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"Comparison of damage caused by beach seining vs. corallivory on coral transplants in Mombasa, Kenya"

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TABLE OF CONTENTS

Comparison of damage caused by beach seining vs. corallivory on coral transplants in Mombasa, Kenya

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ABSTRACT: Two coral species were transplanted into three distinct management areas adjacent to the Mombasa Marine National Park, Kenya: a no fishing MPA, a gear restricted reserve with no beach seining, and a reserve with beach seining. Corallivory from fish or breakage from beach seining were measured by percentage mortality of transplants. The branching species, *Porites cylindrica* was more susceptible to disturbances than the massive species *Porites lutea* which showed no difference in mortality rate between the three sites. Branching *Porites* were more affected by corallivory than by beach seining or by restrictive gear fishing. Corals transplanted into the gear restricted fishing site had the highest survival, although in the past, this site has shown signs of overfishing and phase shifts towards a sea-urchin dominant community. Branching coral transplants suffered the most damage inside the no fishing MPA, however, in the long term corals may suffer more in the two other areas through direct impact by beach seining and indirect impacts from overfishing. Extended experiments of longer duration should be conducted to differentiate between long and short term effects and the rates of recovery for each management strategy.

KEY WORDS: Beach seining · Corallivory · Transplantation · Marine Protected Areas · Kenya

INTRODUCTION

With a growing coastal population of over 1.83 million people, Kenya's marine resources are under increasing pressure from overexploitation (Obura 2001). Although Kenya possesses four marine parks, some of them effective for more than 20 years (McClanahan &

Mangi 2001), a large extent of its reefs are not regulated and destructive fishing gear such as beach seining are being used. Kenyan beach seines, are nets approximately 150m in length with a mesh of approximately 3 cm. They are directly detrimental to fish populations due to their mesh size and non-selectivity (Jennings & Kaiser 1998). This has resulted in the decline of a wide number of species, causing concern to both ecologists and fishermen.

Fish density and diversity are positively correlated with reef complexity and live coral cover (Bell & Galzin 1984, McClanahan & Shafir 1990, McClanahan 1994a, Chabanet et al. 1996, Jennings & Polunin 1996). Beach seining involves dragging the weighted nets across seagrass, sand and coral reef substratum causing direct physical damage to the reef substratum and therefore indirectly impacting fish stocks (Jennings & Polunin 1996, Jennings & Kaiser 1998, McClanahan & Mangi 2001).

Indirect effects of fishing can cause greater damage to corals than direct physical impacts. McClanahan (1994a) describes two terminal states of degradation of coral reefs: one where the reef is dominated by large fleshy algae and the second by abundant sea urchins. In both cases, percentage live coral cover typically declines (Jennings & Kaiser 1998). This is described as a "phase shift" (Done 1992) of the benthic community, away from a coral dominant community. Phase shifts are often the consequence of removal of sea-urchin predators and herbivorous fish (McClanahan & Shafir 1990).

MPAs have been advocated as the most effective management tool because they may offer types of protection not provided by other management strategies: specific protection of critical areas, intrinsic prevention of overfishing and even enhancement of fisheries (Allison et al. 1998). The Mombasa Marine National Park (MMNP) has shown a greater abundance in fish and an increase in live coral cover since its establishment (McClanahan & Arthur 2001). However this can have secondary effects by increasing corallivorous fish which may result in the exclusion of certain species of corals (Cox 1986). Additionally parrotfish may reach high population densities and eliminate entire coral colonies (Miller & Hay 1998).

It is increasingly difficult to separate damage from the different pressures reefs are subjected to. However, in order to use the right management tools and design useful strategies for reef protection and recovery it is essential to understand the ramifications of particular management strategies.

Past studies have measured the impact of beach seining by recording the decrease in catches and monitoring catch weight (McClanahan & Mangi 2001, McClanahan & Arthur 2001, McClanahan et al. 1999). Fewer studies have measured the physical impact on corals. Comparison of substrate between protected and fished reefs have been made, but other effects must be taken into account when interpreting the results since health of corals is dictated by complex interactions. Dusek (2000) attempted to measure the damage from beach seining on existing coral colonies but encountered difficulties by neglecting to take into account the possibility that some corals may have already been excluded from sites (Dusek 2000).

The aim of this experiment is to use coral transplants and experimental cages to measure the direct impact of beach seining on corals in comparison with the damage suffered by corals inside a marine park with no fishing, and a marine reserve with high fishing pressure but restricted gear and no beach seining. The short and long term effects of the three different management strategies on corals are discussed.

MATERIALS AND METHODS

Description of experimental sites

[Fig. 1] The study was conducted on the northern shores of Mombasa City, Kenya (Fig. 1). All three sites were situated in back reef lagoons with shallow water (< 3 m) at low tides (Kenya has a 4 m tidal range) dominated by hard substratum colonized by corals and other benthic invertebrates and algae (McClanahan & Shafir 1990).

A "non fishing" site was located in the Mombasa Marine National Park. This park was established in 1989, and effective enforcement began in 1991. It has been categorized as a transition park in most literature due to both its age and fish community characteristics (McClanahan 1994a, Carreiro-Silva & McClanahan 2001, McClanahan & Arthur 2001). Like the majority of Kenyan coral reefs, it suffered extensive coral mortality during the 1998 El Niño related bleaching event (McClanahan et al. 2001).

A "restricted fishing" site was located at Ras Iwatine, in the Mombasa Marine National Reserve. Although fishing there has recently been limited by excluding beach seining, this site is exposed to high gear restricted fishing intensity (McClanahan & Mangi 2001).

A "beach seining" site was located offshore of Nyali Beach (Fig. 1). This site is still within Mombasa National Marine Reserve but lacked enforcement (Mangi pers. com.). All fishing gears were used at this site. The transplantation area was selected according to two criteria: repeated observation of beach seining and presence of hard substrate and coral heads.

Reef Health

Benthic substratum cover of the dominant benthic life forms was estimated by the line-intercept method using loosely draped 10 m line transects (McClanahan & Shafir 1990). Ten transects were laid at each site. Previous monitoring data was used for the MMNP and Ras Iwatine sites. From these measurements the percentage cover of the various categories was estimated for each reef.

Transplantation

Donor site. Two coral species, branching *Porites cylindrica* and massive *Porites lutea* were collected from Kanamai reef, a lagoonal patch reef 20 km north of Mombasa. Two morphological types of the massive species were collected: "knobby" and "smooth" (Fig. 2
Fig. 2]
[Fig. 3] and 3). Different size fragments ranging from finger size to large heads were broken with a chisel and a hammer. They were directly placed in buckets with sea water. On the same day they were transported by car and boat to MMNP and placed under a large cage without being fixed to the substrate. Transport never took more than one hour and a half (40 minutes by car, 20 minutes by boat). Coral showed no signs of stress apart from the production of mucus.

Transplantation. 270 corals were transplanted at the designated three sites. The corals were randomly selected from the broken colonies placed originally in one large cage. They were set by pairs, one branching and one massive species, in "blocks" of seven pairs
Fig. 4] (Fig. 4). There were six "blocks" in MMNP and Ras Iwatine and nine "blocks" at Nyali. The substrate chosen to support the transplantation was either hard substrate or dead coral colonies and was brushed free of algae. Cement was used to fix the corals to the substrate. The

morphological type of massive species was chosen randomly. Each fragment had 90 - 100% live cover at the time of attachment.

Due to low tide restrictions, corals were kept in large cages and were moved to the next site when one site was finished. Transport between sites never exceeded 40 minutes by boat, during which corals were placed in buckets of water.

Cages. Plastic mesh (2.5 cm) was used to build control cages. Each cage was prism shaped (length = 60 cm, height = 21.6 cm). Larger cages were made of the same material and were on average 50 cm long and 35 cm high. Cages were fixed to the reef snbstrate with U-shaped nails. One cage housed one pair of corals. This was repeated for every seven pair of corals or one cage per "block" (Fig. 4). The cages were fixed immediately after transplantation and were brushed twice a week to prevent algal overgrowth and checked for looseness and repairs.

Data collection. Metal callipers were used to take measurement of longest length, longest perpendicular width and greatest height of live tissue of transplants. This was done the day after transplantation at each site and again the last day of monitoring. Percent live cover was estimated for each coral of each species and the type of damage was recorded into seven categories: damage from transplantation, corallivory from fish (predation), predation from corallivorous gastropods, predation from *Acanthaster planci* (COT), competition with sponge, smothering by sand and damage from beach seining. Measurements were taken twice a week in MMNP and at Ras Iwatine and three times a week at Nyali. Experiments were ended 57 days after day 0.

Statistical analysis. Due to the short time allocated to this study, replication of the three management strategies could not be repeated in order to have replication of treatments. Replication within sites are therefore treated as pseudoreplications and the experiment as a case study.

For all analyses, branching *Porites* were treated separately from massive *Porites*. Controls (caged corals) were not included in any of the statistics except when comparing with the experimental corals (non caged). Measurement of the percentage live coral cover loss between the first measurement and the last measurement (100-((% live cover last day - % live cover 1^{st} day)*100) was used as a measure of "percentage mortality". The average for blocks was taken for each species at each site. This allowed the assumption of normal distributions according to the central distribution theory (ZAR 2000). Nevertheless, normality tests were furthermore carried out and transformations were applied in case of strong deviation from normality.

A one-tailed non parametric Mann-Whitney U test was carried out to find if percentage mortality for the controls (corals in cages) was smaller than for the experiment (corals outside cages). The design of the experiment was too unbalanced, one caged pair versus six non caged pairs, did not allow to carry out an ANOVA using averages per block. Furthermore, the data was not normal, even after transformation and the non-parametrical equivalent of the t-test was used.

Difference of percentage mortality between sites for the experiments was tested with a one-way ANOVA. A post hoc Tukey test was carried out.

A 2-way ANOVA with species and sites as crossed factor was carried out to test for interactions.

Differences in the relationship between size and percentage mortality were tested with an ANCOVA by putting size as a covariate. Corals were assimilated to cubic volume and size was calculated as a volume by multiplying the measures of length, width and height taken on the first measure. Natural log transformations were performed on both size and percentage mortality for normality.

Difference of percentage mortality between the two types of massive *Porites* (knobby and smooth morphologies) was tested first for each site with a one-way ANOVA, then with a 2-way ANOVA with type and location as crossed factors. For both tests, a natural log transformation was carried out for percentage mortality.

Comparison of the types of damage between sites was made using the Bray-Curtis Index of similarity and MDS analysis. The data for percentage mortality was 4^{th} root transformed. ANOSIM test was also carried out on the rank similarity matrix to test for significant difference between sites.

RESULTS

Controls vs. Experiment

The Man-Whitney U comparison of percentage mortality inside and outside cages for both MMNP and Ras Iwatine, where there was no effect of beach seining, showed a significant difference (MMNP/ branching $n = 6$, $W = 51.0$, $p = 0.033$, massive $n = 6$, $W = 55.5$, $p = 0.005$; Ras Iwatine/branching $n = 6$, $W = 55.0$, $p = 0.006$, massive $n = 6$, $W = 51.0$, $p = 0.033$). Corals outside the cage suffered significantly higher percentage mortality than corals inside cages. These controlled successfully for predation.

There was no significant difference between controls and experiments for both branching ($n = 9$, $w = 92.0$, $p = 0.298$) and massive ($n = 9$, $W = 100.0$, $p = 0.108$) in Nyali. This can be explained by the fact that 4 cages in Nyali were ripped out by beach seining and either broke or ripped out the controls (only the base was left). Cages were a non effective means of control against beach seining and a similar mortality rate was found for corals inside and outside cages.

Percentage Mortality vs. Location and Species

There was a significant difference of percentage mortality between sites for branching *Porites* (ANOVA df = 2, F = 11.06, p = 0.001). MMNP suffered the most damage with 39.48% (± 4.93) mortality overall, then Nyali with 31.00% (± 6.49) mortality and finally Ras Iwatine with 16.75% (± 6.37). The main difference lies between MMNP, and Ras Iwatine: as
[Fig. 5] Fig. 5 illustrates, the confidence intervals (C.I.) bars of MMNP and Ras Iwatine do not overlap.

There was no significant difference in percentage mortality between sites for massive *Porites* (ANOVA df = 2, F = 0.71, p = 0.506), with all sites suffering similar mortality: MMNP-9.1% (± 5.31), Nyali-13.55% (± 5.35), Ras Iwatine-13.52% (± 6.36). As Fig. 5 illustrates, the C.I bars overlap for the three sites.

The ANOVA showed that there was a significant interaction effect of species with site (df = 2, F = 8.41, p = 0.01). The interaction term may be due to the response of branching transplants to the different types of damage between sites. Furthermore, massive *Porites* reacted differently to disturbances than branching *Porites*, overall suffering less damage.

Measurement of percentage live cover was subjective to the observer, since the same observer took all the measurements throughout the experiment the errors averaged out. Frequent observation allowed any inconsistencies to be revealed. Finally caged corals were referred to for comparison of branching *Porites* to help estimating the loss.

Effect of Size

There was no significant correlation between percentage mortality and size for either branching (ANCOVA df = 2, F = 0.42, p = 0.524) or massive *Porites* (ANCOVA df = 2, F = 0.31, p = 0.584) (Fig. 6 and 7) thus no comparison of differences in the relationship between size and percentage mortality could be carried out. This is a surprising result: percentage mortality should be related to size since the same amount of grazing for example on a large coral will cause less damage than on a small coral. Size range may have to be larger than the one selected in order to detect a relationship.

[Fig. 6]
[Fig. 7]

Knobby vs. Smooth

There was no significant difference in percentage mortality between morphological types of massive *Porites* in both Nyali (ANOVA df = 1, F = 0.53, p = 0.478) and Ras Iwatine (ANOVA df = 1, F = 0.01, p = 0.935). However, there was a significant difference between knobby and smooth massive *Porites* in MMNP (ANOVA df = 1, F = 8.18, p = 0.019). This is illustrated in Fig. 8 by the overlapping C.I bars.

[Fig. 8]

This result should be carefully interpreted as there were more knobby massive *Porites* transplanted in Nyali than any other site. Furthermore all sample sizes were small which increased type 1 error of not successfully rejecting the null hypothesis. Further experiment should be carried out in order to study the effect of predation on the different morphology of massive *Porites*.

There was no interaction effect between sites and type of massive *Porites* (ANOVA df = 2, F = 1.92, p = 0.162). It was expected that knobby morphologies would show higher damage from predation than smooth morphologies especially in MMNP. The lack of interaction may, again, be due to the small sample size and the unbalanced proportion of knobby transplants in Nyali.

Difference in Types of Damage

Nyali was the only site exposed to beach seining. There was a significant difference in the type of damage between sites for branching *Porites* (ANOSIM R = 0.36, p = 0.001). The [Fig. 9] MDS plot (Fig. 9) illustrates the difference in composition of damage which is significant for [Table 1] the three sites (Table 1, ANOSIM pairwise test). The main types of damage were predation, predation by snails (only in MMNP) and beach seining (Fig. 10). Branching Porites seem to be good indicators of disturbances of the three different management strategies.

Damage from predation was significantly different between the three sites for branching *Porites* (ANOVA df= 2, F = 5.24, p = 0.016), with MMNP being most affected with 31% mortality, then Nyali with 24,32%, and Ras Iwatine with 10.7%. This difference was significant between MMNP and Ras Iwatine but not between Nyali and MMNP (Tukey [Fig. 10] post hoc, p = 0.05). Fishing was only responsible for 3.19% mortality (Fig. 10). Damage from beach seining was therefore less important than predation damage at any of the sites. Although beach seining occurred systematically when monitoring over two sites in Nyali, the frequency of actual impact with corals may have been low. However, when damage did occur it was important. This is compared to frequent, small damage by fish. In a short span of time, although fish damage may well be statistically represented, damage from beach seining will not. Measures will underestimate direct impacts of beach seining.

There was no significant difference in the type of damage causing mortality between Fig. 11] sites for massive *Porites* (ANOSIM R = 0.088, p = 0.118). The MDS plot (Fig. 11) illustrates the difference in composition of damage which was non significant between the three sites (ANOSIM pairwise test). Predation damage was not significantly different between the three

[Fig. 12] sites for massive *Porites* (ANOVA df = 2, F = 1.73, $p = 0.205$) (Fig. 12). Massive *Porites* are poor indicators of disturbances.

Benthic cover: absolute coral cover, diversity and species richness

There was a greater coral abundance in MMNP than in Nyali or Ras Iwatine with a coral cover of 23.2%, 14.0% and 6.2% respectively, although this has not been statistically tested. There was also a greater number of species counted at MMNP (17.3 species/ 100 m^2) than in Nyali (13 species/100 m^2) or Ras Iwatine (13 species/100 m^2). Shannon index was calculated for the three sites. Diversity for MMNP, Nyali and Ras Iwatine were very similar with 0.812, 0.7 and 0.8 respectively. These results have not been tested statistically for differences between sites, but may give a general idea of the coral at each site.

DISCUSSION

Massive vs. Branching corals

Branching morphologies are usually used in regeneration experiments for two reasons: they present a life history with high asexual reproduction by fragmentation (Highsmith 1982, Hughes 1985, Bruno 1998), and have rapid growth and regeneration (Done 1982, Karlson & Hurd 1993). They are also more fragile than other morphologies, often suffering the more from damage from hurricanes and wave action, bleaching (McClanahan 2000), COT predation (Brown 1997, Edinger & Risk 2000) or human disturbances such as diving (Rouphael & Inglis 1997, 2002, Hawkins et al. 1999, Jameson et al. 1999), fishing (Edinger & Risk 2000), and transplantation (Edwards & Clark 1998) than massive morphologies. However, in this study the two species did not show difference in percentage mortality due to beach seining. This is most likely a result of the small sample size and short time period of the experiment. Only a few corals were broken by beach seining and did not allow distiuction between any effects of morphologies. The vertical arborescent structure of branching *Porites* would more likely be caught and ripped out or damaged by seine net than a horizontal structure close to the substrate.

Branching *Porites* were more susceptible to fish predation than the massive species. Massive corals are therefore recommended for transplantation. This shift in strategy is supported by Edwards and Clark (1998) who argue that there has been too much focus on transplanting fast-growing branching corals to create short-term increases in live coral cover, at the expense of slow-growing massive corals. However, massive morphologies such as *Montastrea annularis*, *Montastrea faveolata* and *Colpophylla natans* seem to be the favoured food of parrotfish (Bythell et al. 1993, Bruckner & Bruckner 1998). Therefore, in areas where there are large parrotfish or wrasses, massive *Porites* may be more affected by predation than the branching morphology.

corals (Hixon 1997, Miller & Hay 1998). This suggests greater damage from predation would occur inside MMNP. However, the experiment did not reveal any significant difference of mortality between sites or the type of damage from predation between MMNP and Nyali.

The corals in Nyali were exposed to smaller fish than in MMNP (McClanahan & Mangi 2001). In MMNP, parrotfish "spot biting" (Bruckner & Bruckner 2000) was often [Fig. 16] recognizable on the transplants, especially the massive species (Fig. 16). Fish bites were less recognizable in Nyali and had possibly been made by smaller fish. During this study, no large fish were observed in Nyali (pers. obs.), as it is often the case in over exploited reefs (Jennings et al. 1998). This leads to the hypothesis that although fish were smaller in Nyali, they concentrated on the transplanted corals due to the lack of corals naturally present there, elevating the predation level. This compares with MMNP where fish were larger and more abundant but had more choice. A parallel case may be found in literature with macro algae. Williams and Polunin (2001) designed an experiment where they reduced the area of macroalgae available for grazing. Within three months algae cover had been reduced by 25%

in areas where they had simulated a 25% increase in coral cover compared to control plots. This area attracted a greater abundance of fish. The bloeks of transplanted corals in Nyali may have had a similar effect of a coral "oasis", concentrating a high number of fish in those specific areas.

Branching coral transplants suffered the least damage in Ras Iwatine. Unlike the preceding two sites, very few fish were observed predating on or near the transplanted corals (pers. obs.). On the other hand, a large number of sea urchins were observed near or on coral transplants. Ras Iwatine was qualified as an intensively fished reef (McClanahan 1994a, McClanahan & Arthur 2001) and has showed signs of a "phase shift" (McClanahan & Shafir 1990). It presented a high density of urchins compared to non fished sites such as Mombasa Marine National Park. Phase shift towards sea urchin dominated reefs, can effectively maintain low fish population on overexploited reefs (McClanahan 1994, McClanahan & Arthur 2001, McClanahan & Mangi 2001), especially parrotfish (McClanahan 1994a). Sea urchins are grazers and are usually responsible for the bioerosion of coral skeleton once corals are dead and overgrown by algae (Jennings & Kaiser 1998). The low fish intensity explains the low predation level and overall low damage level at this site.

Invertebrate Corallivores

In the last weeks of monitoring *D. cornus* was found oo one block of transplants
.Fig. 17| predating on branching *Porites* transplants (Fig. 17) in the vicinity of a large patch of dead *Acropora* in MMNP. There were three to four snails on each branching coral and they killed 60 % each corals, mostly at the base. This coincides with previous observation of *D. cornus* preferably on the genus *Acropora* and the family *Pocilloporidae* (McClanahan 1994b, Turner 1994a, Turner 1994b) and on damaged reefs (Turner 1994a, 1994b). In the past damage from

Drupella outbreaks have been compared to damage from COT outbreaks (Cumming 1999). Reports of mass mortality due to this snail have been recorded in Western Australia and Japan (Turner 1994a, 1994b). Outbreaks have been in part attributed to overfishing and the removal of key predators of the snail (Turner 1994a, 1994b, McClanahan 1994b). Further investigation on *Drupella* outbreaks should be considered to improve future management.

In the long term.....

It is necessary to consider both the direct and indirect impact of beach seining to get a broader picture of its effects on corals. Damage from beach seining resulted in the death of the coral colony and the loss of its structural complexity. Dead colonies have been found to maintain their function as habitat for fishes and facilitate larvae and juvenile settlement (Lindahl et al. 2001). Evidence from hurricane damage and shipwrecks (Lindahl 1998, Bowden-Kerby 2001, Precht et al. 2001) shows that the rubble created by beach seining will only cause further damage to neighbouring corals. Furthermore, if high densities of fish in MMNP are responsible for damage through predation, they also maintain coral dominance. Miler and Hay (1998) argue that this may be a fine balance for certain species of corals, nevertheless there is a strong correlation between high coral abundance and fish density (Bell and Galzin 1984, Chabanet et al. 1997), and specifically in Mombasa Marine National Park (McClanahan & Arthur 2001). Grazing fish are essential to maintain low algae densities and allow coral growth and reproduction. In environments like those in Nyali or Ras Iwatine with low densities of grazers or sea urchin dominant communities, damaged corals are likely to be quickly overgrown by algae and experience further tissue loss (Carreiro-Silva & McClanahan 2001).

Rate of calcification plays an important role in assessing the success of coral transplantation (Jennings & Kaiser 1998). Damage by predation may be small enough on each colony to allow net positive growth in the long term. Conversely, complete loss of a community will not allow such a growth.

CONCLUSION

Surprisingly and contrary to predictions, coral transplants in this study were most affected by fish corallivory inside a no fishing MPA. Branching species showed a significant difference in damage between the three management regimes, unlike the massive morphology which seemed less susceptible to corallivory, but showed no difference for direct impacts of beach seining. Due to the short time scale of this study and the lack of replication, it is difficult to generalize from these results. Nevertheless, it is a good example that managers and scientist alike must be aware of secondary effects of marine protected areas on coral reef communities.

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LITERATURE CITED

Allison GW, Lubchenco J, Carr MH (1998) Marine reserves are necessary but not sufficient for marine conservation. Ecol Appl 8:S79-S92

Bell JD, Galzin R (1984) Influence of live coral cover on coral-reef fish communities. Mar Ecol Prog Ser 15:265-274

Bowden-Kerby A (2001) Low-tech coral reef restoration methods modeled after natural fragmentation processes. Bull Mar Sci 69:915-931

Brown BE (1997) Disturbances to reefs in recent times. In: Birkeland C (ed) Life and Death of Coral Reefs. Chapman & Hall, New York, p 354-379

Bruckner AW, Bruckner RJ (1998) Destruction of corals by *Sparisoma viride*. Coral Reefs 17:350

Bruckner AW, Bruckner RJ, Solfins P (2000) Parrotfish predation on live coral: "spot biting" and "focused biting". Coral Reefs 19:50

Bruno JF (1998) Fragmentation in *Madracis mirabilis* (Duchassaing and Michelotti): how common is size-specific fragment survivorship in corals? J Exp Mar Biol Ecol 230:169-181

Bythell JC, Gladfelter EH, Bythell M (1993) Chronic and catastrophic natural mortality of three common Caribbean reef corals. Coral Reefs 12:143-152

Carreiro-Silva M, McClanahan TR (2001) Echinoid bioerosion and herbivory on Kenyan coral reefs: the role of protection from fishing. J Exp Mar Biol Ecol 262:133-153

Chabanet P, Ralambondrainy H, Amanieu M, Faure G, Galzin R (1997) Relationships between coral reef substrata and fish. Coral Reefs 16:93-102

Côté IM, Mosquera I, Reynolds JD (2001) Effects of marine reserve characteristics on the protection of fish populations: a meta-analysis. J Fish Biol 59:178-189

Cox EF (1986) The effects of a selective corallivore on growth-rates and competition for space between 2 species of Hawaiian corals. J Exp Mar Biol Ecol 101:161-174

Cumming RL 1999. Predation on reef-building corals: multiscale variation in the density of three corallivorous gastropods, *Drupella* spp. Coral Reefs 18:147-157

Done TJ (1982) Patterns in the distribution of coral communities across the central Great Barrier Reef. Coral Reefs 1:95-107

Done TJ (1992) Constancy and change in some Great Barrier Reef Coral Communities: 1980-1990. Amer Zool 32:655-662

Dusek E (2000) Effects of fishing gear on coral reef recovery in Kenya following 1998 El Niño induced coral bleaching. Honors Program, Stanford University, Palo Alto, Ca.

Edinger EN, Risk MJ (2000). Reef classification by coral morphology predicts coral reef conservation value. Biol Cons 92:1-13

Hawkins JP, Roberts CM, Van't Hof T, de Meyer K, Tratalos J, Aldam C (1999) Effects

coral and fish communities. Cons Biol

Hughes TP (1985) Life histories and population dynamics of early successional corals. Proc 5th Int Coral Reef Congr 4:101-106

Jameson SC, Ammar MSA, Saadalla E, Mostafa HM, Riegl B (1999) A coral damage index and its application to diving sites in the Egyptian Red Sea. Coral Reefs 18:333-339

Jennings S, Kaiser MJ (1998) The effects of fishing on marine ecosystems. Adv Mar Biol 34:291-302

Jennings S, Polunin NVC (1996) Impacts of fishing on tropical reef ecosystems. Ambio 25:44-49

Karlson RH, Hurd LR (1993) Disturbance, coral reef communities, and changing ecological paradigms. Coral Reefs 12:177-125

Lindahl U (1998) Low-tech rehabilitation of degraded coral reefs through transplantation of Staghorn corals. Ambio 27:645-648

Lindahl U, Öhman MC, Schelten CK (2001) The 1997/1998 mass mortality of corals: effects on fish communities on a Tanzanian coral reef. Mar Poll Bull 42:127-131

McClanahan TR (1994a) Kenyan coral reef lagoon fish: effects of fishing, substrate complexity, and sea urchins. Coral Reefs 13:231-241

McClanahan TR (1994b) Coral-eating snail *Drupella cornus* population increases in Kenyan coral reef lagoons. Mar Ecol Prog Ser 115:131-137

McClanahan TR (2000) Bleaching damage and recovery potential of Maldivian coral reefs. Mar Poll Bull 40:587-597

McClanahan TR, Arthur R (2001) The effect of marine reserves and habitat on populations of East African coral reef fishes. Ecol Appl 11:559-569

McClanahan TR, Mangi S (2001) The effect of a closed area and beach seine exclusion on coral reef fish catches. Fish Manage Ecol 8:107-121

McClanahan TR, Muthiga NA, Mangi S (2001) Coral and algal changes after the 1998 coral bleaching: interaction with reef management and herbivores on Kenyan reefs. Coral Reefs 19:380-391

McClanahan TR, Muthiga NA, Kamukuru AT, Machano H, Kiambo RW (1999) The effects of marine parks and fishing on coral reefs of northern Tanzania. Biol Cons 89:161-182

McClanahan TR, Shafir SH (1990) Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoon. Oecologia 83:362-370

McClanahan TR, (1994) Coral-eating snail *Drupella cornus* population increases in Kenyan coral reef lagoons. Mar Ecol Prog Ser 115:131-137

Miller MW, Hay ME (1998) Effects of fish predation and seaweed competition on the survival and growth of corals. Oecologia 113:231-238

Mosquera I, Côté IM, Jennings S, Reynolds JD (2000) Conservation benefits of marine reserves for fish populations. Anim Cons 3:321-332

Obura DO (2001) Kenya. Mar Poll Bull 42:1264-1278

Rouphael AB, Inglis GJ (1997) Impacts of recreational scuba diving at sites with different reef topographies. Biol Cons 82:329-336

Rouphael AB, Inglis GJ (2002) Increased spatial and temporal variability in coral damage caused by recreational scuba diving. Ecol Appl 12:427-440

Turner SJ (1994a) The biology and population outbreaks of the corallivorous gastropod *Drupella* on Indo-Pacific reefs. Oceanogr Mar Biol Annu Rev 32:461-530

Turner SJ (1994b) Spatial variability in the abundance of the corallivorous gastropod *Drupella cornus*. Coral Reefs 13:41-48

Williams ID, Polunin NVC, Hendrick VJ (2001) Limits to grazing by herbivorous fishes and the impact of low coral cover on macroalgal abundance on a coral reef in Belize. Mar Ecol Prog Ser 222:187-196

Zar JH (2000) Biostatistical analysis. Prentice Hall, New Jersey

Table 1 . ANOSIM pairwise test for difference in types of damages for branching *Porites* between sites. Difference is significant between all three sites ($p < 0.05$).

Sites	R	p
Nyali/MMNP	0.369	0.008
Nyali/Ras Iwatine	0.277	0.025
MMNP/Ras Iwatine	0.452	0.004

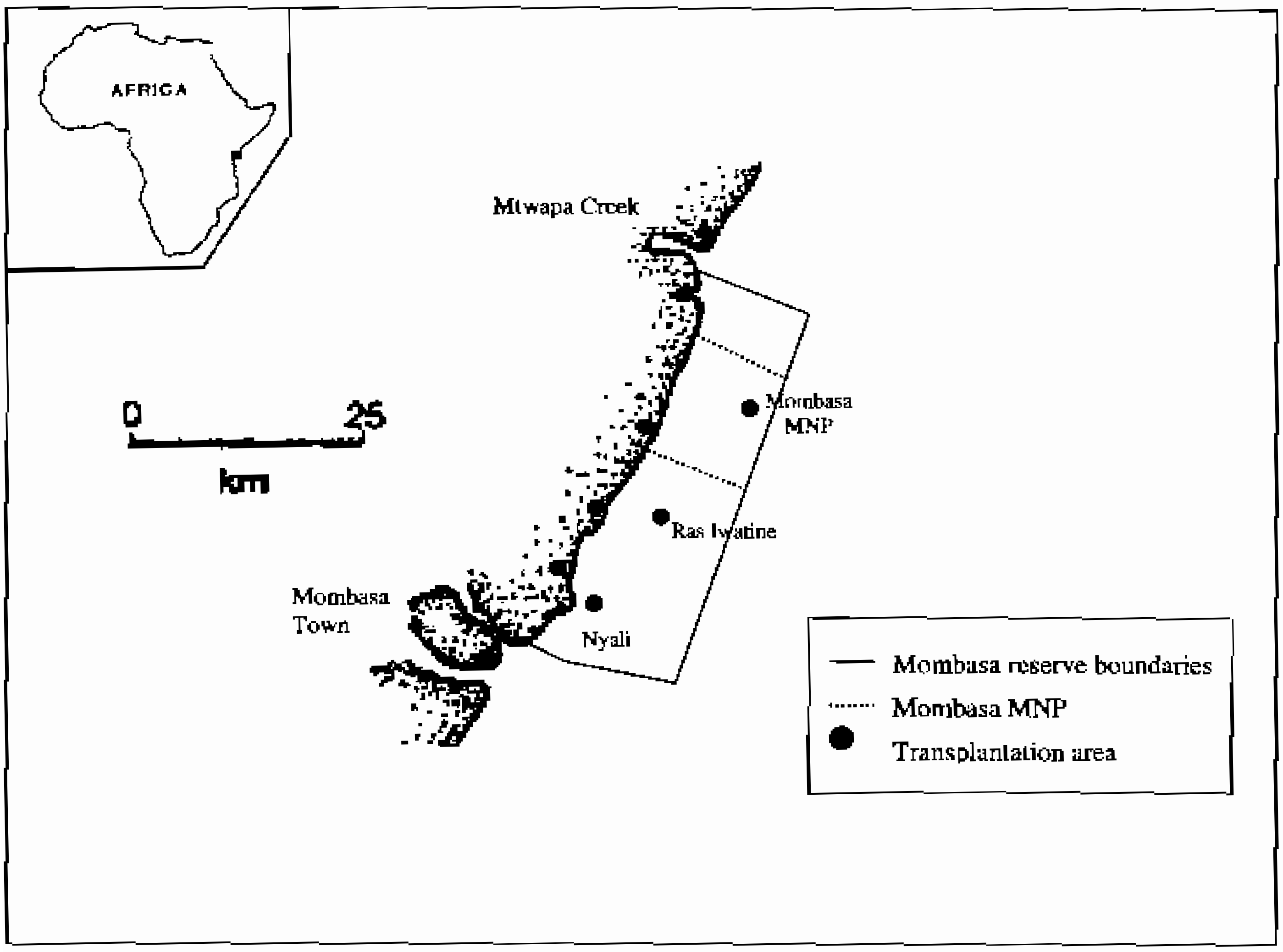

Fig. 1. Map of Mombasa Marine Reserve and transplantation sites, Mombasa, Kenya.

Fig. 2. Bite marks on a transplanted "knobby" massive *Porites* in Mombasa Marine National Park.

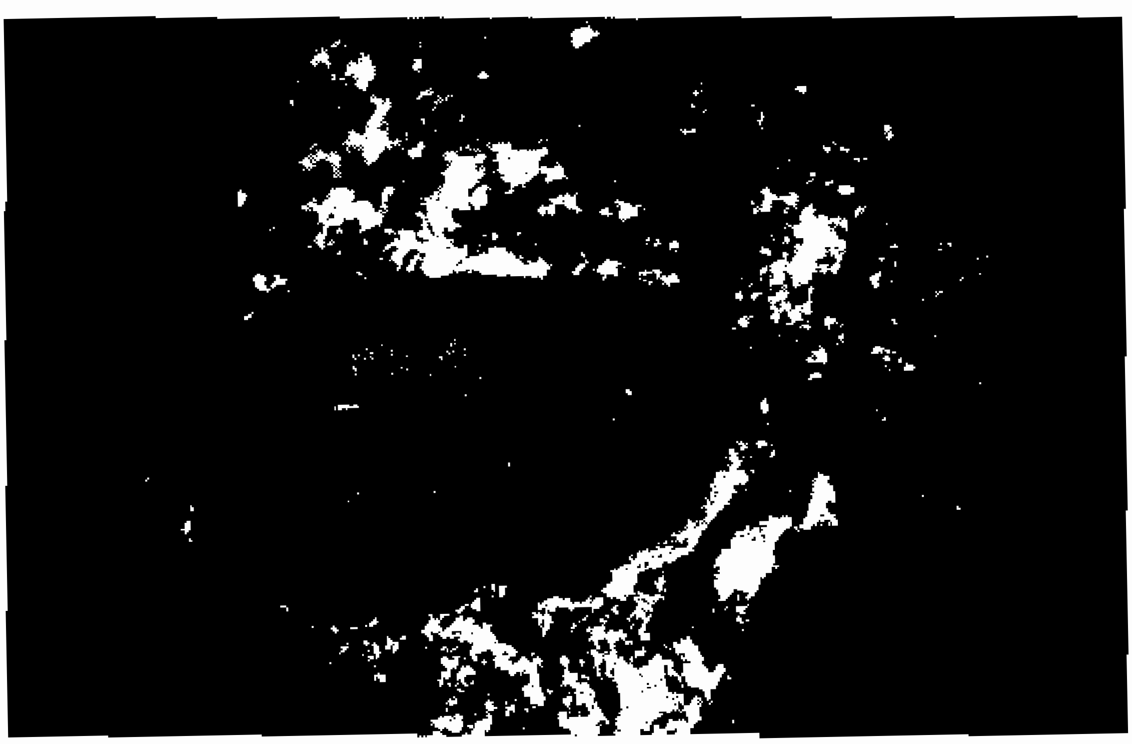

Fig. 3. "Smooth" massive *Porites*, 0% mortality in the Mombasa Marine National Park.

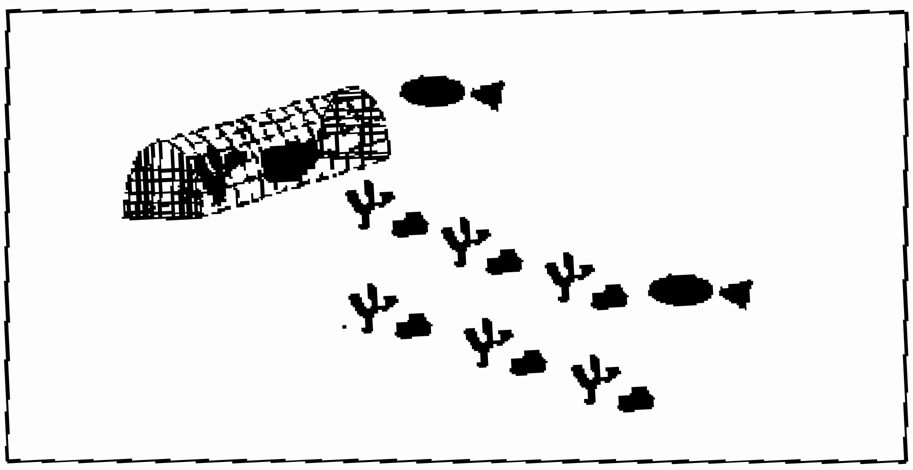

Fig. 4. "Block": seven pairs of corals (one branching *Porites* and one massive *Porites*) were transplanted. One pair was caged to control for predation. This was repeated for every "block". There were six blocks in MMNP and Ras Iwatine and nine blocks in Nyali.

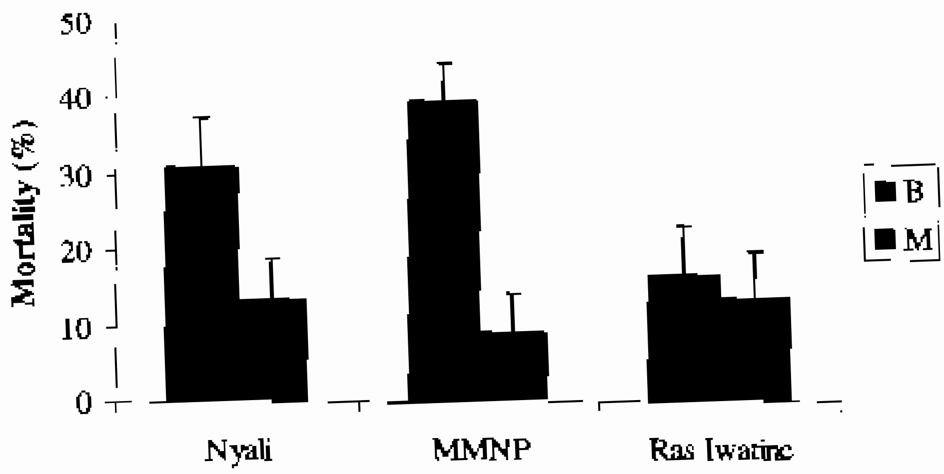

Fig. 5. Percentage mortality between sites for branching (B) and massive (M) *Porites*. Bars represent 95 % C.I. Branching *Porites* suffered a mortality in MMNP of 39.48 % (C.I. ± 4.93), in Nyali of 31.00 % (C.I. ± 6.49) and in Ras Iwatine of 16.75 % (C.I. ± 6.37). For massive *Porites*, MMNP suffered 9.1 % (C.I. ± 5.31) mortality, Nyali 13.55 % (C.I. ± 5.35), and Ras Iwatine 13.52 % (C.I. ± 6.36).

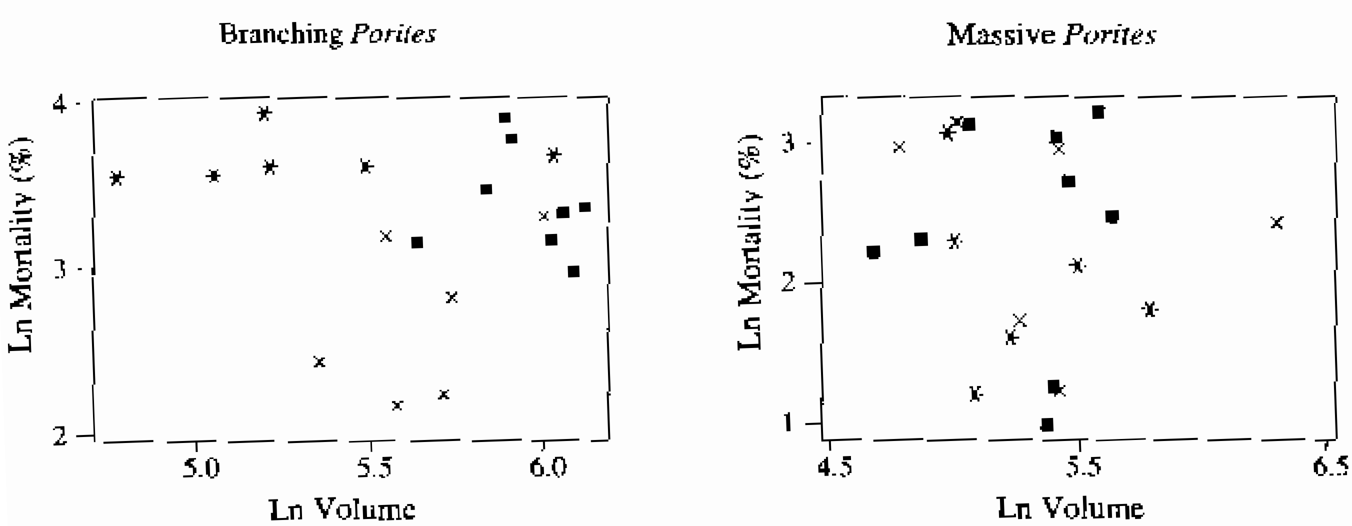

Fig. 6 and 7. Correlation between size (volume) and percentage mortality for the three sites. Full squares represent average volume of corals in one "block" against percentage mortality of the same corals in Nyali (n = 9). Stars represent MMNP (n = 6) and crosses Ras Iwatine (n = 6).

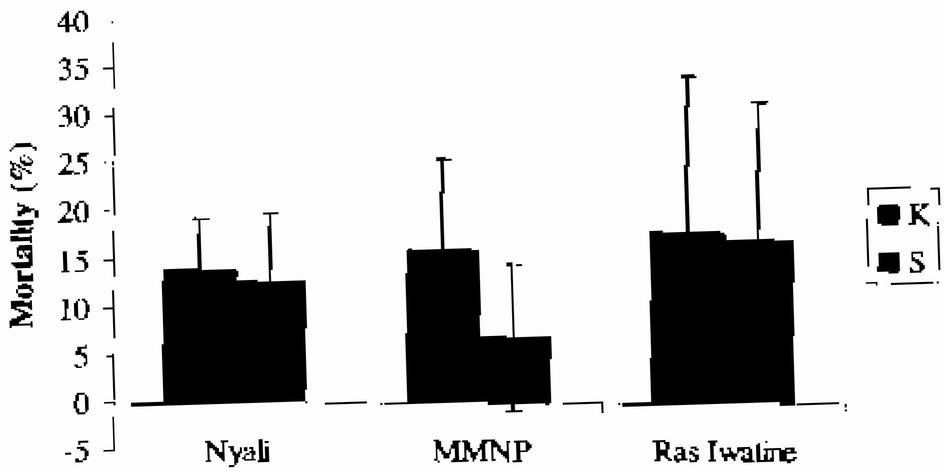

Fig. 8. Difference in percentage mortality between the two morphological types of massive *Porites*: knobby (K) and smooth (S) for Nyali, Ras Iwatine and MMNP. Bars represent 95 % C.I. Corals with knobby morphology suffered a mortality in MMNP of 15.98 % (± 9.47), in Nyali of 13.94 % (± 5.36), in Ras Iwatine of 17.74 % (± 16.42). Corals with smooth morphology suffered a mortality in MMNP of 6.77 % (± 7.71), in Nyali of 12.53 % (± 7.43), in Ras Iwatine of 16.92 % (± 14.64).

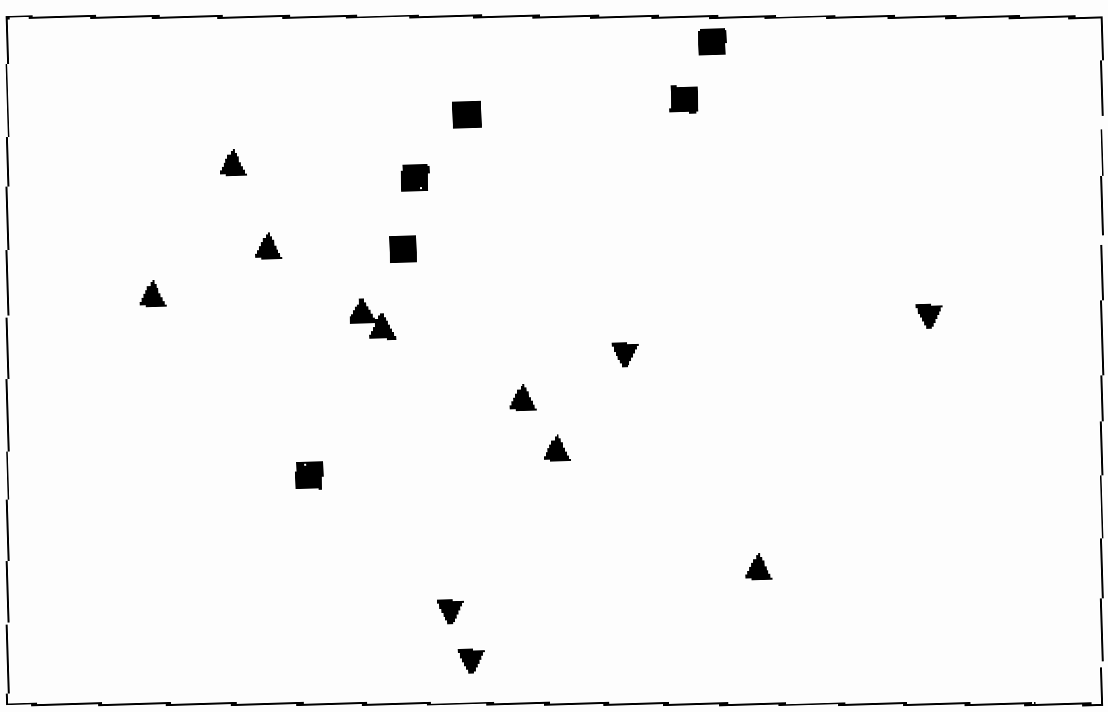

Fig. 9. MDS ordination plot based on Bray–Curtis similarities showing the damage composition for MMNP, Nyali and Ras Iwatine for branching *Porites* (n = 9 for Nyali, n = 6 for MMNP, n = 6 for Ras Iwatine). Stress=0.11. Inverted triangles represent MMNP, dark triangles represent Nyali, squares represent Ras Iwatine.

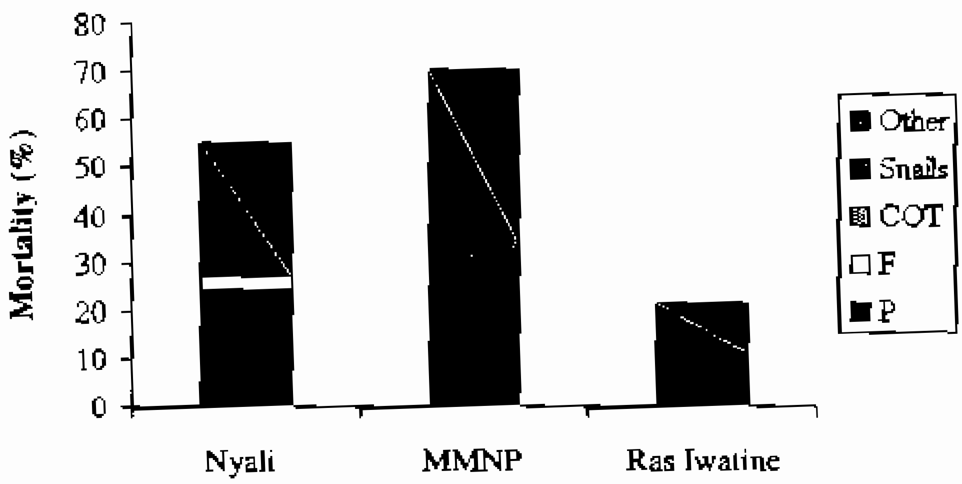

Fig. 10. Percentage mortality of branching *Porites* due to different types of damage for the three sites. (P: predation by fish, F: fishing, COT: crown-of-thorns, "other" includes: transplantation, sand, sponge and unidentified damage).

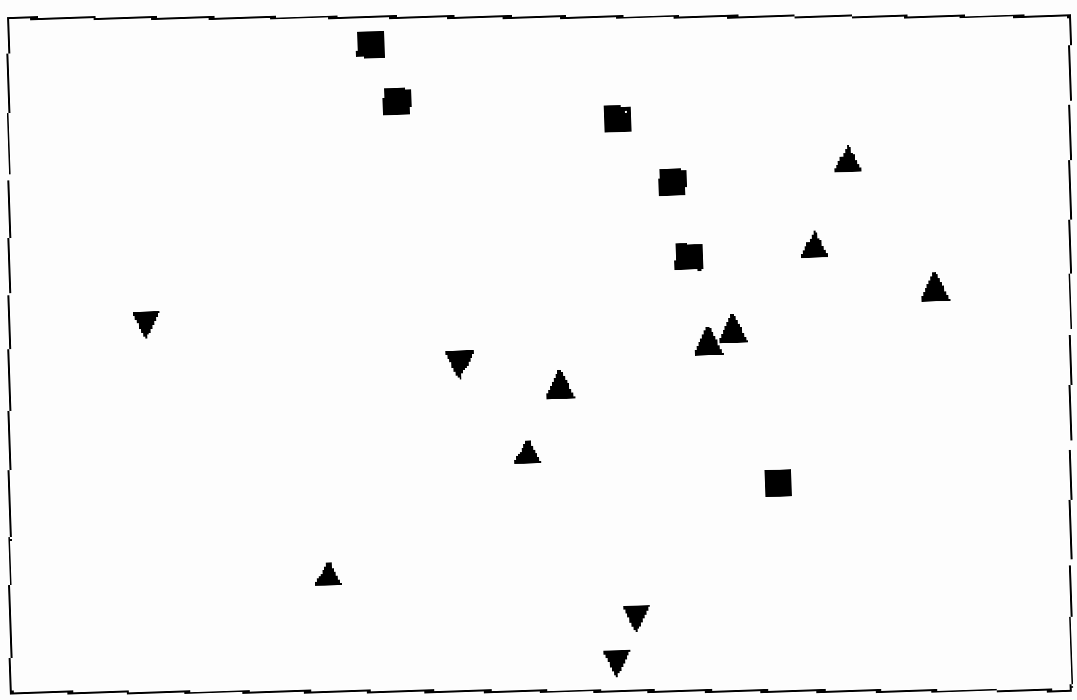

Fig. 11. MDS ordination plot based on Bray–Curtis similarities showing the damage composition for the three sites for massive *Porites*.(n = 9 for Nyali, n = 6 for Park, n = 6 for Ras Iwatine). Stress=0.1. Inverted triangles represent MMNP, dark triangles represent Nyali, squares represent Ras Iwatine.

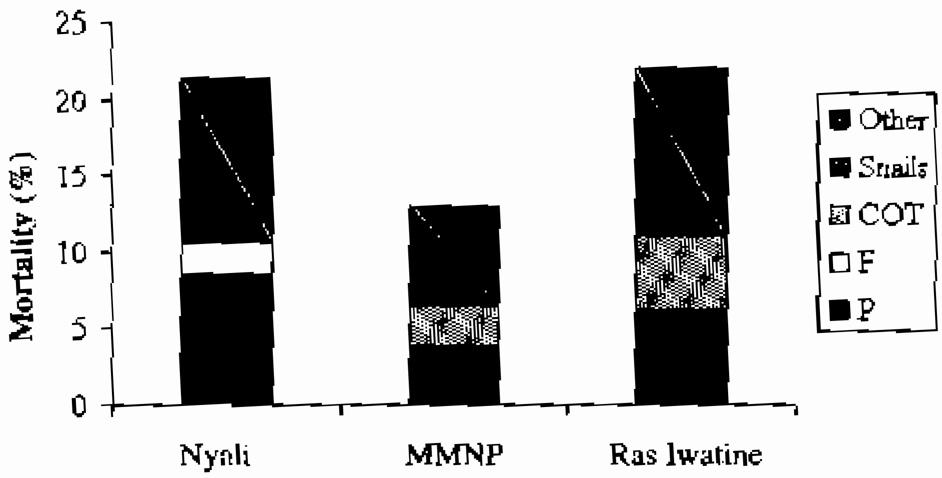

Fig. 12: Percentage mortality of massive *Porites* due to different types of damage for the three sites (P: predation by fish, F: fishing, COT: crown-of-thorns, "other" includes: transplantation, sand, sponge and unidentified damage).

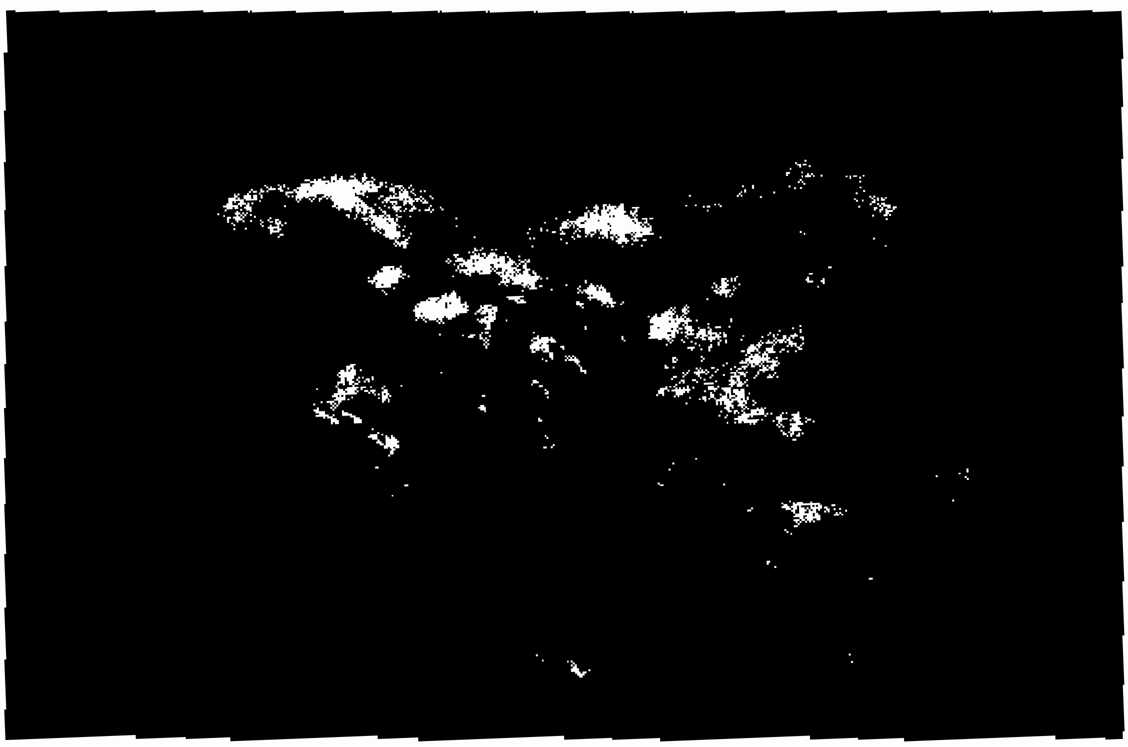

Fig. 13. *Drupella cornus* predation on branching *Porites* in the Mombasa Marine National Park. 6 snails can be observed eating the polyps at the base of branches (bleached branches). Damage was estimated at 60% mortality on the last day.