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Assessment of reef fish and benthic cover of the North and South Dar es Salaam Marine Reserves system before the 2016 El Niño

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Abstract
The status of reef fish density, diversity, species richness, biomass and coral cover was evaluated by comparing the conditions in two Dar es Salaam Marine Reserves (DMRs); the North Dar es Salaam Marine Reserve (NDMR; gazetted in 1975), and the South Dar es Salaam Marine Reserves (SDMRs; gazetted in 2007), before the 2016 El Niño. A 10 m line-intercept transect was used to characterize benthic cover and a 50 m belt transect was used to assess reef fish population status. Sampling occurred between August 2014 and April 2015. The results showed that fish biomass in the (NDMRs) was 2.7 times higher than that in the (SDMRs) and live hard coral cover was 3 times higher. Higher reef fish density, biomass, diversity, species richness and live hard coral cover were revealed before 2016 El Niño in NDMRs as compared to the SDMRs. Differences in status are linked to differences in time of gazetting and level of effective management in the marine protected areas (MPAs), where NDMRs has a General Management Plan (GMP) while SDMRs does not, and the differences in management are likely to have contributed to the differences in fish biomass and coral cover.

Keywords: reef fish, coral reef, marine reserves, conservation, El Niño

Introduction
The DMRs comprising of the NDMRs and SDMRs were gazetted in 1975 and 2007 respectively. The NDMRs has a GMP that was developed in 2005 while the SDMRs lacks a GMP. The GMP serves as guidance to ensure that resource protection and recreational activities and developments remain balanced and compatible with one another. It also sets out an active process which guides subsequent planning and implementation on how to effectively conserve and manage the resources (URT, 2005). Before being gazetted, the DMRs were characterized by unregulated fishing, including widespread use of beach seines and spear fishing.

Informed management intervention in MPAs includes understanding the impacts of El Niño in order to institute adaptive management as part of a disaster response mechanism. The impact could easily be detected if data were collected before the event. El Niño is often termed the "Southern Oscillation", or ENSO, where the atmosphere and ocean collaborate together (Trenberth, 1997). However, some scientists confine the term to the coastal phenomenon, while others use it to refer to the basin wide phenomenon (Trenberth, 1997; Aceituno 1992; Glantz, 1996).

Surface temperatures in the Eastern and Central tropical Pacific Ocean during the ENSO in late 2015 exceeded 2 degrees Celsius above average (Glantz, 1996), providing evidence that the 2015-16 El Niño was one of the strongest on record, comparable with the 1997-98 and 1982-83 events. Subsequently, it led

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to unusually high levels of warming and changes in the local and regional coral reef ecology, including coral bleaching and mortality. It was reported that the threshold on the Sea Surface Temperature (SST) for three consecutive months was only 0.4 °C (Glantz, 1996). Throughout 2014, the inter-tropical Pacific SST rose steadily from the below average values observed in 2013. They remained near borderline values for some time (October to February) before finally breaking the El Niño threshold (+0.5°C) in March 2015 (FAO, 2014).

High SST can lead to coral bleaching, which refers to the loss of the zooxanthellae by the host (i.e. the coral), or the loss of photosynthetic pigments within the coral structure itself, and can cause coral mortality (Muhando, 1999; Wagner, 2004).

Consequently, El Niño events are a serious public concern, and forecasting is critical to highlight the need for society to get ready for the potential impacts of the event. Additionally, El Niño is also responsible for larger magnitude weather anomalies such as floods, drought, heat waves, hurricanes, and tsunamis resulting in disease outbreaks and water shortages, among other challenges. Knowledge on El Niño can provide usable information for decision makers to choose whether to pursue strategic or tactical disaster risk reduction policies (Glantz *et al*., 2018). El Niño intensities can easily be quantified ranging from weak to very strong (Glantz *et al*., 2018). If severe, El Niño can result in coral bleaching and subsequent mass coral mortality. Baseline data from before the event is therefore critical for tracing the impact on the ecosystem.

Studies on coral reef systems have been carried out along the Tanzanian coast to describe the coral fauna of the East African Coast (Hamilton, 1975), to assist with the establishment of MPAs in southern Tanzania (Muhando *et al*, 1998), and assess the status of coral reefs in the DMRs and other MPAs in Tanzania (Muhando and Francis, 2000). These studies have also served to assess coral reef degradation in Tanzania (Mohammed *et al.,* 2000), assist with coral reef management (Wagner, 2004), and to determine the role of improved fisheries management in increasing the biomass of fish and benthic communities in Tanzania (McClanahan *et al,* 2009). Prior to the establishment of the DMRs their natural systems had been degraded due to the widespread use of dynamite and other destructive fishing techniques (Benno, 1992).

Friedlander (2007) reported high species richness, biomass, density, habitat complexity and good habitat quality in protected areas as compared to areas open to fishing in Hawaii. FAO (2011) suggested that the most common types of indicators of biological response within protected areas include increased density, biomass and size of animals. Syms and Jones (2000) observed that disturbance plays a substantial role in structuring communities of coral-reef fishes by modifying both spatial and temporal heterogeneity.

There is substantial scientific evidence that areas with increased management (when designed appropriately) have more and bigger fish and a higher biomass than those without management (Côte *et al*., 2001; Friedlander and De Martini, 2002; Friedlander *et al*., 2003a; Friedlander *et al*., 2003b; Dulvy *et al*., 2004; Kamukuru *et al.,* 2004; McClanahan *et al.,* 1999).

However, only a few studies have been undertaken in the DMRs. Hamilton (1975) and Wagner (2004) indicated that some parts of the DMRs had significant live coral cover which was valuable as a tourist attraction while some areas were already degraded as a result of dynamite fishing. Kamukuru (1997, 2009) assessed the biological status of the coral reefs, the trap fishery and reproductive biology of the white spotted rabbit fish *Siganus sutor* (Siganidae), respectively.

Understanding the relationship between reef habitat and fish population structure is becoming increasingly recognised as important for the sustainable management of fisheries and MPA resources (see for e.g., Anderson and Millar 2004; García-Charton *et al*., 2004). The physical structure of the reef has been observed to play a key role in the organization of fish assemblages, protection of reef fish from predators and providing access to food (Tuya *et al,* 2011). Thus, this study was aimed at investigating and establishing the status of reef fish density, biomass, richness and live hard coral cover in the DMRs before the 2016 El Niño event. El Niño has been reported to cause tremendous impacts including the collapse of coral reef ecosystem. Taking the 1998 El Niño as an example, coral cover was 81.2 % before bleaching, and dropped to 37 % after bleaching at Bongoyo West (Wagner, 2004). Around Mbudya Island, coral mortality was 40-60 % (Wagner *et al*., 2001), while at Pangavini, 77.5 % of the coral reef died (Mrema, 2001). At Fungu Mkadya, 60 % died (Bipa, 2000) while at Fungu Yasini southwest, almost 100 % died (Peter, 2002). Similarly in Mnazi Bay in the Ruvuma Estuary Marine Park only 50 % of the coral reef survived after bleaching (Wilkinson, 1998). Reef fish density has been reported to be correlated with live coral in terms of both density and biomass which implies that when coral reefs are affected, the whole ecosystem is jeopardized (Kamukuru, 1997; Julius *et al.*, 2016). Taking the above into account, coral cover and the associated reef fish were assessed before the predicted 2016 El Niño event.

North Marine Reserve includes Mbudya, Bongoyo and Pangavini Islands, while the Southern Marine Reserve is comprised of Inner and Outer Sinda and Inner and Outer Makatumbe islands (Fig. 1). The respective locations of the islands are given in Table 1. The islands are surrounded by diverse and unique habitats including coral reefs, sea grass beds, sandy

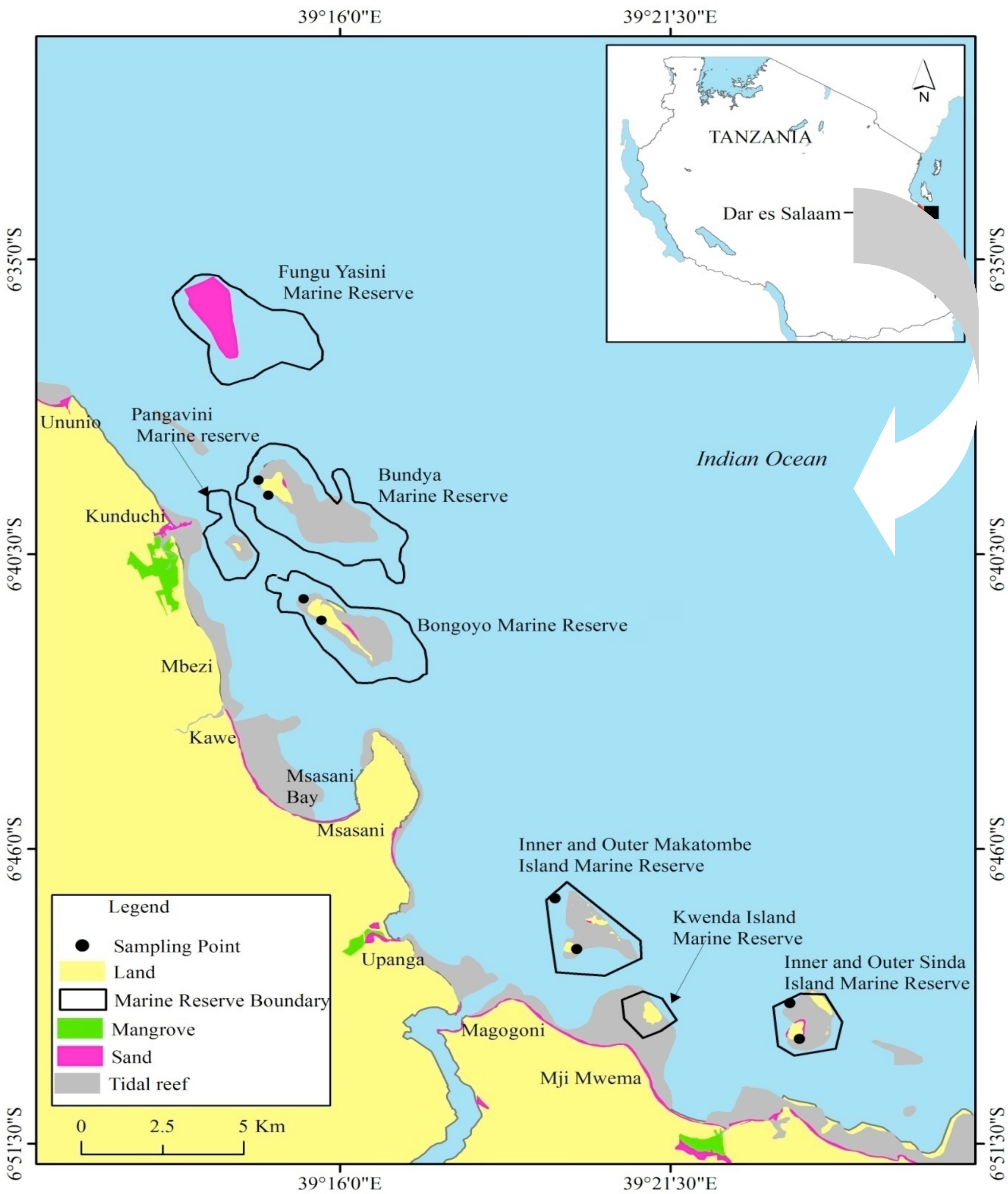

Figure 1. Map of Dar es Salaam coast showing the DMRs study sites.

Study site and methodology

Study site

The study was conducted within the DMRs which extends from north to south of Dar es Salaam City between 06°35' - 06°45' S and 39°13' - 39°17'E. The beaches and rocky shores and lie on a shallow continental shelf with waters of less than 20 metres deep (URT, 2005). Fishing and collection of shellfish in the reserves (no take), recreational and tourist activities are common on and around the islands.

Methodology

Data was collected during low tide. Three methods were used to assess the habitats and resources of the DMRSs. Firstly, a rapid assessment using a Manta tow survey was conducted around all islands (Mbudya, Bongoyo, Makatumbe and Sinda) to select the sampling site. Coordinates of the selected sites were recorded using a GPS. Sampling was done on the southern and western sides of the islands because they were sheltered and easily accessible, and the corals were well-established.

Secondly, a visual census survey was used to assess the habitats and quantify fishes. All fish species observed along each transect were identified with the aid of a field fish identification guide (Richmond, 2002). Fish size was classified based on their total length. Specimens from 1-10 cm were considered as juvenile, from 11 to 20 cm as recruits, and 21cm and above as adult. Slates were used to record the data, which included fish description, size and number of individuals falling of a particular species and size. The fish were counted by tallying the information from the slates and where larger numbers of reef fish were encountered, a rough estimated was done. An underwater camera was used for taking photos of fish species which were not easily identified on the spot. Further detailed identification in the laboratory was carried out using the field guide by Fischer and Bianchi (1984), the Coral Reef Fishes Pocket Guide (Lieske and Myers, 2001), and Bianchi (1985).

Fish counting was undertaken by adopting the method of Samoilys and Carlos (2000) by swimming at a slow, constant speed along the transect at 3-4 metre min^{-1}, depending on fish abundance and complexity of the habitat or rugosity of coral reef, covering 33 m^2 min^{-1}. A break period of 20 minutes between transects was allowed for fish to return to the area.

Coral reef fish diversity was determined by the Shannon index (H'):

$$H' = -\sum_{i=1}^{s} p_i \ln(p_i)$$

Where p_i is the proportion of all observations in the i^{th} species category, and S is the total number of species. The Shannon index measures both richness (the number of species) and evenness, or how evenly individuals are distributed among species. High values of H' denote high biodiversity.

The third method used was a Line Intercept Transect (LIT). A 10 m LIT was used to characterize benthic cover along a 50 m wide belt to assess reef fish density, biomass, species richness and diversity based on Underwater Visual Census (UVC) techniques (English, 1994). Eight belt transects running from immediately above the reef crest to the reef slope were conducted in the study. Two divers recorded data on either side of the transects. Three surveys were conducted from August to September, 2014, January to February, 2015, and April, 2015. A total of 24 swim tracks were performed for each sampling phase per site.

Data analysis

Fish densities obtained from direct field counts (UVC) were organized using Microsoft Excel 2013 before analysis. Fish biomass values were computed from length estimates using the conversion equations (W = a * Lb) of published length-weight relationships from FishBase (www.fishbase.org), where L is fish length in centimetres estimated from the field during data collection, W is fish weight in grams computed from the equation, a is the y-intercept and b is the slope of the equation when the natural logarithm is applied. The values obtained for an individual fish was multiplied by the number of fish of each species counted and sizes observed, providing an estimate of total biomass (g.500m-2) per transect. The mean fish biomass (± standard error) and mean density (± standard error) was also calculated. Live coral benthic cover was also organized in Microsoft Excel in a different file before analysis. Statistical analysis was carried out using Graph Pad Instant Statistical software, version 3.06.

Data were tested for normality before reef fish density, biomass and live coral cover were evaluated for homogeneity and heterogeneity. The Mann-Whitney, Signed-ranks and Kruskal-Wallis tests were used for testing the data.

Reef density and biomass were tested for normality before analysis using Graph Pad Instant Statistical software version 3.06 for statistical analysis. The Mann-Whitney test was used to evaluate differences in reef fish biomass and density between the North and South DMRs, and the Wilcoxon matched-pairs signed-ranks test was used to evaluate coral reef fish population structure between the marine reserves. Live coral cover was tested by the Wilcoxon matched-pairs signed-ranks test.

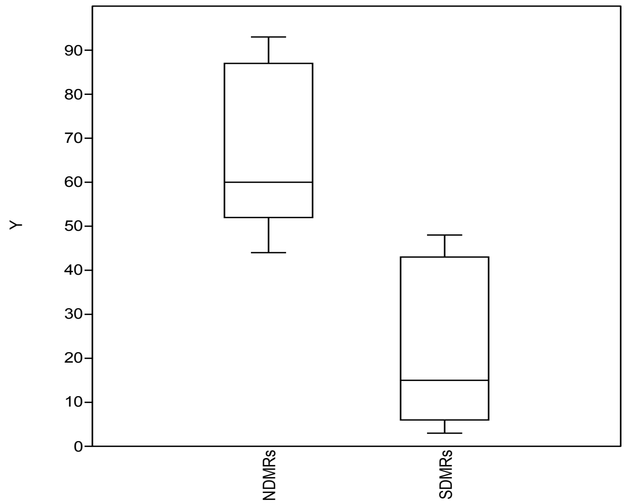

Figure 2. Live coral cover (%) in the DMRs.

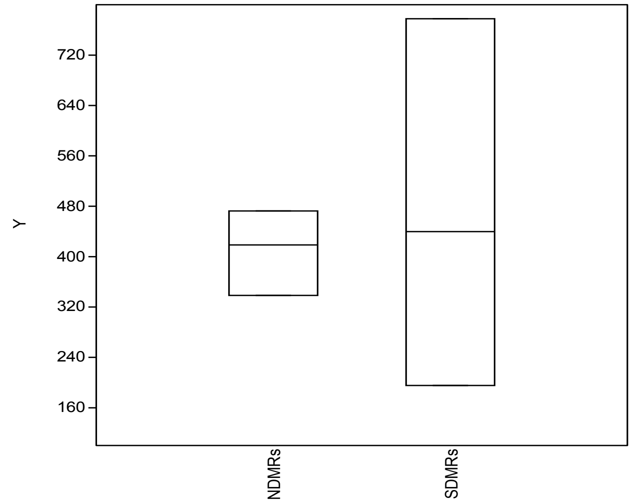

Figure 3. Reef fish density (#/ha) in the DSMs.

Results

There was significant a difference in reef fish density and biomass (Mann-Whitney Test (U); P = 0.0004 for both abundance and biomass). Additionally, hard coral cover was significantly different between sites; higher in NDMRs than in SDMRs (Wilcoxon matched-pairs signed-ranks (T); $p<0.0001$). The test also revealed that similarity existed among the northern islands in the NDMRs (Mbudya and Bongoyo) and in those in the SDMRs (Sinda and Makatube) for both fish biomass and abundance (Mann-Whitney Test (U); $p<0.05$).

Coral cover, fish density and biomass

The mean live coral in % ± SE was significantly higher (Wilcoxon matched-pairs signed-ranks test, $T_{0.05,\ 32}$ = 528, $p<0.0001$) in the NDMRs (69.688 + 3.249) than in the SDMRs (22.969 ± 2.966) (Fig. 2). Likewise, the mean fish density was significantly higher ($U'_{0.05}$ *188. 69 = 8372*; $p < 0.05$) in the NDMRs (442.6 ± 69.4 individuals per 500 m^2) than in the SDMRS (408.4 ± 104.2 individuals per 500 m^2) (Fig. 3). The NDMRs had significantly higher reef fish biomass (27.7 ± 5.4 kg/500 m^2) than the SDMRs (10.1 ± 2.6 kg/500 m^2); $U'_{0.05}$ *188.69 = 8944;* $p < 0.05$ (Fig 4). A similar situation was observed for the juvenile fish at P < 0.0001 (Fig. 5). A very strong positive correlation ($r^2=0.955$) was revealed between live coral cover and fish abundance in both the NDMRs and SDMRs (Figs. 6 and 7).

Reef fish diversity and species richness

The study revealed that there were 59 species within 26 families in NDMRs (Fig. 8) and 40 species within 22 families in SDMRs (Fig. 9). The NDMR was dominated by the butterfly fishes (Chaetodontidae) which contributed 17 %, followed by Pomacentridae at 15 %, and Pomacanthidae at 8 % of fish families. The fish family

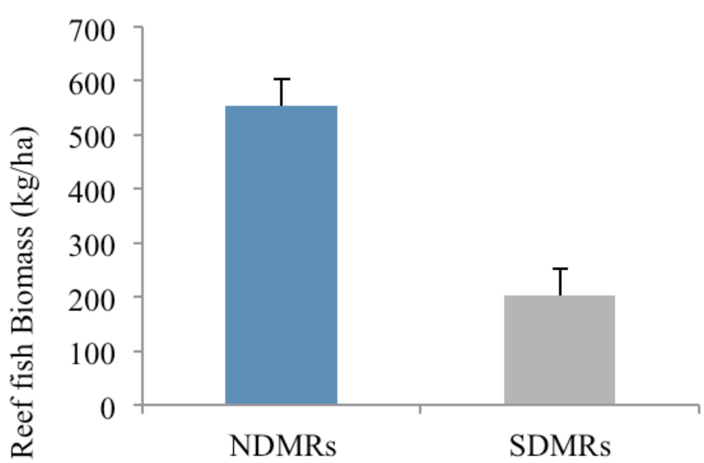

Figure 4. Reef fish biomass in the DMRs.

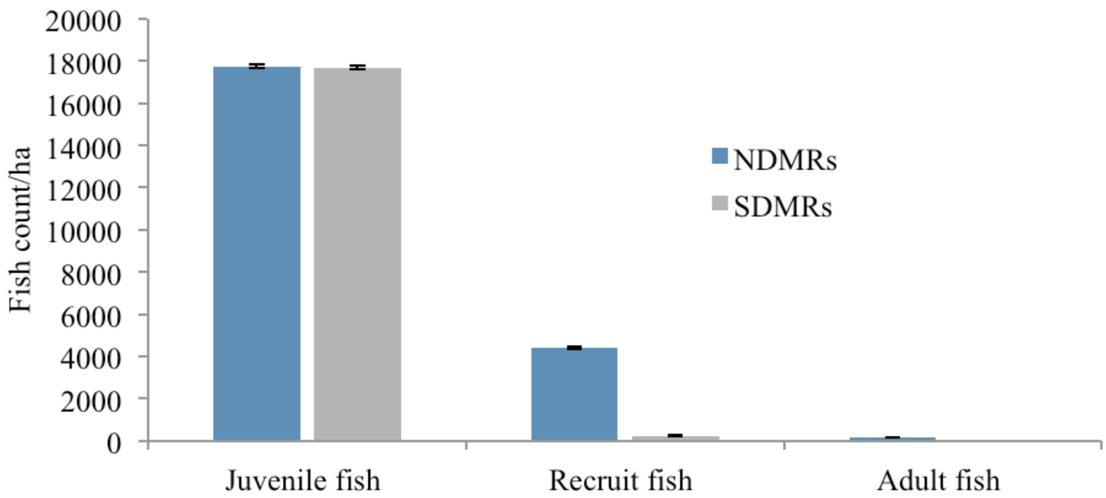

Figure 5. Reef fish population structure in the DMRs.

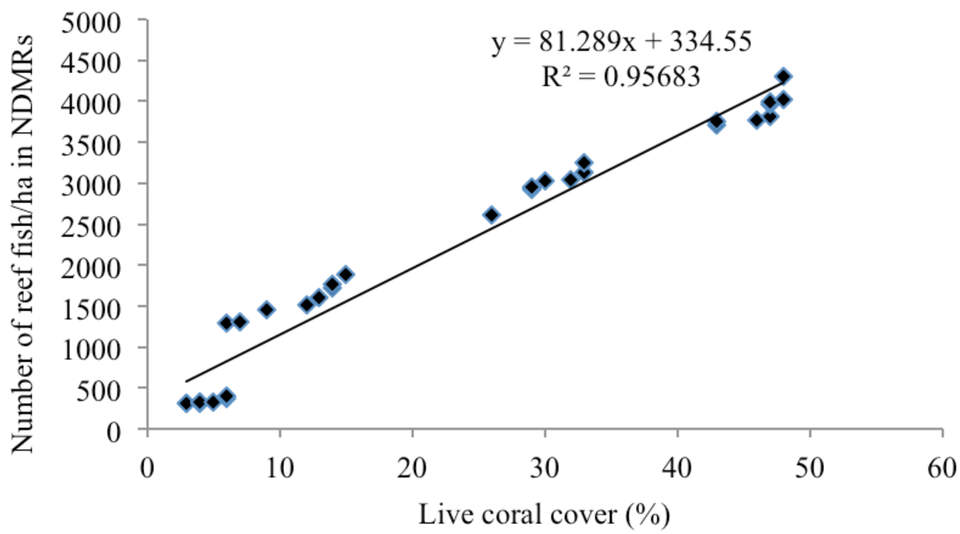

Figure 6. Correlation of reef fish abundance with live coral cover in NDMRs.

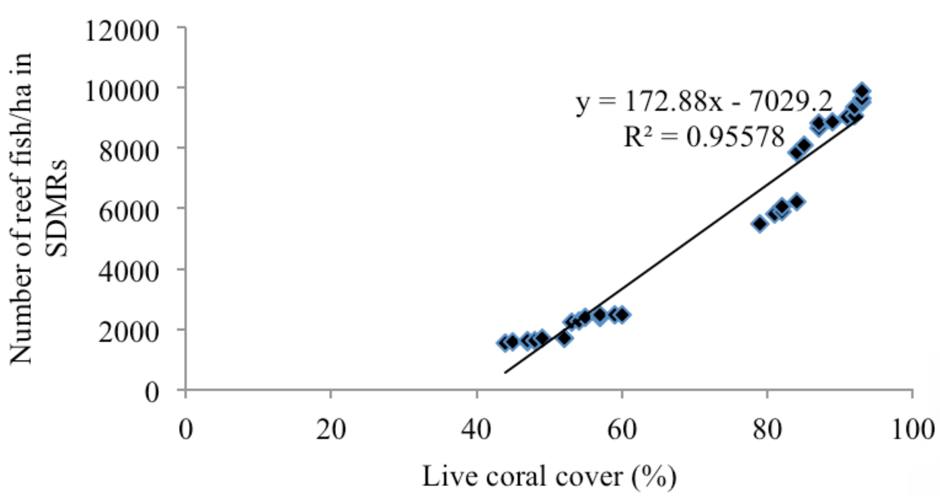

Figure 7. Correlation of reef fish abundance with live coral cover in SDMRs.

Table 1. Number of reef fish families and species recorded in DMRs.

Study site	Shannon index (H´)	No. of Reef fish family	No. of Reef fish species
NDMRs (with GMP)	4.3	26	59
SDMRs (without GMP)	3.2	22	40

composition in SDMRs was dominated by small bodied individuals, namely Pomacentridae (damselfish and clown fishes) at 18 %, followed by Pomacanthidae at 13 %. Mullidae and Labridae both contributed 8 % and the remaining proportion was shared by other groups. The fish diversity was higher in the NDMRs than in the SDMRs with Shannon Wiener diversity indices (H´) of 4.323 and 3.22692, respectively (Table 1, Figs. 7 and 8).

Discussion

The status on reef fish and benthic cover of the North and South DMRs before the 2016 El Niño is now established. The higher reef fish density, biomass, species diversity and live hard coral cover observed in the NDMRs compared to the SDMRs indicates the impact of differences in management effectiveness and the implication of differences in the time since the reserves were gazetted, as well as the level of management between the marine reserves though the guidance of a GMP.

Reef fish were dominated by the families Chaetodontidae, Pomacentridae and Pomacanthidae in the NDMRs; probably because butterflyfishes (Chaetodontidae) have been observed globally to constitute almost half of the coralivorous fish families, followed by other families including the Pomacentridae (Cole *et al.*, 2008). It has been observed by Garpe and Öhman (2003) and Halford *et al.* (2004) that the loss of structural reef complexity often affects the health of fish communities. Sano *et al.* (1987) also observed that the abundance and diversity of the coral reef community was observed to have declined by approximately two-thirds after the reef collapsed into a formless rubble state. The low fish diversity in SDMRS could be a response to loss of coral cover (Cole *et al.*, 2008). Both dominant family groups indicate a disturbed habitat which is attributed to destructive fishing practices impacting coral growth as well as causing physical damage.

Prevalence of juvenile fishes in both sites emphasises the role of coral reefs as nursery grounds (Fig. 5). Higher abundance of both recruits and adult fishes in NDMRs indicates the value of high coral cover. The low number of recruits and adult fish observed in the SDMRs suggests their excessive removal by unregulated fishing activities in the area as a contributing factor. The better biological status in the NDMRs was possibly due to highly regulated fishing activities as well as the older age of the reserve compared to the

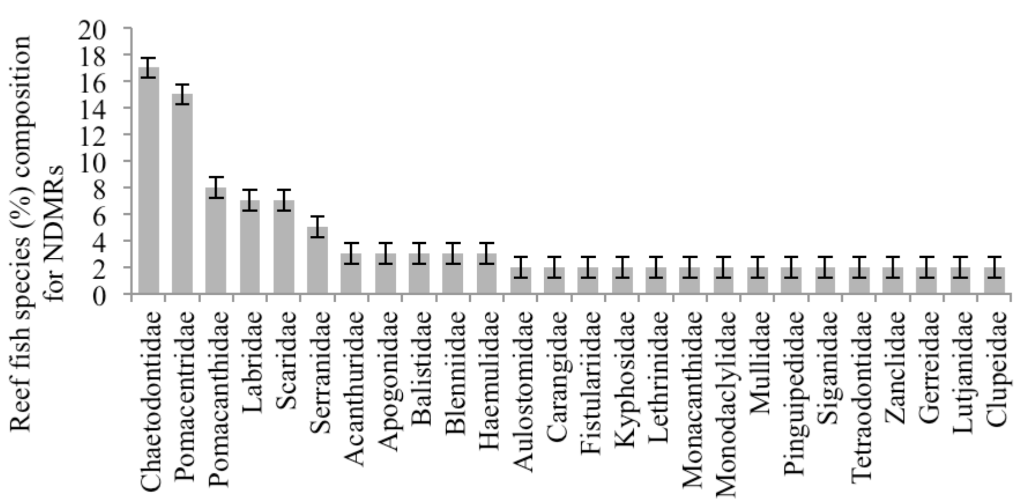

Figure 8. Reef fish families in NDMRs.

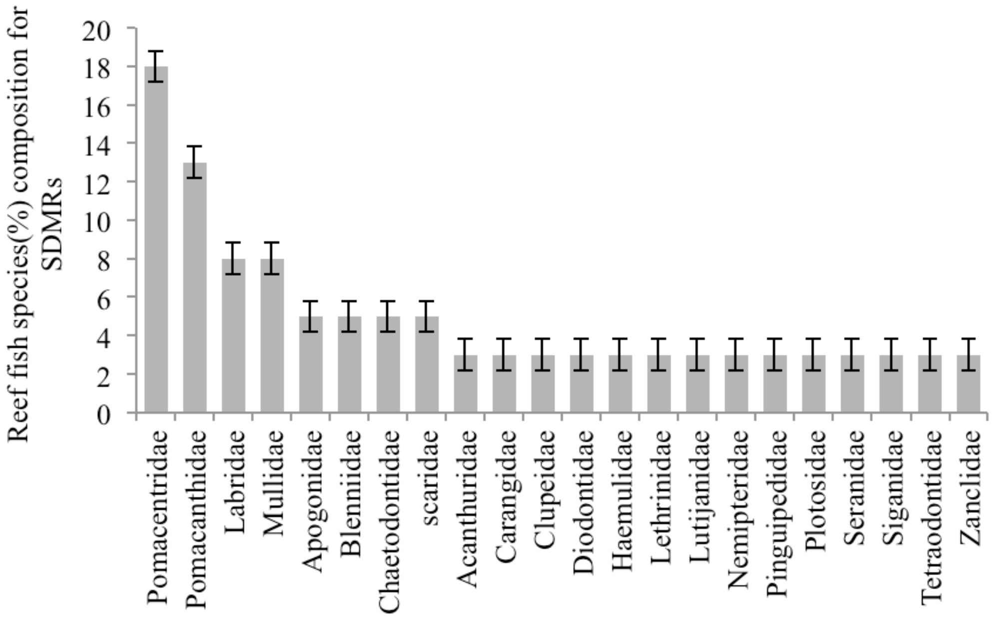

Figure 9. Reef fish families in SDMRs.

SDMRs. In contrast to the SDMRs, the existence of a GMP for the NDMRs provides guidance and attributes accountability to all key stakeholders in the management and conservation of resources in this area.

With respect to fish biomass status in other areas in the WIO, Kamukuru *et al.* (2004) reported the existence of over six times the biomass of *Lutjanus fulviflamma* in Mafia Island Marine Park compared to unprotected areas. McClanahan *et al.* (2009, 2015) reported that the biomass of fish rose continuously from 260 to 770 kg/ha from 1994 to 2007 on Tanga reefs because of stability of coral cover due to increased management, and that the reefs exhibited more resilience due to management.

Reef fishes have been reported to exhibit a strong positive correlation with live hard coral cover substratum, with this being considered critical for the provision of food, shelter and living space for fishes (Beukers and Jones, 1997). Also, Garpe and Öhman (2003) observed that sites with the highest proportion of dead coral exhibited the highest degree of dispersion of fish assemblages. Habitat characteristics play a dominant role in determining fish assemblage composition on coral reefs (Garpe and Öhman, 2003). The high percentage coral reef cover in the NDMRs is associated with the presence of both high reef fish abundance and biomass in NDMRs. This has management implications, as reef fish are automatically conserved if the coral reef is maintained in good condition.

Conclusion

This study revealed that the NDMRs has higher reef fish density, biomass, diversity, species richness and live hard coral cover compared with the SDMRs. This study recommends another survey using similar methods after the 2016 El Niño to assess the impact of the event on the ecology in the DMRs. This will assist in improved management and sustainability of the Marine Reserves through regular documentation on their biological status.

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