since become a dominant species through much of Turkey's south, bringing additional stress to the already marginalized small-scale fishers, particularly by damaging fishing nets, consuming captured fish within nets, and swallowing many longline fishing hooks (Ünal and Göncüoğlu, 2012). Another NIS whose abundance is also increasing in Turkey's southeast is the yellowspotted pufferfish *Torquigener flavimaculosus*. However, not all NIS are pest species, and some even provide economic benefits to fishers, such as Randall's threadfin bream *Nemipterus randalli*, the rabbitfish (*Siganus* spp.), the brushtooth lizardfish *Saurida undosquamis*, the goldband goatfish *Upeneus moluccensis*, the redcoat (squirrelfish) *Sargocentron rubrum*, and the lionfish (*Pterois miles*).

Study Aims

Here we present a first comprehensive stock assessment for Turkey's major and/or important commercial fisheries from the Black Sea, Marmara Sea, and Northeastern Levantine Sea. To do this, we applied the new "CMSY" method (Froese et al., 2017) to 50 years of reported catch data, to better understand the current plight of the stocks and the fishing pressure exerted upon them, to provide management advice and realistic future scenarios aimed at rebuilding these fisheries. The "CMSY" method is a low-cost assessment approach incorporating the "surplus production" stock assessment model, requiring only a long time-series of catch data (Froese et al., 2017). Specifically, this study aimed to: (i) Estimate total current stock biomass (B) and fishing mortality (F) in relation to MSY levels; (ii) Estimate the total potential stock biomass (Bmsy) and the fishing mortality (Fmsy) required to produce MSY; (iii) Calculate the differences between current biomasses and potential biomasses (i.e., recovery ability); and, finally, (iv) Present recovery options for managers by estimating the time needed for stocks' restoration under different fishing mortality scenarios.

MATERIALS AND METHODS

The Dataset

A total of 54 commercial fish and invertebrate stocks belonging to 34 species were analyzed for respective stock status by using a 50 years' time-series of archived national catch data. Turkey began spatially reporting its catches from five distinct marine sub-regions in 1967: Western Black Sea, Eastern Black Sea, Marmara Sea, Aegean Sea, and Levantine Sea (Eastern Mediterranean portion); so this was the initial year used in our study. Underreporting is still serious problem in catch statistics globally. Although the uncertainty levels associated with the Turkish catch data were admittedly higher in the past (50 to 70 years ago), recently this has been somewhat improved upon (see Ulman et al., 2016a,b,c). Nevertheless, national data (from 1967 to 2017) were chosen as the source data to demonstrate how this stock assessment technique could be specifically applied to marine catch data. Catches for the Western and Eastern Black Sea were combined to represent catches from the Black Sea. The Aegean Sea was excluded from this analysis, since its commercial catches are more of an artisanal nature owing to its very narrow continental shelf, and the reporting of industrial catches from there are better represented than small-scale catches (Ulman et al., 2013). Therefore, the Turkish Exclusive Economic Zone (EEZ) in the Black Sea, Marmara Sea and the Levantine Sea constituted the three marine bodies investigated in this study. Stocks were chosen based on either their historical or current commercial influence and/or their hypothesized role in the trophic ecosystem, by relying on expert knowledge. All analyses were done using the R statistical program (v3.6.1; R Core Team, 2014).

Stocks' Status

The CMSY model (Froese et al., 2017) is an open-source stock assessment model for data-limited stocks in fisheries that produces reference points, such as biomass trends, and MSY, based on only catch and species resilience data. Using a time-series of catches can be a very useful indicator of stock status as declining catches generally represent stock reductions, so long as mortality is not decreasing over time, and migration is not a factor (ICES, 2018). Particularly, CMSY-BSM is a Bayesian model that uses a Markov Chain Monte Carlo approach based on the Schaefer SPM (Schaefer, 1954). Species resilience is defined as the "measure of a species' ability to absorb changes in variable states, driving influences and parameters, and still persist" (Holling, 1973). Resilience ranges are assigned by combining several population parameters, such as maximum population growth (rmax), von Bertalanffy growth rate (K), fecundity (total egg production), age of maturity (tm), and maximum age (tmax). The maximum population growth rate (rmax) is defined as the "intrinsic rate of population increase, equal to per capita birth minus death rate," which is closely related to carrying capacity (k), the latter corresponding to when "a population's per capita growth rate gets diminutively smaller as the population size approaches its maximum limit imposed by the limited resources of the environment" (Rees, 1996). Below is the dynamic formula of the Schaefer model using a Bayesian approach for the best estimate of biomass:

$$B_{t+1} = B_t + r\left(1 - \frac{B_t}{k}\right)B_t \ - C_t \tag{1}$$

where B is the biomass (metric tons), C is the catch (metric tons), r is the intrinsic rate of population growth, k is the carrying capacity, and t is the time (years).

For a given species, CMSY requires prior information to be specified for resilience and productivity at the beginning and the end of the catch time-series. Priors are the most important part of this model; in particular, the "resilience prior r" (later represented as r) must be estimated by experts for this CMSY analysis. It should be noted that the reliability of stock assessment results is completely dependent upon setting a proper prior range for r. If only catches are known, a prior r is derived from life-history traits and stock assessment records, while a prior range for k can be derived from the maximum catch. Here, prior estimates for r were taken from the "Estimates based on models" section found on summary pages for species in FishBase[1] and SeaLifeBase[2] (**Table 1**).

Probable ranges for the maximum intrinsic rate of population increase (r) and for the unexploited population size or carrying capacity (k) are filtered with a Monte Carlo approach, to detect "viable" r–k pairs. Their empirical relation is formulated this way:

$$r \approx 2M \ \approx 3\ K \ \approx 3.3/t_{gen} \approx 9/t_{max} \tag{2}$$

where M is the rate of natural mortality, K is the von Bertalanffy growth parameter, *tgen* is the generation time, and tmax is the maximum age (Froese et al., 2017). The CMSY method incorporates a routine for estimating wide (uniform) priors for k (Froese et al., 2017):

$$k_{low} = \left(\frac{C_{max}}{r_{high}}\right);\ k_{high} = \left(\frac{4C_{max}}{r_{low}}\right)_t$$

$$k_{low} = \left(\frac{2C_{max}}{r_{high}}\right);\ k_{high} = \left(\frac{12C_{max}}{r_{low}}\right)_t \tag{3}$$

where *klow* and *khigh* are the lower and upper bounds of the prior range of k; *Cmax* is the recorded maximum catch in the time series; *rlow* and *rhigh* are respectively the lower and upper bounds of the range of r values that the CMSY method explores.

Prior ranges for Bt/k (beginning and end of catch time-series) were derived from expert knowledge. Using a Monte Carlo approach, all r–k combinations that are compatible with the life history traits (r, M, K), the catches (Ct), and the

[1] www.fishbase.org
[2] www.sealifebase.org

TABLE 1 | Scientific names, functional groups, and resilience categories from FishBase (and from SeaLifeBase for invertebrates) with prior ranges as input parameters for r (intrinsic rate of population), based on classification of resilience and for relative biomass (B/k) depending on the depletion status.

Scientific name	Stock name	Functional group	Resilience category	Prior ranges for r	Relative biomass category (depletion)	Prior ranges for B/k
Engraulis encrasicolus	Anchovy	Plankton feeders	Medium	0.39–0.91	Nearly unexploited	0.8–1.0
Sardina pilchardus	Sardine		Medium	0.40–0.90	Nearly unexploited	0.8–1.0
Sprattus sprattus	Sprat		Medium	0.32–0.74	Nearly unexploited	0.8–1.0
Trachurus mediterraneus	MedHMack		Medium	0.33–0.76	Nearly unexploited	0.8–1.0
Trachurus trachurus	HorseMack		Medium	0.31–0.72	Nearly unexploited	0.8–1.0
Scomber colias	AChub		Medium	0.39–0.90	Nearly unexploited	0.8–1.0
Scomber scombrus	Mackerel		Medium	0.31–0.75	Medium depletion	0.2–0.6
Boops boops	Bogue		Medium	0.39–0.89	Nearly unexploited	0.8–1.0
Alosa immaculata	Shad		Medium	0.20–0.80	Nearly unexploited	0.8–1.0
Spicara smaris	Picarel		Medium	0.28–0.63	Medium depletion	0.2–0.6
Pomatomus saltatrix	Bluefish	Large predators	Medium	0.44–1.01	Nearly unexploited	0.8–1.0
Sarda sarda	Bonito		Medium	0.37–0.85	Low depletion	0.4–0.8
Merlangius merlangus	Whiting		Medium	0.33–0.74	Nearly unexploited	0.8–1.0
Merluccius merluccius	Hake		Medium	0.35–0.80	Nearly unexploited	0.8–1.0
Belone belone	Garfish		Medium	0.29–0.65	Medium depletion	0.2–0.6
Dicentrarchus labrax	Seabass		Medium	0.35–0.84	Low depletion	0.4–0.8
Lichia amia	Leerfish		Medium	0.34–0.76	Nearly unexploited	0.8–1.0
Epinephelus marginatus	Grouper		Low	0.24–0.55	Nearly unexploited	0.8–1.0
Scophthalmus maximus	Turbot	Benthic fish and invertebrates	Medium	0.33–0.74	Low depletion	0.4–0.8
Mullus barbatus barbatus	Mullet		Medium	0.42–1.04	Low depletion	0.4–0.8
Chamelea gallina	Venus		Medium	0.32–0.76	Low depletion	0.4–0.8
Mullus surmuletus	Surmullet		Medium	0.20–0.80	Medium depletion	0.2–0.6
Diplodus annularis	Annular		Medium	0.20–0.80	Nearly unexploited	0.8–1.0
Oblada melanura	Saddled		Medium	0.52–1.17	Low depletion	0.4–0.8
Sparus aurata	Seabream		Medium	0.37–0.85	Low depletion	0.4–0.8
Argyrosomus regius	Meager		Low	0.10–0.49	Nearly unexploited	0.8–1.0
Dentex dentex	Dentex		Low	0.24–0.61	Nearly unexploited	0.8–1.0
Conger conger	Conger		Low	0.23–0.53	Nearly unexploited	0.8–1.0
Raja clavata	Raja		Low	0.07–0.45	Medium depletion	0.2–0.6
Squatina squatina	Angelshark		Low	0.16–0.47	Medium depletion	0.2–0.6
Squalus acanthias	Dogfish		Low	0.05–0.50	Medium depletion	0.2–0.6
Parapenaus longirostris	RoseShrimp		High	0.68–1.54	Nearly unexploited	0.8–1.0
Octopus vulgaris	Octopus		High	0.53–1.21	Nearly unexploited	0.8–1.0
Sepia officinalis	Cuttlefish		Medium	0.37–0.84	Low depletion	0.4–0.8

expert knowledge (Bt/k) were identified. Therefore, we followed generic rules established by Froese et al. (2017), for setting the biomass priors based on general available knowledge of the fisheries. Rules for the initial prior biomass range assumed a high initial biomass (0.5–0.9 k) if the time-series of catch data started before 1960, when most fisheries were recovering after World War II (Pauly et al., 2002; Froese et al., 2017; **Supplementary Material**), and medium (0.2–0.6 k) biomass after 1960. For the final biomass, its range was set up using the last catch relative to maximum catch. Final biomass range is set to high if this ratio was over 0.7 and set to low if it was lower than 0.3. Since there is no proper stock assessment research or CPUE data for Turkish stocks available for use, an automatic selection of low, medium, or high prior biomass ranges on the simple default rules described above were used for our CMSY analyses.

In the Schaefer model, 0.5 rmax = Fmsy and 0.5 k = Bmsy (Ricker, 1975). Fmsy is the maximum rate of fishing mortality (i.e., the proportion of caught fish) that eventually ensues, usually over a very long time frame, in a population size of Bmsy (the resulting stock biomass from fishing at Fmsy). The outputs of the analyses produce proxies for MSY, key reference points such as fishing mortality that can produce MSY (Fmsy), biomass that can produce MSY (Bmsy), and indicators such as relative stock size (B/Bmsy) and fishing mortality (F/Fmsy) to better understand current stock trends and status. We also present cumulative results for biomass and MSY, and median fishing mortality by main functional groups (benthic fish and invertebrates, small pelagics, benthopelagics, and large predators) according to sea location.

These criteria were used for evaluating stock status: severely depleted (B < 0.2Bmsy), critical condition (B < 0.2Bmsy, F > Fmsy), exploited outside safe biological limits (B < 0.5Bmsy), subject to overfishing (F > Fmsy), recovering (B < Bmsy, F ≤ Fmsy), and healthy (B > Bmsy, F ≤ Fmsy) (**Table 2**). Overall, 16 stocks were assessed from the Black Sea, 17 stocks from the Marmara Sea, and 21 stocks from the Levantine Sea.

Expected Time to Rebuild Overfished Stocks

In addition to the Fmsy estimation for each stock, recovery options were also considered according to stock biomass (projected for 2018) under four fishing mortality scenarios for 15 year (until 2032) (**Table 3**). The time needed to rebuild stocks so that they are at Bmsy was calculated using outputs of the CMSY model. Thus, the recovery time needed to reach the biomass level (Bmsy) capable of producing MSY, is a function of depletion and remaining fishing mortality (F_{msy} – F), and it was calculated for F equal to 0.5 F_{msy} for every stock according to the this formula:

$$\Delta t = \frac{1}{2F_{msy} - F}\, ln\left(\frac{\frac{B_{msy}}{B_{cur}}\, 2\left(1 - \frac{F}{2F_{msy}}\right) - 1}{2\left(1 - \frac{F}{2F_{msy}}\right) - 1}\right) \tag{4}$$

For a given stock, its potential catch was calculated as the difference between the current catch and the catch at MSY. The final value was conservatively reduced by 10% to account for stocks that cannot reach MSY at the same time.

The CMSY model output showed that, in 2017, of the 54 assessed stocks, 29 stocks (54%) were in critical condition and/or being severely depleted (B < 0.2Bmsy), yet all are still being fished (**Figure 2**). Another 17 stocks (31%) were exploited outside safe biological limits (0.2Bmsy < B < 0.5Bmsy), while five stocks (9%) are over 0.5 Bmsy and two stocks (*Sardina pilchardus* and *Trachurus mediterraneus* from the Marmara Sea) are not overfished and close to a sustainable level (F ≤ Fmsy and B < Bmsy), with just one stock of the 54 seemingly sustainable (*Sprattus sprattus* from Black Sea) (**Figure 2** and **Table 4**). Overall, 46 stocks (85%) were overfished. Catches in 2017 were below the maximum sustainable yield (C/MSY < 1) for 52 stocks—the Marmara Sea sardine being very close to MSY—and above the MSY for the remaining two stocks (Atlantic mackerel and striped Venus clam). We provide plotted outputs of the CMSY models for each stock assessment and its associated management information in the **Supplementary Material**.

Outputs of CMSY analyses by main functional groups (benthic fish and invertebrates, large predators, pelagic plankton feeders) are presented as summed catches, median biomass relative to the level that can produce the MSY (B/B_{msy}), and median

TABLE 3 | Four scenarios using different fishing mortality ratios according to exploitation level of stocks.

Scenarios	Fishing mortality (F)	Biomass (B)
1	0.50	$<0.2B_{msy}$
2	0.60	$\leq 0.5B_{msy}$
3	0.80	$\geq 0.5B_{msy}$
4	0.95	Control ($\sim B_{msy}$)

TABLE 2 | Definition of stock status according to fisheries reference points.

Reference points	Definition
$B > B_{msy}$ and $F \leq F_{msy}$	Healthy (not subject to overfishing and having a biomass higher than necessary to produce the maximum sustainable yield)
$B < B_{msy}$ and $F \leq F_{msy}$	Recovering (not subject to overfishing and having a biomass very close to one that can produce the maximum sustainable yield)
$F > F_{msy}$	Subject to overfishing (fishing mortality higher than the one that can produce the maximum sustainable yield)
$B \leq 0.5\ B_{msy}$	Exploited outside safe biological limits (biomass under half that required to produce the maximum sustainable yield, 0.5 B_{msy} is B_{pa})
$F > F_{msy}$ and $B \leq 0.5\ B_{msy}$ or $B \leq 0.2\ B_{msy}$	Critical condition (being outside safe biological limits and subject to overfishing or being severely depleted)
$B \leq 0.2\ B_{msy}$	Severely depleted (biomass below 20% of that can produce the maximum sustainable yield)

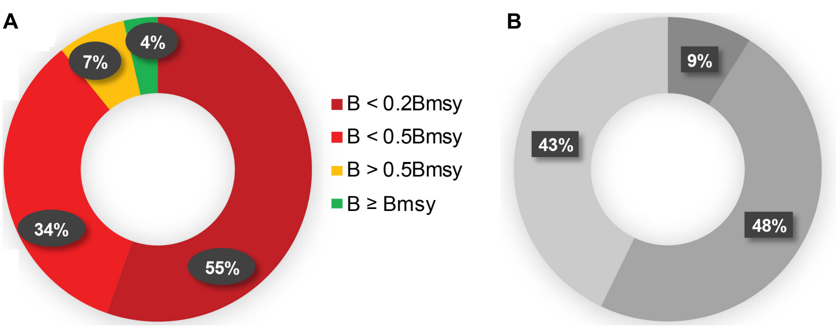

FIGURE 2 | Status of assessed stocks ranked by their **(A)** current biomass levels relative to different B_{msy} levels, for which the dark red area indicates stocks under critical condition and/or severely depleted, the red area the overfished stocks, the yellow area the recovering stocks, and the green area with healthy stocks; and **(B)** fishing mortality over three different F_{msy} levels.

TABLE 4 | CMSY outputs for MSY, B_{msy}, F_{msy}, and r with stock indicators B/B_{msy} and F/F_{msy} for stocks in Turkish seas from 2017.

Scientific name	Region	Year	MSY (t)	Catch (t) 2017	F_{msy}	F/F_{msy}	B_{msy} (t)	B/B_{msy}
Engraulis encrasicolus	Black Sea	2017	232468	133767	0.27	1.06	853250	0.54
Sprattus sprattus	Black Sea	2017	49104	33944	0.28	0.54	166654	1.34
Trachurus mediterraneus	Black Sea	2017	65772	4257	0.22	4.15	289030	0.09
Trachurus trachurus	Black Sea	2017	8035	2167	0.21	2.35	38382	0.24
Scomber colias	Black Sea	2016	6788	1	0.29	0.01	23563	0.07
Alosa immaculata	Black Sea	2017	1181	620	0.15	2.62	8159	0.31
Spicara smaris	Black Sea	2017	1217	5	0.19	0.32	6509	0.08
Pomatomus saltatrix	Black Sea	2017	11707	997	0.34	1.24	34346	0.18
Sarda sarda	Black Sea	2017	13378	5570	0.29	1.43	47019	0.38
Merlangius merlangus	Black Sea	2017	14545	7416	0.21	1.70	69704	0.38
Belone belone	Black Sea	2017	2117	99	0.17	3.36	11671	0.09
Scophthalmus maximus	Black Sea	2017	2617	152	0.23	2.59	11443	0.11
Mullus barbatus barbatus	Black Sea	2017	2636	329	0.26	2.67	10259	0.15
Raja clavata	Black Sea	2017	883	12	0.07	0.78	13382	0.09
Squalus acanthias	Black Sea	2014	2479	1	0.07	0.01	40437	0.09
Chamelea gallina	Black Sea	2017	30830	34941	0.28	1.20	107524	0.95
Engraulis encrasicolus	Marmara Sea	2017	18438	8341	0.34	1.60	54005	0.38
Sardina pilchardus	Marmara Sea	2017	6285	5685	0.34	0.94	18440	0.97
Trachurus trachurus	Marmara Sea	2017	2969	1729	0.24	3.06	12332	0.31
Trachurus mediterraneus	Marmara Sea	2017	4908	2718	0.25	0.96	20327	0.57
Scomber colias	Marmara Sea	2017	9492	147	0.27	1.48	34331	0.07
Scomber scombrus	Marmara Sea	2017	261	387	0.20	7.31	1304	0.32
Pomatomus saltatrix	Marmara Sea	2017	3983	720	0.33	2.44	12502	0.19
Sarda sarda	Marmara Sea	2017	2444	1103	0.24	2.49	10233	0.30
Merlangius merlangus	Marmara Sea	2017	6618	79	0.29	0.70	5427	0.09
Merluccius merluccius	Marmara Sea	2017	1313	248	0.24	2.70	22573	0.19
Scophthalmus maximus	Marmara Sea	2017	158	15	0.26	1.59	605	0.17
Mullus barbatus barbatus	Marmara Sea	2017	216	3	0.31	1.14	701	0.08
Mullus surmuletus	Marmara Sea	2017	605	76	0.14	1.98	4267	0.18
Raja clavata	Marmara Sea	2017	159	145	0.10	1.51	406	0.58
Squalus acanthias	Marmara Sea	2017	191	11	0.06	2.93	3366	0.10
Squatina squatina	Marmara Sea	2017	46	1	0.11	1.29	1586	0.09
Parapenaus longirostris	Marmara Sea	2017	3702	1389	0.60	1.10	6170	0.41
Sardina pilchardus	Levantine Sea	2017	5426	3274	0.34	1.08	15739	0.56
Trachurus trachurus	Levantine Sea	2017	440	227	0.26	3.15	1709	0.28
Trachurus mediterraneus	Levantine Sea	2017	687	234	0.27	2.22	2565	0.27
Boops boops	Levantine Sea	2017	491	237	0.27	2.24	1810	0.33
Sarda sarda	Levantine Sea	2017	595	381	0.32	1.27	1894	0.50
Pomatomus saltatrix	Levantine Sea	2017	275	77	0.33	2.09	844	0.26
Merlangius merlangus	Levantine Sea	2017	403	78	0.27	2.95	1529	0.18
Dicentrarchus labrax	Levantine Sea	2017	675	11	0.27	1.04	2561	0.09
Epinephelus marginatus	Levantine Sea	2017	245	1	0.19	0.25	1296	0.09
Lichia amia	Levantine Sea	2017	678	169	0.23	4.29	2980	0.17
Mullus barbatus barbatus	Levantine Sea	2017	1095	632	0.29	1.74	3866	0.40
Mullus surmuletus	Levantine Sea	2017	808	17	0.39	1.43	2100	0.09
Argyrosomus regius	Levantine Sea	2017	86	10	0.10	5.30	914	0.10
Dentex dentex	Levantine Sea	2017	78	4	0.18	2.60	433	0.10
Conger conger	Levantine Sea	2016	296	108	0.18	3.28	1707	0.23
Diplodus annularis	Levantine Sea	2017	102	24	0.14	5.32	762	0.15
Oblada melanura	Levantine Sea	2017	99	8	0.34	1.37	296	0.17
Sparus aurata	Levantine Sea	2017	604	223	0.24	1.94	2551	0.31
Squalus acanthias	Levantine Sea	2017	155	3	0.05	0.89	3074	0.10
Octopus vulgaris	Levantine Sea	2017	160	3	0.39	0.89	409	0.10
Sepia officinalis	Levantine Sea	2017	813	673	0.31	1.37	2591	0.60

fishing mortality to the MSY level (F/F_{msy}) for stocks according to seas (Black Sea, Marmara Sea, Levantine Sea) (**Figure 3**). Lastly, fishing mortality (F/Fmsy) and biomass state (B/Bmsy), respectively, are presented for each assessed stock by Turkish sea (**Figure 4**).

Black Sea

The 16 Black Sea stocks analyzed consisted of 13 teleosts, two elasmobranches, and one invertebrate. Functionally, they were grouped as follows: seven plankton feeders, four large predators, and five benthic fish and invertebrates. Biomasses of all these stocks have gradually decreased since the 1980s, and, from about 1990, all stocks can be deemed unhealthy, with the exception of sprat that only became a target species in the late 1980s. Beginning in the 1980s, fishing mortality for all 16 stocks exceeded the level of mortality needed to produce MSY, and fishing mortality continued to increase over time. In 2017, 15 stocks (93.8%) were subject to ongoing overfishing of which nine (55%) were fished outside their safe biological limits (**Figures 3A–C**, **4A** and **Table 4**).

Marmara Sea

Of the 17 stocks analyzed for the Sea of Marmara, 13 were teleosts, plus three elasmobranches and one invertebrate. Their functional grouping was as follows: six plankton feeders, four large predators, and seven benthic fish and invertebrates. Two stocks (sardine and Mediterranean horse mackerel) still retain a moderate stock status, but because each is being fished at maximum mortality (F = Fmsy), this leaves little prey for their predators. Ten stocks were subject to ongoing overfishing and 14 stocks were fished outside safe biological limits (85%), of which nine are in critical condition. After 2005, fishing mortality was at a much higher, dangerous level than what is needed to produce MSY, and biomass values were critically low for all stocks, as confirmed by declining catches (**Figures 3D–F**, **4B** and **Table 4**).

Levantine Sea

For the Levantine Sea, the 21 stocks analyzed were composed of 18 teleosts, one elasmobranch, and two invertebrates. The dominant functional group was benthic fish and invertebrates with 11 stocks, followed by six large predators and four plankton feeders. Of the 17 stocks (90%) exploited outside safe biological limits, 10 of them are in critical condition. Since the mid-1990s, biomasses and catches for these three functional groups continued to decrease while fishing mortality increased (**Figures 3G–I**, **4C** and **Table 4**).

Future Scenarios

Future fishing scenarios were projected for four different fishing mortality reduction possibilities (50, 40, 20, and 5% of the stocks'

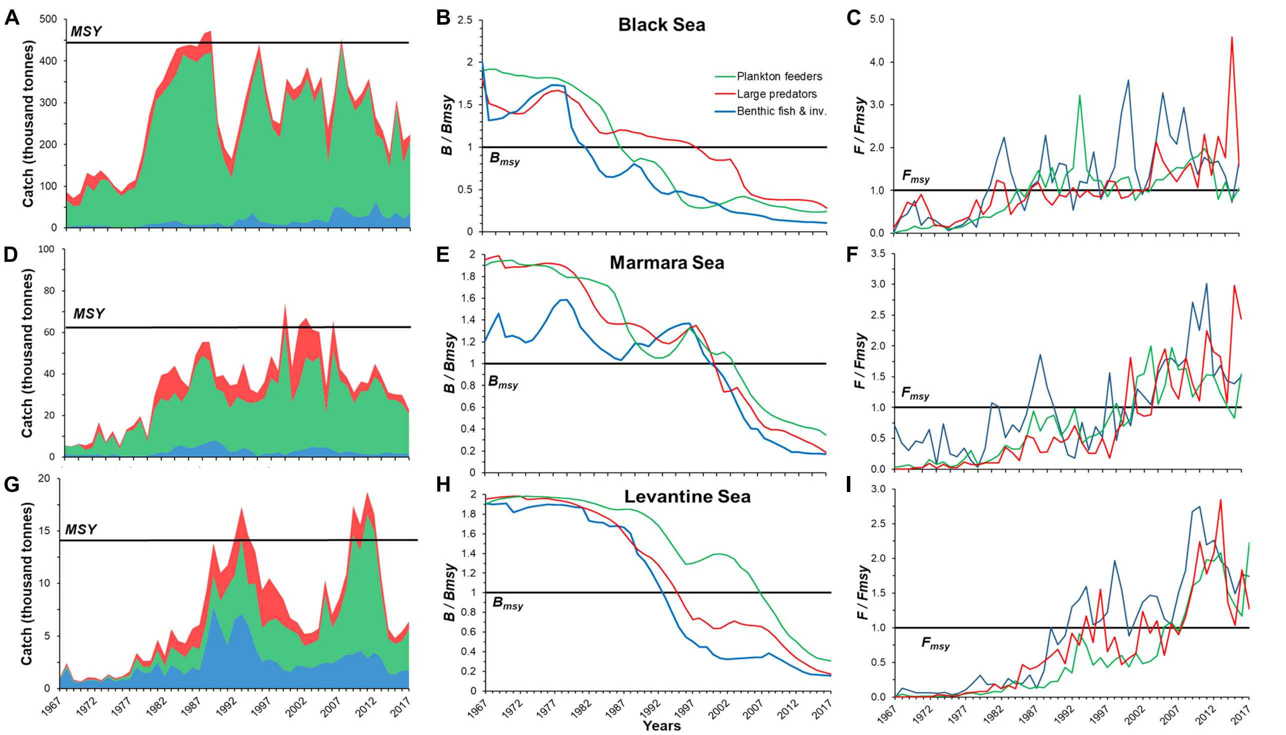

FIGURE 3 | Time-series of stock status by functional group (red: large predators, green: pelagic plankton feeders, blue: benthic species) for the Black Sea, Marmara Sea, and Levantine Sea, from 1967 to 2017. Summed catch **(A,D,G)**, median biomass relative to the level that can produce the MSY (B/Bmsy; **B,E,H**), and median fishing mortality **(C,F,I)** relative to the MSY level (F/Fmsy) is shown respectively for 17 stocks in the Black Sea **(A–C)**; 17 stocks in the Marmara Sea **(D–F)**, and 21 stocks in the Levantine Sea **(G–I)**. The black lines indicate the summed MSY, sustainable stock biomass at Bmsy level, and fishing mortality at Fmsy level, respectively, going from left to right.

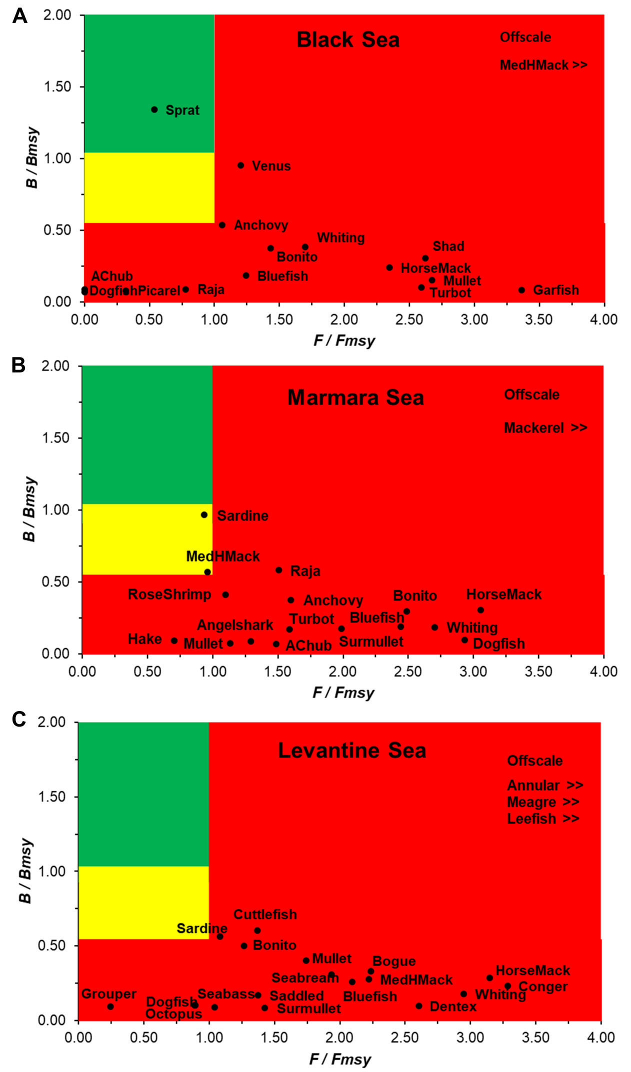

FIGURE 4 | Pressure plots showing fishing mortality (F/Fmsy) and relative stock size (B/Bmsy) for the assessed stocks in the **(A)** Black Sea, **(B)** Marmara Sea, and **(C)** Levantine Sea for 2017. The red area indicates overfished stocks (including those outside of safe biological limits, in critical condition, and severely depleted). The yellow area indicates recovering stocks and the green area shows healthy stocks. Scientific names of the fish were presented in **Table 1**.

respective Fmsy), according to the estimated biomass of the latest year that stocks could reach the biomass necessary to produce MSY (**Figure 5A**) and by reduction in catch necessary for catches to attain an MSY (**Figure 5B**). For the stocks capable of recovery, these scenarios show that if fishing mortality was reduced to 0.5Fmsy, 40% of the 54 stocks could reach the Bmsy level within 10 years, and 60% of them within 15 years. For the 0.6Fmsy scenario, half of the stocks can fully recover by 2032. If fishing mortality is set to 0.8Fmsy, within the next 15 years, 30% of stocks can reach the Bmsy level, for which catch values come closer to MSY levels than under any other options examined. However, a lowering of fishing mortality by only 5% (i.e., the 0.95 Fmsy values) would leave stocks overfished and unable to rebuild, while their catches remain similar to present values.

DISCUSSION

This first large-scale stock assessment of Turkish marine fisheries has revealed the impacts on them arising from overexploitation. Fishing mortality has resulted in unsustainable and rapidly declining stocks, which will inevitably cause further stock collapses without prompt management interventions. Of the 54 stocks assessed here, 85% are clearly overexploited (B < 0.5Bmsy), with only Black Sea sprat sustainably fished, and Marmara Sea Mediterranean horse mackerel and sardine near overfishing levels. It must be stressed that sprat is not consumed as a fish item but instead processed into fishmeal to support aquaculture and used in fish oil products (Ceyhan and Emir, 2015). Wild fish are a cheaper, much healthier alternative to farmed fish not requiring the additional culturing input costs, such as feed and placement, and so they should be prioritized over aquaculture fisheries. For the majority of the Turkish commercial fisheries, their stocks need to rebuild to levels near to or equal to their respective MSY. Our investigation of future scenarios shows that rebuilding the fisheries is indeed possible and it can be achieved through a reduction of effort and mortality to levels that allow stocks to reach their MSY (Merino et al., 2012; Froese et al., 2018).

Our results confirm other similar studies conducted in the Mediterranean and the Black Seas. That body of work shows that declines in catches and stock biomasses are a general theme in the region (Merino et al., 2015; Tsikliras et al., 2015; GFCM, 2016; Guillen et al., 2016; Scientific Technical Economic Committee for Fisheries [STECF], 2017; Food and Agriculture Organization [FAO], 2018), with the consensus being that stocks are exploited outside of safe biological limits (Osio et al., 2015). Further, after analyzing all completed stock assessments for the Mediterranean, Colloca et al. (2013) concluded that over 90% of those stocks are overexploited.

The future scenario outputs presented here suggest that, to rebuild Turkey's fisheries, a 50% reduction in mortality is the fastest way, in that over half of the 54 stocks could recover within 15 years. A 40% (0.6Fmsy) reduction would support the full recovery of half of the overexploited stocks by the year 2032. However, a 20% reduction enables less than half of the stocks to recover, but it would increase catches and revenue within 15 years. Conversely, if fishing continues as normal (F = 0.95Fmsy), stocks may not be able to recover later. Firstly, a clear management objective should be identified for which path to pursue rebuilding these stocks. In addition to mortality reductions, regulations on size selectivity are also needed (Colloca et al., 2013). Stocks currently at critical biomass levels may have much slower recovery rates (Neubauer et al., 2013), since a recovery will also depend on life-history strategies and trophic interactions between species (Hutchings and Reynolds, 2004; Audzijonyte and Kuparinen, 2016), especially for slow-growing and less-fecund species (Hutchings, 2000). Across the Mediterranean, young and juvenile

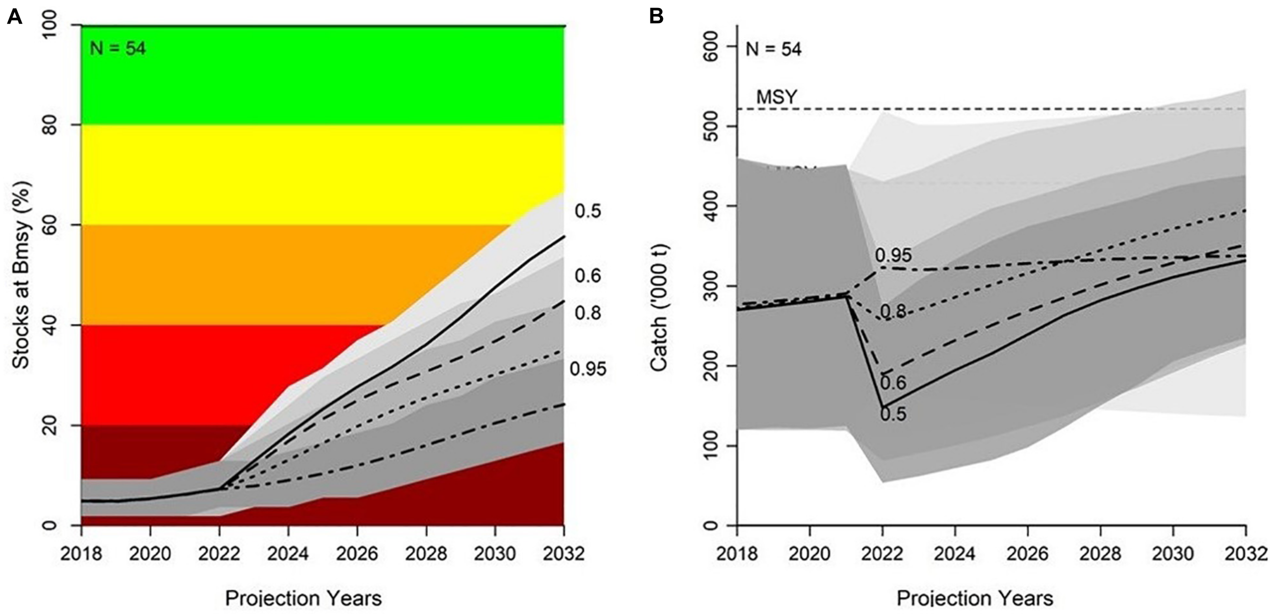

FIGURE 5 | Future scenarios for all of Turkeys' 54 commercial stocks assessed in this study, showing the **(A)** predicted percentage of stocks reaching the Bmsy level, and **(B)** predicted cumulative catch under four different fishing mortality scenarios, ranging from 0.5 to 0.95 Fmsy. Gray areas indicate the approximate 95% CI for each scenario.

fish, mostly ages 1 and 2 years, comprise the dominant portion of catch compositions because of low size selectivity by fishing gears (Colloca et al., 2013). To help improve the standing stock biomass, size regulation for the conservation of large adult females (Birkeland and Dayton, 2005; Froese et al., 2016) and juveniles, fishing restriction on spawning seasons and ensuring protection of nursery habitats should be used as additional measures to improve recruitment opportunities (Caddy, 2015). In Turkey, most commercial taxa (68% according to the 2016–2020 regulations) already have minimum landing size (MLS) regulations, and strengthening the effectiveness of such MLS measures would be quite easy, namely by improving awareness, controls, and fines (Froese et al., 2018).

Regulating fishing mortality to MSY levels is a true investment for the ecosystem and economy (Sumaila et al., 2012); additionally, species-specific MLS regulations must be effective to support the rebuilding of stocks (Froese et al., 2018). Beddington et al. (2007) noted that the definition of successful management is a combination of biological, economic, social, and political objectives. Aside from the obvious ecological benefits regarding the structure and function of marine ecosystems (Murawski, 2000), the economic benefits of exploiting the Mediterranean fish and invertebrate stocks at or near MSY within an 'Ecosystem Approach to Fisheries' (EAF) framework—proper fishing selectivity, restrictions on fishing gear, fishing season, and fishing areas—are enormous (Sumaila et al., 2012; Colloca et al., 2013). These would guarantee revenues and long-term profitability (Ye et al., 2012; Sumaila et al., 2016), with a view to job security as the long-term goal (Beddington et al., 2007) for the sector. Rebuilding the global fisheries is a costly process, estimated at ca. $200 billion, because it may include subsidies, buyback programs, and alternative employment training for fishers, but once stocks reach their MSY levels, profits for the global fishing fleet could increase by ca. $50 billion per year (Sumaila et al., 2016).

Challenges in Assessing Stocks

While we agree that MSY is a less sophisticated model than, for example, the yield-per-recruit model of Beverton and Holt (1957), the MSY concept was nevertheless chosen as the framework for fisheries management by the CFP. Accordingly, we relied on it to make a first evaluation in Turkey for its future fisheries within the framework of data-limited stocks. Note that our results actually confirm several MSY criticisms: while most managers still believe that fishing at F = Fmsy is the best option, our results demonstrate that fishing at that level provides the least desirable outcome. Thus, here we proposed fisheries reference points and indicators derived from the SPM for the MSY concept, which provides assessments for highly fluctuating and high-biomass small pelagics, and also for mixed-fisheries in the Mediterranean, which is a pertinent issue due to their biological and ecological traits. We acknowledge there are limits to production models for providing realistic results if high variability is present in stock productivity or catchability have occurred during the time-series under analysis. As a species-specific example, the disappearance of European hake at the end of the 1990s, due to overfishing, is associated with competition from alien species (mainly lizardfish) and changes in hydrographic conditions in the Levantine Sea (Gücü and Bingel, 2011). As a very unique environment, the Black Sea is an important example of a severe marine ecosystem regime shift, initiated by overfishing that started in the 1950s with the removal of top predators, continued with trophic cascades and eutrophication until the 1980s, followed by an alien ctenophore *Mnemiopsis leidyi* invasion in the 1990s (Daskalov et al., 2007). The Black Sea anchovy fishery is regarded as a very good example of a fishery whose catch is greatly affected by high volatility (Gücü et al., 2017; Libralato et al., 2019). It should also be stressed that fluctuations of small pelagics such as anchovy are more likely to be heavily influenced by climatic variability than by exploitation *per se* (Salihoğlu et al., 2017; Tsikliras et al., 2018). Despite the current lack of operational state indicators for understanding pelagic fish assemblages, the MSFD clearly requires considering the state of the biotic community, the environmental status of pelagic habitats, and the functioning of pelagic components of the food web (Shephard et al., 2014). Another important point: the multi-fleet, multi-species (MF-MS) characteristics of demersal fisheries in the Mediterranean present challenges to achieving MSY for single species under the EAF (Beddington et al., 2007). For example, one species can be targeted by one gear type yet be considered as bycatch by others; or achieving an objective for one species may preclude attaining the objective for another. The fact remains that there is intense competition for resources by nations, sectors, and multi-fleets in the Mediterranean. Colloca et al. (2013) have stated that sudden changes in fishing selectivity and use of Fmsy can be difficult. Hence, the MSY objective alone is insufficient for rebuilding without implementing other supporting measures, such as increasing the size of first capture and better incorporation of the 'Ecosystem Approach to Fisheries.'

CONCLUSION

Following from this first assessment of 54 data-poor fish and invertebrate stocks exploited by Turkish fleets, we warn that if current levels of overexploitation continue unabated, only the sprat stock will remain a viable fishery in the near future. If swift action is not taken soon, many of these overexploited stocks may become too depleted to ever recover. The Turkish fisheries are plagued with excessively high fishing mortality that depletes standing stocks far below the EU and GFCM targets for sustainable fisheries and Good Environmental Status. Based on these results, to rebuild most of the remaining commercial stocks within 15 years, we suggest that fishing mortality must first be reduced to the Fmsy level, accompanied by the second proposed 40%-effort reduction as the best option. Concerning the other proposed options, the 20%-effort reduction would maximize catches over a much longer time frame, and the 50%-effort reduction would achieve sustainability goals in a minimal time frame. But the reduction of fishing mortality should be complemented with other effective management measures, especially improving MLS effectiveness and incorporation of the EAF. These mortality reduction recommendations may appear severe, but the ongoing drastic decline of marine fisheries is not a trivial matter, justifying the necessity of radical policy and intervention.

We are likely nearing the last opportunity to rebuild the Turkish marine fisheries, considering how many stocks of them are in critical condition, but many fishers will require assistance in the transition out of this sector into another for employment. It also must be stressed that the future scenario predictions presented here are likely overly optimistic considering the other anthropogenic stressors involved, such as the use of destructive fishing techniques (i.e., bottom trawling), by-catch, eutrophication, pollution, invasive species, habitat loss, sea warming, and heavy urbanization, none of which were accounted for in our modeled scenarios. Now that the current and future alarming status of Turkey's marine fisheries have been clearly presented, this opportunity should be seized upon to rebuild them for long-term viability, so that wild fish persist into the foreseeable future, valued as protein, biodiversity, biotic resistance to stressors, and an integral part of Turkish culture.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

ND, MZ, and AU collected and organized the data and wrote the manuscript. ND analyzed the data.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00103/full#supplementary-material

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A Baseline for the Blue Economy: Catch and Effort History in the Republic of Seychelles' Domestic Fisheries

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The adoption of sovereign blue bonds by the Republic of Seychelles, hereafter referred to as Seychelles, focuses on resource sustainability and illustrates options for island countries to use their ocean resources for years into the future. The fishing industry is one of the main pillars of Seychelles' economy and is of crucial importance for domestic food- and employment-security. In order to promote long-term ecological sustainability and economic viability of domestic fisheries, accurate and long-term baseline information is required. Such baseline data were derived here with a reconstruction of the Seychelles' domestic fisheries catches and fishing effort within its Exclusive Economic Zone (EEZ) from 1950 to 2017, coupled with resulting Catch Per Unit Effort data (CPUE). The total reconstructed domestic catch was approximately 1.5 times larger than the baseline as reported by the United Nations Food and Agriculture Organization (FAO) on behalf of Seychelles from 1950 to 2017 after adjustment for fully domestic catches within the EEZ. Domestic catches (i.e., excluding the large-scale industrial pelagic catches) increased by over 500% throughout the time period, growing from 1,900 t·year^{-1} in the 1950s to around 11,200 t in 2017. The major targeted taxa were jacks (Carangidae), tuna-like fishes (Scombridae) and snappers (Lutjanidae). Total fishing effort in the form of fishing capacity grew from 21,500 kWdays in 1950 to over 3.4 million kWdays in 2017. The resultant artisanal CPUE displayed a declining trend over time, suggesting a potential decline in relative abundance of fish populations within the Seychelles EEZ or targeted fishing areas.

Keywords: marine capture fisheries, blue bonds, catch reconstruction, catch per unit effort (CPUE), Indian Ocean, small-island country, small-scale fisheries, food security

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INTRODUCTION

The novel blue bonds initiative in the Seychelles has emerged as a threefold strategy to encourage and support long-term sustainability, economic growth and social equity (WorldBank, 2018b; Schutter and Hicks, 2019). In 2018, the Seychelles issued the first sovereign blue bond in history to finance marine resource sustainability and advance the ocean economy, i.e., ocean-based economic activities (WorldBank, 2018a; Cisneros-Montemayor et al., 2019). The blue economy concept describes the use of various ocean resources for economic growth (ocean economy), whilst implementing environmentally sustainable practices and social equity (Cisneros-Montemayor et al., 2019). The development of a blue economy is particularly important to small island countries, as their societies and economies often face special challenges, such as limited terrestrial resources and economic opportunities, small domestic markets, a narrow range of domestic products and high dependence on imported goods (Briguglio, 1995; Jennings et al., 1996b; Rustomjee, 2016). Driven by growth in the tourism sector, the Seychelles became a high-income island state in 2014 with low absolute poverty levels (WorldBank, 2019). Despite a considerable Gross Domestic Product (GDP) per capita increase over the last decade, income inequality persists and poverty remains of national concern (Conceição, 2019; WorldBank, 2019). The socio-economic and environmental strategies from the blue bonds could arguably[1] contribute to narrowing inequality gaps in order to advance a fairer and more inclusive society. Inequality gaps could be addressed through government incentives such as increases in the minimum wage or asset building for fishing families, e.g., home ownership (Powell, 2014).

The Seychelles' blue bonds complement a broad range of sustainability initiatives such as marine spatial planning, debt swap for conservation[2], and the World Bank's Third South West Indian Ocean Fisheries Governance and Shared Growth Project (SWIOFish3)[3]. Fisheries management is a direct focus of the blue bonds and this initiative raised US$15 million to invest in the management of fisheries, marine protected areas, and seafood value expansion, whilst promoting ocean resilience to rebuild fish stocks (Jackson, 2018). The blue bond initiative is the first of its kind, as it attempts to address sustainable use of the marine environment, improve fisheries governance, and the potential impacts that such conservation measures could have on the livelihood of Seychellois, such as displacement from traditional fishing grounds and financial hardship from temporal closures and license restrictions (IFLR, 2019). To build truly effective fisheries management systems which ensure long-term viability and minimize social costs, comprehensive and historically extensive fisheries baseline data are required to inform management (Zeller et al., 2006, 2015; Froese et al., 2017; Palomares et al., 2018). These data will provide a historical baseline for informed insights and assessments of the status of underlying marine resources.

Seychellois and their economy are highly dependent on the Indian Ocean for many marine ecosystem services, including transportation, nutrition, and tourism (Jennings et al., 1996b; Mathieu et al., 2003; WorldBank, 2018b). The large-scale industrial tuna fishing industry is a key contributor to the Seychelles' economy, providing ∼17% of the country's employment and 68% of the entire export trade (Central Intelligence Agency, 2016; WorldBank, 2017). Although these large-scale, almost exclusively export-oriented tuna fisheries make a significant revenue contribution to the economy, they are dominated by foreign fleets and foreign beneficial ownership. Thus, truly domestic small-scale fisheries remain of paramount importance for domestic food security, employment, and cultural heritage in the Seychelles (Robinson et al., 2006). Despite inequality and the historical presence of poverty malnourishment is rare in the Seychelles, which illustrates that even those Seychellois living in poverty have continued access to fish, their primary source of protein and micronutrients (Carroll, 1991; Wilson, 1994; Golden et al., 2016; Hicks et al., 2019; WorldBank, 2019). This reaffirms that the food-security provided by the domestic small-scale artisanal and accompanying take-home catch as well as "true" subsistence fisheries are an integral health, livelihood and food security component in these communities (FAO, 2015; Pauly and Charles, 2015; Pauly and Zeller, 2016a). With careful consideration, the Seychelles' blue bond initiative could provide a unique example for small island countries and emerging economies on how to improve and enhance a sustainable and profitable resource relationship (Ghina, 2003), and could be viewed as a crucial step in the right direction for fisheries policy and management (Zeller and Pauly, 2019).

Both environmental and social challenges are often created through the development of the fishing industry, as technological advancements lead to increased fishing effort, i.e., associated with economic gains, and a reduction in employment opportunities, e.g., mechanized longlines which require fewer fishers, leading to increased social inequality (Kent, 1986; Bailey and Jentoft, 1990; Miyake, 2005; Dunn et al., 2010; Zeller and Pauly, 2019). In addition, long-term sustainable practices and truly effective management has been recognized as a global necessity, due to widespread overfishing in an ultimately limited ecosystem (Jennings et al., 1996a; Robinson et al., 2004; Pauly, 2006; Payet and Agricole, 2006; Chang-Seng, 2007; Daw et al., 2012; Khan and Amelie, 2015; Pauly and Zeller, 2016a,b; Zeller and Pauly, 2019). In the Seychelles, reducing fishing pressure on marine resources has been of national interest since the 1980s (Wakeford, 2001), when concerns arose due to rapid fleet expansion and technological development, i.e., larger motorized vessels primarily focused on the traditional and vulnerable inshore fishing grounds (Grandcourt, 2002). Among attempts to reduce the exploitation of coastal resources, the government issued low-interest loans to encourage the acquisition of larger vessels with the intention of relocating fishing effort to offshore fishing areas that were thought to be more lightly exploited (Carroll, 1991). However, over time such subsidized loans led to an overall increased fishing pressure in both offshore and inshore regions and an overall higher fishing effort (Zeller and Pauly, 2019).

In 1984, the Seychelles established the Seychelles Fishing Authority (SFA), which began collecting data on landed catch, vessel registration, days fished, gears utilized and catch rates (SFA, 1988, 1996, 2005, 2006, 2014, 2017). However, despite intensive and commendable efforts by the SFA to collect and report commercial fisheries data, a comprehensive long-term and high-resolution catch and effort analysis that also considers non-commercial, e.g., subsistence, or discarded catch had not been

[1] http://blueeconomyseychelles.org/item/59-economic-inequality-in-a-high-income-country-can-the-blue-economy-mind-the-gap (accessed on 22/01/2020)

[2] These debt swap agreements are financial transactions whereby a charitable conservation trust is established and agrees to repay a portion of a nation's debt to creditors. The trust then offers more favorable lending terms to the country and directs savings on future debt repayments to fund new projects designed to protect marine life and address effects of climate change

[3] http://projects.worldbank.org/P155642?lang=en

conducted. In addition to collecting and reporting on catch data, fishing fleet statistics are highly useful for providing insights into changes in fishing effort and the status of targeted fish stocks (Davidson et al., 2014). The present study reconstructs and examines catch and fishing effort data from 1950 to 2017 for domestic marine fisheries within Seychelles' EEZ (**Figure 1**). Fishing effort data and the corresponding CPUE were reconstructed for artisanal, recreational sportfishing and industrial fisheries within Seychelles EEZ. The "true" subsistence sector was not considered here but instead take-home catch as a component of the artisanal fishery, and therefore "true" subsistence effort was not presented. The corresponding Catch Per Unit Effort (CPUE) trends were also evaluated using these catch and effort time series data. The aim was to provide a comprehensive historical baseline for each domestic fishing sector (semi-industrial, artisanal, recreational, and subsistence) over a historically and ecologically meaningful time period. The increased focus on marine conservation, sustainability, and long-term fisheries management by the Seychelles' government highlight the importance of comprehensive historical data baselines for informed decision-making.

METHODS

Total domestic marine fisheries catch for 1950–2017 was derived for the Seychelles following the reconstruction approach outlined in Zeller et al. (2016). This catch data reconstruction was built upon the official catch statistics as reported by the Food and Agriculture Organization (FAO) of the United Nations on behalf of the Seychelles, by complementing these official records with the best available time series estimates of all unreported fisheries components using secondary data from the peer-reviewed and gray literature. These secondary data, combined with conservative assumptions allow a comprehensive catch data time series to be developed that can help address existing data gaps (Pauly and Zeller, 2016b). The present study builds on a previous preliminary reconstruction for the Seychelles from 1950 to 2010 by Le Manach et al. (2015b). With newly available literature and government reports, the present study introduces take-home catch estimates since 1950, provides updates to industrial, artisanal, and recreational catches, and improves the taxonomic resolution of the catch data.

The SFA, the FAO, and the Indian Ocean Tuna Commission (IOTC) all report on particular components of Seychelles' fisheries. Data reported by the FAO are the historically longest time series of catches starting in 1950, but only represent reported landings (FAO, 2019a). These data were used as the reported baseline data to the most recently reported FAO data year (here 2017). Official landings data presented by the FAO on behalf of the Seychelles reports catches for both the Eastern (FAO area 57) and Western Indian Ocean (FAO area 51). As all of the Seychelles' EEZ waters are well within the Western Indian Ocean (**Figure 1**), the Eastern Indian Ocean landings were assumed to belong exclusively to large-scale industrial tuna and billfish vessels (Le Manach et al., 2015b). In addition, these large-scale industrial vessels, under majority-foreign beneficial ownership, caught 90% of Seychelles' Western Indian Ocean landings, which were primarily tuna and billfishes. These assumed foreign beneficial ownership tuna and billfish landings from the Eastern and Western Indian Ocean were harmonized with the FAO and the IOTC data to avoid double reporting, and treated separately in a spatially assigned *Sea Around Us* global catch database (Zeller et al., 2016; Coulter et al., 2020). These industrial tuna and billfish data can be accessed via the integrated catch dataset for the Seychelles at www.seaaroundus.org/data/#/eez/690 and www.seaaroundus.org/data/#/fao/51 and www.seaaroundus.org/data/#/fao/57. However, due to the focus on truly domestic, within EEZ fisheries, these catches were excluded from consideration in the current study, and thus, the remaining assumed domestic FAO reported data is henceforth referred to as the "adjusted" FAO reported baseline.

The SFA reports generally consist of three main sectors: industrial, semi-industrial, and artisanal. The industrial sector is the large-scale tuna and billfish fisheries with predominantly foreign beneficial ownership which was treated independently from this study (Coulter et al., 2020). The data for the industrial tuna-associated fishery are integrated and available via www.seaaroundus.org/data/#/eez/690 and www.seaaroundus.org/data/#/fao/51 and www.seaaroundus.org/data/#/fao/57. The semi-industrial and artisanal sectors are domestically owned and operated commercial fleets. The semi-industrial fleet encompasses larger longline boats that fish offshore waters within the EEZ for tuna-like taxa. The semi-industrial sector was considered the "domestic industrial" sector in the present study to align with the globally applied sectoral assignments of the *Sea Around Us* into industrial, artisanal, subsistence and recreational sectors (Zeller et al., 2016). The artisanal sector represents small-scale vessels and fishers operating primarily for commercial purposes in nearshore and inshore waters (Zeller et al., 2016). This sector differentiation is consistent with the previous preliminary reconstruction of the Seychelles by Le Manach et al. (2015b). In addition to commercial fisheries, catches from the non-commercial sectors (recreational and subsistence), not or rarely reported on by the SFA, were also reconstructed in this study. Recreational fisheries were considered exclusively sportfishing charter boat operations and this may under-represent catches due to the absence of non-charter recreational fishers. Subsistence fisheries were considered as take-home catches by commercial fishers only, which provide personal and family home consumption (Zeller et al., 2016). Thus, "true" subsistence fishing by non-commercial fishers was not accessed here. Both missing non-commercial components require further study and dedicated data collection and estimation in the Seychelles. For each fishing sector, the catch data were subdivided as either reported or unreported, landed (i.e., retained) or discarded (dead), and the best available taxonomic category was assigned. In agreement with SFA (2018), we conservatively assumed that the small-scale sectors (artisanal, recreational sportfishing, and take-home subsistence) had no discards, i.e., all catches were considered landed and retained. Discards of dead or dying catch by small-scale fisheries operating in relatively shallow water using predominantly static gears

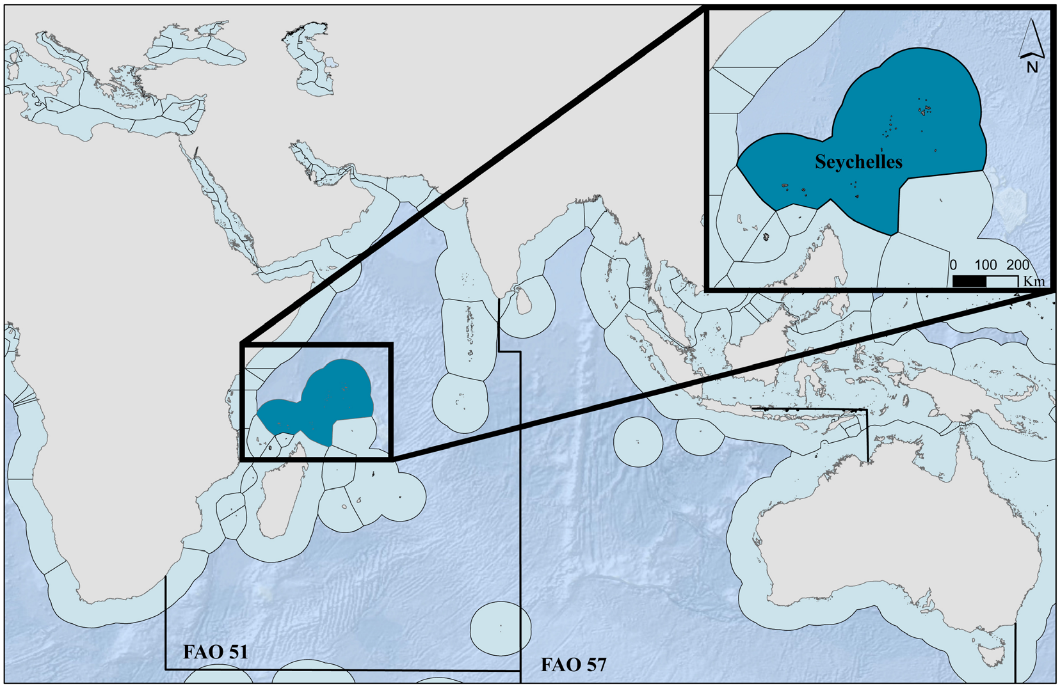

FIGURE 1 | The exclusive economic zone (EEZ) of the Republic of Seychelles.

such as traps (and even handlines) are relatively low, even if unwanted bycatch may exist, albeit with often high survival rates (Zeller et al., 2018).

Domestic Data

Artisanal

Since 1990, the SFA has reported the number of registered vessels by vessel type (pirogue, vessels with outboard motors, schooner, and whaler) and gear (handline, trap, net and miscellaneous). In order to reconstruct the vessel time series prior to 1985 (the first year of reported vessel registrations), the ratio of pirogues by gear type to the total population of the Seychelles was derived (De Moussac, 1987; Payet, 1996; Le Manach et al., 2015b). The ratio of pirogue to total population was doubled for 1950, under the assumption that pirogues were relatively more common in this period, as they were the only fishing vessel type at the time, followed by the introduction of whalers in 1958 (Wakeford, 2001), schooners in 1974 (Payet, 1996), and vessels with outboard motors in 1981 (De Moussac, 1987; Bach and Lablache-Carrara, 1991; Payet, 1996; Wakeford, 2001; Le Manach et al., 2015b). The pirogue to population ratio was interpolated from the anchor points in 1950–1985 and multiplied by the population of Seychelles in each year to estimate the number of pirogue vessels from 1950 to 1984 (Le Manach et al., 2015b; UNSD, 2018; National Bureau of Statistics, 2019). The numbers of whalers, schooners, and vessels with outboard motors were reported from 1985 to 2017, these vessel counts were extrapolated back to their known year of introduction, 1958, 1974, and 1981, respectively (De Moussac, 1987; Bach and Lablache-Carrara, 1991; SFA, 1991; Payet, 1996; Wakeford, 2001; Le Manach et al., 2015b; Assan and Lucas, 2016). Specific gear ratios were reported in 1990 (SFA, 1991) and 2015 (Assan and Lucas, 2016) for all vessel types (pirogues, whalers, schooners, and vessels with outboard motors). All vessel-specific gear ratios were interpolated between the reported ratios in 1990 and 2015, the given trends in gear ratios were carried back to the start year of each vessel type and extrapolated forward to 2017 (SFA, 1991; Assan and Lucas, 2016) to create a complete time series of the artisanal fishing fleet by vessel and gear type (see **Supplementary Table S1**) (De Moussac, 1987; Bach, 1992; Payet, 1996; Wakeford, 2001; Le Manach et al., 2015b).

Catch rates (kg·person^{-1}·day^{-1}) of landed catch per vessel type and gear type were reported for 1990 and 2015 in the SFA statistical reports (SFA, 1991, 2016). To derive a catch rate time series, the rates between these years were interpolated and extrapolated forward to 2017. For each gear type during the early time period (1950–1989), the specific gear type catch rate from 1990 was conserved back to 1970, and then linearly extrapolated to a 25% reduction in 1950 (see **Supplementary Table S2**).

Crew size and numbers of days fished were reported by vessel type for 1990 and 2015 in SFA fisheries statistical reports (SFA, 1991; Assan and Lucas, 2016). For each vessel type, the crew size, and days fished were interpolated between reported years (1990 and 2015). For pirogues and outboard motor-powered vessels, the number of days fished in 1990 was held constant back to 1950 and 1981, respectively, as pirogues were assumed to have been fishing prior to 1950, and outboard motor-powered vessels were reported as fishing full time in the year of introduction (De Moussac, 1987; SFA, 1991; Payet, 1996; Wakeford, 2001).

For whalers and schooners, the 1990 reported number of days fished was linearly carried back to an assumed full-time rate in the year of introduction, 1958 and 1974, respectively (De Moussac, 1987; SFA, 1991; Bach, 1992; Payet, 1996; Wakeford, 2001). To determine crew sizes for pirogues, whalers, schooners, and outboard motor-powered vessels, the reported crew size in 1990 was linearly interpolated to the reported 2015 crew size, and the 1990 reported crew size was held constant back to the year of known introduction for each vessel type, and the 2015 crew size was held constant to 2017 (SFA, 1991, 2016; Wakeford, 2001; Le Manach et al., 2015b.

Finally, for each vessel-gear type, e.g., pirogue-handline, the annual landed catch was calculated using the associated crew size, catch rate, and the number of days fished. For every given year, landed catch for each vessel-gear type was added together to estimate the total reconstructed annual landed catch for the artisanal fishery. This follows the procedures also used in the preliminary reconstruction by Le Manach et al. (2015b).

The adjusted FAO reported baseline (see above for exclusion of foreign beneficial ownership dominated tuna and billfish catches) was assumed to represent artisanal catch only, i.e., it was assumed that non-commercial catches (recreational and subsistence) were not included in the reported data, and thus, any difference between the reported and total reconstructed artisanal catch for any given year was assigned as unreported artisanal landings.

Sportfishing (Recreational)

The recreational fisheries data estimated here were assumed to be entirely tourism-based sport fishers on charter vessels and largely started after the airport inauguration in 1970 (Payet, 1996; Le Manach et al., 2015b). Thus, our estimates for the recreational sector are minimum estimates as they exclude any potential domestic recreational fisheries undertaken outside of charter boat operations. The number of charter vessels were assumed to start in 1970 and were reported by the SFA for the years 1985–1992 and 1996–2001, and by the IOTC in 2017 (Payet, 1996; SFA, 2005; Pepperell et al., 2017; Assan et al., 2018). These reported data indicated that the charter fleet increased rapidly from 7 vessels in 1985 to 38 in 2001 and to 168 vessels in 2017. The number of charter vessels were interpolated between anchor points for unknown years. A comprehensive report of charter vessels and landings for 1985–1990 was released by the SFA (1991); as no other SFA reports specified recreational or sportfishing catches, the data from SFA (1991) were assumed unreported to the FAO, and thus were considered here as unreported catches (Nageon et al., 2014). The 1985–1990 data points from SFA (1991) were used to estimate catch rates per charter vessel, as these rates were variable and showed a declining trend over time they were used as indicators only for the remaining time period. Due to a lack of information, the 1985 catch per vessel ($4.1\ \mathrm{t{\cdot}vessel}^{-1}$) was held constant back to 1970 and multiplied by the interpolated number of charter vessels from 1971 to 1984 to estimate the sportfishing landings in these years. The 1990 catch per vessel ($2.3\ \mathrm{t{\cdot}vessel}^{-1}$) was held constant to 2017 and multiplied by the number of reported and interpolated charter vessels from 1991 to 2017 to estimate the annual sportfishing landings.

Take-Home Catch (Subsistence)

The subsistence catches as estimated here were considered to be entirely take-home catch by artisanal fishers only. Thus, "true" subsistence fishing by non-commercial fishers has yet to be quantified. In 1994, 7% of households in the Seychelles were considered to be commercial fisher households (National Bureau of Statistics, 2012). The percent of fisher households from 1994 was maintained at 7% back to 1950, due to a lack of information on the earlier period and as a lower percentage of fisher households in early periods would be unlikely, due to the traditional reliance on this resource (Louis-Marie, 1987). In 2010, 14% of total households in the Seychelles were reported to be commercial fisher households (National Bureau of Statistics, 2012). Commercial fisher household percentages were interpolated between 1994 (7%) and 2010 (14%), and the 2010 percentage was conservatively held constant to 2017 (Le Manach et al., 2015b; UNSD, 2018; National Bureau of Statistics, 2019).

In 1950, 100% of all commercial fisher households in the Seychelles were assumed to take some catch home for self- and family-consumption. In 2010, a household survey reported that 71% of all fisher households engaged in take-home catch (UNSD, 2018). A linear interpolation of the percentage of households engaging in take-home catch was performed between 1950 (100%) and 2010 (71%), and the trend was carried forward to 2017. The number of total households were reported in 1971, 1977, 1987, 1994, 1997, 2002, and 2010 (National Bureau of Statistics, 2012, 2017) and were utilized to estimate the average family size. This trend in family size was extrapolated back to 1950, and forward to 2017. Family size was multiplied by fisher households engaged in take-home catch to estimate the total number of people consuming take-home catch.

As there were no independent studies for pre-1990, the per capita consumption study by Wilson (1994) with 85 $\mathrm{kg{\cdot}person^{-1}{\cdot}year^{-1}}$ was assumed as the take-home catch consumption for 1950, and this was interpolated to 57 $\mathrm{kg{\cdot}person^{-1}{\cdot}year^{-1}}$ in 2017 as indicated by the WorldBank (2017) to derive a complete time series. The number of people assumed to be consuming take-home catch from 1950 to 2017 (based on family size) was multiplied by the consumption rate time series. Due to the systematic absence of take-home and/or "true" subsistence catch data from the SFA statistical reports, all take-home catches were assumed to be unreported.

Semi-Industrial (Domestic Industrial)

The semi-industrial fishery in the Seychelles, here considered "domestic industrial" in line with the *Sea Around Us* database structure (Zeller et al., 2016), started in 1995 and historically targeted billfish, sharks, and tuna in offshore EEZ waters (SFA, 1991; Kolody et al., 2010; Le Manach et al., 2015b). Due to the relatively small fleet size (5 active vessels in 2016), all semi-industrial landings were assumed to be reported by the SFA (2016) and therefore to the FAO. However, discards were assumed to be unreported, as these are explicitly excluded from FAO data requests (FAO, 2019a) but specifically included by the *Sea Around Us* (Zeller et al., 2016, 2018). Estimates of discarded catch were derived as a proportion of the reported landings from this sector. The discards considered here were based on targeted catch damaged by shark and cetacean depredation (Le Manach et al., 2015b; Rabearisoa et al., 2018). Depredation rates to targeted tuna/billfish catches ranged from 11 to 30% between 1995 and 2000 (SFA, 1996, 1998, 2002; Le Manach et al., 2015b); these rates were applied as is and the 2000 rate of 21% was carried forward to 2017. Any potential discards of non-depredated but unwanted and unmarketable species were not estimated here but require further research. Our discards estimates are therefore likely conservative.

Taxonomic Composition

The taxonomic composition of reported landings (semi-industrial and artisanal sectors), was based on the adjusted FAO baseline data, however, the SFA statistical reports from 1990 SFA (1991), 2013 (Assan and Lucas, 2015), and 2015 (SFA, 2005) had more detailed taxonomic information. Utilizing these detailed SFA reports, the FAO taxonomic categories were disaggregated further. The FAO taxonomic categories were disaggregated by linear interpolation between the taxa reported within each category in 1990 SFA (1991) and an average of the taxa reported within each category in the 2013 and 2015 SFA reports (SFA, 1991; Assan and Lucas, 2015, 2016). The taxonomic disaggregation within FAO categories was conserved (i.e., not altered) from the SFA anchor points for pre-1990 and post-2013. A report on sharks by Nevill et al. (2007) had distinct artisanal and industrial commonly caught shark species, this disaggregation was held constant for reported "sharks and rays" for the entire time period due to the lack of earlier studies on shark composition in the commercial fisheries catch. As the FAO had limited taxonomic disaggregation in the early time period, the taxonomic breakdown for "marine fishes nei" was estimated to be the average reported landings for 1983–2017 by the FAO (2019b) for the entire time period (Le Manach et al., 2015b).

For the unreported catch component estimated for the artisanal sector, the interpolated SFA taxonomic breakdown from 1990 to 2015 was applied (SFA, 1991; Nevill et al., 2007; Assan and Lucas, 2016; SFA, 2016). For 1950–1989 the SFA (1991) disaggregation from 1990 was held constant, and for 2016–2017 the disaggregation from Assan and Lucas (2016) was carried forward (see **Supplementary Table S3**).

The taxonomic composition for the estimated sportfishing catches was based on the reported sportfishing landings from 1990 (SFA, 1991) and anecdotal[4] reports of targeted reef fish (FAO, 2005; Martín, 2011; Breuil and Grima, 2014; Nageon et al., 2014; Pepperell et al., 2017). This disaggregation was held constant for the entire period this fishery was active, i.e., 1971–2017 (SFA, 1988, 1991). See **Supplementary Tables S4** for details.

For the taxonomic composition of the estimated take-home catch, the combined reported and unreported artisanal landings composition (see above) was utilized, but adjusted for the exclusion of high-value taxa, i.e., tunas, lobster, sea cucumber, and sharks (see **Supplementary Table S5**). Thus, lower-value fish were assumed to be the most likely to be used for self- and family-consumption, with high-valued taxa generally sold. This assumption was supported by the high prevalence of poverty affecting fisher families (Muller et al., 2016).

TABLE 1 | Fishing capacity of the artisanal, recreational sportfishing, and semi-industrial fishing sectors of the Republic of Seychelles.

Sector	Length (m)	Engine power (kW)	Days fished	Motorization
Artisanal	4.5	0.89[a]	175	No
	4.5	9.11	175	Yes
	11.3	58.72	175	Yes
Sportfishing	11.3	58.72	104	Yes
	20.0	185.72	104	Yes
Semi-industrial	20.0	185.72	175	Yes

Revised from Greer et al. (2019).
[a] *Engine power based on manpower (Greer et al., 2019), adjusted to 2.4 persons per un-motorized vessel (SFA, 1991).*

Fishing Effort Data

The Seychelles' total fishing effort for its domestic fleet, i.e., excluding the large-scale tuna/billfish fleets with predominantly foreign beneficial ownership (Coulter et al., 2020), was estimated following the global fishing effort method described in Greer et al. (2019) based on the fishing capacity of the artisanal, sportfishing, and semi-industrial fisheries. Since the take-home catch was estimated as a component of the artisanal fishery rather than an independent truly subsistence fishery, a dedicated fishing effort for the take-home catch was not estimated. The fishing effort for each sector was calculated based on the number of vessels, length, motorization (kW), days fished, and type of gear utilized in individual fleets.

The total number of fishing vessels operating in each fleet were those used to estimate the landings described above. Vessel size, motorization, and days fishing data from the *Sea Around Us* global fishing effort database were updated with local information and sources as outlined below (**Table 1**). Artisanal vessels were assigned an average length of 4.5 m or 11.3 m, classified as either unmotorized (i.e., traditional pirogues), or motorized vessels. Charter (sportfishing) vessels were assigned an average length of 11.3 or 20.0 m and semi-industrial vessels were 20.0 m, all motorized (**Table 1**). Engine power in kW was associated with the length of motorized vessels and gear used, while unmotorized vessels were allocated an engine power equivalent based on manpower conversion equivalent to motorization (Greer et al., 2019). Commercial (artisanal and semi-industrial) fishers were assumed to spend 175 days per year fishing, and tourist charter boats were assumed to be fishing recreationally for 104 days per year. Final effective fishing effort (kWdays) was estimated for each individual fishing fleet from 1950 to 2017 by combining the number of vessels, engine power, and days fished.

Catch per Unit Effort (CPUE)

Estimated Catch Per Unit Effort (CPUE) time series were derived for the artisanal, recreational sportfishing, and semi-industrial sectors in an attempt to derive a first-order indicator of likely relative abundance trends of the underlying fish stocks within the Seychelles. The CPUE per sector was derived by dividing the

[4] http://www.seychellesnewsagency.com/articles/6076/Seychelles+to+evaluate+status+of+fish+stock+with+new+tagging+project+in+; https://www.charterworld.com/index.html?sub~=~seychelles-fishing-charter

reconstructed catch time series by the estimated fishing effort from 1950 to 2017 for each sector, excluding the large-scale industrial tuna and billfish fleets dominated by foreign beneficial ownership (Coulter et al., 2020).

Data Uncertainty

Data uncertainty scores in the form of data reliability scores for the catch data were assigned by fishing sector for four time periods (1950–1969, 1970–1989, 1990–2009, and 2010–2017). Uncertainty around, or trust in the data derived here, both reported and unreported, was estimated following a scoring approach adopted from the Intergovernmental Panel on Climate Change (IPCC) as outlined in Zeller et al. (2016). Uncertainty scores range from 1 to 4, representing highest uncertainty (±50%) to lowest uncertainty (±10%) (see **Supplementary Table S7**).

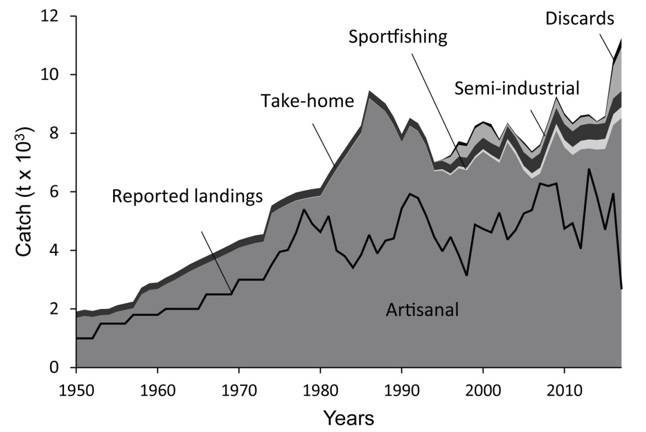

FIGURE 2 | Total reconstructed domestic catch for Seychelles from 1950 to 2017 within Seychelles' EEZ by fishing sector. Landings as reported by FAO on behalf of the Seychelles are represented as the black overlay line. Industrial large-scale tuna and billfish fisheries are excluded and addressed separately in the global spatial tuna database (Coulter et al., 2020).

RESULTS

The reconstructed annual domestic catches, excluding the large-scale industrial tuna and billfish fleets dominated by foreign beneficial ownership (Coulter et al., 2020), across all the sectors considered here increased by more than 500% over the 1950–2017 time period, from 1,900 t in 1950 to 11,200 t in 2017 (**Figure 2**). From 1950–1986, the reconstructed catches increased by approximately 200 t·year^{-1} to the first time series peak of 9,200 t in 1986, and then declined until the 2000s, before increasing again, eventually reaching 11,200 t in 2017. Of the total reconstructed catch, 38% was deemed unreported, with the unreported proportion decreasing from nearly 50% of total catches in 1950 to an average of around 30% in the early 1990s (**Figure 2**). Since 1995, the unreported proportion fluctuated with an average of 38% until 2016, when the adjusted reported FAO baseline data suddenly declined by 45%, resulting in an unreported proportion of 63% for 2017.

The artisanal sector dominated the domestic fisheries within the Seychelles' EEZ and represented over 90% of total truly domestic catches since 1950. The artisanal catch exhibited an increasing trend from 1950 to 1986 when it peaked with an annual catch of 9,200 t (**Figure 2**). From 1987 to 2017 the artisanal catch stabilized with interannual variability at an average of 7,500 t·year^{-1}.

The recreational sportfishing sector steadily increased since its commencement in 1971 and represented around 4% of the entire Seychelles domestic catch in 2017 (**Figure 2**). Sportfishing landings increased from around 60 t in the mid-1990s to almost 400 t in 2017. The total sportfishing catch from 1971 to 2017 was 4,900 t, all of which was considered unreported.

The subsistence take-home consumption increased from 200 t in 1950 to 500 t in 2017 and had an average of 310 t·year^{-1} across the entire time period. The total take-home catch from 1950 to 2017 was 21,400 t, all of which was considered unreported.

The semi-industrial sector increased over time, contributing 16% to the total catch in 2017, compared to 5% when it commenced (**Figure 2**). It displayed particularly strong growth in the last two years. The semi-industrial fishery had an average catch of 330 t·year^{-1} until 2015, however, in 2016 the annual catch increased fourfold with an average catch of 1,600 t·year^{-1} for 2016–2017. This fishery produced almost 1,700 t of discards, i.e., 17% of its landings, over the active time period (1995–2017), which were all deemed unreported.

Taxonomic Composition

The catch composition across all four sectors comprised over 90 taxa, of which 65 were at the informative species level, accounting for 53% of the reconstructed catch. This represents 15 taxonomic categories in the data reported by the FAO on behalf of the Seychelles. The major family was Carangidae, representing 27% of the total catch, with the dominant taxa being *Carangoides* spp. and *Caranx sexfasciatus* (**Figure 3**). The second most important family was Scombridae, which accounted for 23% of the total catch, with *Rastrelliger kanagurta* and *Rastrelliger* spp. being the main taxa (**Figure 3**). Lutjanidae accounted for 16%, with *Aprion virescens* and *Lutjanus* spp. being the most caught taxa. Siganidae and Lethrinidae each represented 8% of the total catch, and Serranidae and Sphyraenidae nearly 4% of the total catch each.

For the artisanal and take-home catch, Carangidae, Scombridae, and Lutjanidae were the main families and represented over 60% of the total catch (**Supplementary Tables S3, S5**). The sportfishing sector primarily targeted Scombridae (*Euthynnus affinis* and *Thunnus albacares*) and Istiophoridae (*Istiophorus platypterus*), these taxa combined represented more than 70% of the targeted sportfishing taxa (see **Supplementary Table S4**). The semi-industrial sector primarily targeted Scombridae with *T. albacares and T. obesus* which represented 38% of the total catch, followed by *Xiphias gladius* 37% and Elasmobranchii 20% (see **Supplementary Table S6**).

Fishing Effort

The total effective fishing effort, here estimated as annual fishing capacity, has grown from 21,500 kWdays in 1950 to over 3.4

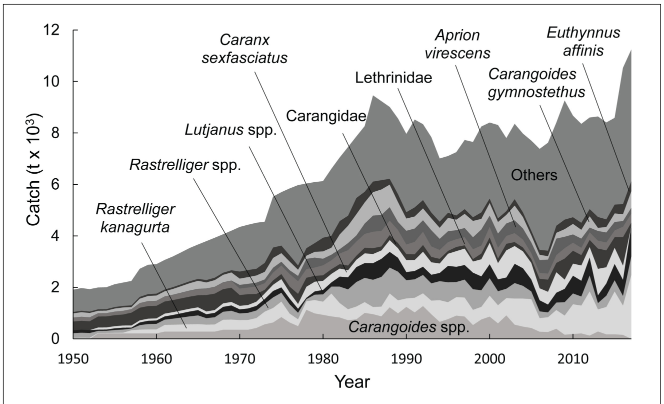

FIGURE 3 | Taxonomic composition of reconstructed total catches for Seychelles for 1950–2017. The category "Others" consists of 82 additional taxa with minor contributions to the overall catch.

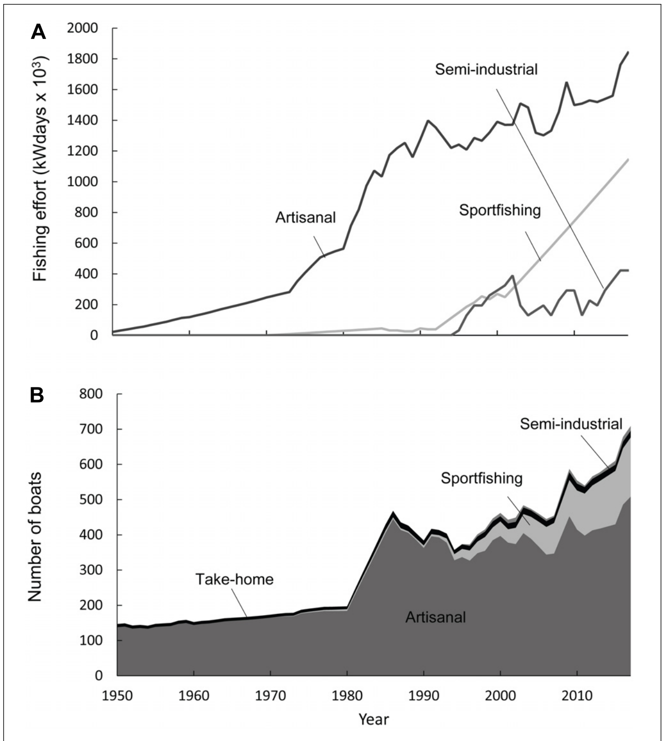

FIGURE 4 | Fishing effort for the domestic fisheries in the Seychelles from 1950 to 2017 as **(A)** effective fishing effort (fishing capacity); and **(B)** total number of vessels by sector. Take-home catch was estimated as a component of the artisanal fishery rather than an independent fishery so the effort for the take-home catch was not estimated.

million kWdays in 2017 (**Figure 4A**). In addition to the gradual increase in the total number of vessels, from 147 vessels in 1950 to 708 vessels in 2017 (**Figure 4B**), effective fishing effort has also increased due to the introduction and growth in vessel motorization. Artisanal (and take-home component) fishing effort represented 100% of the total effective effort in 1950 but declined to 53% in 2017 (**Figure 4A**). This percent decline was primarily due to the establishment and rapid growth of the sportfishing sector and more recently the semi-industrial sector. The sportfishing sector began in 1971 with an estimated 2,800 kWdays of effort and grew to over 1.1 million kWdays in 2017 (**Figure 4A**). The semi-industrial sector began in 1995 and by 2017 accounted for 12% of the total effort (**Figure 4A**).

Catch per Unit Effort (CPUE)

The artisanal CPUE steeply declined from an average of almost 80 kg·kWdays^{-1} in the early 1950s to approximately 14 kg·kWdays^{-1} in the 1970s, followed by a more gradual decline to an average of 5 kg·kWdays^{-1} since the 1990s (**Figure 5A**). This CPUE decline appeared to be driven by the introduction of motorization which led to an increase in effective fishing effort. The sportfishing CPUE displayed a steadily declining trend from around 0.6 kg·kWdays^{-1} in 1971 to around 0.3 kg·kWdays^{-1} by 1990 and has remained constant since (**Figure 5B**). The semi-industrial CPUE was approximately 1.6 kg·kWdays^{-1} from 1995 to 2015 but increased to around 3.7 kg·kWdays^{-1} by 2016 (**Figure 5B**).

DISCUSSION

The reconstructed domestic fisheries catches of the Seychelles for 1950–2017, excluding the large-scale industrial tuna and billfish fisheries with predominantly foreign beneficial ownership (Coulter et al., 2020), were more than 1.5 times the adjusted official statistics as reported by the FAO on behalf of the Seychelles. Such a difference between reported data and likely total catches is not unusual and can be seen across many Indian Ocean small island countries and territories, e.g., catches for Mauritius, La Reunion and Mayotte were 2.4, 1.6 and 1.4 times the data reported by FAO on behalf of these entities, respectively (Boistol et al., 2011; Doherty et al., 2015b; Le Manach et al., 2015a). Non-reporting and under-reporting in small -scale fisheries, especially for non-commercial sectors, is widespread globally (Zeller et al., 2015; Smith and Zeller, 2016).

Despite commendable efforts by the SFA to improve catch statistics by collecting information from different fishing sectors, non-commercial fisheries (recreational and subsistence) were consistently not included in the national reports and therefore not reported to the FAO, despite FAO requesting such data (Garibaldi, 2012), a situation that unfortunately is common throughout the globe (Pauly and Zeller, 2016a). The unreported sectors estimated here (sportfishing and take-home catch) represent nearly 10% of the entire domestic catch in 2017, and catches from these sectors have the potential to increase given the economic opportunities associated with the sportfishery (Mwebaze and MacLeod, 2013). In this study, recreational catches were assumed to be sportfishing from the tourism sector only. The strong reliance of many island countries on the tourism industry creates not only economic opportunities (Cisneros-Montemayor and Sumaila, 2010) but also environmental, social, and economic challenges (Pattullo, 1996; Bhola-Paul, 2015). In addition to tourism-based sportfishing, it is possible that the improved socio-economic conditions in recent years may

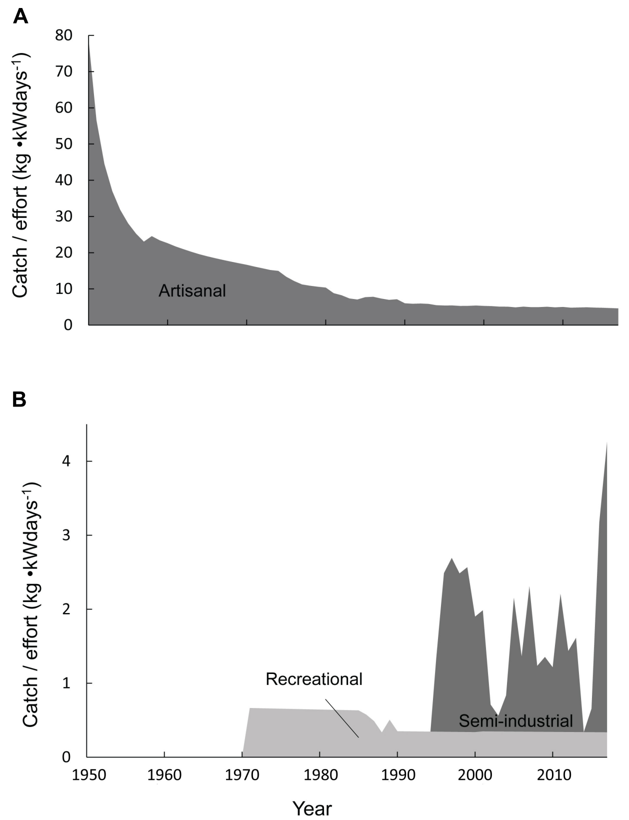

FIGURE 5 | Catch-per-unit-effort (CPUE) for the domestic fisheries in the Seychelles from 1950 to 2017, for **(A)** artisanal; and **(B)** recreational sportfishing and semi-industrial sector.

have created local recreational fishing interests not addressed in this study. Such a potential underestimation of catches for the recreational sector will need examining in future research. To reduce the amount of catch considered unreported, previous fisheries management plans which included detailed monitoring of recreational fisheries should be reactivated in future management plans and such estimates reported to FAO, ideally after adjustment for of the presentist bias (Zeller et al., 2018).

Furthermore, trends in "true" subsistence and take-home catch could be assessed by regular national surveys with questions regarding take-home catch and time spent fishing only for subsistence purposes. These self-consumption fisheries are especially relevant as they are strongly linked to socio-economic indicators in small island countries (Kronen et al., 2010). Given that our estimates of recreational and subsistence only focused on tourism-based sportfishing and commercial take-home catch, respectively, makes our estimates conservative minimum estimates. The total recreational and subsistence catches are therefore likely larger than 10% of the total catches and require further detailed study and management focus.

In addition to unreported landings, discarded catch is also not included in the official catch statistics reported by the FAO on behalf of countries. While this exclusion is in line with FAO's data request (Garibaldi, 2012), it should be considered problematic in today's age of ecosystem-based fisheries management considerations. Depredation of targeted catch by sharks and toothed whales in the Seychelles' semi-industrial longline fishery are amongst the highest (>22%) in the world (Rabearisoa et al., 2018). The discards estimated here were a conservative representation (20.5%) of these rates (Le Manach et al., 2015b).

The artisanal sector dominated the total domestic catch, representing over 90% of total catches over the entire time period. Despite the relatively low economic value when compared to the foreign-owned and operated industrial fishery for high-value tuna, the artisanal fishery is arguably the most important fishery component for the Seychelles' blue bonds initiative to focus on when developing management strategies. The artisanal fishery plays a vital role in Seychelles' domestic food security and local livelihoods, whilst being less wasteful in terms of negligible discards and having lower fuel use and lower CO_2 emission intensity (Zeller and Pauly, 2019). In addition, the linkage between resources and economy suggests that economic growth measured via market-based GDP alone (Jerven, 2013), such as from large industrial fisheries does not reflect the strong dependency on limited natural resources that massively contribute to local food security and human well-being (Costanza et al., 1997; Robinson et al., 2004; Costanza et al., 2014; Zeller and Pauly, 2019; Kalimeris et al., 2020). Seafood represents the primary source of protein for Seychellois, resulting in a high rate of marine fish consumption of 57 $kg \cdot person^{-1} \cdot year^{-1}$ (WorldBank, 2017), despite 21% of household survey respondents reporting an inability to afford to buy sufficient fish and meat (Muller, 2014; National Bureau of Statistics, 2016). In addition, fishers represent the most destitute population sector in the Seychelles, highlighting the paramount importance of artisanal fishing and associated livelihood development (Muller et al., 2016). The Seychelles is highly dependent on foreign imports, e.g., canned meats, and as a result, is particularly susceptible to inflation and global market fluctuations, which can also affect the domestic fish market (Larose, 2003; Philpot et al., 2015). In 2008, the global financial crises triggered a fivefold increase in the inflation rate causing a 60% increase in the price of fish; this rise in cost encouraged ordinary non-fishing locals to engage in fishing endeavors, intensifying fishing effort (Clifton et al., 2012).

Overall, total domestic catch increased steadily since 1950, coupled with intensified fishing effort driven by the expansion of fishing grounds and an increase in fleet size and motorization. The increasing catch trend has been observed also in other island entities such as the Comoros (Doherty et al., 2015a), Maldives (Hemmings et al., 2014), and Fiji (Zylich et al., 2012). The total artisanal catch showed an increasing trend since 1950 until it peaked in 1986, likely associated with government incentives to stimulate the artisanal fisheries (government loans), and modernization of fishing fleets and technology (Wakeford, 2001). Since 2015, the number of outboard motor-powered vessels increased by 28%, indicating that modernization and fleet expansion continues. Despite relatively high reporting by the SFA in previous years, the reporting rate has dropped since 2016, resulting in nearly 70% of the total domestic catch for 2017 likely unreported. The drop in reporting was exemplified by the disappearance of catches of carangids (jacks) in the official statistics reported by the FAO on behalf of the Seychelles, previously one of the most prominent reported groups. The disappearance of the officially reported Carangidae family with no disaggregation to carangid species is unlikely to reflect the disappearance of these taxa from the fisheries and the disappearance was not conserved in the estimation of the unreported component. This change in reporting must be investigated as incorrect trends could improperly influence new blue bond management strategies. In addition, national non-tuna SFA data were revealed to be more taxonomically detailed than the FAO non-tuna data, suggesting a substantial loss in taxonomic resolution between national records (SFA) and data reported by the FAO. This crucial loss of information between national data and FAO records has been observed in numerous other countries (Pauly and Zeller, 2016a). As the artisanal fisheries represent the majority of Seychelles domestic catch, species-specific information is of high importance when making well- informed policy and management plans (Mees and Rousseau, 1997; Pauly and Charles, 2015; Robinson et al., 2020).

While catch reconstructions conducted and led by the *Sea Around Us* over the last 15+ years for every maritime country in the world (Pauly and Zeller, 2016a,b) have attracted some criticisms (e.g., Garibaldi et al., 2014; Chaboud et al., 2015; Ye et al., 2017), these were either shown to be misguided (Al-Abdulrazzak and Pauly, 2014; Belhabib et al., 2015), or a miss-directed pre-occupation with the uncertainty associated with reconstructed data (Pauly and Zeller, 2017). The overall uncertainty range for the Seychelles domestic reconstructed catch data presented here was considered to be up to $\pm$ 40%, and the estimates presented likely underestimate the actual catch given the conservative assumptions made throughout (Pauly and Zeller, 2016a, 2017).

Three different patterns in the CPUE by sector were observed over the entire time period. The artisanal fishery showed a strong decline in CPUE from almost 80 kg·kWdays^{-1} in 1950 to nearly 5 kg·kWdays^{-1} in 2017. The increasing number of fishing vessels and motorization over time intensified the pressure on fish stocks, somewhat reflected in the declining CPUE trend for the artisanal fishery. The relatively flat trend in sportfishing CPUE from 1971 to 1985 and from 1990 to 2017 reflects the data sources and methods used and thus, this trend should be interpreted with caution as it may not indicate the true trend. The semi-industrial long-line fishery displayed considerable CPUE variability, likely due to the strong fluctuation in pelagic catch with relatively stable effort. This is not unusual amongst semi-industrial fisheries due to the high temporal and spatial variability of pelagic species in the open ocean (Ward and Hindmarsh, 2007). The increasing fishing effort trends appear to be strongly influenced by the introduction of outboard motors in the artisanal fleets in the early 1980s, and the increase in the number of vessels within the existing fishing fleets. Despite a slight reduction in effort in relation to the number of boats since 1991, the CPUE has not shown signs of recovery in the small-scale fisheries (**Figure 5A**). The stable low CPUE since 1985 for small-scale sectors could be associated with shifting the targeted demersal species to those previously less exploited or an overall demersal stock depletion (Jennings et al., 1995; SFA, 1996; Nageon and de Lestang, 1998; Spalding and Jarvis, 2002). Since the mid-1980s, fishing effort has steadily increased while CPUE has declined. The Seychelles management strategy to attempt and encourage to re-distribute fishing effort to offshore grounds via financial incentives have done little to halt CPUE declines (Wakeford, 2001), suggesting at least potential failure of this policy, as has been shown globally (Swartz et al., 2010; Zeller and Pauly, 2019).

Given Seychelles' unique opportunity of blue bonds, the accurate assessment of the health of fish populations for fisheries management is critical. Strong monitoring and accurate reporting of all fisheries sectors, including non-commercial sectors, i.e., subsistence and recreational, is crucial to provide accurate data for underlying fisheries stock assessments to ensure long-term resource sustainability and consequently food security. Species and/or stock-specific monitoring and stock assessments are of the utmost importance for effective management plans for sustainable fisheries. While traditional stock assessments are technically challenging and financially problematic for most developing countries and small island entities due to the need for expensive secondary data, novel and easily applicable data-limited stock assessment methods with low data requirements are now readily available. Adopting these well-established data-limited assessment methods will allow valuable insights into stocks with less exhaustive sampling, allowing for fisheries management to promote local sustainable practices (Froese et al., 2017, 2018, 2019).

The opportunity of the blue bonds strategy brings financial funds that enable further research to promote informed decisions regarding fisheries management, but also the design and establishment of other management, livelihood, and conservation strategies, such as marine protected areas, or non-extractive tourism opportunities, could result in improvements for the economy and environment. With such implementation of novel initiatives, the Seychelles fisheries would benefit from refinement and increased frequency of current data collection systems, such as household surveys in addition to annual fisheries reports (Zeller et al., 2015). Given the historical value of the small-scale fisheries as demonstrated here, all small-scale sectors should be highlighted in the blue economy roadmap of the Seychelles along with future research and strong regulations (Secretariat Commonwealth, 2018; Techera, 2019). Small-scale fisheries management could alleviate some significant social inequality and resource overexploitation challenges, and improve the overall quality of life for many Seychellois.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request from the corresponding author and will be freely available at www.seaaroundus.org/data/#/eez/690 during 2020.

AUTHOR CONTRIBUTIONS

HC and RW collaborated to complete the data synthesis and analysis and drafted, reviewed, and edited the manuscript. GV and LH edited the manuscript. DZ conceptualized catch reconstructions, advised on methods, guided analysis, reviewed, and edited the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00269/full#supplementary-material

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Growing Into Poverty: Reconstructing Peruvian Small-Scale Fishing Effort Between 1950 and 2018

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Small-scale fisheries are globally marginalized by management institutions; thus, they have to endure the consequences of ineffective regulations, environmental uncertainty, social traps and market inequity. Small-scale fisheries in Peru, one of the world's leading fishing countries, are important contributors to national employment, food security and gross domestic product. Yet, relatively little is known about these fisheries and their evolution, except for the fact that the Peruvian small-scale fleet size is rapidly increasing. Here, we reconstructed small-scale fishing effort across time and developed several indicators using it to assess changes in the fleet's fishing efficiency and economic performance. Segmented regression analysis was used to identify statistically significant breakpoints and changes in their trajectories between 1950 and 2018. Our results suggest that fishing effort has strongly increased, and at much faster rates than the catches, particularly since 2006. The combined effect of these trends results in significant declines in the fleet's ratio indicators (i.e., catch per unit of effort, revenue per unit of effort, and fisher's incomes relative to Peru's minimum wage), suggesting that the growing fishing effort is unsustainable and uneconomic. The behavior of these indicators differs within the fleet, depending on the vessel's main fishing method. Most small-scale fishers are currently living in relative poverty. Yet, fishers using the least selective fishing gears, or engaged in illegal fishing, had the most stable incomes over the past decade. These findings are discussed in detail by exploring the social, legal and economic drivers fostering fleet growth. Finally, a list of general recommendations aimed at improving fisheries sustainability and fisher's wellbeing was produced, based on the local context, fisheries literature and common sense.

Keywords: small-scale fisheries, fishing effort, catch per unit of effort, revenue per unit of effort, relative income, uneconomic growth, fisheries enforcement, Peru

INTRODUCTION

Small-scale fisheries are globally marginalized by management institutions, and thus have to endure the consequences of ineffective regulations, environmental uncertainty, social traps and market inequity (Pauly, 2006; Salas et al., 2007; Finkbeiner et al., 2017; Chuenpagdee and Jentoft, 2019). Peru has one of the world's largest fisheries catch (FAO, 2018), although most of it is anchoveta (*Engraulis ringens*), a low-value fish mainly caught by industrial vessels and used overwhelmingly for fishmeal production (Gutiérrez et al., 2017). However, most of Peru's marine landings used for direct human consumption (i.e., used to partially satisfy local seafood demand and supply international seafood markets) are caught by the local small-scale fleet (Christensen et al., 2014). According to national regulations, this fleet is composed of small vessels (i.e., total length $\leq$ 15 m, hold capacity $\leq$ 32.6 m^3), equipped with one or multiple manually operated fishing gears, that target marine living resources for commercial purposes (SPDA, 2019).

As in other developing countries (Schuhbauer and Sumaila, 2016), Peruvian small-scale fisheries play an important role in the national economy. In 2009, 54 thousand people were employed as small-scale fishers generating a revenue of 0.61 billion USD (Christensen et al., 2014). Moreover, as their landings provide raw materials for the secondary and tertiary sectors of the economy, 2.2 jobs and 3.5 USD were additionally generated in seafood value chains for every job and dollar made at sea (Christensen et al., 2014). However, small-scale fisheries remain relatively understudied, poorly regulated and subjected to ineffective enforcement of input (e.g., fishing areas and seasons, fleet size, and vessel dimensions) and output (e.g., total allowable catches and minimum-landing sizes) controls (Sueiro and De la Puente, 2015; Gutiérrez et al., 2016; Monteferri et al., 2017; SPDA, 2019).

Recent studies show that the Peruvian small-scale fleet is growing rapidly (Castillo et al., 2018), and now targets a more diverse portfolio of species over fishing grounds that are expanding both geographically and bathymetrically (Marín et al., 2017). This has allowed their catches (in volume) to increase over time, even when the average length and annual landings of traditionally targeted coastal species follow declining trends (CEDEPESCA, 2013; Mendo and Wosnitza-Mendo, 2014). Although these are pressing concerns, several factors have contributed to dulling their relevance in the public eye. These include, but are not limited to: the relative size of the small-scale fishery in comparison to the industrial fishery, the high environmental variability of the Peruvian marine ecosystem, the large number of stakeholders involved in addressing small-scale fisheries management, the limited resources allocated to strengthening and enforcing regulations, and the lack of clear objectives and indicators to assess the success of management strategies over time (Sueiro and De la Puente, 2015).

The reported increases in fishing effort observed around the world (Anticamara et al., 2011; Greer et al., 2019) continue to raise concerns about the sustainability of the targeted resources and the wellbeing of peoples that depend on them for their livelihoods and nutrition (Watson et al., 2012; Link and Watson, 2019). Thus, this paper aims at reconstructing small-scale fishing effort in Peru, seeking to highlight the extent at which it has grown over time and assess its impact on the fishery's performance and fishers' economies.

MATERIALS AND METHODS

Reconstructing Small-Scale Fishing Effort

Peruvian small-scale fishing effort was reconstructed following the *Sea Around Us* fisheries data reconstruction framework (Zeller and Pauly, 2016), aiming to improve the local resolution of previous attempts that sought to estimate global fishing effort (Greer et al., 2019). This process included: identifying and sourcing official and alternative sources of time series data on the number of vessels in the Peruvian small-scale fleet and their characteristics; developing data 'anchor points' in time using all available information; interpolating data for the periods between anchor points; estimating small-scale fishing effort based on validated predictor variables; and constructing confidence intervals for the fishing effort reconstruction over time, by scoring uncertainty in data sources, assumptions and methods used (Zeller and Pauly, 2016; Greer et al., 2019).

Two indicators were used to estimate annual fishing effort: nominal and effective effort. Nominal effort in year t (nE_{t}) is the product of the number of boats in the fleet (N_t), their average capacity (P_t) and the average number of days they spend fishing during the year (D) (Greer et al., 2019).

$$nE_{\mathrm{t}} = N_{\mathrm{t}} \times P_{\mathrm{t}} \times D_{\mathrm{t}} \tag{1}$$

Effective effort in year t (E_{t}) is the product of nE_{t} and a technological creep factor (Tc_{t}). The latter accounts for the progressive increases in fishing power that result from improvements in gear design, fish detection and catch handling methods (Belhabib et al., 2018; Palomares and Pauly, 2019).

$$E_{\mathrm{t}} = nE_{\mathrm{t}} \times Tc_{\mathrm{t}} \tag{2}$$

Local stakeholders and regulators describe fishing operations, and administer the small-scale fleet based upon vessel type, gear and target species (Christensen et al., 2014; Sueiro and De la Puente, 2015; Marín et al., 2017; SPDA, 2019). Thus, this reconstruction aimed at segregating fishing effort by fishing gear and vessel type. This required: (a) estimating changes in small-scale fleet size over time; (b) defining subgroups within the small-scale fleet; (c) estimating vessel capacity from total length; and (d) developing working hypotheses for the extent of their technological creep.

Estimating Changes in Small-Scale Fleet Size Over Time

Data anchor points for the total number of small-scale vessels in Peru were extracted from the literature (**Table 1**). Fleet size was interpolated linearly between anchor points except for two periods: (i) 1988–1995, when ancillary data on 'vessels year of construction,' retrieved from INEI (2012), were used to reconstruct annual fleets size increments, and (ii) 2016–2018

TABLE 1 | Data sources used for the reconstruction and uncertainty 'scores' used for evaluating the quality of the small-scale fleet size time series, with their corresponding confidence intervals, based on Mastrandrea et al. (2010) and Zeller and Pauly (2016).

Data score	Scoring criteria	Confidence interval		Years were applied	Sources of data
		– %	+ %		
Very high	High agreement and robust evidence	10	10	1997–2012	Wosnitza-Mendo, 1992; Escudero, 1997; Estrella et al., 2006, 2010; Marín et al., 2017
				2015	Castillo et al., 2018
High	High agreement and medium evidence or medium agreement and robust evidence	20	20	1982	Wosnitza-Mendo, 1992
				1986–1996	Reconstructed using data from INEI (2012)
				2013–2014	Linear interpolation
Low	High agreement and limited evidence or medium agreement and medium evidence or low agreement and robust evidence	30	30	1953–1969	FAO, 2018; Greer et al., 2019
				1970–1980	Berrios, 1983
				1981	Linear interpolation
				1983–1985	FAO, 2018; Greer et al., 2019
				2016–2018	Projection
Very low	Low agreement and low evidence	40	40	1950	Caravedo, 1979
				1951–1952	Linear interpolation

when fleet size was projected assuming a conservative growth rate of 3% year^{-1}. All vessels reported in fleet size surveys or census were assumed to be in operation, unless otherwise stated in the source data. Confidence intervals were assigned to annual fleet size estimates based on the level of uncertainty in data sources (i.e., very high, high, low, and very low reliability), in accordance to previous fisheries data reconstructions (**Table 1**).

Defining Subgroups Within the Small-Scale Fleet

The Peruvian small-scale fleet is composed of several sub-fleets simultaneously operating at sea (Castillo et al., 2018), and each sub-fleet (e.g., small-scale longliners) is composed of métiers or combinations 'vessel types' and 'fishing methods' (Christensen et al., 2014).

Small-scale vessels were grouped into four categories, or vessel types, based on their length class and propulsion system (i.e., VT1: non-motorized vessel with $L < 8$ m; VT2: non-motorized vessel with $8 \text{ m} \leq L \leq 15$ m; VT3: motorized vessel with $L < 10$ m; VT4: motorized vessel with $10 \text{ m} \leq L \leq 15$ m) (Greer et al., 2019). Anchor points for the number (or percentage) of motorized vessels in the fleet were extracted from literature (Caravedo, 1979; Wosnitza-Mendo, 1992; Estrella et al., 2006, 2010; INEI, 2012; Castillo et al., 2018). Vessel motorization was linearly interpolated between anchor points and projected to 2018 by applying the average annual rate of increase estimated for the 2012–2015 period.

In 2012, a census of small-scale fishermen and their vessels was conducted along the Peruvian coast (INEI, 2012). Using vessel-specific information extracted from this dataset (e.g., its year of construction, total length and propulsion system), we were able to estimate, on annual time steps, the proportion of active motorized vessels according to their length (i.e., VT3 and VT4), and the average vessel length for all vessel types between 1950 and 2012. Annual average vessel lengths by vessel type were projected from 2013 to 2018 by using a 5-year moving average starting with the 2008–2012 period. The rate of change of the proportional contribution of each vessel type to the total fleet exhibited between 2008–2012 was used to project the number of vessels by vessel type between 2013 and 2018 (Castillo et al., 2018).

Next, métiers were defined by allocating fishing methods to vessel types. The main fishing methods used by Peruvian small-scale vessels across the time series include: (i) gillnets, (ii) handlines, (iii) hands or tools (i.e., compressed air divers), (iv) longlines, (v) purse seine nets, (vi) squid jigs, (vii) traps, and (viii) trawl nets (Sueiro and De la Puente, 2015; Marín et al., 2017; Castillo et al., 2018).

Small-scale vessels in Peru can change fishing gears seasonally or use more than one fishing gear at the time (Sueiro and De la Puente, 2015). However, most vessels use a single or main fishing method throughout the year (Estrella et al., 2010). Hence, we assumed that vessels only used one fishing method per year. Annual estimates of the total number of small-scale vessels using individual fishing methods were available for 1982 (Wosnitza-Mendo, 1992), 1996 (Escudero, 1997), 1997–2012 (Estrella et al., 2006, 2010; INEI, 2012; Marín et al., 2017), and 2015 (Castillo et al., 2018). The relative contribution of each fishing method to the total fleet, and vessel types, were estimated for these anchor points. An additional artificial anchor point for these parameters was constructed based on descriptions of the small-scale fleet around the start of the time series (Caravedo, 1979; Coker, 1910). Data were then linearly extrapolated between anchor points (i.e., 1951–1981, 1983–1996, and 2013–2014). Time series were then projected to 2018 using a 3-year moving average (starting from 2013 to 2015) of each methods' proportional contribution to the total fleet. The proportion of vessels using each fishing method by vessel type were carried forward without change (i.e., if 20% of VT3 used gillnets in 2015, we assumed that 20% of VT3 also used gillnets in 2016, 2017, and 2018).

Finally, five assumptions were used to formulate a working hypothesis of the evolution of non-motorized métiers over time: (I) non-motorized vessels only used handlines and gillnets across the time series, (II) the use of gillnets by non-motorized vessels decreased over time and was highest at the start of the time series, (III) the use of handlines by VT2 increased over time, (IV) the easiest path for fishers seeking to become new vessel owners is to obtain a non-motorized vessel and equip it with handlines, and (V) non-motorized vessels equipped with gillnets have a higher likelihood of producing larger yields, allowing their vessel owners to acquire an engine and transition to a different vessel type over time (at a faster rate than unmotorized vessels using handlines).

Estimating Vessel Capacity From Total Length

Annual average vessel capacity (P_t) by métier (in kW $\cdot$ vessel^{-1}) was approximated using the estimated annual average vessel length (L) by vessel type. For métiers using motorized vessels, P_t was inferred from L (in meters) through previously validated constants, such that $P = 0.436 \times L^{2.021}$ (Anticamara et al., 2011; Greer, 2014). Alternatively, constant P_t values were assigned for métiers using non-motorized vessels consistent with those found in published literature (i.e., VT1: P_t = 0.37 kW $\cdot$ vessel^{-1}; VT2: P_t = 0.75 kW $\cdot$ vessel^{-1}; Greer et al., 2019).

Developing a Working Hypothesis for the Technological Creep by Métier

The Peruvian small-scale fleet experienced a slow technological creep over time resulting in larger fishing areas (i.e., increases in engine power), longer fishing trips (i.e., greater use of isothermal boxes with ice or insulated holds) and reductions in the time spent searching for target species (i.e., increased cellphone coverage and/or usage of sounders and GPS navigators) (Alfaro-Shigueto et al., 2010; Estrella and Swartzman, 2010; Sueiro and De la Puente, 2015; Marín et al., 2017; Castillo et al., 2018).

Technological creep trajectories were heterogeneous across métiers (**Table 2**). Annual creep factors (Tc_t) might seem 'conservative' in comparison to other studies (Palomares and Pauly, 2019). However, they were developed based on the local history of the fleet and exclude increases in fishing power associated with increases in vessel size. The latter were directly incorporated in our calculations through the vessels' capacity (P_t).

Non-motorized vessels and their operations have changed very little over time (Coker, 1910; Sueiro and De la Puente, 2015), thus retaining the same Tc_t between 1950 and 1999. Similarly, technological investment by motorized vessels were restricted between 1950 and 1979 due to fishers' limited purchasing power (Caravedo, 1979; Miranda, 2016). Nonetheless, the amount and size of the fishing gear carried by motorized vessels (e.g., the length and number of longlines) started to increase across métiers between 1980 and 1999, with mechanized winches becoming more commonly used by small-scale purse seiners (Sueiro and De la Puente, 2015). Additionally, increases in the use and area of coverage of cellphones led to a faster technological creep across all métiers between 2000 and 2018, and particularly in the last decade. However, the much faster creep experienced by vessels using longlines or squid jigs reflects their investments for improving hold insulation and gaining access to sounders and GPS navigators (INEI, 2012; Marín et al., 2017; Castillo et al., 2018).

Estimating Fishing Effort at Different Scales

Effective and nominal fishing effort time series were estimated for each métier (Eqs. 1–2). The number of days spent at sea per year (D_t), used for these computation were approximated from published literature (**Table 3**; Alfaro-Shigueto et al., 2010; Estrella and Swartzman, 2010; Sueiro and De la Puente, 2015; Marín et al., 2017), and were consistent with those used in global studies (Anticamara et al., 2011). Annual effort estimates were added across all métiers using the same fishing method to compute nE_t and E_t at the sub-fleet level, and across all sub-fleets to estimate the small-scale fleet's total fishing effort. The units for fishing effort are kW $\cdot$ days (Belhabib et al., 2018).

Estimating Catch per Unit of Effort

Catches were divided by nominal and effective fishing effort to produce time series of nominal and effective catch per unit of effort with annual time steps ($nCPUE_t$ and $CPUE_t$, respectively). $nCPUE_t$ and $CPUE_t$ were estimated for each sub-fleet and for the total small-scale fleet. The units for these indicators are expressed in kg $\cdot$ kW^{-1} $\cdot$ days^{-1}. Catch data used in these calculations was extracted from the *Sea Around Us* database.[1]

Peruvian catch data included in the *Sea Around Us* database was reconstructed for the period between 1950 and 2018 using the methods described in Mendo and Wosnitza-Mendo (2014). The starting point of the reconstruction process is data reported by FAO. These are considered 'nominal catches' that are corrected, using ancillary sources of information, so that the 'total reconstructed catch' incorporates previously unreported data (e.g., discards and IUU fishing) (Zeller and Pauly, 2016).

[1] www.seaaroundus.org

TABLE 2 | Annual rates of increase in relative technological power assumed to have been experienced by different métiers in the Peruvian small-scale fleet over time.

Period	Non-motorized vessels (% $\cdot$ year^{-1})	Motorized vessels using hands or tools, handlines, gillnets, traps, or trawl nets (% $\cdot$ year^{-1})	Motorized vessels using longlines or squid jigs (% $\cdot$ year^{-1})	Motorized vessels using purse seine nets (% $\cdot$ year^{-1})
1950–1979	1.0	1.0	1.0	1.0
1980–1989	1.0	1.5	1.5	2.0
1990–1999	1.0	1.5	1.5	2.0
2000–2009	2.0	2.0	3.0	2.5
2010–2018	2.0	2.5	3.5	3.0

TABLE 3 | Additional parameters used to estimate small-scale fishing effort and fishers' incomes, segregated by fishing method.

Fishing method	Days spent fishing (year^{-1})	Crew size (number of fishers)	Fishers' income as a percentage of vessel revenue (%)
Gillnets	200	1–4	50
Handlines	250	1–3	60
Hands or tools (divers)	180	3–5	40
Longlines	290	4–6	40
Purse seine nets	190	5–8	35
Squid jigs	270	4–7	29
Traps	120	3–6	30
Trawl nets	180	5–7	10

For more detailed information on the sources of data used in the Peruvian catch reconstruction, see **Supplementary Material**.

Catches corresponding to the artisanal sector of Peru (i.e., small-scale fisheries) were extracted from the total reconstructed catch, which also includes catches corresponding to the industrial, recreational and subsistence sectors (Mendo and Wosnitza-Mendo, 2014). The artisanal catch was then distributed among fishing gears following the methods described by Cashion et al. (2018). Data on species catches by gear was available for some years of the time series (e.g., 1986–1988, Wosnitza-Mendo et al., 1988; 1996–2012, Marín et al., 2017). These were used as anchor points. The proportion of the catch caught by a given fishing method was estimated for all taxa in years when data was available. These proportions were used together with fishing effort estimates, by sub-fleet responsible for a taxon's annual catch, to determine taxon-specific gear preferences. These were used to infer the distribution of the catch among fishing methods, using fishing effort, for years lacking catch by gear by species data. Confidence intervals for the reconstructed catch data were estimated following the methods described in Zeller and Pauly (2016).

Assessing the Socio-Economic Consequences of Changes in Small-Scale Fishing Effort

The value of the small-scale catch (R_t) was estimated using official off-vessel price data for the Peruvian small-scale fleet produced by the Instituto del Mar del Perú[2] and the reconstructed catch data retrieved from the *Sea Around Us* database. Given the shorter length of the off-vessel price time series, R_t was only estimated for the 2009–2018 period. Price data, originally expressed in Peruvian 'Nuevos Soles,' was converted to United States Dollars using the official exchange rate and then corrected for inflation by dividing it by the Consumer Price Index (IPC, acronym in Spanish) for food items. The official exchange rates and IPC were extracted from the Central Reserve Bank of Peru (BCRP, acronym in Spanish) online databases.[3] The value of R_t was estimated for each sub-fleet and for the whole small-scale fleet and is expressed in real 2009 USD · year^{-1}.

Two indicators were used to assess the socio-economic impact of changes in fishing effort on small-scale fishers: (a) the revenue per unit of effort ($RPUE_t$), computed by dividing R_t by E_t, serving as a proxy for the economic efficiency of the fleet and sub-fleets; and (b) fishers annual incomes relative to minimum wage (rI). The value of rI was computed for each sub-fleet such that:

$$rI_{jt} = R_{jt} \times p_j / n_{jt} \times CS_j \tag{3}$$

Where: rI_{jt} is the average fishers income using fishing method j in year t, p_j is a constant representing the proportion of the vessel's revenue allocated to paying fishers' salaries [retrieved from Christensen et al. (2014) and Sueiro and De la Puente (2015)], n_{jt} is the number of vessels in the sub-fleet, and CS_j is the typical crew size per vessel within the sub-fleet (Estrella and Swartzman, 2010; Sueiro and De la Puente, 2015; Marín et al., 2017). Values for p_j and CS_j are included in **Table 3**. If rI_{jt} falls below the minimum wage (i.e., $rI_{jt} < 1$), it can be assumed that fishers are unable meet the minimum level of living standards compared to other Peruvians and are thus living in 'relative poverty' (Hagenaars and van Praag, 1985).

Assessing Performance Indicators Over Time

Significant breakpoints in nE_t, E_t, $nCPUE_t$, and $CPUE_t$ time series, reflecting changes in trends (i.e., slope) over time, were identified by analyzing the time series trajectories for these parameters using segmented regression (Muggeo, 2003, 2008). The significance of linear trends in $RPUE_t$ within the 2009–2018 period, were assessed using simple regression analysis. All analyses and figures were done in R (Ver. 3.6.3).

RESULTS

A Growing Small-Scale Fishery

The Peruvian small-scale fleet has strongly increased in size over the last seven decades (**Figure 1A**). In 2018, fleet size was estimated to be 5.4 times larger than in 1950. Growth rates, however, were not constant over the years. The fleet grew at a 'moderate' rate of 220 vessels · year^{-1} between 1950 and 1961, but decreased almost as fast between 1962 and 1971 (−180 vessels · year^{-1}). Growth was limited between 1972 and 1990 (70 vessels · year^{-1}). However, the fleet then started to grow much faster (i.e., 1991–2005: 220 vessels · year^{-1}), and grew fastest between 2006 and 2010 (1,390 vessels · year^{-1}). Although growth rates more than halved since, the fleet still grew by 640 vessels · year^{-1} from 2011 to 2018.

Additionally, vessel motorization increased substantially, from 5% in 1950 to 91% in 2018 (**Figure 1A**). The number of new engine-powered vessels entering the fleet or existing boat being fitted with engines, grew steadily across the time series, except for the period between 1982 and 1996. During these 15 years of

[2]www.imarpe.gob.pe

[3]www.bcrp.gob.pe

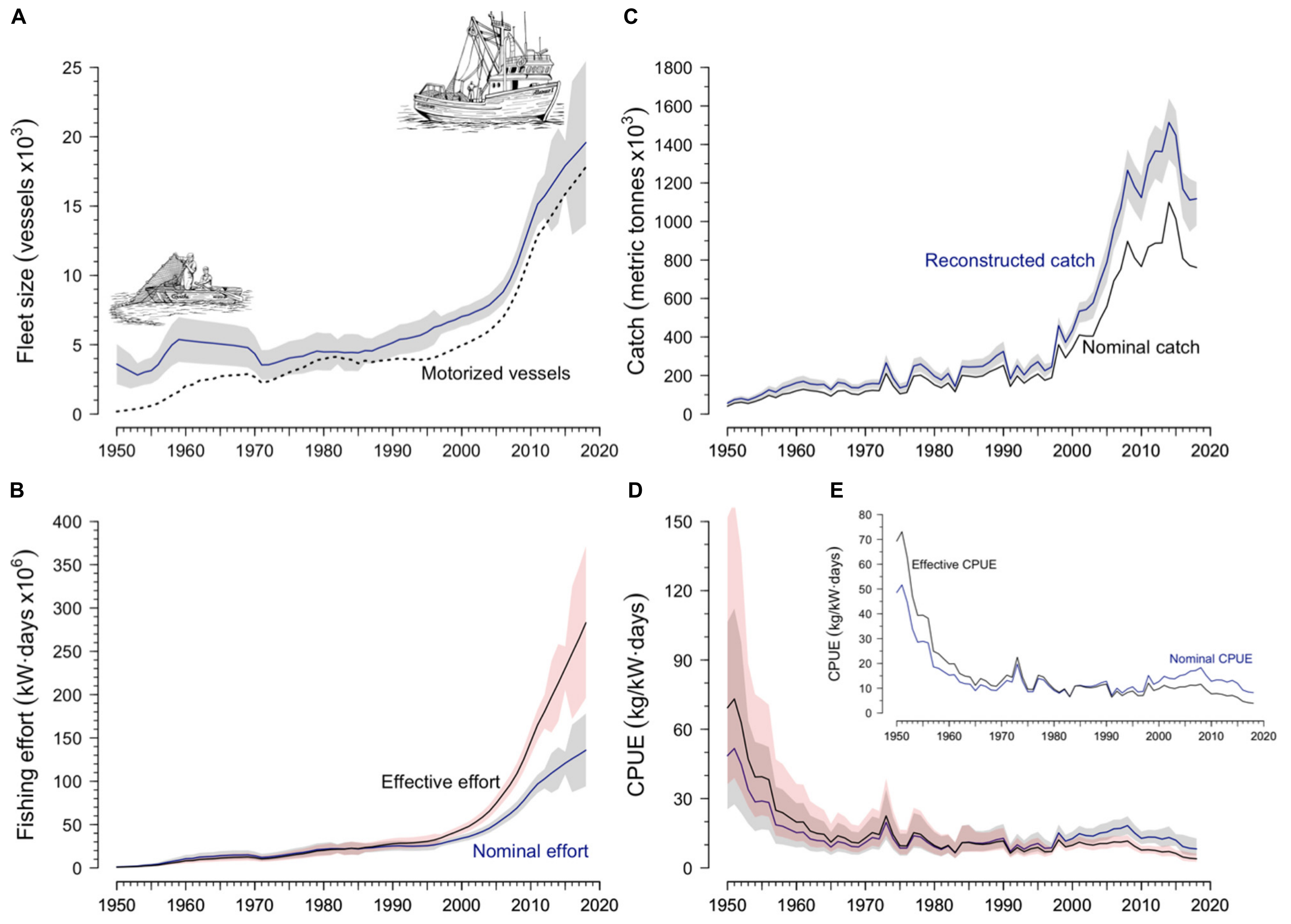

FIGURE 1 | Evolution of Peruvian small-scale fisheries between 1950 and 2018, highlighting changes in: **(A)** vessel numbers and vessel motorization, **(B)** nominal and effective fishing effort, **(C)** catches, and **(D,E)** nominal and effective catches per unit of effort. Polygons denote confidence intervals for reconstructed data. **(D,E)** Thus show the same CPUE trends only differing on whether they include of confidence intervals or not.

limited growth in total fleet size, a relatively large number of non-motorized vessels entered the fleet. Nonetheless, since 1997 the temporary reduction in vessel motorization was overturned.

Moreover, the composition of the fleet by length class also changed in favor of larger vessels. For example, only 2% of the fleet was composed of motorized vessels with total lengths larger than 10 m (VT4) between 1950 and 1969; this type of vessels represented 20% of the fleet between 2000 and 2018.

The reported increases in fleet size, vessel motorization and vessels' total length have contributed to strongly increasing nominal and effective fishing effort (nE_t and E_t, respectively) across the studied period (**Figure 1B**). Fishing effort grew fastest after the turn of the century, and more so if the technological creep was also considered (e.g., nE_t was four times larger in 2018 than in 2000; E_t was two times larger than nE_t in 2018, but only 30% larger than it in 2000).

As expected, increases in fishing effort resulted in greater catches, at least for part of the time series (**Figure 1C**). Reconstructed catches, accounting for the unreported landings and discards, were on average 33% (±2%) larger than the reported landings (i.e., nominal catch) over time. Growth in catches was moderate between 1965 and 1996 (3,761 tons · year^{-1}), and much faster between 1997 and 2014 (74,237 tons · year^{-1}). Catches surpassed the million tons mark in 2007 and have not fallen below this mark since. However, after reaching their peak value in 2014, reconstructed catches exhibit a rapidly declining trend.

For more information regarding: (i) the evolution of the fleet by vessel type, (ii) changes in nE_t and E_t over time by fishing method, (ii) statistically significant breakpoints in effort trends, and (iii) changes in their slopes, see **Supplementary Material**.

Declining Fishing Efficiency

Nominal and effective catches per unit of effort ($nCPUE_t$ and $CPUE_t$, respectively) significantly declined over time (**Figure 1D**). Breakpoints and changes in slope were consistent between both indicators (**Figure 1E**). For information regarding $nCPUE_t$ trajectories, see **Supplementary Material**.

Overall $CPUE_t$ declined until the early 1990s, decreasing fastest at the start of the time series (1950–1959) and at slower rates since 1960. In 1993 the trend reversed, and until 2006 $CPUE_t$ increased at slow annual rates. In 2007 the trend again reversed itself, with $CPUE_t$ declining until the end of the time series (**Table 4**).

The trend in $CPUE_t$ experienced by the small-scale fleet was not shared by all sub-fleets (**Figure 2** and **Table 4**). Vessels using gillnets, handlines and purse seine nets show consistent declines in $CPUE_t$, with steep declines in the first decades and much gentler declines since. As with the previous group, vessels fishing with longlines show declines in $CPUE_t$ at the beginning of the studied period and later (early 1990s) reverse the trend with faint increases in $CPUE_t$. Contrary to this trend is that shown by vessels using hands or tools (i.e., divers) and traps, whose $CPUE_t$ trajectories start with positive slopes, followed by declining trends. Finally, vessels using trawl nets and squid jigs show two trend reversals in their $CPUE_t$ trajectories. They, start with negative slopes, followed by a relatively short period where $CPUE_t$ increases and then by a second period of declining $CPUE_t$.

TABLE 4 | Summary indicators of the segmented regression analysis of effective Catch per unit of Effort ($CPUE_t$) time series.

Fishing methods	Breakpoints (±*SE*)†	Slope (±*SE*)‡	*p*-value§	Adjusted R^2
Gillnets	1950 (±0)	−9.32 (±0.45)	4.55E-29*	0.9734
	1958.33 (±0.33)	−0.85 (±0.47)	4.98E-26*	
	1979.91 (±2.03)	0.31 (±0.25)	1.35E-05*	
	1994.02 (±3.21)	−0.40 (±0.23)	3.58E-03*	
Handlines	1950 (±0)	−2.61 (±1.18)	3.06E-02*	0.6582
	1954 (±1.96)	−0.75 (±1.25)	1.41E-01	
	1961.99 (±2.81)	0.03 (±0.41)	5.89E-02	
	2000.99 (±3.07)	−0.45 (±0.13)	3.18E-04*	
Hands or tools (Divers)	1950 (±0)	0.84 (±0.31)	9.15E-03*	0.8636
	1967 (±1.36)	4.43 (±0.76)	1.40E-05*	
	1976.22 (±0.68)	−2.84 (±0.76)	9.63E-14*	
	1993.07 (±1.54)	−0.29 (±0.36)	1.37E-09*	
Longlines	1950 (±0)	−20.28 (±1.46)	1.43E-20*	0.9609
	1951.15 (±0.07)	−1.05 (±1.46)	1.99E-19*	
	1960.19 (±0.82)	−0.12 (±0.13)	3.90E-09*	
	1990.92 (±2.53)	0.09 (±0.03)	1.17E-08*	
Purse seine nets	1950 (±0)	−26.66 (±2.02)	1.50E-19*	0.9308
	1955.44 (±0.28)	−0.55 (±2.02)	4.02E-19*	
	2005.62 (±2.39)	4.59 (±11.94)	6.68E-01	
	2007.09 (±2.09)	−0.50 (±11.96)	6.72E-01	
Squid jigs	1990 (±0)	−0.88 (±0.38)	3.08E-02*	0.9536
	1997.89 (±0.28)	9.49 (±1.16)	1.42E-08*	
	2002 (±0.27)	−3.17 (±1.16)	4.37E-10*	
	2010 (±1.15)	−1.04 (±0.5)	3.13E-04*	
Traps	1950 (±0)	0.02 (±0.02)	2.98E-01	0.7594
	1986.63 (±1.87)	0.47 (±0.06)	6.99E-06*	
	2006.42 (±0.87)	−2.55 (±2.15)	4.86E-05*	
	2008.11 (±0.69)	0.31 (±2.15)	6.35E-01	
Trawl nets	1970 (±0)	−20.17 (±2)	1.50E-12*	0.8877
	1978.55 (±0.55)	1.64 (±2.27)	6.34E-12*	
	1990 (±2.6)	−2.70 (±1.79)	1.99E-02*	
	2000.22 (±3.43)	0.23 (±1.54)	6.48E-02	
All methods	1950 (±0)	−6.38 (±0.36)	1.87E-25*	0.9596
	1958.56 (±0.35)	−0.27 (±0.37)	2.62E-24*	
	1993.07 (±3.56)	0.26 (±0.21)	1.58E-02*	
	2006.47 (±2.47)	−0.66 (±0.32)	5.01E-03*	

'Breakpoints' reflect changes in the slope of the regression (in years). 'Slope' is the rate of change in $CPUE_t$ between breakpoints. The 'p-values' reflect the statistical significance of the regression, whilst the 'Adjusted R^2' reflects the proportion of the variation in slope explained by the regression. Breakpoints and slopes are presented next to their respective standard errors (SE). †Units in years. ‡Units in (kg/kW · days) · year^{--1}. §Statistically significant p-values (p < 0.05) are indicated with an asterisk (*).

Moreover, fishing efficiency differed substantially across small-scale fishing methods. When comparing sub-fleets' average $CPUE_t$ between 2009 and 2018, three groups emerge: sub-fleets using gillnets (1.7 ± 0.1 kg · kW^{-1} · days^{-1} · year^{-2}), handlines (1.7 ± 0.2 kg · kW^{-1} · days^{-1} · year^{-2}), and longlines (2.5 ± 0.2 kg · kW^{-1} · days^{-1} · year^{-2}) have relatively low fishing efficiency; sub-fleets using traps (4.4 ± 0.3 kg · kW^{-1} · days^{-1} · year^{-2}) and hands or tools (4.8 ± 0.5 kg · kW^{-1} · days^{-1} · year^{-2}) are moderately efficient; and finally, vessels using squid jigs (9.5 ± 0.7 kg · kW^{-1} · days^{-1} · year^{-2}), trawl nets (11 ± 0.8 kg · kW^{-1} · days^{-1} · year^{-2}), and purse seine nets (13.1 ± 0.4 kg · kW^{-1} · days^{-1} · year^{-2}) are the most efficient.

Uneven Economic Performance Within the Small-Scale Fleet

Changes in the economic performance of the small-scale fleet depends on: (1) the annual catch, (2) the species composition of the catch, (2) the prices of landed items, (3) the percentage of the sub-fleets' revenue used for paying fishers salaries (Christensen et al., 2014), and in this case, the Peruvian minimum wage as well.

Catches by sub-fleet were quite stable between 2009 and 2018, except for vessels using trawl nets, whose catch grew significantly ($p = 0.005$) over the decade at an average rate of 3,995 tons · year^{-1}. Vessels using squid jigs and purse seine nets, were responsible for most of the small-scale catch during this period (42 and 36%, respectively), whilst sub-fleets using longlines (7%), hands or tools (5%), gillnets (5%), trawl nets (2%), handlines (2%), and traps (1%) were minor contributors in comparison.

However, things are different in terms of revenue. Small-scale fisheries directly generated a total annual average revenue of $902 million · year^{-1} (±$49 million · year^{-1}) between 2009 and 2018. Vessels equipped with purse seines were responsible for 34% of the total small-scale revenue over the decade, followed by those using squid jigs (17%), longlines (13%), hands or tools (12%), trawl nets (11%), and gillnets (9%). As with total catches, vessels fishing with handlines (3%) and traps (1%) were only minor contributors to the fleet's total revenue.

As expected, due to the observed $CPUE_t$ trajectory, the fleet's $RPUE_t$ significantly declined between 2009 and 2018 ($p = 0.0054$) (**Figure 3**). $RPUE_t$ declines were also significant for vessels using longlines ($p = 0.0001$), divers (hands or tools; $p = 0.004$), gillnets ($p = 0.0055$), and handlines ($p = 0.008$), but not for those fishing with purse seine nets, squid jigs and traps, whose $RPUE_t$ was

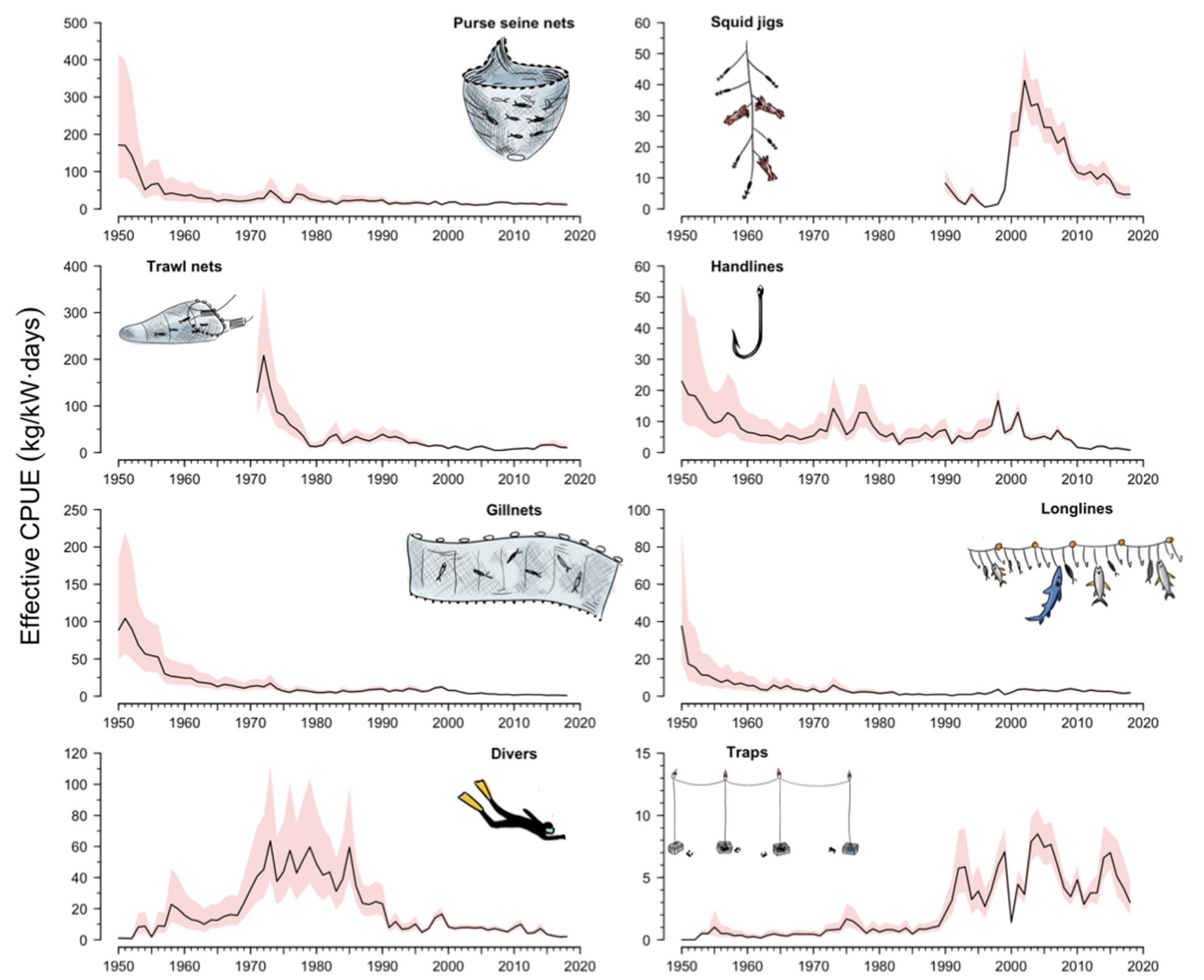

FIGURE 2 | Effective catches per unit of effort (CPUE) of the Peruvian small-scale fleet between 1950 and 2018. Data is presented segregated by fishing method. Polygons denote confidence intervals for reconstructed data.

relatively stable, nor for vessels using trawl nets, whose increase in $RPUE_t$ was almost statistically significant.

These trends affect small-scale fishers' wellbeing in an uneven manner. Looking at their annual incomes relative to Peru's minimum wage (*rI*) reveals an alarming scenario (**Table 5**). Fishers of only two sub-fleets are doing well. Trawl fishers' *rI* has increased since 2013, being over 6 times larger than Peru's minimum wage in 2017 and 2018; while *rI* for fishers using purse seine nets has been consistently above the minimum wage (roughly 2 twice as large) throughout the decade. However, all other small-scale fishers' relative incomes show reductions over time. Annual earnings by trap fishers, fell below the minimum wage for 9 years within the decade. Squid jiggers' *rI* were smaller than the minimum wage between 2012–2013 and 2015–2016. Handliners' and gillnetters' *rI* have been consistently below the minimum wage since 2015. Finally, longliners' *rI*, although showing a 60% decline over the last decade, never went below the minimum wage.

For more information on the catch by sub-fleet and off-vessel prices by target resource, in real US dollars, see **Supplementary Material**.

DISCUSSION

Growing Into Poverty

Unsustainable Fleet Growth

Peruvian small-scale fisheries are experiencing a dangerous and resilient pathology where uncontrolled fleet growth is directly reducing fishing efficiency and fishers' wellbeing. This growth in fishing effort is unsustainable (Pauly, 2009). It is important

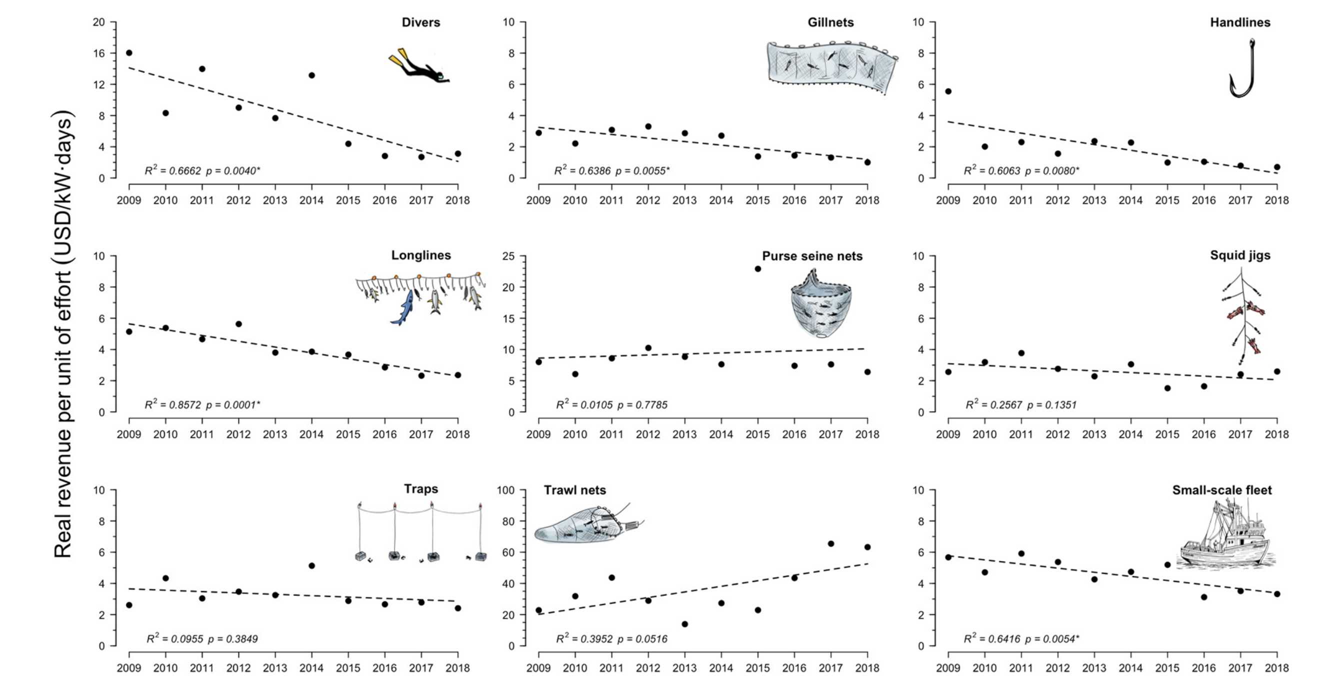

FIGURE 3 | Trends in revenue per unit of effort of the Peruvian small-scale fleet between 2009 and 2018. Points reflect annual estimates in real USD per kW · days. Black lines represent linear trend extracted from the data. The *p*-values reflect the statistical significance of the regressions, whilst the adjusted R^2 reflects the proportion of the variation in slope explained by the regression.

TABLE 5 | Small-scale fishers' annual average incomes relative to the Peruvian minimum wage.

Fishing methods	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Gillnets	1.76	1.16	1.43	1.26	1.21	1.22	0.73	0.72	0.65	0.49
Handlines	1.68	0.96	1.19	0.72	1.20	1.23	0.64	0.65	0.50	0.44
Hands or tools (Divers)	7.08	3.18	4.79	2.53	2.34	4.24	1.66	1.01	0.95	1.08
Longlines	3.59	3.24	2.52	2.51	1.88	2.02	2.27	1.68	1.36	1.38
Purse seine nets	2.99	1.87	2.25	2.22	2.12	1.97	6.89	2.10	2.17	1.83
Squid jigs	1.42	1.53	1.61	0.98	0.92	1.31	0.78	0.80	1.17	1.25
Traps	0.71	1.01	0.62	0.59	0.60	0.98	0.65	0.55	0.58	0.50
Trawl nets	2.96	3.47	4.19	2.26	1.22	2.52	2.45	4.39	6.53	6.25
Peruvian annual minimum wage (in real 2009 USD)	2064	2460	2832	3540	3300	3216	2820	3060	3180	3300

Relative incomes below the minimum wage are highlighted in red.

to clarify that we do not claim that all stocks targeted by the small-scale fleet are overexploited, but rather that current effort levels are excessive for catches to be sustained, let alone to enable a recovery of overexploited stocks. This assertion is supported by the declining trajectory of the effective catch per unit of effort ($CPUE_t$) observed across the time series (**Figure 1**). It showcases that more effort is now required to capture the same amount of fish over time, and hence that less marine living resources are currently available for the growing fleet. A scenario that is further warranted by: (i) the expanding fishing grounds of the fleet (Marín et al., 2017), and (ii) the reduced importance of traditional target species in the catch (Mendo and Wosnitza-Mendo, 2014).

There are several caveats to the use of ratio estimators (e.g., catch per unit of effort) as indicators of relative abundance for targeted stocks. For example, catches can decline in areas that are not representative of the stock's overall distribution, and if only focused on these, declines in CPUE would be unrealistically rapid and unrepresentative of the stock's abundance. This phenomenon is known as hyperdepletion (Walters, 2003).

However, our analysis is not focused on individual stocks. The Peruvian small-scale fleet is part of a multi-species fishery: (i) where most boats target coastal resources but whose fishing area keeps expanding further offshore (Estrella and Swartzman, 2010; Marín et al., 2017), (ii) which lacks effective input and output (SPDA, 2019), (iii) in which most fishers have decades (Castillo et al., 2018) and a vast understanding of the temporal and spatial distribution of resources, i.e., productive fishing areas (Sueiro and De la Puente, 2015). Technological improvements and fishers' growing experience should be increasing the fleet's catchability

(Palomares and Pauly, 2019). Additionally, the fleet's expansion into previously unfished areas should be generating increasing catches if resource abundance within them were high (Hilborn and Walters, 1992). Thus, by pooling together all small-scale catch and effort data within the Peruvian EEZ we should expect the converse of hyperdepletion, i.e., hyperstability in $CPUE_t$. Yet we observe a declining $CPUE_t$ trajectory, and declining catches as well (**Figure 1**).

Nonetheless, statistically significant increases in both fleets' $CPUE_t$ and E_t were registered between 1993 and 2006 (**Table 4**). These favorable conditions can perhaps be explained by the combined effect of: (i) a change in primary productivity and oceanographic conditions (e.g., a regime shift), potentially increasing the carrying capacity for multiple traditionally targeted coastal stocks (Ayón et al., 2011; Bertrand et al., 2011; Salvatteci et al., 2019); (ii) a segment of the fleet starting to venture further offshore (i.e., small-scale squid jiggers and longliners) targeting non-traditional, more abundant, stocks such as jumbo squid (*Dosidicus gigas*), mahi-mahi (*Coryphaena hippurus*), and pelagic oceanic sharks (e.g., *Prionace glauca*, *Isurus oxyrinchus*, and *Alopias vulpinus*) (Estrella and Swartzman, 2010; Mendo and Wosnitza-Mendo, 2014; Marín et al., 2017); and (iii) the implementation of Decreto Supremo No. 017-92-PE, a fisheries regulation that excludes industrial purse seiners and trawlers from the first five nautical miles off the coast, seeking to reduce industrial bycatch and habitat damage in areas known to be important for traditional small-scale fishery resources (SPDA, 2019).

However, fishing effort started growing at faster rates with the turn of the century, and much faster than the catch by 2006 (**Figure 1** and **Table 4**). This caused a trend reversal in the fleet's $CPUE_t$ trajectory, showing that fishing effort was indeed too high, and that the overall surplus production of targeted stocks was declining (Froese et al., 2019).

This explanation is consistent with the $CPUE_t$ trajectories of vessels using gillnets, handlines, hands or tools, and purse seine nets (**Figure 2** and **Table 4**). These sub-fleets make up most of the small-scale fleet and target coastal resources (Marín et al., 2017; Castillo et al., 2018), some of which already show signs of overfishing (CEDEPESCA, 2013; Sueiro and De la Puente, 2015). Conversely, sub-fleets using traps and trawl nets also target coastal resources but show positive, although not statistically significant, trends in $CPUE_t$ over the last decade (**Figure 2** and **Table 4**). These trajectories must be taken with caution, but potentially hint to: (i) the recovery of the punctuated snake-eel (*Ophichthus remiger*), the main species targeted by vessels using traps, after the recent implementation of a rebuilding plan that included input and output controls; and (ii) important increases in the catch of penaeid shrimps by trawl nets, mediated by favorable environmental conditions (IMARPE, 2018).

Sub-fleets whose catch was mostly composed of oceanic resources show mixed trends in $CPUE_t$ over the last two decades (i.e., longlines and squid jigs; **Figure 2** and **Table 4**). However, it is more likely that changes in their $CPUE_t$ are driven by environmental factors modifying their catchability (e.g., by changing the density of targeted schools or the distances at which they are found relative to the shores), rather than suggesting changes in the relative abundance and status of targeted species (Flores et al., 2016; Csirke et al., 2018; Torrejón-Magallanes et al., 2019).

Uneconomic Growth

It is important to also consider socio-economic indicators to define overfishing (Hilborn et al., 2015). We used three indicators of this sort to assess the small-scale fleet behavior, and after looking at their performance over the last 10 years, we claim that recent increases in fishing effort were not only unsustainable but also uneconomic (Daly, 2005). This assertion is supported by the declining trajectories of the small-scale fleet's revenue per unit of effort ($RPUE_t$) (**Figure 3**) and the fishers' relative income to the minimum wage (rI) (**Table 5**), albeit the fleet's total revenue (R_t) remaining stable over the last decade. As vessel owners subtract the operating costs of fishing from R_t before paying salaries to the crew (Sueiro and De la Puente, 2015), it is likely that they are better at withstanding the negative consequences of declining $RPUE_t$.

Significant decreases in $RPUE_t$ were observed for vessels using gillnets, handlines, hands or tools and longlines (**Figure 3**), even when the last of the gears showed a growing $CPUE_t$ trajectory over the same period (**Table 4**). This indicates that the marginal utility of capital investments (i.e., new vessels) was decreasing and furthering economic inefficiency (Daly, 2005, 2013). Nonetheless, declining trajectories in $RPUE_t$ were not observed across all sub-fleets (**Figure 3**). For vessels fishing with squid jigs, price elasticity kept $RPUE_t$ stable, as the unsatisfied demand for jumbo squid by local frozen seafood processing plants (Christensen et al., 2014) resulted in much higher off-prices when catches started to decline (see **Supplementary Material**). Moderate declines in E_t kept $RPUE_t$ stable for the trap sub-fleet, even with highly variable catches. Changes in the catch composition of small-scale trawlers in favor of highly valuable shrimp species, combined with greater landings, prevented declines in $RPUE_t$ albeit significant increases in E_t (see **Supplementary Material**). Finally, the economic contribution of illegal landings of anchoveta by small-scale purse seiners (Guardiola et al., 2012; Grillo et al., 2018), kept them profitable despite increases in E_t, which stabilized their $RPUE_t$ over the last decade (**Figure 3**). This sub-fleet would be significantly less profitable, however, if fisheries regulations were adequately enforced (Sueiro and De la Puente, 2015).

Although the Peruvian small-scale fleet has been able to withstand declines in CPUE and remain profitable in the past (Alfaro-Shigueto et al., 2010), this does not mean that such declines have not taken their toll on fishers' wellbeing. As shown in this study, most sub-fleets' rI has declined significantly over the last decade and many of fishers are already living in relative poverty (Hagenaars and van Praag, 1985). At first, this finding might come across as strange given that recent national surveys indicate that small-scale fishers' incomes have increased since 2012 (Marín et al., 2017; Castillo et al., 2018). However, after correcting for inflation, fishers' incomes and their trajectories over time are consistent with those estimated here. For example, in 2015 people living on the Peruvian minimum wage generated an annual income of real 2009 USD 2,820. During that year, 31% of surveyed small-scale fishers reported annual incomes below \$1500, while 39% reported annual incomes between \$1500 and \$3000 (Castillo et al., 2018). Coincidentally, we found that fishers using gillnets, handlines, squid jigs and traps, which represent ~65% of small-scale fishers working in 2015, earned less than the minimum wage that year (**Table 5**).

It is worth noting that the only fishers who did not experience declining *rI* were using small-scale purse seine and trawl nets. This is problematic as it reveals that incomes could be kept relatively stable, at least for a while, by using the least selective and most ecological damaging fishing methods (Chuenpagdee et al., 2003; Salazar Céspedes, 2019).

The Usual and the Unusual Suspects

Small-scale fishers are a globally vulnerable and marginalized population (Pauly, 2006) whose ability to withstand environmental or economic shocks is curtailed by their limited human capital and economic assets (Finkbeiner et al., 2017; Chuenpagdee and Jentoft, 2019). Their struggles are not evident to others outside the fisheries, including seafood users along the value chain, as markets and social traps distort and muffle them (Salas et al., 2007; Crona et al., 2016a,b). Yet, what are the root causes of the explosive increase in small-scale fishing effort observed in Peru?

As seen in other developing countries, small-scale fisheries tend to absorb a large contingent of the unemployed population migrating to the coasts (Pauly, 2006). In Peru this process started in late 1980s as internal conflicts and terrorism displaced a significant part of the national population from the Andes and Amazon to the coast (Sueiro and López de la Lama, 2014; Sueiro and De la Puente, 2015).

Nonetheless, it is likely that legal changes have had more pervasive impacts than migration. In 1992, a new General Fisheries Law came into effect (Decreto Ley 25977). Through it the government sought to foster the development of the small-scale fleet, and thus waived for them a key administrative requirement for constructing new vessels: the "authorization to increase the fleet size" (in Spanish: 'autorización de incremento de flota'). This allowed for the construction of small-scale vessels without government oversight or fishing licenses. Later, in 1998, the Ministry of Fisheries (now Ministry of Production or PRODUCE) further relaxed regulations to soften the negative economic impacts of the 1997/1998 El Niño event on small-scale fishers, by changing the nature of their fishing licenses from 'single species' to 'multi-species' (Resolución Ministerial 593-98-PE). This modification granted them access to catch all marine living resources if destined for human consumption.

Furthermore, Peru started a process of decentralization in 2002 (Ley 27867), transferring power from the central government to the regional governments ('GOREs'). By 2004, PRODUCE began delegating competences regarding small-scale fisheries monitoring and enforcement to the regions. However, many GOREs did not have the capacities (e.g., training, manpower, or budgets) required for such undertaking, resulting in the weakening of the rule of law (Sueiro and De la Puente, 2015; Monteferri et al., 2017). Although many GOREs have overcome multiple shortcomings (SPDA, 2019), their fisheries-related budgets remain low and insufficient for the tasks at hand (Pajuelo and Sueiro, 2019).

By 2006, PRODUCE sought to curtail the rapidly growing small-scale fleet. First by suspending the construction of new vessels whose hull capacity exceeded 10 m^3 (Decreto Supremo 020-2006-PRODUCE; Decreto Supremo 018-2008-PRODUCE; Decreto Supremo 015-2010-PRODUCE). As smaller boats started to enter the fleet, this suspension was extended to vessels whose holding capacity ranged between 5 and 10 m^3 in 2010 (Decreto Supremo 018-2010-PRODUCE). As the fleet continued to increase, by 2012 PRODUCE prohibited the construction of new small-scale vessels regardless of their size (Decreto Supremo 005-2012-PRODUCE). This prohibition is still in effect (Decreto Supremo 006-2015-PRODUCE), yet the fleet continues to increase.

A factor limiting enforcement for these bans comes from the bylaws specifying fisheries infractions and penalties (Decreto Supremo 016-2007-PRODUCE; Decreto Supremo 017-2017-PRODUCE). Although several regulations deem the construction of new small-scale vessels as illegal since 2006, the infraction is defined in these bylaws as: "constructing or importing fishing vessels without having an authorization to increase the fleet size; as well as modifying or rebuilding small-scale fishing vessels during periods of prohibition or suspension." The wording of this infraction is inadequate, as: (1) only new industrial vessels require authorizations for increasing fleet size, and (2) only small-scale vessel modifications and rebuilding are considered illegal. Thus, this infraction is inapplicable to new small-scale vessels entering the fleet, rendering the bans effectively unenforceable. Paradoxically, the associated penalty (fine) for this infraction is severe (Decreto Supremo 017-2017-PRODUCE).

An additional barrier halting compliance is the top-down governance approach implemented by the Peruvian government. Regulations directly impacting small-scale fishers are not generally drafted in an inclusive or participatory manner (Sueiro and De la Puente, 2015; SPDA, 2019). Hence, they lack legitimacy amongst fishers, who commonly misunderstand, oppose or disregard them albeit their potential benefits (Doherty et al., 2014; Nakandakari et al., 2017; López de la Lama et al., 2018; SPDA, 2019; Mason et al., 2020). Moreover, trust in governing authorities is poor, as they are often seen as inefficient and even corrupt (Sueiro and De la Puente, 2015; Nakandakari et al., 2017; López de la Lama et al., 2018), further incentivizing illegal behaviors (Salas et al., 2007; Finkbeiner et al., 2017). Also, and as seen in other developing countries, social capital is limited among some small-fishing communities further preventing them to find solutions to common problems through self-governance (Chuenpagdee and Jentoft, 2007; Salas et al., 2007; Nakandakari et al., 2017; López de la Lama et al., 2018).

Another dimension of this problem is economic. Battling poverty is a task that relentlessly occupies fishers' attention. Many are so busy working to provide for their families, that coming to terms with the collective consequences (e.g., decreasing *rI*) of individual behaviors (e.g., commissioning a new vessel) becomes a challenge. This results in them blaming external factors, like industrial fisheries and pinniped conservation, for their current circumstances (Sueiro and De la Puente, 2015); and reinforces their pursuit to improve their wellbeing by becoming vessel owners (i.e., increasing the small-scale fleet size).

Without disregarding the validity of fishers claims, it is important to recognize that: (i) recent improvements in industrial fisheries management in Peru have reduced their bycatch and the impact of their catch on local marine ecosystems (Arias-Schreiber, 2012; De la Puente and López de la Lama, 2019), and (ii) pinnipeds have been seen as nefarious for fishers economies and the status of their targeted stocks even when their abundance (and the size small-scale fleet) were both much smaller (Tovar and Fuentes, 1984).

Moreover, the last two decades have been periods of strong economic growth in Peru, leading to investments in the small-scale fleet by stakeholders whose primary sources of income are not necessarily fisheries dependent (Sueiro and De la Puente, 2015). This does not exclude industrial or small-scale fishers from purchasing small-scale vessels (Castillo et al., 2018). Yet, vessel ownership is somewhat concentrated and some boat owners are still able to profit (Christensen et al., 2014; Sueiro and De la Puente, 2015; Castillo et al., 2018). Thus, many small-scale fishers aspire to become vessels owners, but are increasingly having to supplement their incomes with seasonal jobs in agriculture or construction (INEI, 2012; Castillo et al., 2018).

The combined effect of these factors (i.e., internal migration, deficient regulations, top-down governance mechanisms, fishers' economic vulnerability and vessel ownerships aspirations) has resulted in a *de facto* open access regime with strong perverse incentives for increasing small-scale fishing effort – which currently exceeds that of many other fishing countries around the world (Greer, 2014; Belhabib et al., 2018).

Limitations

This study has focused exclusively on assessing the performance of Peruvian small-scale fisheries that use vessels to capture marine living resources. Our findings are thus not representative of shore fishers (methods: beach seines, cast nets, handlines and traps; target group: coastal fishes), coastal gleaners (method: hands or tools; target group: invertebrates of the intertidal zone) and kelp collectors (method: hands or tools; target group: macroalgae). These groups of fishers do not use small-scale vessels and are minor contributors to the catch (INEI, 2012; Christensen et al., 2014; Marín et al., 2017).

Uncertainty exists in this analysis, mainly driven by the reconstruction methods for estimating catch data and inferring fishing effort between data anchor points. For more information on common sources of uncertainty in reconstructions and how they are dealt with please review: Mendo and Wosnitza-Mendo (2014) and Zeller and Pauly (2016). However, the results of this assessment resonate with findings by other researchers within the national and international context.

Nonetheless, it is important to mention that the declining trend in fishers' incomes may be underestimated. On one hand, a fixed cost-income structure (Christensen et al., 2014) was used for each métier between 2009 and 2018. Yet, vessel owners cover their cost (i.e., deduce all operating cost from the revenue) before paying their crews (Sueiro and De la Puente, 2015). Thus, increases in the fishing area covered by the fleet and changes in fuel prices, could have led to reducing the aliquot of the revenue used to pay the crew. Furthermore, wages were assumed to be the same for all crew members within a small-scale fishing vessel. Nonetheless, skippers and motorists are known to have higher earnings than general crew members (Castillo et al., 2018). Hence, for some fishers the reported declines in their income might underrepresent their current financial struggles.

Thus, the authors ask readers to consider the results presented in this paper as a working hypothesis of the state of affairs in Peru. These findings can be used as a tool to communicate the dangers of continued fleet growth on local target resources and fishing communities. However, they should also be regarded as the starting point of a longer discussion, where if strengthened by additional and perhaps more accurate data, and then validated by local stakeholders, could be used as a tool to inform fisheries policy in Peru.

Recommendations

Although this section falls beyond the scope of this paper, we present some recommendations for the Peruvian government - rooted in robust social and fisheries science - aimed at improving small-scale fisher's wellbeing in Peru: (i) strengthen social capital within fishers' assemblies and fishing communities (Nakandakari et al., 2017; López de la Lama et al., 2018); (ii) increase fishers' involvement in research, as well as in the design, implementation and evaluation of fisheries policies and regulations (Punt et al., 2016; Chuenpagdee and Jentoft, 2019; McDonald et al., 2019); (iii) support successful, yet informal, self-governance arrangements currently in effect within fishing communities (Salas et al., 2007; Nakandakari et al., 2017; Chuenpagdee and Jentoft, 2019); (iv) reinforce local transdisciplinary research capacities aimed at improving and incorporating small-scale fisheries related knowledge into management (Pauly, 2006; Hilborn et al., 2015; Chuenpagdee and Jentoft, 2019); (v) promote investments for developing alternative sources of income within small-scale fishing communities (Sueiro and De la Puente, 2015); and (vi) enhance PRODUCE's and regional governments' enforcement capacities by increasing their budgets and modifying the legal tools at their disposal for discouraging illegal behaviors (Pajuelo and Sueiro, 2019; SPDA, 2019).

Conclusion

Small-scale fisheries, and their sustainable development, are highly important for Peruvian food security, economy and culture. However, small-scale fishing effort has significantly increased over time, negatively impacting target stocks, fishing efficiency and fishers' livelihoods. These findings are alarming and require immediate action, as small-scale fishers are a vulnerable population and growing into poverty could drive them further away from becoming resource stewards.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

SD conceptualized the study, analyzed and interpreted the data, and wrote the manuscript. SB reviewed and analyzed the Peruvian fisheries legal framework. RL, JS, and DP revised the manuscript. All authors approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00681/full#supplementary-material

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Using AIS to Attempt a Quantitative Evaluation of Unobserved Trawling Activity in the Mediterranean Sea

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In the past decades, the Automatic Identification System (AIS) has been employed in numerous research fields as a valuable tool for, among other things, Maritime Domain Awareness and Maritime Spatial Planning. In contrast, its use in fisheries management is hampered by coverage and transmission gaps. Transmission gaps may be due to technical limitations (e.g., weak signal or interference with other signals) or to deliberate switching off of the system, to conceal fishing activities. In either case such gaps may result in underestimating fishing effort and pressure. This study was undertaken to map and analyze bottom trawler transmission gaps in terms of duration and distance from the harbor with a view to quantifying unobserved fishing and its effects on overall trawling pressure. Here we present the first map of bottom trawler AIS transmission gaps in the Mediterranean Sea and a revised estimate of fishing effort if some gaps are considered as actual fishing.

Keywords: Automatic Identification System (AIS), data gaps, Maritime Domain Awareness, Monitoring Control and Surveillance (MCS), fishing activity

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INTRODUCTION

The Automatic Identification System (AIS) was originally introduced by the International Maritime Organization to enhance safety at sea by enabling navigators to view the position, identity, and direction of other ships in the area. Its characteristics include a considerable accuracy in providing vessel position (Gioia et al., 2013), high-frequency transmission (James et al., 2018), and accessibility through several online portals (Le Tixerant et al., 2018). Even though AIS was initially developed to avoid collisions between vessels (Goerlandt and Kujala, 2011; Wu et al., 2017), its features make it a useful tool for Maritime Spatial Planning (MSP) and for the management of a number of maritime activities (Shelmerdine, 2015). Indeed, AIS is now employed in a variety of research areas (Svanberg et al., 2019) that range from oil spill prevention (Eide et al., 2007; Schwehr and McGillivary, 2007) to identification of fishing vessel behavior (Natale et al., 2015; De Souza et al., 2016). Since May 2014, all European fishing vessels exceeding 15 m length overall are required to carry and operate an AIS device (European Commission [EC], 2011). The system has been providing valuable Monitoring, Control and Surveillance (MCS) data on a large amount of European fishing activities. Moreover, its voluntary adoption by several non-European fishing fleets has been supplying additional information to map the areas exploited by these fleets, such as the Mediterranean Sea (Ferrà et al., 2018).

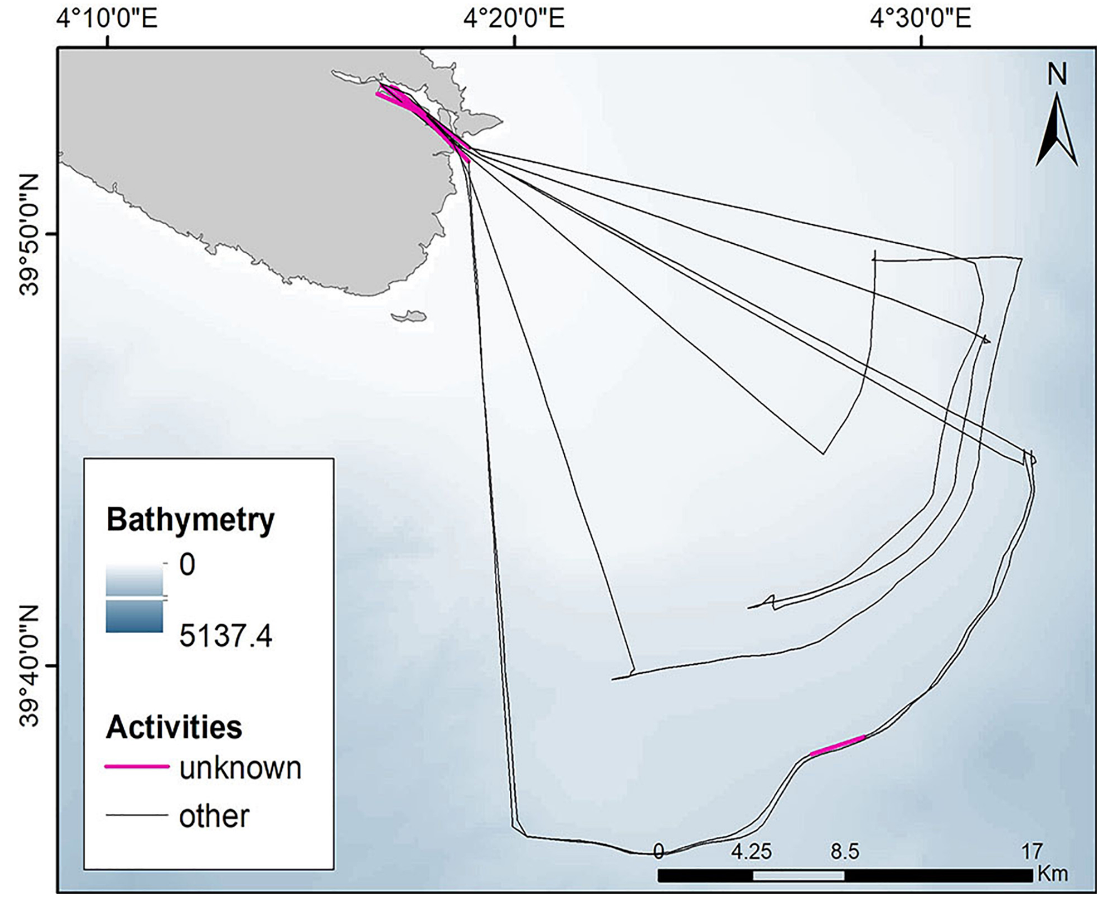

FIGURE 1 | Weekly tracking layer of a trawler. Segments exceeding 30 min were labeled as "unknown".

Automatic Identification System is the only existing large-scale system supplying actively transmitted data (Ford et al., 2018) and provides valuable information to understand maritime routes and fishing activities. Nonetheless, a major disadvantage is that it suffers from transmission gaps. Such gaps may be due either to technical limitations or to deliberate manipulation of the AIS transponder, which may be switched off (Winward, 2015) to obscure the vessel's destination or hide illegal fishing (Ahlberg and Danielsson, 2015). Manipulation is a cause for concern for financial and security stakeholders, since it undermines reliance on the system's ability to track vessels and monitor areas (Winward, 2015). For this reason, identifying vessels engaging in this practice is especially important (Hsu et al., 2019).

The technical limitations of AIS, e.g., incomplete AIS reception, system saturation in areas with high vessel traffic, faulty equipment and/or line of sight limits, have been widely discussed (Harati-Mokhtari et al., 2007; Merchant et al., 2012; Papi et al., 2014). Some data providers have been developing solutions to address them, for example by combining land-based and satellite AIS signals which enable vessel detection hundreds of miles from any land-based receiver (Høye et al., 2008; Pallotta et al., 2013a). Some researchers have proposed a number of approaches that range from statistical models to anomaly detection algorithms (Guerriero et al., 2010; Mazzarella et al., 2017; Ford et al., 2018; Fernandes et al., 2019). Automatic Identification System reception quality maps of fishing vessels in EU (Vespe et al., 2016) and global waters (Kroodsma et al., 2019) have recently been published.

Where fishery management is concerned, mapping transmission gaps – be they accidental or deliberate – and integrating them with high temporal and spatial resolution data of fishing position is essential to gain an exhaustive and spatially explicit knowledge of the fishing grounds exploited by vessels. Moreover, it is a key requirement for the implementation of an effective ecosystem approach to fisheries management in areas under multiple (e.g., local, national and international) jurisdictions and in those involved by competing activities or encompassing sensitive habitats.

Based on such considerations, this study was undertaken to map the AIS transmission gaps of bottom trawlers operating in the Mediterranean Sea with a view to identifying those that may represent potential hidden fishing activities and to quantifying the unobserved fishing effort. The data subset suspected to conceal potential fishing activities was combined with the known fishing dataset (Tassetti et al., 2016; Ferrà et al., 2018), to compute the extent to which the overall trawling activity would

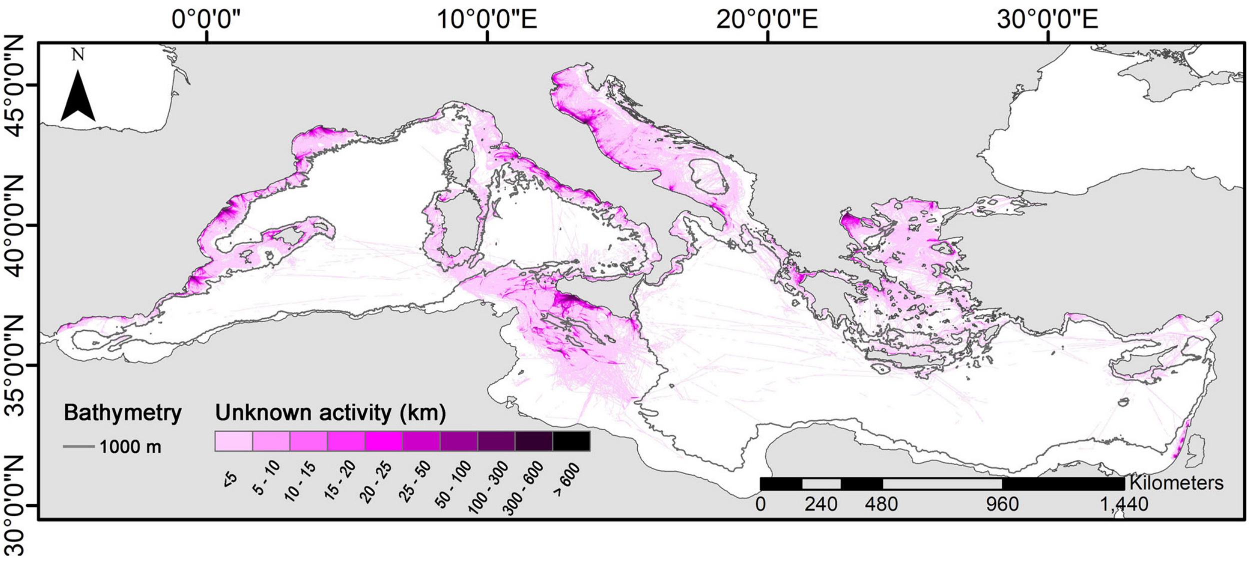

FIGURE 2 | AIS-based unknown activity map (km) aggregating the unknown tracks.

increase if some of these gaps were considered as actual fishing activities. Accurate estimation of such increase would provide valuable information for Mediterranean fisheries, where trawling is the main fishing activity (Merino et al., 2019) and where in some areas its footprint exceeds 80% of the continental shelf (Amoroso et al., 2018).

MATERIALS AND METHODS

Terrestrial AIS data at 5-min resolution were purchased from a private provider[1] for all the fishing vessels (AIS type 30) operating in the Mediterranean Sea in 2014. Gaps in tracking data were identified by pre-processing the raw dataset with the statistical software R (R Development Core Team and R Core Team, 2018). Repeated points and outliers with a speed >20 knots (kn) or located inland were removed; the remaining pings were filtered so as to retain only the records of the bottom trawlers identified in a previous study by our group (Ferrà et al., 2018).

Subsequently, the Maritime Mobile Service Identity (MMSI) was used to create a tracking layer by ordering each MMSI record by datetime. Assuming that vessels travel in a straight line, track duration and vessel speed were computed for each segment based on the difference between consecutive pings. Since the acquired reporting rate was 5 min, tracks with a duration exceeding a predefined threshold of 30 min were tagged as "unknown," retained and considered as transmission gaps (**Figure 1**). Computation of the monthly ratio of total unknown time/total AIS time in the Mediterranean (tests for differences in monthly patterns of gaps) demonstrated a nearly constant pattern and allowed subsequent analyses to be performed on a yearly scale.

[1] http://www.astrapaging.com/

TABLE 1 | Frequency of unknown tracks (%) in terms of duration (t) and distance (d) from harbors.

Distance from harbors (nm)	Frequency of unknown tracks (%)				
	$\leq$1 h	1 < t $\leq$ 2 h	2 < t $\leq$ 24 h	>24 h	Total
0 < d $\leq$ 1	2.8	6.8	35.2	58.3	13.7
1 < d $\leq$ 5	16.7	21.4	23.9	22.7	19.2
d > 5	80.5	71.8	40.9	19.0	67.1

The annual unknown activity was quantified by intersecting all unknown segments with a 1 km × 1 km grid of the Mediterranean Sea by spatial joining each cell with overlapping portions of unknown segments and then summing their relative length (in km) (Ferrà et al., 2018; Tassetti et al., 2019).

The unknown segments were grouped according to their duration (t) into four categories (t $\leq$ 1 h; 1 < t $\leq$ 2 h; 2 < t $\leq$ 24 h; > 24 h) and according to distance from the harbor into four categories (in the harbor, within 1 nautical mile [nm] of the harbor, between 1and 5 nm, and >5 nm from the harbor). The percent frequency of each category was then computed.

To establish whether some gaps, identified at the sites of the main fishing grounds, could conceal fishing activities, the unknown tracks overlaying the trawl footprint (Ferrà et al., 2018) hotspots were analyzed for speed and duration. Those inside harbors were excluded. The most frequent duration and speed values were extracted to create a subset of unknown segments. After their validation based on the common speed of bottom trawlers in the Mediterranean Sea (2–5 kn), the segments were considered as representing potential hidden fishing activity. They were then added to the trawling dataset (Ferrà et al., 2018), to estimate their contribution to fishing activity.

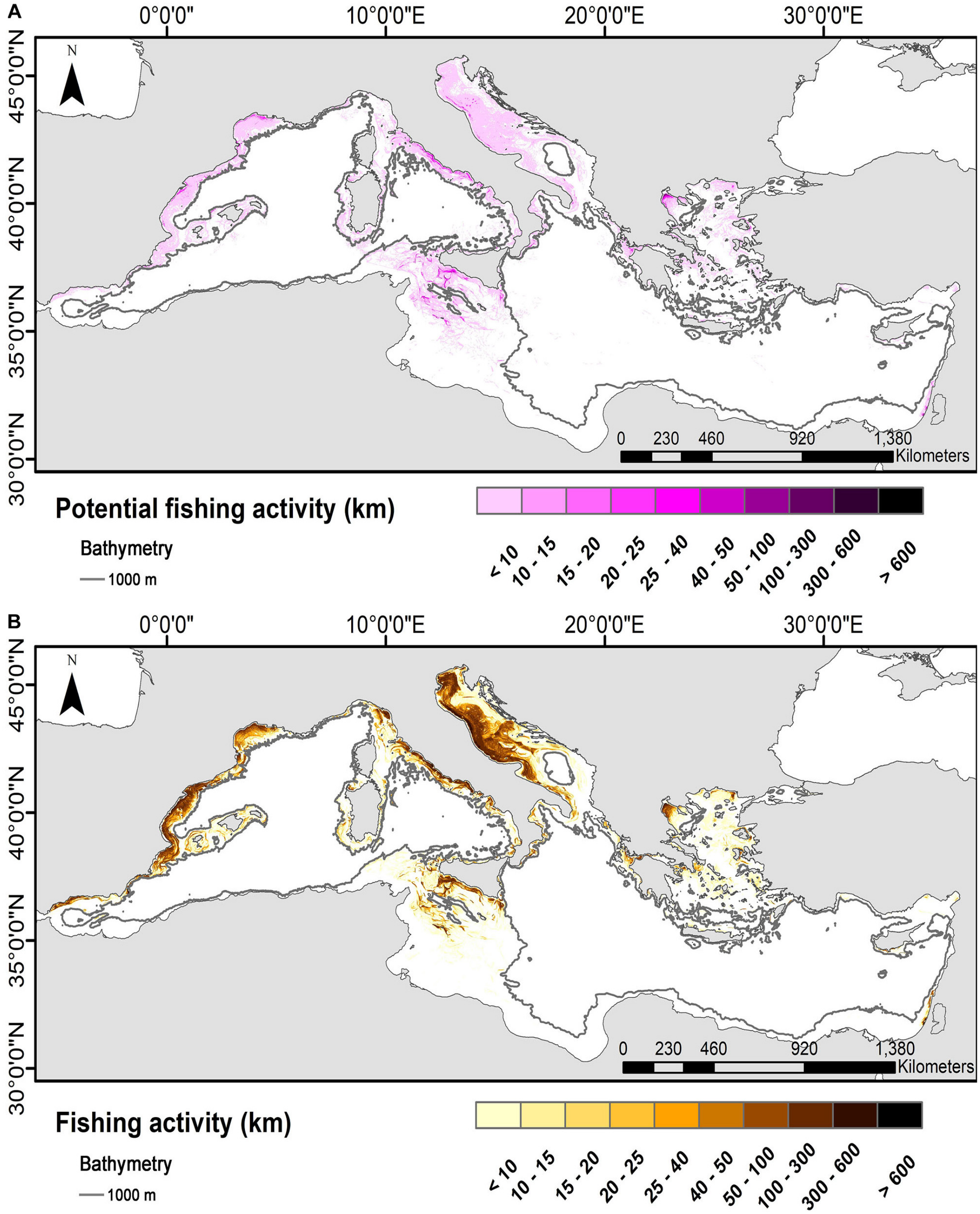

FIGURE 3 | Potential **(A)** and known annual fishing activity **(**Ferrà et al., 2018, **B)** in the Mediterranean Sea in 2014.

Finally, the percent potential fishing activities over total fishing (known fishing plus potential fishing), were mapped and aggregated in each 30 × 30 min cell of the General Fisheries Commission for the Mediterranean (GFCM) Statistical Grid.[2] The procedure allowed identifying those areas where potential hidden fishing was more intense. One such area, the Strait of Sicily, was selected as a case study and analyzed in detailed maps combining known and potential fishing activity (sum and ratio: potential/[known + potential]).

RESULTS

The initial dataset consisted of 57,029,372 position records of 4,343 vessels tracked in the Mediterranean Sea throughout 2014. Pre-processing allowed selecting 37,322,648 records broadcast

[2]http://www.fao.org/gfcm/data/maps/grid

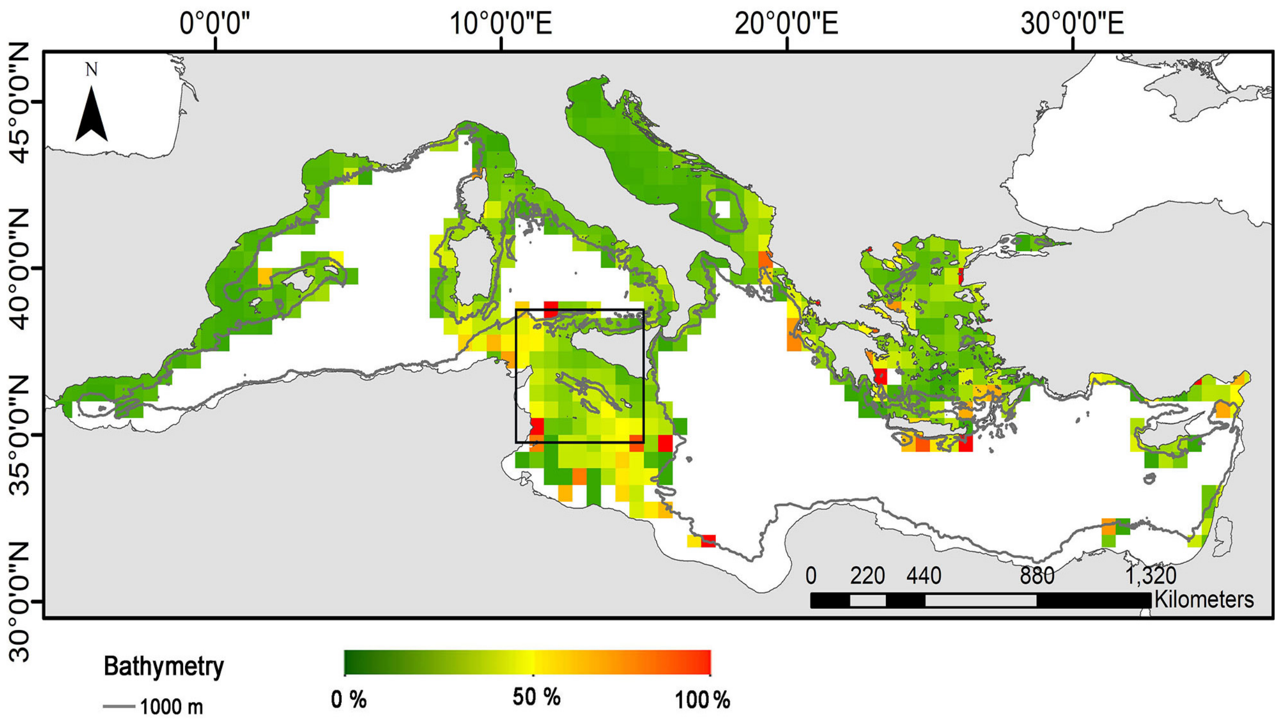

FIGURE 4 | Mean percent values of potential hidden fishing out of total fishing activity (2014).

by 2,123 bottom trawlers identified in a previous study (Ferrà et al., 2018), which accounted for 65% of the initial records. These records were retained and processed to estimate tracking gaps. The procedure allowed identifying 733,143 unknown tracks connecting 1,466,286 pings, corresponding to 4% of the AIS bottom trawl records.

The AIS data coverage in each month of 2014, expressed as the ratio of the duration of the unknown tracks to the duration of total activity, showed an almost constant value that ranged from a minimum of 65% (May) to a maximum of 72% (September). Altogether, 68% of the total time recorded by AIS in 2014 was categorized as unknown.

Unknown tracks were identified in all European, Turkish, and Israeli coastal areas, and rarely in areas off the coast, at depths up to 1,000 m (**Figure 2**). In the Mediterranean Sea they accounted for 3,954,925.2 km, of which 19% (741,114.47 km) were concentrated within 3 nm from the coast or up to a depth of 50 m. The unknown tracks were most numerous in the Thermaic Gulf, the western Mediterranean, the western Adriatic, and the Tyrrhenian coast. The southern area of the basin was largely free of unknown tracks except for the Strait of Sicily.

Slightly more than 60% of unknown tracks had a duration of ≤2 h (mean, 45.8 min); of these, about 50% lasted ≤1 h (mean, 38.6 min). The mean duration of gaps lasting 1–2 h was 79 min.

With regard to gaps >2 h, those ranging from 2 to 24 h had a mean duration of 594.7 min and accounted for 29% of all unknown tracks, whereas those >24 h had a mean duration of 3 days.

Tracks lying wholly or partially in harbors accounted for 32% and were excluded from subsequent analyses. Of the unknown tracks identified outside harbors, 35 and 58% of those lasting respectively 2–24 h and >24 h were wholly or partially located within 1 nm of a harbor (**Table 1**). Notably, the number of tracks >24 h diminished as the distance from the harbor increased, whereas those lasting 2–24 h were less numerous 1–5 nm from the harbor and more numerous at greater distances. In contrast, the number of gaps lasting up to 2 h increased gradually as vessels left the coastal waters (within 1 nm of the harbor) and sailed out to sea. The increment was very large for tracks of up to 1 h and slightly smaller for tracks whose duration was 1–2 h.

More than 78% of the unknown tracks overlapping trawling hotspots lasted up to 2 h and the interquartile range of speed was 2–5 kn. This type of tracks was filtered and mapped as representing potential hidden fishing (**Figure 3**). Summation of this track subset to the known fishing activities involved an 8% increase in the trawling effort in the Mediterranean basin as a whole.

A more detailed analysis using the GFCM grid highlighted a greater increase in the total trawling effort in some areas such as the Strait of Sicily, the Gulf of Alexandretta, and the waters south of Crete (**Figure 4**).

In the Strait of Sicily (black bounding box in **Figure 4**), the potential fishing subset added to the known fishing would increase the total fishing activity by around 20% (**Figure 5**). The potential hidden activity in this area showed no distinctive spatial pattern, even though it was very limited even in the most extensively exploited fishing grounds (**Figure 6**).

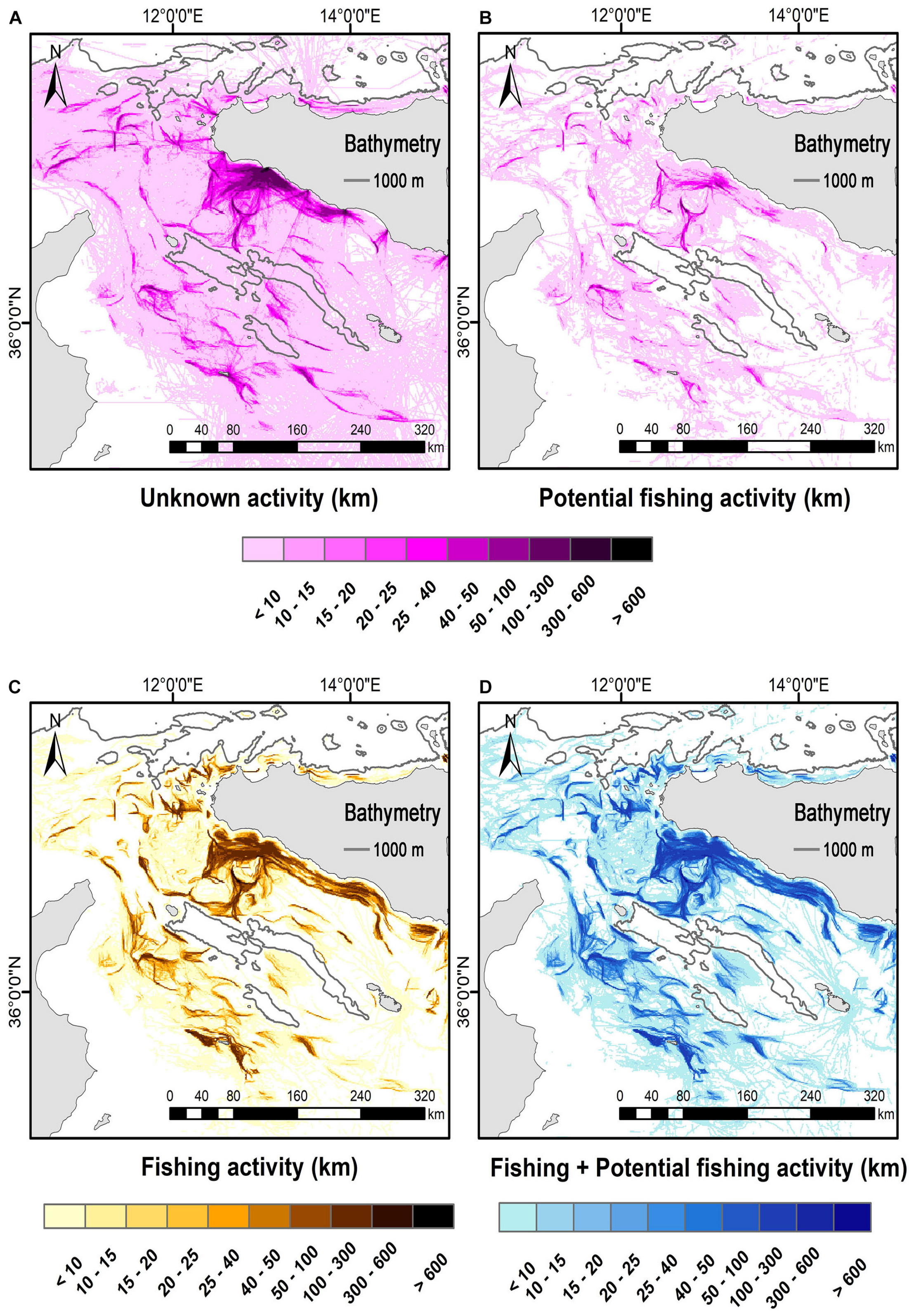

FIGURE 5 | Unknown annual activity **(A)**, potential **(B)**, known **(C)**, and total (known + potential) annual fishing activity **(D)** in the Strait of Sicily (2014).

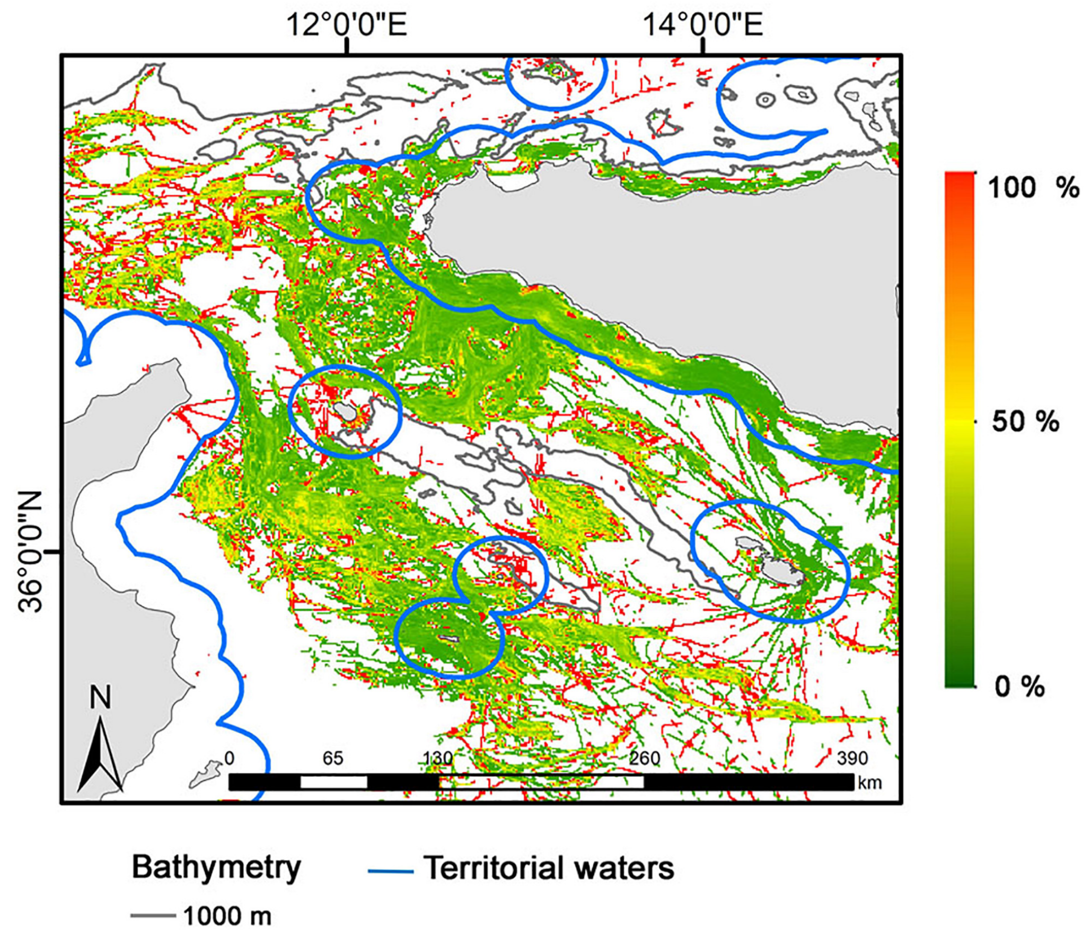

FIGURE 6 | Map of unobserved out of total fishing activity in the Strait of Sicily (percent values, 2014).

DISCUSSION

The use of AIS tracking data for Maritime Domain Awareness involves two major drawbacks: transmission gaps and the ability of the system to be switched off (McCauley et al., 2016; Shepperson et al., 2018). These disadvantages are particularly important in fisheries mapping, since they result in underestimating the extension of fishing grounds and total fishing effort and pressure.

Although AIS reception quality in the Mediterranean Sea has recently been mapped by FAO (Kroodsma et al., 2019), it involves all fishing activities, whereas maps for specific gears or studies aimed at interpreting them are not yet available. The aim of this study was to provide a map of the spatial distribution of bottom trawler transmission gaps in the Mediterranean in 2014, to identify any potential unobserved fishing activity and to assess the extent to which it contributes to the trawling pressure.

Clusters of high values of unknown activity were mostly found in the northern Mediterranean in correspondence to the hotspots identified by Ferrà et al. (2018). Even though these gaps were a small proportion of the total number of AIS records, they accounted for a large proportion of coverage in terms of duration, since some lasted more than 30 days. Considering that most clusters coincided with areas where signal strength is expected to be high (Vespe et al., 2016), it is reasonable to hypothesize that the gaps were due either to deliberate manipulation of the device or to technical problems related to transmission in high-density traffic areas (ICES, 2015; Plass et al., 2015) or to poor weather conditions (Pelich et al., 2014; Nguyen et al., 2020), rather than to inadequate coverage (Mazzarella et al., 2016). In contrast, the limited known and unknown activity seen in the southern Mediterranean is likely due to the fact that the area is largely exploited by trawlers from non-EU Mediterranean countries, few of which broadcast AIS.

The proportion of unknown tracks having a duration of up to 2 h was limited in harbors, it increased with increasing distance from the harbor, and it peaked in areas farther than 5 nm from the home harbor. The large proportion of relatively short unknown tracks within a 5 nm radius from harbors may be due to illegal fishing (European Commission [EC], 2006), delayed switching on of the device after departure, or early switching off at arrival. In contrast, the unknown tracks exceeding 24 h were more numerous in harbors and less numerous at a greater distance from them. These data can be explained by the switching off of all electric devices while at mooring. However, part of the transmission gaps can probably be explained by the fact that, since vessels can be tracked in real time through free-access AIS websites, fishermen switch the AIS off when leaving harbor to prevent competitors from learning which fishing grounds they will exploit. Finally, the irregular pattern of tracks lasting 2–24 h probably reflects a greater variability of the duration of this category of tracks rather than fishermen's behavior.

A considerable proportion of the unknown tracks overlapping known fishing hotspots lasted no more than 2 h and their mean speed was 2–5 kn. Since these values are typical of the fishing operations of most Mediterranean bottom trawlers (Lucchetti, 2008; Lambert et al., 2012; Keskin et al., 2014; Arjona-Camas et al., 2019), this subset was felt to represent hidden fishing and was used to sketch a more realistic picture of the overall fishing activity.

Although towing duration can last more than 2 h, especially in the deeper fishing grounds, and although a vessel is unlikely to spend more than 24 h at sea with its AIS switched off without fishing, the segments longer than 24 h were not considered as potential fishing. This may have resulted in underestimating total fishing pressure; yet such segments were not felt to represent potential fishing, because vessels are unlikely to keep a straight course for long periods (Lambert et al., 2012) and because in such long periods they can fish and sail several times.

When potentially hidden and known fishing activities were summed, the percent increase of the fishing effort was not significant in the Mediterranean Sea as a whole. However, use of the 1 nm × 1 nm reference grid enabled a fine analysis of the spatial localization of potentially unobserved fishing and showed that in some areas it was quite large. In the Strait of Sicily the increase was 20%; yet, in the hotspots of this area, potential fishing contributed little to total trawling activity and was randomly distributed. This suggests that the unknown tracks were largely to be ascribed to system saturation in this heavily trafficked area rather than to deliberate switching off of the AIS device.

In conclusion, the present findings confirm that the reasoned interpretation of tracking gaps in relation to fishing data can provide more accurate information on fishing ground extension, fishermen's behavior and fishing pressure. This information is useful to implement fisheries management strategies both on the regional and the local scale, as also required by EU regulations (European Commission [EC], 2002, 2013; The European Parliament and the Council of the European Union, 2008; European Parliament and the Council of the European Union, 2014).

Notably, quantification of the unobserved activity of each vessel would allow identifying those vessels that consistently switch the AIS device off, thus increasing transparency at sea (Malarky and Lowell, 2018) and contributing to curb Illegal Unreported and Unregulated fishing (Bastani and de Zegher, 2019). This is even more important in regulated areas, where AIS data gaps could unmask attempts to conceal illegal activities (Tassetti et al., 2019), and would allow decision-makers to implement mitigation strategies and MCS plans.

AIS optimization would be of great interest to policymakers and also to environmental and fisheries scientists (Svanberg et al., 2019), who are often called upon to help develop management measures aimed at marine environment preservation and fisheries regulation. Whereas the tracking gaps are expected to be overcome by satellite AIS (Pallotta et al., 2013b; Wu et al., 2017), strengthening policy provisions seems essential to enforce the proper use of AIS and reduce deliberate switching off of the device (McCauley et al., 2016).

DATA AVAILABILITY STATEMENT

The data analyzed in this study is subject to the following licenses/restrictions: AIS raw data is not anonymous and owned by CNR. Requests to access the aggregated datasets should be directed to CF, carmen.ferravega@cnr.it.

AUTHOR CONTRIBUTIONS

CF and ANT conceived the ideas and designed the methodology. CF collected and analyzed the data. CF, ANT, and GF wrote the manucript. GS, ENA, and AG contributed critically to the manuscript drafting. All authors gave final approval for publication.

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Fishing Effort and Associated Catch per Unit Effort for Small-Scale Fisheries in the Mozambique Channel Region: 1950–2016

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The Mozambique Channel region in East Africa has diverse marine ecosystems and serves as a migratory corridor for economically important species. Local and foreign industrial fisheries operate in the Mozambique Channel, but regional small-scale fisheries are the crucially important fisheries that provide food security, livelihoods, and economic opportunities for rural coastal communities. This study reconstructed and investigated trends in the fishing effort and catch per unit effort (CPUE) of small-scale marine fisheries in four Exclusive Economic Zones (EEZ) that constitute the Mozambique Channel, i.e., Union of Comoros, Madagascar, Mayotte, and Mozambique, from 1950 to 2016. Effective fishing effort for small-scale fisheries in the form of fishing capacity in kWdays (i.e., kilowatt days) was derived using the number, length, motorization (engine power) by fishing vessels, as well as an approximate human-powered equivalent for shore-based fishers without vessels, as well as days of fishing per year. Effective small-scale fishing effort in the Mozambique Channel increased by nearly 60 times from just over 386,000 kWdays in 1950 to over 23 million kWdays in 2016. Correspondingly, the overall small-scale CPUE, based on previously and independently reconstructed catch data declined by 91% in the region as a whole, from just under 175 kg·kWday^{-1} in the early 1950s to just over 15 kg·kWday^{-1} in recent years. All four EEZs showed the strongest declines in the small-scale CPUE in the earlier decades, driven by motorization and growth in vessel numbers impacting effective fishing effort. Increased motorization combined with a substantial growth in overall vessel numbers were the drivers of the increasing fishing effort and decreasing CPUE, and clearly suggest that continuing to increase the fishing capacity of small-scale fisheries in the absence of effective and restrictive management actions may exacerbate overexploitation risk.

Keywords: artisanal fisheries, CPUE, fishing capacity, Madagascar, Mayotte, Mozambique, subsistence fisheries, Union of Comoros

INTRODUCTION

The Mozambique Channel region in East Africa separates Madagascar from the African continent (**Figure 1**) and is characterized by high marine biodiversity and a variety of ecosystems, including a large proportion of the Indian Ocean's coral reefs, mangroves, and seagrass beds (Nunes and Ghermandi, 2015). It is also an important corridor for migratory species, such as tuna (Nunes and Ghermandi, 2015). The four countries/territories directly associated with the Mozambique Channel region, namely the Union of Comoros, Madagascar, Mayotte, and Mozambique all heavily depend on small-scale domestic fishing (artisanal and subsistence fishing). For example, the Comoros lacks domestic industrial fisheries and an aquaculture sector (FAO, 2015a); however, between 2005 and 2009, domestic marine fish resources accounted for 60–70% of the animal protein consumed by Comoros islanders (Béné and Heck, 2005; Kurien and López Ríos, 2013; Breuil and Grima, 2014a). For Madagascar, the Food and Agriculture Organization of the United Nations (FAO) reports that the small-scale fishing fleet was responsible for over half of total domestic marine fish catches in 2008 (FAO, 2009). Despite their importance to food security and local livelihoods, small-scale fisheries catches in the region have been widely underreported (Jacquet et al., 2010; Le Manach et al., 2012b; Genay and Merceron, 2017; Anon, 2019c). For example, the reconstructed catch data of the *Sea Around Us*,[1] which complement officially reported data with comprehensive estimates of unreported catches (Zeller et al., 2016), suggest that small-scale catches in Madagascar actually may account for over 80% of the total domestic marine fish catch (Le Manach et al., 2012b), as opposed to the 53% suggested by FAO (2009) based on officially reported data. The underrepresentation of small-scale fisheries is a common issue globally, which contributes to the marginalization in socio-economic and political considerations (Pauly, 2006; Schuhbauer and Sumaila, 2016; Teh et al., 2020) as well as in official statistics (Pauly and Charles, 2015; Zeller et al., 2015; Pauly and Zeller, 2016a), despite more recent efforts to begin addressing this issue (FAO, 2015b).

Since 1950, the coastal populations of the four inhabited countries/territories associated with the Mozambique Channel have increased, and marine catches have correspondingly also grown (Jacquet et al., 2010; Le Manach et al., 2012b; Doherty et al., 2015a,b; Genay and Merceron, 2017). The exceptions are the uninhabited islands of Juan de Nova, Europa, and Bassas da India that are territorial possessions of France, here referred to as the "Mozambique Channel Isl. (France)" (**Figure 1**). These waters are not fished by "domestic" fleets due to the uninhabited nature of the islands, but are accessed by fishers from surrounding areas as well as by distant water fleets (Le Manach and Pauly, 2015). Given the crucial importance of locally sourced seafood for domestic food security and nutrient security for the Mozambique Channel region, demand will likely continue to increase. However, official fisheries statistics describing the small-scale fisheries catches in this region remain limited.

[1] www.seaaroundus.org

Historically, FAO presumed that small-scale artisanal and subsistence fishers were likely only temporary features during an anticipated transition to industrial fisheries (Panayotou, 1982). Therefore, institutional and political support had been, and often still is, skewed toward large-scale (i.e., industrial) fisheries due to perceived higher direct macroeconomic contributions to national or government income (Zeller and Pauly, 2019). As such, industrial fisheries often have easier access to subsidies, including subsidized development loans (Panayotou, 1982; Harper et al., 2012; Bellmann et al., 2016; Sumaila et al., 2019). More recently, the international community began to recognize the crucial nutrient security and food security, livelihood, and socio-economic importance of small-scale fisheries (FAO, 2015b; Golden et al., 2016; Teh et al., 2020; Vianna et al., 2020), although officially reported statistics as published by the FAO on behalf of countries continue to be hampered by an absence of fishing sector differentiation (Pauly and Charles, 2015). In contrast, such sectoral differentiation is an integral and core feature in the *Sea Around Us* reconstructed catch data for all countries (Pauly and Zeller, 2016a,b), which allows for a more accurate and comprehensive examination of the importance of small-scale sectors at the country, regional and global level (Zeller and Pauly, 2019), including their economic importance (Zeller et al., 2006).

Here, a data reconstruction approach based on Zeller et al. (2016) was used to derive time series estimates of small-scale fishing effort from 1950 to 2016 for each of the four inhabited Mozambique Channel countries/territories. These effort estimates were then combined with independently reconstructed small-scale catch data from the *Sea Around Us* for each of these countries/territories (Pauly and Zeller, 2016a,b) to derive small-scale catch per unit effort (CPUE) estimates for the same time period.

MATERIALS AND METHODS

In this study, small-scale fishing effort for each country/territory was reconstructed using the same general data reconstruction approaches and principles as used earlier and independently by the *Sea Around Us* for catch data reconstruction (Zeller et al., 2016). Data reconstructions essentially aim to complement officially reported data with best estimates of unreported components, using a wide variety of secondary data and information sources combined with carefully vetted and clearly stated, conservative assumptions (Zeller et al., 2016). To ensure independence of the two key datasets being used here (small-scale catches and fishing effort), fishing effort was reconstructed without considering previously reconstructed small-scale catch data in each country (Pauly and Zeller, 2016a,b).

Study Area

The Mozambique Channel region (IHO, 1953) lies between the East African coast and the west coast of Madagascar (**Figure 1**). The northern boundary of the channel is marked by the estuary of the River Rovuma in Mozambique (10.46° S and 40.43° E) and the north west point of Grande

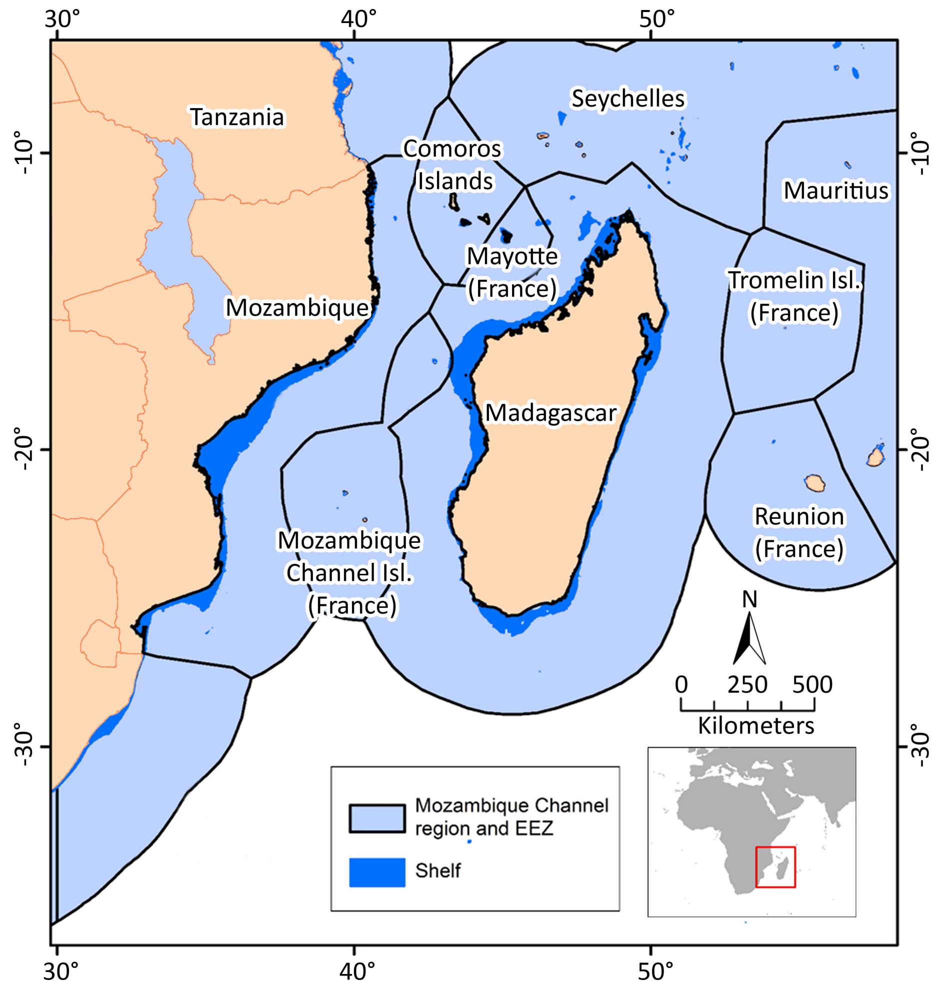

FIGURE 1 | The Mozambique Channel region and the EEZs of the countries or territories considered here: Union of the Comoros, Madagascar, Mayotte, and Mozambique. The uninhabited Mozambique Channel Islands administered by France are indicated as well, but have no domestic, resident small-scale fisheries and thus no resident small-scale effort or catches.

Comore/Ngazidja (the northernmost island of the Comoros Isl. Group; 11.95° S and 49.28° E). The southern boundary of the channel runs from the southern tip of Madagascar to Ponta do Ouro, Mozambique (26.88° S and 32.93° E). Within the channel, there are five Exclusive Economic Zones (EEZs): Union of Comoros Islands, Madagascar, Mayotte, Mozambique, and the minor Mozambique Channel Islands, comprising the uninhabited French dependencies of Juan de Nova, Bassas da India, and Europa Islands (**Figure 1**).

The focus of this study was on small-scale domestic fisheries by resident fishers, and included the EEZs of the Union of Comoros, Madagascar, Mayotte, and Mozambique. The French dependencies here referred to as the Mozambique Channel Islands are uninhabited, and therefore by the definition of small-scale sectors used here (Zeller et al., 2016) have no locally based domestic small-scale fleets, and thus did not contribute to this study. However, their waters are being fished by neighboring countries, including by artisanal and recreational vessels, as well as French and distant-water industrial fleets (Le Manach and Pauly, 2015).

Boat-Based Fishing Effort

Nominal fishing effort (kW), as defined in this study, is the product of total small-scale fleet capacity. A fleet is the number of vessels with a similar capacity, i.e., in the same length class and motorization category, and utilizing the same or similar fishing gears. Engine capacity per fishing vessel (kW) in a given fleet was determined by length and motorization (**Table 1**). Thus, total nominal fishing effort is the product of the engine capacity and the number of boats operating within a fleet segment in a given year. Non-motorized vessels were considered to have equivalent capacity of 0.37 $kW \cdot vessel^{-1}$ for vessels of length class 1 and 0.75 $kW \cdot vessel^{-1}$ for vessels of length class 2 (**Table 1**), based on Greer et al. (2019a).

Effective fishing effort (kWdays) is the product of nominal fishing effort (engine capacity × number of boats within a fleet segment) in kW and the number of days spent fishing per year.

The number of days spent fishing per year (i.e., fishing trip days) for each fleet segment was a major aspect of the research in this project (see country details in **Supplementary Material**). Unfortunately, there was a notable knowledge and data gap in this metric in the literature, thus requiring conservative assumptions to be able to derive likely time series for fishing days. If no specific data were found for a given fleet segment, conservative assumptions, and approximations were used to derive an average number of fishing trip days per year. Due to the generally low technological development of most small-scale vessels in the Mozambique Channel region, we did not apply a "technology creep" factor to our effort estimates (Palomares and Pauly, 2019).

Shore-Based Fishing Effort

Fishing and seafood collecting from shore without the use of a boat, often largely conducted by women, is a regular activity in many parts of the study region and globally (Harper et al., 2013, 2020; Zeller et al., 2015). Given its widespread existence, this fishing component needs to be accounted for in the estimation of overall small-scale fishing effort.

The *nominal fishing effort* or fishing capacity of one shore-based fisher is assumed to be approximately 0.08 kW of engine equivalence per work (fishing) day (Krendel et al., 2007). Thus, shore-based fishing activities were converted to the equivalent nominal fishing effort (kW) by multiplying the number of shore-based fishers by their assumed engine equivalency of 0.08 kW (**Table 1**).

Effective fishing effort (kWdays) was calculated by multiplying the nominal fishing effort for shore-based fishers with the average number of days per year for which fishing from shore occurred. Fishing days for shore-based fishers were determined from country-specific sources wherever available (see country details in **Supplementary Material**). If no specific data were found for a given country, conservative assumptions, and approximations were used to derive an average number of fishing trip days per year. We recognize that shore-based fishers do not necessarily fish all day, i.e., for a full "work day," with regards to the engine equivalency used above. However, as we remained conservative in our estimation of the number of shore-based fishers throughout, we consider our assumption of a full work day engine power equivalency for shore-based fishers to be an acceptable approximation.

TABLE 1 | Assumed and derived engine capacity of motorized and non-motorized fishing vessels by length class for small-scale vessels based on Greer et al. (2019a), as well as engine capacity-equivalency for shore-based fishers.

Length class	Length (m)	Mean length(m)	Capacity (kW)	
			Motorized	Non-motorized
1	<7.9	4.5	9.11	0.37
2	8–15.9	11.3	58.70	0.75
Shore-based	NA	NA	NA	0.08

Length classes over 15.9 m were considered to be industrial vessels and thus not used here.

Country Details

Union of Comoros

Prior to gaining independence in 1975 as the Union of Comoros, the Comoros Archipelago was a French colony. The Comoros Archipelago includes the island of Mayotte, which remains a French colony and is thus examined separately (see **Figure 1** and below). The population of the Union of Comoros is concentrated on three islands: Ngazidja in the northwest, Mwali, located centrally, and Nzwani in the east (**Figure 1**). Comoros declared its 165,000 km^2 EEZ in 1976, and has an *Inshore Fishing Area* of around 1,500 km^2.[2] *Inshore Fishing Areas* are defined as the area that extends from shore to either 50 km offshore or to the 200 m depth contour, whichever comes first (Chuenpagdee et al., 2006). *Inshore Fishing Areas* are thought to represent the majority coastal sea-space along inhabited coastlines within which small-scale fisheries would likely operate (Chuenpagdee et al., 2006; Chuenpagdee, 2011).

Approximately 18% of the population of Comoros was estimated to live below the international poverty line in 2014 (Anon, 2019a). Marine fisheries are crucially important for food security, and exports and imports of fish products are limited, thus most fish are consumed directly in the country (FAO, 2015a). Comoros does not have a truly domestic industrial fleet, or distinct recreational fishing. Fishing is characterized by small-scale artisanal and subsistence activities of local fishers, predominantly using hand lines, gillnets and surface nets.

The detailed estimation methods for the small-scale fishing effort are presented in **Supplementary Material**. There were several data sources stretching back to the mid-1950s for the number of small-scale fishing vessels in the Comoros, with motorization starting in the late 1970s (**Supplementary Table 1**). Several types of vessels are used in the Comoros: the more traditional motorized or non-motorized *pirogues* of up to 7 m length, or the more modern motorized *vedettes* of between 6 and 12 m in length. Data on shore-based fishers was limited, and required assumptions to estimate shore-based fishing activity (**Supplementary Table 2**). Women actively fish and collect seafood (i.e., gleaning) from shore in the Comoros (Hauzer et al., 2013), which we estimated as part of total shore-based effort (**Supplementary Table 2**).

Madagascar

Madagascar represents the eastern boundary of the Mozambique Channel (**Figure 1**), with an EEZ of over 1.2 million km^2 and an *Inshore Fishing Area* (Chuenpagdee et al., 2006) of over 113,000 km^2.[3]

Until its independence in the 1960s, Madagascar was a French colony. Madagascar's political history since then has been marked with multiple regime changes, and the country remained politically unstable (Ploch and Cook, 2012). There is a high rate of poverty, with an estimated 75% of the population living under the international poverty line in 2018 (Anon, 2019b). Although the east is more densely populated, most of the fishing is conducted off the west coast (Le Manach et al., 2012b). In

[2] www.seaaroundus.org/data/#/eez/174

[3] www.seaaroundus.org/data/#/eez/450

Madagascar, marine fishing is an important source of food and exports, particularly in coastal areas where agriculture cannot or is not practised (Le Manach et al., 2012b).

The small-scale fishing sector consists of fishers on foot as well as "traditional" boat-based fishing in wooden dugout canoes (these can be motorized), with both these components engaging in artisanal as well as subsistence fishing. Locally, the artisanal sector is considered to include motorized boats with engine power up to 50–60 HP, including shrimp trawlers and "catchers," dories and collection vessels for fisheries products such as sea cucumbers (Andrianaivojaona et al., 1992; ASCLME, 2011). Although the FAO/Indian Ocean Commission's *Smartfish* program consider smaller shrimp trawlers to be artisanal (Andrianaivojaona et al., 1992; Mngulwi, 2006; Breuil and Grima, 2014b), the *Sea Around Us* follows the definition of Martín (2012) and classifies all trawler catch as being industrial regardless of vessel size (Zeller et al., 2016). Therefore, shrimp trawlers were not included in the calculation of small-scale fishing effort and CPUE in this study.

The detailed estimation methods for the small-scale fishing effort are presented in **Supplementary Material**. There were several data sources for the number of fishing vessels in Madagascar, stretching back to the mid-1960s, with motorization starting in 1970 (**Supplementary Tables S3, 4**). A substantial number of people in Madagascar fish without boats (Andrianaivojaona et al., 1992; Breuil and Grima, 2014b), and the number of shore-based fishers over time was assumed to correlate to the total Madagascan population (**Supplementary Table 3**). There was very little information on the number of days fished per year by small-scale vessels in Madagascar, with reports from the 1980s suggesting between 150 and 220 days per year (Rey, 1982), while shore-based fishers were thought to fish around 20 days per month in 2009 (Barnes and Rawlinson, 2009) or 240 days per year (**Supplementary Table 5**).

Mayotte

Mayotte, a French territory, is an island located east of the Union of Comoros within the Comoros Archipelago (**Figure 1**). The EEZ of Mayotte, declared in 1978, has an area of nearly 63,000 km^2 and an *Inshore Fishing Area* (Chuenpagdee et al., 2006) of roughly 1,600 km^2.[4] Mayotte is the poorest of France's overseas territories; however, the Gross Domestic Product of the island is higher than that of the other countries in the Mozambique Channel, and it is generally considered to be more developed than its neighbors. As a result, Mayotte is a major destination for illegal immigration (Genay and Merceron, 2017). The large and growing population and resultant food security strain on the marine environment pushed Mayotte in 2010 to declare its entire EEZ as a marine protected area (Anon, 2012). Mayotte's population is dependent on marine fisheries as a primary source of protein, and many of its villages are concentrated along the coast (Guézel et al., 2009).

The detailed estimation methods for the small-scale fishing effort are presented in **Supplementary Material**. As a French territory, data for Mayotte are incorporated in national French statistics; however, these data are not easily disaggregated between Mayotte and other French territories in the Indian Ocean. Reported numbers of small-scale fishing boats exist in five categories: small, medium, and large *pirogues*, and *barques*, as well as small longliners in recent years. These data were available for multiple years between 1962 and 2015 from various sources (**Supplementary Table 6**). Motorization started in the 1970s through the introduction of outboard motors for large pirogues (Jacquemart, 1980), with a small proportion of small and medium pirogues being motorized in subsequent years (**Supplementary Table 7**). There was minimal quantitative information on shore-based fishing in Mayotte, and the number of shore-based fishers was assumed proportionate to the total Mayotte population over time. Accurate estimates of the number of fishing days were not available for all small-scale fleet components in Mayotte, and approximations were required based on similar fleet components in Mayotte or the Mozambique Channel region.

Mozambique

Mozambique is located on the southeast coast of mainland Africa and makes up the western border of the Mozambique Channel. The EEZ of Mozambique, declared in 1976, has an area of over 571,000 km^2 and the *Inshore Fishing Area* (Chuenpagdee et al., 2006) is over 68,000 km^2.[5] Formerly a Portuguese colony, the country gained its independence following a decade-long war from 1964 to 1974. Shortly thereafter, a civil war erupted from 1977 to 1992. Since the first democratic elections in 1994, Mozambique has been relatively stable politically, with economic reform and the resettlement of civil war refugees leading to a high growth rate in both the population and the economy (Bueno et al., 2015). However, the country suffers from wealth inequality (Anon, 2018). Since the mid-2010s the country has faced a growing insurgency by Islamist groups, mainly in the northern region of Cabo Delgado, which has increased the political and socio-economic instability. Small-scale fisheries continue to contribute substantially to food security and livelihoods in coastal communities, while government support is often directed to industrial, cash-revenue generating fisheries (Jacquet and Zeller, 2007; Jacquet et al., 2010). Wooden canoes make up most of the small-scale fleet, and the most widespread gears used are gillnets and beach seines (Oceanic Développement and Mega Pesca Lda, 2014).

The detailed estimation methods for the small-scale fishing effort are presented in **Supplementary Material**. There were data for the number of fishing vessels in Mozambique for multiple years back to the early 1950s (**Supplementary Table 8**), and unlike the other countries/territories in this study, motorization started before 1950 (FAO, 1958). The total number of shore-based fishers were reported for multiple years since 1995, but required adjustments to exclude freshwater statistics from the combined data in 2007 and 2012. Prior to 1995, the number of shore-based fishers was assumed to correlate with the total population back to 1950 (**Supplementary Table 9**). This proportion was adjusted across several time periods to account for increased migration to coastal regions due to the internal conflicts. As described in

[4]www.seaaroundus.org/data/#/eez/175

[5]www.seaaroundus.org/data/#/eez/508

Supplementary Table 10, the average number of days per year spent fishing was approximated based on Hara et al. (2001) and insights from Jacquet and Zeller (2007).

Catch per Unit Effort

Catch per unit of effort is a basic fisheries science measure often used as a first-order evaluation of broad trends in likely relative abundance or relative biomass trends over time of the underlying fish stocks (Skalski et al., 2005; Belhabib et al., 2018). While CPUE has its limitations, it can be useful in long-term trend monitoring of a fishery, particularly where more detailed data and stock assessments do not exist (Stamatopoulos, 2002) and open or semi-open access fisheries predominate, i.e., where restrictive fisheries management actions may be absent or ineffective. Here, the small-scale fisheries catches for each country/territory in the Mozambique Channel region were extracted from the *Sea Around Us* global reconstructed catch database (see text footnote 1) for 1950–2016 (Pauly and Zeller, 2016a,b). These catch data were previously and independently reconstructed for the Union of Comoros (Doherty et al., 2015a), Mayotte (Doherty et al., 2015b), Mozambique (Jacquet and Zeller, 2007; Jacquet et al., 2010), and Madagascar (Le Manach et al., 2011, 2012a,b), and conservatively updated to 2016 (Derrick et al., 2020). CPUE is the quotient of the annual small-scale total catch per country as derived independently through *Sea Around Us* catch reconstructions and the annual effective small-scale fishing effort as reconstructed here.

RESULTS

Fishing Effort

Effective small-scale fishing effort in the entire Mozambique Channel region grew slowly but steadily from around 386,000 kWdays in 1950 to around 2.6 million kWdays in the mid-1970s, increasing more steeply thereafter and reaching approximately 23 million kWdays by 2016 (**Figure 2A** and **Table 2**). Thus, overall small-scale fishing effort in the Mozambique Channel region increased nearly 60-fold since 1950. This increase was dominated by growth in fishing effort in Madagascar since the mid-1980s and in Mozambique since the 1990s, although the other countries/territories also grew in effective effort over the 67-year period considered here (**Figure 2A**).

For the region as a whole, shore-based effective fishing effort was almost equal to boat-based effective fishing effort in the first few years, with boat-based effort growing 96-fold from around 220,000 kWdays in 1950 to approximately 21 million kWdays by 2016 (**Figure 2B** and **Table 2**). This substantial growth in boat-based fishing effort was driven by massive increases in the number of boats, as well as the motorization trend in many of the countries after 1970, especially the Comoros and Mayotte (**Figures 2C–E** and **Table 2**). Only Mozambique maintained a substantial shore-based effort trend over time, although boat-based effort increased strongly after the late 1990s (**Figure 2F**). Importantly, however, both Mozambique and Madagascar seem to be experiencing an increase in shore-based fishing effort in the most recent years (**Figures 2D,F**).

Catch per Unit Effort

The previously and independently reconstructed small-scale marine catches for the Mozambique Channel countries/territories displayed an overall fivefold increase over the 1950–2016 time period, growing from approximately 67,000 tonnes in 1950 to over 350,000 tonnes in 2016 (**Figure 3A**). Small-scale catches in the Mozambique Channel region were dominated by Mozambique and Madagascar (**Figure 3A**), which was to be expected, given their substantially larger populations, long coastlines and large coastal fishing areas. Small-scale catches in the Comoros Islands increased slowly from around 1,000 tonnes in 1950 to just over 4,000 tonnes in 1978, before increasingly strongly through the 1980s and early 1990s. Growth in catches slowed through the 1990s, and catches plateaued around 18,500 tonnes year^{-1} in the mid-2010s (**Figure 3B**). The catches of small-scale fisheries in Madagascar grew near linearly from around 14,000 tonnes in 1950 to 136,000 tonnes in 2016 (**Figure 3C**). Catches in Mayotte increased from 237 tonnes in 1950 to just over 2,600 tonnes in 1995 (**Figure 3D**). Thereafter, catches entered a period of strong fluctuations throughout the late 1990s and early 2000s, before increasing again strongly to over 3,100 tonnes by 2016 (**Figure 3D**). Mozambique's small-scale catches were strongly influenced by the civil war driven coastal migrations during the 1960–1990 period (Blythe et al., 2013), with catches first increasing from around 51,500 tonnes in 1950 to an initial peak of just under 147,500 tonnes in 1979 (**Figure 3E**). Thereafter, catches declined to approximately 79,700 tonnes in 2005, before increasing again during the 2000s and especially since 2013 to a new all-time peak of just over 192,500 tonnes in 2016 (**Figure 3E**).

Overall, the small-scale CPUE estimates for the entire Mozambique Channel region, as derived from the combination of the previously reconstructed catches and presently reconstructed fishing effort data suggested a decline of 91% between 1950 and 2016, from around 174 kg·kWday^{-1} in 1950 to just over 15 kg·kWday^{-1} by 2016 (**Figure 4A**). The strongest decline in CPUE was observed during the 1950s, after which the rate of decline slowed until the mid-1980s; thereafter, the CPUE continued to decline at a slower rate (**Figure 4A**).

The small-scale fisheries CPUE in the Comoros Islands initially experienced an increase, from 19.3 kg·kWday^{-1} in 1950 to the time series maximum of 27 kg·kWday^{-1} in 1962, after which the CPUE declined strongly to a low of 6.3 kg·kWday^{-1} in 1989 (**Figure 4B**). The CPUE recovered slightly in the early 1990s before starting a steady decline to the all-time low of 4.3 kg·kWday^{-1} by 2016. Overall, the small-scale CPUE in the Comoros Islands declined by approximately 78% between 1950 and 2016 (**Figure 4B**). The CPUE for the small-scale fisheries on Madagascar stayed relatively constant at a high level of around 110 kg·kWday^{-1} during the 1950s and 1960s, reaching a time series maximum of 117 kg·kWday^{-1} in 1969 (**Figure 4C**). Starting in 1970, the CPUE began a strong decline to around 32 kg·kWday^{-1} in the late 1980s, after which the decline tapered off somewhat, before leveling out at around

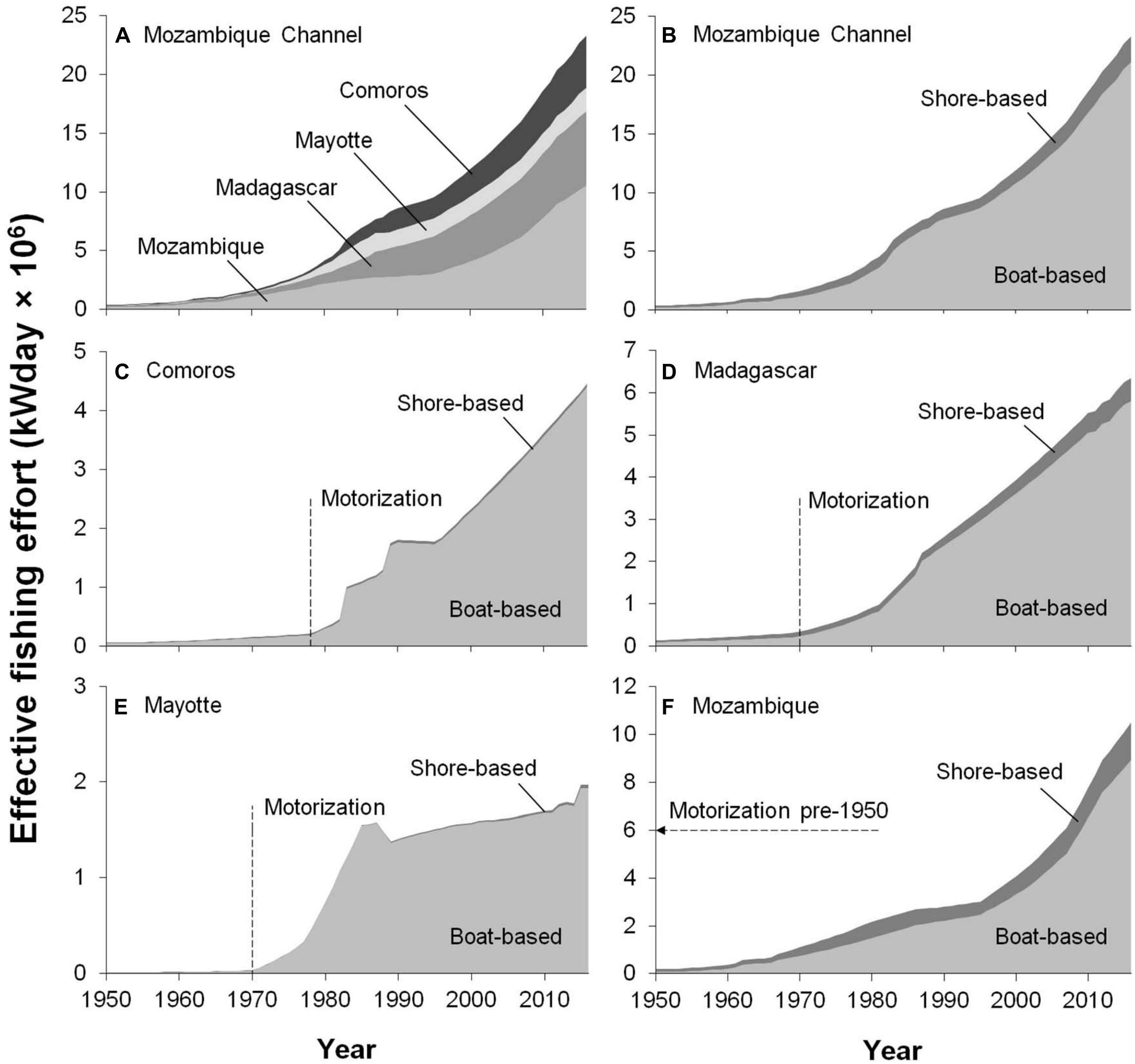

FIGURE 2 | Annual effective fishing effort (kWday × 10^6) between 1950 and 2016 for small-scale fisheries in **(A)** the Mozambique Channel region by country; **(B)** the Mozambique Channel region by boat- versus shore-based fisheries; **(C)** Comoros; **(D)** Madagascar; **(E)** Mayotte; and **(F)** Mozambique. Start year of motorization is indicated.

TABLE 2 | Effective fishing effort (kWdays) of the four Mozambique Channel entities and the entire Mozambique Channel region examined for 1950 and 2016.

Area	Shore-based effective fishing effort (kWdays × 10^6)		Boat-based effective fishing effort (kWdays × 10^6)		Total effective fishing effort (kWdays × 10^6)	
	1950	2016	1950	2016	1950	2016
Comoros	0.010 (6%)	0.06 (3%)	0.05 (22%)	4.40 (21%)	0.06 (15%)	4.46 (19%)
Madagascar	0.045 (27%)	0.56 (25%)	0.08 (39%)	5.79 (28%)	0.13 (34%)	6.35 (27%)
Mayotte	0.001 (1%)	0.03 (1%)	0.01 (3%)	1.94 (9%)	0.01 (2%)	1.97 (8%)
Mozambique	0.111 (66%)	1.57 (71%)	0.08 (36%)	8.93 (42%)	0.19 (49%)	10.51 (45%)
Region	0.17	2.22	0.22	21.07	0.39	23.29

Percentage values indicate the contribution to the total regional effective fishing effort of each component in each year.

22 kg·kWday^{-1} by 2016 (**Figure 4C**). The overall decline of small-scale CPUE on Madagascar was 80% over the 60+ year time period considered here. The CPUE of Mayotte's small-scale fisheries declined starting in 1950 from 30 kg·kWday^{-1} through the 1950s before tapering off in the late 1960s at around 21 kg·kWdays^{-1} (**Figure 4D**). The introduction of motorization

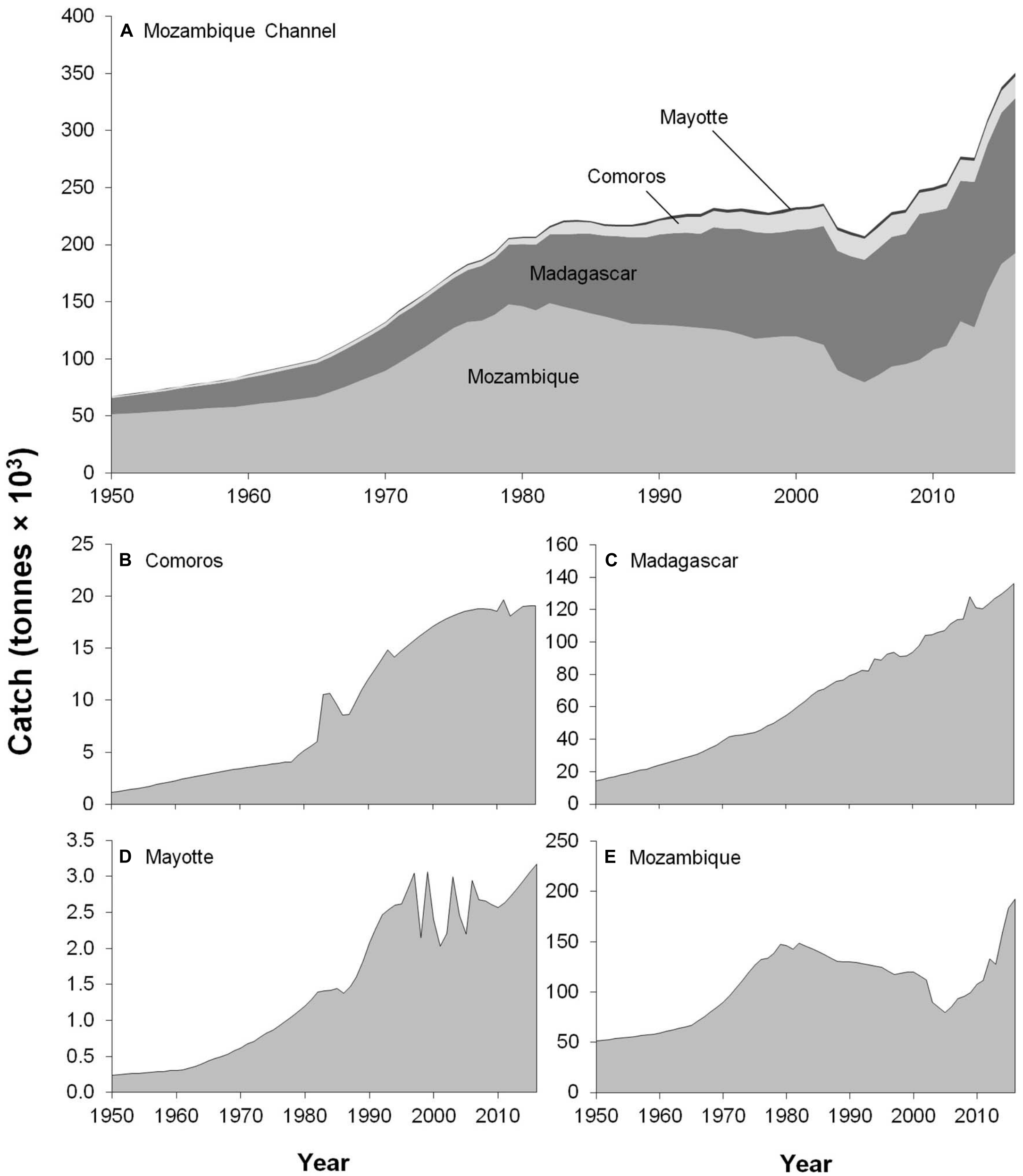

FIGURE 3 | Previously and independently reconstructed catches (tonnes × 10^3) for small-scale fisheries from 1950 to 2016 for **(A)** the Mozambique Channel region by country; **(B)** Comoros; **(C)** Madagascar; **(D)** Mayotte; and **(E)** Mozambique.

in Mayotte's fleets after 1970 led to a very rapid and massive decline in the derived CPUE in the early-1970s, before reaching a very low level of around 1.5 kg·kWday^{-1} in the early 1980s (**Figure 4D**). The CPUE fluctuated slightly around this level ever since, resulting in an overall CPUE decline of 95% since 1950 (**Figure 4D**). The CPUE of the small-scale fisheries in Mozambique declined strongly and rapidly from very high levels of 272 kg·kWday^{-1} in 1950 to just over 100 kg·kWday^{-1} by the early 1960s (**Figure 4E**). It continued to decline more gradually over the remainder of the time period to reach very low levels of around 13 kg·kWday^{-1} by the early-2010s, before increasing again very modestly to 18 kg·kWday^{-1} by 2016 (**Figure 4E**). Overall, this resulted in a decline in CPUE of 93% since 1950 (**Figure 4E**).

DISCUSSION

A considerable variety of different and non-standardized measures of fishing effort are used around the world, which makes global comparisons difficult. Fortunately, a standardized measure of effort using the power input in fisheries, i.e., kWday,

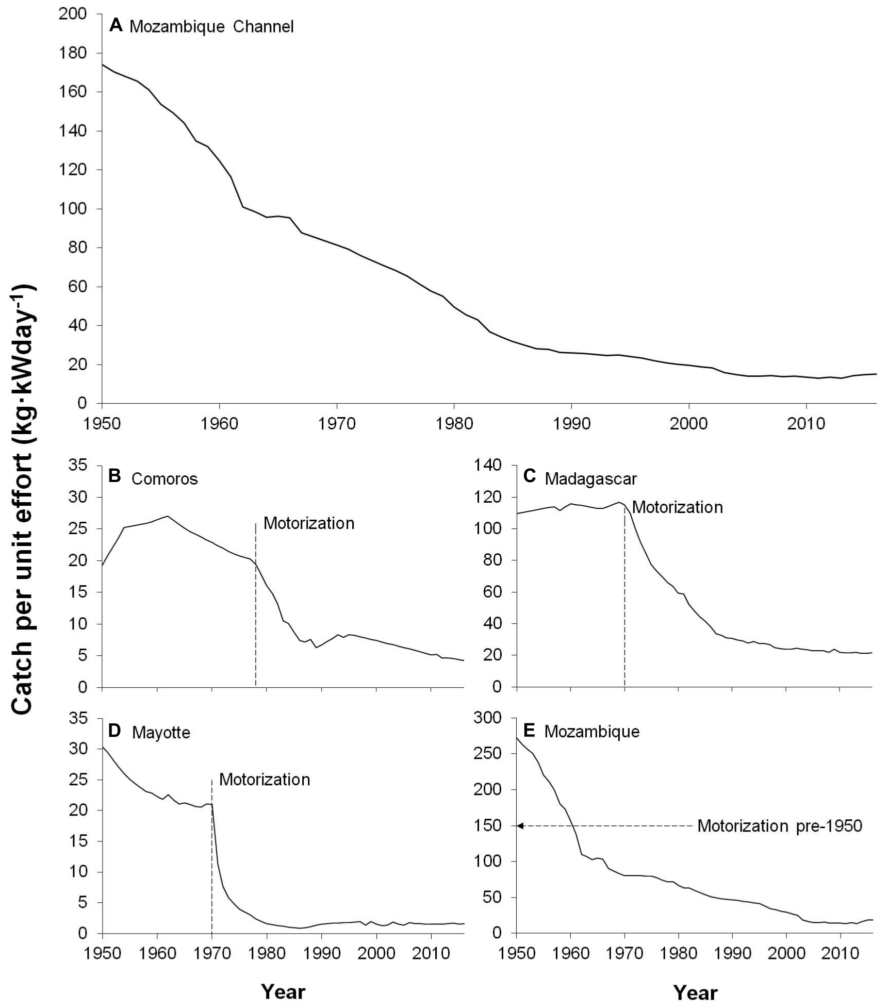

FIGURE 4 | The catch per unit effort (CPUE) for small-scale fisheries (kg·kWday^{-1}) from 1950 to 2016 for **(A)** the complete Mozambique Channel region; **(B)** Comoros; **(C)** Madagascar; **(D)** Mayotte; and **(E)** Mozambique. Start year of motorization is indicated.

is now available globally (Parker et al., 2018; Greer et al., 2019a,b), enabling international comparisons. This finally permits direct comparisons between fisheries, gear-types, regions countries, and globally to be undertaken in a standardized fashion. Here, we used a data reconstruction approach (Zeller et al., 2016) to refine and update the preliminary fishing effort data of Greer et al. (2019a) for small-scale fisheries from 1950 to 2016 in the four inhabited countries or territories that make up the Mozambique Channel region off East Africa (**Figure 1**). Furthermore, we combined the small-scale fishing effort time series data derived here with previously and independently estimated small-scale catch data for these countries/territories (Jacquet and Zeller, 2007; Jacquet et al., 2010; Le Manach et al., 2011, 2012b; Doherty et al., 2015a,b,c), obtained using the catch data reconstruction approach of Zeller et al. (2016), to develop CPUE time series for the small-scale fisheries in the Mozambique Channel region from 1950 to 2016.

Overall, our results illustrated consistent and strong increases in fishing effort over the 67-year period considered here. This growth in effort was driven not only by increasing motorization, but also by considerable increases in the number of small-scale fishing boats being used. Crucially, the combined catch and fishing effort data clearly demonstrated consistent and strong declines in the CPUE of small-scale fisheries in every country or territory examined, with CPUE declining by 91% across the region. The geographically smaller entities of Comoros and Mayotte had CPUE declines of 78 and 95%, respectively, while CPUE declines of 80 and 93% were observed for Madagascar and Mozambique, respectively. Declining CPUE time series trends, especially in the absence of effective effort-controlling fisheries management, generally suggest declining relative abundances of the underlying fished stocks (Hoggarth, 2006). Thus, the pattern of CPUE observed here suggests a strongly declining fisheries resource base for the small-scale fisheries in these countries/territories over the last 60+ years, impacting a crucial food security sector in this region of Africa. This trend is in general agreement with a recent assessment of the biomass patterns of exploited stocks for the tropical Indian Ocean region, which suggested general declines in stock biomass of around 60–70% from levels in 1950 (Palomares et al., 2020). Furthermore, the overall levels of catches, including industrial and foreign fishing, in the Agulhas Current Large Marine Ecosystem, which includes the Mozambique Channel entities considered here, have been declining steadily since peaking in the late 1960s (**Figure 3** in Zeller et al., 2020).

Effective fishing effort was calculated here using four core fishing capacity parameters: (1) the number of boats, (2) the length of boats, (3) the engine power (kW) of boats, with non-motorized boats being assigned an approximate human-power equivalent, and (4) the number of days boats fish in a year (fishing days). Shore-based small-scale fishing without boats was also included in the fishing effort estimation, using the number of shore-based fishers, shore-based fishing days and a human-power equivalent. This ensured comprehensive coverage of all small-scale fishing in each country and territory. Estimating broad-scale fishing effort based on indirect methods such as used here relies on assumptions that, no matter how carefully and conservatively made, are inherently uncertain, which has generated some debate in regards to the potential accuracy of the data (Greer et al., 2019b; Ziegler et al., 2019). However, our extensive use of a range of available fishing capacity parameters sourced at the fisheries- and country-level may provide some advantages over more generalized estimation techniques, and is thought to provide a reasonable representation of the fishing effort trends in the region over time.

The increase in small-scale fishing effort we documented here was the combined result of growth in both the number of boats as well as motorization of fleets. The introduction and popularization of motorization of traditional vessels, and the introduction of newer, larger powered vessels were important factors driving fishing effort trends. Mayotte, for example, had the lowest effective fishing effort in 1950, with no motorized fishing vessels. With the introduction of motorization after 1970, effective fishing effort steeply increased over the subsequent decade as fleets rapidly motorized. In contrast, the small-scale fishing fleets in Madagascar were the largest in terms of number of boats, and although its effective fishing effort is still increasing due to continuing growth in the number of non-motorized vessels, the very low proportion of vessels that are motorized (0.03% in 2016) limits the fishing power and fishing capacity of its large small-scale fleet.

Clearly, the four parameters used here to estimate effective fishing effort do not necessarily capture all aspects of vessel and fishing capacity. Most fishing effort models, including the one we used here, do not implicitly account for the impacts of technological advances on effort measures over time (i.e., technology creep), such as the introduction and spread of synthetic material in fishing gear, sonar, GPS, refrigeration or fish aggregating devices (Palomares and Pauly, 2019). Technology creep is pronounced in industrial fishing fleets over time, and is likely to have increased the fishing capacity of industrial vessels in the region, further contributing to overfishing and general decline in stock biomass in the Mozambique Channel. However, the available literature suggests that the small-scale fleets in the Mozambique Channel region may not have implemented many of these capacity-enhancing technologies at this point, other than motorization and the use of synthetic fishing gear materials. For example, in the Comoros and Mozambique, refrigeration is restricted to only a few boats (Chacate and Mutombene, 2013; Breuil and Grima, 2014a), and Madagascar's fleet is largely non-motorized and thus unlikely to have widespread refrigeration. Such technological advances do result in effective effort growth of fleets over time even when the number and size of vessels remains the same, and would need to be accounted for in long time series comparisons (Palomares and Pauly, 2019; Scherrer and Galbraith, 2020). Due to the general absence, for the time being, of major technological advances in the small-scale fleets considered here, other than motorization and synthetic materials, the present study did not include any adjustment factors for technology creep (Palomares and Pauly, 2019; Scherrer and Galbraith, 2020).

A limitation of the approach used by us is the assumption of equal capacity and predictive ability of vessel length to estimate effective fishing effort *between gears*; this may not always be reflective of reality. For example, while vessel characteristics such as length and engine power alone may predict effective fishing effort well for some gears, e.g., for industrial trawlers (which were not evaluated here), effective fishing effort for vessels using, e.g., longlines is more often dependent on the length and number of longlines deployed and the number of hooks used per line rather than vessel length or engine power alone (Bell et al., 2017). Furthermore, for estimating overall effort for longline fishing, one will need to include the effective effort (in kWday) of the vessels which caught the bait used by the longline vessels, which is most likely proportional to the number of hooks used overall across all sets. Until recently, it was deemed not viable to estimate cumulative effective fishing effort of all gears and fleets across all countries, as standardized global conversions between different traditional measures of effective fishing effort were lacking. Fortunately, this has now been addressed *via* the standardization of fishing effort as kWday, as shown here and in other recent examples (Piroddi et al., 2015; Belhabib et al., 2018; Greer et al., 2019a,b). Fundamentally, in order to catch fish, one needs an input of energy. Prior to motorization, this energy was derived through human power (e.g., oars) or wind-power, which in our study was approximated through a human-power equivalent. Since motorization, most of this energy is derived from fuel, which has become the most expensive input in fishing. Therefore, using kWday as the key fishing effort measure for all gears is reasonable, and also provides an immediate indication of the fishing cost. Expressing the returns (i.e., CPUE) in $kg{\cdot}kWday^{-1}$ allows easy and standardized comparison and visualization across gears, fleets, countries, and time periods. The direct connection between this measure of effort and fuel use, and hence fuel cost as well as CO_2 emissions, also allows fishing effort and CPUE to be readily re-expressed in economic and climate impact terms, such as $kg{\cdot}\$^{-1}$ of fuel cost and tCO_2 per unit effort or unit catch.

Here, data from boats using various different gear types were used to estimate effort from length and engine power, and as such the kW per vessel predicted by length is thought to represent the average vessel capacity. Because small-scale fisheries often may switch between various fishing gears, and motorized small-scale vessels most commonly use outboard motors with lower engine power (kW) than other types of motorization, the impact of gear type on average vessel capacity is not expected to greatly impact our results.

Infrequent and incomplete data on boat types and lengths were commonplace in our study, and fishers who do not use vessels, particularly women (Harper et al., 2013, 2017, 2020), are often not considered at all in many countries' data and information sources (Hauzer et al., 2013). Women fishers are frequently underrepresented in fisheries statistics and policy (Harper et al., 2013, 2017, 2020). Many traditional perspectives of what constitute "fisheries" and "fishers" tend to exclude or downplay small-scale activities such as collection of seafood from shore (gleaning) from the definition of fishing, when in reality there is a long tradition of women and children collecting seafood from shores in Mozambique Channel coastal communities (e.g., Hauzer et al., 2013) and elsewhere (Harper et al., 2013, 2020). We have accounted for the contributions of shore-based fishers in this study as best as possible based on available information. This is particularly important as shore-based fishing may represent the major source of protein and micro-nutrients to the poorest sectors of coastal communities (Golden et al., 2016; Hicks et al., 2019; Pauly, 2019), which may not have access to boats or elaborate fishing gear. A greater emphasis should be given by governments' fisheries departments and by the scientific community to the recording, estimating and inclusion of all small-scale fisheries in studies and data to evaluate the impact, the food security and nutrient security as well as livelihood importance of these fisheries as a whole (Vianna et al., 2020).

This study focused on the small-scale fisheries in the Mozambique Channel region of East Africa only, and did not account for recreational or large-scale, industrial fisheries, whether domestic or foreign. Resource overlap between the small- and large-scale sectors is nearly global in its occurrence and constitutes a growing source of conflict in many regions (Le Manach et al., 2012b; Belhabib et al., 2014). Small-scale and industrial fishers both often target similar species, e.g., tuna and tuna-like species in the Comoros (Breuil and Grima, 2014a), or shrimp in Madagascar (Cripps, 2009), as well as likely elsewhere. Such resource competition, if not strictly controlled and managed, will continue to increase the political and socio-economic marginalization of small-scale sectors (Pauly, 2006).

African countries are also particularly vulnerable to illegal and unregulated foreign fishing (Kurien and López Ríos, 2013). The waters of Comoros, Mozambique, and Madagascar have all been targeted by Asian tuna longline fleets with and without prior access agreements (Anon, 1995; Cox, 2012; Breuil and Grima, 2014a,b). When agreements do exist for foreign fleets, monitoring, compliance and enforcement is difficult and generally very limited or widely absent, and the catch is often not landed or processed in the host country (UNCTAD, 2017). For example, tuna caught by European Union vessels in Comoros waters are not landed in the Comoros, nor are there fully trained onboard observers from the Comoros aboard these vessels, despite it having been shown that full observer coverage is necessary for equitable resource use (Zeller et al., 2011). Thus, the foreign tuna fisheries in the Comoros does not create any employment or livelihood support for the local population, despite the agreement between the European Union and Comoros being fair and transparent, and the access fees collected promoting modernization of the domestic fishing sector (UNCTAD, 2017). On the other hand, while European Union fishing quotas in Madagascar had increased by 30% between 1986 and 2010, the fees paid by the European Union to Madagascar had decreased by 20% (Le Manach et al., 2013). Governments in this region have historically lacked the organizational structures and resources to engage effectively in fisheries governance (Cox, 2012), in part due to decades of political instability. Considering the often-overlooked importance of small-scale fisheries to domestic livelihoods and food security in coastal communities, the countries and territories in the Mozambique Channel region may want to carefully consider the interaction between domestic and foreign fisheries when renewing or establishing future fishing agreements.

Given the considerable increases in effective fishing effort observed here for small-scale fisheries in the Mozambique Channel region, it is not surprising that the CPUE has been declining substantially for decades now. The findings of this study are consistent with previous suggestions by Watson et al. (2013) who, based on a shorter time series (1950–2006), suggested that CPUE had been declining globally since 1950, including in the Indian Ocean. Declining CPUE trends in the absence of effective effort restrictions generally are indicators of declining relative abundance of fished stocks (Hoggarth, 2006). Given the country-level focus in the present study, it is challenging to draw specific conclusions on the status of any particular stock or species, but the overall declining CPUE in all EEZs points to serious concerns about the status of the fish stocks underlying and supporting small-scale fisheries in the Mozambique Channel region, as it does elsewhere in the western Indian Ocean (Christ et al., 2020). We suggest that this decline in CPUE should be viewed as a serious warning sign of decreased abundance and biomass of the exploited fish populations. Thus, local governments should consider implementing policies to promote recovery of fish populations, i.e., stock rebuilding, to abundance levels that readily support maximum sustainable yields (Pauly and Froese, 2020). This will require minimizing or restricting further growth in fishing effort until the biomass and abundance of local stocks can recover. Enforcing gear restrictions (e.g., larger mesh and hook sizes) and permanent no-take zones would assist in rebuilding fish populations. Such effort and spatial access restrictions should also apply to foreign fleets due to the limited direct local benefits provided and possible interaction and conflict with domestic small-scale fisheries. Given the importance of small-scale fisheries in the Mozambique Channel, the adoption and enforcement of such policies could result in considerable improvements in food and nutrient security as well as the socio-economic condition of coastal communities in the region.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

DZ developed and formalized the fisheries data reconstruction approach and conceptualized, drafted, revised, and edited the manuscript. DZ, MP, and DP conceptualized the development of the fishing effort data reconstruction, developed the methodological approach, and conceptualized, edited, and revised the manuscript. MA assembled catch data and reconstructed effort data, synthesized CPUE data, prepared the figures and **Supplementary Material**, and drafted, revised, and edited the manuscript. KG advised on effort reconstruction methods. AC, BD, GV, and S-LN contributed to the effort data reconstruction, assembled catch data, and edited the manuscript. AZ collected and reconstructed fishing effort data and edited the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.707999/full#supplementary-material

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Angling to Reach a Destination to Fish—Exploring the Land and Water Travel Dynamics of Recreational Fishers in Port Phillip Bay, Australia

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Recreational fishing is a popular pastime and multibillion dollar industry in Australia, playing a key economic role, especially in regional areas. In the State of Victoria, Port Phillip Bay (PPB), bordered by Melbourne and its suburbs, is the largest of the State's marine recreational fisheries. At present, little is known about the spatial and temporal dimensions of angler travel from origins to destinations, and the applicability of such spatial knowledge in fisheries management. To address this lack of information we assessed spatiotemporal dynamics and patterns in fishing trips, based upon travel distances on land and water, to acquire insight into the spatial ranges over which anglers residing in various locations travel to fishing destinations in the environs of PPB. Data for each angler per fishing trip, from 6,035 boat-based creel surveys, collected at 20 boat ramps in PPB during a 10-year period from 2010 to 2019, were analyzed by applying geospatial modeling. Differences were observed in both land and water travel distance by region and popular target species, with anglers who launched from Bellarine region traveling further on land, and those who targeted snapper traveling further on water. It was also evident that most anglers resided within close proximity of PPB, often less than 50 km, although some anglers traveled long distances across the State to access fishing locations, particularly when targeting snapper. This work further highlights the importance of spatially explicit approaches to inform fisheries management by identifying users across different landscape and seascape scales, and out-of-region or State fishing trips, which may especially impact coastal communities and benefit local businesses.

Keywords: recreational fishing, angler behavior, travel dynamics, spatial modeling, network analysis, fishing destination

INTRODUCTION

Recreational fishing is a popular pastime with profound socio-cultural impacts, and it contributes considerably to the world economy (Cisneros-Montemayor and Sumaila, 2010; Cooke et al., 2018; Hyder et al., 2018; Lewin et al., 2019). Globally, more than 100 million people participate in recreational fishing annually, landing ∼0.9 million tonnes per annum with participation and catch rates differing substantially among countries (Pauly and Zeller, 2016; Freire et al., 2020). Although annual recreational landings are not large (∼1%) compared to the commercial catch (> 100 million tonnes per year), value adding derived from recreational activities can create substantial economic onflows (Cisneros-Montemayor and Sumaila, 2010; Potts et al., 2020).

There has recently been concerted effort, mainly in the industrialized world, to develop comprehensive governance structures for recreational fisheries given that most people fishing today do so recreationally (Arlinghaus et al., 2015, 2019). Yet, the recreational fishing sector generally lags behind the commercial sector in their understanding of fisheries policy and management. This can be best redressed by involving recreational fishers in monitoring and decision-making processes and regulatory agencies can improve their engagement of anglers by gaining a better understanding of recreational fisher behavior (Hunt et al., 2013; van Poorten and Camp, 2019). The underpinning operational policy objective is to enable fishing stakeholder needs to be met whilst satisfying community expectations for sustainability and socio-economic benefits[1], particularly within proximity of urbanized coastal areas that provide easy access to marine resources. This is of particular importance given that in Australia the amount of recreational catch may exceed commercial catch in highly populated coastal areas, or that angling may be the only, or dominant, recreational activity supporting tourism and holiday visitation within less populous areas (McPhee et al., 2002; McPhee, 2017). In this country, recreational fishing is very popular compared to global norms with > 19.5% of the population partaking, which is facilitated by highly efficient implementation of recreational fisheries policies and promotional initiatives (Cooke and Cowx, 2004; McPhee, 2017; Cooke et al., 2018; Lynch et al., 2019).

A major recreational fishing industry (i.e., tackle, bait, equipment, charter, and boat sales) exists in the State of Victoria and the direct and indirect economic output of its recreational fishing activities was estimated to be about AUD $7.5 billion in 2018/19 arising from more than six million fishing trips, generating > 14,000 direct jobs (Huang et al., 2020; VFA, 2020). Centrally located in Victoria, Port Phillip Bay (PPB; **Figure 1**) is a large marine embayment bordered by Greater Melbourne and Geelong regions extending along the Mornington and Bellarine Peninsulas on its eastern and western sides, respectively (ASGS, 2021). Due to PPB's sheltered waters, diversity of habitats, and proximity to Melbourne and suburbs, with a population exceeding 4 million (ABS, 2016), recreational fishing is extremely popular. Thus, PPB plays a key role from both an urban and regional perspective because it creates numerous tourism and economic opportunities (Sampson et al., 2014). Although PPB supports diverse ecologically and economically important species, the main species of interest to recreational anglers are snapper (*Pagrus auratus*), southern calamari (*Sepioteuthis australis*), King George whiting (KGW) (*Sillaginodes punctatus*), and sand flathead (*Platycephalus bassensis*).

Little is known about angler behavior and travel characteristics within space and time in PPB. Understanding angler behavior and responses to the implementation of varying policies and strategies is important due to the critical role that some anglers play in many aspects of fisheries governance (Fulton et al., 2011; Hunt et al., 2013; Camp et al., 2018; van Poorten and Camp, 2019). Spatial dynamics in fisheries resources, including spatiotemporal patterns in catch and effort have been widely investigated in fisheries science (McCluskey and Lewison, 2008; Stelzenmüller et al., 2008; Stewart et al., 2011; Post and Parkinson, 2012; Aidoo et al., 2015; Jalali et al., 2015, 2018). However, less attention has been devoted to the anglers' behavioral dimensions of their recreational fishing activities such as their travel distance, trip dynamics and fishing behaviors, largely due to the paucity of information available in most instances. This is of significance because the spatial context of the socioecological systems in which marine recreational fisheries operate can impose additional challenges to fisheries management and regulatory agencies due to heterogeneity of human uses and biological communities as well as key dynamic processes that maintain them (Pereira and Hansen, 2003; Carpenter and Brock, 2004; Crowder et al., 2006; Crowder and Norse, 2008; Lorenzen et al., 2010).

Economists and human geographers have traditionally used data from angler travel to assess the value of recreational fisheries and to evaluate the impacts of fishing on landscapes (Post et al., 2002, 2008; Hunt, 2005; Post and Parkinson, 2012). However, having a detailed understanding about the spatial dimensions of angler travel patterns is of further significance as it can assist to determine the spatial size, extent, and boundaries of areas targeted by anglers. Spatially explicit marine planning policies have been promulgated as a means for more effective fisheries management delivered at geographic scales that align with fish stock boundaries and varying demographics of anglers that harvest those stocks (Young et al., 2007; Pomeroy and Douvere, 2008; Klein et al., 2010; Lorenzen et al., 2010; Stelzenmüller et al., 2013; Camp et al., 2018). Spatially resolved approaches can more meaningfully engage stakeholders in decision making processes (Pomeroy and Berkes, 1997; Olsen et al., 2011), and the identification of the spatial origins of anglers targeting certain fishing grounds can foster effective allocation of resources. Moreover, given that the statistical properties of angler travel dynamics may differ considerably among target species, across regions, or through time, greater resolution can provide improved insights about how fishing effort distributes across landscapes and how it can contribute to local economies.

Challenges in understanding angler behavior stem from a lack of information which cannot be addressed in the absence of a comprehensive and well-structured monitoring program that acquires detailed multi-year data about travel dynamics. In this context, annual creel surveys collecting a variety of angler demographics and fishing behavior related data in PPB were undertaken. The primary objectives of this study were to use these data to estimate and describe anglers' travel distances from residential postcodes to boat ramps and from boat ramps to fishing locations, and to find out how these differ temporally, spatially and by target species, to assist resource managers and policymakers in making evidence-based decisions regarding recreational fisheries in PPB.

[1] https://vfa.vic.gov.au/operational-policy; accessed 23/12/21.

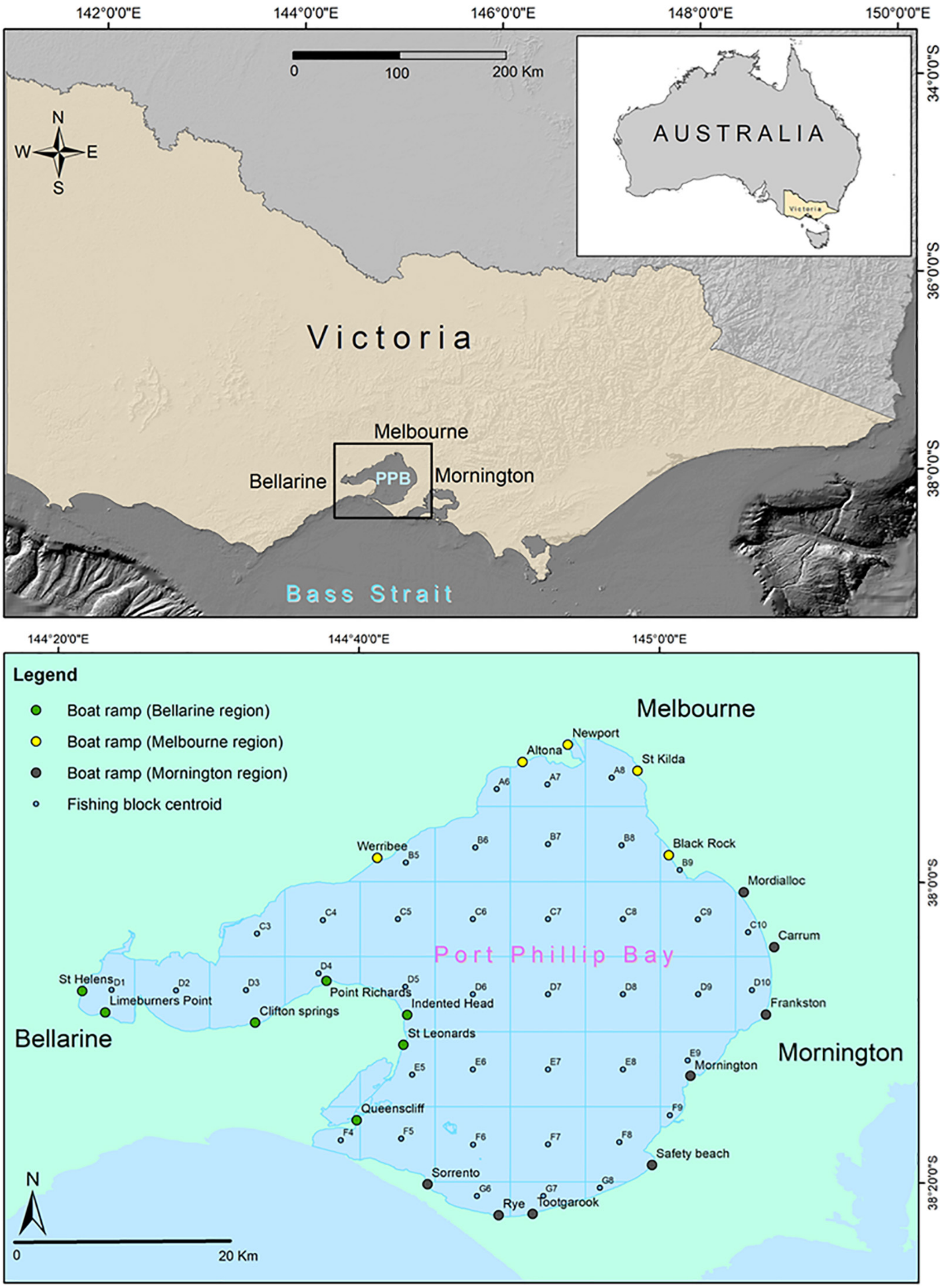

FIGURE 1 | Map of the study area overlaid across the shaded relief of land and sea, zoomed over the State of Victoria and Port Phillip Bay in the lower panel showing the three residential regions of Bellarine, Melbourne and Mornington. Fishing block codes and boat ramp names are also shown.

MATERIALS AND METHODS

Survey Area

Port Phillip Bay (PPB) is centrally located on the southern coastline of Victoria (**Figure 1**). It is a large marine embayment with an area of approximately 1,930 km^2 and 264 km shoreline with an average depth of 13 m, and the deepest part being ∼24 m. PPB connects to Bass Strait through a narrow deep channel (∼3.5 km across but up to nearly 100 m deep × 800 m wide at its most navigable passage)

TABLE 1 | Fishing area and the proportion of anglers surveyed as well as the proportion of anglers in age groups for each fishing area based on 6,035 fishing trip records in Port Phillip Bay (Victoria) from 2010/2011 to 2018/19.

Region	Proportion (%) of anglers in each age group				Proportion (%) of anglers
	Age < 18	Age 18–49	Age 50–69	Age ≥ 70	
Bellarine	2.5	46.4	48.5	2.6	33.0
Melbourne	5.4	63.3	27.4	3.9	36.3
Mornington	3.7	69.2	26.6	0.5	30.7

exchanging oceanic water (Holdgate et al., 2001; Sampson et al., 2014).

Creel Survey Data

Creel surveys of boat-based recreational fishing are conducted annually in PPB. The surveys are undertaken at 20 actively used boat ramps (i.e., Clifton Springs, Limeburners Point, St Leonards, Queenscliff, St Helens, Point Richards, Indented Head, Altona, Werribee, St Kilda, Newport, Black Rock, Carrum, Sorrento, Mornington, Rye, Mordialloc, Safety Beach, Frankston, Tootgarook) around PPB's coastline that can be divided into three regions (Melbourne, Bellarine and Mornington) based on geography and urbanicity (**Table 1**). The sampling design followed the approach described by Chen and Woolcock (1999) to ensure that estimates of fishing effort are unbiased. Creel surveys are conducted when anglers return from a fishing trip, mostly on weekends and during peak fishing days e.g., public holidays over a 6-month period (November to April), mainly to monitor harvest rates of key recreationally important species (Ryan and Conron, 2019). During survey interviews, anglers are asked to provide their residential postcode and information about their completed trip including fishing block, target species and the amount of time that they spent fishing, and if an angler changed their fishing block or target species during the fishing trip, then the main fishing block or target species was recorded. In this study, creel survey data were obtained from the years 2010/2011 to 2018/19 and only properly completed interviews with all questions answered were used for analyses. Most of the survey data (approximately 85%) related to frequently targeted species: snapper, southern calamari, King George whiting (KGW), and flathead (*Platycephalus spp.)*. Sand flathead and southern bluespotted flathead (*Platycephalus speculator*), were combined as "flathead" given that the two species are often caught concurrently and most anglers do not differentiate between them. Due to the limited number of fishing trips where other species were targeted, only the four abovementioned species were used to evaluate patterns and differences in travel distance. This resulted in 6,035 of the 7,495 fishing trip records being analyzed across three survey regions (i.e., Bellarine, Melbourne and Mornington) based on residential postcode.

Fishing blocks for each trip were assigned to one or more of 40 defined individually numbered rectangular fishing blocks (approximately 7 km by 9 km) in PPB (**Figure 1**) in order to spatially resolve each angler's daily activity on water. Every vessel launched from a boat ramp in PPB was assumed to be capable of accessing any of the fishing blocks, yielding 800 potential alternative ramp-block and 1,600 block-block combinations from which each angler might choose (Huang et al., 2020).

Spatial and Statistical Analyses

Spatial

The centroid of each angler's postcode polygon was generated in ArcGIS (version 10.7, ESRI) to provide proxies for anglers' residential postcode. Travel distance was calculated across two separate networks of land and water using the origin-destination (OD) cost matrix in ArcGIS Network Analyst. The OD cost matrix identifies and measures the least-cost paths along the network from multiple origins to multiple destinations (Abrahamsson, 1998; Bera and Rao, 2011). To develop a land network, the Vicmap transport road network dataset[2] for the state of Victoria was obtained to calculate land travel distance (km) from each angler's residential postcode to the destination boat ramps. The origins in the road network were the nearest road to the centroid of each postcode polygon and the destinations were the boat ramp locations of the creel surveys. After creating origin-destination routes, results were validated by randomly selecting the network analysis outputs and comparing these with the Google Maps application, a web mapping service developed by Google as a reliable and popular mobile application for route planning[3].

A waterway network across PPB was created with polylines connecting each boat ramp to the centroid of each fishing block, and fishing blocks collectively, as there were trip records with anglers targeting two or three fishing blocks, thereby enabling estimation of the entire distance traveled over water. Moreover, nautical charts and a light detection and ranging (LiDAR) bathymetric layer (5 m spatial resolution) were used to check that waterways were correctly defined through the navigable channels due to challenges in navigating some areas of PPB, particularly among sand shoals in its southern region. The OD cost matrix was also applied as per the land network to measure the travel distance on water, and to estimate distances traveled from the boat ramps to the centroid of targeted fishing blocks. The number of anglers from each postcode were spatially joined by their respective postcode polygon to generate a heatmap of angler distribution over the landscape.

Statistical

All statistical analysis were undertaken in R version 4.0.3 (R Core Team, 2021). Multivariate analysis of non-metric multidimensional scaling (nMDS) was used to identify target species patterns among boat ramps. Data were square root transformed and then subjected to Wisconsin double standardization. A Bray–Curtis dissimilarity matrix was set for nMDS, and analyses were performed using the "vegan" package within the suite of R statistical software (Oksanen et al., 2020). Generalized linear models (GLM) were used to model differences in travel distance by region and target species on

[2]https://www.data.vic.gov.au/; accessed 20/03/2020.
[3]google.com/maps

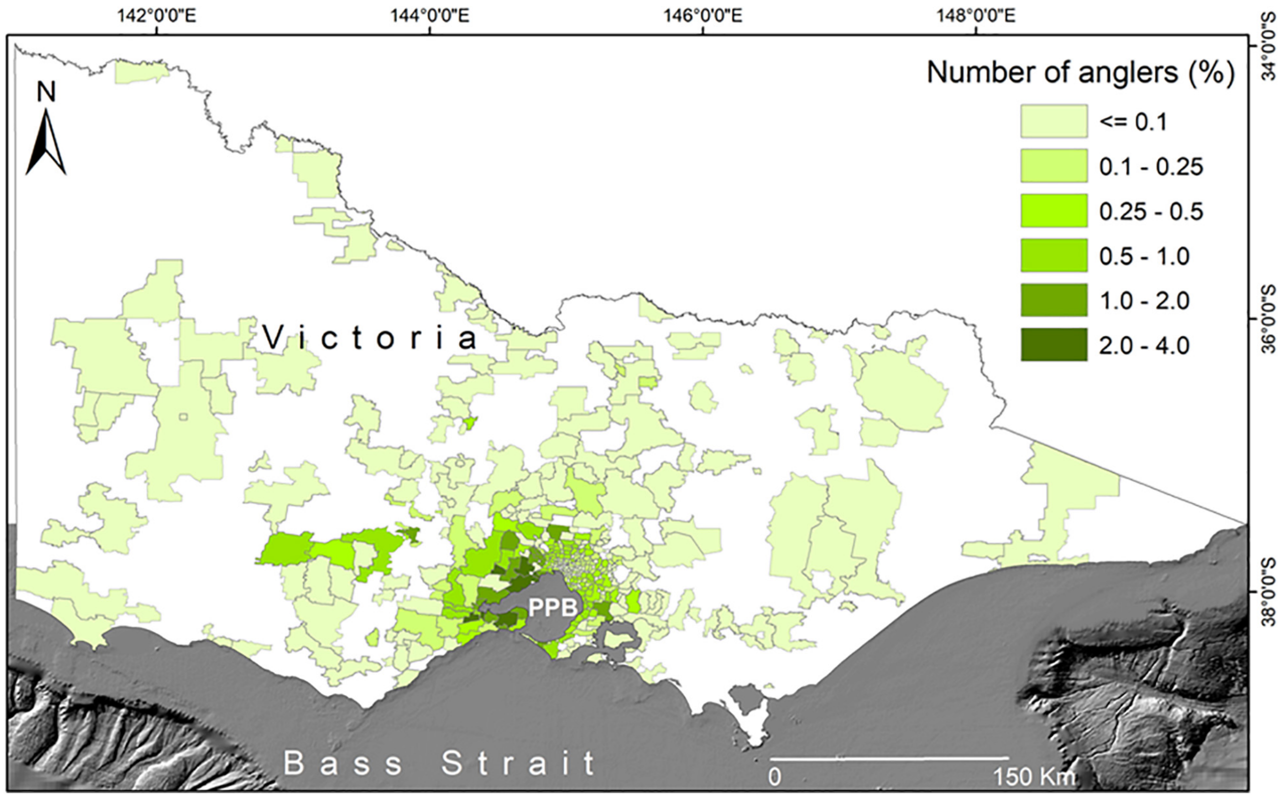

FIGURE 2 | Classified heatmap showing patterns in the percentage of interviewed anglers targeting Port Phillip Bay, based on their residential postcodes in Victoria with the data being pooled among years and species.

land and water. A Gaussian distribution was selected to best represent the distortion of the response variable with a log link function. The "ggplot2" data visualization package (Wickham, 2016) was applied to plot temporal (annual) trends in travel distance. In addition, regional and species differences in travel distance over land and water were evaluated using analysis of variance (ANOVA) and the non-parametric Kruskal–Wallis test, depending on the results of Shapiro–Wilk and Bartlett tests for normality and homogeneity of variance. Statistical differences in travel distance were then assessed between regions by applying *post hoc* Duncan and Wilcoxon tests at 95% confidence intervals ($P < 0.05$).

RESULTS

Most anglers fishing in PPB were residents from postcodes within relatively close proximity of PPB particularly within Melbourne, Mornington and Southwest Victorian regions as well as Geelong, Bellarine and Surf Coast Shire (**Figure 2**). The number of anglers aboard each vessel varied widely, from one individual to 12, with an average of 2.1 anglers per trip (**Table 2**). The amount of time spent fishing in PPB was 4.3 ± 2.0 (mean $\pm$ SD) hours on average per trip (**Table 2**), but there were a limited number of anglers (approximately one percent of fishing trips) who reportedly fished for considerably longer periods of time. Although overnight trips are occasionally undertaken the veracity of these reports of extreme fishing duration is unknown. Variability in effort (time spent fishing) among the fishing blocks was high with anglers typically spending more time fishing in the central areas of PPB (**Figure 3**).

TABLE 2 | The average number of anglers per trip and average (±SD) time spent fishing per trip in Port Phillip Bay by residential regions and overall.

Region	Number of anglers per trip	Hours spent fishing
Bellarine	2.0	4.4 ± 2.1
Melbourne	2.1	4.5 ± 2.0
Mornington	2.1	4.1 ± 1.9
Overall	2.1	4.3 ± 2.0

Regional patterns and variability in target species were evident in the nMDS with three clusters identified based on ordination of target species toward boat ramps within the three regions (**Figure 4**). Snapper were the dominant, or second most popular target species for anglers launching from most boat ramps, especially in the Melbourne and Mornington regions of PPB such as Carrum, Black Rock and Newport. This was followed by KGW and southern calamari, which were among the most popular target species for anglers who launched from the Bellarine and Mornington regions of PPB, particularly at Rye, Sorrento, Queenscliff and St Leonards boat ramps (**Figure 4**).

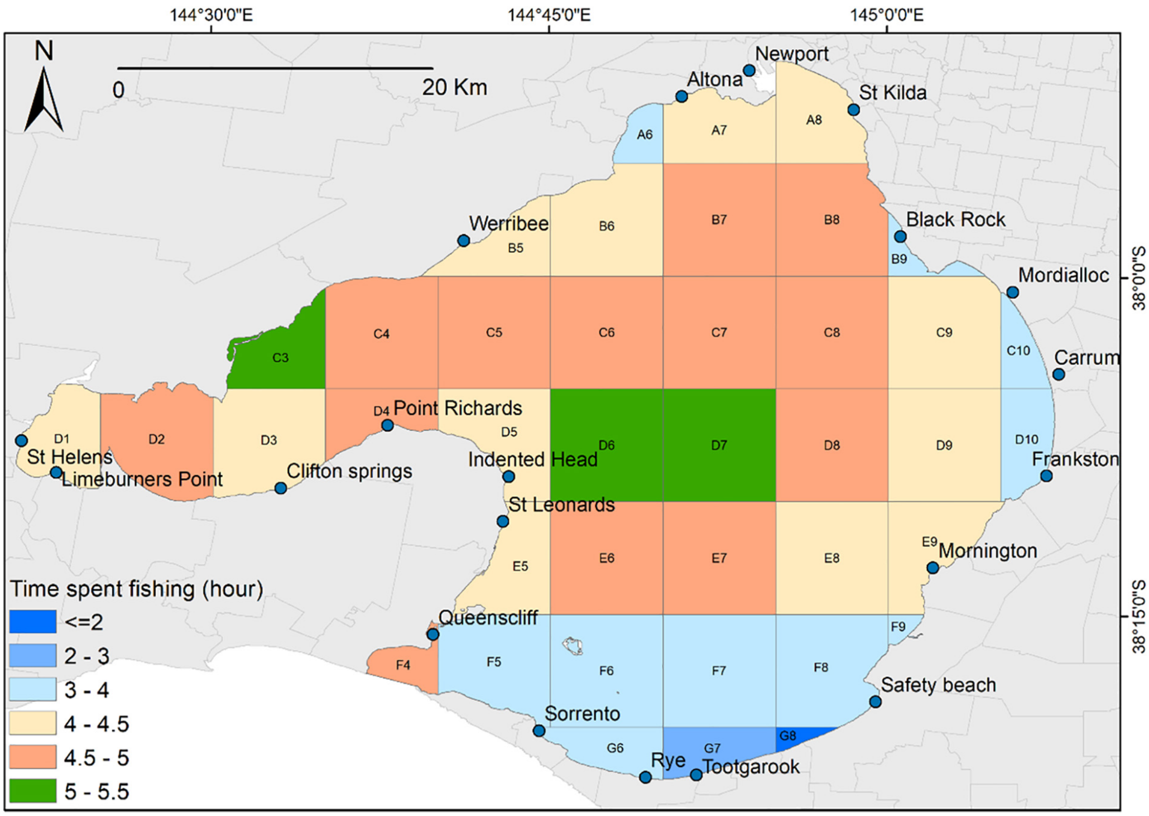

FIGURE 3 | Average amount of time (hours) spent fishing per fishing block in Port Phillip Bay based on the data pooled among years and species.

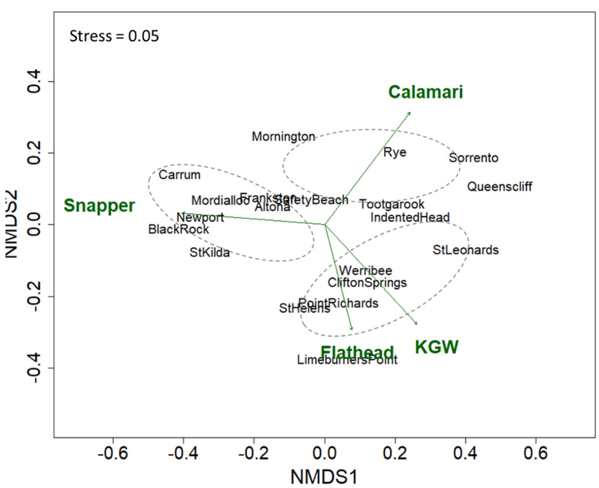

FIGURE 4 | Ordinations of nMDS analyses illustrating patterns of four main target species across boat ramps based on the survey data combined across years. Ellipses represent 95% confidence intervals further indicating the clusters of boat ramps deriving patterns in target species.

Targeting of flatheads was more common at boat ramps located in the Bellarine region including Limeburners Point, St Helens and Point Richards. Assessment of anglers' travel distance and routes revealed that they traveled from throughout the State to fish in PPB (**Figure 5**). Spatial patterns in anglers' travel dynamics were related to target species with most fishing trips commencing from postcodes adjacent to PPB or from northern and western residential regions, especially by anglers targeting snapper. By calculating travel distance, we found that average distance traveled on land was 40.5 km per trip, ranging from about one km to over 580 km, with most anglers residing within a proximity of about 50 km from their launch destination. Calculation of distance traveled on water with fishing trips starting from boat ramps to the centroid of fishing blocks showed that anglers traveled 11.1 km on average per trip one way.

The GLM results indicated significant differences by region and target species in travel distance on land and water (P-value < 0.001, **Table 3**). Furthermore, anglers who launched from boat ramps in the Bellarine region traveled significantly longer distances on land to access PPB (**Figure 6**). In comparison, travel distance on water was significantly longer for anglers who launched from boat ramps in the Mornington and Melbourne regions (**Figure 6**).

In terms of travel distance by region and target species, we observed a significant increase in distance traveled *via* land by anglers who started their trip from the Bellarine region to target snapper (**Table 3** and **Figure 7**). Similar regional differences were observed for other species (i.e., southern calamari, flathead and KGW) with longer distances traveled on land by anglers who

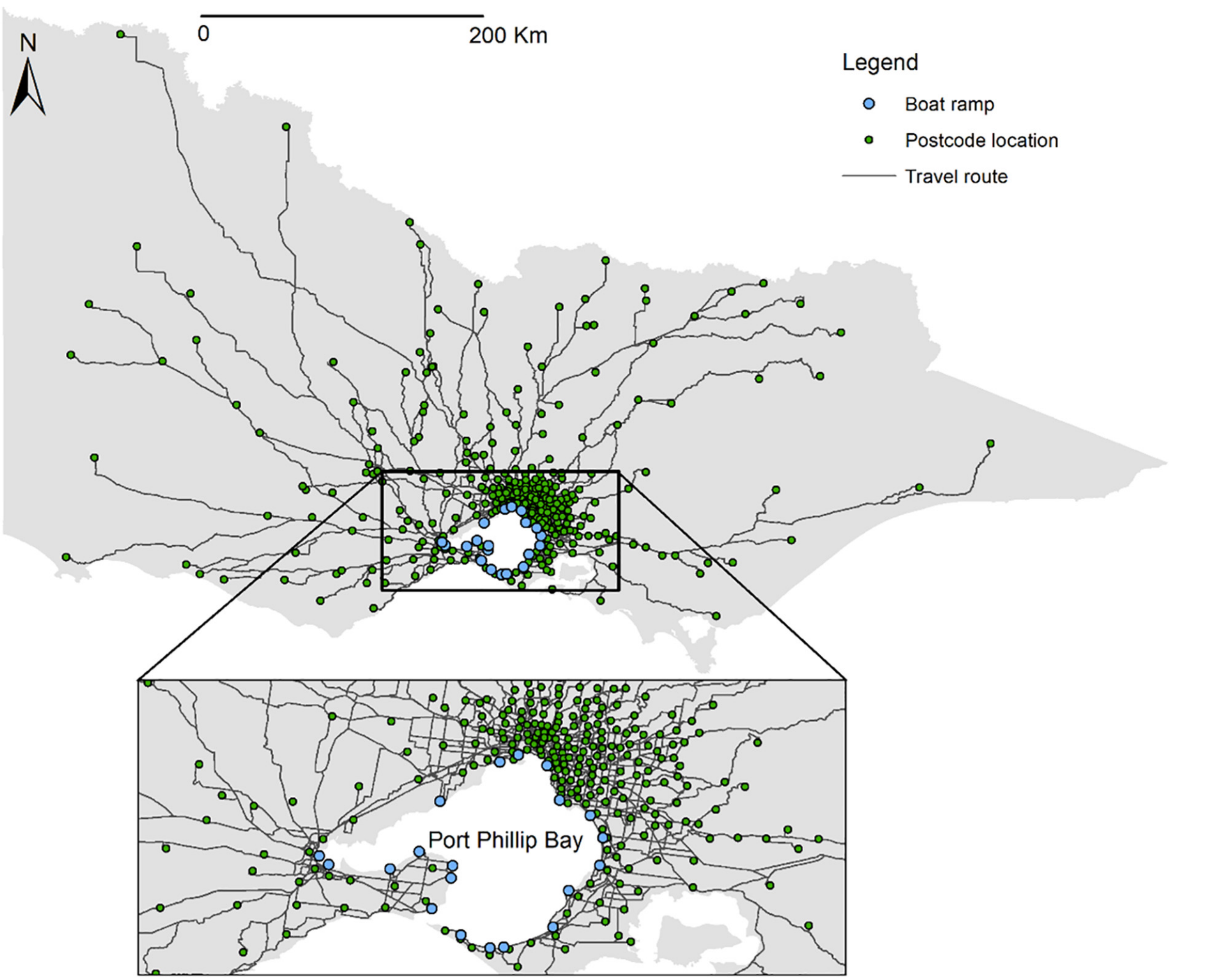

FIGURE 5 | Map showing land travel routes modeled using network analyst from residential postcodes of anglers to boat ramps throughout the State of Victoria. Data were pooled among species and years. Zoom box is drawn over Port Phillip Bay.

launched from boat ramps in the Bellarine region. In contrast, anglers who launched from boat ramps in the Melbourne and Mornington regions traveled significantly further on water for almost all target species (**Table 3** and **Figure 7**). Longest travel distances on water occurred for snapper (14.3 km on average) by anglers from the Mornington region and the shortest fishing trips on water (7.2 km on average) were made by anglers who launched from the Bellarine region to target southern calamari.

Temporal (annual) assessment of travel distance on land and water revealed significant variation by region and target species. Travel distance on land increased through time in the Bellarine region, but the travel distance on water remained shorter compared to the Melbourne and Mornington regions (**Figure 8**). Relatively similar annual trends were evident when travel distance patterns were compared by region and target species (**Figure 9**), revealing longer travel distance on land, and shorter distance on water for anglers from Bellarine, though some trends did not appear to vary through time. Interestingly, an increasing annual trend in travel distance on water occurred for anglers launching from Mornington who targeted highly mobile snapper, whereas a decreasing annual trend in travel distance on water was detected for anglers launching from Melbourne who targeted southern calamari and flathead (**Figure 9**). There were clear consistencies in land and water travel distance trends for anglers from Melbourne and Mornington for flathead and KGW.

The most popular fishing trips by postcode to boat ramps and fishing blocks revealed over 80% of fishing trips were the same on water and about 50% the same *via* land (**Figure 10**). Most of the popular travel routes over land and water for snapper occurred in the east and north of PPB by anglers residing mostly in the Melbourne metropolitan area. Their fishing trips on water were predominately in the northern and eastern parts of PPB and ventured further into the deeper central areas of PPB (**Figure 10**). Different trip patterns on land and water were evident when anglers targeted calamari as most of these anglers were from postcodes in the west, north and southeast of PPB, and most

TABLE 3 | Summary statistics of generalized linear models (GLM) describing travel distance on land and water by region and species, and statistical differences in average travel distance (km) on land and water between regions by target species in PPB.

Travel distance on land

Variable		Standard error	Deviance	*p*-value
Region	Melbourne	0.039	994,956	< 0.001
	Mornington	0.033		
Species	Flathead	0.041	228,068	< 0.001
	KGW	0.038		
	Snapper	0.035		

Travel distance on water

Variable		Standard error	Deviance	*p*-value
Region	Melbourne	0.024	20,099	< 0.001
	Mornington	0.025		
Species	Flathead	0.039	11,246	< 0.001
	KGW	0.038		
	Snapper	0.030		

Average travel distance (km) on land

Species	Region		
	Bellarine	Melbourne	Mornington
Snapper	$49.4^{\#}$	25.8^{ns}	29.6^{ns}
Calamari	$68.8^{\#}$	27.7^{*}	$54.7^{@}$
Flathead	$60.0^{\#}$	31.7^{ns}	42.2^{ns}
KGW	$58.3^{\#}$	28.9^{ns}	37.5^{ns}

Average travel distance (km) on water

Species	Region		
	Bellarine	Melbourne	Mornington
Snapper	$11.1^{\#}$	13.1^{ns}	14.3^{ns}
Calamari	$7.2^{\#}$	12.1^{ns}	9.4^{ns}
Flathead	$7.9^{\#}$	11.4^{ns}	11.7^{ns}
KGW	$7.8^{\#}$	12.5^{ns}	10.2^{ns}

*Data are based on anglers' average travel distance combined across years. Different symbols (#, *, and @) in each row indicate significant differences in travel distance between regions for the species in a given row at a 5% level of significance, with "ns" indicating differences among regions that were non-significant.*

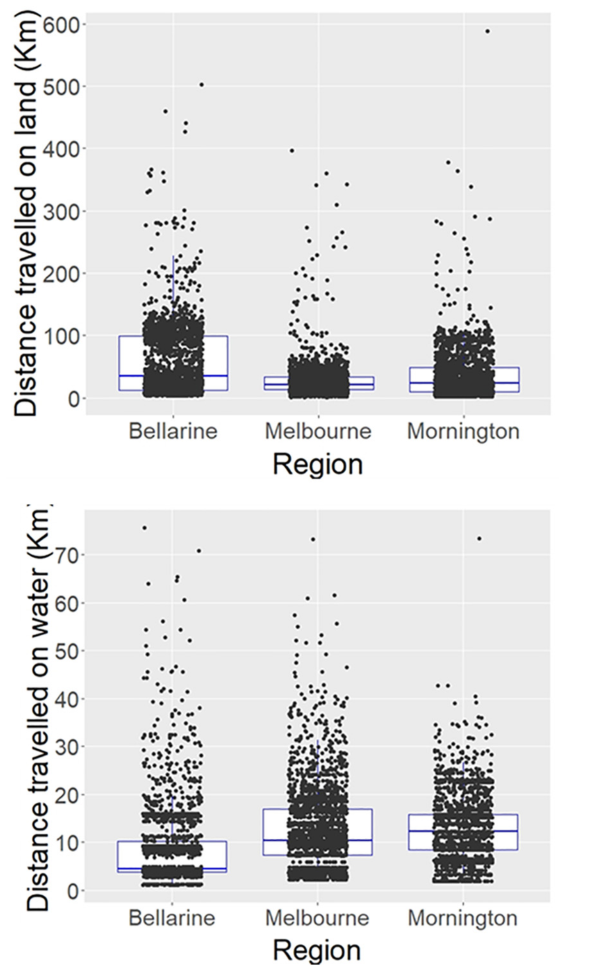

FIGURE 6 | Regional patterns (Bellarine, Melbourne and Mornington) in angler travel distance on land and water pooled among species and years. Each dot indicates an angler trip.

of their fishing trips occurred in shallow inshore waters in the south and western areas of PPB (**Figure 10**). In contrast, fishing trips targeting flathead and KGW were popular among anglers from postcodes from suburbs farther north and west of PPB (**Figure 10**), and fishing trips on water were also dominated in the north and west, with anglers who were targeting flathead generally traveling further into deeper water.

DISCUSSION

The present study has improved our understanding of marine recreational fishers' behavior in PPB, including spatial and temporal attributes of distances traveled on land and water to target particular species. In many instances, travel patterns varied considerably based on all these factors. Such knowledge is useful for managers and policy makers as it identifies the boundaries as well as dimensions of the areas explored by resource users across seascape and landscape scales.

This kind of information has guided planning and implementation of recent initiatives to upgrade boat ramps by adding additional lanes and trailer parking bays, and to improve land-based facilities for anglers, such as fish cleaning tables, fish waste and tackle disposal bins, and boat washdown stations, at the more popular launching locations around PPB (Better Boating Victoria, 2021).

Spatial-based analysis and mapping are useful for better understanding the dynamics of recreational fishing trips because when the results are used in conjunction with stakeholder engagement, they can become a key component of successful fisheries co-management (Pomeroy and Berkes, 1997; Pomeroy and Douvere, 2008; Pınarbaşı et al., 2017). Particular attention

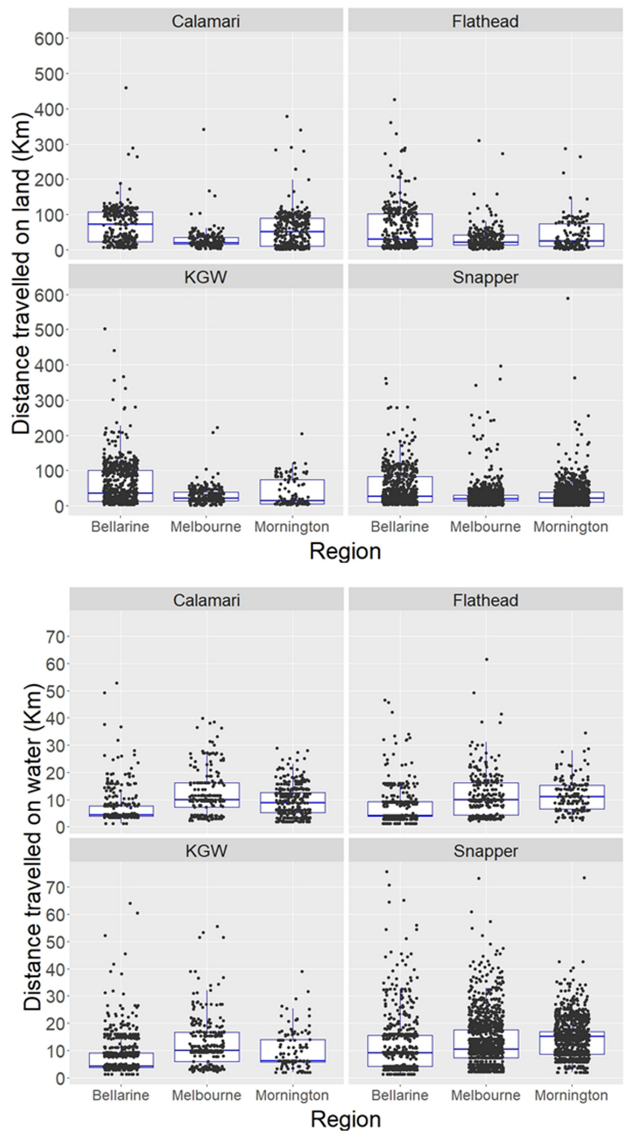

FIGURE 7 | Regional patterns (Bellarine, Melbourne and Mornington) in angler travel distance on land and water by individual key species in Port Phillip Bay combined across years. Each dot indicates an angler trip.

has been given to marine spatial planning in recent times, incorporating human activities, for supporting sustainable resource harvest and conservation (Douvere, 2008; Katsanevakis et al., 2011; Rassweiler et al., 2014; Domínguez-Tejo et al., 2016). In this respect, spatiotemporal assessment of anglers' trip and effort patterns may assist in achieving better user and stock support as some fisheries can transcend spatial boundaries, requiring coordination throughout land and water management areas. For instance, urban residents from Melbourne may travel further by land to target southern calamari and KGW in southern PPB, which can concomitantly generate increased tourism-based spending in regional Victoria whilst imposing further pressure on stocks in those areas. A quantitative understanding of these dynamics will enable resource managers to incorporate these factors into future decision-making processes, which hitherto had not been possible.

In addition, model outputs and maps generated from spatial analyses can reflect the complexity and patterns in the distribution of anglers at local and out-of-region levels. This may help predict how the behavior of these users and its impact on resources change and evolve through space and time, which is not generally well understood in recreational fisheries. Some anglers, especially those who launched their vessels from boat ramps in the Bellarine region to fish the western parts of PPB, traveled longer distances on land throughout regional Victorian to reach fishing destinations. Given such differences, co-management strategies may also differ in how to effectively engage these users as, for example, it has been suggested that stakeholders who are local residents and reside in closer proximity to the resources they use may have a greater tendency to be involved in co-management actions (Gutiérrez et al., 2011). Local users may also be more accessible and easier to engage, whereas more distant users from larger geographic scales may be less apt, or at least more difficult to engage in co-management activities (Hammitt et al., 2004; Cheng and Daniels, 2005; Hart et al., 2015). Distant users may also be less informed about fishing regulations and require different strategies to access information, noting that fishing regulations are provided on a mobile phone application that covers the entire State. In addition, local stakeholders may share more cohesive cognitions and have common preferences or requirements related to a given geographical region to produce greater resource stewardship in contrast to visitors from other residential regions. This may diverge even further between urban and regional residents, for instance, in terms of the levels of dependency, sentiments and attitudes toward local natural resources (Bonaiuto et al., 2002), as people may be more likely to support the protection of the local environment if they feel attached to a given geographic region or place (Faccioli et al., 2020). On the other hand, local people may not perceive resource protection initiatives, like MPAs, as useful, especially whenever a lack of access to former fishing grounds seems to impinge on community livelihood or managerial governance of MPAs is considered to be inadequate (Bennett and Dearden, 2014).

The present study has shown that, unsurprisingly, spatial patterns in anglers' fishing effort are linked to the target species' productivity and habitat preferences, as well as their mobility e.g., snapper (Hamer and Mills, 2017), under the assumption that the regions in which fishers are operating are more likely to support higher densities of targeted species. Herein, anglers launching from the Bellarine region were more likely to target KGW and calamari where seagrass and/or reef habitats are more abundant (see Mazor et al., 2021). Previous investigations of KGW habitat suitability indicated high suitability of the western and southern areas of PPB supporting seagrass or seagrass-edge habitats (Morris and Ball, 2006). Reduced fishing trips for KGW were evident in the central and eastern areas of PPB that are less suitable for KGW due to the deeper substratum and/or relatively bare sand habitat (Hamer et al., 2004; Morris and Ball, 2006).

Most fishing trips for southern calamari occurred adjacent to Rye, Sorrento, Queenscliff and to a lesser extent several other boat

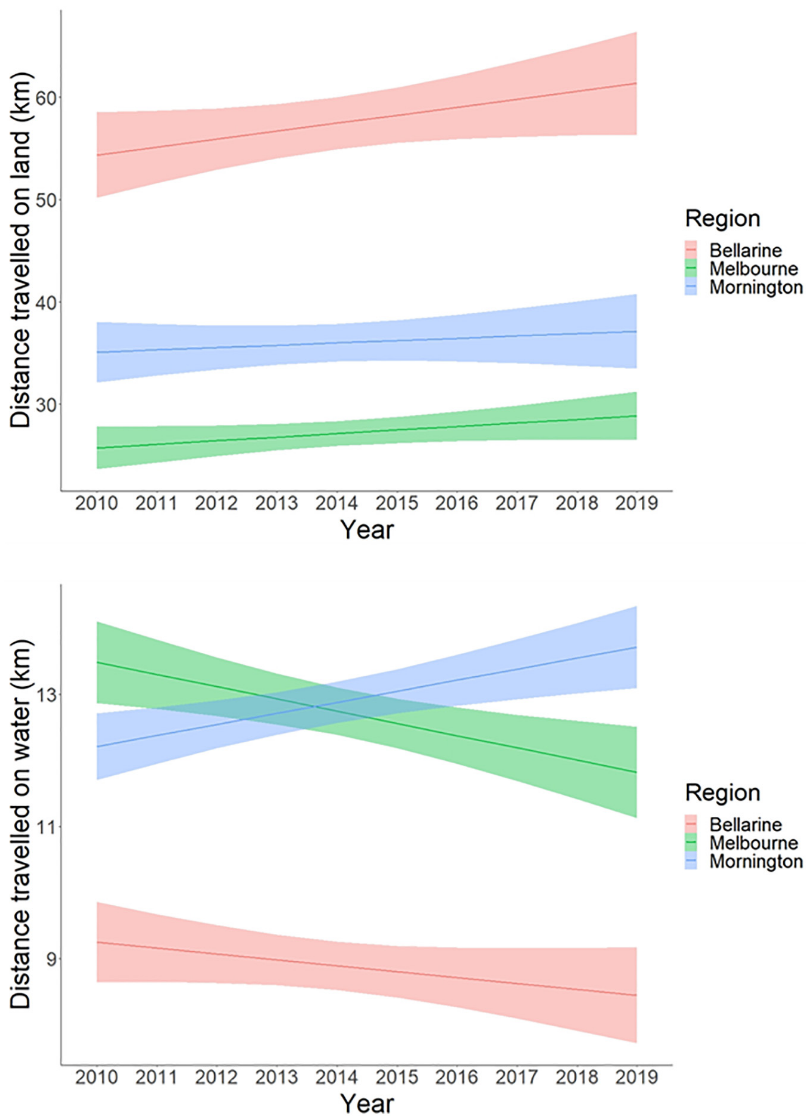

FIGURE 8 | GLM plots of temporal trends in the anglers' average travel distance (km) on land and water by residential region, combined across species. Shaded areas represent the 95% confidence intervals.

ramps in Bellarine region. Southern calamari is a demersal species that inhabits shallow inshore waters aggregating on seagrass beds to spawn (Smith et al., 2015). Therefore, shallow reef structures and seagrass beds surrounding the edges of PPB are more likely to be of interest to anglers who do not generally need to travel long distances on water to reach their fishing spots, especially those launching from Bellarine region as was observed in travel distance modeling.

Flathead, as interpreted in the present study, represents an assemblage of species, which are a popular table fish with sand flathead being the most popular among them in PPB because they are easy to catch and delectable (Fishing World, 2021). Flathead also show much higher CPUE in PPB than the other main target species, including snapper and KGW (Conron et al., 2020). Flathead can occupy a range of habitats, being found in estuaries, on bare sandy, muddy and weedy bottoms as well as in deeper areas (Jordan, 2001; Imamura, 2015). Such behavior and habitat preferences could potentially address variability in anglers' effort patterns throughout PPB and longer distances traveled to facilitate access to intermediate and deep waters, where they are particularly abundant (Parry et al., 1995).

In contrast, anglers' trips and fishing effort for snapper were spread throughout PPB especially within fishing blocks in the deeper areas in the east and north where the majority of snapper spawn (Hamer and Mills, 2017). PPB is the center of recreational snapper fishing in Victoria, which is highly valued by anglers, and the snapper fishery in PPB varies both spatially and temporally (Hamer and Mills, 2017). In this respect, it has been demonstrated that preferred habitats of adult snapper

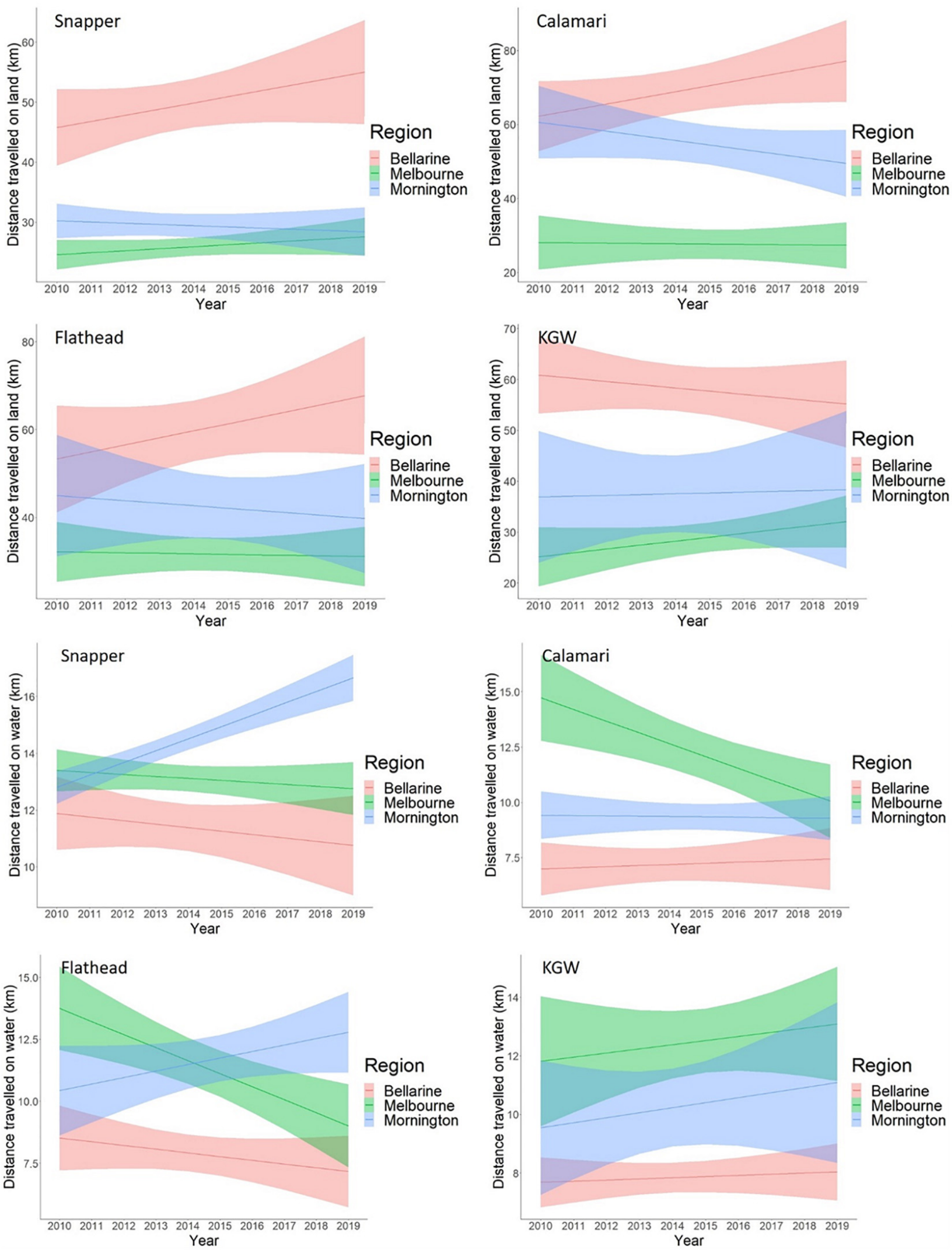

FIGURE 9 | GLM plots of temporal trends in the anglers' average travel distance (km) on land and on water by residential region and target species. Shaded areas represent the 95% confidence intervals.

differ from sub-adult snapper, with the deeper areas being more favorable, whereas shallow seagrass habitat and substrates of coarser sediment less suitable for adult snapper and larger size/age groups (Morris and Ball, 2006). This species' habitat preference, in turn, influences anglers' fishing decisions and travel dynamics as revealed in the most popular trip maps with > 80% of fishing trips targeting fishing blocks in the north and east of PPB. Anglers targeting snapper traveled further than those targeting other species, both because snapper frequented deeper waters further from shore, and also because both sub-adult and adult snapper are aggregative and highly mobile, moving considerable distances utilizing a range of habitats (Wilson, 1986; Parry et al., 1995; Conron and Coutin, 1998; Hamer and Mills, 2017). Although spatial and temporal trends in fishing trips can be linked to suitable habitat for each target species, there could also be other factors at play such as travel time at sea and associated costs, suitability of vessels for long range travel, prevailing weather conditions, time constraints, anglers' knowledge, seafaring experience, and potentially others, that play a role in anglers' decisions and

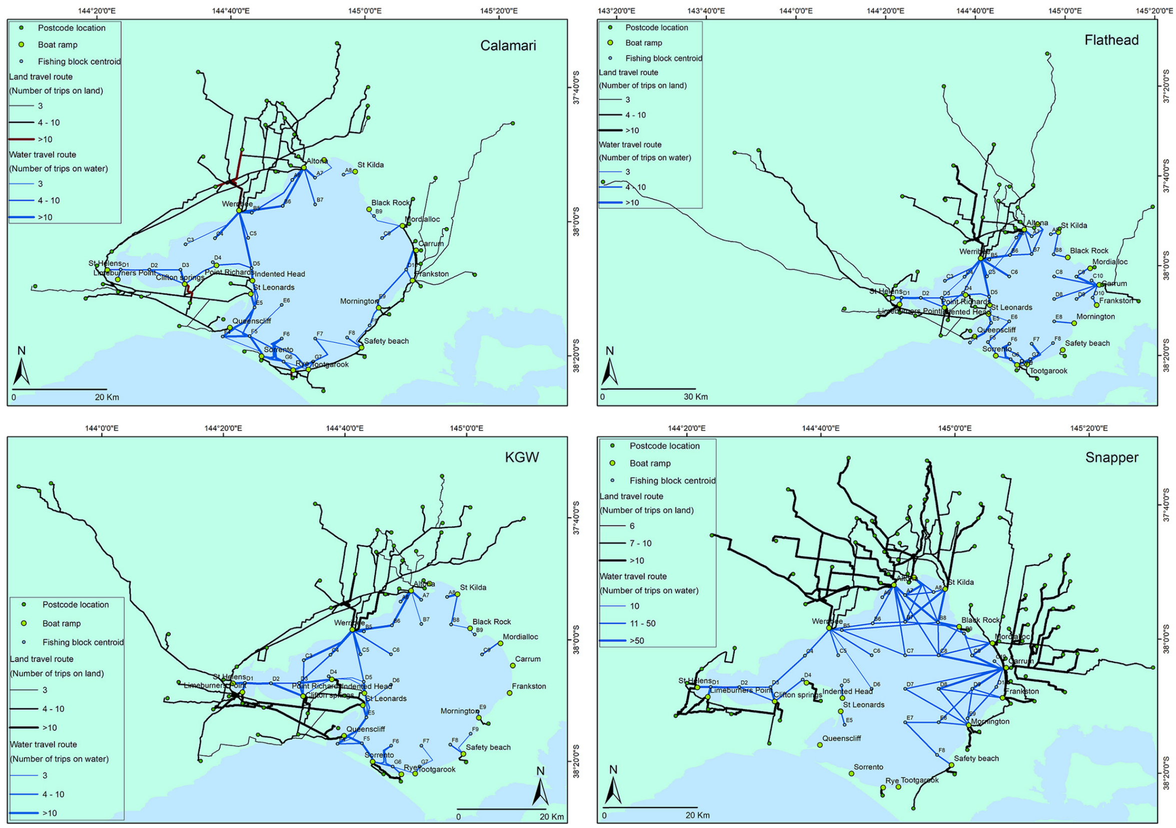

FIGURE 10 | Map of the most popular fishing trips over land and water (from residential postcode to the boat ramps and fishing blocks) to target the four main recreationally targeted species in Port Phillip Bay combined across years. These routes incorporate around 50% of all fishing trips on land, and over 80% of all fishing trips on water.

choices in fishing location. Nevertheless, both the number of anglers, and the time spent fishing, was relatively similar among years regardless of residential region, fishing block or target species, indicating that actual fishing time appears to be independent of travel time to and from the fishing grounds. Thus, it appears that this fishing time is sufficient to satisfy anglers expectations in terms of the overall experience, including the number of fish caught, or persistence if none are caught, and the need to allow sufficient time to return to shore, clean, pack up, and drive home.

Although we did not investigate angler travel patterns from economic perspectives, angler travel dynamics influences the spatial redistribution of their expenditure, which may support local businesses, further highlighting its importance to governing bodies, as has been revealed in other geographies such as Europe and United States (Steinback, 1999; Butler et al., 2009; Hyder et al., 2018; Pita et al., 2018). Given that millions of recreational fishing trips occur annually worldwide across diverse urban, regional and remote geographic regions to access convenient or prime fishing destinations, this pastime can create many direct and indirect job opportunities with substantial economic yield, which may be particularly important for the prosperity of coastal communities (McPhee, 2017; Camp et al., 2018; Pita et al., 2018). Impacts on local market activity will differ by species or region as all fisheries are unlikely to have the same potential spatial impact from the economic benefits derived from angling. It follows that economic investigations are required to provide policy makers with objective information obtained from comprehensive and robust analysis of angler demographics, expenditure patterns and socioeconomic dependencies of local and regional communities.

DATA AVAILABILITY STATEMENT

The de-identified raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements. Ethical review and approval was not required for the animal study because no direct sampling of vertebrate animals or cephalopods was undertaken by the researchers.

AUTHOR CONTRIBUTIONS

SC conceived the study, supervised the data collection, and oversaw the drafting process. AJ was involved in all phases of design, analysis, interpretation, and writing. JB supervised the quantitative analysis, interpretation of the results and assisted with drafting and refining the text. HG helped with interpreting the results, drafting and refining the text, and preparing tables and figures for submission and KG selected the appropriate statistical analysis, interpreted the analytical outputs and helped to accurately describe these in the methods and results. All authors contributed to the article and approved the submitted version.

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The impacts of long-term changes in weather on small-scale fishers' available fishing hours in Nosy Barren, Madagascar

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Small-scale fisheries (SSF) are highly susceptible to changes in weather patterns. For example, in Nosy Barren, Madagascar, SSF use traditional pirogues with handcrafted sails that rely on seasonal wind and sea conditions. As climate change is expected to increase the intensity and frequency of severe weather, it is important to understand how changes in weather affect SSF fishing efforts. Yet, a gap exists in the understanding of how changes in meteorological conditions affect small scale fishers. This study combines fishers' meteorological knowledge of weather conditions that allow for small-scale fishing with long-term remotely sensed meteorological data to quantify how fishing effort, defined as available fishing hours, of SSF in coastal Madagascar has changed between 1979-2020 in response to long-term weather trends. Results show a significant decrease in available fishing hours over the examined time period. Particularly, we found that a decrease in available fishing hours between 1979-2020 with a loss of 21.7 available fishing hours per year. Increased adverse weather conditions, likely associated with climate change, could decrease fishers access to crucial resources needed for the food and livelihood security. Climate change adaptation strategies will need to account for changing weather impacts on fishing availability.

KEYWORDS

small-scale fisheries, weather, climate change, non-motorized, Madagasacar, fishing effort, traditional knowledge (TK)

Introduction

Climate change, such as changes in ocean temperatures and ocean acidification can affect marine fisheries in multiple ways including changes in fish species distribution, fish reproduction, fish-species composition (Lam et al., 2020), distribution (Perry et al., 2005; Munday et al., 2008; Daw et al., 2009), increased mortality of larval fish or alteration in the composition and productivity of fish habitats (Blanchard et al., 2012; Barange et al., 2014). Yet, knowledge about impacts of climate change on the fishers themselves are less understood compared to ecological impacts, in particular for more long-term meteorological changes. An increase in extreme weather events or bad weather days, for example, has the potential to affect fishers' infrastructure, including boats and gear, disrupt fishing effort, and cause physical harm to the fishers themselves (Daw et al., 2009; Sumaila and Cheung, 2010; Sainsbury et al., 2018; Heck et al., 2021). Small-scale fishers (SSF) in particular, are highly susceptible to changes in weather and climate conditions, due to their high dependency on resources, exposure to the elements, and sensitivity to impacts (Huber and Gulledge, 2011; Onyango et al., 2012; Limuwa et al., 2018; Freduah et al., 2019; Thoya and Daw, 2019; Karlsson and Mclean, 2020; Ramenzoni et al., 2020; Turner et al., 2020). This is concerning given that over half of all fish caught in developing countries is produced by SSF and up to 95 percent of these landings are for local consumption (The World Bank, 2012). Thus, SFF play an important role in many societies and any threats to this role could have severe consequences for the food and livelihood security of millions of people.

SSF can be defined in multiple ways (Smith and Basurto, 2019). In this paper, we adopt the FAO definition that defines SSF as traditional fisheries that involve fishing households, use relatively small amount of capital and energy, relatively small fishing vessels (if any), make short fishing trips close to shore, and mainly fish for local consumption (FAO, 1994). SSF are embedded in complex, dynamic social-ecological systems (Chuenpagdee, 2011) that are highly nested in the local context. Climate change impacts on SSF thus might not only affect the sustainability of SSF but also have larger social impacts such as change in food availability and security in coastal areas (Allison et al., 2009).

This study explores changes in SFF fishing effort due to long-term changes in weather conditions. Previous work has already highlighted how climate change can reduce the efficiency of SSF fishing and consequently reduce food production (Tidd et al., 2022). Yet, given the lack of data on many SSF activities and SSF are diverse in nature, there is a need to investigate weather impacts on SSF effort in more local, placed-based approaches. We investigate this question in Madagascar, a small island state on the East coast of Africa that has an estimated 1.5 million people dependent on fisheries (Obura et al., 2017). At the same time, the country is highly vulnerable to climate change impacts on its fisheries (Heck et al., 2021). Some coastal communities in southwest Madagascar have already perceived an increased in bad weather in recent decades and report that it has reduced their ability to fish (Farquhar, personal communication). In Southwest and West Madagascar, fishing is mainly done by the *Vezo* people who use a traditional canoe, a "laka", carved out of a single tree (Astuti, 1995; Gough et al., 2009). A mast, sail, and seat are attached to the laka then, using seasonal winds and celestial navigation, these vessels allow them to reach the fishing grounds (Astuti, 1995). Given that these SSF are using non-motorized fishing vessels, they are likely to be more highly affected by changes in adverse weather conditions. Yet, knowledge about impacts on weather conditions on such SSF is hardly understood, partly because both historical weather and fishing effort data for SSF are limited. This study thus combines fishers' knowledge of ideal weather conditions with long-term remotely sensed meteorological data to quantify how fishing effort, defined as available fishing hours, of SSF in coastal Madagascar has changed over time in response to long-term weather trends.

Methods

Study site

This study focused on the fishing activity occurring in the Barren Isles or Nosy Barren archipelago that is located in the Mozambique Channel off the city of Maintirano, in the Melaky region of Madagascar (Figure 1). This 4632.0 km^2 area includes large coral reef and mangrove habitats which supports over 4000 traditional fishers for livelihood (Cripps, 2010). The region has a tropical savannah climate with distinct wet and dry seasons (Peel et al., 2007). The majority of fishers are Vezo people who reside in nearby villages on the coast of Madagascar or come from the southwest. Both travel to the islands to fish. While the majority typically fish seasonally between April and December during the dry season, others frequent the islands year-round for fishing activities (Cripps and Gardner, 2016). Fishing trips usually last 1-2 days, occurring during the day or at night in near shore areas. Fishers use a variety of gears including gill nets, hand lines, or spearguns. (Cripps and Gardner, 2016).

Data collection

Interviews

Fishers' meteorological knowledge (FMK) of weather conditions and associated impacts on fishing was assessed using semi-structured interviews. Similarly to fishers ecological knowledge (FEK), which has been used to reduce uncertainties and increase salience and credibility of models in data poor situations (Lavides et al., 2016; Lopes et al., 2019; Leduc et al.,

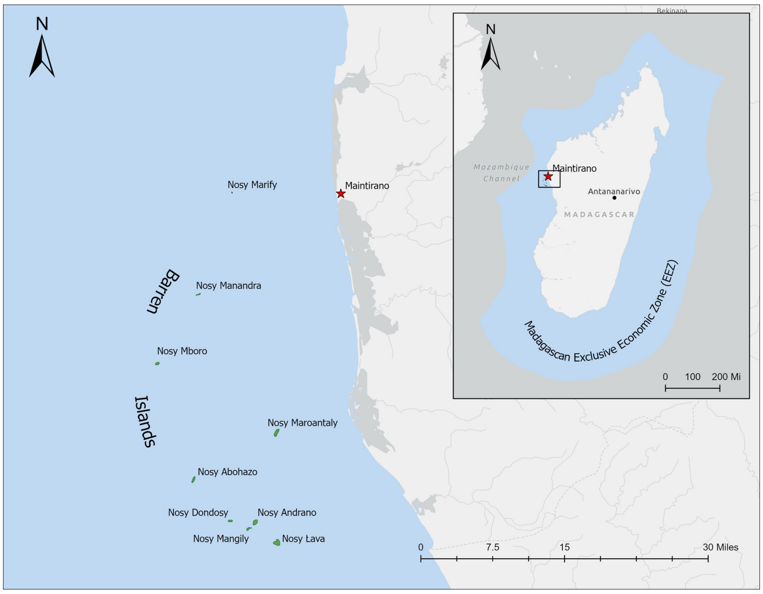

FIGURE 1
Map of Nosy Barren, also known as the Barren Iles or Barren Islands, of Madagascar.

2021), we assessed FMK to identify which weather conditions allow or prohibit SSF fishing activities at sea. We interviewed fishers that live in the neighborhood of Ampasimandroro of the nearby city of Maintirano where fishers who frequent the Barren Isles as their primary fishing area live. Interview participants were sampled using a snowball sampling method in which known contacts to author A.N. introduced him to more fishers who then agreed to be interviewed as key-informants. Fishers were determined to be informants if they have fished in the Nosy Barren for more than 15 years. A total of 23 key-informant interviews were completed between October 2021 and December 2021. Given that over 1000 fishers are thought to live in Maintirano, this sample represents a small proportion of the fishing community. All interviews were conducted in Sakalava Malagasy language.

To identify how weather affects the ability of fishers to go fishing, we asked fishers to identify (1) What wind speed is too strong to go to sea?; (2) What wind speed is too weak to go to sea?; (3) What wave height is too high to go to sea?; and (4) Is there a certain wind direction that prevents you from going to sea? Questions were asked in semi-structured interview format so we could gain both specific numerical estimates, but also understand in-depth information on local descriptions and terminologies for specific weather conditions that allow or hinder fishing at sea.

Remote sensing data

As Madagascar does not have observed historical maritime weather data records (i.e. marine meteorological buoys), we used modeled remote sensing data to assess historical marine weather conditions. We used modeled, 10m u-component of wind, 10m v-component of wind, and significant height of combined wind waves and swell available from the fifth generation of European Centre for Medium-Range Weather Forecasts (ECMWF) reanalysis (ERA5) single hourly dataset (Hersbach et al., 2018) (https://cds.climate.copernicus.eu/cdsapp#!/dataset/reanalysis-era5-single-levels?tab=form) for years 1979-2020. The reanalysis improves the accuracy from previous modeled historical weather datasets (Hersbach et al., 2020). The 10m u-component of wind is the horizontal speed of air moving towards the east, at a height of ten meters above the surface of the Earth. Similarly, 10m v-component of wind is the horizontal speed of air moving towards the north at ten meters above the surface. Significant height of combined wind waves and swell is the average height of the highest third of surface ocean/sea waves generated by wind and swell. It accounts for both surface waves and wind-sea waves. For wind components, data was available at a 0.25 degrees horizontal resolution. For significant height of combined wind waves and swell, data was available at a 0.50 degrees horizontal resolution. The uncertainty estimate for the ERA5 data is 0.5 degrees for wind data and 1 degrees for ocean wave data (Hersbach et al., 2018). Given the relatively small area of the Barren Iles, only six locations of data were available within the study site for wind data whereas three locations were available for wave data.

The data was imported into MATLAB for analysis. In order to obtain long-term data for wind speed and direction, the u-component and v-component of wind were calculated using two formulas following Guillory and Giusti (2020):

$$\text{Windspeed} = \text{sqrt}(u10.^2 + v10.^2)$$

$$\text{Wind direction} = \text{mod}(180 + 180 * \text{atan2}(u10, v10)/\pi, 360)$$

We then calculated the mean wind speed, wind direction, and significant height of combined wind waves and swell across all locations within the Barren Iles to get hourly datasets of the three weather parameters between 1979-2020. The mean was taken to help standardize the data across all locations within study area. However, due to the spatial uncertainty, it is likely that weather conditions outside of our defined study area are incorporated into our data. The estimated the spatial uncertainty is approximately 111km for wave data and 55km for wind data. Our study area is about 4632.0 km². We assume that the weather conditions are relatively similar between the study area and outside the study area.

Data Analysis

Calculating available fishing hours

Based on the interview data, we developed thresholds for wind speed, wind direction, and wave height that prevent fishers of going out to sea. These thresholds were used to determine the range of weather conditions that allow fishers to go out to sea. When weather conditions met these thresholds for the hour, it was considered an 'available fishing hour'. Allowable ranges were determined by using the mean values for minimum wind speed, maximum wind speed, and wave height reported from interviews and fishers' insights on wind directions that prohibit fishing in this calculation. Based on these data, the following parameters were used to calculate available fishing hours: wind speeds between 5.4-30.8 km/h; wave heights between 0-1.3m; and wind direction that was blowing from any direction except the south at 20km/h or greater. Given the technology the fishers are using, the traditional pirogues and sails, has not changed, it is assumed that these thresholds remain constant throughout time. Similarly, while there is some seasonality and cultural components that govern the individual preference of fishers' decision to go to sea, our analysis does not attempt to incorporate these because some fishers also choose to fish day or night and year-round. Thus, for simplicity we consider all months and times in our analysis. Available fishing hours were calculated based on the sum of all hours that fell within the weather conditions derived from the FMK. Available fishing hours were aggregated by each year and by each month between 1979-2020.

Calculating annual change in available fishing hours

Yearly available fishing hours between 1979-2020 were used to calculate annual change over time. To calculate the annual change in available fishing hours, we used a linear regression where the dependent variable was the total available fishing hours per year and the independent variable is time. Regression was appropriate because all the test regarding normality, stationarity, and linear assumptions were met: 1) there is no seasonal pattern in the yearly data; 2) the results of the augmented Dickey-Fuller test indicated that the data series is stationary and 3) any autocorrelation in the series could be attributed to weather changes driven by broader weather phenomenon whose affect lasts longer than one year (i.e., Indian Ocean Dipole). Furthermore, the residual plots showed that the fishing hours can be expressed as a linear function of time. Further information on the linear regression analysis can be seen in the Supplementary Information file.

Calculating decadal change in available fishing hours

Monthly available fishing hours between 1979-2020 were used to calculated decadal change over time. For each decadal period, 1979-1989, 1990-1999, 2000-2009, and 2010-2020, the monthly available fishing hours was averaged. Next, a T-test was used to understand if there was a significant difference in available fishing hours between the most recent decade (2010-2020) and the oldest (1979-1989). Further information for the T-test analysis can be found in the Supplementary Information file.

Results

Interview findings indicate a range in wind speed and wave height conditions that fishers identify as safe for operating their boats (Table 1). Mean wind conditions that were mentioned as good conditions for SSF were between 5.4 to 30.8 km/h and a maximum mean wave height of 1.30m. Standard deviations shows that fishers' perceptions of safe fishing conditions varied in particular for maximum wind speed (Std. = 7.23) and less for maximum wave height (Std. = 0.58).

Fishers described the dynamics of weather as "masay" when the ocean is very calm and "molenge" when the wind conditions are so calm that fishers cannot operate. Fishers' report that molenge conditions happen at all times of the year but usually are most common around noon when winds are shifting from the morning winds to the afternoon winds. The majority of interviewees (96%) further emphasized that the condition that prevents them the most from going to sea is "valaza", when winds become too strong—sometimes for weeks at a time. Valaza conditions are associated with strong winds coming from the south typically during May, June, July, and August.

While available fishing hours showed some cyclic behavior of years with high available fishing hours followed by and low available fishing hours, we found an overall decline in available fishing hours. Linear regression showed a significant decrease in

TABLE 1 Weather thresholds for wind speed and wave height given by fishers' meteorological knowledge (N=23).

	Wind Speed Min (km/h)	Wind Speed Max (km/h)	Wave Height (m)
Mean	5.40	30.80	1.30
Standard Error	0.33	1.50	0.12
Mode	5	30	1
Standard Deviation	1.62	7.23	0.58
Sample Variance	2.62	52.30	0.34
Kurtosis	4.26	2.88	0.70
Skewness	2.00	1.54	1.03
Range	7	30	2.30
Minimum	3	20	0.70
Maximum	10	50	3

available fishing hours between 1979-2020 at the 99% confidence interval (R^2 = 0.39, p<.005, t (41) = -5.13) (Figure 2) with a loss of 21.7 available fishing hours per year (See supplement for additional statistical information).

Mean fishing hours by month for each decade showed that more available fishing hours occurred during the rainy season (November-April) than the dry season (May to October). Mean values for fishing hours showed a decline in available fishing hours between decades (Figure 3) with a significant difference in mean values of available fishing hours between decadal periods 1979-1989 and 2010-2020 (t (11) = 6.17, p<.005) (See supplement for additional statistical information).

Discussion

Changes in weather can have significant impacts on fisheries, in particular SSF. Based on the combination of historical remotely sensed weather data and weather parameters defined by fishers, we found that available fishing hours have declined over the past four decades due to worsening weather conditions. Changing weather patterns thus can also significantly affect SSF as fishers may need to choose between reduced access to the resource or an increase in physical risks (Sainsbury et al., 2021).

As fishers are usually averse to higher wind and waves (Sainsbury et al., 2021), changing weather conditions may reduce access to the resource, which could have implications for food production, food access, and food stability in coastal areas. Because many small-scale fishers in Nosy Barren also sell fish for income to exporters, lack of fishing could reduce income for households and economic livelihood assets. Yet, perceptions of what was deemed as safe to operate varied among fishers—in particular for maximum wind speed. Variations could be driven by individual risk perceptions based on social and cultural factors (Salas et al., 2004; Thoya and Daw, 2019; Pfeiffer, 2020).

Adaptation strategies will be needed that help fishers cope with disruption of fishing activities to prevent an increase in physical risks that fishers take to sustain their livelihoods. However, traditional fishers might not be able to easily switch to less sensitive gear types or vessel sizes which requires substantial financial means.

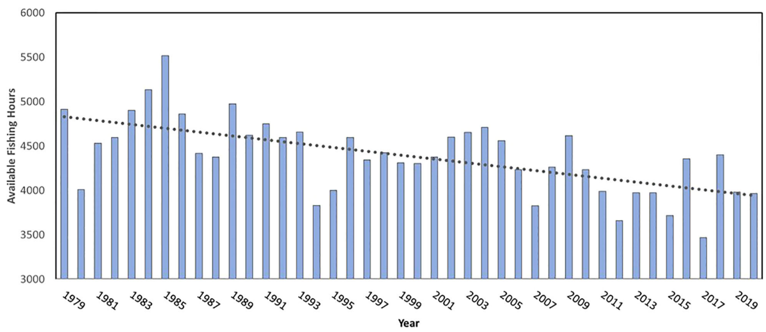

FIGURE 2
Total available fishing hours by year for the Nosy Barren area between years 1979-2020. The trendline represents the linear regression (y = -21.71 + 47787) showing an overall significant decrease in available fishing hours over time.

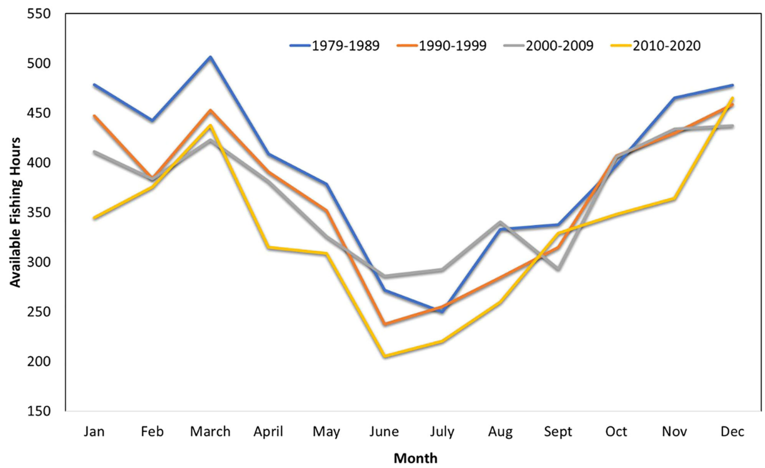

FIGURE 3
Decadal averages of available fishing hours by month for the Nosy Barren area.

In Madagascar, there has been efforts to use marine protected areas to help SSF adapt to climate change by promoting sustainable fisheries management, ecological restoration, adaptive management governing bodies, and livelihood diversification. Some initiatives have aimed to help SSF diversify their income specifically through seaweed and sea cucumber aquaculture ventures (Rönnbäck et al., 2002; Rasolofonirina et al., 2004; Robinson and Pascal, 2009; Ateweberhan et al., 2015). Yet, developing alternative livelihoods can be challenging depending on the social-ecological context of the fisheries. For example, the water quality near Maintirano is not conducive to sea cucumber aquaculture due the high sediment load from the nearby rivers that empty into the ocean ("Maintirano" literally translates to "black water"). Another challenge are financial issues. For example, aquaculture on the Barren Iles themselves may be possible, but would require significant infrastructure investment. Environmental or economic conditions thus can limit the use of aquaculture as an alternative livelihood strategy to marine fishing and might not be a suitable livelihood alternative in some coastal areas.

Methodologically, this study illustrates how FMK may be combined with remote sensing data to understand changes in fishing trends in data-poor regions. Given that SSF often exist in data-poor contexts, this methodology could be applied in other coastal communities. Future research could build on this study by including more details of fishing behavior and other weather conditions to calculate available fishing hours (e.g., temperature, precipitation, lunar cycles, culturally significant holidays, etc.). For example, our study found that more available fishing hours were available in the rainy season than the dry season. This is likely because our model did not include fisher's preferences and FMK for fishing during rain events.

Given that this study was intended to be a proof-of-concept, it would also be beneficial to further test and validate the data and the methodology in the future. This study used weather data that was remote sensing data and at a coarse resolution. As mentioned earlier, the uncertainty estimate for the ERA5 data is 0.5 degrees for wind data and 1 degrees for ocean wave data (Hersbach et al., 2018). This does not account for how local features, such as the islands, may affect the weather patterns. Additionally, due to the spatial uncertainty, weather data from outside of our defined study area may have been included in our analysis. However, given that it is known that SSF in this area are highly migratory and operate in other places outside the Nosy Barren, these results are still meaningful. Yet, future work thus could include observed, *in-situ* weather data where available to ground truth and reduce uncertainty of remote sensing data.

Similarly, the model used the average weather parameters based on 23 interviews. Within each of these parameters was some variability which affected the calculation of available fishing hours. For example, while there was little standard deviation on minimum windspeed needed to go to sea (1.62 km/h), a larger standard deviation was found in regards to the maximum windspeed that prevented fishers from going to sea (7.23 km/h). Such deviation increases the uncertainty of these results. A larger interview sample size across multiple fishing communities could provide more empirical insights into fishers' definition of weather condition thresholds. Also, the study only investigated changes in available fishing hours, not actual fisheries production. Future research could explore whether fisheries production have changed due to changes in available fishing hours and assess the indirect and direct social, economic, ecological, cultural consequences of weather impacts on SSF to better understand how SSF are affected not just by extreme events but also changes in long-term weather patterns.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

SF designed the study and calculated available fishing hours. AN collected the interview data, completed all translations, and organized data for processing. SF and NH wrote the manuscript. MS conducted statistical analysis. YX created cartographic figure. All authors read and approved the final version of the manuscript.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.841048/full#supplementary-material

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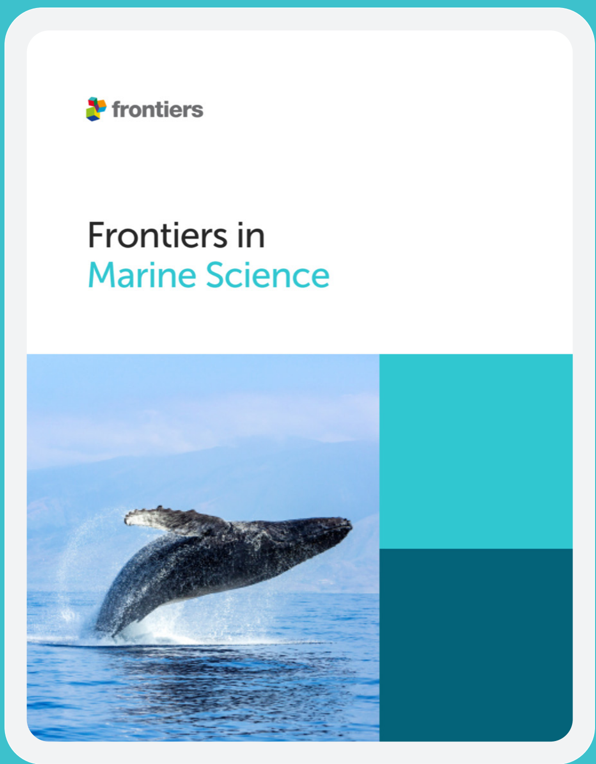

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