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Contrasting patterns in the abundance of fish communities targeted by fishers on two coral reefs in southern Mozambique

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Abstract :

Coastal populations of maritime countries in eastern Africa rely on fish as a primary source of protein, but baseline information on the abundance of fish communities on these coastlines is often lacking. We used baited remote underwater video stations to compare the abundance and diversity of reef fishes targeted by fishing at two sites in southern Mozambique, one at Lighthouse Reef within the Bazaruto Archipelago National Park and the other to the south at San Sebastian Reef on the San Sebastian Peninsula. Fish that are known targets of fisheries (mostly small-scale and artisanal) had an abundance that was almost three-times greater at San Sebastian Reef (80.22 ind. h⁻¹ [SE 18.00]) than at Lighthouse Reef (29.70 ind. h⁻¹ [SE 8.91]). Similarly, there was greater mean species richness at San Sebastian Reef (38.74 species h⁻¹ [SE 2.79]) than at Lighthouse Reef (25.37 species h⁻¹ [SE 3.66]). The main drivers of targeted fish abundance were habitat and depth, with shallow (<15 m) and mixed reef areas having the greatest abundance and richness. More sampling was done over sand habitat at Lighthouse Reef, which likely led to the lower abundance and species richness observed at this site; however, that finding could also be attributable to the fact that protection is provided to only a section of available coral reef habitat in a small area. Nevertheless, fish community structure was comparable between the sites, with similar proportions of carnivores (78–81%), herbivores (12–14%) and omnivores (7–8%). Our findings highlight the variation in species abundance and assemblages of coral-reef fish targeted by fishing in Mozambique and emphasise the importance of localised environmental variables as a driver of these patterns. To

ensure maximum protection of Lighthouse Reef fish communities, we recommend an extension of the no-take zone to include the entire reef complex.

Keywords : baited remote underwater video, Bazaruto Archipelago, biodiversity, fish community structure, fishing pressure, marine reserve, rarefaction curves, western Indian Ocean

Introduction

Fish harvested on coral reefs constitute one of the main sources of protein for communities in developing countries throughout the world (Teh et al. 2013). This is particularly evident in Africa where maritime countries derive the majority of their protein from fish (Belhabib et al. 2019). However, in many instances **short-term [check]** food security is prioritised over conservation efforts (McClanahan et al 2011), and this has been an important factor leading to the overexploitation of many stocks of coral-reef fishes in East Africa (e.g. Kaunda-Arara et al. 2003; Tuda and Wolff 2015; Samoily et al. 2017; Belhabib et al. 2019). Poor **compliance, resulting from inadequate [check]** enforcement, and a lack of baseline information, which is essential for describing long-term trends in abundance, often hamper the success of management strategies and conservation efforts (McClanahan 1999; MacNeil et al. 2015).

Marine reserves **[These are not synonymous; some marine reserves allow some exploitation, including the BANP.]** have been established as a means of management and conservation for species targeted **by fishing** throughout many regions of East Africa (e.g. McClanahan and Arthur 2001), with varying levels of success (McClanahan et al. 1999; McClanahan et al. 2007; Currie et al. 2012). In general, such areas prohibit extractive or disruptive activities and provide refuge for maintenance or recovery of marine populations (Ballantine and Langlois 2008). Although the cost of establishing a marine reserve is often lower than **that of** traditional fisheries management strategies (Roberts and Polunin 1991), effective implementation requires an understanding of traditional rights **and** international laws, as well as regional fishing practices (Wells et al. 2016). Where reserves have been managed successfully, benefits have included positive effects on biodiversity and ecosystem health over time (e.g. Wells et al. 2016), **with** an increase in species richness, biomass and density of exploited families of fishes, and protection of habitats for **[Associated with what? Do you mean flora and fauna that occur in the reserves?]** flora and fauna within the reserves (McClanahan and Kaunda-Arara 1996; McClanahan et al. 2007; Caveen et al. 2015). It is widely accepted that larger reserves can **result in increased** resilience of coral-reef fish communities to climate change, improve the spillover effect (movement of fishes beyond the boundary of the reserve **to** where they can be captured by fishers), and reduce the stress induced by the edge effect of the boundary on populations (Wilhelm et al. 2014). However, the placement of a marine reserve is also important, with the selection of high-quality habitat, **appropriate** water depths and increased distance to human populations and activities **all** thought to have a positive effect on reserve performance (Wells et al. 2016; Cinner et al. 2018).

There have been numerous studies of coral-reef fish communities within and outside marine reserves along the coast of eastern Africa (e.g. McClanahan and Arthur 2001). Most of these have focussed on a broad range of demersal fishes associated with coral reefs, without a specific focus on species that are important targets of reef fisheries (e.g. Maggs et al. 2010; Chabanet et al. 2016; Samoilyis et al. 2019, although see McClanahan and Jadot 2017). This lack of data on targeted species is concerning, given the importance of such data to baseline monitoring of trends in the status of fish communities. Furthermore, studies to date have generally used underwater visual census (UVC) techniques to survey reefs, which often preferentially record the smaller, site-attached species (Colton and Swearer 2010), rather than larger-bodied and often wider-ranging carnivores, many of which are targeted by fisheries. Better assessments of the status of these targeted species could be obtained with baited remote underwater video stations (BRUVS), which use baits to attract fishes into the field of view of a camera. This technique mostly samples these larger, omnivorous and carnivorous fishes and thus may provide insights into the abundance of the species preferentially targeted in reef fisheries (Watson et al. 2005; Harvey et al. 2007; Bernard et al. 2014). To date, fishery-independent surveys using BRUVS have been used to quantify fish abundance and biomass in temperate regions of eastern South Africa [If all these studies were conducted in South Africa, it is more precise to write 'eastern South Africa'.] (e.g. Bernard and Gotz 2012; De Vos et al. 2014; Roberson et al. 2015; Heyns-Veale et al. 2016). However, comparatively few surveys have been conducted further north along the tropical coastlines of Mozambique, Kenya, and Tanzania (e.g. Unsworth et al. 2015).

Here, we use BRUVS to describe the coastal fish communities of coral reefs in southern Mozambique. We focus on species targeted by fisheries and we sample both within and outside a marine reserve present along the coast. Our aim is to provide one of the first baseline assessments of the status of targeted fish species and an assessment of the possible role of marine reserves in the management of these resources. Specifically, we hypothesised that: a) targeted fish would be more abundant inside the marine reserve, where fishing practices are limited, compared to outside the marine reserve, which was open to fishing, and b) overall fish abundance and diversity would also be greater inside the marine reserve compared to outside the reserve. [You refer later to testing your hypotheses. You need to specify here what those hypotheses are. Did you hypothesise that fish abundance and species richness (both of targeted species and of all species) are higher inside a no-take area within the BANP than they are outside the BANP? But please see further queries below, under 'Study sites'. It is not clear whether you regard your Lighthouse Reef site as being in the no-take area or not.] The results of our study could be used to inform management and to infer current pressures in the region. In addition to focussing on targeted

species, we also sampled non-target species, which could be used to provide a more holistic overview of species diversity and relative abundance in the region.

Materials and methods

Mozambique has a history of research on reef fishes that dates back to the 1930s (Pereira 2000). Much of this work has focussed on assessments of biodiversity (e.g. Maggs et al. 2010), rather than the species targeted by fisheries. The country has more-developed management of marine resources than some other areas of East Africa, and there are a number of well-established marine reserves in Mozambique, such as the Ponto Do Ouro Partial Marine Reserve in the extreme south, and around the Bazaruto Archipelago. The archipelago, composed of five islands (Figure 1) has had protected status since 1971, although initially this only included three islands (Benguérua, Magaruque and Bangué) (Maggs et al. 2010) [reference?]. In 2001, the protected area was enlarged to include all five islands and the legislation was strengthened with the official creation of the Bazaruto Archipelago National Park (BANP) (Maggs et al. 2010) [reference?]. Bazaruto Archipelago National Park was not designed specifically to protect or strengthen resources for artisanal fisheries (Santos 2008), but rather to protect suitable habitat for threatened, vulnerable, and endangered marine megafauna such as dugongs *Dugong dugon* and sea turtles (green *Chelonia mydas*, hawksbill *Eretmochelys imbricata*, loggerhead *Caretta caretta*, olive ridley *Lepidochelys olivacea* and leatherback *Dermochelys coriacea*) (MITUR 2008; van der Elst and Afonso 2008).

Although the initial management goal of the BANP was not to promote the sustainability of fish stocks, in theory the protection offered by the implementation of the park and no-take areas within the park should have flow-on benefits that address some of the issues surrounding overfishing in the region (Russ 1991). However, there has been concern in recent years of a decline in marine resources within BANP including fish, which may have led to some of the impetus for re-zoning and increasing protection of no-take areas or 'Total Marine Protection Zones (ZPTM)' (Diaz et al. 2016). Furthermore, the most recent park management plan also discusses large species of cod / 'rockfish' that are afforded protection throughout all of the park (Diaz et al. 2016). [You refer to the 'initial' management goal. Please could you clarify whether the protection of fish stocks has since explicitly been added to the management goals of the BANP, or whether such protection is just an unintended benefit of protecting megafauna habitat. If it is still not an explicit goal to protect fish stocks, why have no-take areas been declared? This is not clear at present.]

Study sites

Two reef sites were sampled during August 2016; one on Bazaruto Island inside the BANP, and the other outside the BANP on the San Sebastian Peninsula (Figure 1). Lighthouse Reef (21°31'00" S, 35°30'14" E) is situated inside the BANP near the north-eastern tip of Bazaruto Island (Maggs et al. 2010). Here, habitats are mainly composed of hard coral (43%), bare reef, rubble and sand (32%), dead coral (9%) and soft coral (7%) (MH Schleyer and JQ Maggs, Oceanographic Research Institute, Durban, unpublished data). In general, the park aims to restrict fishing activities such as gillnetting and semi-industrial or industrial fishing. **[This is why it is important to clarify why the park aims to restrict certain types of fishing.]** Mozambican fishers who do not live inside the BANP are only allowed to use handlines when inside the park, and beach-seines are only permitted in a small zone in the south of Magaruque, the second biggest island in the BANP (Everett et al. 2008) **[Are there Mozambican fishers who do live inside the BANP? If so, what restrictions apply to them?]**. The latest estimate of the population of residents living within the BANP is > 5,000 people, who are the only people permitted to fish within a specific zone called 'Zones of Use Limited to Local Communities' (Diaz et al. 2016). Fishing is also allowed inside the BANP by non-residents **[Please clarify exactly what type of fishing is allowed. You have indicated already that gillnetting and (semi-)industrial fishing are not allowed, and that local fishers can use handlines and, in one location, beach-seines. What else is allowed? Artisanal and recreational linefishing?]**, although it is totally excluded inside no-take areas or 'Total Marine Protection Zones' (ZPTM), one of which was located at our sampling site at Lighthouse Reef. **[When you say 'adjacent to', this implies that your sampling site was not actually in the no-take area. Consistent with this, in Figure 1a, you show the deployment locations of the BRUVS as being outside the no-take area. In the Discussion, however, you imply that your site was in the no-take area. Although you go on to say, below, that the boundaries of the no-take area are unclear, you need to specify whether you considered the Lighthouse Reef site to have been in the no-take area or not. This is particularly important if you were testing the hypothesis that abundance and species richness are higher inside a no-take area within the BANP than they are outside the BANP.]** The only activity allowed inside the BANP no-take areas is recreational SCUBA diving and snorkelling. The Lighthouse Reef no-take area includes the main reef complex that is adjacent to the coastline, albeit only covering a section of the available coral reef habitat, which is < 3 km² (Diaz et al. 2016). In comparison, San Sebastian Reef (22°09'45" S, 35°33'76" E) is situated to the south of the archipelago **[Please clarify. Is San Sebastian Reef inside the Bazaruto Archipelago but outside the BANP?]** within the San Sebastian Coastal Reserve, which was created to afford protection to terrestrial and littoral habitats. The coastal waters beyond the immediate shoreline, including our site, were

not officially protected at the time of the study and fishing was permitted. [This reference to Lighthouse Reef does not seem to be strictly correct. The waters immediately offshore of the no-take area would not have had complete protection but, because they were in the BANP, there was some official protection (through the restrictions on fishing gear that you describe earlier).] These two reefs were selected for comparison because of similarities in habitat features such as depth (Lighthouse mean = 13.4 m and San Sebastian mean = 11.1 m) and benthic cover (predominantly coral reef and sand). However, it must be acknowledged that Lighthouse Reef within the BANP is situated on an island (Bazaruto), which at the southern end has a channel separating it with Benguerra island [What bay? Can you label it on Figure 1?], whereas San Sebastian Reef is adjacent to the coastline of the mainland, and as such is likely to experience different hydrological patterns and be characterised by differing geomorphology compared to the BANP (See Everett et al. 2008). [Is there a reference that describes the geomorphology?] [You refer in the Discussion to the 'isolation' of San Sebastian Reef, although it is not clear what is meant by the term. You need to describe that here.].

Data collection and video interrogation

Data were collected over a two-week period from 12 to 20 August 2016 [Table S2 shows 12–20 August.]. At each site, BRUVS were used to sample reef fish communities (Supplementary Figure S1). This sampling method uses bait to attract fishes, which makes the technique particularly suited for sampling predatory species that are often targeted by fisheries (Langlois et al. 2012; Goetze et al. 2015), without precluding herbivorous species (Harvey et al. 2007). Our equipment consisted of high-definition video cameras (GoPro Hero4 Silver) in underwater housings, secured in an aluminium frame with a bait bag placed in front of the cameras to attract fish (Supplementary Figure S1). The bait was composed of approximately 1 kg of crushed mackerel (Scombridae) or sardine (Clupeidae). The use of more than one type of bait was not considered to be problematic, as both types had similar properties in terms of oil content, and studies comparing less-similar baits found no differences in the assemblage of fishes attracted to BRUVS (Dorman et al. 2012). At each site, a set of six BRUVS were deployed, separated by a minimum of 500 m and left to record for approximately 60 min on the seafloor (between the hours of 07:00 and 16:00 to avoid potential changes in behaviour of fishes under conditions of low light [Myers et al. 2016]). A total of 33 BRUVS (five x set of six deployments) were deployed at Lighthouse Reef and 48 (eight x set of six deployments) at San Sebastian Reef during the study (Table S1). The dataset from Bazaruto Island was a subset from a larger dataset, which was reduced to focus on the area in and around the no-take area at Lighthouse Reef (Full set of deployments at <https://globalarchive.org/geodata/data/campaign/get/590>). [Please clarify how you get from

a set of 6 BRUVS to a total of 30 BRUVS. Did you have five deployments of the set of six? Also, you have not cited Supplementary Table S2 in the text. Presumably it should be cited here, in which case it would become Table S1 and the current Table S1 would become Table S2.]

Where possible, deployments were made near randomly generated points created using a random point generator in ArcGIS® software by Esri. This was not always logistically possible, and in instances where it was impractical or hazardous, BRUVS were deployed on reef patches or contiguous reef as near as possible to the randomly generated points. This resulted in uneven sampling among different habitat types, with more deployments in sand habitat at Lighthouse Reef compared to San Sebastian Reef (30% and 10% respectively) and more deployments over reef habitat at San Sebastian compared to Lighthouse Reef (60% and 39% respectively). EventMeasure software (SeaGIS™) was used to identify and calculate the abundance of each fish species observed across the 81 samples. All fishes recorded on video were identified to species level where possible. The maximum number of individuals for each species viewed at any one time (MaxN; Priede et al. 1994) was recorded from viewing the footage and used as a measure of relative abundance averaged over a sixty-minute period, expressed as individuals per hour [check].

Data processing

In order to compare functional groups between the two sites, fishes were assigned to one of eight categories (herbivores, omnivores, browsers of sessile invertebrates, diurnal carnivores, nocturnal carnivores, piscivores, diurnal planktivores and nocturnal planktivores; see Supplementary Table S1) based on classifications used in previous studies (Hiatt and Strasberg 1960; Hobson 1974; Harmelin-Vivien 1979; Myers 1999) and completed using information on functional groups [check] from Vivien (1973), Parrish and Boland (2004), Rajasuriya (2013), Tickler (2015) and Fordyce (2016). In order to compare functional-group structure with other studies in the western Indian Ocean (WIO), the number of categories was reduced to three (herbivores, omnivores and carnivores). For these broader groupings, all categories except herbivores and omnivores were pooled into the carnivore group. Fishes were assigned to the target-fish group (Supplementary Table S1) based on information contained in Balidy et al. (2007), Claquin (2008), Doherty et al. (1950), Everett et al. (2008), and FAO (2007). [check edit]. [Why are these two references in a separate sentence? Why not just add them to the previous sentence?] Images of the habitat present in each BRUVS deployment were used to categorise deployments into one of three categories: 'reef', 'sand', or 'mixed' (both reef and sand) based on which contributed the highest percentage. When it was difficult to assign a representative habitat (i.e. where the distribution was close to

50/50), it was designated as 'mixed'. **Determination** of habitat category was done by the same observer, to limit **bias**.

Fish abundance modelling

A multi-model approach using generalised linear models (GLM) with a negative binomial distribution was initially adopted to test our hypothesis **[singular or plural?]** relating to targeted fish abundance inside and outside of the BANP. **[See comment at end of Introduction. You have not stated there what your hypotheses are, but you need to. However, as I understand it, you are not comparing simply 'inside the BANP' with 'outside the BANP'; instead you are comparing 'inside a no-take area within the BANP' with 'outside the BANP'. But then that raises the question of whether your Lighthouse Reef sampling site was inside or outside the no-take area.]** Negative binomial GLMs were chosen over Poisson GLMs because of poor deviance explained (22.12 DE) and large overdispersion (43.01). Negative binomial GLM analyses **were conducted using** the 'glm.nb' function **in** the R package 'MASS' (Venables and Ripley 2002). To address any differences in deployment **duration** of BRUVS (range 57–87 mins), 'soak time' was used as an offset within the model structure. Four data points **representing** targeted fish exceeding 300 individuals per deployment **[Is 'drop' synonymous with 'deployment'? Use consistent terminology.]** were removed from the model dataset to avoid influencing model assumptions and fit (Zuur et al. 2009). Four deployments were also removed from the San Sebastian dataset due to these deployments landing in comparatively rare rocky habitat, and a further three deployments were removed from Lighthouse Reef for the same reason. Five models were fitted, including an intercept-only model, as well as single-term models with 'site', 'depth', **or** 'habitat', and a saturated model including all three variables. Models were then ranked using Akaike's information criterion weights corrected for small sample sizes (wAICc) (Burnham and Anderson 2002) and the model with the highest wAICc value was selected as the 'best model'. Variance partitioning was used to assess the effects of independent explanatory variables within the model structure. This procedure used the 'hier.part' package in R (Walsh et al. 2013).

Differences in mean relative abundance of targeted species of fish between Lighthouse Reef and San Sebastian Reef, and among differing habitats, were estimated using partial residuals plots from **the** GLMs using the 'Visreg' package in R (Breheny and Burchett 2017**[not in References]**). Mean relative abundance was also calculated for all fish species found at each site. **[See query in 'Study sites' section regarding whether San Sebastian Reef is in the Bazaruto Archipelago. Are you saying in this final sentence that a mean relative**

abundance per species was calculated for Lighthouse and San Sebastian reefs combined?]

Analyses of fish community structure

Species rarefaction curves recording the rate at which new species (y) are added with extended sampling effort (x) were used to compare the richness of fish species between sites (Thompson and Withers 2003). Curves that asymptote provide support that all species have been sampled, whereas curves that fail to asymptote suggest that an incomplete set of species have been observed.

Prior to **conducting** multivariate analyses, relative abundance was transformed using a common logarithm (+1) to create Bray-Curtis similarity matrices. A logarithmic transformation was chosen **because** it ensured that contributions of uncommon genera were accounted for, while reducing the importance of dominant genera in the analysis (Clarke and Warwick 2001[**not in References**]). All statistical tests used the software R (R Core Team 2019).[**Use of R itself should be cited as 'R Core Team (year)'. The Pinheiro reference is for the package 'nmlc', specifically.**]. To explore patterns in the community structure of fish at the level of individual deployments of BRUVS across all habitats sampled at Lighthouse Reef and San Sebastian Reef, the Bray-Curtis distance matrix derived from transformed values was subjected to hierarchical agglomerative clustering with group average linking (function 'hclust' from the package 'stats') and an associated similarity profiles test (function 'simprof' from the package 'clustsig'; Clarke et al. 2008). Similarity profiles were used to identify whether BRUVS from particular sites and habitats clustered together, and **hence** did not differ significantly in their species composition. This analysis was a permutation test that determined whether any significant group structure existed, within the samples, for which there was no *a priori* grouping hypothesis (Clarke et al. 2008). In addition to cluster analysis, a principal components analysis (PCA), using the function 'pco' in the package 'ecodist' (Goslee and Urban 2007), was used to examine differences in community structure between sites and habitats. The 'vf' function in this package was used to find the maximum correlation of the individual variables with the configuration of BRUVS deployments in ordination space.

Results

Species rarefaction curves

~~Species rarefaction curves for each location approached asymptotes, indicating that a large portion of available species were sampled by our study (Figure 2).~~ [**Deleted sentence duplicates final sentence of paragraph**] Lighthouse Reef and San Sebastian Reef

displayed similar curves, although there was a slight difference in the total number of species recorded (Figure 2). Both sites approached an asymptote, suggesting that, although some latent biodiversity was not described, the majority of fishes detectable by BRUVS deployments were recorded.

Species richness

We recorded a total of 16 412 fishes, comprising 220 species and 49 families. Seven families represented more than half of the total number of species observed (54%). These included 34 species in the Labridae, 21 Chaetodontidae, 18 Acanthuridae and 14 Serranidae and 11 species in each of the Carangidae, Lutjanidae and Scaridae (Supplementary Table S2). Mean species richness ~~per hour~~ at Lighthouse Reef was 25.81 species h⁻¹ (SE 3.44) and at San Sebastian Reef was 38.74 species h⁻¹ (SE 2.79). Overall, San Sebastian Reef had higher numbers of species than Lighthouse Reef (Supplementary Table S2) **[This table does not show genera or families.]** When combining overall species richness across both sites, the greatest richness occurred over reef habitat ($n = 213$), followed by mixed habitat ($n = 171$), and then sand habitat ($n = 81$).

Functional group structure

[This paragraph duplicates all the information, including the percentages, contained in Figure 3. Repetition should be avoided, so please reduce the paragraph to summary statements only. However, please check that the percentages shown in Figure 3 are correct, because there are numerous discrepancies between the figure and the text, as shown below.] Using the most basic grouping of functional categories, 80% of the fish community at Lighthouse Reef was composed of carnivores, 7% **[Figure 3 shows 7%]** of omnivores, and 13% **[13% in Figure 3]** herbivores (Figure 3). Community composition was similar at San Sebastian Reef, consisting of 81% carnivores, 8% **[8% in Figure 3]** omnivores, and 11% **[11% in Figure 3]** herbivores (Figure 3). **[Some disagreement in numbers with Supplementary Table S3, too.]** Separation of fishes into finer-scale categories revealed that, at both sites, the largest number of species recorded by the BRUVS were diurnal carnivores, which were mostly labrids. Browsers of sessile invertebrates (Chaetodontidae), nocturnal carnivores (e.g. Lutjanidae) and herbivores (Acanthuridae and Scaridae) were also in comparable proportions at each site. In addition, the functional groups containing the fewest species at each site were similar (i.e. Omnivores such as Pomacentridae, piscivores such as Serranidae and Carcharhinidae, diurnal planktivores such as Caesionidae, and nocturnal planktivores such as Pempheridae).

Fish abundance in relation to habitat characteristics

The effect of habitat variables on relative abundance of target fishes was examined using negative binomial generalised linear models (GLMs). The model that provided the top-ranked fit (wAICc = 0.97) and explained the greatest amount of deviance (28.5% DE) for relative fish abundance included the explanatory variables habitat, depth, and site (Table 1). Variance partitioning of explanatory variables indicated that the 59% of the variance was explained by depth, which was followed by habitat at 26.5%, and then site at 14.4% (Supplementary Figure S2 [Figures and tables should be introduced in numerical order. This should be renumbered as Figure S2]).

The mean relative abundance of all target species was 2.7 times greater at San Sebastian Reef (80.22 ind. h⁻¹ [SE 18.00]) compared to Lighthouse Reef (29.70 ind. h⁻¹ [SE 8.91]), and a similar pattern was observed for all species combined (target and non-target) (Lighthouse Reef 95.10 ind. h⁻¹ [SE 24.21]; San Sebastian Reef 219.12 ind. h⁻¹ [SE 33.12]). When accounting for habitat and depth effects on the relative abundance of targeted fish at each site, using the top-ranked model, there were still more than double the number of targeted fish at San Sebastian Reef compared to Lighthouse Reef (Figure 4a). Targeted fishes occurred in greater relative abundance in mixed and reef habitats, according to the model, a pattern that was consistent both at Lighthouse and San Sebastian reefs (Figure 4b). The latter reef consistently had greater relative abundance of target fishes across all habitats and depths (Figure 4b [This should be '4b, c'. Figure 4c does not include habitat.]).

The most common targeted species belonged to the carnivorous functional group and all occurred in higher relative abundance at San Sebastian Reef. These included *Aprion virescens*, *Lutjanus gibbus*, *Thalassoma hebraicum* and *T. lunare* (Figure 5a-d). In contrast, two of the most commonly targeted herbivores, *Acanthurus tennentii* and *Scarus rubroviolaceus* showed patterns of slightly increased relative abundance at Lighthouse Reef compared to San Sebastian (Figure 5e and f).

Fish community composition

The composition of fish communities showed some overlap between Lighthouse and San Sebastian reefs and varied among habitats (Figure 6; Supplementary Figure S3[Should be renumbered S3]). Communities at the two reefs had similar levels of complexity and diversity. Those at Lighthouse Reef were dominated by assemblages found in mixed habitats, whereas those at San Sebastian Reef were dominated by reef assemblages. There was considerable overlap in fish community composition between sites in reef and mixed habitats (Supplementary Figure S3[Should become S3]). The most complex and diverse communities

of fishes were found in reef habitats, followed by mixed habitats and then sand habitats. PCA ordination suggested that there were two groupings of fish communities; (i) reef and mixed habitats; and (ii) sand habitats (Figure 6).

Discussion

The findings of our study provide preliminary estimates of relative abundance and diversity of species targeted by fishing at sites and across multiple habitats in southern Mozambique using a fishery-independent sampling technique. Total and targeted species richness and relative abundance were both greater at San Sebastian Reef than at Lighthouse Reef, a difference that was largely driven by localised variation in depth and habitat. The availability of optimal habitats, particularly reef areas, was better represented at San Sebastian Reef than at Lighthouse Reef. **[Have you described anywhere in the Methods how depth and habitat differed at the two sites? This information seems to be missing.]** Previous studies in the region have also found that benthic habitat can have a strong effect on patterns in fish density and assemblage structure (e.g. Currie et al. 2012; Osuka et al. 2018), and this variable can explain large proportions of variation in models (30–65%) (Pinca et al. 2012; Russ et al. 2015, although see Hawkins et al. 2006). Higher numbers of fishes in reef habitats is likely a result of the numerous complex micro-habitats and the greater variety of food sources compared to those available in less-complex habitats (Gratwicke and Speight 2005). Reef size has also been shown to have a positive effect on the abundance and biomass of several species of predatory fish in protected areas of the WIO (Dames et al. 2019). Unfortunately, there were no detailed benthic habitat maps available for both of our sites and so we were unable to model the potential effects of reef size on relative abundance. However, we sampled reef structures that ran over a similar length of coastline at each site, albeit a slightly longer section at San Sebastian (Lighthouse Reef = 8.5 km and San Sebastian Reef = 10.5 km). **[Have you described/documentated how reef size differed between your two sides?]** Sandy habitats had the lowest community richness of fish in our study. **[Is this documented in the Results?]** This pattern of lower abundance in sandy habitats has also been noted by previous studies (e.g. Carpenter et al. 1981). In contrast, Samoilys et al. (2019) found no effect of habitat variables on fish assemblage in the WIO region; rather, geomorphology, exposure and fishing pressure drove patterns. Indeed, these factors probably contributed to the 14% of site variation in fish abundance in our models, after accounting for the influence of habitat type and depth.

Our findings provide some of the first estimates of targeted-reef-fish abundance for Mozambique using a fishery-independent assessment method, which is critical to ensure effective management of this resource. Our estimates of mean targeted-fish abundance varied

between 29.70 ind. h⁻¹ [SE 8.91] (Lighthouse Reef) and 80.22 ind. h⁻¹ [SE 18.00] (San Sebastian Reef). It is unknown how this range compares to other areas within the WIO region, although some estimates exist for commonly targeted species. For example, *A. virescens* is a known target, particularly for recreational fishers, and was found in greater average abundance in our study (1–1.7 ind. h⁻¹) than in other studies of protected areas in South Africa (0.6–0.9 ind. h⁻¹) (Dames et al. 2019). In fact, the common species in our study ranged between mean abundances of 1 and 4.5 ind. h⁻¹, which is high compared to estimates from other areas in the region where BRUVS were used [What method? BRUVS, or fishery-independent assessment in general?] to assess the abundance of targeted reef fish such as members of the families Lutjanidae, Lethrinidae, and Serranidae. The study by Dames et al. (2019) found that the mean abundances of common targeted fish were all <2.5 ind. h⁻¹ in protected zones of the iSimangaliso Wetland Marine Park in South Africa. Similarly, a study by Unsworth et al. (2015) in Palma Bay in northern Mozambique observed that species such as *Lethrinus variegatus*, *Lutjanus ehrenbergii* and *Lutjanus kasmira* ranged in average abundance between 0.4 and 0.66 ind. h⁻¹. Furthermore, Unsworth et al. (2015) recorded an average total fish abundance of 30 ind. h⁻¹, which was at the lower end of the range in our study (30–80 ind. h⁻¹). Although it was not possible to draw a strict comparison between results from our study and those from studies at these other sites (i.e. in northern Mozambique and northern South Africa), our estimates for common species of targeted fish in the same general region (southwest Indian Ocean) provide some confidence that protective measures in the BANP and the isolation of San Sebastian from high-density human populations [In what way is it isolated? You need to mention isolation in the description of study sites in Methods.] are likely to be providing benefits for some targeted species at a local scale.

Our study observed 189 species on BRUVS within the BANP at Lighthouse Reef, which was considerably more than was observed with UVC (103 species) by Maggs et al. (2010). Baited video surveys have recorded a greater proportion of predatory species when compared to UVC (Langlois et al. 2006; Goetze et al. 2015). Species accumulation curves from our study suggest that the majority of the species richness at Lighthouse Reef that is detectable by BRUVS was captured by our sampling. It is also important to note that sampling by Maggs et al. (2010) was conducted on small patches of reef at Lighthouse Reef, and hence their study would not have covered as great an area as our study. At the scale of the southwest Indian Ocean, our sampling produced fewer species overall (220) than did other surveys of fish communities in the region (Supplementary Table S3, Figure S4), most of which used underwater visual census on outer reef slopes. These studies found that species richness ranged between 225 and 334 species per location. Interpretation of these comparisons should be made with caution, however, as BRUVS preferentially sample fishes attracted to bait, rather

than the complete community (Harvey et al. 2007), and our study covered a different range of habitats and depths than the previous studies. ~~A previous study found differences in species richness between sites outside BANP (Twelve-mile Reef) and sites inside BANP in the no-take areas, but also among sites within the no-take areas (Maggs et al. 2010).~~ **[Text deleted because the information is contained in the next sentence.]** Contrary to our findings of lower species richness within the BANP at Lighthouse Reef compared to outside the BANP at San Sebastian, Maggs et al (2010) observed that species richness in sites outside the BANP was usually lower than sites inside the BANP **[It is this sentence, where you say 'contrary to our findings', that implies that you consider your Lighthouse Reef sampling site to be inside the no-take area.]** In addition, within the no-take areas, Maggs et al (2010) found that Two-Mile Reef had twice the number of species as the no-take area bordering Lighthouse Reef. Although we were unable to focus on fish communities at Two-Mile Reef in our study due to limited resources, abundance and community structure should be the focus of future work in this area.

Unequal proportions of the different categories of carnivores was one of the main differences in functional composition between the two sites in our study, with more nocturnal and diurnal carnivores and fewer browsers of sessile invertebrates at San Sebastian Reef than at Lighthouse Reef. This is important because diurnal and particularly nocturnal carnivores represent the majority of species targeted by fisheries in the region, in contrast to browsers of sessile invertebrates (Supplementary Table S1). Of the common species targeted by fisheries, carnivores were in higher relative abundance at San Sebastian Reef, while common herbivores were more abundant at Lighthouse Reef. This likely reflects differences in the availability of habitat at the two sites suitable for the differing foraging behaviour **[check]** and diets of these functional groups. **[I have suggested the addition of 'behaviour' to clarify why (I think) you have included both 'foraging' and 'diets'.]**

Fish community structure and diversity were comparable between sites although proportions of these groups differed in each. Cluster analysis identified that there was a large overlap between reef-fish assemblages and dominant clusters at San Sebastian Reef, which was due to the greater proportion of reef habitats sampled at San Sebastian than at Lighthouse Reef. **[Again, did you present this information in Methods, i.e. re proportion of habitats sampled?]** In contrast to our findings, previous work by Maggs et al. (2010) found that the no-take areas inside the BANP at Two-Mile Reef and Lighthouse Reef had greater proportions of target species of carnivores (74% and 66%, respectively) compared to Twelve-Mile Reef (63% carnivores) outside the BANP. The functional-group structure of fish communities in our study was comparable with locations across the southwest Indian Ocean that had been surveyed

using UVC (Supplementary Table S3), where carnivores comprised between 51 and 80% of the fish community.

One of the potential factors that might have led to fewer species, reduced abundance, and lower proportions of carnivores within Lighthouse Reef in the BANP compared to San Sebastian Reef could be lack of enforcement of reserve [This acronym would need to be defined. However, this is the only time that you refer to 'marine protected area'. Best to stick to terms that you have used previously.] regulations or differing human impacts at these two sites or in their vicinities (e.g. Cinner et al. 2018). Such impacts could include fishing effort, which might well be greater around Lighthouse Reef compared to San Sebastian Reef due to its close proximity to the coastal town of Vilanculos. Although the BANP was created in 1971, the first conservation and development plan was not implemented until 1989 [Development of the plan, or implementation of it?]. Furthermore, park boundaries and management were not originally well defined or structured. Park zone boundaries have undergone re-zoning through time (See MITUR 2002 and 2008; Diaz et al. 2016), which has helped to clarify the boundaries. [This reference is 12 years old, and hence cannot be referred to as 'today'. Also, in your description of your study site in the Methods, you say that the boundaries of the no-take area are not clear. Here you are saying that the boundaries of the BANP as a whole are not clear. Is that correct?]. Enforcement of park regulations was originally the responsibility of the warden (marine park ranger), whose role was limited to that of intermediary between tourism concessions and local populations (Claquin 2008). BANP management staff highlighted a need to improve the patrol level (MITUR 2002), which led to numerous private companies, NGOs, and institutions providing logistical support to the BANP in the form of a patrol boat and training (Diaz et al. 2016). [That was 18 years ago! No update? Even a personal communication?]

A nominal proportion of the variation in our model was attributed to other site-specific variables, which might include subtle localised habitat differences not adequately captured in our coarse habitat categories, or differing levels of protection from fishing practices. Enforcement and compliance with park regulations by fishers are common issues in marine reserves throughout the world (Bergseth et al. 2015). The performance of a marine reserve can be measured by how well it preserves density, biomass, size, and diversity of organisms as well as how it serves [check] human needs through time (Halpern 2003). Although we did not measure biomass or size of individual fish in our survey, lower relative abundance and diversity inside the park suggests that there is limited suitable habitat at Lighthouse Reef to support ['support?'] diverse fish communities. A few studies of marine reserves have found little or no difference in fish communities inside and outside of no-take areas (e.g. Advani et

al. 2015; Cox et al. 2017), although instances where fish are significantly lower in abundance and diversity inside no-take areas are rare (e.g. Gilby et al. 2017). As with our study, Gilby et al. (2017) found that increased abundance of fish outside of reserves was largely driven by inadequate coverage of optimal habitats, which is an issue relating to reserve placement. If rezoning or expansion of no-take areas was to occur in the future, we recommend **expansion** to the south of current boundaries of Lighthouse Reef to include all reef habitat in the northeast section of Bazaruto Island. **[Can you suggest what that depth might be? You refer in the Conclusions to the ‘main reef complex’. At what depth is that?]**

It is important to reiterate here that not all of **the** BANP is a no-take area; with less than 1% (1,100 ha) of overall park area encompassed by such areas (‘Total Marine Protection Zones’) (Diaz et al. 2016). In the remaining areas of the park, only specific fishing techniques are allowed in Multiple Use Zones (e.g. line fishing is permitted), and inhabitant rights have been established that allow residents within the park to harvest resources in specific zones (e.g. Zones of Use Limited to Local Communities (Diaz et al. 2016)). **[It is not clear what is meant by ‘inhabitant rights’. In the description of your study sites, you refer to Mozambican fishers who do not live inside the BANP, but you do not refer to inhabitants of the BANP.]**

Conclusions

Our findings provide a detailed assessment of fish communities at two sites along the southern Mozambique coastline and highlight that localised depth and habitat variables explained most of the differences in community abundance and diversity. The current suite of habitats encompassed by Lighthouse Reef within the BANP **might** not be appropriate for improving targeted-reef-fish stocks, if this becomes a park management priority. Furthermore, the Lighthouse Reef no-take area **[Which is where you sampled?]** currently provides protection **only** to fish in a small area on the northeast corner of Bazaruto Island, and does not include reef habitat to the south. **[I think it would be useful to include this information about the main reef complex in your description of the study site.]** The no-take zone here could be extended to include the entire reef complex to provide greater protection. In contrast, San Sebastian Reef, outside of **the** BANP, appears to have **a greater** reef area and has higher species richness and, most importantly, **a higher abundance of all fishes and of targeted fishes** than at Lighthouse Reef. Managers of the San Sebastian Coastal Reserve have considered extending **the** boundaries offshore onto this reef to create a no-take area. However, the implementation of a no-take area in **the** San Sebastian **reserve** would be even more difficult to manage and enforce **than in the BANP**, given the increased distance from the town of

Vilanculos [Distance from where? Presumably this is what you refer to earlier as 'isolation'.] and more difficult access [Do you mean 'more difficult access'?]. Ongoing monitoring of fish communities and increasing the area of suitable target-reef-fish habitat and depth with no-take status within the park could provide more positive conservation benefits in the long-term. Incorporation of results from previous studies in the region, together with information about local habitat conditions and pressures, would assist in improved management of the BANP. However, the small size of the no-take area at Lighthouse Reef and incomplete coverage of all available coral reef habitat in the immediate vicinity provides additional justification for rezoning of boundaries [Please see earlier query about whether it is just the boundaries of the no-take area that are not clear or whether it is the boundaries of the whole BANP. This is not clear at the moment.].

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Figure legends

Figure 1: Bazaruto Archipelago National Park (BANP) in southern Mozambique, showing the park boundary (solid black line) [Not dashed in figure] and positions of deployments of baited remote underwater video stations (white dots) at two sites: (a) Lighthouse Reef and (b) San Sebastian Reef. Lighthouse Reef is inside the BANP, in which fishing activities are restricted, whereas San Sebastian Reef is outside the BANP and is open to all fishing beyond the immediate shoreline. The boundary of the no-take area or 'Total Marine Protection Zone' at Lighthouse Reef (solid white line) extends offshore to include the main reef complex (See Diaz et al. 2016). All fishing activities are prohibited inside the no-take area. [Please check edits] [Source of images in (a) and (b)?]

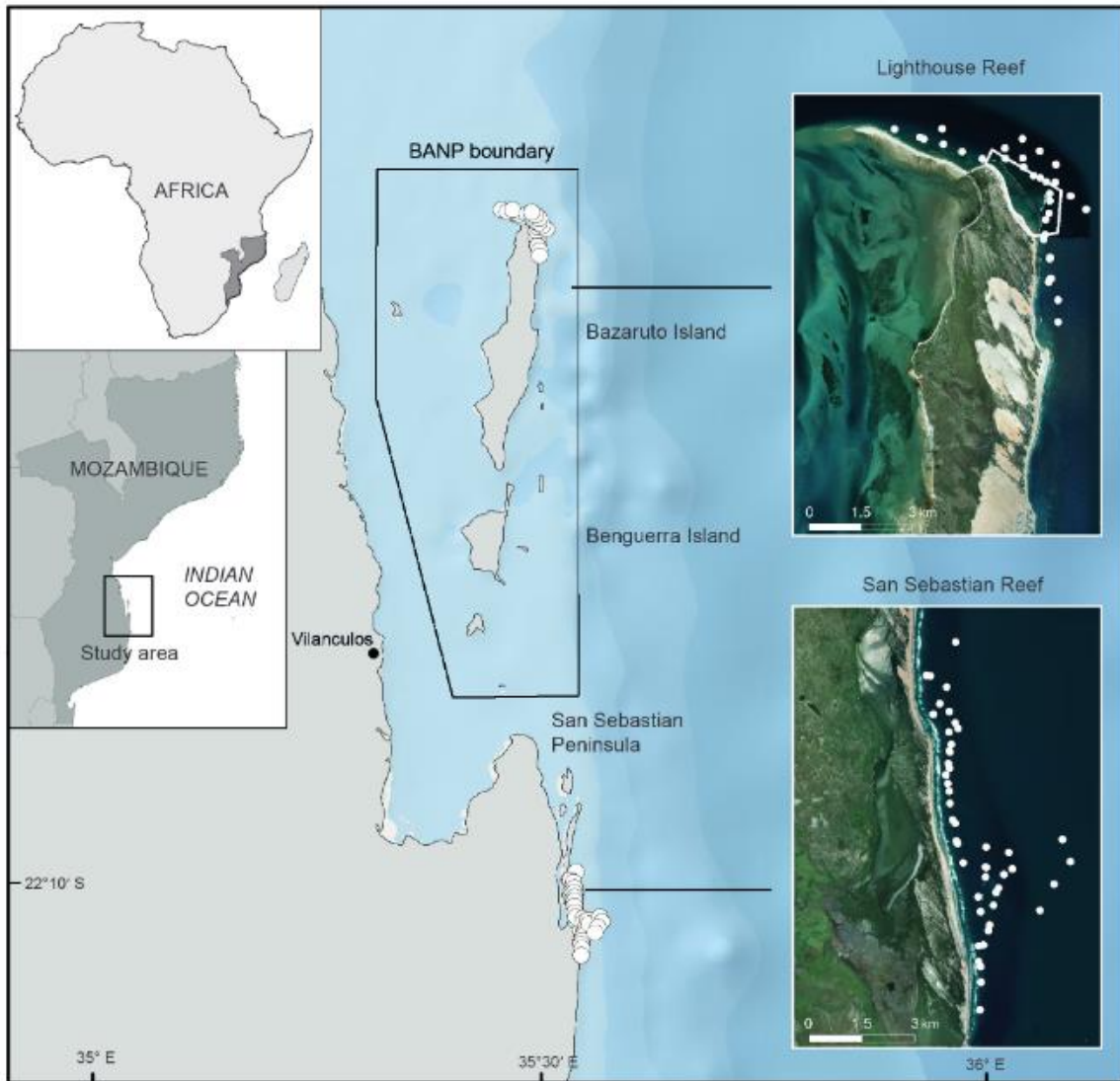


Figure 2: Species rarefaction curves for San Sebastian Reef and Lighthouse Reef, central Mozambique, in relation to the number of baited remote underwater video stations (BRUVs) deployed. Inset shows schematic of BRUVs (see also Supplementary Figure S1)

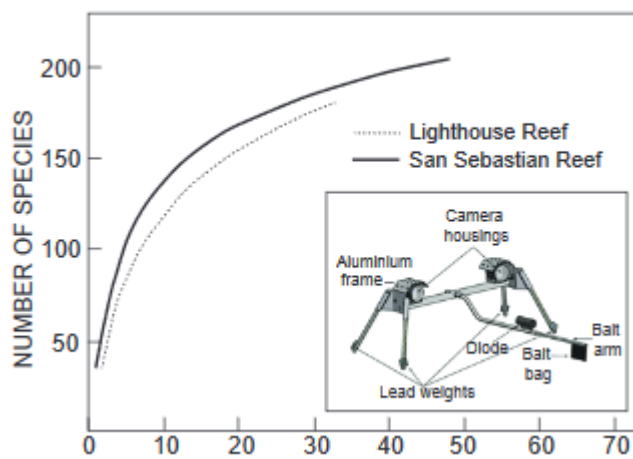


Figure 3: Overall functional structure of reef-fish communities at the two sites investigated using baited remote underwater video stations: (a) Lighthouse Reef and (b) San Sebastian Reef

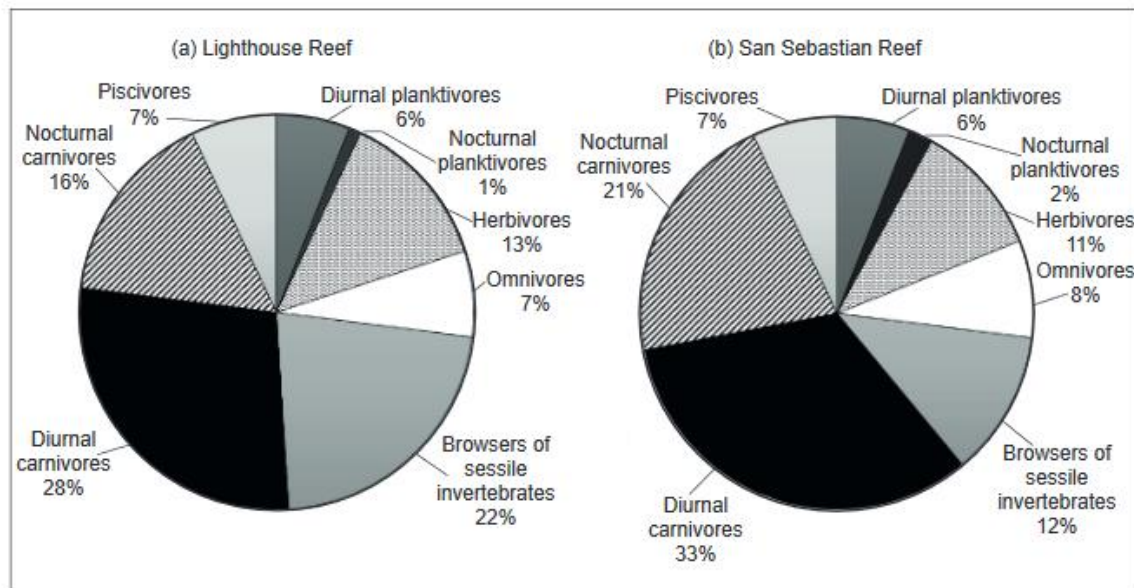


Figure 4: Partial residual plots relative to the independent variable (mean MaxN of targeted fishes [Does the y-axis have units? ind. h⁻¹?]) in the top-ranked negative binomial generalised linear model, which included: (a) site; (b) habitat type; and (c) depth. Bold line represents the fitted line (predicted MaxN values) and grey shaded areas represent the 95% confidence intervals [When you say the fitted line, do you mean the value of MaxN predicted by the model? Might that be more informative wording?]

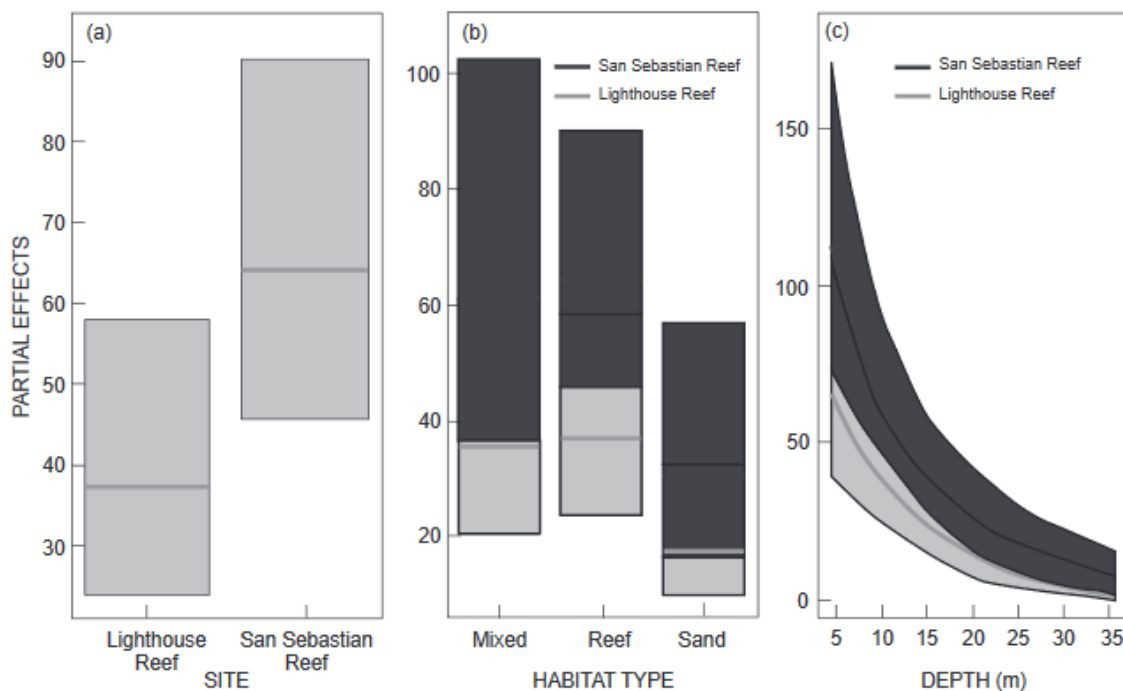


Figure 5: The six most common and abundant [Please explain the difference between 'common' and 'abundant'. Also, please clarify the name of the y-axis variable, and also the units. Would it be 'Mean MaxN (ind. h⁻¹)?'] target species observed on baited remote underwater video stations (BRUVS) in southern Mozambique. These were (a) *Aprion virescens*, (b) *Lutjanus gibbus*, (c) *Thalassoma hebraicum*, (d) *Thalassoma lunare*, (e) *Acanthurus tennentii* and (f) *Scarus rubroviolaceus*. These species occurred in >40% of all deployments combined [at both sites?]

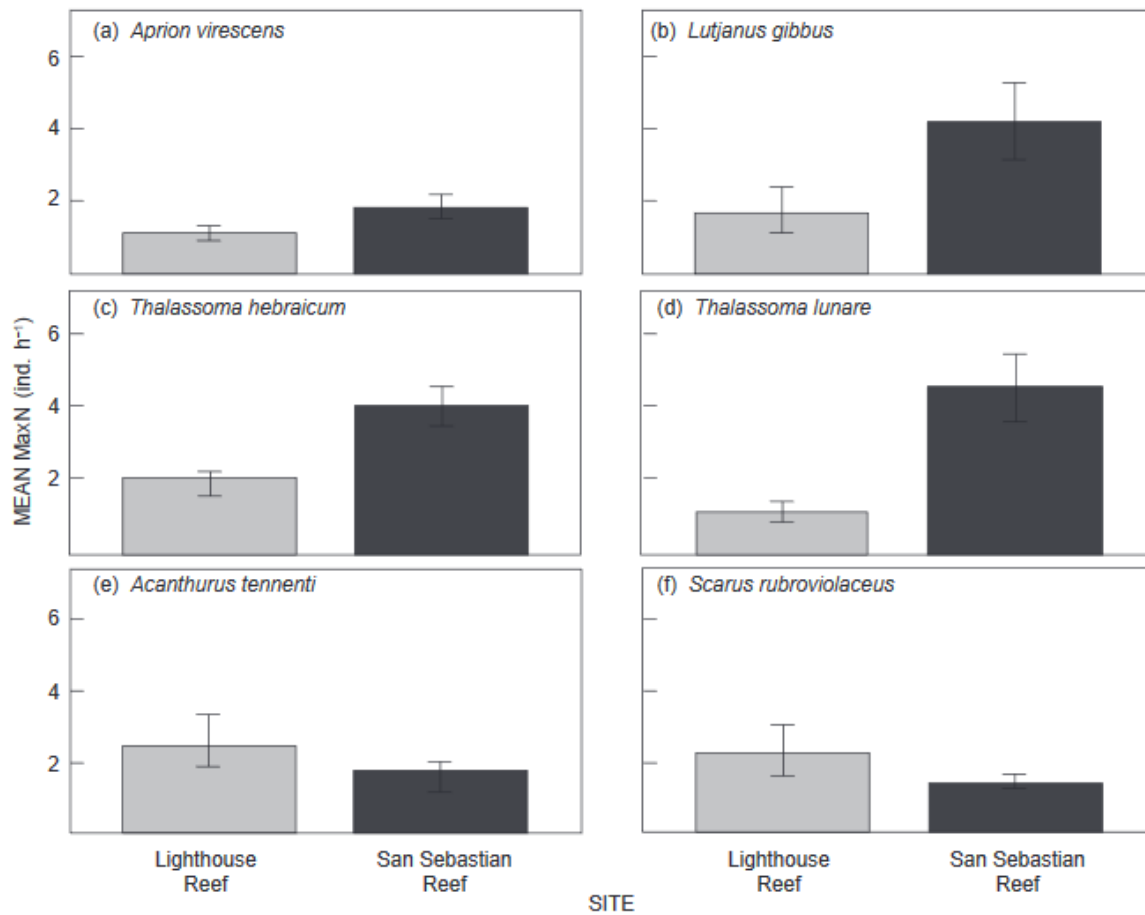


Figure 6: PCO ordination showing the fish community structure recorded at all baited remote underwater video stations, based on (a) site (San Sebastian and Lighthouse reefs), and (b) habitat

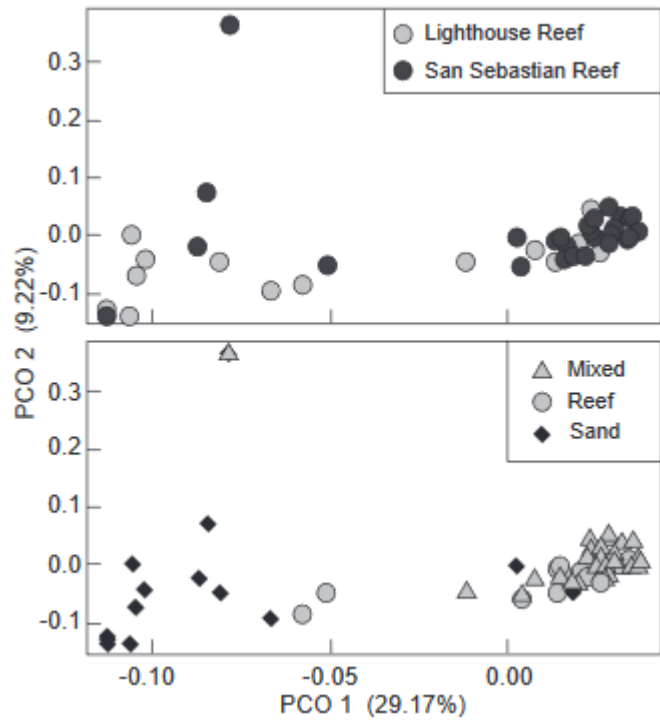


Table 1: Summary results of negative binomial generalised linear models used to test for effects of site **[This is called 'Site' in the text and the table. Should be consistent]**, habitat, and depth variables on the MaxN of targeted fish **recorded by** baited remote underwater video stations (BRUVS) in southern Mozambique. **The top-ranked model is highlighted in bold font** **[What is 'K?']**

Model	<i>K</i>	LogLik	AICc	Δ AICc	wAICc
Site + Habitat + Depth	5	-320.28	651.49	0	0.97
Depth	3	-326.2	658.76	7.27	0.03
Habitat	4	-331.12	670.86	19.38	0
Site	3	-333.31	672.99	21.5	0
Intercept	2	-334.92	674.02	22.53	0

Supplementary material

Table S1: List of all deployments of **baited remote underwater video stations at Lighthouse and San Sebastian reefs, southern Mozambique, during August 2016**

Deployment	Date	Latitude (°S)	Longitude (°E)	Depth(m)	Soak time	Reef
CMBAZ_037	12-08-2016	-21.543613	35.497866	9.1	64	Lighthouse
CMBAZ_038	12-08-2016	-21.537923	35.497562	8.9	64	Lighthouse
CMBAZ_039	12-08-2016	-21.531871	35.496314	9	62	Lighthouse
CMBAZ_040	12-08-2016	-21.526161	35.497324	11.4	61	Lighthouse
CMBAZ_041	12-08-2016	-21.520927	35.497491	8.7	63	Lighthouse
CMBAZ_042	12-08-2016	-21.5167	35.495038	12.2	61	Lighthouse
CMBAZ_043	12-08-2016	-21.513809	35.490626	15.7	63	Lighthouse
CMBAZ_044	12-08-2016	-21.511632	35.485889	9.4	65	Lighthouse
CMBAZ_045	12-08-2016	-21.511643	35.479827	4.2	65	Lighthouse
CMBAZ_046	12-08-2016	-21.509738	35.474485	5.1	67	Lighthouse
CMBAZ_047	12-08-2016	-21.50772	35.469592	4.3	67	Lighthouse
CMBAZ_048	12-08-2016	35.464442	-21.506327	5.8	67	Lighthouse
CMBAZ_049	12-08-2016	-21.503732	35.456768	5.8	69	Lighthouse
CMBAZ_050	12-08-2016	35.463431	-21.505907	5.9	66	Lighthouse
CMBAZ_051	12-08-2016	-21.503619	35.46931	21	66	Lighthouse
CMBAZ_052	12-08-2016	-21.508848	35.48576	25	60	Lighthouse
CMBAZ_053	12-08-2016	-21.511495	35.490411	14.6	70	Lighthouse
CMBAZ_054	12-08-2016	-21.51617	35.49327	13.7	57	Lighthouse
CMBAZ_055	12-08-2016	-21.543781	35.497515	9.8	60	Lighthouse
CMBAZ_056	12-08-2016	-21.538299	35.497525	9.2	62	Lighthouse
CMBAZ_057	12-08-2016	-21.533159	35.495939	7	63	Lighthouse
CMBAZ_058	12-08-2016	35.496427	-21.527914	5.7	62	Lighthouse
CMBAZ_059	12-08-2016	-21.522838	35.497534	8.7	62	Lighthouse
CMBAZ_060	12-08-2016	-21.517732	35.496564	10.1	62	Lighthouse
CMBAZ_061	13-08-2016	-21.525113	35.50705	30.8	67	Lighthouse
CMBAZ_062	13-08-2016	-21.521549	35.502966	27.9	67	Lighthouse
CMBAZ_063	13-08-2016	-21.517941	35.498976	15.3	64	Lighthouse
CMBAZ_064	13-08-2016	-21.51316	35.499162	24.6	69	Lighthouse
CMBAZ_065	13-08-2016	-21.50957	35.494884	35.7	69	Lighthouse
CMBAZ_066	13-08-2016	-21.506307	35.490353	33.1	72	Lighthouse
CMBAZ_067	13-08-2016	-21.544535	35.497224	9.8	84	Lighthouse
CMBAZ_068	13-08-2016	-21.54934	35.499673	14.1	86	Lighthouse
CMBAZ_069	13-08-2016	-21.555157	35.499679	11.8	87	Lighthouse
CMLHR_085	15-08-2016	-22.262801	35.534222	8.9	72	San Sebastian
CMLHR_086	15-08-2016	-22.264943	35.539756	13.2	68	San Sebastian
CMLHR_087	15-08-2016	-22.270402	35.538763	10.8	66	San Sebastian
CMLHR_088	15-08-2016	-22.275284	35.538027	8.4	64	San Sebastian
CMLHR_089	15-08-2016	-22.280338	35.537885	8.5	64	San Sebastian
CMLHR_090	15-08-2016	-22.285354	35.538455	8.5	63	San Sebastian

CMLHR_091	15-08-2016	-22.290424	35.539797	10.3	61	San Sebastian
CMLHR_092	15-08-2016	-22.295514	35.540386	11.3	64	San Sebastian
CMLHR_093	15-08-2016	-22.300414	35.541788	10.4	64	San Sebastian
CMLHR_094	15-08-2016	-22.301438	35.54745	9.9	63	San Sebastian
CMLHR_095	15-08-2016	-22.303363	35.552102	11.8	64	San Sebastian
CMLHR_096	15-08-2016	-22.307897	35.55012	6.7	67	San Sebastian
CMLHR_097	15-08-2016	-22.326697	35.546211	7.2	62	San Sebastian
CMLHR_098	15-08-2016	-22.321433	35.54566	6.7	62	San Sebastian
CMLHR_099	15-08-2016	-22.317545	35.548227	9.4	64	San Sebastian
CMLHR_100	15-08-2016	-22.312746	35.546194	8.8	67	San Sebastian
CMLHR_101	15-08-2016	-22.309045	35.545577	9.8	66	San Sebastian
CMLHR_102	15-08-2016	-22.303983	35.547463	9.5	66	San Sebastian
CMLHR_103	15-08-2016	-22.244527	35.539859	17.1	66	San Sebastian
CMLHR_104	15-08-2016	-22.252933	35.532477	10.5	67	San Sebastian
CMLHR_105	15-08-2016	-22.260094	35.535833	11.1	66	San Sebastian
CMLHR_106	15-08-2016	-22.266386	35.540508	15.2	63	San Sebastian
CMLHR_107	15-08-2016	-22.272334	35.538457	11.2	60	San Sebastian
CMLHR_108	15-08-2016	-22.278166	35.537386	8.6	60	San Sebastian
CMLHR_122	19-08-2016	-22.253062	35.533501	8.8	65	San Sebastian
CMLHR_123	19-08-2016	-22.255887	35.537599	11.3	65	San Sebastian
CMLHR_124	19-08-2016	-22.262139	35.53829	10.4	66	San Sebastian
CMLHR_125	19-08-2016	-22.267276	35.538181	9.7	66	San Sebastian
CMLHR_126	19-08-2016	-22.272065	35.53831	9.1	61	San Sebastian
CMLHR_127	19-08-2016	-22.276431	35.53837	10.3	66	San Sebastian
CMLHR_128	19-08-2016	-22.282278	35.538283	7.4	60	San Sebastian
CMLHR_129	19-08-2016	-22.289479	35.539114	8.6	61	San Sebastian
CMLHR_130	19-08-2016	-22.294898	35.540105	10.3	62	San Sebastian
CMLHR_131	19-08-2016	-22.296379	35.547619	10.4	62	San Sebastian
CMLHR_132	19-08-2016	-22.301537	35.547651	6.8	62	San Sebastian
CMLHR_133	19-08-2016	35.550743	-22.306796	8.2	58	San Sebastian
CMLHR_134	19-08-2016	35.549456	-22.311136	7.3	60	San Sebastian
CMLHR_135	19-08-2016	35.548494	-22.316135	9.4	60	San Sebastian
CMLHR_136	19-08-2016	-22.321193	35.546931	10.7	61	San Sebastian
CMLHR_137	19-08-2016	-22.325398	35.545687	4.8	60	San Sebastian
CMLHR_138	19-08-2016	-22.330622	35.546321	7.2	61	San Sebastian
CMLHR_139	19-08-2016	-22.337616	35.546014	7.6	62	San Sebastian
CMLHR_142	20-08-2016	-22.29778	35.55336	15	82	San Sebastian
CMLHR_143	20-08-2016	-22.301957	35.554236	17.8	74	San Sebastian
CMLHR_144	20-08-2016	-22.294463	35.566941	25	72	San Sebastian
CMLHR_145	20-08-2016	-22.300117	35.568934	28.5	71	San Sebastian
CMLHR_146	20-08-2016	35.564765	-22.305804	22.4	69	San Sebastian
CMLHR_147	20-08-2016	-22.31246	35.561234	21.9	63	San Sebastian

Table S2: Functional categories, **targeting status** (T = targeted) and relative abundance of **fish species observed** using baited remote underwater video stations at Lighthouse and San Sebastian reefs, **southern Mozambique**. Functional categories: H = herbivore; O = omnivore; BSI = browser of sessile invertebrates; DC = diurnal carnivore; NC = nocturnal carnivore; PI = piscivore; DP = diurnal planktivore; and NP = nocturnal planktivore

[Please check spelling of Bennet/Bennett. Not consistent. With regard to when to use parenthesis for a particular naming authority and date, please see ‘Species nomenclature’ in the journal’s latest Instructions to Authors. This has not been applied correctly here. However, in a non-taxonomic paper, the authorities are not necessary, and hence a much simpler option would simply be to delete all of them. It would have the advantage of shortening the table considerably.]

Taxon	Functional category	Targeting status	Relative abundance			
			Lighthouse Reef mean	Lighthouse Reef SE	San Sebastian Reef mean	San Sebastian Reef SE
ACANTHURIDAE						
<i>Acanthurus dussumieri</i>	H		1.67	0.82	3.66	0.54
<i>Acanthurus leucocheilus</i>	H		1.83	1.24	1.59	0.51
<i>Acanthurus leucosternon</i>	H		0.07	0.07	0.57	0.21
<i>Acanthurus mata</i>	DP		1.6	0.82	2.05	0.72
<i>Acanthurus tennentii</i>	H	T	2.43	0.73	1.68	0.42
<i>Acanthurus thompsoni</i>	H		0.43	0.21	0.07	0.04
<i>Acanthurus triostegus</i>	H		1.77	1.2	0.07	0.07
<i>Acanthurus xanthopterus</i>	DP		0.17	0.14	1.02	0.43
<i>Ctenochaetus striatus</i>	H		5.13	1.1	4.41	0.86
<i>Ctenochaetus truncatus</i>	H		0	0	0.02	0.02
<i>Naso brachycentron</i>	H		0.43	0.25	0.34	0.11
<i>Naso brevirostris</i>	H		0.5	0.23	1.39	0.61
<i>Naso elegans</i>	H		0.17	0.08	0.77	0.22
<i>Naso hexacanthus</i>	H		0.13	0.09	0	0
<i>Naso tonganus</i>	H		0.07	0.05	0.02	0.02
<i>Naso unicornis</i>	H		1.97	0.7	1.64	0.6
<i>Zebrasoma scopas</i>	H		0.1	0.06	0	0
ANTENNARIIDAE						
<i>Antennarius nummifer</i>	PI		0.03	0.03	0	0
APOGONIDAE						
<i>Apogon aureus</i>	DC		0.23	0.23	2.91	2.91
AULOSTOMIDAE						
<i>Aulostomus chinensis</i>	PI		0.03	0.03	0.02	0.02
BALISTIDAE						
<i>Abalistes stellatus</i>	DC		0.03	0.03	0	0
<i>Balistapus undulatus</i>	DC		0.1	0.06	0.41	0.11
<i>Balistoides conspicillum</i>	DC		0.07	0.07	0.34	0.09
<i>Balistoides viridescens</i>	DC		0.3	0.15	0.3	0.08
<i>Melichthys indicus</i>	O		0.2	0.09	0.07	0.05
<i>Odonus niger</i>	DP		1.77	1.43	3.11	1.86
<i>Pseudobalistes flavimarginatus</i>	DC		0.07	0.05	0.18	0.07

<i>Pseudobalistes fuscus</i>	DC		0	0	0.05	0.05
<i>Sufflamen chrysopterus</i>	DC		1.37	0.26	1.41	0.14
<i>Sufflamen fraenatum</i>	DC		0.33	0.12	0.84	0.16
BLENNIIDAE						
<i>Ecsenius midas</i>	H		0	0	0.02	0.02
<i>Ecsenius nalolo</i>	H		0.13	0.1	0	0
<i>Plagiotremus rhinorhynchus</i>	NP		0	0	0.02	0.02
<i>Plagiotremus tapeinosoma</i>	O		0	0	0.16	0.07
CAESIONIDAE						
<i>Caesio caeruleaurea</i>	DP		6.73	6.73	2.34	2.08
<i>Caesio lunaris</i>	DP		2.87	2.21	10.18	4.73
<i>Caesio striata</i>	DP		0.67	0.67	4.95	2.2
<i>Caesio xanthonota</i>	DP		3.17	3.1	3.41	1.82
<i>Pterocaesio marri</i>	DP		0.07	0.07	0	0
CARANGIDAE						
<i>Carangoides ferdau</i>	DC	T	0.03	0.03	0.02	0.02
<i>Carangoides fulvoguttatus</i>	DC	T	0.03	0.03	0	0
<i>Carangoides gymnostethus</i>	DC	T	0.93	0.32	1.52	1.45
<i>Caranx heberi</i>	DC	T	0	0	1.91	1.23
<i>Caranx melampygus</i>	DC	T	0.5	0.23	1.45	0.4
<i>Caranx sexfasciatus</i>	PI	T	0.1	0.1	0	0
<i>Decapterus macarellus</i>	DP	T	0.1	0.1	0.05	0.03
<i>Gnathanodon speciosus</i>	DC	T	0	0	0.45	0.45
<i>Scomberoides lysan</i>	PI	T	0.13	0.06	0.02	0.02
<i>Seriolina nigrofasciata</i>	DC	T	0	0	0.09	0.07
CARCHARHINIDAE						
<i>Carcharhinus amblyrhynchus</i>	PI	T	0.23	0.09	0.14	0.06
<i>Carcharhinus leucas</i>	PI	T	0.03	0.03	0.07	0.04
<i>Carcharhinus melanopterus</i>	PI	T	0.07	0.05	0	0
<i>Triaenodon obesus</i>	DC		0	0	0.16	0.06
CHAETODONTIDAE						
<i>Chaetodon auriga</i>	BSI		0.87	0.18	1.05	0.15
<i>Chaetodon bennetti</i>	BSI		0.07	0.07	0	0
<i>Chaetodon blackburnii</i>	BSI		0.13	0.09	0.18	0.07
<i>Chaetodon falcula</i>	BSI		0.13	0.09	0.02	0.02
<i>Chaetodon guttatissimus</i>	BSI		0.43	0.17	0.98	0.18
<i>Chaetodon interruptus</i>	BSI		0	0	0.36	0.13
<i>Chaetodon kleinii</i>	BSI		0.87	0.26	1.95	0.33
<i>Chaetodon lineolatus</i>	BSI		0.07	0.07	0	0
<i>Chaetodon lunula</i>	BSI		0.17	0.08	0.82	0.19
<i>Chaetodon madagaskariensis</i>	BSI		0.73	0.22	1.14	0.14
<i>Chaetodon melannotus</i>	BSI		0.03	0.03	0	0
<i>Chaetodon meyeri</i>	BSI		0.2	0.09	0.41	0.11
<i>Chaetodon oxycephalus</i>	BSI		0	0	0.02	0.02
<i>Chaetodon trifascialis</i>	BSI		0.23	0.1	0.07	0.04
<i>Chaetodon trifasciatus</i>	BSI		0.33	0.14	0.09	0.06
<i>Chaetodon vagabundus</i>	BSI		0.23	0.11	0.75	0.15
<i>Chaetodon xanthocephalus</i>	BSI		0.03	0.03	0	0
<i>Chaetodon zanzibarensis</i>	BSI		0.07	0.07	0.02	0.02
<i>Forcipiger flavissimus</i>	BSI		0.1	0.07	0.57	0.15
<i>Hemitaenichthys zoster</i>	DP		0	0	0.07	0.05
<i>Heniochus acuminatus</i>	BSI		0.9	0.29	0.48	0.15
<i>Heniochus monoceros</i>	BSI		0	0	0.14	0.07
CIRRHITIDAE						
<i>Cirrhitichthys oxycephalus</i>	DC		0.17	0.11	0.48	0.13

<i>Paracirrhites arcatus</i>	DC		0.1	0.07	0.16	0.06
<i>Paracirrhites forsteri</i>	DC		0.07	0.07	0.09	0.04
DASYATIDAE						
<i>Himantura fai</i>	NC	T	0	0	0.05	0.03
<i>Himantura jenkinsii</i>	NC	T	0.03	0.03	0	0
<i>Himantura uarnak</i>	NC	T	0	0	0.02	0.02
DIODONTIDAE						
<i>Diodon hystrix</i>	NC		0	0	0.11	0.05
ECHENEIDAE						
<i>Echeneis naucrates</i>	NC		0.27	0.08	0.45	0.11
EPHIPPIDAE						
<i>Platax orbicularis</i>	O	T	0.1	0.06	0	0
<i>Tripteron orbis</i>	DC	T	0	0	0.09	0.09
FISTULARIIDAE						
<i>Fistularia commersonii</i>	PI		0.17	0.07	0.07	0.04
HAEMULIDAE						
<i>Plectorhinchus flavomaculatus</i>	NC	T	0.47	0.18	0.57	0.11
<i>Plectorhinchus gaterinus</i>	NC	T	0.07	0.07	0	0
<i>Plectorhinchus plagiodesmus</i>	NC	T	0.17	0.14	0.07	0.05
<i>Plectorhinchus playfairi</i>	DC	T	0.27	0.17	0.52	0.11
<i>Plectorhinchus schotaf</i>	NC	T				
HOLOCENTRIDAE						
<i>Myripristis murdjan</i>	NP		0.17	0.17	0.05	0.05
<i>Sargocentron caudimaculatum</i>	NC		0.2	0.1	0.05	0.03
<i>Sargocentron diadema</i>	NC		0.03	0.03	0.41	0.21
KYPHOSIDAE						
<i>Kyphosus cinerascens</i>	H		0.83	0.83	0.66	0.27
LABRIDAE						
<i>Anampses caeruleopunctatus</i>	DC		0.03	0.03	0.09	0.04
<i>Anampses meleagrides</i>	DC		0.1	0.06	0.09	0.04
<i>Bodianus anthioides</i>	DC		0	0	0.02	0.02
<i>Bodianus axillaris</i>	DC		0.17	0.11	0.16	0.06
<i>Bodianus bilunulatus</i>	DC		0.17	0.08	0.52	0.11
<i>Bodianus diana</i>	DC		0.53	0.17	0.66	0.13
<i>Bodianus perditio</i>	DC		0	0	0.07	0.05
<i>Cheilinus trilobatus</i>	DC		0.2	0.07	0.05	0.03
<i>Cheilio inermis</i>	DC		0.27	0.13	0.16	0.06
<i>Cirrhilabrus exquisitus</i>	DC		0	0	0.14	0.06
<i>Coris aygula</i>	DC		0	0	0.23	0.08
<i>Coris caudimacula</i>	DC		0.47	0.19	0.82	0.17
<i>Coris cuvieri</i>	DC		0.07	0.05	0.02	0.02
<i>Coris formosa</i>	DC					
<i>Gomphosus caeruleus</i>	DC		0.5	0.13	0.59	0.13
<i>Halichoeres cosmetus</i>	DC		0	0	0.43	0.1
<i>Halichoeres hortulanus</i>	DC		0.73	0.17	1.11	0.17
<i>Halichoeres iridis</i>	DC		0.17	0.17	0.05	0.05
<i>Halichoeres nebulosus</i>	DC		0.13	0.13	0.39	0.12
<i>Halichoeres scapularis</i>	DC		0	0	0.16	0.16
<i>Hemigymnus fasciatus</i>	DC		0.2	0.09	0.32	0.1
<i>Hologymnosus annulatus</i>	DC		0.13	0.06	0.2	0.06
<i>Hologymnosus doliatus</i>	DC		0.03	0.03	0.07	0.04
<i>Iniistius pavo</i>	DC		0	0	0.02	0.02

<i>Iniistius pentadactylus</i>	DC		0.23	0.12	0.02	0.02
<i>Labroides bicolor</i>	DC		0.3	0.15	0.3	0.09
<i>Labroides dimidiatus</i>	DC		0.87	0.25	1.34	0.29
<i>Macropharyngodon bipartitus</i>	DC		0	0	0.02	0.02
<i>Novaculichthys taeniourus</i>	DC		0	0	0.02	0.02
<i>Pseudodax moluccanus</i>	O		0.03	0.03	0.45	0.11
<i>Stethojulis albovittata</i>	DC		0.4	0.15	0.27	0.08
<i>Thalassoma amblycephalum</i>	DC		0.83	0.34	5.57	1.68
<i>Thalassoma hebraicum</i>	DC	T	1.8	0.36	3.98	0.62
<i>Thalassoma lunare</i>	DC	T	1	0.27	4.57	0.93
LETHRINIDAE						
<i>Lethrinus crocineus</i>	NC	T	1	0.31	0.59	0.16
<i>Lethrinus harak</i>	NC	T	1.1	0.42	0.41	0.11
<i>Lethrinus microdon</i>	NC	T	0.1	0.07	0.91	0.27
<i>Monotaxis grandoculis</i>	NC	T	0.3	0.12	0.64	0.35
LUTJANIDAE						
<i>Aphareus furca</i>	PI	T	0.1	0.06	0.11	0.06
<i>Aprion virescens</i>	PI	T	1	0.2	1.75	0.33
<i>Lutjanus bengalensis</i>	NC	T	0	0	0.61	0.5
<i>Lutjanus bohar</i>	NC	T	0.37	0.16	0.93	0.19
<i>Lutjanus ehrenbergii</i>	NC	T	0.07	0.07	0	0
<i>Lutjanus fulviflamma</i>	NC	T	0.53	0.5	2.66	1.72
<i>Lutjanus gibbus</i>	NC	T	1.67	0.61	4.16	1.08
<i>Lutjanus kasmira</i>	NC	T	2.6	1.58	9.61	3.63
<i>Lutjanus lutjanus</i>	NC	T	9.9	6.81	29.07	12.54
<i>Lutjanus rivulatus</i>	NC	T	0	0	0.14	0.1
<i>Macolor niger</i>	NC	T	0.2	0.15	0.11	0.06
MALACANTHIDAE						
<i>Malacanthus brevirostris</i>	DC		0	0	0.02	0.02
<i>Malacanthus latovittatus</i>	NC		0.27	0.16	0.27	0.09
MICRODESMIDAE						
<i>Gunnellichthys monostigma</i>	DP		0.6	0.6	0	0
MOBULIDAE						
<i>Manta birostris</i>	DP	T	0	0	0.05	0.03
MONACANTHIDAE						
<i>Aluterus scriptus</i>	O		0.07	0.07	0.05	0.03
<i>Cantherhines dumerilii</i>	BSI		0	0	0.02	0.02
<i>Cantherhines pardalis</i>	BSI		0.33	0.1	0.32	0.07
MONODACTYLIDAE						
<i>Monodactylus argenteus</i>	DP		0	0	3.18	3.18
MULLIDAE						
<i>Mulloidichthys flavolineatus</i>	NC		0.13	0.08	3.43	1.23
<i>Mulloidichthys vanicolensis</i>	NC		0.73	0.73	0.05	0.03
<i>Parupeneus barberinus</i>	DC		0.43	0.14	0.41	0.23
<i>Parupeneus cyclostomus</i>	PI		0.63	0.18	0.61	0.12
<i>Parupeneus indicus</i>	DC		0.17	0.08	0.45	0.14
<i>Parupeneus macronema</i>	DC		1.87	0.44	2.52	0.37
<i>Parupeneus pleurostigma</i>	DC		0.43	0.31	0.23	0.14
<i>Parupeneus rubescens</i>	DC		0.13	0.08	0.2	0.12
<i>Parupeneus trifasciatus</i>	DC		0.43	0.13	0.34	0.1
MURAENIDAE						
<i>Gymnothorax favagineus</i>	NC		0	0	0.2	0.06
<i>Gymnothorax flavimarginatus</i>	NC		0	0	0.32	0.09

<i>Gymnothorax javanicus</i>	NC		0	0	0.02	0.02
MYLIOBATIDAE						
<i>Aetobatus narinari</i>	DC	T	0.03	0.03	0.02	0.02
NEMIPTERIDAE						
<i>Scolopsis bimaculata</i>	DC		0.07	0.05	0.5	0.25
<i>Scolopsis ghanam</i>	DC		2.2	1.55	0.16	0.08
OPLEGNATHIDAE						
<i>Oplegnathus robinsoni</i>	O		0	0	0.05	0.03
OSTRACIIDAE						
<i>Ostracion cubicus</i>	BSI		0.07	0.05	0.36	0.07
<i>Ostracion meleagris</i>	BSI		0.13	0.06	0.02	0.02
PEMPHERIDAE						
<i>Parapricanthus ransonneti</i>	NP		0.03	0.03	29.3	23.51
<i>Pempheris adusta</i>	NP		0	0	0.02	0.02
PINGUIPEDIDAE						
<i>Parapercis clathrata</i>	DC		0	0	0.02	0.02
<i>Parapercis hexophthalma</i>	DC		0.13	0.09	0.05	0.03
POMACANTHIDAE						
<i>Apolemichthys trimaculatus</i>	O		0	0	0.18	0.1
<i>Centropyge multispinis</i>	O		0.23	0.08	0.43	0.11
<i>Pomacanthus imperator</i>	BSI		0.47	0.09	0.89	0.1
<i>Pomacanthus rhomboides</i>	DC		0	0	0.05	0.03
<i>Pomacanthus semicirculatus</i>	BSI		0.37	0.12	0.89	0.13
POMACENTRIDAE						
<i>Abudefduf sparoides</i>	O		0.03	0.03	0	0
<i>Abudefduf vaigiensis</i>	O		0.17	0.12	5.86	1.54
<i>Amphiprion allardi</i>	O		0	0	0.16	0.08
<i>Chromis fieldi</i>	O		1.3	0.53	2.77	0.69
<i>Chromis weberi</i>	DP		2.77	1.41	8.55	3.44
<i>Dascyllus trimaculatus</i>	O		0.6	0.41	1.41	0.33
<i>Neopomacentrus cyanomos</i>	O		0	0	0.61	0.35
<i>Pomacentrus caeruleus</i>	O		0.23	0.12	0.86	0.39
PRIACANTHIDAE						
<i>Priacanthus hamrur</i>	NC		0	0	1.48	0.91
PTEROIDAE						
<i>Pterois miles</i>	PI		0	0	0.02	0.02
RHINOBATIDAE						
<i>Rhynchobatus djiddensis</i>	NC	T	0	0	0.02	0.02
SCARIDAE						
<i>Calotomus carolinus</i>	H	T	0.07	0.07	0.14	0.08
<i>Chlorurus sordidus</i>	H	T	0.17	0.08	0.48	0.2
<i>Scarus caudofasciatus</i>	H	T	0.03	0.03	0.02	0.02
<i>Scarus frenatus</i>	H	T	0.03	0.03	0	0
<i>Scarus ghobban</i>	H	T	0.3	0.13	0.66	0.19
<i>Scarus persicus</i>	H	T	0.03	0.03	0.09	0.05
<i>Scarus rubroviolaceus</i>	H	T	2.23	0.75	1.3	0.18
<i>Scarus tricolor</i>	H	T	0	0	0.02	0.02
SCOMBRIDAE						
<i>Scomberomorus commerson</i>	PI	T	0.2	0.09	0.09	0.04

SERRANIDAE							
<i>Aethaloperca rogae</i>	NC	T	0.03	0.03	0.25	0.07	
<i>Cephalopholis argus</i>	PI	T	0.4	0.15	0.07	0.04	
<i>Cephalopholis miniata</i>	NC	T	0.17	0.1	0.36	0.12	
<i>Cephalopholis nigripinnis</i>	NC	T	0.13	0.08	0.02	0.02	
<i>Epinephelus coeruleopunctatus</i>	PI		0	0	0.05	0.05	
<i>Epinephelus fasciatus</i>	NC	T	0.07	0.05	0.43	0.15	
<i>Epinephelus lanceolatus</i>	NC	T	0.03	0.03	0.16	0.06	
<i>Epinephelus macrospilos</i>	DC	T	0.03	0.03	0.02	0.02	
<i>Epinephelus malabaricus</i>	NC	T	0.33	0.11	0.3	0.09	
<i>Epinephelus multinotatus</i>	PI	T	0	0	0.02	0.02	
<i>Epinephelus tukula</i>	NC	T	0	0	0.14	0.05	
<i>Pseudanthias squamipinnis</i>	DP		2.2	1.32	4.23	1.93	
<i>Variola louti</i>	PI	T	0	0	0.14	0.05	
SIGANIDAE							
<i>Siganus luridus</i>	H		0.03	0.03	0.11	0.07	
<i>Siganus sutor</i>	H	T	1.67	0.45	6.66	2.27	
SPARIDAE							
<i>Rhabdosargus sarba</i>	DC	T	0	0	0.02	0.02	
SPHYRAENIDAE							
<i>Sphyraena barracuda</i>	DC	T	0.03	0.03	0	0	
<i>Sphyraena putnamae</i>	NC	T	0	0	7.41	7.41	
TETRAODONTIDAE							
<i>Arothron hispidus</i>	NC		0	0	0.07	0.04	
<i>Arothron mappa</i>	NC		0.03	0.03	0.02	0.02	
<i>Arothron nigropunctatus</i>	NC		0	0	0.07	0.04	
<i>Canthigaster bennetti</i>	O		0.5	0.16	0.14	0.05	
<i>Canthigaster valentini</i>	O		0.1	0.07	0.25	0.08	
ZANCLIDAE							
<i>Zanclus cornutus</i>	BSI		5.33	4.1	1.61	0.27	

Table S3: Functional structure of fish communities on southwest Indian Ocean reefs, expressed as percentages of total numbers of species (species richness). In the previous studies data were collected on the outer reef slope using underwater visual census (UVC), whereas baited remote underwater video stations (BRUVS) were used in the present study

Site	Study	Technique	Carnivores	Omnivores	Herbivores	No. species
Tofo/Barra	Fordyce (2016)	UVC	80%	8%	9%	324
Ponta Malongane	Floros et al. (2012)	UVC	78%	11%	11%	284
Andavadoaka	Gillibrand et al. (2007)	UVC	76%	11%	13%	334
Bazaruto Archipelago	Maggs et al. (2010)	UVC	76%	12%	12%	249
Tuléar	Harmelin-Vivien (1979)	UVC	74%	13.5%	12.5%	323
Juan de Nova	Chabanet-Durville (2005)	UVC	73%	11%	16%	299
Glorieuses	Durville et al. (2003)	UVC	73%	12%	15%	332
Mayotte	Chabanet (2002)	UVC	69%	12.5%	18.5%	225
Geyser et Zéléé	Chabanet et al. (2002)	UVC	69%	16%	15%	294
Réunion	Chabanet (1994)	UVC	51%	24%	25%	257
Lighthouse Reef	Present study	BRUVS	80%	7%	13%	189
San Sebastian Reef	Present study	BRUVS	81%	8%	11%	195

Figure S1: Schematic of lightweight aluminium baited remote underwater video station (BRUVS) (AIMS Engineering Workshops®) used during deployments in southern Mozambique, showing frame, underwater camera housings, and bait arm with bait bag attached

Figure S2[renamed, was Figure S3]: Variance partitioning of explanatory variables in top-ranked model (negative binomial generalised linear model) of target fish abundance (MaxN) in southern Mozambique

Figure S3[renamed, was Figure S2]: Cluster analysis, using SIMPROF, of the compositions by genera of reef-fish communities at Lighthouse and San Sebastian reefs in 2016. The different shades indicate groups that SIMPROF determined were significantly different. The percentage of baited remote underwater video stations (BRUVS) occurring in each significant cluster are shown at the two sites (San Sebastian Reef and Lighthouse Reef) (middle graph) and in the three major habitats (lower graph)

Figure S4: Map of regional study sites listed in Supplementary Table S3