Coral Recruitment and Coral Reef Resilience on Pemba Island, Tanzania

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Abstract

This study explores the patterns of coral recruitment, survivorship and resilience on the coral reefs of the west coast of Pemba Island, Tanzania. The results show that recovery from the 1998 mass coral bleaching event has been patchy, with great variation in coral cover among sites, and a generally high macroalgal cover. Sites with low coral recovery were found to exhibit higher numbers of coral recruits but lower survivorship, implying that larval supply is not impeding recovery but rather that local stressors are reducing coral reef resilience. The main stressors observed were predation by *Acanthaster plancii*, overfishing and use of destructive fishing methods (including dynamite fishing). *A. plancii* predation was shown to negatively correlate with coral recruit survivorship, implying that it is a potential cause of failure of corals to reach adult sizes. Fish surveys showed that Pemba is being overfished, with the vast majority of fish observed less than 10 cm in length, and only 4 individuals larger than 40 cm recorded throughout the whole survey. It is recommended that the two major, and potentially synergistic, stressors of *A. plancii* predation and overfishing/destructive fishing are addressed in order to avoid loss of Pemba's coral reefs. Land-ocean connections are also explored in this context.

Keywords

Coral recruitment • Resilience • Pemba

Introduction

The island of Pemba lies 50 km off the northern Tanzanian Coast in the Indian Ocean, and forms part of the Zanzibar Archipelago. The climate is tropical and can be broadly divided into two monsoon periods, the northeast monsoon ('kaskazi') with trade winds blowing from the northeast between December and April, and the southwest monsoon ('kusi') with trade winds blowing from the southwest between May and November. The Northeast monsoon is generally characterized by lower wind speeds, calmer seas and higher sea surface temperatures. The doldrums at the end of the northeast monsoon is the usual coral bleaching period in this region. The southwest monsoon is generally characterized by higher wind speeds, rougher seas and lower water temperature (Newell 1959).

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The shoreline consists of stretches of sandy beach interspersed with low limestone cliffs and headlands. Offshore, there are shallow fringing reef flats which drop off rapidly into the ~2,000 metre deep Pemba Channel. There are several smaller islands in the area to the west of Pemba, creating tidal channels between them and sheltered lagoon-like areas behind them. The west coast of Pemba is more protected from oceanic swell and generally exhibits lower wave energy than the east coast, which is more exposed to oceanic swell and where wave energy is generally higher. The tidal range for Pemba is relatively large, with the neighbouring Zanzibar Island (Unguja) harbour exhibiting a mean spring tidal range of 3.9 m (Nhnyete and Mahongo 2007). The islands (from north to south) to the west of Pemba include Njao, Fundo (the largest of the islands), Uvinje and Misali. Coral reefs surround the islands and are present in the tidal channels, lagoons, bays and fringing the western edges of the islands (Grimsditch et al. 2009).

The coral reefs of Pemba are not in the centre of coral diversity for the Western Indian Ocean, which lies in the northern Mozambique channel and from which diversity decreases radially (Obura 2012). However, on the gradient of diversity they are among the ‘second tier’ of the most diverse coral reefs in East Africa and locally extremely important. The local human population relies on them heavily for food security and income from fishing, aquaculture and increasingly SCUBA diving tourism. However, these corals are also vulnerable to bleaching caused by warming sea surface temperatures. In 1998 they bleached heavily and live coral cover decreased from 54% average around the island to 12% in 1999 (Obura 2002). In one particular site on Misali, a study found that live coral cover decreased from 74% to 17% during 1998 (Muhando 2003). In the 5 years after bleaching, the corals around Pemba recovered modestly up to an average of 16% live cover in 2002 with variable recovery around the island (Obura 2002). Overfishing and the use destructive fishing methods are additional threats to the corals of Pemba. Most fishers in the area have low incomes and use traditional fishing boats such as outrigger and dugout sailed canoes with hand lines, beach seines and fish traps. The fishers cannot usually access offshore areas due to their boats’ constraints, so most fishing is carried out relatively close to shore and the nearby reefs are thus intensely fished. Beach seines, gill nets, and dynamite fishing are typical of the destructive methods in the area that cause significant damage to the coral reef structure and populations. Furthermore, population outbreaks of the corallivorous *Acanthaster plancii* have occurred on Pemba’s reefs in the last decade (Obura et al. 2004). Although this organism is a natural part of the coral reef ecosystem, population outbreaks can cause severe and widespread coral mortality. These drivers of coral mortality can reduce coral reef resilience over time, and lead to trajectories of ecological degradation that can have negative socioeconomic implications as ecosystem services associated with coral reefs (for example shoreline protection, food security from fisheries, biodiversity values, recreational values and medicinal values) are reduced and eventually lost. Understanding the factors affecting coral mortality and coral reef recovery or resilience is important for improving the management of coral reefs. In particular, it is important to understand whether coral recruitment is a limiting factor in coral reef recovery or whether mortality (and which causes of mortality) are hindering coral reef recovery.

Coral reef resilience can be defined as ‘following mortality of corals, the ability of the reef community to maintain or restore structure and function and remain in an equivalent ‘phase’ as before the coral mortality’ (Obura and Grimsditch 2009). Mortality could occur, for example, due to bleaching, *A. plancii* predation and/or destructive fishing practices. Coral reef resilience is dependent on a myriad of ecological, physical, oceanographic and anthropogenic factors and the complex interactions between them (Obura 2005). Because of the complex nature of these interactions, it is challenging to determine which factors are the most important for driving resilience. One important factor is coral recruitment that replenishes coral populations after a mortality event, and this paper will present an analysis of spatial patterns for coral recruitment for selected sites along the west coast of Pemba Island, linking recruitment patterns to potential resilience of the sites surveyed and potential drivers of coral reef degradation. The potential for land-ocean connections to affect coral recruitment in Pemba is also discussed.

Materials and Methods

Study sites – Surveys were conducted in February 2009 on 13 sites along the fringing reefs along the west coast of Pemba Island. Table 1 names, describes and locates the sites surveyed, as well as the survey depth. Sites sampled could broadly be categorized into three geomorphological types: sheltered bays (The Hole, Fundo Lagoon and Msuka Bay), fringing reef slopes (Simba Wall, Paradise, Swiss, Mandela, Fundo Outer and Misali) and tidal channels (Njao Gap, Manta, Fundo Inner and Kokota). The depth of the surveys varied according to site in order to sample the most representative areas of the reef ecosystem.

General sampling design – Fieldwork followed the methods outlined in the ‘Resilience Assessment for Coral Reefs’ protocol (Obura and Grimsditch 2009) and is described below.

Benthic surveys – Coral size classes were measured along two or three 25 × 1 m transects per site counting all corals larger than 10 cm, identifying them to genus level and placing them in appropriate size classes according to

Table 1 Shows the sites, sampling depths, geographical position and habitat for the sites surveyed for this study

Site name	Sampling depth (m)	Lat. (S)	Long. (E)	Site description
The Hole	6	4.88720	39.67632	Sheltered bay
Simba Wall	9	4.87575	39.67349	Fringing reef slope
Paradise	13	4.91282	39.67093	Fringing reef slope
Swiss	18	4.86786	39.67046	Fringing reef slope
Njao Gap	10	4.95911	39.66748	Tidal channel
Mandela	8	4.99694	39.65576	Fringing reef slope
Manta	10	5.00146	39.65607	Tidal channel
Fundo Inner	9	5.00924	39.66755	Tidal channel
Fundo Outer	10	5.04207	39.64161	Fringing reef slope
Fundo Lagoon	3	5.02569	39.67309	Sheltered bay
Msuka Bay	4	4.86164	39.7036	Sheltered bay
Misali	9	5.23958	39.5952	Fringing reef slope
Kokota	11	5.1374	39.63824	Tidal channel

diameter (10–20 cm, 20–40 cm, 40–80 cm, 80–160 cm, 160–320 cm and >320 cm). Corals smaller than 10 cm were measured in six 1 m^2 quadrats placed along the same transects. These smaller corals were also divided into size classes (0–2.5 cm, 2.5–5 cm and 5–10 cm). Size class data were collected for a restricted set of commonly occurring coral genera representing a range from low to high bleaching susceptibility as described in the literature for the region (McClanahan 2004; Obura 2001). The genera sampled were grouped into 4 distinct groups: Susceptible to bleaching stress (*Acropora*, *Pocillopora*, *Stylophora*, *Seriatopora* and *Montipora*), Resistant (*Porites* massive and *Pavona*), Moderate tolerance Faviidae (*Favia, Favites*, *Leptastrea*, *Echinopora* and *Platygyra*) and Moderate tolerance non-Faviidae (*Galaxea*, *Porites* branching, *Lobophyllia*, *Fungia*, *Hydnophora* and *Coscinarea*). Incidence of coral disease, predation or bleaching were also recorded along the transects and presented as a percentage of all colonies in the belt.

Fish surveys – Sampling was combined in one long swim, to maximize sampling of the large mobile fish (e.g. bumphead parrotfish), with 3 replicate transects for density estimates of fish. The long swim consisted of a 20 minute timed swim at a standardized swimming speed parallel to the reef axis. The area sampled was approximately 10 m on either side of the observer and only the largest size classes of key genera were recorded to genus level. Three transects were then undertaken using 50 × 5 m belt transects for the remainder of target families (Acanthuridae, Scaridae, Kyphosidae, Siganidae, Serranidae, Haemulidae and Mullidae), and all fish observed were recorded to genus level, their length estimated and placed into appropriate 5 cm size classes (5–10 cm, 10–15 cm, 15–20 cm, 20–25 cm, 25–30 cm, etc). Herbivorous fish were classified into different functional groups depending on their feeding modes and preferred diet (large excavators, small excavators, scrapers, grazers and browsers; Green and Bellwood 2009). Predatory fish which are commercially important and good indicators of fishing pressure were also surveyed.

Results and Discussion

The average live hard coral cover around the island was found to be 23%, with large variations from 86% in the Misali (a no-take area) to only 3% and 5% in highly degraded sites such as Paradise and Fundo Outer, respectively (Fig. 1). The overall trend shows further recovery from the 1998 bleaching event compared to the 16% live cover recorded in 2002; however, the figures are highly variable and specific comparisons for sites are often not possible (Obura 2002). One site that can be compared directly to previous literature is Misali, where live coral cover was recorded to 17% shortly after the 1998 bleaching (Muhando 2003), and was recorded to 86% in 2009. Coral reef conditions were highly variable, with some sites (Misali, Mandela and Manta) being dominated by hard coral, whilst others (Paradise and Fundo Outer) were dominated by rubble and turf algae. In total, 47 hard coral genera were recorded, with Misali having the highest diversity with 42 genera recorded and Paradise having the lowest diversity with 23 genera recorded (Fig. 2). *Acropora*, massive *Porites* and *Ecninopora* dominated the hard coral cover, accounting for 46% of coral area, while *Pocillopora* were by far the most numerous colonies, accounting for 24% of all coral colonies (Fig. 3). This corresponds to the life history strategies of the genera. *Pocillopora* is an early colonizer that reproduces quickly and colonizes disturbed environments but does not grow to a large size or old age compared to other genera. *Acropora* is fast-growing, susceptible to bleaching and typical for an undisturbed reef. At the time of the survey, only Misali and Mandela were dominated by *Acropora*, whilst the rest of the sites were dominated by more stress-resistant genera such as the slow-growing massive *Porites* (Fig. 4).

Coral recruitment is considered an important characteristic for driving ecological resilience of coral reef systems by allowing coral populations to replenish after a mortality event (Lukoschek et al. 2013; Nyström et al. 2008). This not only encompasses a plentiful larval supply and the successful settling of larvae at a site, but also post settlement

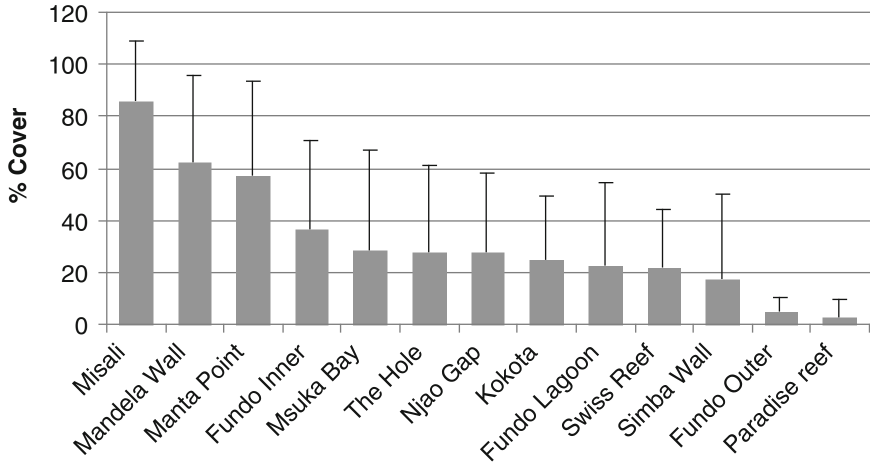


Fig. 1 Depicts live hard coral cover per site. Coral cover varied greatly from a high cover of 86% in Misali (no-take zone) to low cover of 3% in Paradise Reef and 5% in Fundo Outer, two highly degraded sites

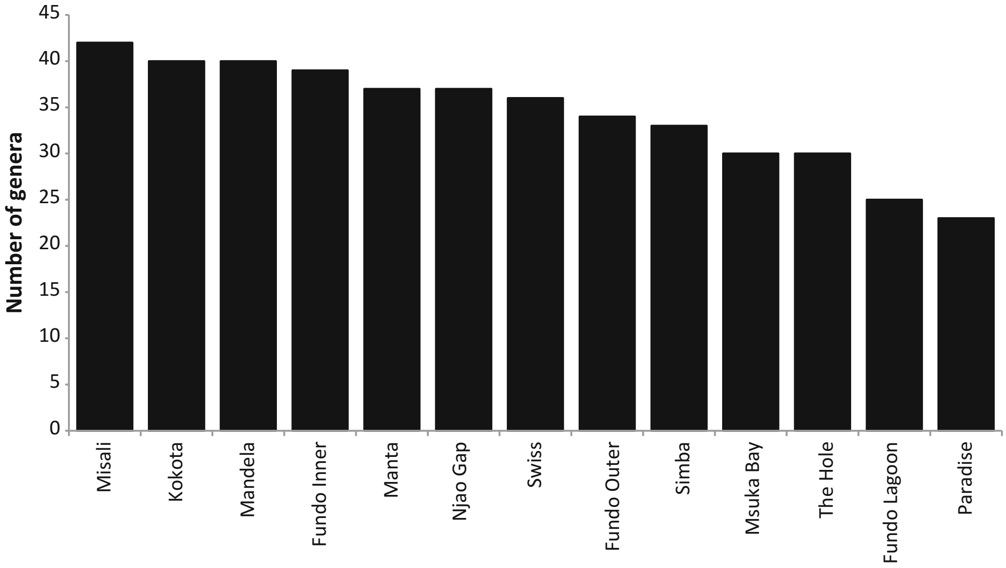


Fig. 2 Depicts coral diversity by site. The total number of coral genera found was 47, with a maximum of 42 found in Misali and a minimum of 23 in Paradise. Fundo Lagoon, The Hole and Msuka Bay also had relatively low diversity

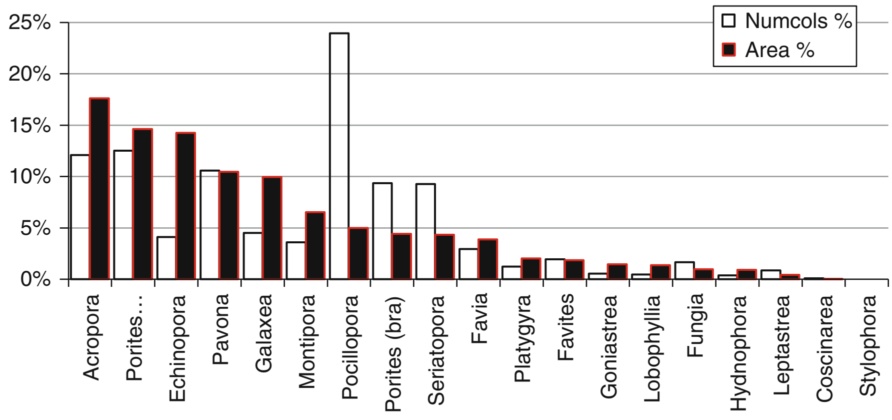


Fig. 3 Depicts the number of colonies and area covered by genus. Acropora, Porites massive and Ecninopora dominate coral cover, accounting for 46% of the total coral area, while Pocillopora is by far the most numerous genus, accounting for 24% of all coral colonies

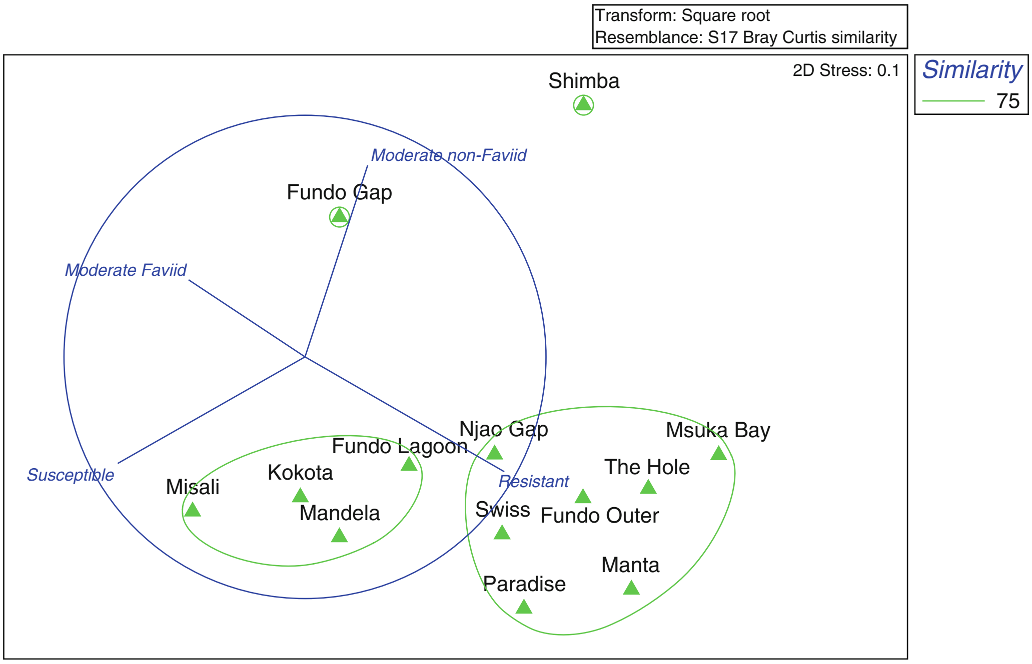
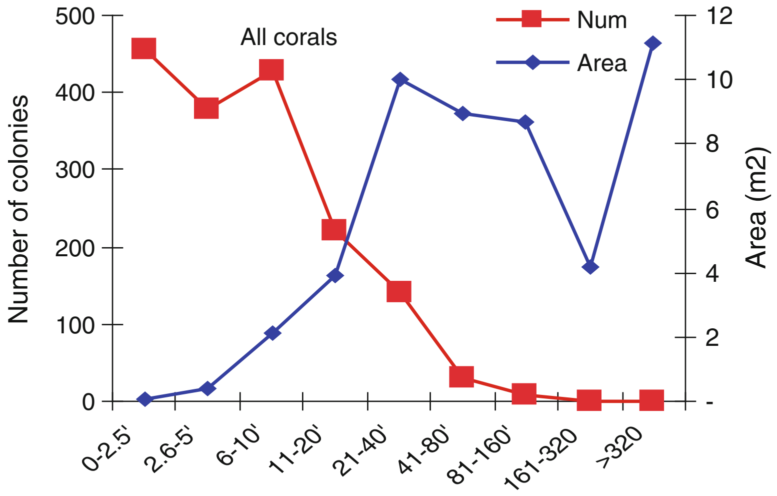


Fig. 4 Depicts genus susceptibility by site. Genera were categorized into groups depending on their bleaching responses. Four groups were identified – Susceptible (Acropora, Pocillopora, Stylophora, Seriatopora and Montipora), Resistant (Porites massive and Pavona), Moderate tolerance Faviidae (Favia, Favites, Leptastrea, Echinopora and Platygyra) and Moderate tolerance non-Faviidae (Galaxea, Porites branching, Lobophyllia, Fungia, Hydnophora and Coscinarea). The proportion of total coral cover occupied by each bleaching response group was calculated, and sites were compared using Multi-Dimensional Scaling analysis

Fig. 5 Depicts overall size class distribution for all coral colonies recorded. The distribution of size classes is shown by number of colonies, and by area of colonies for all size classes. On average, there were 1665 colonies in an area of 100 m^2, corresponding to 49.5 m^2 of coral colony surface. The dominant size classes by area, were >320 cm, 21–40 cm, 41–80 cm and 81–160 cm



survival and growth into juveniles and then adults (Ho and Dai 2014; Martinez and Abelson 2013). Coral size class distributions can be indicative of the history of mortality of reefs' coral populations (Zvuloni et al. 2008). Large-scale coral mortality events caused by bleaching or other factors are known to cause reduced fecundity and recruitment in coral populations (Hoegh-Guldberg 1999). Periods of mortality could thus be reflected in the size class structure of Pemba's coral reef community. This study found that the coral size class distribution of Pemba's reefs showed reduced numbers of corals sized 2.5 to 5 cm and 1.6 to 3.2 m (Fig. 5). The coral size class analysis was conducted both including and excluding *Pocillopora* colonies in order to ascertain the influence on coral size distribution exerted by *Pocillopora* given that it is the genus with the highest number of colonies and could disproportionately drive patterns; however, the same pattern was apparent even excluding *Pocillopora* from the analysis. The dip in the

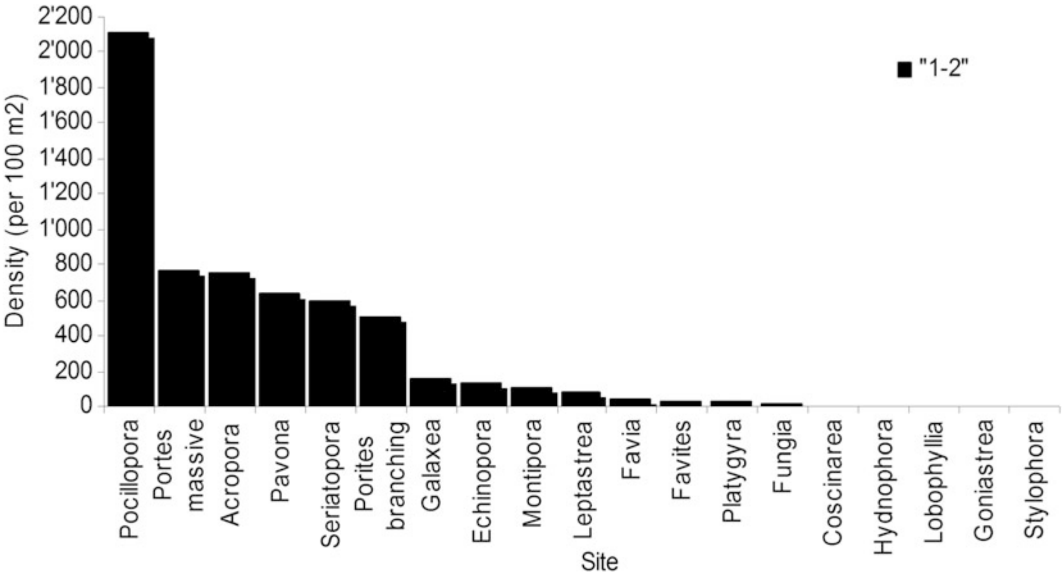


Fig. 6 Shows that recruitment by genus. Recruitment is largely dominated by Pocillopora, followed by Acropora, Porites massive, Pavona, Seriatopora and Porites branching

population of corals sized 1.6 to 3.2 m is probably indicative of a past mass mortality event, possibly the 1998 bleaching event, which is known to have greatly affected Pemba's coral populations (Muhando 2003). However, the dip in the population of corals sized 2.5 to 5 cm indicates more recent failures in recruitment and survivorship, and is probably related to recent 'pulse' stress events such as bleaching and *A. plancii* outbreaks or more ongoing 'press' mortality and stress from destructive fishing practices (Visram et al. 2008; Kayal et al. 2012; Fox and Caldwell 2006). This failure in recent recruitment and survivorship indicates that these stressors could further affect future recovery of Pemba's coral populations. Overall, coral recruitment is largely dominated by *Pocillopora* (Fig. 6), an early colonizer that can reproduce asexually via fragmentation as well as sexually through spawning and production of larvae.

Recruitment patterns also vary spatially, giving indications of recovery potential at different sites and for different reef types. We observed that in most sites the number of small recruits (0–2.5 cm) was healthy and comparable to results of other studies on recruitment in the region (Tamelander 2002); however, post-settlement mortality is not allowing recruits to survive into adults in degraded sites. It was found that coral recruitment was generally higher at degraded sites than in sites with higher live coral cover. A correlation analysis between coral cover and recruitment found a weak negative correlation between the two variables ($r = -0.16$), indicating that sites with higher coral cover can display lower recruitment. The sites with the highest recruitment rates were Simba, Fundo Outer and Paradise (670, 641 and 623 corals sized 0–2.5 cm per 100 m^2 per site, respectively), but these sites had the lowest coral cover (18%, 5% and 3%, respectively) and were instead dominated by turf algae. The sites with the highest live coral cover, i.e. Misali, Manta or Mandela (86%, 59% and 62% respectively) were all found to have relatively low recruitment rates (385, 418 and 572 corals sized 0–2.5 cm per 100 m^2, respectively), possibly because less suitable substrate was available for coral larvae to colonize as the area has already being occupied by larger corals (Fig. 7). This indicates that the low coral cover in the identified degraded sites (Simba, Fundo Outer and Paradise) was not due to a lack of larval supply and recruitment but rather to a local stress on the site that can have caused mortality of young corals and not allowing them to grow into adults.

Two sites stand out as having relatively low recruitment; Msuka Bay and Fundo Lagoon (Fig. 7), two sheltered and shallow sites with 58 and 85 corals sized 0–2.5 cm per 100 m^2, respectively. This is possibly due to their geographical positioning and substrata. Msuka Bay is located at the north of the island, not well connected to the dominant currents in the area, and is characterized by high wave energy and dominated by *Sargassum* (35% *Sargassum* cover recorded) where it is difficult for coral recruits to settle and grow. Fundo Lagoon is located in a sheltered bay area, and although there is tidal exchange of water, it appears that coral larvae do not successfully settle there. The case of Fundo Lagoon is interesting given that the tidal flushing and exchange is high, so it could be assumed that coral larvae do reach the site. However, Fundo Lagoon was observed to have the highest sediment- and turbidity load of all sites surveyed, and this could be related to lower settlement and survival of small recruits.

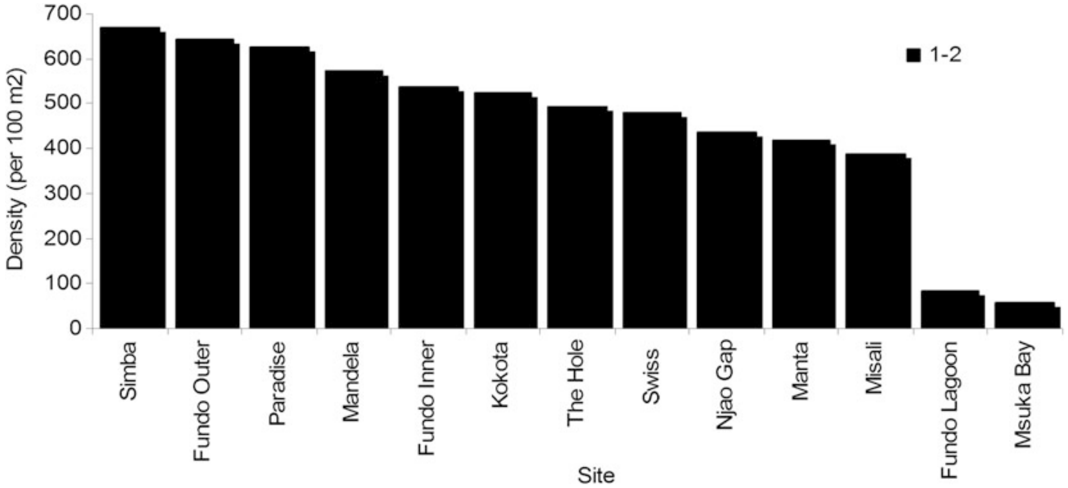


Fig. 7 Depicts coral recruitment. Simba, Fundo Outer, Paradise and Mandela had the highest coral recruitment (0–2.5 cm size of corals) despite having lower coral cover than sites such as Misali, Mandela and Manta

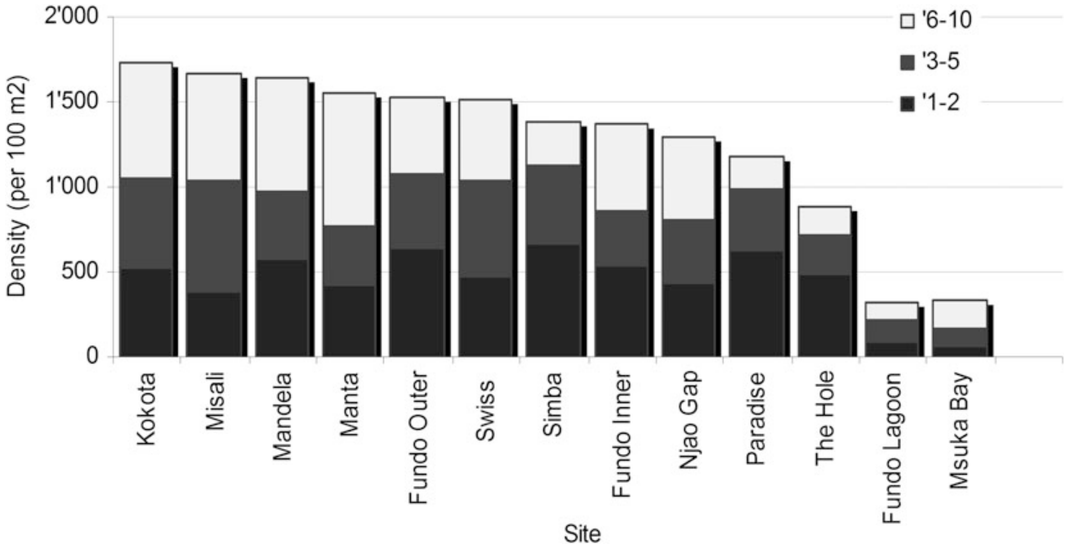


Fig. 8 Depicts the density of smaller corals at different size classes (0–2.5 cm, 2.5-5 cm and 5–10 cm)

The findings suggest that in degraded sites such as Paradise, Simba, Fundo Lagoon and Fundo Outer, lack of coral recruitment is not driving the lack of recovery, but rather that poor survivorship of coral recruits does not allow the reef to regain coral cover and diversity. This finding is further accentuated when we examine the fate of the recruits as they grow older. Although recruitment and the number of corals smaller than 2.5 cm may be higher in degraded sites such as Paradise, Simba and Fundo Outer, the data show that recruit survivorship was much lower than in other sites, meaning that the number of corals sized 2.5–5 cm and 5–10 cm decreases drastically in these degraded sites. If this trend continues, fewer and fewer corals attain larger sizes at these sites (Fig. 8). Figure 9 shows the mortality rates of small corals from the 0–2.5 cm size class to the 5–10 cm size class. Degraded sites such as Paradise, Simba and Fundo Outer exhibited high mortality rates (69%, 62% and 31% respectively), while most other sites did not display such high rates of recruit mortality and instead high survival rates were observed. It is thus apparent that local stress factors and mortality of small corals are eroding resilience in the degraded sites, rather than a lack of larval supply.

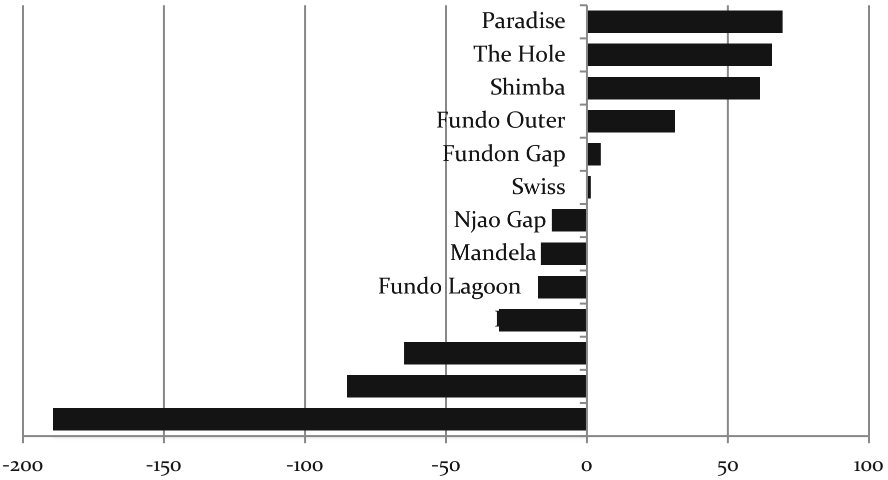


Fig. 9 Depicts the mortality rates (%) of small corals in different sites. Positive numbers mean high mortality, while negative numbers mean low mortality and high survivorship. The figure presents mortality rates from the 0–2.5 cm size class to the 5–10 cm size class

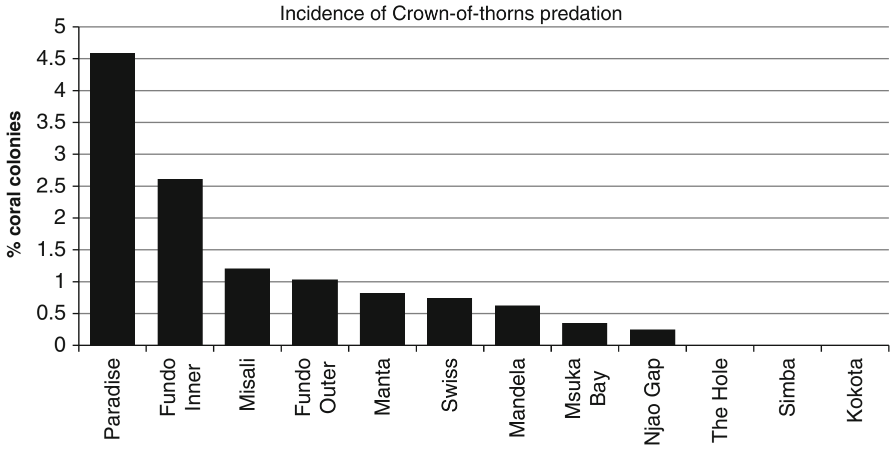


Fig. 10 Shows predation on corals by crown-of-thorns starfish. Paradise had the highest incidence of crown-of-thorns starfish feeding scars observed on coral colonies, with 5% of colonies being predated

Plausible explanations include predation by *A. plancii*, overfishing, destructive fishing and/or further bleaching events since 1998.

At Paradise the high incidence of *A. plancii* predation scars on corals (5% of all colonies surveyed were predated compared to 2.5% at Fundo Inner or 1.2% at Misali, the two sites with the second and third highest incidences of coral predation; Fig. 10) indicates that the corallivores are one cause of coral recruit mortality. An *A. plancii* outbreak was observed at Fundo Inner, with over 50 individuals sighted in one dive. It is unknown what the trigger for population outbreaks is. Removal of predators of *A. plancii* (e.g. triton shells or triggerfish) through overfishing, improved survival of larvae due to land-based nutrient inputs and increasing sea-surface temperature have all been postulated as potential triggers of *A. plancii* outbreaks (Moran 1986; Uthicke et al. 2009). Although no outbreak was observed at Paradise, it still exhibited the highest percentage of feeding scars. Paradise had the lowest coral cover of all sites, and the coral population comprised mostly of smaller sized corals (no corals larger than 40 cm in diameter were observed, and total coral area mostly consisted of colonies of 11–20 cm in size). These small corals were being predated opportunistically by *A. plancii*, showing that even in the absence of an outbreak, the corallivores were causing mortality of coral recruits and exerting pressure on the coral population. A correlation analysis between survivorship of smaller coral size classes and *A. plancii* predation across all sites showed a weak negative correlation between survivorship of the 0–2.5 cm coral size class and *A. plancii* predation ($r = -0.16$), while a moderate negative correlation between survivorship of the 2.5-5 cm coral size class and *A. plancii* predation ($r = -0.32$) was found. *A. plancii* predation thus does appear to be driving mortality of smaller corals to some extent, and more so when corals attain sizes larger than 5 cm.

Another major cause of ecological degradation in Pemba is overfishing and destructive fishing. The data presented

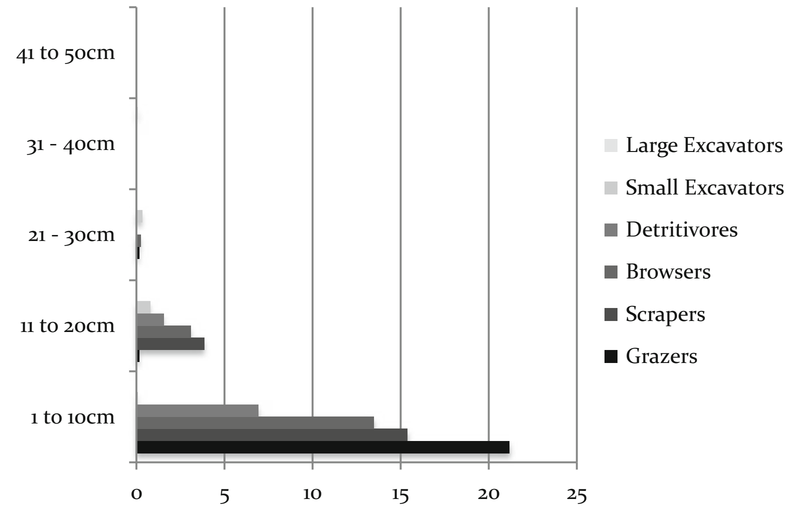


Fig. 11 Shows the overall abundance of herbivores (separated into functional groups) in Pemba according to size class. The vast majority of herbivorous fish on Pemba seen were <10 cm long, indicating overfishing. The absence of large fish is ubiquitous across predators and herbivorous functional groups

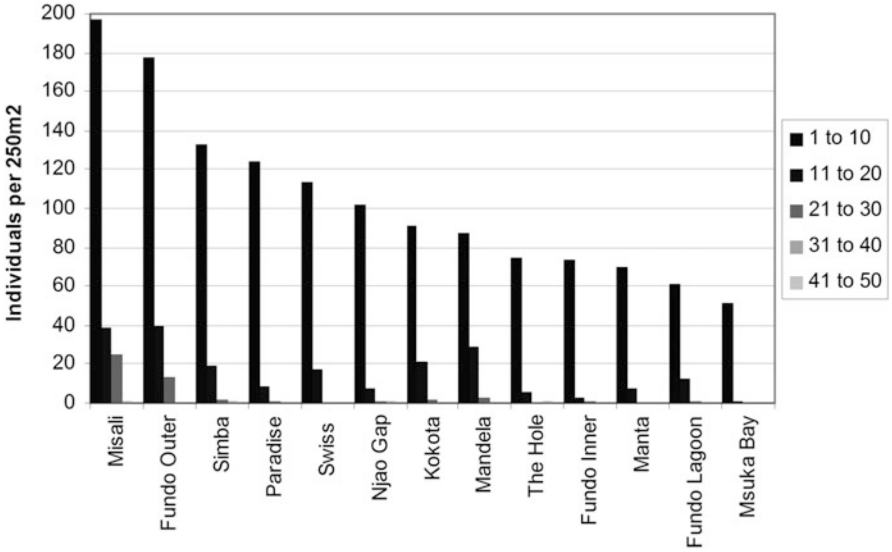


Fig. 12 Shows fish abundance for all species of fish (herbivores and predators) recorded. Misali, which is a no-take zone, had the highest number of large bodied fish, showing that management can impact fish populations

indicates that the fish populations in Pemba are being harvested aggressively. Fish populations in Pemba varied greatly among sites surveyed, from over 250 individuals of all species per 250 m^2 observed in Misali to 50 individuals per 250 m^2 observed in Msuka Bay for all species measured. These fish abundance numbers are comparable to studies from Kenya with the lower numbers comparable to non-protected areas and the higher numbers comparable to protected areas (Munga et al. 2012). Small-bodied herbivorous Acanthuridae and Scaridae were the most common fish observed. Very few (<5 individuals per 250 m^2) were large excavators or commercially valuable Serranidae, Haemulidae or Mullidae, and no sharks were seen; this indicates the overfishing of large bodied predators, herbivores and commercially valuable species. Within the herbivorous functional groups, grazers and scrapers were the most abundant, followed by browsers, while small excavators (<35 cm) were twice as abundant as large excavators. *Bolbometopon* spp., which is a large excavator, was absent (Fig. 11). Again, this is an indication of overfishing, as large bodied excavators are usually among the first to disappear on an overfished reef. Misali, the no-take zone, exhibited the highest fish densities (Fig. 12) but overall the vast majority of fish observed on Pemba were <10 cm long, and only four individuals larger than 40 cm were recorded during the entire survey (Fig. 11). Herbivorous fish are important in regulating algal-coral dynamics, and healthy herbivore populations are important for coral reef resilience as algae can outcompete coral recruits and reduce their survivorship (West and Salm 2003). Evidence of frequent and/or continuous destructive fishing was also evident from anecdotal visual observations. Illegal beach seines have routinely being used in the area and dynamite blasts were regularly heard during dives. These indiscriminate destructive methods not only destroy coral habitat but also remove sexually immature juveniles as well as rarer species. At this point, however, there is insufficient data to correlate overfishing and destructive fishing pressure with coral recruitment and survivorship, and more targeted studies are needed in this area for Pemba. Nevertheless, Paradise and Fundo Outer serve as warnings of what currently ‘healthy’ sites could look like if ecological resilience continues to be eroded through destructive practices.

Land-based sources of pollution were not observed to have major impacts on Pemba’s coral reefs. There is very little coastal development, with only few hotels and fishing villages on the coast. However, nutrients from the seaweed farms that are common in Pemba could be contributing to algal growth. Macroalgal cover was found to be high, with an average of 9% of the sites covered by macroalgae (mostly *Dictyota*, *Cyanophyta*, *Sargassum* and *Jania*), and an average of 28% turf algae cover. Lack of herbivorous fish and grazing is probably the major cause of the high macroalgae cover, with some influence from nutrients; however, more data is needed to understand the cause-effect relationship. Higher nutrient levels have also been linked to *A. plancii* outbreaks due to improved larval survival (Brodie et al. 2005), but there is no evidence of this in Pemba and more research is needed.

Despite the degradation observed in many sites, we have continued to observe live coral cover increasing compared to other surveys since 1998, indicating that some coral populations on parts of the island are able to recover. For example, Misali is currently a no-take zone that has been offered higher protection than other sites surveyed, and it has exhibited significantly healthier populations of both coral and fish. Currently, degraded sites such as Paradise and Fundo Outer could, in theory, also recover to the extent of other sites if the stressors currently causing mortality of recruits and juveniles are identified and, if possible, eliminated. Some of the major stressors found to be driving mortality of coral recruits were *A. plancii* predation, the use of destructive fishing methods and overfishing. Dynamite fishing is illegal in Tanzania with a penalty of at least a 5-year prison term according to the Tanzania Fisheries Act of 2003, yet there was frequent evidence of its use in Pemba. Fishing activity using gill nets and beach seine was also observed in supposedly protected areas. We recommend that these anthropogenic stressors should be monitored and managed more closely in order to avoid further degradation of the coral reefs of Pemba.

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