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Trends in recreational fisheries and reef fish community structure indicate decline in target species population in an isolated tropical oceanic island

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ABSTRACT

Marine recreational fisheries and fish communities have been mostly studied separately, severely hampering the detection of possible interactions. Here we monitored recreational fishery landings (angling and spearfishing) and assessed the reef fish community through underwater visual censuses in Trindade, an isolated oceanic island in the southwestern Atlantic Ocean. The study was conducted in zones of high and low fishing effort, along three different years spread over a seven-year period. We found high catch per unit effort (CPUE; kg*fisher^{-1}*hour^{-1}) associated with high fishing intensity. However, the biomass of the targeted species in the natural environment decreased by 58% over time, while fishing effort increased about 270%, and CPUE decreased by 40%. Inverse relationships between effort and CPUE, and effort and biomass in the environment, were stronger in zones of high fishing effort. We conclude that the recreational fishery activity quickly responded to overfishing, creating a harvesting process that negatively impacted reef fish communities in this isolated oceanic island. Thus, conservation strategies should incorporate long-term recreational catch and fish community monitoring programs. Here we discuss options for the sustainability of the recreational fisheries and reinforce the importance of dialogue among scientists, managers and users for the achievement of conservation goals.

1. Introduction

Despite their social and economic importance worldwide, fishery activities are a major source of disturbances on marine ecosystems (Cesar et al., 2003; Jackson et al., 2001; Pauly et al., 1998). While the impacts of commercial fisheries have been broadly studied (Garrison and Link, 2000; Hilborn et al., 2003; Pauly et al., 2005, 1998), much less attention has been given to the effects of recreational fishing (Cooke and Cowx, 2006). This lack of information is worrisome since recreational activities can reach yields similar to those of commercial fisheries targeting the same species (Coleman et al., 2004; Pinheiro and Joyeux, 2015). Thus, incorporating recreational yields into monitoring programs and sustainable harvest models is crucial to a better stock management (Post et al., 2002). Whereas appropriate, catch quotas, spatial and temporal fishing closures, the establishment of Marine Protected Areas (MPAs), among other measures, could provide options for managing recreational fisheries (Frisch et al., 2012).

Studies assessing recreational fisheries' impacts have generally monitored landings or the population structure of captured species (Frisch et al., 2012, 2008; Marengo et al., 2015; Meyer, 2007; Nunes et al., 2012; Pinheiro et al., 2010a). However, a comprehensive understanding of the interactions between fish landings and population's status is mostly lacking (but see Edgar et al., 2018 for a counter-exemple). The use of non-destructive monitoring methods, such as underwater visual census (UVCs), for the generation of data that can help to establish fishing limits remains challenging (Frisch et al., 2012).

Reefs of remote oceanic islands are commonly reported as pristine and healthy (DeMartini et al., 2008; Knowlton and Jackson, 2008), mainly when compared with mainland coastal reefs. However, oceanic islands are also fragile due to their isolation, being highly vulnerable to overfishing even when facing low fishing effort (Luiz and Edwards, 2011; Sandin et al., 2008). In front of the global coral reef crisis

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(Bellwood et al., 2004; Hughes et al., 2017; Knowlton and Jackson, 2008), regulation, management and enforcement of all fishery activities are urgently needed to improve and maintain the resilience of reef ecosystems.

In this context, the present work investigated the linkages between trends in recreational fisheries catches and fish community structure around Trindade Island over a seven-year period. The recreational activity is practiced mostly by the military personnel that maintain a Navy station in this remote oceanic island. Commercial fisheries, now prohibited with the creation of the MPAs around Trindade Island (Giglio et al., 2018), used to target different reef fishes in deeper waters (Pinheiro et al., 2010b), not offering conflict or overlap to the recreation activity. Our research aims to: 1) quantify recreational fishery landings; 2) analyze the temporal variation in fishing effort and captures of two recreational fishing activities (spearfishing and angling); and 3) evaluate the spatial and temporal variation in biomass of species targeted by recreational fisheries (all species combined and key species: *Cephalopholis fulva*, *Epinephelus adscensionis* and *Caranx lugubris*). Finally, we discuss the importance of recreational fisheries in shaping target species populations, suggesting management options to stakeholders (Brazilian

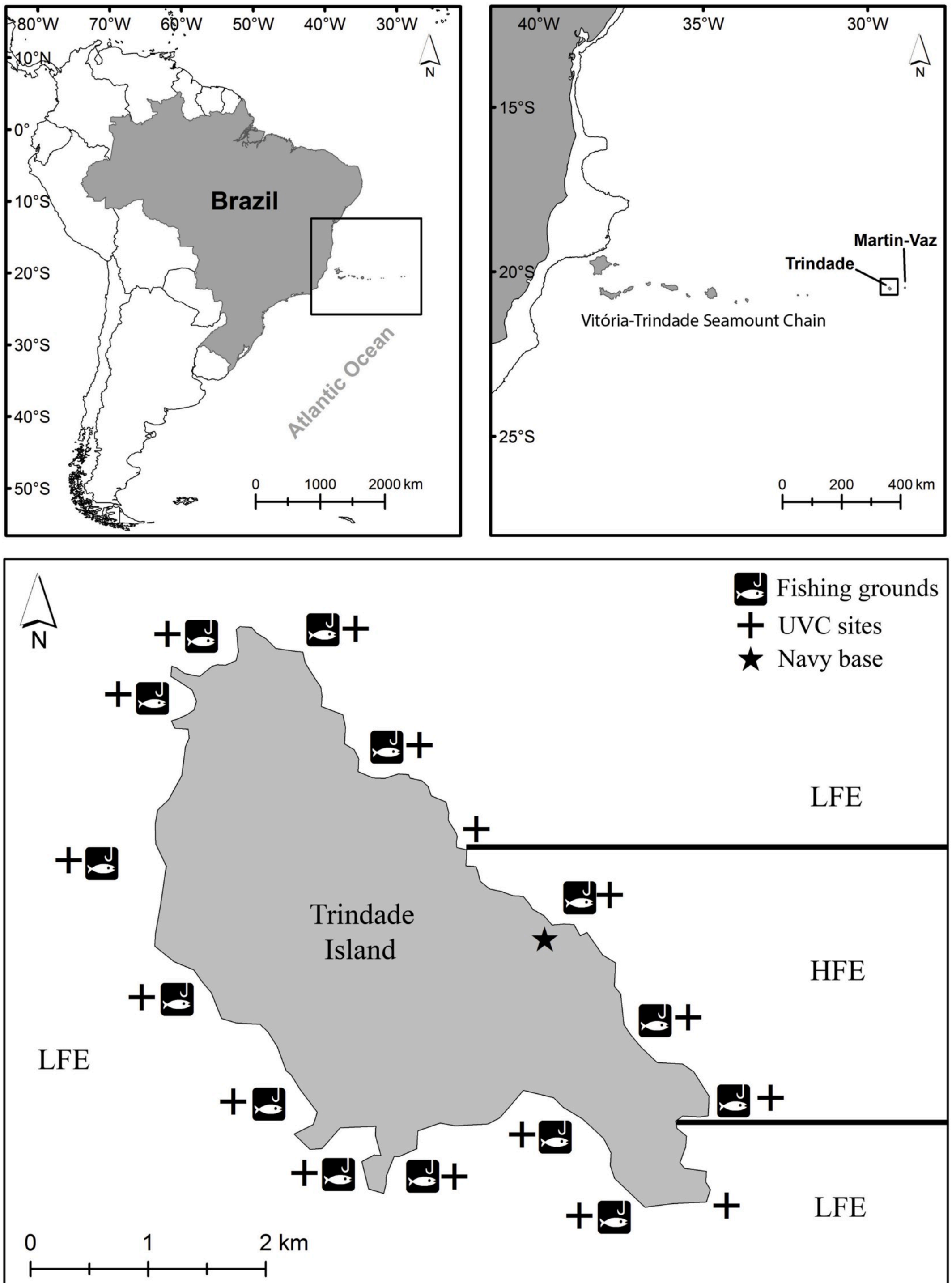

Fig. 1. Localization of Trindade and Martin Vaz archipelago and map of Trindade Island showing recreational fishing grounds and sampled sites for underwater visual census (UVCs). Higher and lower fishing effort zones (HFE and LFE, respectively) and the location of the Navy base are also indicated.

Navy and environmental agency - ICMBio) aiming the sustainability, recovery and conservation of overexploited resources.

2. Materials and methods

2.1. Study area

Trindade Island and Martin Vaz Archipelago are located on the eastern end of the Vitória-Trindade Seamount Chain (Fig. 1), approximately 1140 km off the coast of Brazil, to whose territory they belong (Almeida, 1961). Compared to other oceanic islands of the South Atlantic, Trindade shelters one of the highest biodiversity levels of reef fishes (Gasparini and Floeter, 2001; Pinheiro et al., 2015; Simon et al., 2013), coralline algae (Sissini et al., 2017), scleractinian corals (Santos et al., 2020), zoantharians (Santos et al., 2016, 2019), as well as a high richness and endemism of marine mollusks (Gomes et al., 2017).

Since 1957, Trindade Island harbors a Navy base (Fig. 1) that also maintains an oceanographic program (Pedroso et al., 2017). Although there is no native population, around 40 Brazilian Navy personnel and eight civilians (mainly researchers) stay on the island throughout the year (military crew is changed every 4 months), all housed in the military base. Island-based recreational fishing is practiced in shallow waters (<20 m depth) while deeper areas were mainly targeted by commercial semi-industrial boats (Pinheiro et al., 2010b). Since commercial and recreational fisheries target different areas and species (Pinheiro et al., 2010b; Pinheiro and Joyeux, 2015), their effects are likely to be isolated.

Recreational fishing is one of the main leisure activities in the island, with spearfishing and angling being the most employed fishing techniques. Other gears, such as throw net and hand nets, are used on a minor scale targeting small fishes for bait (e.g. *Harengula jaguana*). A high quantity of processed (gutted/beheaded or filleted) fish is periodically transported to the continent (Pinheiro and Gasparini, 2009). With the exception of rare recreational fishing excursions coming from the mainland to Martin Vaz Archipelago (situated 50 km east of Trindade), recreational fishing is restricted to Trindade (authors' personal observation). In March 2018, the Brazilian government created a large MPA (including no take and sustainable use zones) encompassing almost the entire Brazilian Exclusive Economic Zone around Trindade and Martin Vaz Archipelago (Brasil, 2018; Soares and Lucas, 2018). However, most of the shallow waters and shoreline around Trindade were excluded from any type of protection (to allow military use), and subsistence fishing was allowed inside the MPAs (see Giglio et al., 2018 for more details about fishing regulation in this MPA). Nevertheless, by September 2018, the Brazilian environmental agency (ICMBio) established commercial fishing regulations that constrain the activity around the island, and the Navy committed to prohibit the spearfishing practice as well as the stocking and transportation of fishes to the continent (ICMBio, 2018).

2.2. Data collection

Data of Trindade recreational fisheries (spearfishing and angling) and community structure of reef fishes were collected during three expeditions along a seven-year period. Expeditions were conducted in 2007 (February to April), 2012 (June to August) and 2013 (April to June), accounting for a total of 135 days of fishing monitoring (n = 149 fishery activities monitored) and 415 underwater visual censuses (UVCs).

Effort (number of fishers and time spent in each activity; in fisher*hour^{-1}) and location of fishing grounds were acquired from 149 recreational activities, both for angling and spearfishing. Each fisher in both activities usually only employed a single fishing gear. All fishes captured were identified to the species level, counted, measured (total length to the nearest mm) and weighted (with precision of 5 g). In thirty-two fishing activities, only the number of fishes and the catch composition were recorded, and this data was excluded from weight analyses.

The composition and distribution of reef fish assemblages around Trindade Island was assessed through UVCs (n = 415) consisting in strip transects measuring 20 × 2 m (Pinheiro et al., 2011). In each transect, all fishes were counted and identified to the species level. Fish lengths were grouped into 10 cm classes with the first two classes grouped into 5 cm (1–5 cm and 6–10 cm). Biomass was estimated through length-weight relationships (Froese, 2006) using the center of the length class. The allometric coefficients for the species were obtained from FishBase (Froese and Pauly, 2018) and from the literature (Frota et al., 2004; Guabiroba and Joyeux, 2018). Although Martin Vaz Archipelago is located near Trindade, it has no research facilities or accommodations and it is extremely difficult to access but by helicopter. Despite these logistic restrictions, we were able to perform 47 UVCs in Martin Vaz in 2007 and 2013, but none in 2012. This data is not included in Trindade's dataset (415 transects), and was used as a control site, without recreational fishing.

2.3. Data analysis

Sites around Trindade Island were segregated into two different zones according to accessibility (i.e., distance from Navy base and necessary logistics) and, consequently, to fishing pressure (Fig. 1). The Higher Fishing Effort (HFE) zone encompassed sites close to the base (<2 km and easily accessed on foot with no need of boats), that experienced more than 0.85 fishing activities per day (an average pulling all monitored years together). The Lower Fishing Effort (LFE) zone grouped all other sites farther from the base (>2 km away, mainly accessed by boat), and that experienced less than 0.32 fishing activities per day. Access to LFE sharply increased from 2009 onwards with the improvement of the logistics brought to support scientific research, which included inflatable boats, dive gears and trained personnel for boat operations.

Catch per Unit Effort (CPUE) was established as the number and weight of fishes caught per fisher per hour (hereafter number and biomass CPUE, respectively). The Shannon-Wiener diversity index {H' = -SUM(Pi*log(Pi))} of captures was calculated using logarithms of base *e*. Annual catch estimates were calculated for each species multiplying mean individual weight by number of captures, averaged per day of monitoring and then extrapolated for one year. The effort was considered constant for this purpose (in order to be conservative), once number of military crew in the island did not change and fishing was observed as a daily leisure activity. Estimates were performed separately for each year (considering only data from the given year) and also for the whole period (using data of all three years). Species recorded using UVCs were categorized in either "captured" (species targeted by recreational fisheries at least once) and "non-captured" (species never caught by fishers). Three species represented more than 60% of the biomass harvested by recreational fisheries and were defined hereafter as key species: *Cephalopholis fulva*, *Epinephelus adscensionis* and *Caranx lugubris*. Although these key species were included in the analyses of captured species (above), their biomass was also analyzed separately afterwards.

As fishery and visual census data did not fit normality assumptions, Permutational Analyzes of Variance (PERMANOVAs; Anderson et al., 2008) were performed to test whether the fishing effort, CPUE (both in number of individuals and in biomass) and biomass data derived from visual census (all captured species, non-captured species and key species biomass) differed significantly between factors (see details below).

For fishery related data (*i.e.* effort, CPUE, diversity index and mean size of catches) a PERMANOVA model was built considering year (three levels, 2007, 2012 and 2013) and gear type (two levels, angling and spearfishing) as fixed factors. *A posteriori* paired-wise comparisons were performed with distance-based permutational *t*-tests (Anderson et al., 2008) to address significant differences between each year. Regarding UVCs related data (i.e. captured, non-captured and key species biomass), a PERMANOVA model was built considering year (three levels, 2007, 2012 and 2013) and fishing zones (two levels, LFE and HFE zones) as fixed factors. *A posteriori* paired-wise comparisons were made as the same way above to address significant differences between each year. Prior to PERMANOVA analyzes, raw data were square root transformed and resemblance matrices based on Euclidian distance between samples were constructed for each variable related both to fishery and UVCs data. PERMANOVAs were run using 999 permutations of residuals under a reduced model and type III of sums of squares.

3. Results

3.1. Recreational fisheries landings

A total of 2424 fish specimens caught by recreational fisheries were analyzed, corresponding to 37 species belonging to 18 families. In average, each fishing activity employed 3 fishers and captured approximately 14 kg of fishes. The key species, *Cephalopholis fulva*, *Epinephelus adscensionis* and *Caranx lugubris*, made up more than 70% of the total number and 60% of the total biomass harvested. Annual estimates of total biomass harvested increased more than 80% along the study period. Including data of all sampled years, recreational fisheries in Trindade were estimated to have caught yearly more than 6500 specimens and 7.6 tons (Table 1).

3.2. Trends in fisheries catches and effort

Total biomass (i.e., considering all captured species) CPUE decreased significantly from 4.39 to 2.60 kg*fisher^{-1}*hour^{-1} (approximately 40%) between 2007 and 2013 (Fig. 2b; appendix A). Spearfishing showed higher biomass CPUE than angling in all years (Fig. 2a) and was responsible for 68.2% of the total biomass harvested by recreational activities in Trindade, with an average of 2.96 ± 0.27 (SE) kg*fisher^{-1}*hour^{-1}. Angling represented 31.8% of harvested biomass, with an average of 1.34 ± 0.27 kg*fisher^{-1}*hour^{-1} (appendix A). Total number CPUE (number of fishes*fisher^{-1}*hour^{-1}) did not present significant differences between years and gear types (appendix B).

Fishing effort (fisher*hour^{-1}) increased more than 270% between 2007 and 2013 (*pseudo-F* = 6.2; $p = 0.003$). Angling effort was significantly higher than spearfishing only in 2012 (Fig. 2). Spearfishing effort showed a sharp increase from 2007 onwards while angling effort was higher in 2012 (Fig. 2e).

Diversity of captures, as measured by the Shannon-Wiener diversity index, did not differ among years, but showed a strong difference between gear types (appendix B). Spearfishing targeted more species (33) than angling (19 species), including ecologically important herbivores like *Sparisoma amplum* and *S. axillare*. Moreover, 21 species were captured in LFE zone while 36 species were captured in HFE zone. In general, spearfishing caught larger fishes than angling (*pseudo-F* = 18.03; $p = 0.001$). Fishes caught by spearfishing were larger in 2007 than in the other years, while those captured by angling did not differ

Table 1
Annual estimates in biomass (kg) and abundance (number) of reef fishes species captured by recreational fisheries in Trindade Island, southwestern Atlantic. Total refers to estimates using data from the three sampling years (2007, 2012, 2013). Empty cells indicate no capture and dash signs "-" indicate that fishes were counted but not weighted.

Species	2007		2012		2013		Total	
	Biomass	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass	Abundance
Abudefduf saxatilis			9.58	49.77			3.12	16.22
Acanthurus coeruleus	3.52	13.04					0.73	2.70
Anisotremus surinamensis	201.51	104.29	197.56	132.73	229.42	150.63	215.32	135.19
Balistes vetula	78.48	39.11	196.14	107.84	111.26	52.14	130.59	67.59
Bothus sp.			1.41	8.30			0.46	2.70
Canthidermis sufflamen	241.69	143.39	92.41	33.18	571.13	266.51	345.56	164.93
Caranx crysos					27.05	40.56	12.63	18.93
Caranx latus			106.93	24.89	48.50	11.59	57.49	13.52
Caranx lugubris	1121.30	1147.14	2468.74	2389.09	3169.13	2375.40	2448.53	2125.11
Caranx ruber	69.90	39.11	9.00	8.30	140.37	57.94	79.43	37.85
Carcharhinus perezi	109.19	13.04	91.25	16.59	171.48	23.17	132.41	18.93
Cephalopholis fulva	138.06	273.75	1506.71	3840.80	726.79	1402.06	797.28	1962.89
Clepticus brasiliensis					–	5.79	–	2.70
Diplodus argenteus			24.68	58.07	2.78	5.79	9.34	21.63
Echidna catenata			10.45	33.18			3.41	10.81
Elagatis bipinnulata					–	5.79	–	2.70
Epinephelus adscensionis	914.78	338.93	567.86	315.23	1313.79	677.86	996.47	489.37
Gymnothorax moringa	132.57	65.18	175.55	58.07	–	17.38	102.55	40.56
Halichoeres brasiliensis	22.54	26.07			2.73	5.79	5.95	8.11
Hemiramphus brasiliensis					–	5.79	–	2.70
Heteropriacanthus cruentatus	10.05	26.07	20.24	41.48	30.08	63.73	22.65	48.67
Hybrid (*Cephalopholis fulva* with *Paranthias furcifer*)			10.95	33.18			3.57	10.81
Holocentrus adscensionis	19.82	52.14	46.75	107.84	39.85	69.52	36.44	78.41
Kyphosus sp.	48.47	26.07	133.47	141.02	–	5.79	57.35	54.07
Malacanthus plumieri			3.19	8.30	–	5.79	2.08	5.41
Melichthys niger			663.54	2214.89	56.71	92.70	236.93	765.15
Mycteroperca interstitialis	37.48	13.04					7.77	2.70
Myripristis jacobus	6.48	13.04					1.34	2.70
Pseudupeneus maculatus			3.15	8.30			1.03	2.70
Rypticus saponaceus			3.15	8.30			1.03	2.70
Selar crumenophthalmus					9.39	11.59	4.38	5.41
Seriola rivoliana			244.76	74.66	660.26	168.02	370.28	102.74
Sparisoma amplum	855.58	417.14			656.47	254.92	480.93	205.48
Sparisoma axillare	163.08	91.25			50.70	40.56	57.49	37.85
Sphyraena barracuda	680.17	104.29	115.64	24.89	282.78	81.11	334.43	67.59
Thunnus obesus					28.07	5.79	13.10	2.70
Uraspis secunda					20.22	23.17	9.44	10.81
Total	**5254.07**	**2946.07**	**7404.12**	**9738.86**	**9754.87**	**5932.70**	**7629.52**	**6553.78**

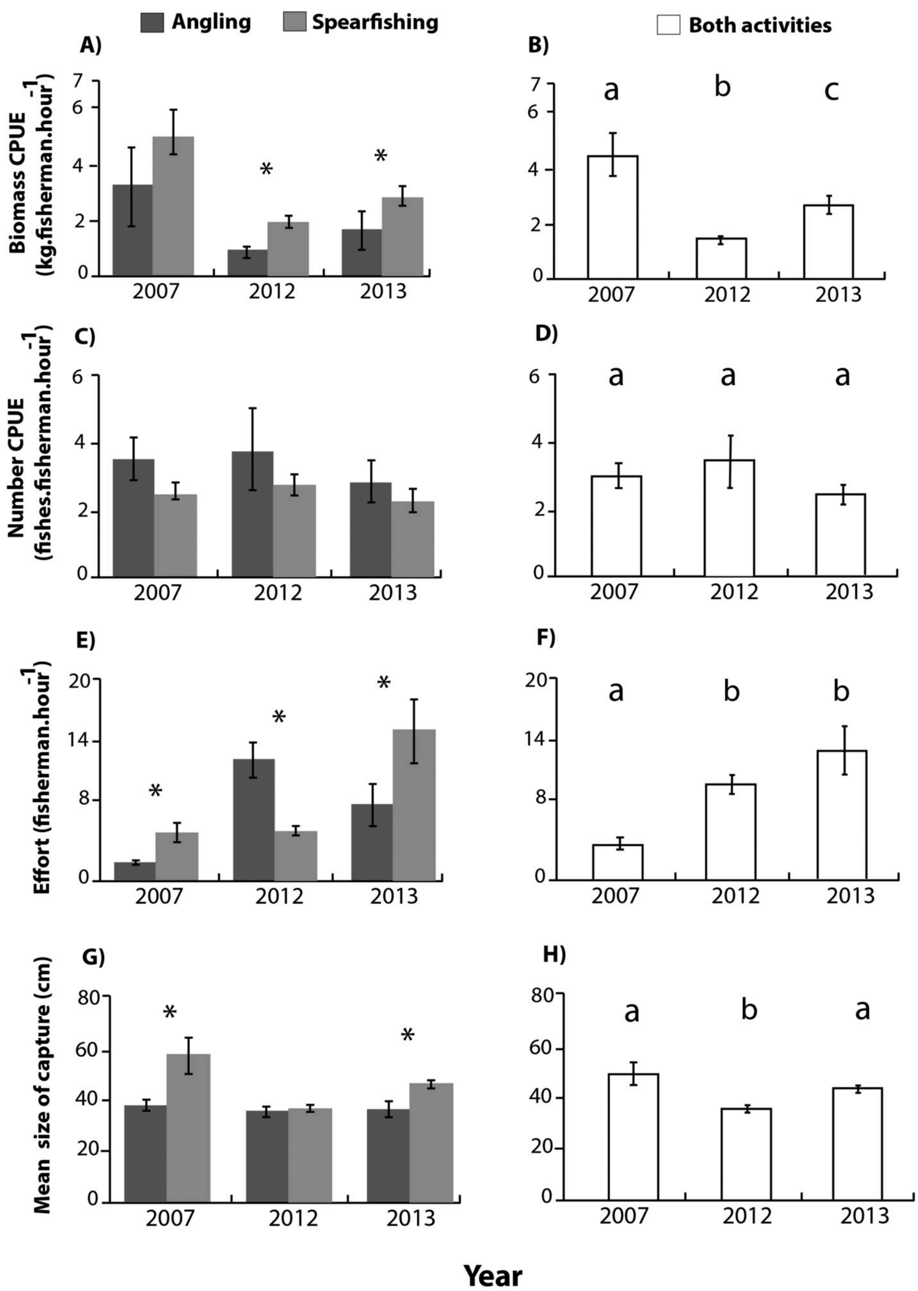

Fig. 2. Mean biomass (A and B) and fish number (C and D) catches per unit of effort (CPUE), effort (E and F) and mean size of captures (G and H) of angling (dark gray), spearfishing (light gray) and both fishing gears (white) in Trindade Island. Asterisks indicate significant differences ($p < 0.05$ between gears for the given year). Lower-case letters indicate whether the years present significant differences ($p < 0.05$: different letters) or not (same letter). Error bars show standard errors of the means.

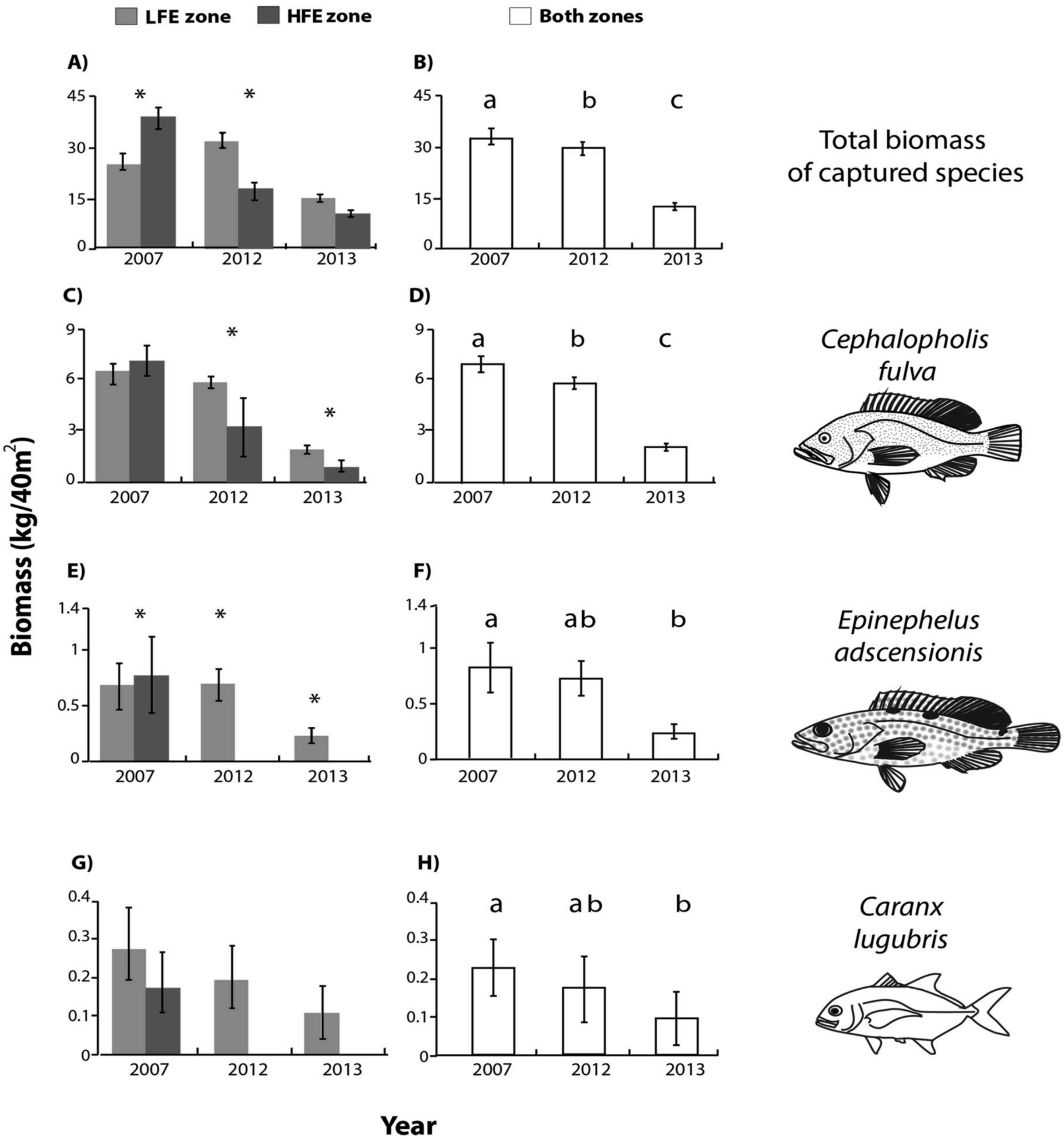

Fig. 3. Mean biomass of reef fishes observed in underwater visual census over three years of sampling: total biomass considering all captured species (A and B); and of the three most captured species, *Cephalopholis fulva* (C and D), *Epinephelus adscensionis* (E and F) and *Caranx lugubris* (E and F). Light gray bars refer to Lower Fishing Effort zone, dark gray bars to Higher Fishing Effort zone and white bars included both zones. The asterisks indicate significant differences ($p < 0.05$) between zones for the year considered. Lower-case letters indicate whether the years present significant differences ($p < 0.05$: different letters) or not (same letter). Bars show the standard error.

among years (Fig. 2).

3.3. Trends in captured and key species biomass

Total biomass per underwater visual census (UVC) of the captured species (i.e., considering all species captured by the recreational fishery) decreased significantly on transects (approximately 58%) between 2007 and 2013 (*pseudo-F* = 38.13; $p = 0.001$; Fig. 3a and b). On the other hand, total biomass of the non-captured species did not differ between years (*pseudo-F* = 0.51; $p = 0.596$) nor zones (*pseudo-F* = 1.54; $p = 0.205$). Contrasting with the decline in captured species biomass observed in Trindade, Martin Vaz Archipelago showed no significant difference in captured species' biomass between 2007 and 2013 (*pseudo-F* = 0.12; $p = 0.706$).

The overall biomass of the coney *Cephalopholis fulva* showed a sharp and significant (*pseudo-F* = 64.62; $p = 0.001$) decrease among years (more than 70% between 2007 and 2013). Such a decline in mean biomass per census was more pronounced in HFE zone (Fig. 3c and d), with significant difference among zones recorded in 2012 and 2013 (for PERMANOVAs results see appendix C).

The biomass of the rock hind grouper *Epinephelus adscensionis* also declined by approximately 70% among years of monitoring (*pseudo-F* = 7.03; $p = 0.001$) and zones (*pseudo-F* = 5.50; $p = 0.02$). Not a single individual was recorded by UVC in the HFE zone in 2012 and 2013. The biomass of the black jack *Caranx lugubris* declined almost 60% between 2007 and 2013 (*pseudo-F* = 4.05; $p = 0.023$). Although no individuals of *C. lugubris* were registered in UVC conducted in HFE zone in 2012 and 2013, no significant differences were found among zones (*pseudo-F* = 1.86; $p = 0.165$) (Fig. 3g, appendix D).

4. Discussion

High annual catch rates sustained by a decreasing CPUE and a coincident decline of key species biomass are strong evidence that recreational fisheries adapted quickly to the alterations in the reef fish community. The maintenance of 1) "non-captured" species biomass through years and zones in Trindade and 2) "captured" species biomass in Martin Vaz through the years indicates that the decline observed in captured species in Trindade was a consequence of recreational fisheries instead of natural variability. Thus, these activities are pointed as one of the main causes of changes reported to the fish communities.

Fishers' activity changed among years as the fish community structure evolved in response to exploitation. As the biomass of captured species and the CPUE declined, fishers compensated by employing higher effort. This cyclic effect has been widely reported for commercial fisheries, where fishermen increase their effort when confronted to a decline in stock biomass, then move down the food web catching species of lower trophic guilds after the collapse of their primary target resources (Myers and Worm, 2005, 2003; Pauly et al., 2005, 1998).

High CPUE associated with high fishing intensity can yield annual catch rates comparable to those of commercial fisheries (Coleman et al., 2004). Indeed, on an annual basis, estimates for recreative harvest (Table 1) represented approximately 8% of the biomass harvested by commercial fisheries that operated around Trindade (Freire and Pauly, 2015). On the other hand, the annual catch estimate in Trindade is approximately 4–7.5-fold higher than that from 85 recreational fishers in Bermuda (1350 kg year^{-1}; Pitt and Trott, 2012). Thus, enforced managed reefs located next to large human populations (e.g., Bermuda) can yield lower catches than more remote locations without proper management such as Trindade.

The effects of recreational and commercial fisheries in Trindade could be isolated since these two fishery types primarily target different species, with commercial activities focusing mainly in sharks and swordfish (Pinheiro et al., 2010b). When both activities have the same targets (e.g., *Caranx lugubris and Epinephelus adscensionis*), the area or habitat is not shared, since commercial fisheries usually operate in deeper areas (20–150 m), while recreational ones are deployed in shallow areas (<20 m depth) closer to shorelines. Moreover, commercial fishing vessels do not usually fish next to the HFE zone (the most impacted zone) because of the presence of the Navy base, which intimidates approaching vessels.

Although *E. adscensionis* and *C. lugubris* were still being captured by recreational fisheries in HFE zone, their absence in 2012 and 2013 UVC records indicates that abundance of both species had severely declined. On the same track, one of the most abundant species of Trindade (Pinheiro et al., 2011), the coney *C. fulva*, experienced a severe decline from 2007 onwards, but was still present in both fishing zones. As these key species are predators, they exert an important ecological role in reef systems controlling fish community structure (Dulvy et al., 2004; Jennings and Polunin, 1997), and management actions are crucial in order to prevent a population collapse and community cascade effects.

Recreational fisheries are an important social and cultural activity practiced along coastlines all around the world (Cooke and Cowx, 2004; Lewin et al., 2006; Post et al., 2002), but have a relevant environmental impact even in remote oceanic islands (Jiménez-Alvarado et al., 2019). Our results pointed out that the management of recreational activities is required in order to ensure a proper protection of species and ecosystem functions in such islands. The success of fishery management and MPA conservation are often built on the users' participation and collaboration (Hernandez and Kempton, 2003; Pereira and Hansen, 2003). Current dialogue among scientists, MPA managers and military authorities in Trindade is yielding positive perspectives and hope for conservation goals achievement and MPA rules enforcement. However, while the scenario is promising, management and enforcement represent a major challenge for all involved parties.

Nevertheless, the establishment of Trindade's MPAs should be considered a strong motivation for the necessary paradigm conversion of recreative activities in priority areas for conservation. Nowadays, where all Brazilian oceanic islands compose different MPAs, the Brazilian navy should not only guarantee sovereignty at sea but also support Brazilian environmental agencies through surveillance actions and law enforcement. Fishing exclusion zones, protection of key and endangered species as well as a monitoring program for recreational fisheries landings are among priority actions for conservation to be taken. Concomitantly to fishing management, a shift toward sustainable and non-destructive activities must be encouraged as leisure activities in isolated islands. Snorkeling and recreational SCUBA diving, underwater photography and citizen-science activities, such as Reef Check, iNaturalist, among others, need to receive incentive and logistic support.

Declaration of competing interest

None.

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Appendix A. Catch per unit effort (kg*fisher^{-1}*hour^{-1} ± SE) of reef fishes species captured by recreational fisheries in Trindade Island, Southwestern Atlantic

Species	2007			2012			2013			Total
	Angling	Spearfishing	Total	Angling	Spearfishing	Total	Angling	Spearfishing	Total	
Abudefduf saxatilis	–	–	–	0.01 ± 0.01	–	0.01 ± 0.01	–	–	–	<0.01
Acanthurus coeruleus	–	<0.01	<0.01	–	–	–	–	–	–	<0.01
Anisotremus surinamensis	0.26 ± 0.17	0.09 ± 0.07	0.16 ± 0.08	<0.01	0.11 ± 0.04	0.04 ± 0.02	–	0.07 ± 0.02	0.05 ± 0.02	0.06 ± 0.02
Balistes vetula	–	0.05 ± 0.05	0.03 ± 0.03	0.01 ± 0.01	0.09 ± 0.04	0.04 ± 0.02	–	0.03 ± 0.02	0.03 ± 0.02	0.03 ± 0.01
Bothus sp.	–	–	–	–	<0.01	<0.01	–	–	–	<0.01
Canthidermis sufflamen	–	0.33 ± 0.16	0.19 ± 0.10	0.02 ± 0.01	0.05 ± 0.05	0.03 ± 0.02	–	0.56 ± 0.24	0.41 ± 0.19	0.14 ± 0.05
Caranx crysos	–	–	–	–	–	–	0.02 ± 0.02	0.02 ± 0.02	0.02 ± 0.01	<0.01
Caranx latus	–	–	–	0.01 ± 0.01	–	<0.01	0.02 ± 0.02	0.01 ± 0.01	0.01 ± 0.01	<0.01
Caranx lugubris	1.24 ± 0.35	0.58 ± 0.23	0.86 ± 0.21	0.42 ± 0.11	0.30 ± 0.09	0.38 ± 0.07	1.40 ± 0.69	1.15 ± 0.25	1.21 ± 0.25	0.65 ± 0.09
Caranx ruber	–	0.17 ± 0.10	0.10 ± 0.06	–	<0.01	<0.01	0.15 ± 0.15	0.03 ± 0.02	0.06 ± 0.04	0.03 ± 0.01
Carcharhinus perezi	0.52 ± 0.52	–	0.22 ± 0.22	0.02 ± 0.01	–	0.01 ± 0.01	–	0.07 ± 0.05	0.05 ± 0.04	0.05 ± 0.04
Cephalopholis fulva	0.45 ± 0.17	0.01 ± 0.01	0.19 ± 0.09	0.17 ± 0.03	0.57 ± 0.11	0.32 ± 0.05	0.06 ± 0.03	0.10 ± 0.05	0.09 ± 0.04	0.25 ± 0.04
Clepticus brasiliensis	–	–	–	–	–	–	–	–	–	–
Diplodus argenteus	–	–	–	–	0.02 ± 0.01	<0.01	–	<0.01	<0.01	<0.01
Echidna catenata	–	–	–	<0.01	–	<0.01	–	–	–	<0.01
Elagatis bipinnulata	–	–	–	–	–	–	–	–	–	–
Epinephelus adscensionis	0.47 ± 0.31	0.60 ± 0.20	0.54 ± 0.17	0.05 ± 0.03	0.47 ± 0.14	0.21 ± 0.06	0.08 ± 0.08	0.37 ± 0.09	0.30 ± 0.07	0.28 ± 0.05
Gymnothorax moringa	0.02 ± 0.02	0.31 ± 0.25	0.19 ± 0.15	<0.01	0.16 ± 0.11	0.06 ± 0.04	–	–	–	0.07 ± 0.04
Halichoeres brasiliensis	–	0.02 ± 0.02	0.01 ± 0.01	–	–	–	–	<0.01	<0.01	<0.01
Hemiramphus brasiliensis	–	–	–	–	–	–	–	–	–	–
Heteropriacanthus cruentatus	–	0.01 ± 0.01	<0.01	–	0.01 ± 0.01	<0.01	–	0.01 ± 0.01	0.01 ± 0.01	<0.01
Hybrid	–	–	–	<0.01	–	<0.01	–	–	–	<0.01
Holocentrus adscensionis	0.06 ± 0.06	<0.01	0.03 ± 0.03	<0.01	0.02 ± 0.01	<0.01	–	<0.01	<0.01	<0.01
Kyphosus sp.	0.23 ± 0.23	–	0.1 ± 0.1	0.04 ± 0.02	0.01 ± 0.01	0.03 ± 0.01	–	–	–	0.03 ± 0.02
Malacanthus plumieri	–	–	–	–	<0.01	<0.01	–	–	–	<0.01
Melichthys niger	–	–	–	0.18 ± 0.17	<0.01	0.11 ± 0.11	0.02 ± 0.01	–	<0.01	0.07 ± 0.06
Mycteroperca interstitialis	–	0.09 ± 0.09	0.05 ± 0.05	–	–	–	–	–	–	0.01 ± 0.01
Myripristis jacobus	–	<0.01	<0.01	–	–	–	–	–	–	<0.01
Pseudupeneus maculatus	–	–	–	–	<0.01	<0.01	–	–	–	<0.01
Rypticus saponaceus	–	–	–	<0.01	–	<0.01	–	–	–	<0.01
Selar crumenophthalmus	–	–	–	–	–	–	–	0.01 ± 0.01	<0.01	<0.01
Seriola rivoliana	–	–	–	–	0.11 ± 0.05	0.04 ± 0.02	–	0.05 ± 0.03	0.04 ± 0.02	0.03 ± 0.01
Sparisoma amplum	–	0.68 ± 0.19	0.39 ± 0.14	–	–	–	–	0.32 ± 0.13	0.24 ± 0.10	0.12 ± 0.03
Sparisoma axillare	–	0.15 ± 0.11	0.09 ± 0.07	–	–	–	–	0.02 ± 0.01	0.01 ± 0.01	0.02 ± 0.01
Sphyraena barracuda	–	2.12 ± 1.13	1.23 ± 0.69	–	0.08 ± 0.05	0.03 ± 0.02	–	0.08 ± 0.04	0.06 ± 0.03	0.23 ± 0.12
Thunnus obesus	–	–	–	–	–	–	–	0.01 ± 0.01	0.01 ± 0.01	<0.01
Uraspis secunda	–	–	–	–	–	–	–	0.01 ± 0.01	<0.01	<0.01

Appendix B. Results of PERMANOVA analyzes on recreational fisheries variables: biomass and number CPUEs; fishing effort; mean size of captures; diversity index of captures (Shannon-Wiener). Acronyms as follow: df: degrees of freedom; SS: sum of squares; MS: mean sum of squares

VariablesFactors	Df	SS	MS	Pseudo-F	p-value	Permutations
Biomass CPUE						
Year	2	9.8223	4.9111	18	0.001	999
Gear	1	5.4495	5.4495	20	0.001	998
Year*Gear	2	0.0336	0.0168	0.060973	0.937	998
Residual	111	30.5840	0.27554			
Total	116	51.7880				
Number CPUE						
Year	2	0.86124	0.43062	0.55338	0.596	998
Gear	1	0.29959	0.29959	0.38499	0.536	998
Year*Gear	2	0.73416	0.36708	0.47172	0.634	998
Residual	143	111.28	0.77817			
Total	148	112.90				
Fishing effort						
Year	2	27.992	13.996	9.6473	0.001	997
Gear	1	1.5503	1.5503	1.0686	0.287	998
Year*Gear	2	27.11	13.555	9.3435	0.001	998
Residual	143	207.46	1.4508			
Total	148	265.13				
Mean size of captures						
Year	2	11.2120	5.6058	9.2944	0.002	999
Gear	1	10.8770	10.8770	18.0340	0.001	998
Year*Gear	2	5.8644	2.9322	4.8616	0.019	998
Residual	111	66.9480	0.60314			
Total	116	96.9520				
Diversity index						
Year	2	0.022714	0.011357	0.060227	0.944	999
Gear	1	3.8057	3.8057	20.182	0.001	998
Year*Gear	2	0.53798	0.26899	1.4265	0.232	999
Residual	143	26.965	0.18857			
Total	148	31.898				

Appendix C. Results of PERMANOVA analyzes on reef fish community parameters (total biomass of captured species, total biomass of non-captured species), and biomass of key species *Cephalopholis fulva*, *Epinephelus adscensionis* and *Caranx lugubris* in Trindade island. Acronyms as follow: df: degrees of freedom; SS: sum of squares; MS: mean sum of squares

Variables factors	Df	SS	MS	Pseudo-F	p-value	Permutations
Total biomass of captured species						
Year	2	229.37	114.69	38.137	0.001	998
Zone	1	3.4594	3.4594	1.1504	0.242	994
Year*Zone	2	54.498	29.249	9.7263	0.001	999
Residual	409	1230	3.0072			
Total	414	1677.7				
Total biomass of non-captured species						
Year	2	0.58439	0.2922	0.51519	0.596	998
Zone	1	0.87434	0.87434	1.54160	0.205	994
Year*Zone	2	0.25318	0.12659	0.22320	0.786	999
Residual	409	231.97	0.56716			
Total	414	234.29				
Cephalopholis fulva						
Year	2	98.2920	49.1460	64.6200	0.001	998
Zone	1	1.7610	1.7610	2.3154	0.125	996
Year*Zone	2	7.4552	3.7276	4.9013	0.009	999
Residual	409	311.06	0.7605			
Total	414	465.27				
Epinephelus adscensionis						
Year	2	6.2640	3.1320	7.0371	0.001	999
Zone	1	2.4511	2.4511	5.5072	0.020	998
Year*Zone	2	1.5025	0.7513	1.6880	0.199	998
Residual	409	182.03	0.4451			
Total	414	193.23				
Caranx lugubris						
Year	2	1.129600	0.5648	4.0559	0.023	999
Zone	1	0.260370	0.2604	1.8697	0.165	997
Year*Zone	2	0.018011	0.0090	0.0647	0.938	998
Residual	409	56.957	0.1393			
Total	414	58.159				

Appendix D. Mean biomass (±standard error; kg*40m^{-2}) of reef fishes species captured by recreational fisheries recorded per underwater visual census in Trindade, Southwestern Atlantic. LFE: lower fishing effort zone; HFE: higher fishing effort zone

Species	2007			2012			2013			Total
	LFE	HFE	Total	LFE	HFE	Total	LFE	HFE	Total	
Abudefduf saxatilis	0.12 ± 0.03	0.24 ± 0.05	0.20 ± 0.04	0.34 ± 0.05	0.60 ± 0.19	0.37 ± 0.05	0.31 ± 0.04	0.51 ± 0.07	0.35 ± 0.03	0.33 ± 0.02
Acanthurus coeruleus	0.32 ± 0.14	0.36 ± 0.1	0.34 ± 0.08	0.25 ± 0.08	0.10 ± 0.10	0.23 ± 0.07	0.19 ± 0.05	0.09 ± 0.04	0.17 ± 0.04	0.22 ± 0.04
Anisotremus surinamensis	0.04 ± 0.04	0.35 ± 0.17	0.23 ± 0.11	0.31 ± 0.10	–	0.27 ± 0.09	0.06 ± 0.03	–	0.04 ± 0.02	0.17 ± 0.04
Balistes vetula	0.04 ± 0.04	0.31 ± 0.09	0.22 ± 0.06	0.25 ± 0.09	–	0.22 ± 0.07	0.07 ± 0.04	0.01 ± 0.01	0.06 ± 0.03	0.15 ± 0.03
Bothus sp.	–	–	–	–	–	–	–	0.01 ± 0.01	<0.01	<0.01
Canthidermis sufflamen	0.39 ± 0.19	0.54 ± 0.14	0.49 ± 0.11	0.04 ± 0.03	–	0.03 ± 0.02	0.11 ± 0.04	0.24 ± 0.09	0.14 ± 0.04	0.16 ± 0.03
Caranx crysos	–	–	–	–	–	–	0.01 ± 0.01	0.04 ± 0.02	0.01 ± 0.01	0.01 ± 0.003
Caranx latus	–	0.06 ± 0.04	0.04 ± 0.03	–	–	–	–	–	–	0.01 ± 0.01
Caranx lugubris	0.25 ± 0.13	0.21 ± 0.09	0.23 ± 0.07	0.20 ± 0.10	–	0.17 ± 0.09	0.12 ± 0.09	–	0.09 ± 0.07	0.15 ± 0.05
Carcharhinus perezi	–	–	–	0.09 ± 0.09	–	0.08 ± 0.08	–	0.61 ± 0.61	0.13 ± 0.13	0.09 ± 0.07
Caranx ruber	–	0.08 ± 0.04	0.05 ± 0.02	0.02 ± 0.01	–	0.02 ± 0.01	0.04 ± 0.02	0.01 ± 0.01	0.03 ± 0.02	0.03 ± 0.01
Cephalopholis fulva	5.89 ± 0.86	7.42 ± 0.61	6.87 ± 0.50	5.93 ± 0.35	4.71 ± 1.31	5.77 ± 0.35	2.18 ± 0.25	1.11 ± 0.23	1.94 ± 0.21	4.29 ± 0.21
Diplodus argenteus	0.14 ± 0.09	0.25 ± 0.13	0.21 ± 0.09	0.04 ± 0.02	–	0.04 ± 0.02	0.12 ± 0.06	–	0.09 ± 0.05	0.09 ± 0.03
Echidna catenata	–	–	–	<0.01	–	<0.01	<0.01	–	<0.01	<0.01
Epinephelus adscensionis	0.8 ± 0.33	0.87 ± 0.3	0.84 ± 0.22	0.87 ± 0.18	–	0.75 ± 0.16	0.31 ± 0.09	–	0.24 ± 0.07	0.54 ± 0.08
Gymnothorax moringa	<0.01	<0.01	<0.01	0.36 ± 0.11	0.47 ± 0.31	0.38 ± 0.1	0.18 ± 0.15	0.17 ± 0.11	0.18 ± 0.12	0.22 ± 0.07
Halichoeres brasiliensis	0.49 ± 0.12	0.26 ± 0.05	0.35 ± 0.06	0.27 ± 0.05	0.17 ± 0.08	0.25 ± 0.05	0.28 ± 0.04	0.26 ± 0.05	0.28 ± 0.03	0.28 ± 0.03
Hemiramphus brasiliensis	–	<0.01	<0.01	–	–	–	–	–	–	<0.01
Heteropriacanthus cruentatus	0.08 ± 0.04	0.19 ± 0.06	0.15 ± 0.04	0.06 ± 0.03	–	0.05 ± 0.02	0.04 ± 0.02	–	0.03 ± 0.02	0.06 ± 0.01
Hybrid	–	–	–	0.11 ± 0.03	0.02 ± 0.02	0.1 ± 0.03	0.02 ± 0.01	0.03 ± 0.02	0.02 ± 0.01	0.05 ± 0.01
Holocentrus adscensionis	0.72 ± 0.21	1.24 ± 0.21	1.05 ± 0.16	1.35 ± 0.45	0.54 ± 0.12	1.24 ± 0.39	0.48 ± 0.11	0.42 ± 0.09	0.47 ± 0.08	0.87 ± 0.16
Kyphosus sp.	1.50 ± 0.63	11.87 ± 2.54	8.13 ± 1.73	5.63 ± 1.04	0.40 ± 0.40	4.94 ± 0.91	3.35 ± 0.64	1.44 ± 0.73	2.93 ± 0.52	4.63 ± 0.53
Malacanthus plumieri	0.02 ± 0.02	0.01 ± 0.01	0.01 ± 0.01	0.02 ± 0.01		0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.01
Melichthys niger	12.19 ± 2.06	10.05 ± 0.77	10.82 ± 0.89	12.84 ± 1.71	8.36 ± 1.81	12.25 ± 1.5	5.83 ± 0.61	2.92 ± 0.54	5.19 ± 0.5	8.89 ± 0.65
Myripristis jacobus	0.64 ± 0.20	1.56 ± 0.31	1.23 ± 0.21	1.19 ± 0.20	0.51 ± 0.33	1.1 ± 0.18	0.32 ± 0.08	0.46 ± 0.15	0.35 ± 0.07	0.79 ± 0.09
Rypticus saponaceus	0.04 ± 0.02	0.03 ± 0.01	0.03 ± 0.01	0.09 ± 0.02	0.09 ± 0.05	0.09 ± 0.02	0.13 ± 0.02	0.07 ± 0.03	0.11 ± 0.02	0.09 ± 0.01
Seriola rivoliana	–	–	–	0.05 ± 0.03	–	0.05 ± 0.03	–	–	–	0.02 ± 0.01
Sparisoma amplum	0.21 ± 0.11	1.55 ± 0.65	1.06 ± 0.42	0.64 ± 0.19	0.68 ± 0.38	0.64 ± 0.17	0.33 ± 0.07	1.32 ± 0.48	0.55 ± 0.12	0.68 ± 0.11
Sparisoma axillare	0.29 ± 0.13	0.15 ± 0.05	0.21 ± 0.06	0.44 ± 0.08	0.32 ± 0.17	0.42 ± 0.07	0.11 ± 0.04	0.05 ± 0.04	0.09 ± 0.03	0.24 ± 0.03
Sphyraena barracuda	0.32 ± 0.32	0.74 ± 0.25	0.59 ± 0.21	0.32 ± 0.21	–	0.28 ± 0.17	0.07 ± 0.03	0.21 ± 0.13	0.11 ± 0.04	0.25 ± 0.08

References

Almeida, F.F.M., 1961. Geologia e Petrologia Da Ilha de Trindade, 19. Ministério das Minas e Energia, Dep. Nac. da Prod. Miner., pp. 1–249

Anderson, M.J., Goerley, R.N., Clarke, K.R., 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods, first. PRIMER-E Ltd, Plymouth, UK, pp. 1–214.

Bellwood, D.R., Hughes, T.P., Folke, C., Nyström, M., 2004. Confronting the coral reef crisis. Nature 429, 827–833. https://doi.org/10.1038/nature02691.

Brasil, 2018. Decreto de criação da área de proteção ambiental do arquipélago de Trindade e Martim Vaz e o ,monumento natural das ilhas de Trindade e Martim Vaz e do Monte Columbia. Diário Oficial da União, Brasil.

Cesar, H., Burke, L., Pet-Soede, L., 2003. The economics of worldwide coral reef degradation. Cesar Environmental Economics Consulting, Netherlands, pp. 1–24.

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

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

Cooke, S.J., Cowx, I.G., 2006. Contrasting recreational and commercial fishing: searching for common issues to promote unified conservation of fisheries resources and aquatic environments. Biol. Conserv. 128, 93–108. https://doi.org/10.1016/j.biocon.2005.09.019.

DeMartini, E.E., Friedlander, A.M., Sandin, S.A., Sala, E., 2008. Differences in fish-assemblage structure between fished and unfished atolls in the northern Line Islands, central Pacific. Mar. Ecol. Prog. Ser. 365, 199–215. https://doi.org/10.3354/meps07501.

Dulvy, N.K., Freckleton, R.P., Polunin, N.V.C., 2004. Coral reef cascades and the indirect effects of predator removal by exploitation. Ecol. Lett. 7, 410–416. https://doi.org/10.1111/j.1461-0248.2004.00593.x.

Edgar, G.J., Ward, T.J., Stuart-Smith, R.D., 2018. Rapid declines across Australian fishery stocks indicate global sustainability targets will not be achieved without an expanded network of 'no-fishing' reserves. Aquat. Conserv. Mar. Freshw. Ecosyst. 28, 1337–1350. https://doi.org/10.1002/aqc.2934.

Freire, K.M.F., Pauly, D., 2015. Fisheries Catch Reconstructions for Brazil'S Mainland and Oceanic Islands. Fisheries Centre Research Reports.

Frisch, A.J., Baker, R., Hobbs, J.P.A., Nankervis, L., 2008. A quantitative comparison of recreational spearfishing and linefishing on the Great Barrier Reef: implications for management of multi-sector coral reef fisheries. Coral Reefs 27, 85–95. https://doi.org/10.1007/s00338-007-0293-z.

Frisch, A.J., Cole, A.J., Hobbs, J.P.A., Rizzari, J.R., Munkres, K.P., 2012. Effects of spearfishing on reef fish populations in a multi-use conservation area. PloS One 7. https://doi.org/10.1371/journal.pone.0051938.

Froese, R., 2006. Cube law, condition factor and weight-length relationships: history, meta-analysis and recommendations. J. Appl. Ichthyol. 22, 241–253. https://doi.org/10.1111/j.1439-0426.2006.00805.x.

Froese, R., Pauly, D., 2018. FishBase. www.fishbase.org (accessed 10.2.18).

Frota, L.O., Costa, P.A.S., Braga, A.C., 2004. Length-weight relationships of marine fishes from the central Brazilian coast. Naga WorldFish Cent. Q. 27, 20–26.

Garrison, L.P., Link, J.S., 2000. Fishing effects on spatial distribution and trophic guild structure of the fish community in the Georges Bank region. ICES J. Mar. Sci. 57, 723–730. https://doi.org/10.1006/jmsc.2000.0713.

Gasparini, J.L., Floeter, S.R., 2001. The shore fishes of trindade island, western South Atlantic. J. Nat. Hist. 35, 1639–1656. https://doi.org/10.1080/002229301317092379.

Giglio, V.J., Pinheiro, H.T., Bender, M.G., Bonaldo, R.M., Costa-Lotufo, L.V., Ferreira, C.E.L., Floeter, S.R., Freire, A., Gasparini, J.L., Joyeux, J.C., Krajewski, J.P., Lindner, A., Longo, G.O., Lotufo, T.M.C., Loyola, R., Luiz, O.J., Macieira, R.M., Magris, R.A., Mello, T.J., Quimbayo, J.P., Rocha, L.A., Segal, B., Teixeira, J.B., Vila-Nova, D.A., Vilar, C.C., Zilberberg, C., Francini-Filho, R.B., 2018. Large and remote marine protected areas in the South Atlantic Ocean are flawed and raise concerns: comments on Soares and Lucas. Mar. Pol. 96, 13–17. https://doi.org/10.1016/j.marpol.2018.07.017 (2018).

Gomes, R., dos, S., Lima, F.D., Barbosa, J.C., Spotorno-Oliveira, P., Costa, P.M.S., 2017. Molucos da Ilha Trindade e Martin Vaz. In: Costa-Abrantes, S., Sissini, M.N. (Eds.), Protrindade: Programa de Pesquisas Científicas Na Ilha Da Trindade. 10 Anos de Pesquisas. SECIRM, Brasília, pp. 133–137.

Guabiroba, H.C., Joyeux, J.-C., 2018. Length-weight relationships for reef fishes in a southwestern Atlantic tropical oceanic island. Pan Am. J. Aquat. Sci. 13, 84–87.

Hernandez, A., Kempton, W., 2003. Changes in fisheries management in Mexico: effects of increasing scientific input and public participation. Ocean Coast Manag. 46, 507–526. https://doi.org/10.1016/S0964-5691(03)00032-2.

Hilborn, R., Branch, T.A., Ernst, B., Magnusson, A., Minte-Vera, C.V., Scheuerell, M.D., Valero, J.L., 2003. State of the world's fisheries. Annu. Rev. Environ. Resour. 28, 359–399. https://doi.org/10.1146/annurev.energy.28.050302.105509.

Hughes, T.P., Barnes, M.L., Bellwood, D.R., Cinner, J.E., Cumming, G.S., Jackson, J.B.C., Kleypas, J., Van De Leemput, I.A., Lough, J.M., Morrison, T.H., Palumbi, S.R., Van Nes, E.H., Scheffer, M., 2017. Coral reefs in the anthropocene. Nature 546, 82–90. https://doi.org/10.1038/nature22901.

ICMBio, 2018. Pesca é regulamentada em áreas protegidas marinhas. http://www.icmbio.gov.br/portal/ultimas-noticias/20-geral/9997-pesca-e-regulamentada-em-novas-areas-protegidas-marinhas (accessed 10.20.18).

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., Warner, R.R., 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293, 629–637. https://doi.org/10.1126/science.1059199.

Jennings, S., Polunin, N.V.C., 1997. Impacts of predator depletion by fishing on the biomass and diversity of non-target reef fish communities. Coral Reefs 16, 71–82. https://doi.org/10.1007/s003380050061.

Jiménez-Alvarado, D., Sarmiento-Lezcano, A., Guerra-Marrero, A., Tuya, F., Santana del Pino, Á., Sealey, M.J., Castro, J.J., 2019. Historical photographs of captures of recreational Fishers indicate overexploitation of nearshore resources at an oceanic island. J. Fish. Biol. https://doi.org/10.1111/jfb.13969 jfb.13969.

Knowlton, N., Jackson, J.B.C., 2008. Shifting baselines, local impacts, and global change on coral reefs. PLoS Biol. 6 https://doi.org/10.1371/journal.pbio.0060054, 0215–0220.

Lewin, W.-C., Arlinghaus, R., Mehner, T., 2006. Documented and potential biological impacts of recreational fishing: insights for management and conservation. Rev. Fish. Sci. 14, 305–367. https://doi.org/10.1080/10641260600886455.

Luiz, O.J., Edwards, A.J., 2011. Extinction of a shark population in the Archipelago of Saint Paul's Rocks (equatorial Atlantic) inferred from the historical record. Biol. Conserv. 144, 2873–2881. https://doi.org/10.1016/j.biocon.2011.08.004.

Marengo, M., Culioli, J.M., Santoni, M.C., Marchand, B., Durieux, E.D.H., 2015. Comparative analysis of artisanal and recreational fisheries for Dentex dentex in a Marine Protected Area. Fish. Manag. Ecol. 22, 249–260. https://doi.org/10.1111/fme.12110.

Meyer, C.G., 2007. The impacts of spear and other recreational Fishers on a small permanent Marine Protected Area and adjacent pulse fished area. Fish. Res. 84, 301–307. https://doi.org/10.1016/j.fishres.2006.11.004.

Myers, R.A., Worm, B., 2003. Rapid worldwide depletion of predatory fish communities. Nature 423, 280–283. https://doi.org/10.1038/nature01610.

Myers, R.A., Worm, B., 2005. Extinction, survival or recovery of large predatory fishes. Philos. Trans. R. Soc. B Biol. Sci. 360, 13–20. https://doi.org/10.1098/rstb.2004.1573.

Nunes, J.A.C.C., Medeiros, D.V., Reis-Filho, J.A., Sampaio, C.L.S., Barros, F., 2012. Reef fishes captured by recreational spearfishing on reefs of Bahia State, northeast Brazil. Biota Neotropica 12, 179–185. https://doi.org/10.1590/S1676-06032012000100014.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., Torres Jr., F., 1998. Fishing down marine food webs. Science 279, 860–863. https://doi.org/10.1126/science.279.5352.860.

Pauly, D., Watson, R., Alder, J., 2005. Global trends in world fisheries: impacts on marine ecosystems and food security. Philos. Trans. R. Soc. B Biol. Sci. 360, 5–12. https://doi.org/10.1098/rstb.2004.1574.

Pedroso, D., Panisset, J. de S., Abdo, L.B.B., 2017. Climatologia da Ilha da Trindade. In: Costa-Abrantes, S., Sissini, M.N. (Eds.), Protrindade: Programa de Pesquisas Científicas Na Ilha Da Trindade. 10 Anos de Pesquisas. SECIRM, Brasília, pp. 27–32.

Pereira, D.L., Hansen, M.J., 2003. A perspective on challenges to recreational fisheries management: summary of the symposium on active management of recreational fisheries. N. Am. J. Fish. Manag. 23, 1276–1282. https://doi.org/10.1577/M01-234.

Pinheiro, H.T., Gasparini, J.L., 2009. Peixes Recifais do Complexo Insular Oceânico Trindade-Martin Vaz: Novas Ocorrências, Atividades de Pesca, Mortandade Natural e Conservação. In: Mohr, L.V., Castro, J.W.A., Costa, P.M.S., Alves, R.J.V. (Eds.), Ilhas Oceânicas Brasileiras: Da Pesquisa Ao Manejo. MMA Secretaria de Biodiversidade e Florestas., Brasília, pp. 135–153.

Pinheiro, H.T., Joyeux, J.C., 2015. The role of recreational fishermen in the removal of target reef fishes. Ocean Coast Manag. 112, 12–17. https://doi.org/10.1016/j.ocecoaman.2015.04.015.

Pinheiro, H.T., Joyeux, J.C., Martins, A.S., 2010a. Reef fisheries and underwater surveys indicate overfishing of a Brazilian Coastal Island. Nat. Conserv. 8, 151–159. https://doi.org/10.4322/natcon.00802008.

Pinheiro, H.T., Martins, A.S., Gasparini, J.L., 2010b. Impact of commercial fishing on Trindade Island and Martin Vaz Archipelago, Brazil: characteristics, conservation status of the species involved and prospects for preservation. Braz. Arch. Biol. Technol. 53, 1417–1423. https://doi.org/10.1590/S1516-89132010000600018.

Pinheiro, H.T., Ferreira, C.E.L., Joyeux, J.-C., Santos, R.G., Horta, P.A., 2011. Reef fish structure and distribution in a south-western Atlantic Ocean tropical island. J. Fish. Biol. 79, 1984–2006. https://doi.org/10.1111/j.1095-8649.2011.03138.x.

Pinheiro, H.T., Mazzei, E., Moura, R.L., Amado-Filho, G.M., Carvalho-Filho, A., Braga, A.C., Costa, P.A.S., Ferreira, B.P., Ferreira, C.E.L., Floeter, S.R., Francini-Filho, R.B., Gasparini, J.L., Macieira, R.M., Martins, A.S., Olavo, G., Pimentel, C.R., Rocha, L.A., Sazima, I., Simon, T., Teixeira, J.B., Xavier, L.B., Joyeux, J.-C., 2015. Fish biodiversity of the Vitória-trindade Seamount Chain, southwestern Atlantic: an updated database. PloS One 10. https://doi.org/10.1371/journal.pone.0118180 e0118180.

Pitt, J.M., Trott, T.M., 2012. Insights from a survey of the recreational fishery in Bermuda. In: Proceedings of the Sixty-Fifth Annual Gulf and Caribbean Fisheries Institute, pp. 254–261.

Post, J.R., Sullivan, M., Cox, S., Lester, N.P., Walters, C.J., Parkinson, E.A., Paul, A.J., Jackson, L., Shuter, B.J., 2002. Canada's recreational fisheries: the invisible collapse? Fisheries 27, 6–17. https://doi.org/10.1577/1548-8446(2002)027<0006:crf>2.0.co;2.

Sandin, S.A., Smith, J.E., DeMartini, E.E., Dinsdale, E.A., Donner, S.D., Friedlander, A.M., Konotchick, T., Malay, M., Maragos, J.E., Obura, D., Pantos, O., Paulay, G., Richie, M., Rohwer, F., Schroeder, R.E., Walsh, S., Jackson, J.B.C., Knowlton, N., Sala, E., 2008. Baselines and degradation of coral reefs in the northern line islands. PloS One 3. https://doi.org/10.1371/journal.pone.0001548.

Santos, M.E.A., Kitahara, M.V., Lindner, A., Reimer, J.D., 2016. Overview of the order zoantharia (Cnidaria: anthozoa) in Brazil. Mar. Biodivers. 46, 547–559. https://doi.org/10.1007/s12526-015-0396-7.

Santos, M.E.A., Wirtz, P., Montenegro, J., Kise, H., López, C., Brown, J., Reimer, J.D., 2019. Diversity of Saint Helena island and zoogeography of zoantharians in the Atlantic ocean: jigsaw falling into place. Syst. Biodivers. 17, 165–178. https://doi.org/10.1080/14772000.2019.1572667.

Santos, M.E.A., Faria-Junior, E., Aued, A.W., Peluso, L., Kitahara, M.V., Pires, D.O., Zilberberg, C., 2020. Benthic Cnidaria community in the oceanic archipelago of trindade and Martin Vaz, southwestern Atlantic ocean. Reg. Stud. Mar. Sci. 33, 1–8. https://doi.org/10.1016/j.rsma.2019.100895.

Simon, T., Macieira, R.M., Joyeux, J.C., 2013. The shore fishes of the Trindade-Martin Vaz insular complex: an update. J. Fish. Biol. 82, 2113–2127. https://doi.org/10.1111/jfb.12126.

Sissini, M.N., Oliveira, M.C. de, Horta, P.A., Pellizzari, F., 2017. Macroalgas da Ilha da Trindade. In: Costa-Abrantes, S., Sissini, M.N. (Eds.), Protrindade: Programa de Pesquisas Científicas Na Ilha Da Trindade. 10 Anos de Pesquisas. SECIRM, Brasília, pp. 99–107.

Soares, M. de O., Lucas, C.C., 2018. Towards large and remote protected areas in the south Atlantic ocean: st. Peter and st. Paul's archipelago and the Vitória-trindade Seamount Chain. Mar. Pol. 93, 101–103. https://doi.org/10.1016/j.marpol.2018.04.004.