Status of Coral Reefs in Trincomalee, Sri Lanka

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INTRODUCTION

Trincomalee is located on the east coast of Sri Lanka. It is known for its deepwater natural harbour, abundant fish resources, and large marine fauna such as whales and dolphins. It is an important location for tourism and has many attractions on land as well as underwater. The rocky seabed of Trincomalee supports extensive reef habitats, the majority of which are large boulder type reefs of crystalline rock. Coral reefs are few and their distribution is patchy along the coast. The main coral reefs are located at Nilaveli, (Pigeon Island and Coral Island), Dutch Bay, Back Bay, Coral Cove and Foul Point, and along the coast south of Foul Point to Batticaloa (figure 1) (Swan, 1983; Rajasuriya & Premaratne, 2000). There are also some small coral patches within the Trincomalee Harbour and in embayments along the eastern coast outside the harbour. There are no coral reefs in Koddiyar Bay due to freshwater and sediment input from the Mahaweli River. A very deep canyon more than 1 000 m deep extends in a north-easterly direction from Koddiyar Bay.

There are extensive rock and sandstone reef habitats, both inshore and offshore. Narrow fringing coral reefs have developed on rock substrates, extending about 200 m from the coast to a depth of about 8 m, while offshore reef habitats are found to a depth of more than 50 m. To date, over 100 species of corals and more than 300 species of reef fish have been identified in Trincomalee and surrounding areas. Rajasuriya and Karunarathne (2000) reported that corals in Trincomalee were not af-

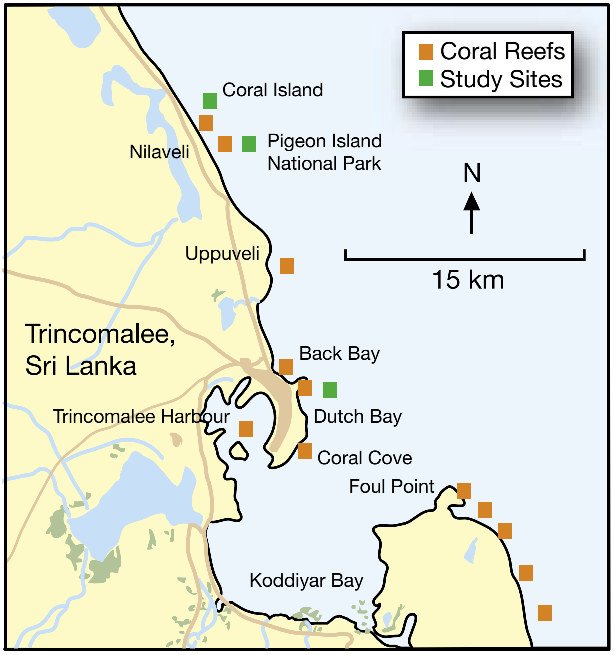

Figure 1. Map of Trincomalee, eastern Sri Lanka showing coral reef areas and study sites.

fected during the major coral bleaching event in 1998, whilst most reefs elsewhere in the country were severely damaged. However, the area experienced major reef degradation in the 1970s due to a crown-of-thorns starfish (*Acanthaster planci*; COTS) infestation (De Bruin, 1972; Rajasuriya & White, 1995).

Reef resources are utilised extensively. Fishing for edible species and ornamental fish collecting are the most common extractive reef uses in Trincomalee, while extensive harvesting of sea cucumber and chanks (*Turbinella pyrum*) is a relatively recent phenomenon that has evolved during the last 5 years. As elsewhere in Sri Lanka, reefs in the area are heavily affected by human activities, especially destructive fishing practices and over harvesting of resources (Rajasuriya *et al.*, 1995), and urban pollution affects reefs close to the town of Trincomalee. Pigeon Island is popular among tourists, both local and foreign, who damage the coral reef by trampling and collecting souvenirs. Visitors also dump refuse on the island.

Scientific studies of the reefs in Trincomalee are lacking because the past two decades of internal conflict in the country has rendered the area inaccessible. Thus the coral reefs were monitored only after the 1998 bleaching event, and due to the prevailing security concerns, reef surveys have been confined to the area between the Dutch Bay and Coral Island. The first survey was carried out at Pigeon Islands in 1999 (Christoffelsz *et al.*, 2000). Thereafter, two more surveys were conducted at this location, Dutch Bay and Coral Island were surveyed in 2003, and several other reef sites in the area were investigated without conducting detailed surveys. All surveys were carried out jointly by the National Aquatic Resources Research and Development Agency (NARA) and the Sri Lanka Sub-Aqua Club (SLSAC). In 2003 and 2004, the Hong Kong and Shanghai Banking Corporation also participated in investigating reefs at the study sites.

SITES

Pigeon Island National Park (N 8 43.198 E 81 12.101) is made up of two small islands (Large and Small Pigeon Island) located about 1 km offshore. There are several rock outcrops about 300–500 m to the south and south-east of Large Pigeon Island, the one on the southern side is called the 'Salabalas Rocks'. Large Pigeon Island has two small beaches on the south-western and northern flanks of the island. The main coral reef is in front of the south-western beach. It is about 200 m long and 100 m wide, with a depth of between 1 and 6 m. There are no large coral patches around Small Pigeon Island. Much of the surrounding area contains rocky reef habitats interspersed with old limestone reef structures and sandy patches. The two islands were declared a sanctuary in 1963 for the purpose of protecting birds. In 2003, Pigeon Island and the surrounding area within a one-mile radius, including its coral reefs, were declared a National Park under the Fauna and Flora Protection Ordinance of the Department of Wildlife Conservation.

Coral Island (N 8 44.200 E 81 10.590) is located about 500 m north of Pigeon Island and about 300 m offshore. It is a small rocky outcrop without sandy beaches. Fringing coral reefs have developed on the northern and southern sides, and small coral patches also occur around the island to a distance of about 200 m. Most of these coral areas are in very shallow water, the near shore fringing reef patches at depths of 2–4 m and the coral patches in the surrounding at depths of 4–6 m.

Dutch Bay contains a shallow coral patch at its northern end (N 8 34.404 E 81 14.373). The rest of the bay is made up of rocky substrate interspersed with patches of sand. In the centre of the bay, there are large *Porites* domes reaching a diameter of about 4 m.

SURVEY METHODS

The Reef Check method (Hodgson, 1999; 2003) was used for surveying the coral reefs as it is a rapid survey technique that allows coverage of large areas, and provides basic information about reef status and health. Point intercept transects 100 m in length were deployed between 3 and 7 m depth, and the following benthic cover categories recorded: hard coral (HC), dead coral

(DC), recently killed coral (RKC), soft coral (SC), sponge (SP), fleshy algae (FS), other organisms (OT) such as corallimorpharians and tunicates, coral rubble (RB), eroded dead coral blocks or rock (RC), sand (SD) and silt (SI). Brief visual surveys were also conducted on reef sites at Uppuveli and Back Bay in 2003.

The coral reefs around Pigeon Island were surveyed in 1999 and 2003 while Coral Island and Dutch Bay were surveyed only in 2003. Pigeon Island and Dutch Bay were also investigated in 2004 to record human influences, types of resource extraction and utilization of coral reefs for recreational purposes.

RESULTS

The surveys revealed that the investigated reef sites were relatively healthy. Surveys at Pigeon Islands indicated that there was little change in the status of live corals between 1999 and 2003, and the reefs that were degraded due to the COTS infestations in the 1970's (De Bruin, 1972) had recovered well. The benthic cover at each reef is presented separately below.

Pigeon Islands

Living hard coral cover on the shallow reef at Pigeon Islands was about 51.3% in 1999, increasing slightly to about 54.4% in 2003 (figure 2). The area is dominated by *Acropora* spp. (figure 3 on next page). The rock and limestone reef section between large Pigeon Island and the Salabalas Rocks was surveyed in 1999 only. Its living hard coral cover was about 24.4%, dominated by corals belonging to the families Faviidae, Mussidae and Poritidae. Notably, coral rubble constituted more than 30% of the benthic cover, while the cover of other organisms (e.g. corallimorpharians) was ~15% (figure 4 on next page).

Coral Island

The shallow fringing coral reefs on both the northern and southern sides of Coral Island were surveyed in 2003. The combined hard coral cover was 58%, mostly consisting of

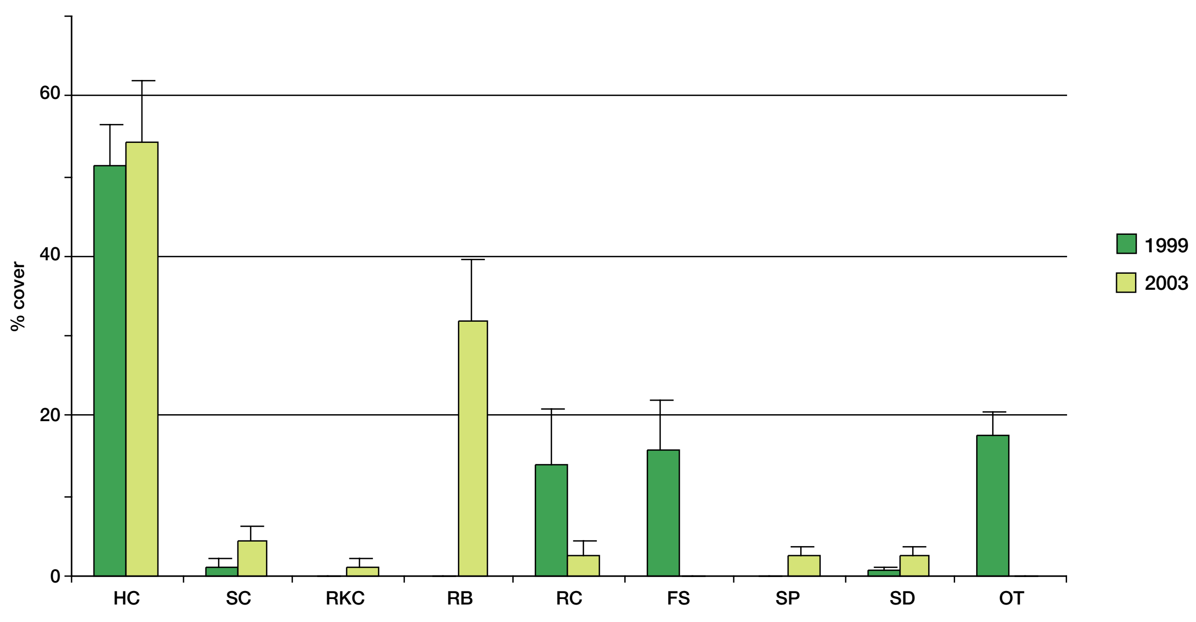

Figure 2. Percentage cover of each substrate category recorded at Pigeon Island during surveys conducted in 1999 and 2003.

Figure 3. *Acropora* dominated coral reef at Pigeon Island.

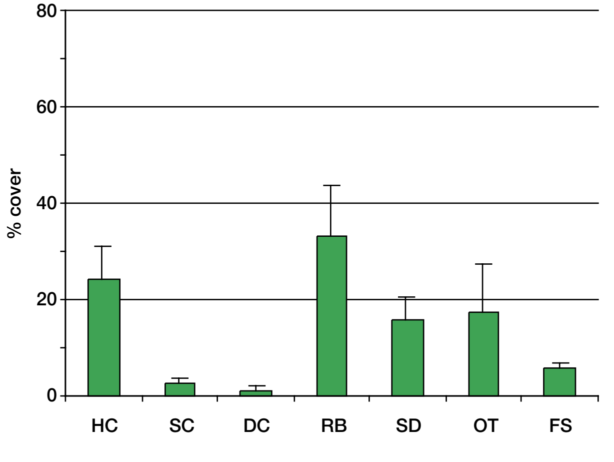

Figure 4. Percentage cover of each substrate category recorded between Pigeon Island and Slabalas Rocks during surveys conducted in 1999.

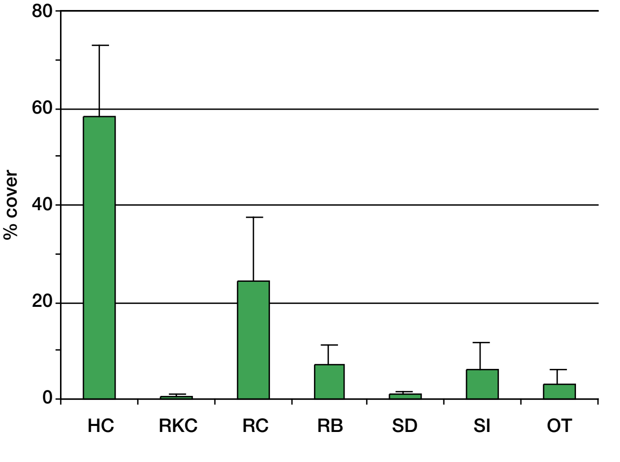

Figure 5. Percentage cover of each substrate category recorded at Coral Island during surveys conducted in 2003.

branching *Acropora* species, while ~24% consisted of rocky substrate devoid of corals (figure 5). Some dead coral and coral rubble was recorded. The northern side of the island exhibited higher coral cover, while rubble and non-coral benthic fauna cover was higher on the southern side.

Dutch Bay

The shallow coral patch was surveyed in 2003. Benthic cover was determined along two line intercept transects at an average depth of 3.5 m. Hard coral cover was 51.5%, with foliose *Montipora* the most abundant coral (figure 6).

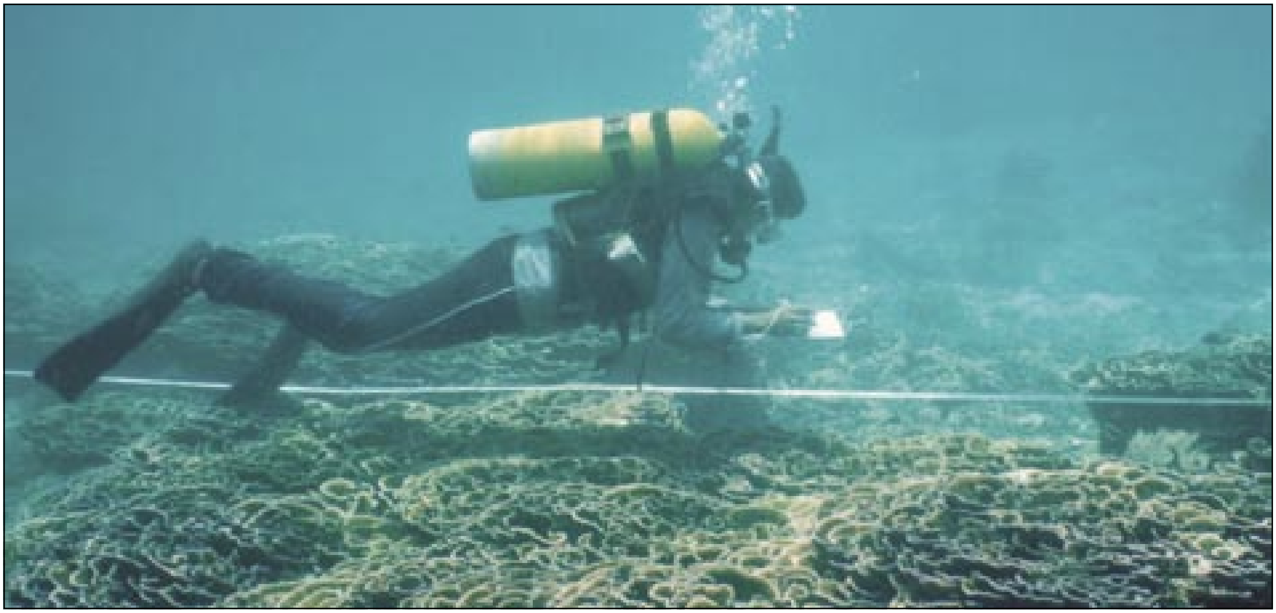

Figure 6. *Montipora* dominated habitat at Dutch Bay in 2003.

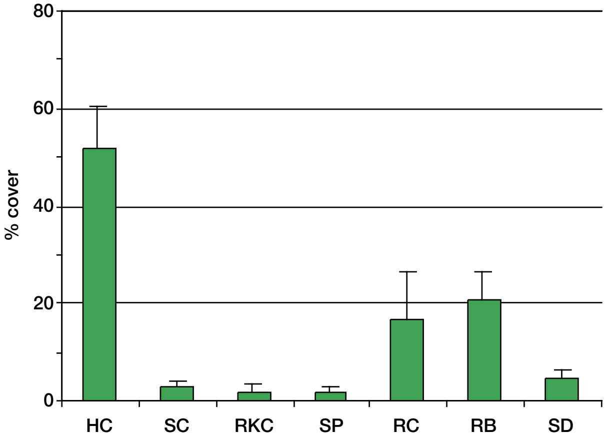

Figure 7. Percentage cover of each substrate category recorded at Dutch Bay during surveys conducted in 2003.

There were also many branching and digitate *Acropora* colonies. Coral rubble and rock substrates were 20.9% and 16.5% respectively (figure 7). The recently killed coral, making up about 1.8% of the total benthic cover, can be attributed to presence of *Acanthaster planci*.

RESOURCE USE

Fishing and ornamental fish collecting are the main extractive reef uses in Trincomalee. There is also widespread exploitation of sea cucumber particularly from the coral reefs in Nilaveli and Foul Point. Commonly used fishing gears are fish traps, rod and line, and a variety of nets including gill nets and bottom set nets. The damage caused by these nets in other areas of Sri Lanka have been highlighted previously (Rajasuriya *et al.*, 1995; Rajasuriya & Wood, 1997).

Major Causes of Reef Degradation in Trincomalee

- Blast fishing
- Use of banned fishing gears such as the bottom set nets used to harvest spiny lobsters and reef fishes and the use of *moxy* nets (a diver deployed underwater drop net) to collect aquarium fishes from shallow coral reef habitats.
- Uncontrolled resource exploitation
- Urban pollution
- Boat anchoring
- Visitor pressure on Pigeon Island and Coral Island (Trampling, souvenir collecting and pollution).

CONCLUSIONS

The coral reefs in Trincomalee have recovered from the COTS infestations that were first observed in the 1960s and reaching a peak in the early 1970s (De Bruin, 1972. This coincided with large scale COTS outbreaks on the Great Barrier Reef (Endean & Cameron, 1985). All the study sites showed healthy coral growth 20 years after the COTS infestation and they were dominated by fast growing species of *Acropora* and *Montipora*. Similar recovery has been observed elsewhere; on the mid-shelf reefs in the Great Barrier Reef, coral habitats damaged by COTS recovered to a level comparable with unaffected mid-shelf reefs after about 10–12 years (Moran *et al.*, 1985) due to the rapid growth of branching corals. A few COTS were present on all the sites surveyed, but no large-scale damage to any of the reef habitats was observed.

Coral reefs in Trincomalee are the only known reefs in Sri Lanka that escaped the major coral bleaching event in 1998 (Wilkinson, 1998; Rajasuriya *et al.*, 2000). Minor coral bleaching was observed at depths of about 1.5 m in the Dutch Bay and at a depth of about 3 m in the Trincomalee harbour in May 2004, which is the same time of year the mass coral bleaching occurred in 1998. Similar minor bleaching was also reported from the southern coast at Rumassala in Galle Bay in 2004.

However, despite recovering from COTS infestations

Blast Fishing in Trincomalee

Blast fishing is a widespread fishing method in Sri Lanka (Rajasuriya *et al.*, 2002), despite being banned under both the Fisheries Act and the Fauna and Flora Protection Ordinance. In Trincomalee too, this is the most damaging fishing practice at present, carried out daily especially north of Dutch Bay. It is most prevalent in the offshore area (2–5 km) from Uppuveli to north of Nilaveli, including in Pigeon Islands National Park.

Blast fishing destroys all forms of marine life and damages reef structures. It is also extremely harmful to divers. Today, all dive operators in Trincomalee have experienced the negative impact of blast fishing and have already lost business and potential revenue from tourists due to this practice.

There is also little interest in protecting and managing the marine environment other than to strengthen legislation periodically. Recently, the penalty for blast fishing was increased through an amendment to the Fisheries Act, but regulations are not enforced. A general lack of resources, manpower and equipment for management of fisheries and marine protected areas, and consequent inability to arrest blast fishermen at sea, means destructive practices continue to threaten coral reefs in Sri Lanka.

and escaping the bleaching in 1998, the area faces a severe threat. Pigeon Island has been declared a marine National Park, but there is no management plan nor are the park regulations implemented. At present, it is unguarded and visitors freely take corals and other souvenirs from the reef and pollute the island by discarding polythene bags and other non-biodegradable refuse. Nevertheless, the most serious threat to the health of biota in Trincomalee is rampant blast fishing. There is an urgent need to strengthen political will to implement and enforce existing fisheries regulations and provide Pigeon Island the protection its National Park status warrants, in order to prevent rapid decline of reef health and fish populations. Lastly, Trincomalee and the surrounding areas have not been studied in the same detail as some other locations in Sri Lanka, e.g. Bar Reef Marine Sanctuary. In order to strengthen and support management, long-term studies on coral reef health and resource use should be initiated in this area.

ACKNOWLEDGEMENTS

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A Preliminary Baseline Survey of the Coral Reefs of Passikuda, Batticaloa

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key words: Passikudah, coral reef status, tourism impacts, solid waste

INTRODUCTION

Study area

Sri Lanka is situated between 5° and 10° north of the equator and south of the Indian subcontinent and has a total land area of 65 000 km^2. The coastline of the island, which is about 1 585 km, supports highly productive ecosystems such as coral reefs, mangroves, seagrass beds and marshy lands. The continental shelf of Sri Lanka is relatively narrow averaging only a few km in width (Rajasuriya *et al.*, 1995).

Well-developed coral reefs exist along much of the coast of Sri Lanka and, along the east coast, they occur between Vaharai and Valaichechenai and from Batticaloa to Kalmunai. These reefs are important sources of natural resources. For example, corals are mined to produce lime for the construction industry, the export of spiny lobsters, sea cucumber and ornamental fish have contributed significantly to the foreign exchange earnings of the country and tourism benefits from the aesthetic value of coral reefs. Despite their value, reef habitats are being degraded at an alarming rate due to unmanaged resource use. Unfortunately however, systematic baseline surveys of the condition of the reefs found along the north and east coast have not been conducted due to the civil conflicts during the last 20 years between the State and the LTTE.

Degradation of habitats at Passikudah

Passikudah Reef is located 28 km north of Batticaloa in a relatively calm bay which is a favoured spot for bathing. During the civil war that has been fought over the last two decades, the fisher families that comprise the civilian population at Passikudah were deprived of their main livelihood of fishing because of restrictions imposed by the military. In addition, tourism had virtually ceased further denying them of any supplementary income from the tourist industry. As a consequence, these people resorted to large scale coral mining of both extant and fossil coral reefs in the area in order to produce lime illegally. As there were severe restrictions in the southern part of the country on lime production, the lime produced at Passikudah was sent to the south in a surreptitious manner (Dharmaretnam & Kirupairajah, 2003). Large-scale live coral mining at Passikudah was stopped only recently.

With the recent cessation of hostilities between the LTTE and the State, local tourists have been arriving at Passikudah in increasing numbers. This large scale influx has had a negative impact on the environment as there were not adequate mechanisms to cope with the local tourists. Large quantities of polythene plastics and other solid wastes left by the tourists now litter the reefs in the bay.

The coral reefs of Sri Lanka were badly affected during the coral bleaching event of 1998. Coral bleaching was reported at sites in the west and south of Sri Lanka such as Bar Reef Marine Sanctuary, Hikkaduwa and Unawatuna (Rajasuriya, 2002). However, the reefs on the east coast at Trincomalee (Pigeon Island) did not seem to be affected by the coral bleaching event. To date, the reefs along the east coast south of Trincomalee have not been surveyed. Thus, the aim of the project was to conduct a baseline survey of the reef at Passikudah to determine its condition.

METHODOLOGY:

At Passikudah Reef, surveys were conducted between January and June 2004 to determine the condition of the reef. Beginning in the centre of the bay and moving eastwards towards Kalkudah, seven 50 m line intercept transects were laid perpendicular to the shore and separated by 100 m. The linear extent of each substrate type bisected by each transect was recorded by snorkel divers using the following codes from English *et al.*, 1997:

LC = Live Coral;
DC = Dead Coral (recently dead white to dirty white);
DCA = Dead Coral with Algae (this coral is standing, skeletal structure can still be seen);
R = Rubble (Unconsolidated coral fragments);
S = Sand;
SL = Silt;
AA = Algal Assemblage (consisting more than one species);
HA = *Halimeda*;
MA = Macro Algae (weedy/fleshy browns, reds, etc.);
TA = Turf Algae (lush filamentous algae, often found inside damselfish territories);
SP = Sponges;
OT = Others (Ascidians, anemones, gorgonians, giant calms).

The mean percent cover was calculated for each substrate type. The frequency with which different varieties of solid wastes occurred along each transect was also noted.

Salinity, temperature, turbidity and water depth were also measured at each transect using a refracto salinometer, a submersible thermometer, a secchi disc and a tape measure respectively.

RESULTS AND DISCUSSION

The overall cover of live coral was 15.81% (± 2.96 S.E.) (figure 1 on next page) and ranged from 8.88% to 29.6% (figure 2 on next page). The cover of dead coral, dead coral with algae and sand ranged between 10.6%–29.66%, 3.22%–8.6% and 6.2%–28.86% respectively and was generally greater towards the western boundary of the study site. The predominantly sandy substrate near the western most transects ensures that this area is the preferred site for bathing and where most tourist activities are concentrated. Coral rubble also occupies a considerable proportion of the substrate with the mean percent cover being 18.87% (± 1.57 S.E.) (figure 1) but is fairly evenly distributed between transects probably indicating skeletal breakdown following coral bleaching (figure 2). Despite the even distribution of coral rubble across the study site, the declining live coral cover and corresponding increases in dead coral with the concentration of tourist activities makes it quite evident that the abundance of live coral is being affected by trampling and siltation caused by the tourists. The impact of live coral mining would have also contributed to the status of the coral reefs of Passikudah.

The cover of *Halimeda* varied considerably between transects. In 3 transects, the cover exceeded 15% while in each of the remaining 4 transects, the cover was 5% or less (figure 2). There is a possibility that *Halimeda* is overgrowing new coral recruits at Passikudah, similar to that reported from the shallow waters of Kandakuliya in 2000 (Rajasuriya, 2002). Algal assemblages, macro-algae and turf algae comprised 6.1% of the substrate. Sponges and others consisted of 2.84% (figure 1).

The average salinity was 32‰ and visibility ranged from 75–88 cm. Water surface temperature was 28.6°C.

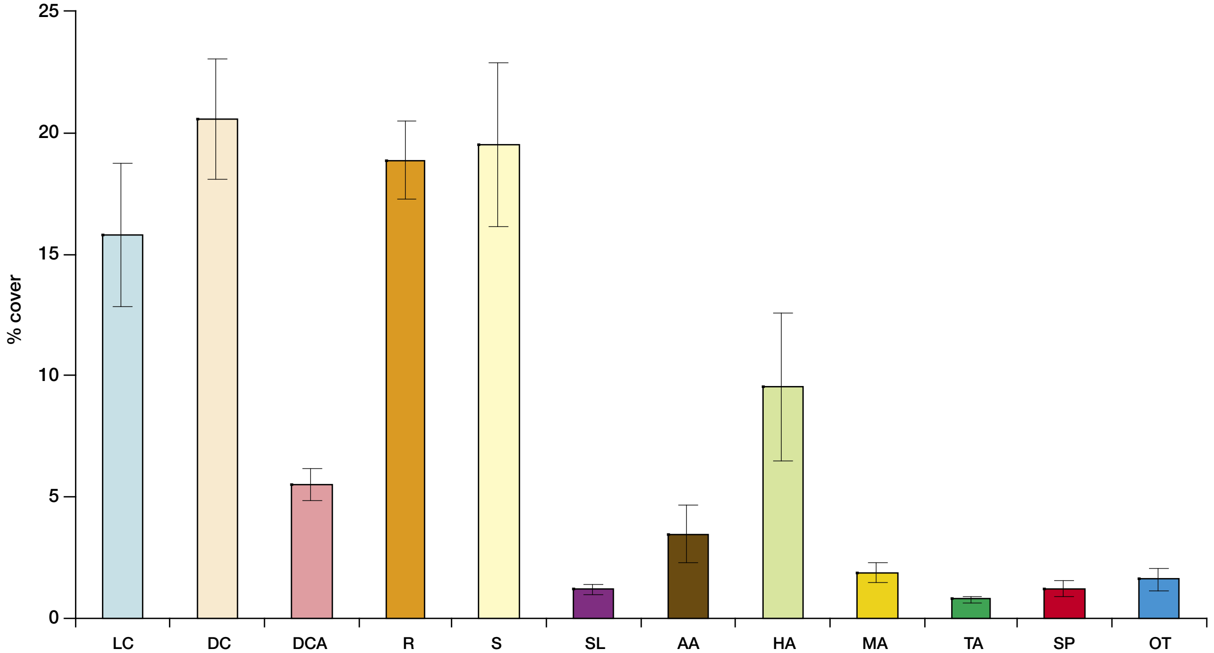

Figure 1. Mean percent cover (±S.E.) of each substrate type recorded from all transects conducted at Passikudah Reef.

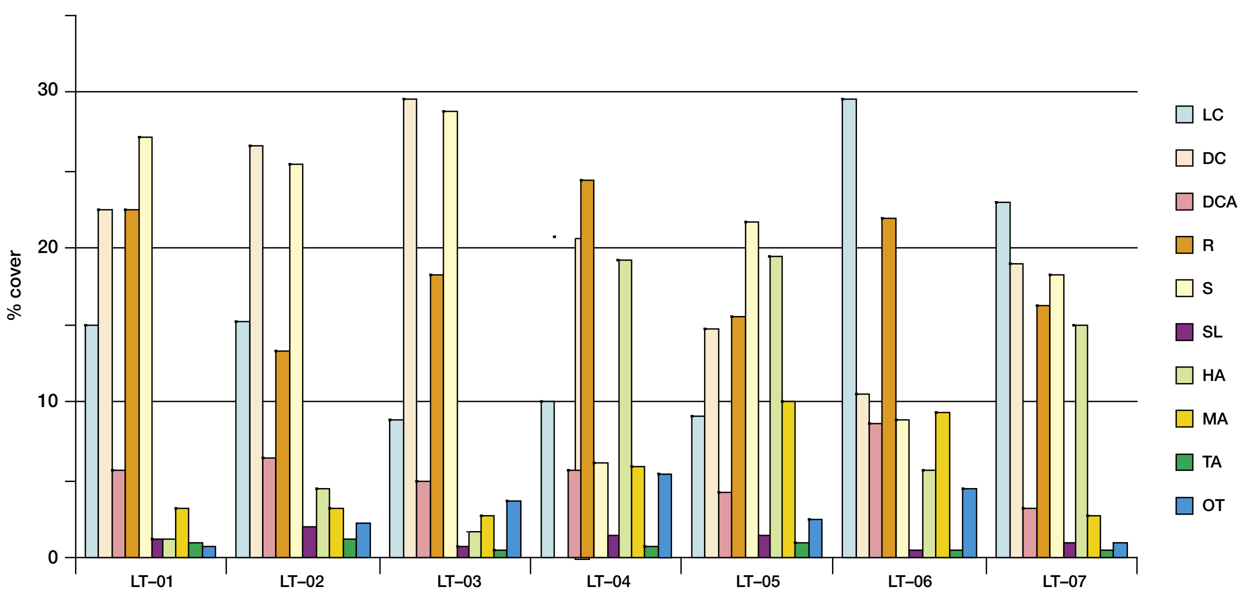

Figure 2. The percent cover of each substrate type recorded from each transect conducted at Passikudah Reef. Note: data for algal assemblages and macroalgae and for sponges and other have been combined and are presented as macroalgae and other respectively.

water depth ranged from 66–92 cm at the beginning of the transect and it was 90–151 cm at the end of the transect towards the sea.

The percentage cover of live coral at Passikudah in 2004 was similar to that of reefs in the southern part of the country that were damaged by coral bleaching. In 2000, live coral cover at Negambo Reef was 14%, while at Hikkaduwa in 2001, coral cover had increased from 7% immediately following the bleaching event in 1998 to 12% (Rajasuriya, 2002). Further systematic observations of Passikudah Reef are necessary to understand the dynamics of this reef.

Polythene plastics, including bags, papers and plastic bottles, were the most common solid wastes found on Passikudah Reef and, on average, were recorded approximately 5 times on each transect (figure 3). Although less frequent, wastes discarded from fishing activities such as longline filament, nets and anchors were the second most common pollutant occurring, on average, almost 4 times per transect. Bones, twigs, glass bottles and bricks comprised the remaining types of solid waste found on Passikudah Reef occurring 0.857, 2.286, 3.571 and 3.143 respectively.

CONCLUSION

The results of surveys presented in this paper are the first obtained from any coral reef along the east coast of Sri Lanka. It seems possible from comparisons with other studies done in southern part of Sri Lanka that the reef at Passikudah could have been damaged by the coral bleaching event. The impact on the live coral mining industry is also a factor to be considered.

Any recovery of these damaged reefs is currently threatened by activities of increasing local tourist population. The presence of discarded plastic material and bottles from tourists is very much higher than the material left by fishermen. The local authorities establish more

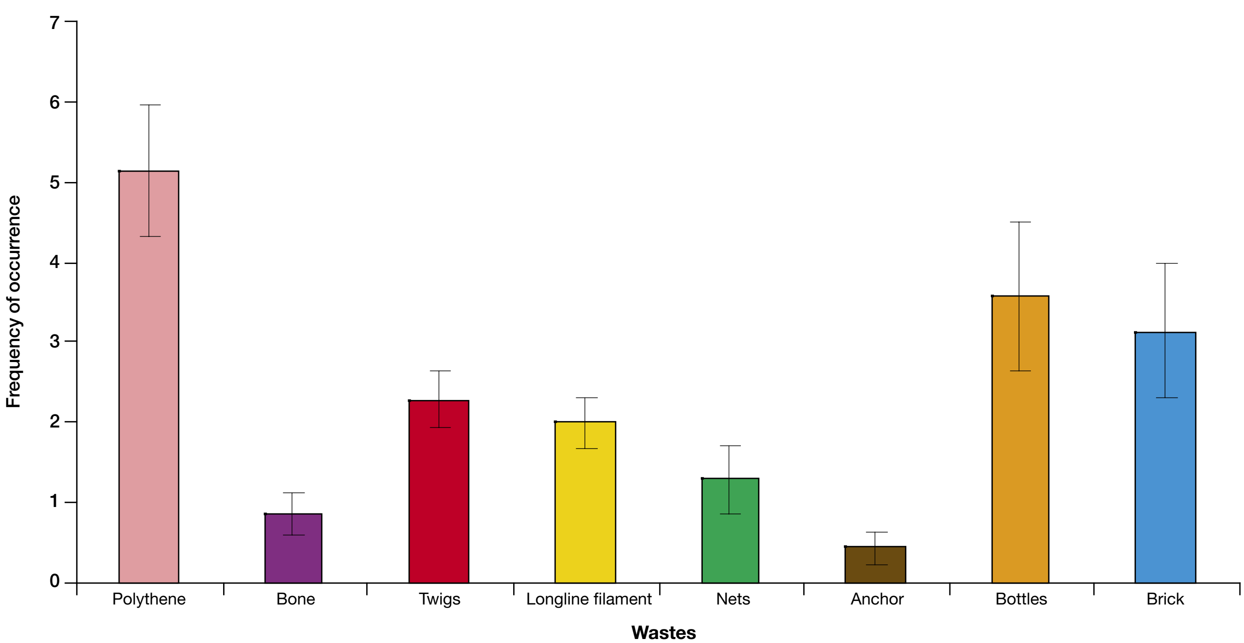

Figure 3. Mean frequency of occurrence (±S.E.) of solid waste material recorded from all transects conducted at Passikudah Reef.

stringent measures to ensure that the plastic material left by tourists does not reach the sea. The plastics are a threat to the growing corals.

Declaring Passikudah a marine sanctuary to protect the growth of the corals should be considered seriously by the authorities. Restricting the use of the beach for a period will no doubt contribute to the regeneration of the degraded coral reefs of Passikudah.

Almost all of the families in the village depend on fishing. There is no information regarding the impact of coral bleaching on fisheries. Studies revealing such impacts and the recruitment of corals are a necessity to conserve the beautiful coral reefs of Passikudah.

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Status of Coral Reefs of the Maldives: Recovery Since the 1998 Mass Bleaching and the Impacts of the Indian Ocean Tsunami 2004

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key words: coral bleaching, reef recovery, tsunami, Maldives

INTRODUCTION

The unprecedented magnitude of the coral bleaching event caused by the El Niño in 1998 affected several coral reef ecosystems throughout the world (Wilkinson *et al.*, 1999). Severe bleaching and subsequent mortality of large areas of coral reefs in the Indian Ocean have been reported (McClanahan, 2000; Edwards *et al.*, 2001; Sheppard *et al.*, 2002).

Coral reefs in the Maldives were severely affected by this El Niño event causing severe bleaching and mortality of more than 90% of the shallow water coral communities (Zahir, 2000; 2002). Reef recovery has been variable among several locations that have been monitored annually since the initial bleaching impact study in 1998. Re-colonisation of fast growing branching growth forms of corals has been reported (Edwards *et al.*, 2001; Zahir, 2002). It is likely the reefs may be modified as a result of the bleaching event changing the community structure. Preliminary findings indicate that the reefs are now being dominated by slow growing coral species such as agariciids and faviids rather than the branching acroporids and pocilloporids that were prevalent previously.

The long term objective of the monitoring program is to examine the processes of reef recovery in terms of coral cover and other benthic communities so that the ecological processes that influence the reef recovery and recolonisation are better understood for science and management needs.

This report describes the patterns of reef recovery since the bleaching event in 1998 over a period of 6 years. In addition, the impacts of tsunami of December 26th, 2004, on one of the regions included in this monitoring program are also described.

METHODS AND SURVEY LOCATIONS

Site Selection

Maldives comprised of 26 natural atolls stretching an area of over 8 000 km^2 in the northern and central Indian Ocean. The 15 sampling sites were chosen in the following 5 regions to cover a large spatial area (figure 1 on next page):

1 **Haa Dhaal** (north and a regional development target)
   Hondaafusi
   Finey
   Hirimaradhoo
2 **Male atoll** (east central with intensive tourism and other commercial activities)
   Emboodhoo
   Bandos
   Udhafush

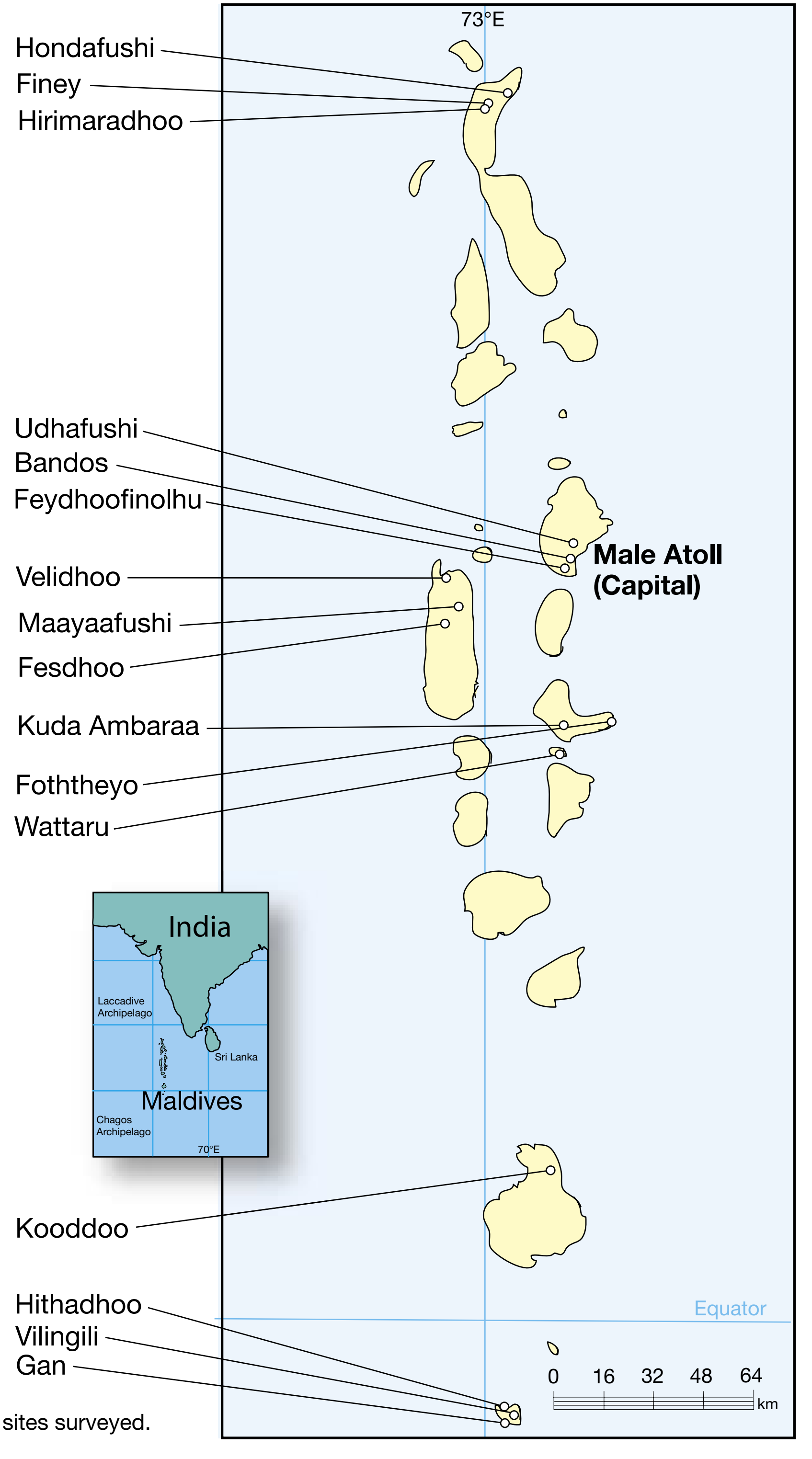


Figure 1. Map of Maldives illustrating the sites surveyed.

3 **Ari atoll** (east central with intensive existing tourism development)
Fesdhoo
Mayaafushi
Velidhoo
4 **Vaavu atoll** (figure 2; south central with a community-based integrated island resource management project underway)
Kuda Ambaraa
Wattaru
Foththeyo
5 **Addu-Gaaf Alif atoll** (south and a regional development target)
Gan
Villigili
Koddoo

These 15 sites were selected initially to ensure that reefs that had been surveyed prior to 1998 bleaching event were included thus providing baseline data against which monitoring data could be compared in order to assess the impacts of the bleaching event (Allison, 1999). In order to be comparable with all previous studies conducted on the coral reefs of Maldives and for logistical efficiency, all surveys were conducted on the reef tops. Surveys were also confined to inner reefs within the atolls because this is where past surveys had been conducted and also because the surge caused by oceanic swells ensures that working in shallow water on outer reefs is usually impossible.

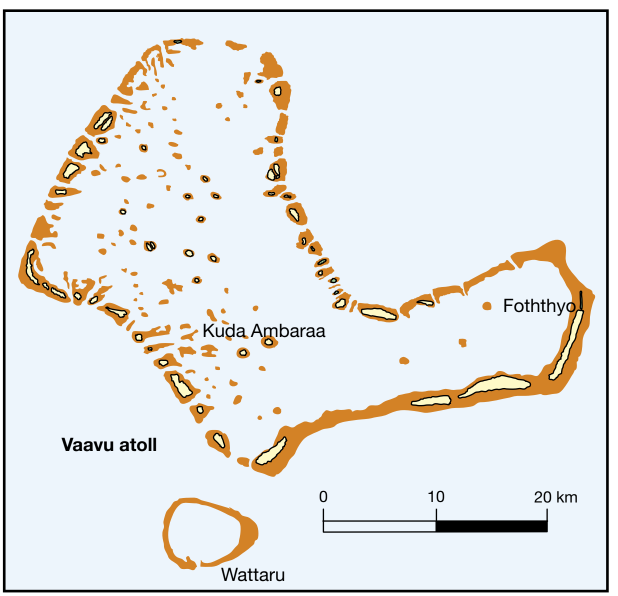


Figure 2. Reef monitoring sites in Vaavu atoll.

Survey Method

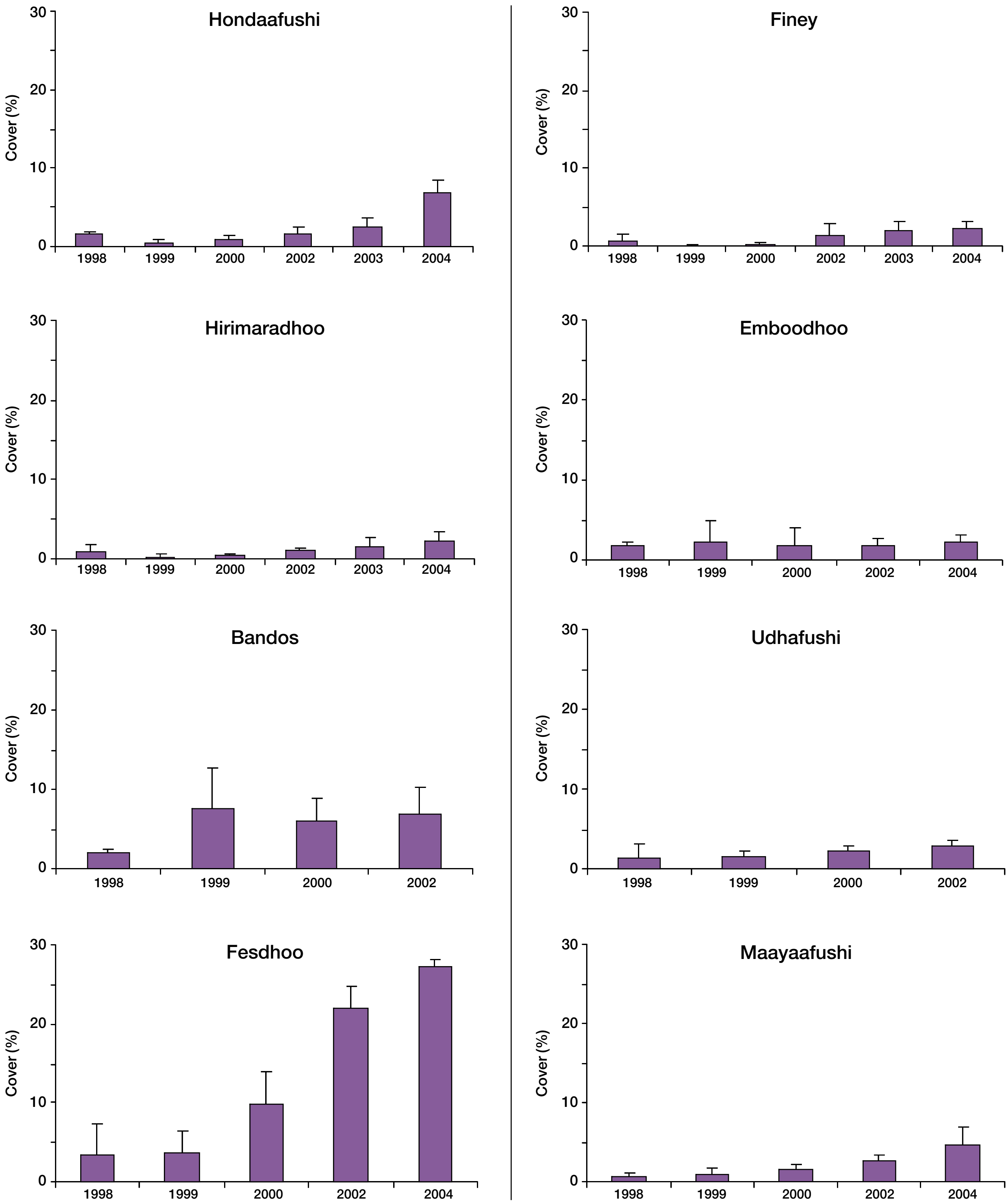
On each reef surveyed, data from three line intercept transects of 50 m (English *et al.*, 1997) were recorded in areas near the location of past survey sites and where physical conditions such as wave action permitted. Occasionally, when it was efficient to do so, a 50 m point intercept transect was used. Initial monitoring was done in 1998 soon after the bleaching, and surveys were repeated at the same sites in 1999, 2000, 2002, 2003 and 2004 to provide an insight into the processes of reef recovery, especially after the bleaching in 1998. Comparative surveys of three sites in Vaavu atoll (Kuda Ambaraa, Wattaru and Foththeyo) were carried out in January 2005 to assess the impact of tsunami of December 2004.

RESULTS

Reef Recovery Since 1998

Coral cover was variable among the 15 sites surveyed (figure 3 on next pages). The average percent cover of live coral for all sites surveyed was approximately 2%. The highest overall cover was recorded at the two southern sites, Gan and Villigili and a western central site, Fesdhoo. Overall coral cover increased approximately 7% in 2004 but the rates of increase varied between different sites. A 3–4 fold increase in coral cover was recorded at Gan, Villigili and Koddoo between 1998 and 2004. An 8 fold increase in coral cover from 3.34% to 27.16% was observed at Fesdhoo, between 1998 and 2004.

Overall, there was an increasing trend in the live coral cover at the 15 sites surveyed over the past 6 years since the 1998 bleaching event. In 1998, coral cover ranged be-


Hondaafushi
Cover (%)
30
20
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Finey
Cover (%)
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Hirimaradhoo
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Emboodhoo
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Udhafushi
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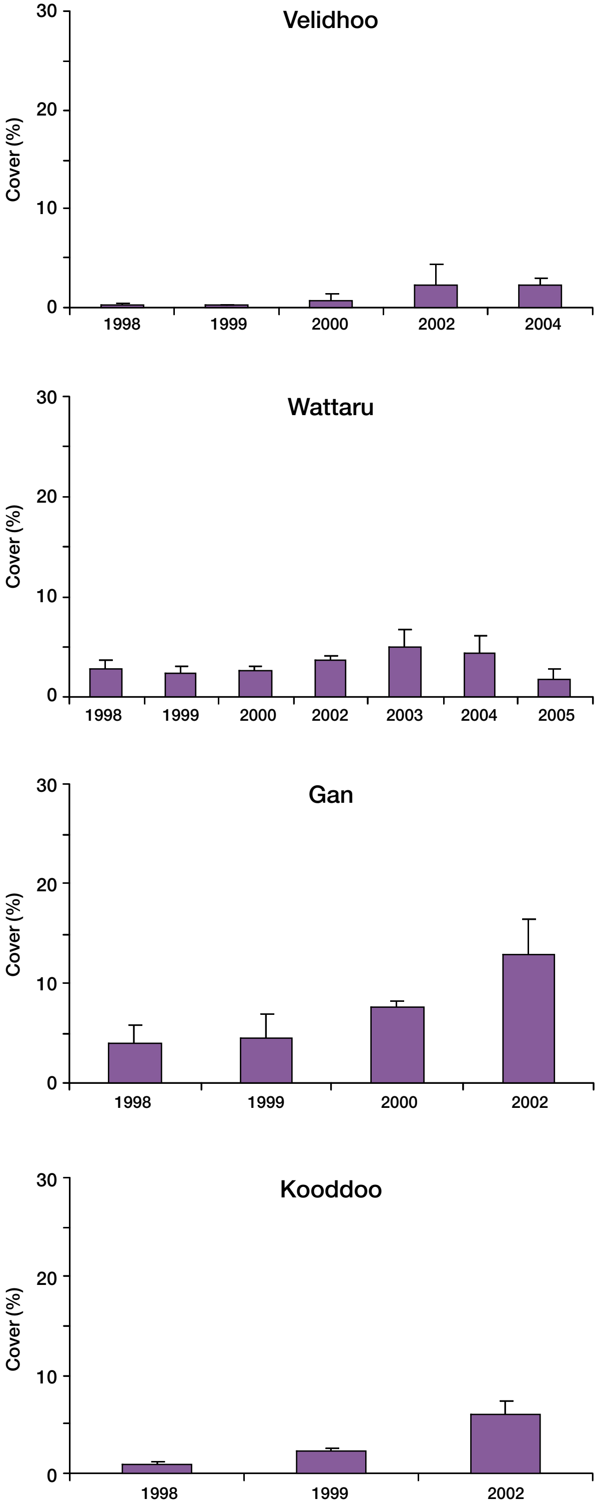

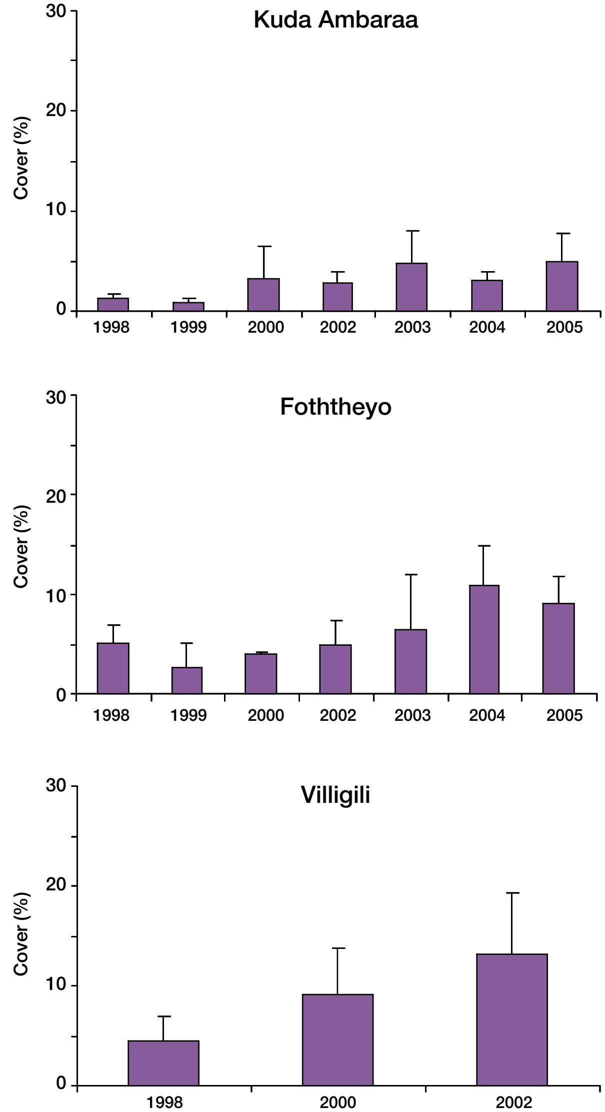

Figure 3. Coral reef recovery during the past 6 years (1998-2005) at the 15 permanent monitoring sites. Note the sampling frequency is not equal for all sites. Kuda Ambaraa, Wattaru and Foththeyo graphs include the coral cover of 2005 (Post tsunami). Figures are mean percent live coral cover and error bars illustrate the standard deviations.

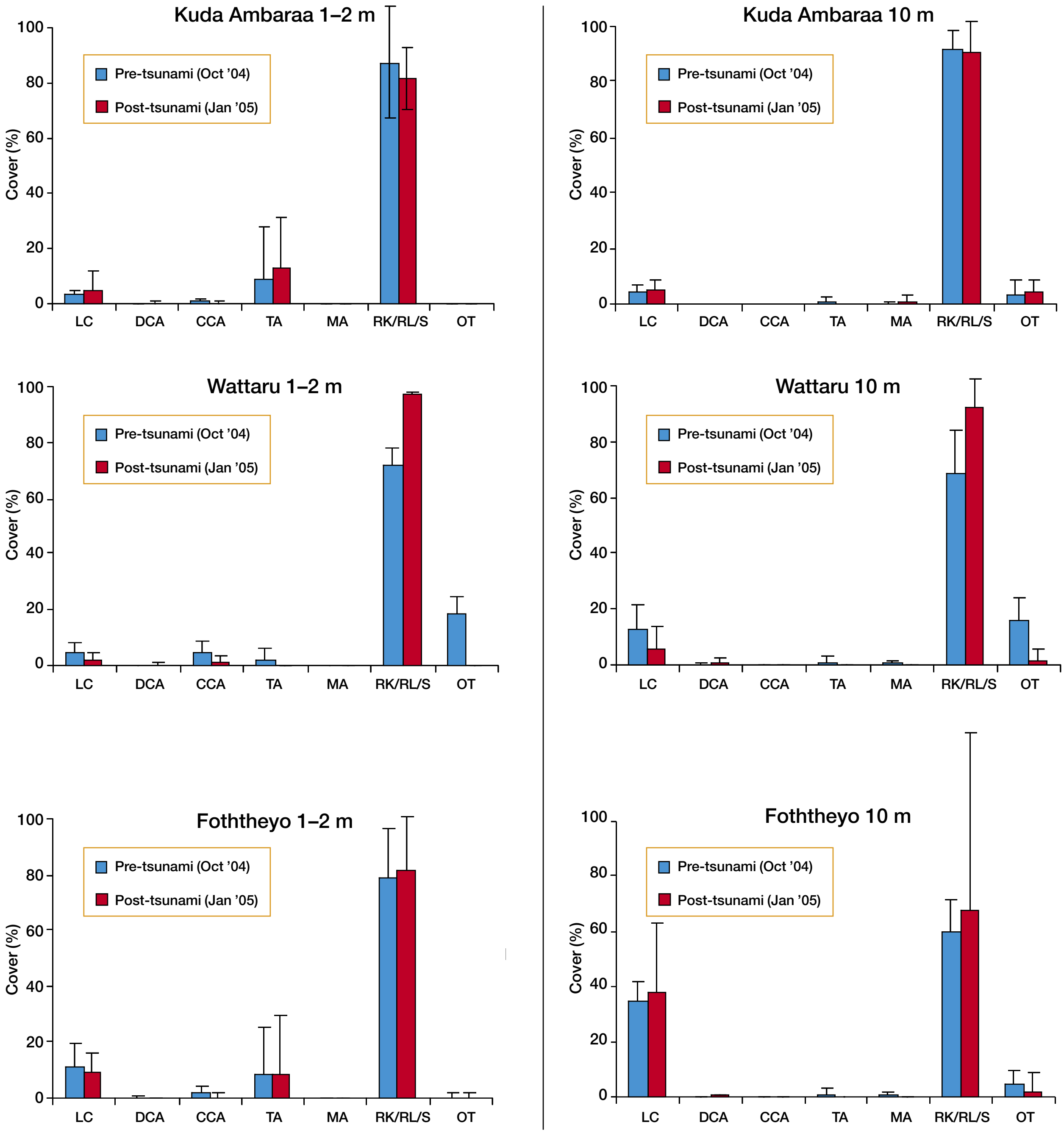

Figure 4. Comparison of the benthic community structure at three monitoring sites in Vaavu atoll before and after the tsunami of 26th December 2004. LC = Live coral, DCA = Dead coral algae, CCA = Crustose coralline algae, TA = Turf algae, MA = Macro algae, Rk/Rl/S = rock/rubble/sand and OT = Others. Values are mean percent cover. Error bars are 95% CL.

tween 0.2% at Velidhoo and 5% at Foththeyo. In 2004, the coral cover is still low at Emboodhoo (2.23%) but is moderate at Fesdhoo (27.16%).

The sites where the impacts of the tsunami were assessed, Kuda Ambaraa, Wattaru and Foththeyo at Vaavu atoll, showed variable coral cover in 2005 (post tsunami) compared with 2004 (pre tsunami). Kuda Ambaraa showed a slight increase in coral cover, from 3.09% ± 0.85 SD to 4.91% ± 2.80 SD and Foththeyo a slight decrease 10.81% ± 4.00 SD to 9.04% ± 2.77 SD. Wattaru suffered the most severe damage to the reef structure shown by the decrease in coral cover from 4.43% ± 1.75 SD in October 2004 (pre tsunami) to 1.80% ± 0.96 SD in January 2005 (post tsunami).

Tsunami Impact at Vaavu Atoll

Analysis of the LIT data obtained from the three sites surveyed at Vaavu atoll showed variable results among the sites. Pre and post tsunami surveys of these sites indicated an increase in live coral cover both at Kuda Ambraa and Foththeyo but a decrease in Wattaru (figure 4). Coral cover increased from 3.1% to 4.9% at 1–2 m and 4.0% to 5.2% at 10 m depth at Kuda Ambaraa. At Foththeyo, coral cover increased from 34.4% to 38.2% at 10 m but decreased from 11% to 9% at 1–2 m depth. In contrast, at Wattaru, coral cover decreased at both depths, from 4.4% to 1.8% at 1–2 m and from 13.2% to 6% at 10 m. The amount of bare substrate also increased at Wattaru compared with the other two sites.

The physical damage to the reef substrate varied significantly between the three sites. The number live coral colonies varied between sites; Kuda Ambaraa (32), Wattaru (27), Foththeyo (102) at 10 m and Kuda Ambaraa (63), Wattaru (22) and Foththeyo (164) at 1–2 m depth. The degree of the physical damage to the coral colonies was contrastingly different at the three sites. The reef at Wattaru exhibited considerably more damage than was observed at Kuda Ambaraa or Foththeyo (figure 5). The average number of live coral colonies also varied among the three sites; 10, 9, 34 at 10 m and 21, 7, 54 at 1–2 m for Kuda Ambaraa, Wattaru and Foththeyo respectively. The least number of live coral colonies was recorded at Wat-

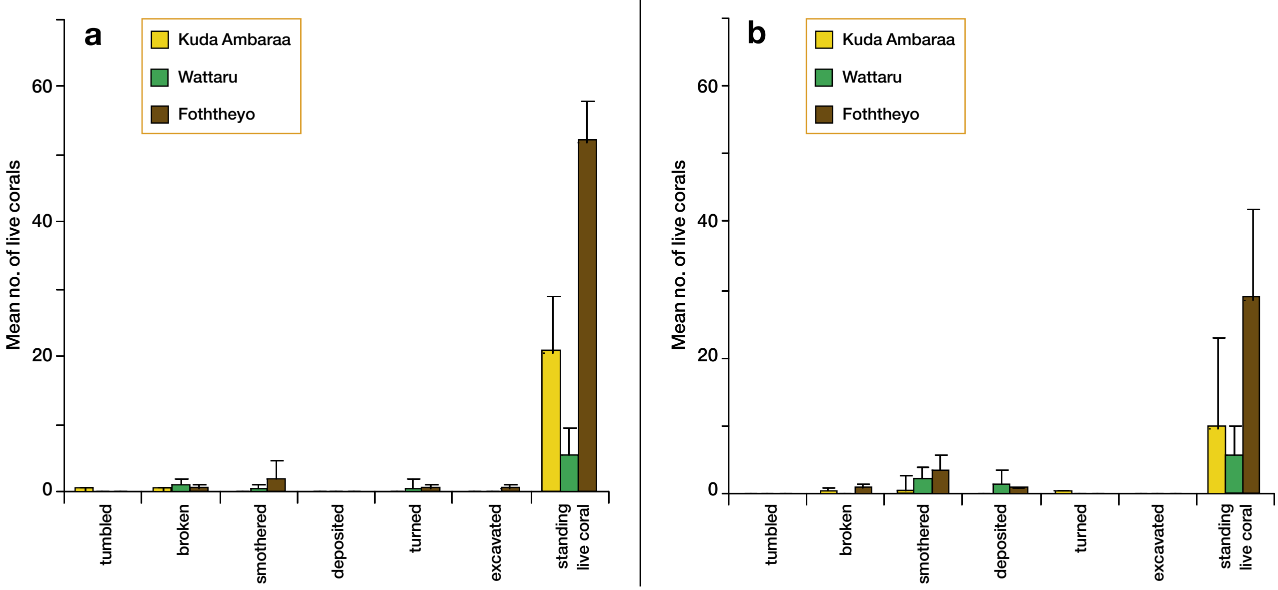

Figure 5. Proportion of the physical damage to the reef structure at three monitoring sites in Vaavu atoll. Depths are plotted separately, a = shallow (1–2m), b = deep (7–10meters).

taru. The percentage of damage to the coral substrate at Wattaru ranged from 27–37% for both depths (1–2 m and 10 m) compared with 3–9% for Kuda Ambaraa and 5–15% for Foththeyo.

The extent of mortality and physical damage to the live coral and reef substrate was also assessed for these three sites. A total of 410 coral colonies were used to assess the extent of mortality mainly due to smothering by sediments. Approximately 10% of the coral colonies were affected due to sedimentation inflicting varying levels of mortality. Among this 10%, only approximately 4% of the colonies sustained more than 50% mortality (partial death of coral colony, loss of tissue and colouration) whilst the remaining 6% exhibited less that 50% mortality. The extent of mortality was highest at Wattaru, moderate at Foththeyo and lowest at Kuda Ambaraa (figure 6).

DISCUSSION

The overall coral cover increased from approximately 2 to 28% between 1998 and 2005 among the 15 sites sur-

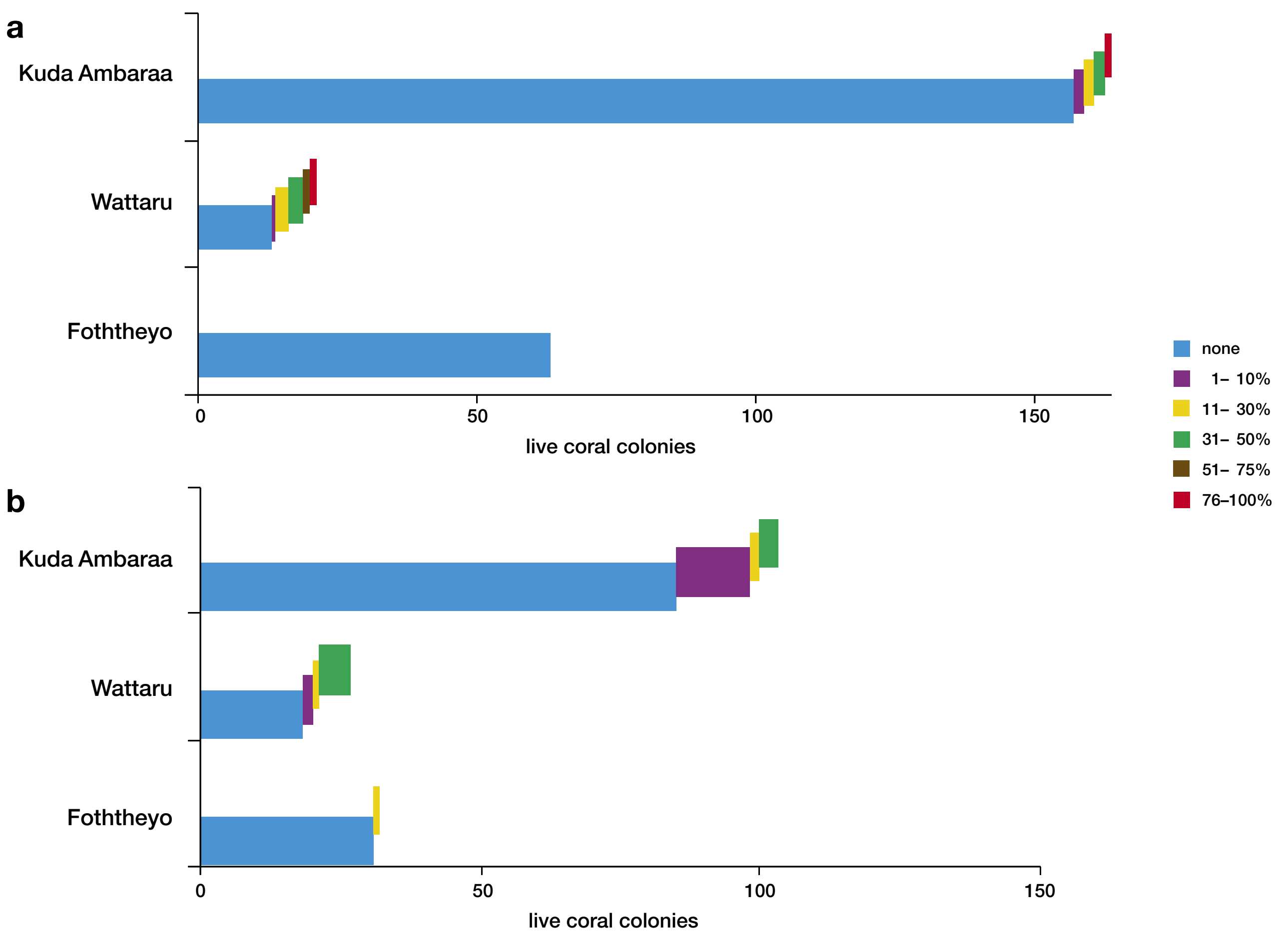

Figure 6. Coral mortality caused by smothering by sediment stirred up by the tsunami at three monitoring sites in Vaavu atoll. Coral mortality for each depth plotted separately, a = shallow (1–2m), b = deep (7–10m).

veyed. Reef recovery in terms of coral cover has been variable across the 15 sites. The 1999 survey indicated less live coral cover in several sites indicating subsequent mortality since the bleaching and the absence of measurable recovery at these sites. The subsequent decline in coral cover in 1999 and the variable recovery at several sites may be a good indication of the severity of the bleaching during the 1998 El Niño.

Strong recruitment of corals has been reported from a parallel study carried out to assess the reef recovery processes since the bleaching event in 1998 (Zahir *et al.*, 2002). Size frequency analysis of the recruits from this study indicated that the coral community is dominated by the influx of new and small recruits. Although the coral recruits are dominated by encrusting forms (e.g. agariciids, mostly *Pavona* spp.) indicating that a shift in coral community composition from the previously dominant branching and fast growing forms (e.g. acroporids), there is a good evidence that the reefs are recovering slowly. The recruitment success at these study sites also indicates that the reefs are not recruitment limited and a source of larvae is in the vicinity, although variable environmental factors are likely to affect the rate of reef recovery, especially on reefs recovering from severe disturbance.

The overall physical damage caused by the tsunami of 2004 was negligible (AusAID and MMRC, 2005). Broad-scale assessment of the damage of the tsunami to reefs showed similar patterns of coral cover through out the Maldives that are comparable to the results of this long- term monitoring program (figure 7).

One hundred and twenty four reef sites were surveyed in seven atolls, covering about 170 km of reef margin. Although there was damage to coral and movement of sediments in all regions these perturbations varied in extent and intensity. Even so, surveys generally indicated that direct damage to reefs from the tsunami was minor. However, the reefs of the Maldives are in the early stages of recovery from the massive bleaching in 1998 and the most significant consequence of the tsunami may be to hamper this process. Many survey sites had a light coating of sand. Small coral recruits are most vulnerable to smothering by sand and rubble and even a light coating of sand may make reef surfaces unsuitable for future settlement. This powdering effect imposes a considerable concern that may hamper the survival of coral and other benthic recruits in these locations. A recruitment failure due to recurring environmental perturbation would have a significant impact on the reef recovery process.

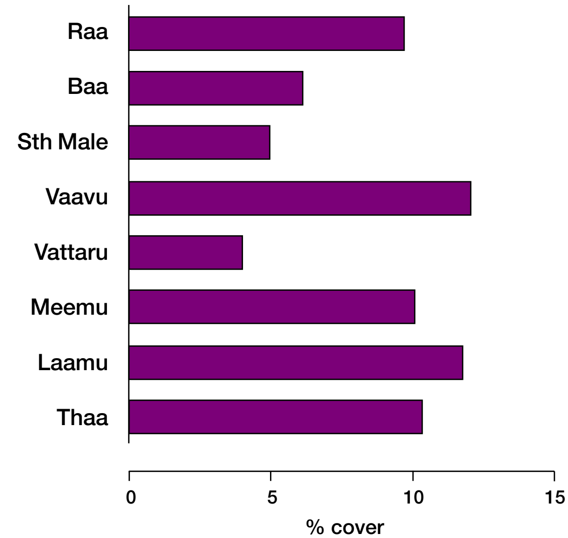

Figure 7. Live coral cover at each of the atolls surveyed by AusAID/ MRC to assess the impacts of the tsunami.

In general, little is known of the biodiversity or prior ecosystem status and past changes on coral reefs of the Maldives, especially after the tsunami. The results presented here provide important information with regard to the damage caused by the tsunami where the sites are directly compared before and after the tsunami.

ACKNOWLEDGEMENTS

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Status of Coral Reefs of the Tuticorin Coast,

Gulf of Mannar, Southeast Coast of India

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key words: Tuticorin coast, coral status, coral diversity

INTRODUCTION

The coral reefs of the Gulf of Mannar along the Indian coast are mainly found scattered around the 21 islands that are distributed in an 8 km wide band between Pamban and Tuticorin (figure 1 on next page). Tuticorin (Lat. 8° 45' N, Long. 78° 10' E) is located at the southern end of the Gulf of Mannar. The floral components comprises of economically viable species of seaweeds such as *Gracilaria* sp., *Gelidiella* sp., *Caulerpa* sp., *Sargassum* sp. and *Turbinaria* sp. The sea grass communities of this region are the most diverse in India with the highest number of sea grass species recorded, providing important feeding grounds for the endangered *Dugong dugon.* The Tuticorin group includes four islands namely, Vaan (Lat 8° 50' N, Long. 78° 13' E), Koswari (Lat. 8° 52' N, Long.78° 13' E), Kariyachalli (Lat. 8° 57' N, Long. 78° 15' E) and Vilanguchalli (Lat. 8° 56' N, Long. 78° 15' E). As a result of soil erosion caused by excessive coral mining, Vilanguchalli now lies 1 m below mean low tide level. The islands have fringing and patch reefs around them. Narrow fringing reefs are located mostly at a distance of 100–150 m from the islands. Patch reefs rise from depths of 2–9 m and extend to 1–2 km in length with widths as much as 50 m.

There are five important fishing villages depending on the Tuticorin coast namely, Pudhukadarkarai (Inico Nagar), Thirespuram, Siluvaipatti, Vellapatti and Tharuvaikulam. There are 42 551 registered fishermen from these villages who depend solely on fishing around these four islands and along the coast for their livelihood. Fishing is carried out mainly in and around Tuticorin. The main fishery targets finfishes and shellfishes.

Fishing is mainly done by trawling or using gillnets or traps. The reef fishery is not very important in terms of total landings or earnings when compared with other fisheries, such as the demersal fishery in the shallow coastal areas or the tuna fishery. Many commercially exploited shoaling fishes, like sardines, mackerels, anchovies, snappers, and fast swimming pelagic forms, like tuna, billfish and sailfishes, are abundant in this region, forming a major fishery. However, it is expected that the focus on the Tuticorin fisheries is likely to change due to increasing demand from foreign markets, for high quality reef fishes, such as grouper, snapper, and shellfishes, like lobsters and crabs.

The reefs of Tuticorin are under severe threat due to natural and human interference. The local fisherfolk of this region have traditionally had a close relationship with the sea resulting in strong cultural and economic links with maritime activities such as fishing, pearl (*Pinctata fucata*) and chank (*Xancus pyrum*) diving. The over-exploitation of seaweeds, sacred chanks, pearl oysters and seahorses by the locals has made them commercially threatened. Coral mining and the use of destruc-

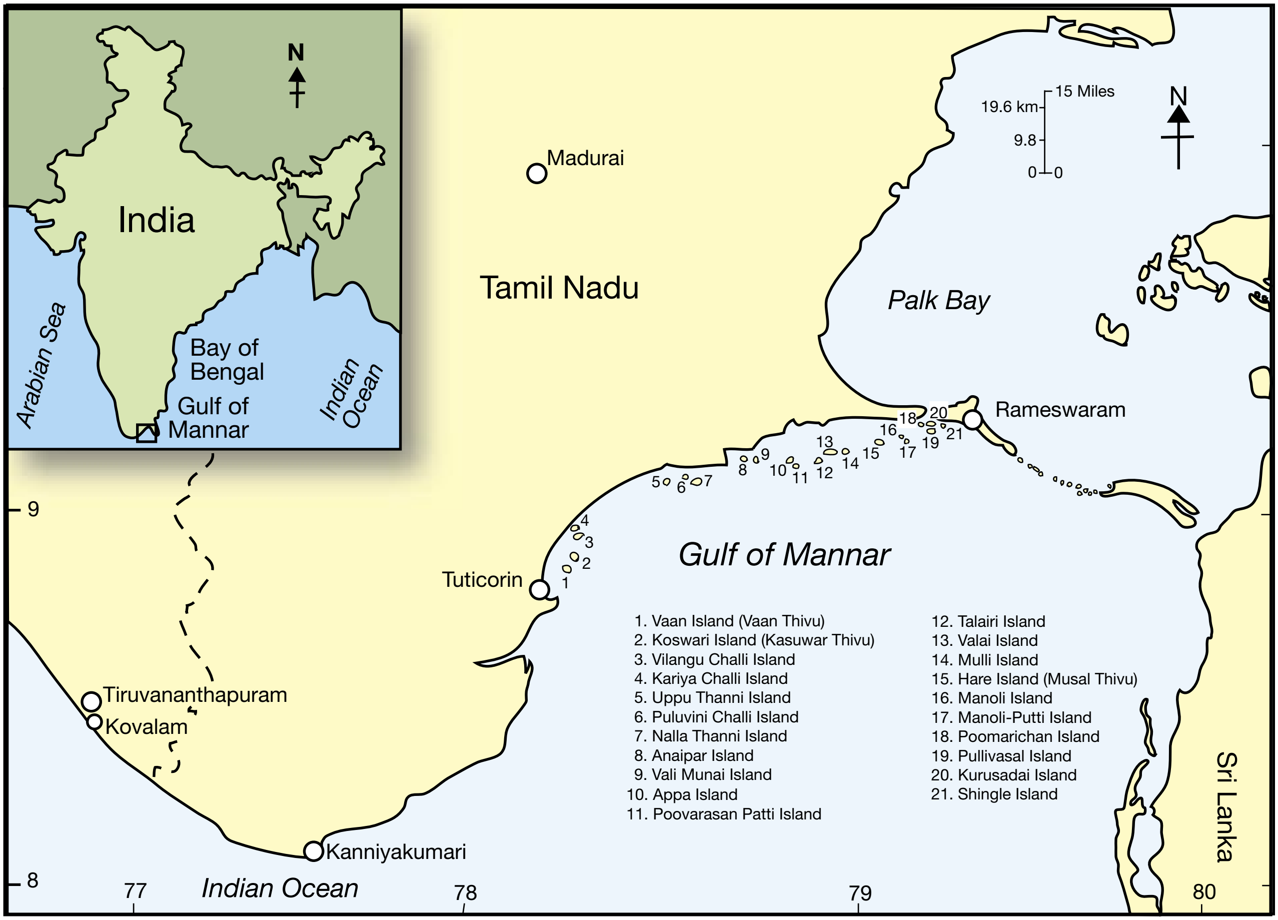

Figure 1.

tive fishing methods are prevalent in the area (Patterson, 2002).

In this present survey, the status of corals reefs along the Tuticorin coast was assessed, as an initiation of a longer-term monitoring effort, mainly around the islands and mainland areas

METHODS

Study Area

The Tuticorin region consists of 7 islands, which are categorized into two groups namely, the Tuticorin Group (4 islands) and the Vembar Group (3 islands). The Tuticorin Group includes two patch reefs, one found near Tharuvaikulam fishing village, which is approximately 300 m offshore and 3 km long, and another found near the Tuticorin harbour area, which is 1.2 km offshore and 5 km long. Each patch reef is found at different depths.

Surveys were conducted between January 2003 and February 2004 at each of the four islands of the Tuticorin group and the two patch reefs, Harbour and Tharuvaikulam.

Line Intercept Transect Method

The surveys were initiated by mapping the patch reefs and island reef areas, using manta tows (Done *et al.*, 1982).

After confirming the positions of the coral reef areas, the sessile benthic community were surveyed using the line intercept transect (LIT) (English *et al.*, 1997). Transects of 30 m in length were laid along the reef area, parallel to the depth contours of the reefs, using a flexible fibreglass measuring tape. The tape was laid at depths ranging between 1 m and 6 m. Depending on the size of the reefs, between 8 and 20 transects were laid on each of the four islands and the two patch reefs. Divers using SCUBA recorded each change of life-form category (table 1),

Table 1. Lifeform categories and codes.

Categories		Code	Notes/Remarks
Hard Coral			
Dead Coral		DC	recently dead, white to dirty white
Dead Coral with Algae		DCA	this coral is standing, skeletal structure can still be seen
Acropora	Branching	ACB	at least 2° branching, e.g. *Acropora palmate*, *A. formosa*
	Encrusting	ACE	usually the base-plate of immature *Acropora* forms, e.g. *A. palifera* and *A. cuneata*
	Sub massive	ACS	robust with knob or wedge-like form e.g. *A. palifera*
	Digitate	ACD	no 2° branching, typically includes *A. humilis*, *A. digitifera* and *A. gemmifera*
	Tabular	ACT	horizontal flattened plates e.g. *A. hyacinthus*
Non – *Acropora*	Branching	CB	at least 2° branching e.g. *Seriatopora hystrix*
	Encrusting	CE	major portion attached to substratum as a laminar plate e.g. *Porites vaughani*, *Montipora undata*
	Foliose	CF	Coral attached at one or more points, leaf-like, or plate-like appearance e.g. *Merulina ampliata*, *Montipora aequituberculata*
Submassive		CS	tends to form small columns, knobs, or wedges e.g. *Porites lichen*, *Psammocora digitata*
	Mushroom	CMR	solitary, free-living corals of the *Fungia*
	Heliopora	CHL	blue coral
	Millepora	CME	fire coral
	Tubipora	CTU	organ-pipe coral, *Tubipora musica*
Other Fauna			
Soft Coral		SC	soft bodied coral
Sponge		SP	
Zoanthids		ZO	examples are *Platythoa*, *Protopalythoa*
Others		OT	Ascidians, anemones, gorgonians, giant clams etc.
Algae	Algal Assemblage	AA	consists of more than one species
	Coralline Algae	CA	
	Halimeda	HA	
	Macroalgae	MA	weedy/fleshy browns, reds, etc.
	Turf Algae	TA	lush filamentous algae, often found inside damselfish territories
Abiotic	Sand	S	
	Rubble	R	unconsolidated coral fragments
	Silt	SI	
	Water	WA	fissures deeper than 50 cm
	Rock	RCK	
Other		DDD	Missing data

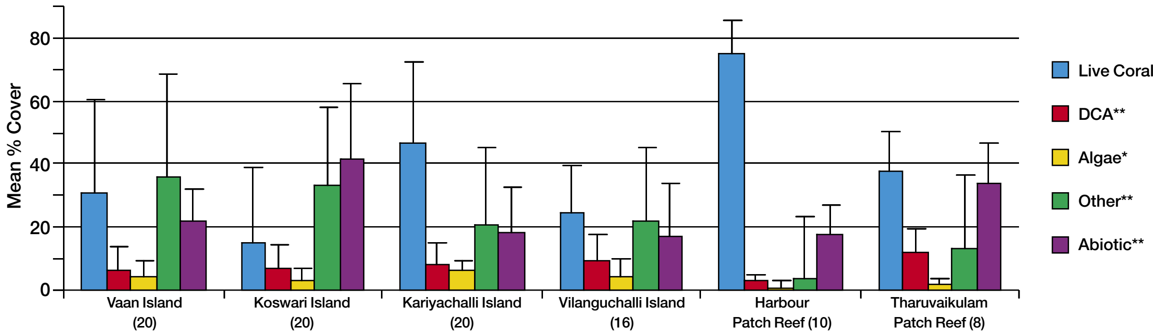

Figure 2. Mean percent cover (±SD) of each of the main habitat components at each site surveyed along the coast of Tuticorin. Differences in the cover of each substrate type between sites was tested using Kruskal-Wallis rank test (** indicates a significance level of $p<0.01$, while * indicates $p<0.05$). Numerals in parentheses indicate the number of transects recorded at each site.

along the transect. When *in situ* identification of coral genera was not possible, samples were collected for analysis in the laboratory. The percent cover of each life-form category was then calculated following the method of English *et al.* (1997). Differences in percent cover of the main benthic categories between the reef areas were tested using Mann-Whitney U-tests. Further, to compare the similarity of the benthic composition between the reef areas, a multivariate analysis was used. Multi Dimensional Scaling (MDS), based on rank similarity matrices using Euclidian Distance measure (Clarke, 1993) was performed on square root transformed data. The variables Halimeda (HA), Zoanthids (ZO) and Coralline algae (CA) were excluded from the analysis due to their low abundances. Differences between groups of samples were subsequently tested using the ANOSIM permutation test (Analysis of Similarities) (Clarke & Green, 1998).

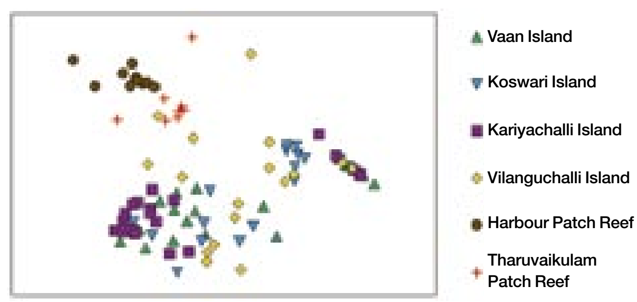

Figure 3. MDS plot illustrating the relative similarity between the benthic communities of each site, based on Euclidian distance similarities and square root transformed data. Each point represents one sample. The distances between the points display the relative similarities between the different samples. The closer the points are to each other, the more similar the samples.

RESULTS

Current Status of the Reefs

The total cover of each benthic life form recorded from each site surveyed is presented in figure 2. The multivariate analysis showed significant differences in the composition of the benthic community ($p>0.05$, Global $Rho>0.3$) between sites. The greatest differences occurred between the patch reefs and the island sites ($p<0.05$, $Rho>0,0,4–0,8$), and were caused by relatively small areas on each patch reef occupied by mono-specific stands of *Turbinaria mesenterina* and *T. peltata*. The differences between sites are clearly illustrated in figure 3 by the distinct grouping of the samples from these reefs.

The Harbour patch reef had significantly greater live coral cover ($p<0.001$) than all other sites, while the reef at Koswari Island had significantly less live coral than all other reefs except Vaan Island ($p<0.05$, Mann-Whitney U-test; figure 2).

The coral communities of the four island sites were dominated by massive corals (CM) followed by foliose (CF) forms and tabulate acroporids (ACT) (figure 4). The two patch reefs, on the other hand, were totally dominated by foliose *Turbinaria* spp. Branching acroporids (ACB) and other branching (CB) and encrusting (CE) forms contributed least to the cover of live coral suggesting that these corals are the worst affected by environmental stress.

Massive corals contributed the greatest cover of live coral at Kariyachalli Island, Vaan Island and Koswari Island. Foliose corals were the dominant life form at both patch reefs and Vilanguchalli Island. The highest cover of encrusting corals occurred at Koswari Island. The greatest percent cover of branching and tabulate acroporids occurred at Kariyachalli Island, while the greatest cover of branching non-acroporid corals was recorded at Vaan Island. Sub-massive corals were totally absent in the Tuticorin group of islands. Tabulate acroporids were only absent at Koswari Island.

The Scleractinian Fauna of Tuticorin

Pillai (1998) provided a comprehensive account of the coral fauna of the Gulf of Mannar in which 94 species belonging to 37 genera were reported. The scleractinian corals of the Tuticorin have been very little studied in the past. The first mention of corals from this area is that of Pillai (1972) who reported 21 species. During the past 30 years, little work has been conducted on the coral diversity of the region. The latest report is that of Patterson (2002) who reported 22 species from many localities in the Tuticorin region. However, the species composition of both reports shows little coincidence.

The present survey recorded 53 species of coral belonging to 22 genera (table 2 on next pages), of which 50 species from 19 genera were hermatypic and the remainder were ahermatypic. Among the 53 species, 42 were recorded for the first time in Tuticorin region and, of these, 10 species were recorded in the Gulf of Mannar for the first time.

DISCUSSION

Island Reefs

Among the islands, Kariyachalli exhibited the greatest live coral cover (46.6%) while Koswari exhibited the least

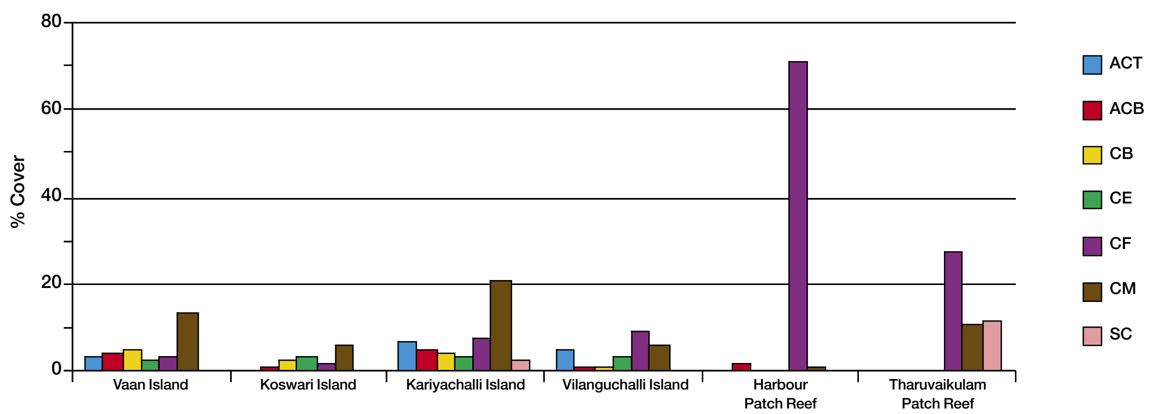

Figure 4. Percent cover of each live coral category, including soft coral, recorded from the Tuticorin group of islands and patch reefs in the Gulf of Mannar.

(15.29%). Generally, massive corals (11.32%) dominated the reefs of the Tuticorin Group of islands.

According to Venkataraman (2000), the bleaching event in 1998 destroyed most shallow water corals in the Gulf of Mannar, causing a significant reduction in the live coral cover, particularly among branching *Acropora* species of Tuticorin which declined to less than 1% of the total cover, and the local extinction of other branching species. Contrary to his report, the present survey, recorded 6.35% branching *Acropora* (ACB) and 3.26% cover of non-acroporid branching corals (CB) in this region and no signs of any mass mortality might have occurred when sea surface temperatures were elevated across the Indian Ocean in 1998.

Our results clearly demonstrate that, among the reefs of the four islands surveyed, only Koswari Island exhibited a low percentage cover of live coral (15.29 %). Coral mining around this island during the past two decades has reduced both the live cover and diversity of coral. At present, only the southwest corner of the island has live coral. Bleaching has not been noticed in Tuticorin Group of islands and, as a result, the damage to reefs in the re-

Table 2. List of scleractinian corals found along the Tuticorin coast

Species Name	Vaan Island	Koswari Island	Kariyachalli Island	Vilanguchalli Island	Harbour Patch reef	Tharuvaikulam Patch reef
Family: Pocilloporidae						
Pocillopora damicornis	–	+	+	+	–	–
Family: Acroporidae						
Acropora formosa	+	+	+	+	+	–
Acropora intermedia	–	–	–	–	+	–
Acropora valenciennesi	–	–	–	–	+	–
Acropora microphthalma	–	–	–	–	+	–
Acropora sp. novo	–	–	–	–	+	–
Acropora corymbosa	–	–	+	–	–	–
Acropora nobilis	+	–	+	+	+	–
Acropora humilis	+	–	–	–	+	–
Acropora valida	+	–	+	–	+	–
Acropora hemprichi	+	–	+	–	–	–
Acropora stoddarti	–	–	–	–	+	–
Acropora diversa	–	–	+	–	–	–
Acropora cytherea	+	+	+	+	+	–
Acropora Pillai sp.nov	–	–	–	–	+	–
Montipora subtilis	–	–	+	–	+	–
Montipora digitata	+	+	+	+	–	–
Montipora divaricata	+	+	+	+	–	–
Montipora jonesi	+	–	+	–	–	–
Montipora hispida	+	+	+	+	+	–
Montipora foliosa	+	–	+	+	–	–
Family: Siderastreidae						
Coscinaraea monile	+	–	+	–	+	+

Species Name	Vaan Island	Koswari Island	Kariyachalli Island	Vilanguchalli Island	Harbour Patch reef	Tharuvaikulam Patch reef
Family: Fungiidae						
Cycloseris cyclolites	+	–	–	–	–	–
Family: Poritidae						
Goniopora minor	–	–	+	–	–	+
Goniopora stutchburyi	+	+	+	+	+	+
Porites solida	+	–	+	+	+	–
Porites lutea	+	+	+	+	+	+
Porites lichen	–	–	+	–	–	–
Family: Faviidae						
Favia pallida	+	+	+	+	+	+
Favia favus	+	–	+	+	+	+
Favia matthaii	–	–	+	–	–	–
Favites abdita	+	+	+	+	+	–
Favites complanata	–	–	+	–	–	–
Favites flexuosa	+	–	+	–	–	–
Goniastrea pectinata	+	+	+	+	+	+
Goniastrea retiformis	+	+	+	+	+	–
Platygyra daedalea	+	+	+	+	+	–
Platygyra sinensis	–	–	+	–	–	–
Leptoria phrygia	+	–	+	+	–	–
Hydnophora microconos	+	–	+	+	–	–
Hydnophora exesa	+	+	+	+	–	–
Leptastrea transversa	–	–	+	–	–	–
Leptastrea purpurea	+	–	+	–	–	–
Cyphastrea serailia	+	+	+	+	+	–
Echinopora lamellosaa	+	–	–	–	–	–
Plesiastrea versipora	–	–	+	–	+	–
Family: Oculinidae						
Galaxea fascicularis	–	–	+	–	–	–
Family: Mussidae						
Symphyllia radians	–	–	+	–	–	–
Family: Caryophylliidae						
Polycyathus verrilli	–	+	–	–	–	–
Family: Dendrophylliidae						
Dendrophyllia indica	+	–	–	–	–	–
Turbinaria crater	+	+	+	+	+	+
Turbinaria peltata	+	+	+	+	+	+
Turbinaria mesenterina	+	+	+	+	+	+

+ Recorded. – Not recorded.

gion is attributable to mining and destructive fishing particularly bottom trawling in the reef area.

Patch Reefs

The results of surveys of the coral fauna conducted in this study are the first obtained from the patch reefs of Tuticorin. The two patch reefs, Tharuvaikulam and Harbour, appear to be healthy and the latter supports 10 species that have not previously been recorded in the region. There was no sign of impacts attributable to the 1998 bleaching event on these patch reefs, which are mostly covered with cup-shaped coral such as *Turbinaria mesenterina* and *T. peltata* and some massive and branching corals. However, sediment accumulating in the cup-shaped structures of colonies of *Turbinaria* seem reduce their growth rates resulting in a slow death.

Branching corals were totally absent from the patch reef of the Harbour area while encrusting corals were poorly represented. On the Tharuvaikulam patch reef, encrusting corals, branching and tabulate forms of *Acropora*, and non-acroporid branching corals were absent.

The commercially important seaweeds are found growing on the surface of the dead coral beds. Sea grass beds are common near all the islands and support a number of associated fauna such as sea anemones, the bivalve *Pinna* sp., and a number of species of starfish, sea cucumbers and sea urchins. Other life forms such as sponges, soft corals and reef fishes were also recorded.

Management Issues

The Gulf of Mannar is one of the most heavily stressed coral reef regions in India, with impacts from destructive fishing, pollution, coral mining, industrial effluent discharges, and domestic sewage pollution. Local fishermen complain that fish catches have declined both on the near shore and offshore coral banks and islands. The islands of Tuticorin have been affected by industrial pollution released by large number of factories that are located along the coast. Sewage has also resulted in the overgrowth by mats of green algae on the dead corals. Bottom trawling is also a major threat to the reefs. Trawlers are now fitted with a wheel at the footrope (Roller madi), which aids in jumping over coral reefs thereby causing destruction to the entire reef. Coral collection for the production of lime and the damage by trawlers continue to degrade the coastal reefs. Black, white and yellow band coral diseases have also been observed.

The live export of lobsters and fishes from this area in the recent years is also causing damage to corals. Fishermen set lobster and fish traps on all available coral areas. The setting and retrieving of cages causes damage to live corals. Ola valai, a type of beach seine, is not intrinsically destructive while the process of shore seine operation is largely responsible for the destruction of new colonies of corals.

It is evident that some of the islands, which are close to the mainland, have fewer live corals. Banning illegal coral mining and destructive fishing practices is the only way to protect coral diversity.

ACKNOWLEDGEMENTS

The authors are highly thankful to Mr. N.K. Raghupathy, Chairman, Tuticorin Port Trust, Dr. Sukhdev, Chief Wildlife Warden, Tamil Nadu Forest Department and Mr. V. Naganathan, Wildlife Warden, Gulf of Mannar Marine National Park for permission to conduct this research and Professor Olof Linden, Co-ordinator, CORDIO for financial support.

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Indian Ocean Island – Summary

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key words: Indian Ocean Islands, coral reefs, coral bleaching, tsunami, climate change

INTRODUCTION

The Indian Ocean Islands CORDIO node consists of Comoros, Madagascar, Mauritius and Seychelles, all located within the western Indian Ocean. Although their combined total Exclusive Economic Zone (EEZ) exceed 4.1 million km², coral reefs cover only about 3 500 km², with the largest area being in Madagascar (see table 1).

Significant areas of coral reefs occur in the sub-region, in particular within the Seychelles archipelago, Madagascar, Mauritius, and Comoros. Madagascar has the largest area of coral reefs in the sub-region, mostly dominant along the eastern coast. Most of the granitic islands of the Seychelles are encircled by discontinuous, fringing reefs. Along the east coast of Mahé, reef flats reaching over 2 km in width and terminating in a high algal ridge which descends down a reef slope to a floor typically at 8 to 12 m are observed. In the coral islands, the types of reefs are highly varied from true atolls, raised atolls to submerged or partially submerged atolls and sand banks. Mauritius is almost completely encircled by fringing reefs, with substantial lagoon and barrier reef development on the east and southwest coasts (Salm, 1976). Rodrigues Island (Mauritius) is totally encircled by reefs, with wide shallow reef flats extending from the shore, with its widest extent reaching 10 km in the west (Spalding *et al.*, 2001). The main types of reefs in the Comoros are discontinuous fringing reefs, ranging from 15 m to several kilometres from the coastline. In Comoros, reef cover is most extensive on the island of Anjouan (Scetauroute, 1999).

There are a total of about 14 marine protected areas (MPAs) in the region, covering over 800 km² of ocean.

Table 1. Estimated coral cover in the Indian Ocean Islands

Countries	Land Area (km²)	Coastline (km)	Est. Coral Cover	No. of Species	No. of Genera
Comoros	2 230	469	432 km² (Anjouan)	N.A	N.A
Madagascar	581 540	9 935	~2 000 km²	112	57
Mauritius	2 030	496	~500 km²	133	47
Seychelles	450	747	~577 km²	174	55

Source: McClanahan *et al.*, 2000.

Table 2. Characteristics of marine protected areas in the sub-region

Country	Name	Year Est.	Size (km^2)
Comoros	Moheli Marine Park	2001	404
Madagascar	Nosy Atafana Marine Park	1989	10
	Masoala Marine Park	1997	100
Mauritius	Fishing Reserves (Port Louis, Grand Port, Black River, Poudre d'Or, Poste Lafayette, & Trou d'Eau Douce)	1983	63.2
	Blue Bay Marine Park	1997	3.5
	Balaclava Marine Park	1997	5
Seychelles	St Anne Marine National Park	1973	14
	Aride Island Special Reserve	1979	0.1
	Baie Ternay Marine National Park	1979	1
	Cousin Island Special Reserve	1979	1
	Curieuse Marine National Park	1979	16
	Port Launay Marine National Park	1979	1.5
	Aldabra Special Nature Reserve/World Heritage Site	1981	190
	Silhouette Marine National Park	1987	

Source: Francis *et al.*, 2002.

All of these MPAs include substantial areas of coral reefs, however, recent assessments indicate that there are still a number of important coral reefs areas which should be included in MPAs in all of these countries (Payet, 2004). Within its research programme CORDIO has assisted and supported monitoring within and outside MPA's.

STATUS OF THE REEFS

The status of coral reefs in the Indian Ocean is reported in the 'State of the Coral Reefs 2004' report, through the contribution of CORDIO experts (Ahamada *et al.*, 2004). This summary provides an update to that report.

Comoros

Monitoring of coral reefs in Comoros is undertaken at 20 sites on the three main islands in the group. Monitoring has been ongoing since the 1998 mass coral bleaching event, and in many areas coral recovery has been observed.

However, reported coral recovery has been modest. In some areas (Isandra Island), coral cover has increased from 36% in 2003 to 42% in 2004. Ouani (Ajounan Island) remains one of the most intact and diverse reef within the Comoros which deserves better management, although it was also affected by the 1998 bleaching. In some areas such as Bimbini reef (Anjouan Island), live coral cover has actually decreased from 24% in 2003 to 18% in 2004, primarily due to a proliferation of sea urchins and also pressure from trampling and anchor damage. Conservation efforts at the Moheli Marine Park (Moheli Island) indicate that coral reef recovery is enhanced when areas are protected and human intervention reduced.

Surveys undertaken in 2005 (Ahamada, 2005) indicate a 48.8% increase in coral cover in Isandra Island, a slight increase over 2004. However, in Ouani, the extent of recovery from 2003 to 2005 ranges up to 61%. Extensive stands of branching and tubular *Acropora* species which are currently unprotected at this site continue to be threatened by human intervention.

Madagascar

Due to its large coastline, coral reef monitoring sites around Madagascar are separated by large distances and also exposed to various local conditions which can influence recovery. For example, sites such as Dzamandjar (on the north-west coast) saw a decline in live hard coral cover (LHC) in 2004, whilst in Foulpointe (on the east coast) LHC has increased, despite high levels of sediment input in that region. At the 'Grand Recif' in Toliara (on the south-west coast), no significant change in coral cover has been reported. Overall human impacts on coral reefs in Madagascar include sediment discharge from unsustainable land-use practices and fishing pressure has not diminished and remains largely unmanaged. Natural events such as cyclones also impact on coral reefs, in particular unconsolidated ones.

Mauritius

Coral bleaching was also observed in the lagoons of Mauritius in 1998 during regular coral reef monitoring. However, the percentage of bleached corals was less than 5% at all the sites surveyed (7 sites). Follow up surveys in 1999 showed that the coral reefs exhibited marked recovery. In 2003, further bleaching of corals was observed in late February but by June, 97% of the bleached corals had recovered. Coral cover dropped by 11% to 37% in 2002. Likewise, in 2004 almost 60% of the corals were affected by bleaching during the warmest month (March) but by July most of these affected corals had recovered. Overall, Mauritius reported a higher coral mortality at all of the sites due to the 2004 bleaching episode than previous episodes.

Coral cover at the Blue Bay Marine Park remained stable at 91%. Substantial stand of *Acropora* sp. (59%) remain, primarily as a result of intensive conservation efforts by the Mauritius Government from human intervention, mainly from hotel and tourism development.

Seychelles

Most of the shallow reefs in the Seychelles archipelago were bleached in 1998. Seven years after the bleaching event, recovery of coral communities has been variable, although recovery has been hampered by recurring bleaching events in 2002 and 2003. In 2000, mean LHC was only 3% (surveys done at 22 sites), but in 2004 mean LHC was 10.2% (surveys done at 48 sites) a significant increase despite the recurring bleaching events.

Detailed coral reef surveys of Cosmoledo Atoll in 2002 showed that bleaching-related mortality had been quite severe, despite its remoteness from human population (Souter *et al.*, this volume). Coral mortality in the lagoon was very high, with 95% of the large colonies of *Acropora* completely decimated.

Recovery rates on carbonate reefs were found to be much slower than on granitic reefs. This may be due to the greater stability of granitic reefs compared with carbonate reefs (Payet *et al.*, this volume). The majority of reefs with high rates of recovery are found in MPAs.

ASSESSMENT OF TSUNAMI DAMAGE

Seychelles was the only country within this CORDIO node to have reported damage to its coral reefs as a result of the tsunami of 26 December 2004 that affected many countries in south-east Asia and the Indian Ocean. A rapid assessment of the damage was undertaken by CORDIO and IUCN in February 2005 (Obura & Abdullah, 2005). Coral reefs were found to be particularly vulnerable to physical damage from the tsunami waves due to the weakened reef structure and bio-erosion as a result of the recent bleaching events. The survey revealed little direct damage caused by the tsunami on coral reef habitats, with the majority experiencing 5% reduction in coral cover, especially in unconsolidated reef areas. However, greater than 50% substrate damage and greater than 25% of direct damage to corals was observed in northern and eastern-facing carbonate reef sites.

CLIMATE CHANGE IMPACTS

As a result of the 1998 coral bleaching due to elevated sea surface temperatures (SST), research aimed at predicting the occurrence of such bleaching events is being undertaken. Sheppard (2003) using mean historical SSTs (from 1871 to 1999) in combination with the HadCM3 climate model (IS92a climate scenario) generated forecast SST for the period 2010–2025. The results of this modelling work indicated that reefs found at latitudes between 10–15° south in the western Indian Ocean will be affected by elevated SST every 5 years. Although areas outside of this geographical range will also be affected, the model does not give clear results. Such predicted coral bleaching events will have serious impacts on ongoing conservation efforts and coral recovery.

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Status and Recovery of Carbonate and Granitic Reefs

in the Seychelles Inner Islands and Implications for Management

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key words: Seychelles, reef, carbonate, granitic, recovery, resilience, management

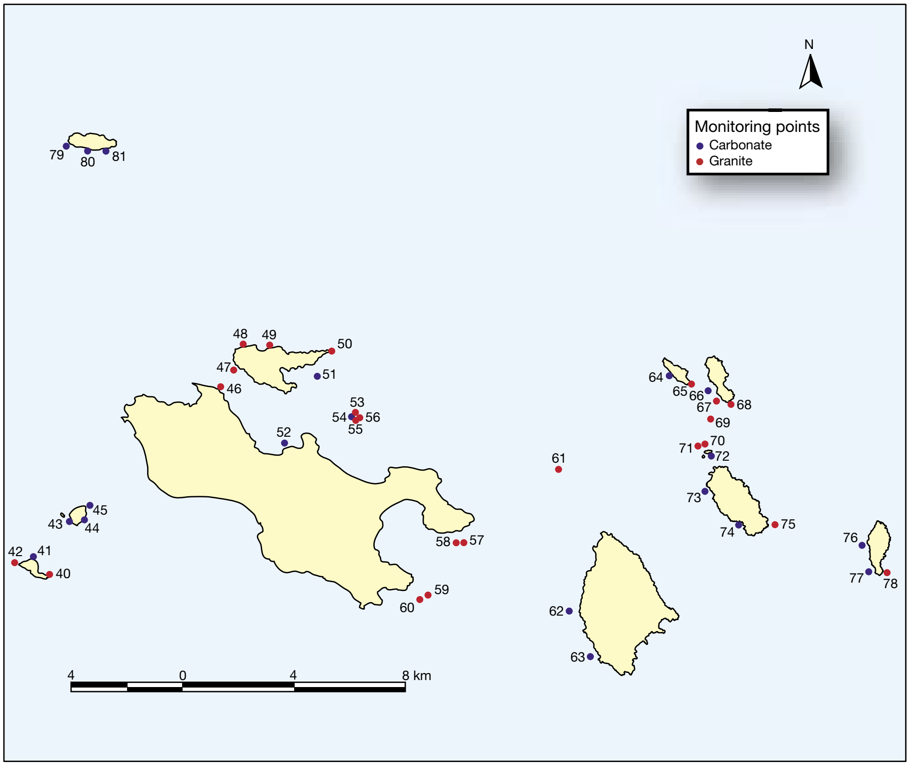
ABSTRACT

The majority of shallow reefs in the Seychelles suffered extensive damage as a result of the 1998 mass coral bleaching event, despite low levels of direct human impacts. This paper explores the historical status of coral reefs of Seychelles and considers the nature and future trends in factors that will affect recovery of coral reefs in the Seychelles inner islands. We have used data obtained from several coral reef monitoring programmes undertaken after the 1998 bleaching event to examine recovery rates on granitic and carbonate reefs and in protected and unprotected areas. Seven years after the bleaching event, recovery of coral communities has been variable. Recovery rates on carbonate reefs were much slower than on granitic reefs, which might be attributable to the greater stability of granitic reefs compared with carbonate reefs. The majority of reefs with high rates of recovery are found in Marine Protected Areas (MPAs). These findings have serious implications for management and also require further studies. Subsequent phases of the CORDIO project will investigate how and why these recovery rates are being observed, and what management options can be considered to maintain or increase those rates of recovery.

INTRODUCTION

The coral reefs of the Seychelles have been described as being one of the most extensive networks in the Western Indian Ocean (Jennings *et al.*, 2000). However, most historical studies and monitoring, as early as 1820, have primarily focused on the granitic islands and Aldabra (Stoddart, 1970). This is probably due to the accessibility of these islands. Recently, studies have also focused on Cosmoledo (Sheppard & Obura, 2005), several atolls in the Amirantes group (Wendling *et al.*, 2003) and Alphonse (Hagan, 2004) in order to better understand the impacts of the 1998 mass coral bleaching and the subsequent recovery of coral reef communities. The most comprehensive assessment of the coral reefs within the inner granitic islands was done through the Global Environment Facility (GEF) funded Seychelles Marine Ecosystem Management Programme (SEYMEMP) from 2000-2004, where 81 coral reef sites (figure 1, table 1 on page 134–135) were monitored using fine-scale monitoring techniques. The Regional Coral Reef Monitoring Programme of the Indian Ocean Commission (COI) also monitored several sites in the Seychelles, especially in the marine parks, and formed the basis for the production of reports to the Global Coral Reef Monitoring Network (GCRMN) (Ahamada *et al.*, 2002, 2004). The CORDIO programme in Seychelles focussed its work on socio-economic aspects and an expedition to Cosmoledo atoll in 2002. The current focus of CORDIO in Seychelles is to study coral reef recovery, management approaches to enhance recovery and the socio-economic

Figure 1. Location of the coral reef monitoring sites between 2000 and 2004.



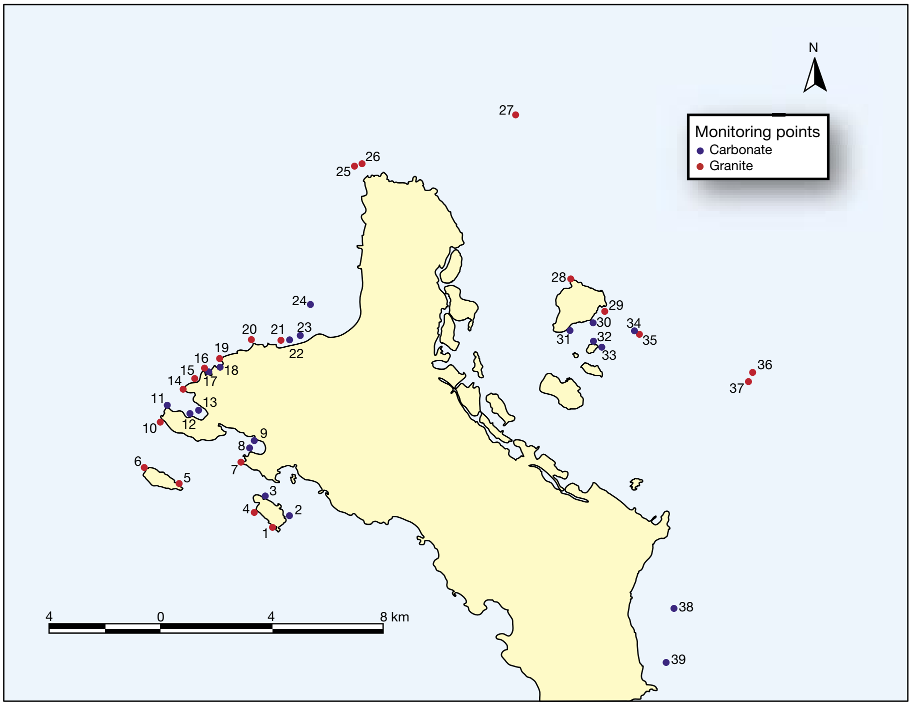


Table 1. Location and geological base of the coral reef sites monitored between 2000 and 2004. Under 'Type' C represents carbonate reefs and G represents Granitic reefs

Number	Name	Type	Latitude (UTM)	Longitude (UTM)
1	Ile Therese South point	G	322781.15	9482596.07
2	Ile Therese South East reef	C	323407.21	9483025.23
3	Ile Therese north East reef	C	322528.98	9483762.70
4	Ile Therese North rocks	G	322125.53	9483134.42
5	Conception North East point	G	319326.27	9484210.40
6	Conception North West rocks	G	318057.61	9484803.72
7	Port Launay South West point	G	321615.17	9485006.30
8	Port Launay South reef	C	321954.00	9485526.00
9	Port Launay South East reef	C	322125.00	9485802.00
10	Cap Matoopa North West point	G	318638.38	9486480.60
11	Baie Ternay North West reef	C	318906.08	9487091.73
12	Baie Ternay South West reef	C	319738.70	9486801.64
13	Baie Ternay Central reef	C	320081.35	9486913.92
14	Baie Ternay North East rock	G	319489.28	9487699.72
15	Rays point	G	319919.73	9488093.54
16	Willie's bay point	G	320272.39	9488465.73
17	Willie's bay reef	C	320429.00	9488352.00
18	Anse Major reef	C	320869.46	9488527.60
19	Anse Major point	G	320832.64	9488836.86
20	Black Rock	G	321999.14	9489510.04
21	White House rock	G	323100.22	9489487.56
22	Auberge reef	C	323414.81	9489510.04
23	Corsaire reef	C	323819.29	9489644.86
24	Aquarium reef	C	324192.87	9490810.55
25	L'ilot West rock	G	325819.07	9495914.53
26	L'Ilot North West rocks	G	326085.07	9496019.53
27	Brissare North rock	G	331776.85	9497832.75
28	Ste Anne North point	G	333793.92	9491777.09
29	Ste Anne South East point	G	335060.71	9490572.41
30	Ste Anne South Central reef	C	334627.86	9490161.70
31	Ste Anne South reef	C	333796.43	9489869.57
32	Moyenne North East reef	C	334650.00	9489468.00
33	Moyenne East reef	C	334950.88	9489237.57
34	Beacon North West reef	C	336147.00	9489862.00
35	Beacon South West rock	G	336335.66	9489734.75
36	Harrisson East rock	G	340521.00	9488311.00
37	Harrisson South rock	G	340345.00	9487985.00
38	Anse Aux Pins Central reef	C	337622.00	9479602.00
39	Anse Aux Pins South reef	C	337326.00	9477607.00
40	Cousine South East point	G	350550.73	9518981.49
41	Cousine North East reef	C	349976.03	9519607.23

Number	Name	Type	Latitude (UTM)	Longitude (UTM)
42	Cousine North West point	G	349311.49	9519416.35
43	Cousin South West reef	C	351271.91	9520863.19
44	Cousin East reef	C	351823.00	9520935.00
45	Cousin North East reef	C	352015.00	9521450.00
46	Praslin North East rock	G	356675.39	9525727.00
47	Curieuse West rock	G	357165.33	9526318.64
48	Curieuse North West rock	G	357495.61	9527247.89
49	Curieuse North Central rock	G	358453.00	9527210.18
50	Curieuse Point Rouge	G	360670.00	9526976.00
51	Coral Garden	C	360150.00	9526094.00
52	Anse Petit Cours	C	358977.00	9523699.00
53	St Pierre North East rock	G	361528.00	9524783.00
54	St Pierre North West reef	C	361386.43	9524636.61
55	St Pierre South East rock	G	361544.04	9524517.36
56	St Pierre East rock	G	361684.00	9524612.00
57	Round island South East rock	G	365404.96	9520123.88
58	Round island South West rock	G	365149.35	9520123.88
59	Ave Maria South rock	G	364114.00	9518234.00
60	Ave Maria South West rock	G	363836.50	9518080.07
61	Roches Boquet	G	368812.00	9522764.00
62	La Digue Central West reef	C	369188.75	9517665.05
63	La Digue South West reef	C	369957.07	9516046.90
64	Petite Soeur Central West reef	C	372791.00	9526100.00
65	Petite Soeur South East rock	G	373569.02	9525806.52
66	Grand Soeur South West reef	C	374176.00	9525558.00
67	Grand Soeur South West rock	G	374467.25	9525181.71
68	Grand Soeur South East rock	G	374985.65	9525086.05
69	Albatross rock	G	374269.91	9524542.55
70	Ile La Fouche North East rock	G	374049.00	9523638.00
71	Ile La Fouche South East rock	G	373827.00	9523566.00
72	Coco Island South East reef	C	374272.00	9523244.00
73	Felicite North West reef	C	374063.00	9521954.00
74	Felicite South West reef	C	375274.00	9520728.00
75	Felicte South East rock	G	376575.59	9520759.34
76	Marianne Central West reef	C	379696.00	9520037.00
77	Marianne South West reef	C	379938.00	9519087.00
78	Marianne South East rock	G	380571.00	9519026.00
79	Aride South West reef	C	351175.67	9534363.61
80	Aride South Central reef	C	351915.73	9534170.47
81	Aride South East reef	C	352596.00	9534149.48

impact of coral bleaching on local communities, especially fishermen, as well as supporting basic monitoring and capacity building.

This paper presents an overview of the historical status of coral reefs in Seychelles, and how this has changed as a result of local impacts and also the effects of mass bleaching. Using data collected since the bleaching event, recovery rates of coral reefs around the granitic Seychelles was computed and analysed against future trends and likely scenarios of elevated sea-surface temperature extremes.

THE STATUS OF CORAL REEFS IN THE SEYCHELLES

Coral reefs in Seychelles can be classified into three main groups: fringing reefs – characteristic of the granitic islands, atolls (e.g. Aldabra and Cosmoledo) and platform reefs (e.g. around islands of the Amirantes). The coral reef areas of the Seychelles have an estimated cover of about 1 690km² (Spalding *et al.*, 2001) of which 40 km² are in the granitics (Jennings *et al.*, 2000). There are more than 300 known species of corals from the Seychelles (Veron & Stafford-Smith, 2000). A recent expedition to the southern Seychelles atoll of Aldabra and Cosmoledo recorded 201 of these species (Sheppard & Obura, 2005). Table 2 summarises the characteristics of those reefs.

Prior to the 1998 coral bleaching event the coral reefs of the Seychelles was described as supporting a growing artisanal fisheries sector and to have remained relatively undamaged except for a certain amount of anchor damage, land reclamation on the east coast of Mahe and land based sources of pollution (Salm *et al.*, 1998). The most

Table 2. Reef characteristics around some of the main islands of the Seychelles

Island	Reef Type	Total area (km²)	Total area of island (km²)	Area of reef (km²)	Reef width (m)	Lagoon area (km²)
Mahe	Fringing	173.91	156.88	17.03	75– 1 400	–
Praslin	Fringing	66.02	37.85	28.17	135– 3 100	–
La Digue	Fringing	12.75	9.82	2.93	50– 630	–
Curieuse	Fringing	3.47	2.74	0.73	40– 275	–
St.Anne	Fringing	2.85	2.19	0.66	45– 440	–
Cousine	Fringing	0.74	0.24	0.50	70– 440	–
Cousin	Fringing	0.78	0.29	0.49	110– 305	–
African Banks	Platform	4.14	4.12	0.02	300– 1 050	–
Coetivy	Platform	15.44	6.50	8.94	230– 600	–
D'Arros	Platform	3.81	2.21	1.60	100– 750	–
Platte	Platform	10.36	9.91	0.45	230– 1 630	–
Providence-Cerf	Platform	138.07	135.75	2.32	Not Available	–
Aldabra	Raised platform	371.89	152.55	28.94	60– 660	190.40
Assumption	Raised platform	14.43	11.01	3.42	100– 390	–
Astove	Raised platform	16.16	5.36	4.20	135– 440	6.60
Cosmoledo	Raised platform	139.17	4.44	70.36	600– 5 000	64.37
St Pierre	Raised platform	1.71	1.71	–	– –	
Alphonse	Raised platform	19.26	1.60	12.79	100– 2 100	4.87
Farquhar	Raised platform	268.48	7.37	160.55	4 400–11 000	100.56
St Francois	Raised platform	48.73	0.32	34.45	200– 3 500	13.96
St Joseph	Raised platform	23.09	1.35	17.34	220– 2 700	4.40

significant impact on coral reefs occurred in 1998 when a widespread bleaching event resulting from globally elevated sea surface temperatures caused extensive mortality among corals affecting as much as 90% of the coral cover down to depths exceeding 15m in some areas of the Seychelles (Lindén & Sporrong, 1999).

Due to the recent bleaching event in 1998, the status of Seychelles reefs has significantly changed. Long-term analysis of sea surface temperature data extracted from the Hadley Centre in UK suggests that the 1997–1998 warming was the highest during the last 37 years (Spencer *et al.*, 2000), and seems to be closely associated with El Niño years, although Webster *et al.* (1999) has argued that ocean warming in the Indian Ocean may occur even in the absence of El Niño.

The status of coral reefs around Mahe was assessed in late 1999 by Turner *et al.* (2000) and Engelhardt (2000), which was focussed on the east coast and Ste Anne Marine Park and the north-west coast respectively. They concluded that the living hard coral cover (LHCC) on most of the shallow coral reefs of the granitic islands had declined to less than 10%, which in effect constituted a loss of between 15% and 70% of the coral cover in many areas. It was found that the branching and tabular *Acropora*, and branching *Pocillopora* species suffered the most while *Porites* was more resilient. Similar observations were reported by the Regional Coral Reef Monitoring Programme of the Indian Ocean Commission (COI) which recorded 95% bleaching-related mortality of *Acropora* (Bigot *et al.*, 2000). However, massive corals from the genus *Porites* and *Goniopora* were observed to have survived throughout the inner islands.

Although mortality among corals was extensive and the diversity at most sites surveyed was low following the 1998 mass bleaching event, no extinctions have been reported, rather the abundance and distribution of species have reduced. This finding has important implications for recovery and the probability of future recruitment from within the region (Engelhardt, 2000; Lindén & Sporrong, 1999).

CURRENT AND FUTURE THREATS TO CORAL REEFS

The quality of coral reefs in the Seychelles, particularly in the granitic islands, has declined significantly as a result of the 1998 coral bleaching event and growing human impacts (Engelhardt, 2004). Whilst the majority of threats act directly on coral reefs, the threat of global warming and further elevation of the sea surface temperature for sustained periods is the sole threat that could virtually eliminate viable coral populations in the Western Indian Ocean region (Sheppard, 2003). It is also feared that further development of coastal fisheries and tourism could result in considerable degradation of the remaining and more resilient reefs. At least five important threats to the coral reefs of the Seychelles are emphasized here:

1. Reclamation, Mining and Sedimentation

Extensive reclamation has occurred on the east coast of Mahe, and small areas have also been reclaimed on the other granitic islands to meet demand for land, since land is extremely scarce in the Seychelles (Payet, 2003). Reclamation on shallow reef flats completely eliminates coral reef ecosystems, and associated impacts such as silt from dredging can affect adjoining reef areas for many years if proper mitigation measures are not implemented. The Ste Anne Marine Park, for example, shows evidence of such sediment-related stress following chronic sedimentation from dredging activities on the east coast of Mahe since the early 1980s (Robinson, 1999).

Sedimentation from land-based activities further inland is also of concern as this fine silt is carried by rivers during periods of heavy rainfall. A red colouration due to erosion of the lateritic soils is also observed within several kilometres of the coast following intense rainfall on the granitic islands. Such sedimentation is mainly a result of heightened development and construction on the steep slopes, as opposed to deforestation which leads to similar effects in other countries. Strict guidelines for minor works and rainy season restrictions have been implemented to reduce the impacts of these types of developments.

2. Impacts from Tourism Activities

Tourism activities have several impacts on coral reefs in Seychelles. These include direct damage from anchors and trampling during snorkelling and diving, and indirectly during hotel construction and operation. Whilst a series of mooring buoys are currently being installed in the critical coral reef areas, there are still reports of anchor damage (Bijoux, J., *pers. obs.*). Trampling through poor visitor management at coral sites is a common problem caused by tour operators who wish to bring the maximum number of passengers to a particular site. Tourism can also affect coral reefs through the discharge of untreated sewage and sediments. This was the case with hotels that were built before the 1990s in the Seychelles. Today, it is a legal requirement that all hotels have to meet stringent effluent water quality standards and offshore outfalls are not encouraged. The environment impact assessment (EIA) process is mandatory for all such large developments and many of these threats are addressed in such a process (Payet, 2003).

3. Fishing Pressure

Reef fisheries have shown a considerable decline in catch per unit effort (CPUE) over the last 10 years (Grandcourt & Cesar, 2003). This is related to increased fishing pressure but also to the overall degradation of coral reef health. There is also evidence of trampling by fishermen when laying and removing their traps. All forms of destructive fishing, including spear guns, are banned in the Seychelles, and there are few records of historical use of such destructive practices. Recently, inhabitants of Praslin have used the argument of declining fish catches to lobby the Government of Seychelles for fishing rights to be granted within marine parks. The Government refused on the basis that the particular marine park in question protected highly resilient reefs that supported high coral cover and species diversity and were recovering rapidly from the 1998 mass bleaching event and that marine parks, in general, play an important role in fisheries.

4. Disease and Invasive Species

Disease and other invasive species have not been observed in any significant abundance on the reefs of the Seychelles. Localised outbreaks of the crown-of-thorns starfish (COTS, *Acanthaster planci*) have been reported since 1996 (Engelhardt, 2000). Populations of COTS are controlled through physical removal. Coral diseases are not widespread, but black-band and white-band diseases have been observed in several areas around Mahe (Engelhardt, 2004). The threat from the release of invasive species from ballast water is real, and a research project with CORDIO-IUCN support is currently being undertaken to determine the extent of this threat.

5. Climate Change and Global Warming

Coral reefs have narrow temperature tolerances, and many SST projections (Nurse *et al.*, 2001) suggest that the thermal tolerance of reef-building corals will be exceeded within the next few decades. Hoegh-Guldberg (1999) predicts that the incidence of bleaching will increase, and recent evidence indicates that the 'episodic' warming of the sea-surface (e.g. during El Niño years) can lead to significant coral bleaching.

Increase of CO_2 concentrations in the oceans is thought to have a direct impact on calcification processes in coral reefs. No such work has yet been done in the Seychelles but Kleypas *et al.* (1999) estimate that the calcification rate of corals would decline by approximately 14–30% by 2050.

METHODS

Coral Reef Monitoring

Coral reef monitoring was conducted at 78 sites in the Seychelles granitic islands from November 2000 to February 2004 as part of the coral reef component of the Global Environmental Facility (GEF) sponsored Seychelles Marine Ecosystem Management Programme (SEYMEMP). The number of sites monitored for each

Table 3. Number of coral reef sites surveyed in each of the monitoring sessions

	Nov –00	Jul –01	Feb –02	Jul –02	Feb –03	Aug –03	Feb –04
Technique used	VET	VET	LIT	LIT	LIT	LIT	LIT
Total No. Reefs surveyed	20	40	43	44	42	36	50
No. carbonate reefs surveyed	15	22	20	20	20	16	26
No. granitic reefs surveyed	5	18	23	24	22	20	24

period is given in table 3. Sites were monitored biannually, annually or as one-offs for greater geographical coverage. The design and subsequent implementation of this component was contracted to Reefcare International based in Townsville, Australia. Field data was collected with the support of the Marine Unit in the Ministry of Environment and various other organisations on a less formal basis. The visual estimation technique (VET) was used for data collection in 2000 and 2001 while line intercept transects (LIT) modified according to Engelhardt (2004) were used from 2002 to 2004.

Visual Estimation Technique (VET)

The VET estimated live hard coral cover (LHCC) in incremental classes of 10%. A small increment of 5% was used where LHCC was found to be extremely low (Engelhardt, 2004). For each site, the mean LHCC was calculated from the estimated LHCC from ten 10 m segments using the mid-point of the appropriate class as the estimate of LHCC for each segment.

Line Intercept Transect

For each 50 m LIT, each substrate type bisected by the transect was recorded in three 10 m segments that were separated by 10 m. All transects were laid obliquely so that the entire depth profile of the reef could be sampled and as such the 0–10m transect always corresponded to the shallowest depth and the 40–50 m transect always corresponded to the deepest depth. At each site, data were collected from a minimum of six 10 m segments. Data describing the live benthic cover were recorded according to 9 categories: coral of the genus *Acropora*, coral of the genus *Pocillopora*, branching non-*Acropora*, encrusting coral, massive coral, fungiid corals, soft coral, macro algae and zoanthids, and corallimorphs.

Calculating Pooled Yearly Recovery Rate

Two different rates of recovery were calculated. The first approach calculated the pooled yearly rate of recovery (PYRR) for all carbonate reefs and all granitic reefs between each survey using the equation:

$$\text{PYRR} = 2\,(\textit{Mean LHCC for period X} - \textit{Mean LHCC for period } (X{-}1))$$

where X is a monitoring session and X – 1 is the monitoring session 6 months earlier.

Since monitoring was done every 6 months, the calculated rate was multiplied by 2 to get a yearly rate.

Calculating Site Specific Recovery Rate

A sub-sample of the data was used for calculating site specific recovery rate. Only sites at which monitoring had been carried out in 2004 and where monitoring has been conducted at least three times since November 2000 were selected. This was to ensure that the status of the reefs in 2004 was taken into account and that there were sufficient yearly recovery rates from which longer-term (4 years) mean recovery rate could be calculated.

Site specific recovery rate (SSRR) was calculated using the equation:

$$\text{SSR} = 12\left(\frac{\textit{Mean LHCC for period X} - \textit{Mean LHCC for period } (X{-}1)}{\textit{Number of months between monitoring sessions}}\right)$$

where X is a monitoring session and X – 1 is one monitoring session before and 12 is the number of months per year.

For both PYRR and SSRR, value of less than 0% was taken to indicated a negative rate of recovery, whereas value between 0–2% , 2.1–5% and >5% were taken to represent low, medium and high rate of recovery respectively.

RESULTS

Monitoring the recovery of coral reefs from the devastating effect of the 1998 mass coral bleaching event in the Seychelles inner islands began in 2000 (Engelhardt, 2000). Overall, a positive trend in recovery is being observed despite the fact that the reefs were again affected by coral bleaching events of 2002 and 2003 brought about by sustained and elevated sea-surface temperatures (figure 2). Combined data for carbonate and granitic reef is showing an exponential increase in mean live hard coral cover. However, there are considerable differences in the rate of recovery between carbonate and granitic reefs. The granitic reefs are experiencing a strong exponential increase (R^2 = 0.917) in live hard coral cover whereas carbonate reefs are experiencing a weak linear increase (R^2 = 0.5454) (figure 3).

Calculated yearly rates of increase in LHCC was affected by the previously mentioned coral bleaching events in 2002 and 2003 on both carbonate and granitic reefs. The 2002 bleaching event, in particular, caused a negative rate of recovery in carbonate reefs, where it dropped from 1.47% yr^{-1} for the Jul 01 to Feb 02 period to –3.84% yr^{-1} for the Feb 02 to Jul 02 (table 4), the period in which the bleaching event occurred. On the other hand, results indicate that on the granitic reefs, the 2002 coral bleaching event did cause a reduction in the recovery rate but did not result in an overall negative rate of recovery for the period from 2000 to 2002 in which recovery was being measured. The 2003 coral bleaching event had a greater negative impact on granitic reefs as it reduced recovery rate from 6.02% yr^{-1} for the Jul 02–Feb 03 period to a negative recovery of –2.06% yr^{-1}, implying that the effect of this post-1988 bleaching event actually resulted in an effective decrease of 8.08% yr^{-1} in coral reef recovery. It is interesting to note that after the 2002 coral bleaching events both carbonate and granitic reefs entered into a phase of high recovery with 7.72% and 6.02% yr^{-1} LHCC recovery respectively. High rate of LHCC recovery was again observed for granitic reefs after the 2003 coral bleaching event with recovery reaching 11.90% yr^{-1}, the highest level recorded since the 1998 coral bleaching

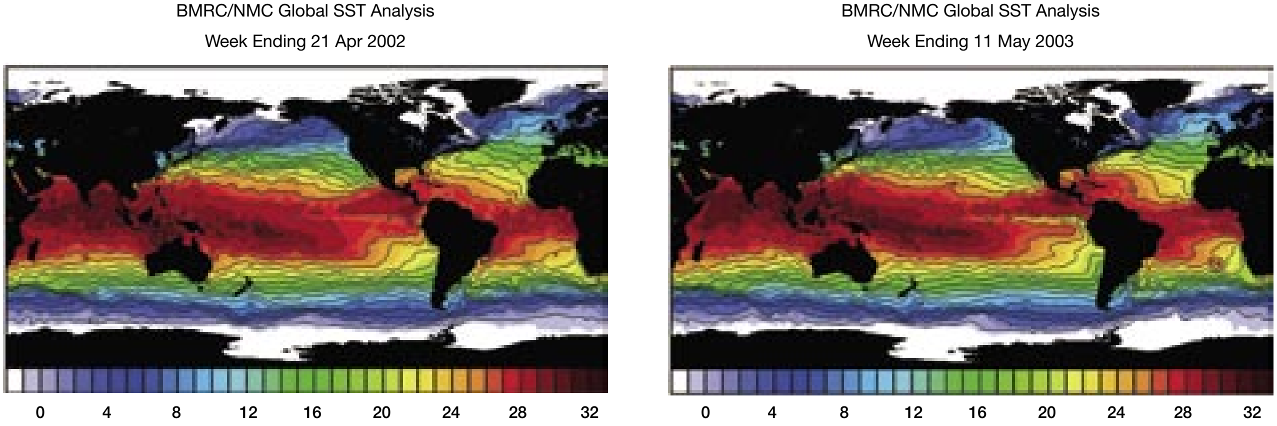


Figure 2. Remotely sensed sea surface temperature (SST) data identifying areas of warm sea water above 30°C during the coral bleaching event in a) 2002 and b) 2003. Source: Australian Bureau of Meteorology.

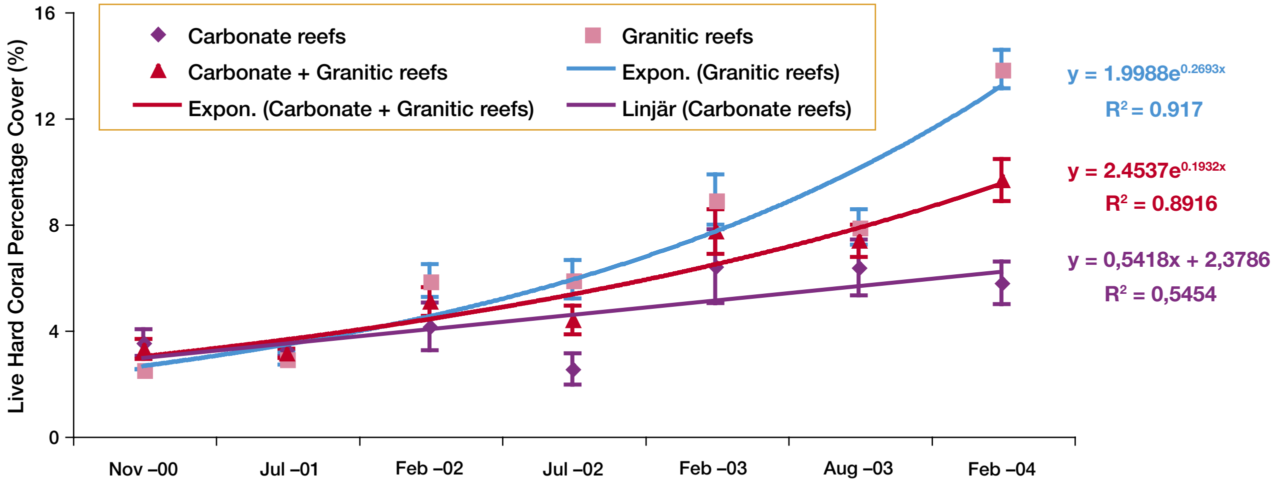


Figure 3. Trends in the recovery of carbonate and granitic reefs in the Seychelles inner islands between November 2000 and February 2004.

Table 4. Percentage yearly rate of increase in LHCC (% yr^{-1})

	Carbonate Reefs	Granitic Reefs
Nov 00–Jul 01	–0.38	0.60
Jul 01–Feb 02	1.47	5.04
Feb 02–Jul 02	–3.84	0.10
Jul 02–Feb 03	7.72	6.02
Feb 03–Aug 03	–0.08	–2.06
Aug 03–Feb 04	–1.16	11.90

Table 5. Percentage of carbonate and granitic reefs within the different calculated recovery potential categories

Mean rate of recovery	Carbonate Reefs (n = 16)	Granitic Reefs (n = 13)
Negative (<0% yr^{-1})	25	7.69
Low (0–2% yr^{-1})	37.5	7.69
Medium(2–5% yr^{-1})	31.25	46.15
High (<5% yr^{-1})	6.25	38.46

event. This was not the case for carbonate reefs on which a general trend of degradation was observed on subsequent surveys after the event.

Site specific mean rate of recovery calculated from a sub-sample of 29 reef sites (13 granitic, 16 carbonate) showed that 17% of all sites were undergoing further degradation (negative recovery) while 24% of the sites had low rates of recovery of between 0–2 % yr^{-1} and 38 and 21% of the sites had medium (2–5% yr^{-1}) and high rates (>5% yr^{-1}) of recovery respectively (table 5). Comparison between carbonate and granitic reefs showed that 84.5% of the granitic reefs had recovery rates which were higher than 2% yr^{-1} compared to 37.5% of carbonate reefs. Importantly 25% of carbonate reefs was degrading (negative recovery) as opposed to 7.69% of granitic reefs.

From our sub-sample, 8 sites stood out from the rest with high rates of recovery gaining close to or above 5% LHCC per year (figure 4 on next page). Out of these 8 reefs, 6 were of granite base and 2 of carbonate base, displaying once again the overall greater ability of granite based reefs to recover from coral bleaching events. Further more 5 of the sites are found in formally protected

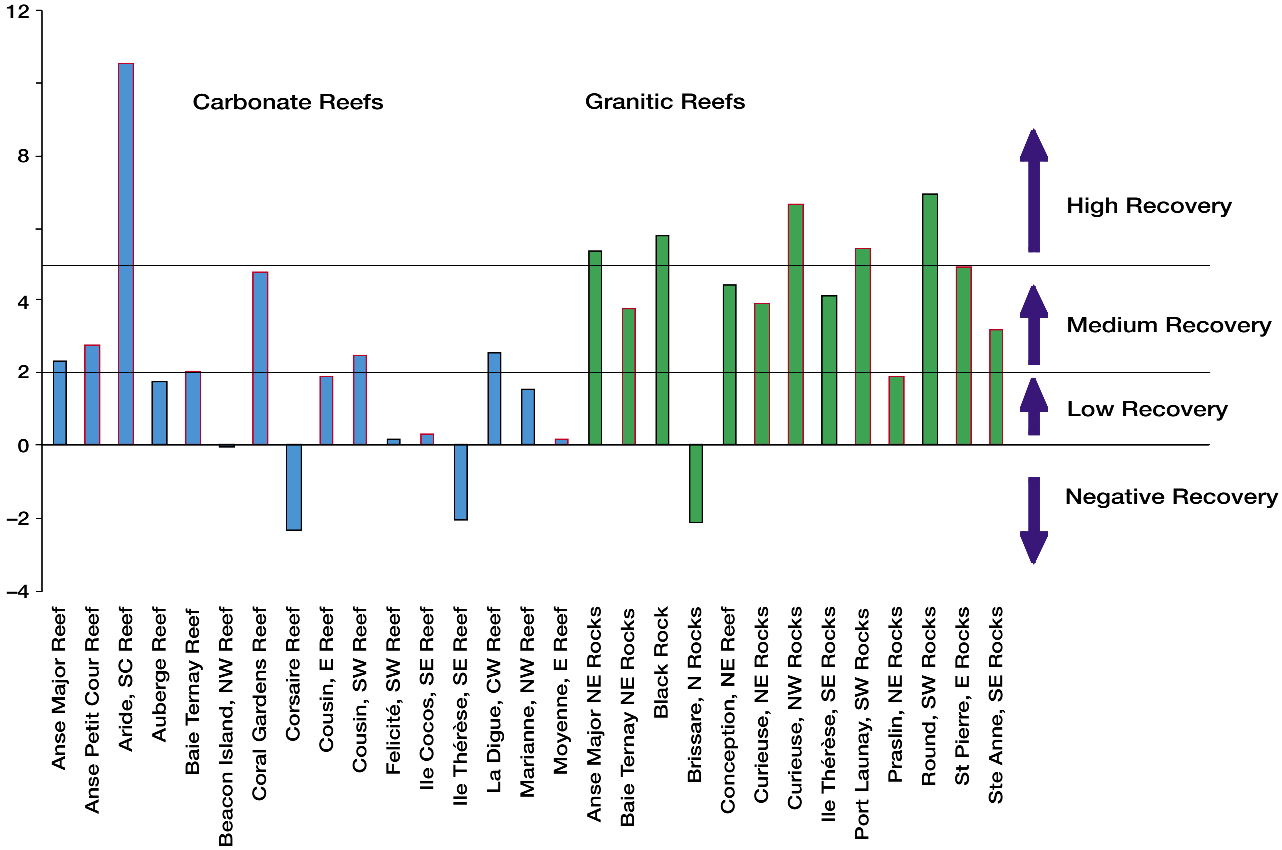


Figure 4. Site specific medium term (4 years) yearly recovery potential of sub-sample of monitored reef sites. Sites from Marine Protected Areas have red borders on the figure.

areas. It should be noted that high rates of recovery however does not always relate to high percentage LHCC.

DISCUSSION

Coral reef recovery rates have been determined using short-term monitoring data, without recourse to changes in other factors such as environmental and natural growth conditions of particular reef sites, as well as species diversity (e.g. Jennings *et al.*, 2000). However, the objective was to be able to determine whether such an approach could show the nature of this variability and how it may relate to management of bleached areas and also long-term prospects.

Analysis of monitoring data since 2000 (2 years post-bleaching) indicates very clearly that coral reefs in the Seychelles inner islands are recovering. The overall exponential increase in % LHCC from 2000 to 2004 is strongly influenced by granite base reef sites for which pooled data indicate an even higher rate of exponential increase. The strong exponential rate of increase in % LHCC for granite based reefs as opposed to the weak linear increase for carbonate reefs clearly outline the greater resilience of granite reef to carbonate reefs. There are several possible explanations for the observed difference in mean % LHCC between the granite based and carbonate based reefs.

The greater stability of granitic reefs over carbonate reefs is thought to be an important factor for the observed differences. More stable granitic reefs means that there are more suitable areas for the recruitment of coral spat and lower post recruitment mortality resulting from movement of rubble resulting from wave action. It is expected that granitic reefs are also suffering from lower level of indiscriminate grazing by invertebrates such as the black spined sea urchins (*Diadema* sp. and *Echinotrix* sp.) as compared to carbonate reefs. This is mainly due to the higher 3-dimensional nature of granite based reefs which limits access to grazing invertebrates. Better shedding of sediments, as a result of their high 3-dimensional morphology, by granitic reefs may also be significant in promoting reef recovery as it provide bare sediment free substrate for improved coral spat recruitment as few coral larvae settle on sediment-covered surfaces (Fabricius, 2005.).

However, all of these reefs will continue to be exposed to future coral bleaching events, such as those that were recorded in April 2002 and 2003. Whilst the 2002 bleaching reduced the rate of coral reef recovery on both granitic and carbonate reefs, recovery was actually negative on carbonate reefs and on granitic reefs in 2003. Overall, both granitic and carbonate reefs bounced back after the 2002 event with high rates of LHCC recovery. The period after the 2002 bleaching event exhibited the highest ever rate of recovery for carbonate reefs and it was the only period in which overall recovery on carbonate reefs (7.72 % yr^{-1}) exceeded that on granitic reefs (6.02% yr^{-1}). A somewhat different scenario was seen in the period after the 2003 event when recovery on granitic reefs reached its highest level so far recorded at 11.90% yr^{-1}. The carbonate reefs however experienced continued overall degradation at a rate of –1.16% yr^{-1}.

At present, there is no scientific explanation with regards to the increase rate of recovery observed on granitic and carbonate reefs after the 2002 bleaching event and on granitic reefs after the 2003 event, although it may indicate the acquisition of resilience by the reef system. Continuous decreases in mean LHCC on carbonate reefs could be the result of degradation of the carbonate reef structure which is being aggravated through time especially through bio-erosion which is not as important on granite reefs.

Site specific rates of recovery for the chosen subsample of carbonate and granitic reefs showed that 17% (25% carbonate and 6.69% granitic reef sites) of the reef sites were not showing any real signs of recovering. Reefs showing relatively high rate of negative recovery are clearly noticeable on figure 4 and includes the carbonate reefs of Corsaire and Ile Thérèse and the granite reef of Brissare.

Degradation of the carbonate Corsaire reef can be attributed to high cover of soft coral which is slowly taking over the reef, as a result of high levels of nutrients found in the bay of Beau Vallon (Jennings et al., 2000). In 2004, the Beau Vallon area was integrated in the wider Greater Victoria centralised sewage treatment system thus limiting the amount of sewage derived nutrients into the bay. It is expected that within a short time period the water quality of the area will improve and will create better conditions for reef building organisms to resettle.

The carbonate reef at Ile Thérèse is more or less 2-dimensional in nature with a gentle slope and lots of coral rubble. The coral rubble may be breaking adult corals and killing recruits when it is moved around by wave action. For these types of reefs, with low rugosity and lots of rubble, recovery is seriously hampered, and in many cases only artificial substrate stabilisation may be used to promote substrate recovery. Pilot studies by the Seychelles Centre for Marine Research and Technology are currently looking at a number of rubble slope stabilisation methods as ways of promoting reef recovery. The most successful method will be used on reefs encountering the same problem as Ile Thérèse.

Degradation of the granitic Brissare reef is probably due to its high composition of *Pocillopora* corals. *Pocillopora* corals are brooders and fast growing. As such, they are able to cause a rapid increase in LHCC on a reef. However, Pocilloporids are highly susceptible to elevated SSTs and bleach easily (McClanahan *et al.*, 2004). The

2002 and 2003 coral bleaching events almost wiped out the whole *Pocillopora* community at this site resulting in the observed degradation.

Five of the eight sites with rates of recovery close to or higher than 5 % yr^{-1} are found in Marine Protected Areas (MPAs), indicating that protection can have a positive effect on recovery potential of bleached coral reefs. Whilst, recovery rates are not the only important indicator of coral reef health, such a result provides an important rationale for continued protection of those reefs. The annual rate of recovery as used here can be an important tool for detecting reefs undergoing degradation or rapid recovery. However, prioritising sites for management on this basis only is not recommended as other factors such as species diversity (corals as well as other marine fauna), representativeness, ecological functioning and socio-economic value should also be considered.

Previous CORDIO status reports and a number of other scientific publications have been largely silent on quantifying the nature of the recovery process, primarily as a result of lack of long-term data. Reef recovery rate information is important for both coral reef managers and policy makers in providing a more complete health assessment of impacted coral reefs (Payet, 2004). Importantly now, is the need to identify other factors, apart from geological base and protection, that are promoting or limiting recovery of coral reefs. These will provide valuable insights into management actions needed to address any major problem and to encourage optimum reef recovery. This paper has stressed the importance of having a long-term and timely implemented coral reef monitoring programme. Future data collection, especially in the face of the predicted increase in the frequency of coral bleaching events, will shed more light on whether there is increased resilience to regain LHCC following successive but low-levels of bleaching episodes. One of the important arguments for conservation of resilient or unbleached corals is their potential roles in facilitating coral reef recovery. Hence, there is a need to identify the more resilient reefs and provide them with higher levels of protection so that they can effectively act as a source of coral larvae to bleached and degraded coral reefs. There is a need to take an integrated approach to coral reef management in the Seychelles as it is well known that there are various other factors that influence the health of coral reefs (see Souter & Lindén, 2000).

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Status of Cosmoledo Atoll,
Southern Seychelles,
Four Years after Bleaching-Related Mass Coral Mortality

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keywords: Cosmoledo Atoll, Aldabra, coral community structure, coral recruitment, coral bleaching, fish assemblages, Seychelles

ABSTRACT

Coral and fish community structure was examined at five permanent monitoring sites and several rapid assessment sites on the leeward reef slope and in the lagoon of Cosmoledo Atoll in the southern Seychelles in order to assess its current status and establish a long term monitoring and conservation programme. The atoll and reef structure of Cosmoledo are characteristic of atoll reefs, with a steep reef slope below 25–30 m, a central lagoon that is connected to the surrounding ocean by two major passes through the reef rim, and several islands distributed around the atoll rim. Almost 200 species of coral were recorded during the expedition. Coral communities were severely affected by the El Niño of 1998 showing near 100% mortality in the lagoon and on reef slopes to a depth of 8 m. Deeper reef slopes (>15 m) supported 20–25% live coral cover, which decreased to 1–5% at depths <10 m. The average number of coral recruits ranged between 5 and 6.7 m^{-2} on the slope and 6.8 m^{-2} in the lagoon. On reef slopes, recruitment was greater at 20 m than 10 m. The species composition of recruits differed from the pre-bleaching adult community indicating that a shift in the species composition of the coral community is underway. In shallow waters on the reef slope, pocilloporids dominated recruit assemblages while faviids were most abundant at depth. In the lagoon, *Porites* and *Fungia* recruits were most abundant while the previously dominant acroporids were rare. More than 200 species of fish were recorded. Acanthurids were common, large and medium sized serranids were recorded at all sites and lutjanids and lethrinids were frequently sighted. The obligate corallivores *Chaetodon trifascialis* and *C. trifasciatus* were rare even where coral cover was greater. Not a single shark was sighted. While recovery from coral bleaching impacts is evident, especially below 10 m, recovery has been slow, particularly in shallow water (<5 m).

INTRODUCTION

During 1998, coral reefs throughout the central and western Indian Ocean suffered severe coral mortality as a result of bleaching caused by anomalously high sea temperatures (see Lindén & Sporrong, 1999, and papers therein). Some of the reefs of the Seychelles were among the worst affected with coral mortality exceeding 90% in some places (Quod, 1999; Turner *et al.*, 2000). Recovery from this bleaching event on other reefs within the inner granitic islands of Seychelles (Payet *et al.*, this volume) and on other reefs in the Indian Ocean (Obura; Rajasuriya, Suleiman *et al.*, Zahir *et al.*, all this volume) has been slow and highly variable between sites. Often, this variation is attributable to differences in the magnitude of the impacts of human activities. Recovery on reefs that are easily accessible and heavily overexploited or subjected to destructive fishing, coral mining, pollution or sedimentation is often negligible, while on those reefs that are managed properly or have escaped serious degradation, recovery is progressing well. Investigation of the status of coral and associated fish communities on inaccessible reefs that are isolated from land-based disturbances provides an opportunity to examine the patterns and rate of recovery in an unperturbed environment enabling an assessment of the influence of human activities on the recovery of reefs elsewhere. In addition, the acquisition of baseline data describing the current status of coral and fish populations on remote reefs enables the assessment of the impacts of climate change and the subsequent changes in community structure and composition in an environment that is not confounded by the influence of human activities.

Cosmoledo Atoll, along with Astove, Assumption and Aldabra itself belongs to the Aldabra Group of islands located in the southern Seychelles. The Aldabra Group lies east of the Mascarene plateau at approximately S12° E45°, about midway between the northern tip of Madagascar and northern Mozambique in the western Indian Ocean. Their isolation from significant land masses and dependent human populations makes them ideal locations to study the impacts of climate change on coral reef ecosystems and the mechanisms by which they are recovering and adapting. Although a number of surveys of the impacts of bleaching-related mortality of coral and fish communities have been conducted on Aldabra Atoll (Spencer *et al.*, 2000; Teleki *et al.*, 2000; Stobart *et al.*, 2001, 2002), the condition of coral and fish communities on Cosmoledo Atoll following the 1998 El Niño is unknown.

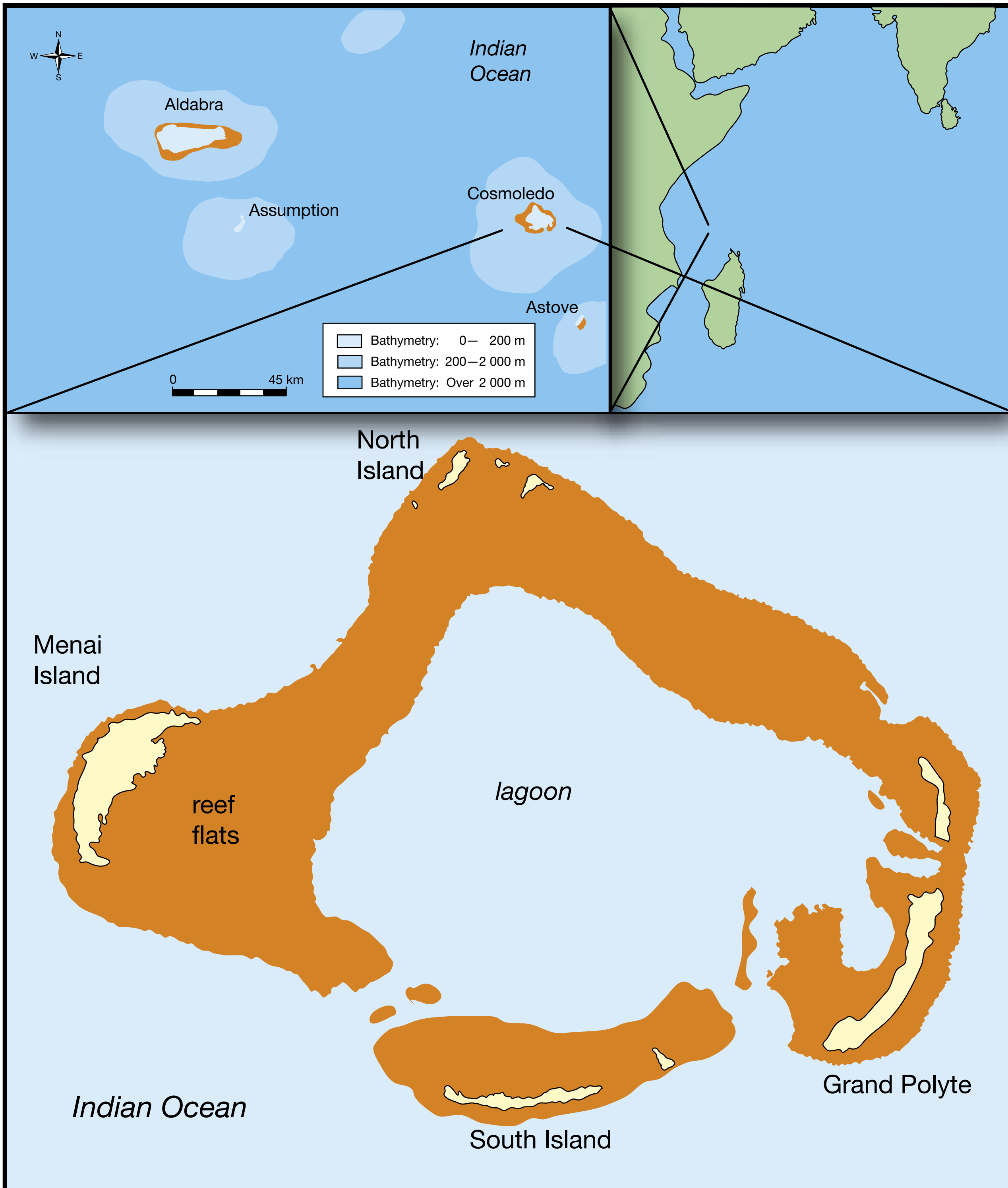
The data presented in this report were obtained during the first scientific expedition to Cosmoledo Atoll. The purpose of this expedition was to determine the status of both the marine and terrestrial environment of Cosmoledo Atoll, including the coral reefs and their associated fish and invertebrate fauna, turtle and bird populations, plants and terrestrial invertebrates. The data gathered during the expedition established a baseline against which changes in community structure can be compared during future monitoring. Another purpose of the expedition was to assess the species diversity of the atoll (see Obura & Sheppard, 2005) and make conservation management recommendations (see Obura *et al.*, in press). This paper describes patterns of reef zonation and reports the current condition of coral and fish communities and patterns of coral recruitment and reef recovery.

ATOLL STRUCTURE AND DESCRIPTION

Comprehensive descriptions of the granitic and outer islands of the Seychelles were obtained during the Indian Ocean expedition (Barnes *et al.*, 1971; Braithwaite, 1971; Rosen, 1971a, b) and a variety of other investigations conducted over the last 100 years (Jennings *et al.*, 2000). A large literature of marine studies conducted on Aldabra Atoll has been summarized by Teleki *et al.* (1999), however, Cosmoledo, Assumption and Astove, the other three islands comprising the Aldabra Group, have received little attention.

The four islands of the Aldabra Group are located on a submarine mountain that rises from the ocean floor 4 000 m below and are separated by distances ranging from 20 to 60 nautical miles. The islands differ considerably from each other (Stoddard, 1984), with Aldabra being the largest (365 km^2) and having an almost totally enclosed lagoon, a deep channel and significant land area (155 km^2), and Cosmoledo (152 km^2) having a submerged atoll rim with scattered small islands (land area 5 km^2). Assumption and Astove are single islands with similar total size (14.2 and 16 km^2, respectively), but with a smaller total land area on Astove (11.5 and 5.6 km^2, respectively). Astove has a central small lagoon with a single outlet to the sea.

Geologically, Cosmoledo is an uplifted coral reef similar to Aldabra, and displays many similarities in terms of marine and terrestrial habitats (Bayne *et al.*, 1970). It consists of thirteen small scattered islands and islets on the atoll rim. The largest island is Menai (figure 1 on next page), which has an area of about 240 ha and an elevation of 3.6 to 4.6 m above mean sea level. The second largest island is Grand Ile (192 ha). The total land area of the atoll is 5 km^2. The lagoon covers an area of about 80 km^2. Inside the atoll rim and the inner shores of the islands extensive shallows several kilometres wide are exposed at low tide. The central part of the lagoon supports


Indian Ocean
Aldabra
Assumption
Cosmoledo
Astove
N
S
E
W
0
45 km
Bathymetry: 0— 200 m
Bathymetry: 200—2 000 m
Bathymetry: Over 2 000 m
North Island
Menai Island
reef flats
lagoon
Grand Polyte
South Island
Indian Ocean

extensive seagrass beds, sand flats and patch reefs that reach the surface. Two primary passes between 10–15 m deep are situated in the southeast and southwest and permit water exchange between the lagoon and surrounding ocean. Complex channels bisect the central lagoon connecting the two passes, with minimum depths of 2–3 m in the centre of the lagoon.

All islands on Cosmoledo Atoll are vegetated and have nesting sea birds and migratory birds in significant numbers (Rocamora, 2002). Although the atoll has been uninhabited since 1990, periodically there have been human settlements on some of the islands during the last 100 years. The dilapidated remains of buildings on Menai and Grande Ile, and the presence of introduced plant species on the other islands are evidence of the last period of human occupation during the 1950s. The inhabitants made a living from guano, sisal and copra production and, of course, fishing. Today, the absence of a permanent settlement or regular policing of activities conducted on the atoll leaves it open for poaching of sharks, turtles and birds.

METHODS

All surveys were carried out on the leeward western fore reef slope, the lagoon and the passes (figure 1). Poor weather and sea conditions prevented investigation of the northern and eastern slopes. Permanent transects for long term monitoring were established at 10 m and 20 m on the leeward reef slope off Menai and North Islands, and in the lagoon. All quantitative surveys of benthic and fish communities were conduced along these permanent transects.

Benthic Community

Digital videography was used to record the benthic habitat bisected by each permanent 50 m transect. All video footage was obtained while swimming at slow speeds in order to maintain high image quality and to keep the area of reef recorded by the camera constant. In order to describe the benthic community, between 48 and 50 still images were extracted at timed intervals from the video footage of each transect. The benthic composition of each image was analysed by laying a transparent film over the image on which 20 evenly spaced points were marked. The substrate type under each point was recorded according to the life form categories described by English *et al.* (1997). Data were recorded from approximately 1 000 points from each transect. Additional qualitative observations of reef profile and structure were obtained at other locations in the lagoon, on the northwest reef slope, and in the southeast and southwest reef passes.

Figure 1. Map of Cosmoledo Atoll showing its location in the Aldabra Group of islands, top left (source: WCMC/Reefbase) and the western Indian Ocean, top right. The atoll measures approx. 10 km across from east to west.

Differences in the composition of the benthic community were analysed using a non-parametric permutation Analysis of Similarities (ANOSIM) procedure (Clarke, 1993) based on a rank similarity matrix using the Bray-Curtis similarity measure calculated from square-root transformed data. Data obtained from each image was treated as a single sample giving approximately 50 samples from each permanent transect. For these analyses, all forms of living hard coral were aggregated into a single category 'Live Hard Coral' and because of their low abundance and lack of power to discriminate between sites, data describing the cover of macro-algae, sponges, seagrasses, and other benthic invertebrates such as corallimorpharians and zoanthids were combined into the single category 'Other'. The resulting benthic cover categories used to examine differences between sites were: Live Hard Coral (LHC); Dead Coral with Algae (DCA); Coralline Algae (CA); Soft Coral (SC); Other (O); Rubble (R); and Sand (S). A post-hoc SIMPER analysis (Clarke, 1993) was conducted to identify the substrate types responsible for differences in the composition of the benthic community between sites.

In order to estimate the rate of recovery of coral populations since the coral bleaching event of 1998, the number of coral recruits within 6–12 1 m^2 quadrats placed haphazardly in the vicinity of each permanent transect was recorded. All colonies of coral smaller than 5 cm in diameter were identified to family level and counted. A 2-way nested ANOSIM procedure based on a similarity matrix using the Bray-Curtis similarity measure calculated using square-root transformed data was used to determine first, if there were significant differences in the composition of the recruit community at different depths at Menai and North Islands, and second, if there were differences between these two reef slope sites. Differences in the composition of coral recruits between the reef slope sites and the lagoon site were investigated using a 1-way ANOSIM procedure.

Fish Community

The fish community was recorded at four locations along each permanent transect. Using a modification of a point based visual census technique (Jennings *et al.*, 1996; Samoilys & Carlos, 2000), the abundances of 11 target families (Acanthuridae. Balistidae, Carangidae, Chaetodontidae, Serranidae (in this survey limited to the tribe Epinephelini), Haemulidae, Labridae, Lethrinidae, Lutjanidae, Pomacanthidae, Scaridae) within a circle with a radius of 7 m (153 m^2), including the overlying water column, were recorded by two divers conducting two censuses each. Fish smaller than 10 cm were not recorded.

Fish abundances were analysed using simple descriptive statistics. Since the data did not approximate normality, non-parametric Kruskal-Wallis ANOVA and Mann-Whitney U-tests were used to examine differences in abundances of fish among sites. Four families were excluded from these analyses either because of the low frequency with which representatives occurred in census areas (Haemulidae and Carangidae) or the enormous variation in abundances between locations resulting from their schooling behaviour (Lutjanidae and Lethrinidae). Similarities in fish community structure among samples were investigated using non-metric multidimensional scaling (MDS), based on rank similarity matrices using Bray-Curtis similarity measure (Clarke, 1993) calculated using square-root transformed data.

In addition, a cumulative list of fish species was developed for Cosmoledo Atoll from the results of fish censuses and also from species identified from video recordings made elsewhere around the atoll, predominantly in relatively shallow waters (5–10 m).

RESULTS

Bathymetry

The bathymetry of Cosmoledo atoll is typical for oceanic atolls, with a steep wall and reef slope with an inclination of between 80–90° at 40 m and deeper (figure 2). The depth at which the transition from the wall to the shallow platform occurred varied between 10 m and 25 m, and from a sharp edge at the top of the wall to a gradual decrease in slope to 10–20°. The main reef platform at 10–20 m depth sloped up gradually to a reef crest that was subtidal (about 1 m midway between Menai and North Islands) to inter-tidal at the islands and scattered islets. In-

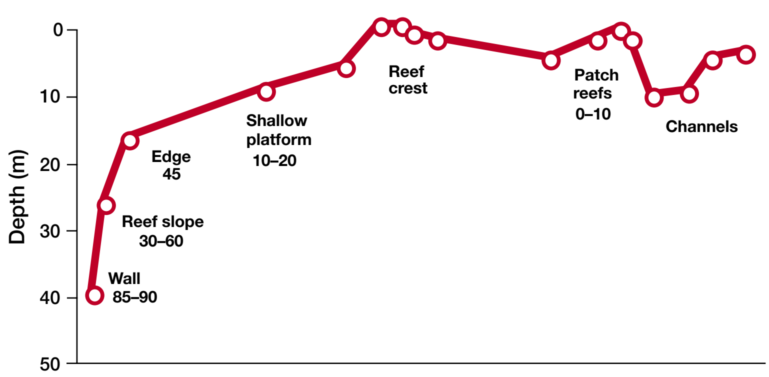


Figure 2. Typical reef profile across Cosmoledo Atoll from the outer reef wall to the inner lagoon. Note: horizontal axis is not to scale.

shore of the reef crest, the atoll has an extensive back reef area with depths <1 m which, in some places, was several kilometres wide according to navigational charts. The central part of the lagoon is about 8–10 km across and was predominantly 5–6 m deep with complex channels and patch reefs that extend to the surface. The majority of the lagoon floor was sandy and supported extensive seagrass beds, predominantly *Thalassondendron ciliatum*.

Benthic Community

Almost 200 species of hard coral were recorded during surveys conducted during this expedition (Obura & Sheppard, 2005). On the leeward reef slopes at depths shallower than 10 m, the cover of live hard coral (LHC) was generally only 5% or less, with occasional patches up to 20%. Evidence of widespread bleaching associated mortality of corals was common at these depths, with small and large coral heads covered by mature coralline and turf algal communities. In particular, algal-covered skeletons with the thick-columnar form of *Acropora palifera* were abundant in the shallow water north of the anchorage at Menai and along the leeward reef slope. Significant partial mortality of many colonies several years previously was also suggested by the existence of remnant patches of living tissue on some of these predominantly dead skeletons.

The cover of LHC was considerably more extensive on the reef slope deeper than ~8 m and was greater at 10 m than it was at 20 m. At 10 m, North Island and Menai Island exhibited 33.2% and 29.2% cover of LHC respectively, while at Menai at 20 m, LHC occupied 23.4% of the substrate (figure 3). The coral community was dominated by massive corals (figure 4), particularly poritids

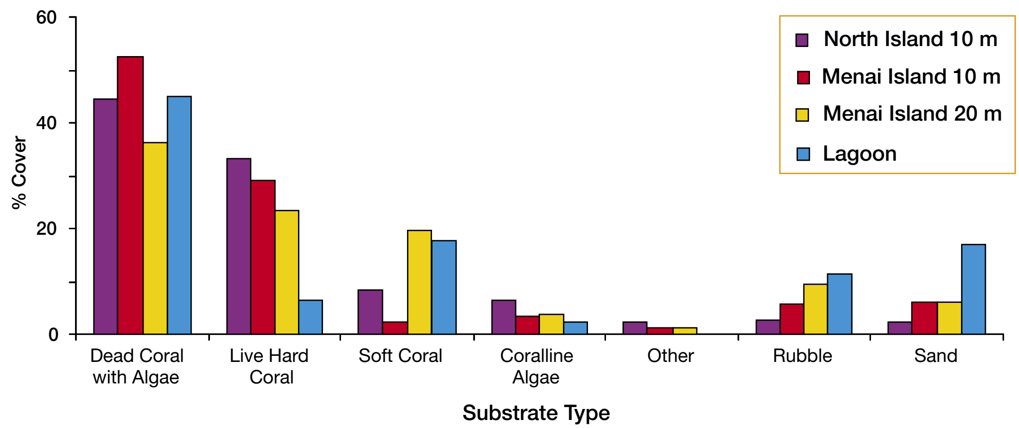


Figure 3. Percent cover of each substrate type along each permanent transect.

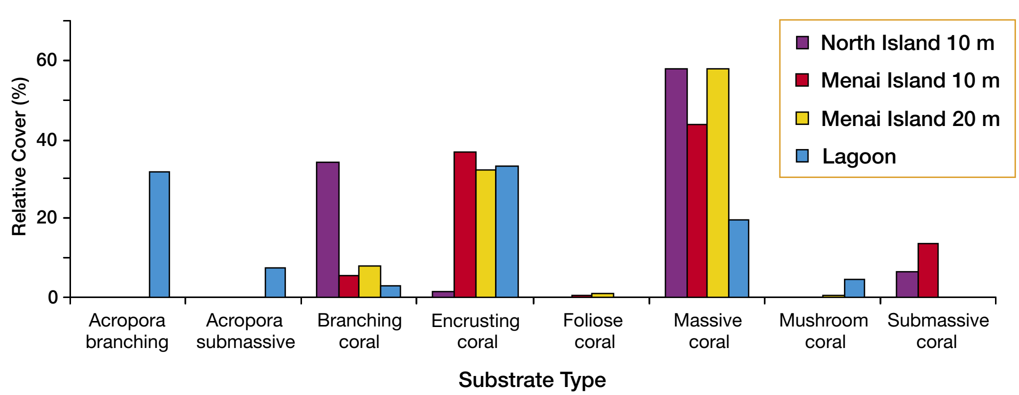


Figure 4. Relative contribution of each type of live hard coral to the total cover of live hard coral along each permanent transect.

Table 1. Summary of the results of SIMPER analyses investigating the contribution of variation in the cover of each substrate type to the overall differences in benthic communities observed between sites

Substrate Type	Average Dissimilarity	S.D.	Contribution %
Lagoon – Menai Island 20 m Average Dissimilarity = 48.64			
SC	10.22	1.13	21.02
LHC	9.43	1.4	19.38
S	8.30	1.16	17.07
DCA	8.19	1.23	16.84
R	7.46	1.14	15.34
CA	3.83	0.77	7.88
Lagoon – Menai Island 10 m Average Dissimilarity = 47.77			
LHC	10.82	1.49	22.64
SC	8.57	1.01	17.94
S	8.4	1.17	17.57
DCA	7.7	1.11	16.11
R	7.47	1.02	15.64
CA	3.68	0.73	7.7
Lagoon – North Island 10 m Average Dissimilarity = 55.71			
LHC	12.09	1.35	21.71
SC	10.02	0.98	17.98
S	9.41	1.12	16.89
DCA	9.4	1.17	16.88
R	7.17	1.02	12.87
CA	5.83	1.13	10.47

and a variety of faviids. Branching poritids were common along the transect at North Island, while encrusting forms constituted a considerable proportion of the cover of LHC along both transects (10 m & 20 m) at Menai Island. Acroporids were universally rare and were observed on the reef slope only in rapid visual surveys conducted away from the permanent transects. No acroporid corals were recorded along any permanent transects surveyed on the reef slope. Species diversity was noticeably low in the genera *Favia*, *Fungia*, *Pavona*, *Millepora*, *Alveopora*, and *Goniopora* and some species typically common on western Indian Ocean reefs were absent or rare, such as *Galaxea fascicularis* (rare), *G. astreata* (absent) and *Goniastrea pectinata* (absent) and *Lobophyllia* spp.

Between depths of 20 m and 30 m the reef slope varied: a broader shelf at the anchorage at Menai Island was dominated by *Halimeda*, while steeper slopes exhibited a more diverse coral community, with greater cover of hard and soft corals and coralline algae. At depths greater than 30 m, the substrate of the reef slope was dominated by coralline algae and bare rock surfaces, with only a small proportion of LHC (<5%). Black coral and gorgonian soft corals were abundant with some colonies exceeding 2 m in diameter.

In the lagoon, the cover of LHC along the permanent transect was only 6.6% (figure 3) and was ubiquitously low throughout the remainder of the lagoon. As on the shallow outer reefs, evidence of severe coral mortality approaching 100% was ubiquitous. At the boat anchorage, large dead *Millepora* colonies up to 1–2 m high were common. Living colonies of *Acropora* were recorded along the transect (figure 4), although most either were

Substrate Type	Average Dissimilarity	S.D.	Contribution %
Menai Island 10 m – Menai Island 20 m Average Dissimilarity = 43.05			
SC	8.87	0.91	20.60
LHC	8.26	1.25	19.19
DCA	7.30	1.15	16.95
R	6.69	0.92	15.55
S	5.71	1.07	13.27
CA	4.28	0.90	9.95
Menai Island 10 m – North Island 10 m Average Dissimilarity = 40.81			
LHC	9.75	1.34	23.89
DCA	8.29	1.10	20.3
CA	5.74	1.17	14.06
S	5.33	0.93	13.06
SC	4.93	0.55	12.07
R	4.3	0.60	10.54
Menai Island 20 m – North Island 10 m Average Dissimilarity = 48.42			
LHC	10.32	1.29	21.31
SC	10.16	0.90	20.98
DCA	8.54	1.19	17.64
R	6.28	0.92	12.96
CA	5.65	1.18	11.68
S	4.98	0.85	10.29

small and likely to have settled after the 1998 bleaching event or were surviving sub-massive colonies of *A. palifera.* The dead skeletons of branching *Acropora formosa*, *A. grandis* and *A nobilis* killed during the 1998 bleaching event were still recognisable. Dead coral covered with algal turf (DCA), sand, rubble and soft corals were the dominant substrate types within the lagoon (figure 3). The high proportion of sandy substrate in the lagoon would also limit the availability of suitable substrate for hard coral growth. Extensive seagrass and (several 100 m across) *Halimeda* beds growing on sandy substrate were also encountered in the lagoon.

The major reef passes, to the southeast and southwest, were surveyed on drift dives. Channel mouths were not steep-sided but instead were smoothly sloping from the reef crests on each side. The bottom substrate was predominantly rocky, with algal turf, coralline algae, *Halimeda* and seagrass. In places, extensive carpeting soft corals was common, and patches with small low-growing hard corals.

Limited coral bleaching was observed during surveys. Coral disease and signs of other damage to corals were absent.

The composition of the benthic community along each of the permanent transects surveyed was significantly different (Global R = 0.151; p = 0.001). Pairwise comparison between the transects showed that the greatest differences occurred between the lagoon and the two shallower reef slope sites (Lagoon – North Is. 10 m: R = 0.296; p = 0.001. Lagoon – Menai 10 m: R = 0.24; p = 0.001) and were attributable to the greater abundance of LHC on the reef slope and more soft coral and sand in the lagoon (table 1). Although the greater abundance of

LHC at Menai at 20 m contributed to differences in the benthic community between this site and the lagoon (Lagoon – Menai 20 m: R = 0.107; p = 0.001), the primary discriminating factor was the greater abundance of soft coral at depth on the reef slope. The greater abundance of soft coral at this deeper site also distinguished it from the two shallow reef slope sites, although the greater abundance of LHC and DCA at 10 m were additional contributing factors. (Menai 20 m - Menai 10 m: R = 0.082; p = 0.001. Menai 20 m – North Is. 10 m: R = 0.105; p = 0.001). Significant differences in the benthic composition of the two shallow reef slope sites (Menai 10 m – North Is. 10 m: R = 0.079; p = 0.001) were attributable to the greater abundance of LHC and coralline algae at North Island and the greater abundance of DCA at Menai.

Coral Recruitment

The average density of coral recruits varied little among sites, ranging between 5.1 m^{-2} (± 2.93 s.d.) at 10 m at Menai Island to 6.8 m^{-2} in the lagoon (figure 5). On the reef slope, recruitment was slightly greater at 20 m than 10 m, although the differences were not statistically significant (ANOVA, p >0.10). The taxonomic composition of the recruit community on the reef slope did not differ significantly between depths within sites (Global R = 0.082; p = 0.095) or between sites (Global R = 0.25; p = 0.667). One-way ANOSIM comparing the composition of coral recruits between sites confirmed that there were no differences between the two reef slope sites (R = -0.04; p = 0.025) but that the recruit community of both reef slope sites was significantly different from that of the lagoon (Menai – Lagoon: R = 0.884; p = 0.001; North Is. – Lagoon: R = 0.531; p = 0.001). Pocilloporids were the dominant family amongst recruits with the greatest average abundances (3 recruits·m^{-2} ± 2.70) recorded on the reef slopes at 10 m, averaging (figure 6). At 20 m, faviids were dominant ranging between 2.3 recruits·m^{-2} (± 1.89) and 2.5 recruits·m^{-2} (± 1.87) followed by pocilloporids and agariciids. Within the lagoon, recruitment was dominated by poritids and fungiids, at 4.1 recruits·m^{-2} (± 3.58) and 2.2 recruits·m^{-2} (± 3.51) respectively. The average number of acroporid recruits per m^2 did not exceed 0.5 at any of the locations sampled. The result of a subsequent multi-dimensional scaling analysis illustrates the relative similarity of the samples within and between sites on a 2-dimensional plot (figure 7). The intermixing of the

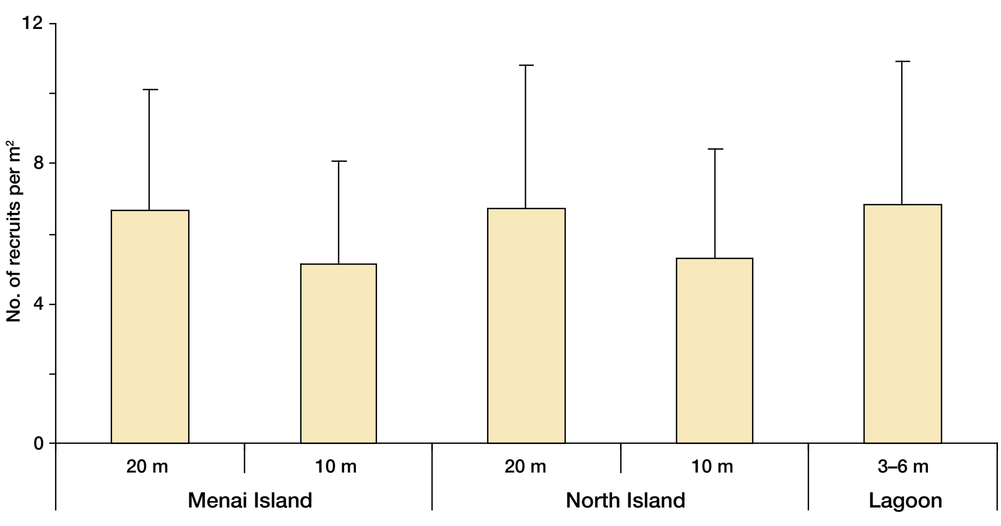


Figure 5. Mean number of coral recruits per m^2 (± SD) recorded in the vicinity of each permanent transect.

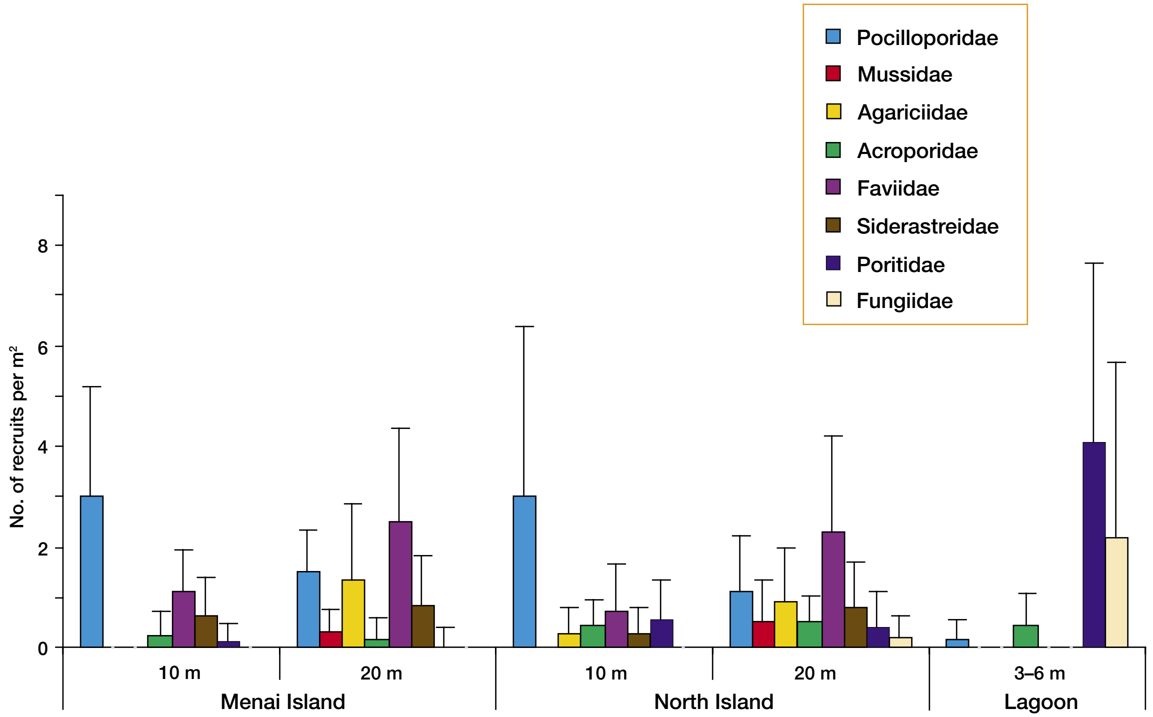


Figure 6. Mean number of coral recruits from each family per m² (± SD) recorded in the vicinity of each permanent transect.

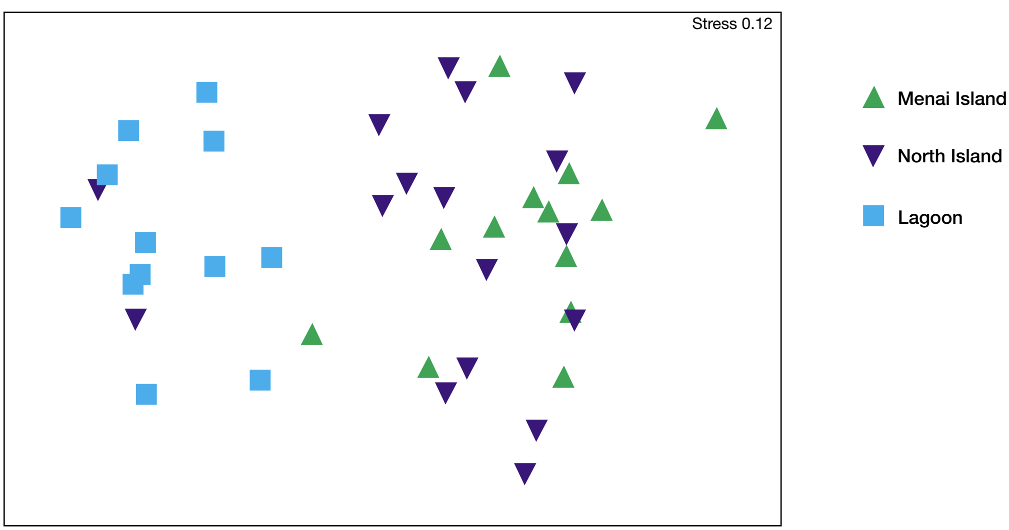


Figure 7. Multi-dimensional Scaling (MDS) plot illustrating the relative similarity of the assemblage of coral recruits recorded in each quadrat sampled along each permanent transect. The MDS plot is based on a Bray-Curtis similarity matrix calculated from square-root transformed data.

Table 2. Summary of the results of SIMPER analyses investigating the contribution of variation in the abundance of each family of recruits to the overall differences in recruit assemblages observed between sites

Family	Average Dissimilarity	S.D.	Contribution %
Menai Island – North Island Average Dissimilarity = 62.69			
Pocilloporidae	20.50	1.17	32.70
Faviidae	15.19	1.21	24.23
Siderastreidae	7.06	1.01	11.26
Agariciidae	6.83	0.88	10.89
Poritidae	5.30	0.61	8.45
Acroporidae	4.40	0.81	7.01
Lagoon – Menai Island Average Dissimilarity = 93.97			
Poritidae	31.46	1.46	33.48
Pocilloporidae	19.10	1.24	20.33
Fungiidae	15.72	0.74	16.73
Faviidae	14.18	1.19	15.09
Siderastreidae	5.47	0.83	5.82
Lagoon – North Island Average Dissimilarity = 87.18			
Poritidae	29.24	1.34	33.54
Fungiidae	15.54	0.74	17.82
Pocilloporidae	14.78	0.81	16.97
Faviidae	12.30	1.03	14.11
Acroporidae	4.64	0.89	5.32
Siderastreidae	4.50	0.77	5.16

samples obtained from the two reef slope sites and the distinct separation of the samples obtained from the reef slope from those obtained from the lagoon illustrates the results of the ANOSIM. Subsequent comparison of the species composition of recruit assemblages at each sites using SIMPER indicated that the difference in the abundance of recruits belonging to the families Poritidae, Pocilloporidae and Fungiidae accounted for approximately 70% of the variation in recruit composition between the reef slope sites and the lagoon (table 2).

The community composition of recruits differed considerably from that of the adult coral community that existed prior to the bleaching event of 1998. Previously, the shallow sites on the reef slope and the lagoon site were dominated by large colonies of branching and tabulate *Acropora*, as seen by the dead standing corals and rubble visible at the time of sampling. Among recruits, however, acroporids, while present at each site, were rare.

Fish abundances and community structure

A total of 172 fish species, belonging to 97 genera and 37 different families were recorded (table 3). Generally, acanthurids were the most abundant at each site, particularly in the lagoon (figure 8). Only at North Island at 20 m, where several schools of lethrinids and lutjanids were encountered within census areas, were acanthurids displaced as the most abundant family. Large and medium sized serranids were commonly recorded at all sites, particularly specimens of *Aethaloperca rogaa* and *Cephalopholis argus* on the reef slope sites and *Plectropomus punctatus* in the lagoon. Chaetodonts were commonly represented in each census, although species identified tended to be omnivorous. Obligate corallivores such as

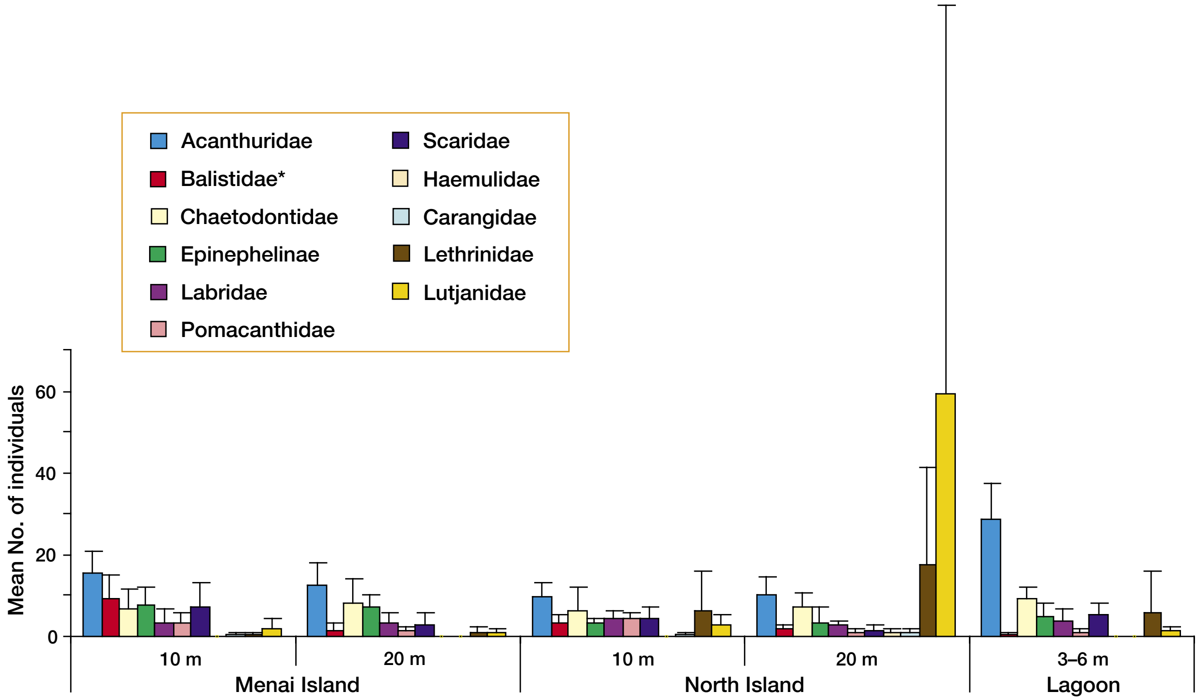


Figure 8. Mean number of individuals (±SD) from each family of fish recorded per fish census (153 m^2) conducted at each site. *The only family of fish to show significant differences in abundance between sites was the Balistidae.

Table 3. Summary of the number of genera and species recorded within each family of fish.

Family	No. of genera	No. of species	Family	No. of genera	No. of species
Acanthuridae	5	12	Muraenidae	2	4
Apogonidae	2	5	Nemipteridae	1	1
Balistidae	5	8	Ostraciidae	1	1
Belonidae	1	1	Pempheridae	1	1
Blenniidae	2	3	Pomacanthidae	4	5
Caesionidae	2	2	Pomacentridae	7	13
Carangidae	2	3	Priacanthidae	1	1
Chaetodontidae	4	18	Scaridae	3	7
Chanidae	1	1	Scombridae	2	2
Cirrhitidae	2	2	Scorpaenidae	3	4
Congridae	1	1	Serranidae	8	17
Haemulidae	1	2	Siganidae	1	2
Holocentridae	2	4	Sphyraenidae	1	1
Labridae	13	21	Syngnatide	1	1
Lethrinidae	3	7	Synodontidae	1	2
Lutjanidae	4	8	Tetraodontidae	2	2
Microdesmidae	1	1	Torpenidae	1	1
Monocanthidae	1	1	Zanclidae	1	1
Mullidae	2	6			

Chaetodon trifascialis and *Chaetodon trifasciatus* were scarce. Labrids, pomacanthids and scarids were recorded at each site with little variation in their abundances between sites. Lutjanids and lethrinids were frequently sighted, although their abundances within census areas varied enormously with the occurrence of large schools. Haemulids and carangids were rarely recorded in censuses, although they were commonly sighted elsewhere around the atoll. Not a single shark was sighted during approximately 90 person-dives conducted at Cosmoledo. The methods used leave some uncertainty in the species identification, especially within the families Scaridae and Labridae, and thus some records were excluded.

No significant differences in total fish abundance were found between sites (figure 9) although the small sample size and area surveyed (4 counts of 153 m^2) limited the statistical power of the data allowing the outcome of the analysis to be heavily influenced by the enormous variation between samples within sites caused by the irregular occurrence of large schools of fish in some censuses. The highest average density of fish was recorded at North Island at 20 m (106.8 fish per 153 $m^2 \pm 116.7$), although this is attributable to the occurrence of schooling lutjanids and lethrinids in one sample, resulting in a very high standard deviation. The remaining 4 sites had densities of about 50 fish per 153 m^2. The Ballistidae was the only family to show statistically significant variation in abundance between sites (figure 8), with fewer representatives in the lagoon compared with the sites surveyed on the reef slope. The composition of the fish community in the lagoon was significantly different from all other sites except Menai Island at 10 m ($p<0.05$) primarily due to the greater abundance of acanthurids within the lagoon. In addition, the tighter grouping of the samples obtained from the lagoon in the MDS plot (figure 10) illustrates that the composition of the fish assemblage surveyed in the lagoon was more homogenous than those surveyed at the reef slope sites. In contrast, the disaggregated appearance of samples obtained from Menai Island at 10 m illustrates the considerable variation in the composition of the fish community observed at this site, and also explains why no significant differences in the fish assemblages were found between these samples and those obtained from the lagoon.

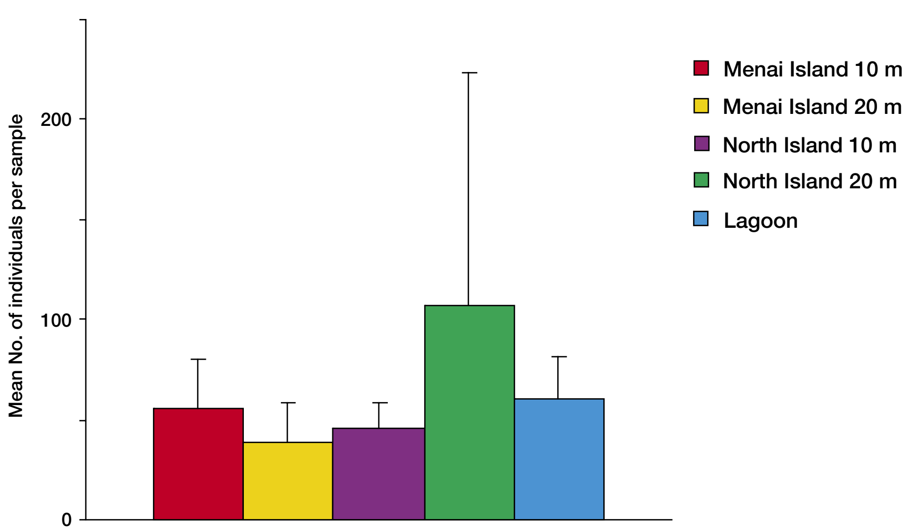


Figure 9. Mean total abundance of fish per 153 m^2 (±SD) sample recorded at each site.

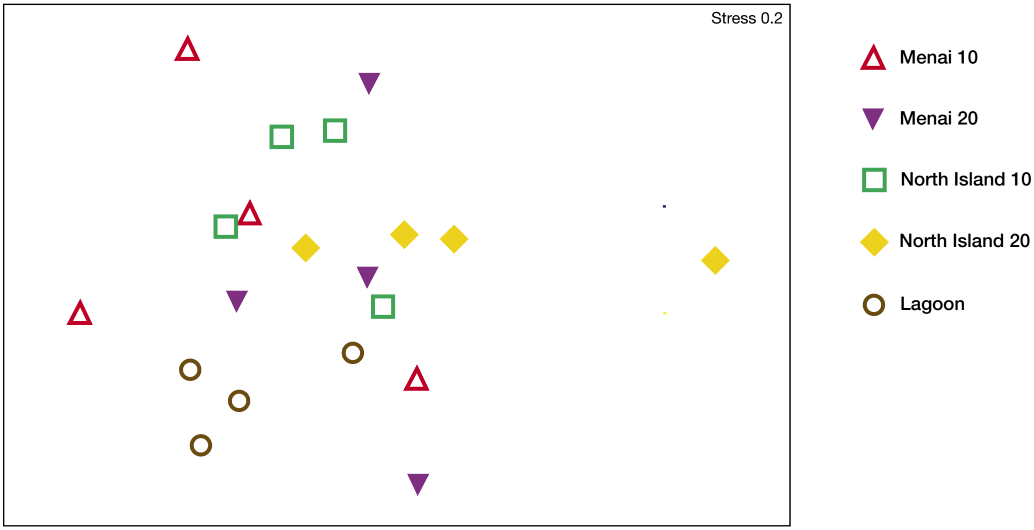


Figure 10. Multi-Dimensional Scaling (MDS) plot illustrating the relative similarities in the fish community composition recorded in each census (4 replicates at each of 5 sites). The MDS plot is based on a Bray-Curtis similarity matrix calculated using square root transformed data. Each point represents one census.

DISCUSSION

The coral reefs of Cosmoledo Atoll show characteristics consistent with those reported for the central Indian Ocean and consistent with early studies of the Seychelles islands (Barnes *et al.*, 1971) and recent studies of Aldabra (Teleki *et al.*, 2000; Downing *et al.*, in review). Highest coral cover (25–40%) and diversity was observed at intermediate depths on the reef slope, where the shallow reef platform curves downwards towards the deep reef wall, although evidence of up to 50% mortality of corals at these depths and even greater mortality in the lagoon and in shallower waters (<10 m) on the reef slope was abundant (Sheppard & Obura, 2005). The appearance of dead coral heads with mature algal communities and many young corals estimated at 2–4 years old provided additional compelling evidence that the mortality of corals on Cosmoledo occurred during the El Niño of 1998, coincident with widespread coral mortality throughout the Indian Ocean (Lindén & Sporrong, 1999; Wilkinson *et al.*, 1999). The level of mortality was similar to that reported for Aldabra Atoll during the El Niño (Spencer *et al.*, 2000; Teleki *et al.*, 1999, 2000), but less severe than that which occurred in the inner granitic islands of the Seychelles (Quod, 1999; Turner *et al.*, 2000).

Almost 200 species of scleractinian corals were found during this expedition (see also Sheppard & Obura, 2005). Many *Acropora* species were notably absent (e.g. the staghorn *A. formosa*, *A. grandis* and *A. nobilis*) though their dead skeletons were still recognizable in the lagoon. The coral communities on the reef slope at 10–20 m were dominated by massive, encrusting, sub-massive and branching coral forms, including *Porites* species and a variety of faviids. Species diversity was noticeably low in the genera *Favia, Fungia, Pavona, Millepora, Alveopora*, and *Goniopora*, which may be due to the isolation of the atoll, or mortality during the El Niño (Sheppard & Obura, 2005). Some species typically common in western Indian Ocean reefs (Rosen, 1971b) were absent or rare, most noticeably *Galaxea fascicularis* (rare), *G. astreata* (absent) and *Goniastrea pectinata* (absent) and *Lobophyllia*.

Acropora, which older literature suggests was once the dominant genus in shallow waters of reefs of this region (Rosen, 1971b), was the genus most severely affected. *Acropora* colonies were virtually eliminated in 1998. In contrast, the genus *Porites* was one of the best survivors and dominates recruitment in the lagoon, and many faviid corals also survived well. The sizes of most corals in waters less than 8 m deep indicated that they were mainly less than 4 years old. On reef slopes at depths of 10 m and greater, survival had been considerably greater preserving significantly higher coral cover and coral species diversity (Sheppard & Obura, 2005). Consistent with many oceanic atolls, Sheppard and Obura (2005) hypothesize that 10 m represents a 'transition depth' between upper waters that suffered near 100% mortality of corals in 1998 and deeper waters where mortality was high but not catastrophic, perhaps at levels of 50% and decreasing to zero below 30 m. This suggests that at Cosmoledo, the thermocline between upper warmer waters and deeper cooler waters varies around 10 m and deeper, with the top 10 m becoming superheated during El Niño doldrums conditions. As a consequence, the entire lagoon and the shallower portions of the reef slope suffered near total mortality of corals. The presence of remnant and diverse coral communities below 10 m on Cosmoledo likely supplies the reservoir of larvae which are recolonising the lagoon and the shallow reef slope. This pattern appears consistent at many atolls, and presents some hope that the El Niño and climate change-related threat may be less severe for oceanic atolls than continental reefs, especially if acclimatization of corals and zooxanthellae to warming conditions occurs (Coles & Brown, 2003; Hughes *et al.*, 2003). However, Cosmoledo is situated at 12°S, which is predicted to be the most vulnerable latitude in the western Indian Ocean to warming sea surface temperatures (Sheppard, 2003). Conditions similar to those in 1998 are predicted to recur approximately every 5 years by 2012–2015, thus preventing successful reproduction and recovery of coral communities.

At present though, coral recruitment is not limiting with recruit densities ranging between 5 and 6.8 recruits per m^2, which were similar to recent surveys conducted at neighbouring Aldabra Atoll (Teleki *et al.*, 2000; Stobart *et al.*, 2001, 2002) and the granitic islands focusing on Mahé (Wendling *et al.*, 2002; Engelhardt, 2003). On the reef slope, the greater recruitment at 20 m compared with 10 m suggests either that the greater abundance of adult colonies at depth act as a reservoir for coral recruitment or that the deeper environment is more conducive to settlement and survival. These preliminary surveys of coral recruitment determined that there were positive signs of recovery from the extensive mortality caused by the 1998 bleaching event.

Prior to the 1998 bleaching event, the shallow reef areas of Cosmoledo were dominated by branching and tabulate forms of *Acropora*. However, all genera belonging to the family Acroporidae were rare at all sites sampled. In shallow water, where mortality of corals was most severe, pocilloporids were the most abundant recruits on the reef slope while poritids and fungiids dominated the lagoon. At deeper sites on the reef slope, differences between the community composition of adults and recruits were less obvious. Faviids were the most abundant recruits and were also well represented among adult colonies. The commonness of the agariciid genus *Pavona* and the siderastreid genus *Coscinaraea* among recruits on the reef slope is consistent with other surveys of coral recruitment conducted at Aldabra (Stobart *et al.*, 2001), the Maldives (Clark, 2000; Zahir *et al.*, 2002) and in northern Kenya (Obura, 2002; and this volume). The variation between the communities of adult corals, which were remnants of the community that existed prior to 1998, and post-bleaching recruits, particularly in shallow waters, suggests that Cosmoledo's reefs are experiencing a shift in species composition and that the reefs that are now developing may become quite different to those that existed prior to 1998.

More than 170 species of fish were recorded at Cosmoledo Atoll and although the time available for rigorous surveys was small, this is comparable with the 212 species counted for Aldabra Atoll in 12 days of surveys in November 1999 (Teleki *et al.*, 2000), 205 species in 2001 (Stobart *et al.*, 2001) and 221 species in 2002 (Stobart *et* *al.*, 2002).With increased surveys, these numbers may approach estimates of fish diversity of >350 species for similar-sized areas on the nearby Tanzanian mainland (Garpe & Ohman, 2003; Obura *et al.*, in review) and an estimated figure of about 400 species for the Mascarene plateau (Jennings *et al.*, 2000).

The fish families Serranidae, Lutjanidae, and Lethrinidae include many economically valuable species targeted by artisanal fishermen (Jennings *et al.*, 1996). High abundances of species within these families at Cosmoledo suggest that the atoll is currently subject only to low or moderate fishing pressure. Although it was not possible to examine long-term trends in fish densities, fishermen active in the area claim that the average body size of fishes in these families has declined considerably during the past two decades. In addition, the absence of sharks during 90 person-dives is cause for concern, particularly considering that sharks were abundant at Cosmoledos during the 1980s and remained so through to at least the early 1990s (Mortimer J., pers comm.). Legal fishing has been a regular activity on Cosmoledo Atoll for decades, and while its isolation and lack of enforcement lends itself to the activities of poachers and illegal fisherman from other countries (Government of Seychelles, 2002), it also confers a degree of protection through its inaccessibility to anything other than larger ocean-going fishing vessels. The reef fish communities at Cosmoledo Atoll are relatively healthy although the absence of sharks indicates that fishing pressure is having some influence.

It is also evident that the extreme coral mortality in shallow waters suffered during 1998 has also affected the coral associated fish community. The abundance of *Chaetodon trifascialis* and *C. trifasciatus,* often numerically dominant among chaetodonts on relatively undisturbed coral reefs (Bouchon-Navaro & Bouchon, 1985; Chabanet *et al.*, 1997; Öhman *et al.*, 1998), has been shown to be positively correlated with the amount of live coral (Jennings *et al.,* 1996). The low abundance of *C. trifascialis,* an obligate predator of *Acropora* polyps (Findley & Findley, 1989), is likely to be a consequence of the almost total absence of this coral genus at Cosmoledo. In addition, increases in abundance of herbivorous acanthurids and, in some cases, scarids, can be attributed to the increased abundance of filamentous algae growing on coral substrata killed during the El Niño of 1998 (Lindahl *et al.*, 2000; Chabanet, 2002; McClanahan *et al.*, 2002; Sheppard *et al.*, 2002). It would be interesting to study the changes in the abundance of herbivorous fish communities in response to potential changes in coral and algae cover at the sites to understand the higher order impacts of bleaching-related coral mortality and reef recovery.

It is evident that the coral communities at Cosmoledo, particularly in the lagoon and on the shallow reef slope (<10 m) were severely affected by the increase sea temperatures caused by the El Niño of 1998 and that this, in turn, has influenced the abundance of both corallivorous and herbivorous fish. However, despite these changes, it is also evident that the coral community is recovering, albeit slowly and with a different species assemblage. The data obtained during this first scientific expedition describes the current condition of the coral and fish communities at Cosmoledo Atoll and has established a baseline against which future changes and recovery can be compared. In addition, these data enable a comparison of the patterns and rate of recovery between reefs that are relatively undisturbed by human activities with those that are heavily influenced, such as those along the coast of east Africa. Moreover, regular monitoring of the condition of Cosmoledo Atoll conducted over a long period will be invaluable for the future management of this biodiverse atoll and for the investigation of the influences of global climate change on coral reef habitats.

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Section 2

Thematic Reports

Coral Settlement Patterns in the Mombasa Marine National Park

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key words: coral recruitment, settlement, seasonal variation

ABSTRACT

Coral settlement patterns were measured at two sites in the Mombasa Marine National Park for a 2-year period from May 2001 to February 2003. Artificial settlement tiles were deployed for approximately 3-month periods and were collected in February, May, August and November of each year. The mean number of coral spat settled on collected tiles varied from 0.75 (± 0.79 s.d.) per tile in August 2001 to a maximum of 16.70 (± 7.53 s.d.) in November 2002, corresponding to mean densities of 8–740 m^{-2}. The maximum number of spat recorded on a single tile was 38 (November 2002). Although peak settlement rates were recorded in November of each year, settlement was sufficiently variable between months and years to obscure a clear seasonal cycle. Settlement was highest at the study site with the best water flow and exchange with the open ocean. Pocilloporids (*Pocillopora* spp.) dominated settlement (76%), followed by poritids (19%), then 'others'. Patterns suggest peak coral recruitment in September–November each year when water temperatures are increasing the fastest prior to reaching their seasonal maxima in March–April, but with substantial recruitment of pocilloporids and poritids throughout the year.

INTRODUCTION

Coral recruitment refers to the addition of new individuals to the coral population and can be defined as the initial sighting of recently settled juveniles in the adult habitat (Caley *et al.*, 1996). Recruitment has a major influence on the adult coral community structure (Birkeland *et al.*, 1981), with temporal and spatial variations in recruitment pattern playing an important role in the population dynamics of coral communities. Local populations may be regulated by recruitment such that when recruitment limitations occur, there are likely to be fluctuations in population size (Hughes *et al.*, 1999).

Artificial settlement plates for coral recruits can be used to provide a measure of the relative abundance of coral recruits in time and space (Harrison & Wallace, 1990), as an indicator of potential recruitment to natural surfaces. Genetic studies suggest that most recruitment to reefs is from larvae produced locally (Ayre *et al.*, 1997), thus the settlement of coral planulae on or near colonies of their own species is a common occurrence (Birkeland *et al.*, 1981). However, the extent to which variation in recruitment reflects variation in adult populations is unclear.

This report explores the spatial and temporal patterns of recruitment in the Mombasa Marine National Park (MMNP) through the use of artificial settlement tiles. Two sites are studied, with data presented here for two years (May 2001 to February 2003) of quarterly deployments of the tiles. The general objective of this project was to identify patterns of coral recruitment at the two sites by investigating:

- Variation in the number of coral recruits between the two sites;
- Variation in the number of coral recruits among the eight samples obtained over two years;
- The composition of coral recruits at the family level;
- Size relationships of recruited corals.

Recent studies at the MMNP have examined the recovery of the coral reef following mass bleaching and mortality during the El Niño of 1998, focusing on overall benthic cover (McClanahan *et al.*, 2001), and dynamics of coral recruits on natural surfaces (Tamelander, 2002). This study focuses on an earlier phase of coral life history and investigates settlement patterns at early stages of recruitment.

METHOD

The Mombasa Marine National Park (4° 0.0' S, 39° 45' E; figure 1) was established and formally gazetted as a protected area in 1986, and encloses part of the lagoon, back reef and reef crest habitats of the Bamburi-Nyali fringing reef. The marine environment is characterized by warm tropical conditions varying at the surface between 25° C and 31° C during the year, stable salinity regimes and moderately high nutrient levels from terrestrial runoff and groundwater. The area has semi-diurnal tidal regimes varying from 1.5 to 4 m amplitude from neap to spring tides, creating extensive intertidal platform and rocky-shore communities exposed twice-daily during low tides.

Monsoon winds are the dominant climatic influence in this area, blowing from the northeast (December–March) and southeast (May–October) with 1–2 months in between characterized by variable and gentler winds. Two study sites located within the back reef/lagoon areas were used. The first, Coral Gardens, is situated adjacent to a small depression forming a small channel through the reef crest that enables water exchange on rising and falling tides. The second, Starfish, is farther inside the lagoon and has less vigorous water exchange.

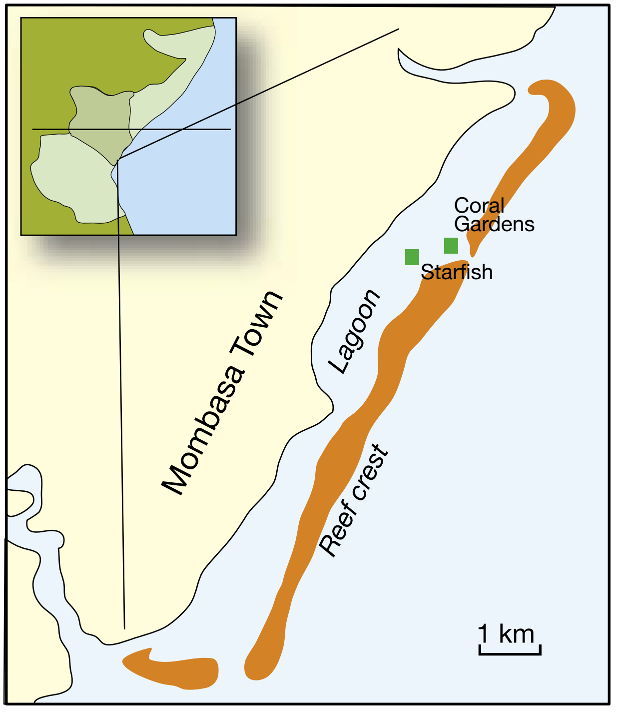

Figure 1. Map of the Mombasa Marine National Park, showing the reef crest, the enclosed lagoon, and the location of the two study sites. Coral Gardens is located near a shallow channel through the reef crest allowing high water exchange, while Starfish is located in the lagoon.

This report presents data obtained between May 2001 and February 2003 when settlement tiles were deployed concurrently at both sites. The study used settlement tile methods modified from English *et al.* (1997). Wire mesh racks were firmly attached to the bottom using metal pegs at a depth of approximately 0.5 m at maximum low tide. Terracotta bathroom tiles (15 cm x 15 cm) were attached to these racks inclined at 45 degrees with the unglazed side facing downwards. Tiles were attached to the racks using wire ties and deployed for a 3 month period. Once collected from the field, the tiles were placed in household bleach for 2–3 days and then dried. Dried tiles were examined using a dissecting microscope. The number of coral spat on each tile was recorded along with the surface (rough/smooth) of settlement. Each coral was identified to family level and its maximum diameter measured. Other organisms settled on tiles such as tubeworms, bivalves and oysters were counted and general observations were made on the amount of debris present.

RESULTS

Deployment and recovery dates and durations of tiles are shown in table 1, which have been compiled into approximate quarterly periods or 'seasons' for further analysis. The seasons are numbered 1–4 starting with the first tile recovery each year, which corresponds to the early onset of the northeast monsoon and settlement of spat from approximately November to February each year. The appearance of the racks and tiles after three months, a 2 mm coral recruit on the tiles and natural recruits of *Pocillopora* and *Porites* are shown in figure 2 on next page.

The mean settlement rates of coral spat per tile varied from 0.75 (± 0.79 s.d., August 2001) to a maximum of 16.70 (± 7.53 s.d.; figure 3 on next page). The greatest number of spat recorded on a single tile was 38 (November 2002). Recruitment was consistently higher at Coral Gardens compared with Starfish, ranging from 2.1 to 27 times higher. Each factor, site, year, season and coral family had a significant effect on recruitment rates, though a fully factorial ANOVA was not possible due to missing samples (table 2 on page 171). By inspection of figure 3, a strong interaction effect of site*year*season*taxon is evident, being highest for pocilloporids in the northeast monsoon (Dec–Feb) of 2002 at the Coral Gardens. Abundance of coral spat on tiles corresponded to recruit densities of 30–740 m^{-2} at Coral Gardens and 8–60 m^{-2} at Starfish.

Recruitment was overwhelmingly dominated by corals of the family Pocilloporidae (76%, figure 4 on page 171) and Poritidae (19%) and only 5% from other families. Non-pocilloporids were only prominent among recruits during the September–November season in both years and in June–August in 2002, and all 'others' were noted only during this latter period. Starfish had a lower diversity of corals than Coral Gardens, with 93% pocilloporids, 5% poritids and 1.8% 'other'.

The average size of coral spat on the tiles varied from 1 to 3 mm in diameter, with a maximum recorded size of 17.5 mm for a pocilloporid in February 2002. Spat sizes were smallest in November–February when density was highest (table 3 on page 171), and highest a few months later in June–August. Pocilloporids were significantly larger, and poritids significantly smaller than other coral spat. No annual effect was seen, and spat sizes were larger at Coral Gardens than at Starfish, most likely due to the predominance of larger pocilloporids. There was high

Table 1. Sampling details of settlement tiles in the Mombasa Marine National Park. 'Month' indicates the standardized periods used in the paper

Year	Season	Date		Coral Gardens		Starfish	
		immersed	retrieved	# tiles	# corals	# tiles	# corals
2001	May	13-Mar-01	22-Jun-01	21	101	20	32
	Aug	22-Jun-01	20-Oct-01	20	14		
	Nov	1-Oct-01	18-Jan-02	18	96	20	4
2002	Feb	18-Jan-02	25-Apr-02	21	94	20	27
	May	25-Apr-02	13-Aug-02	20	112	20	15
	Aug	13-Aug-02	12-Nov-02	20	200	19	24
	Nov	12-Nov-02	24-Feb-03	20	329	20	28
2003	Feb	17-Feb-03	19-May-03	24	174	20	22
Grand Total				1 120		152	

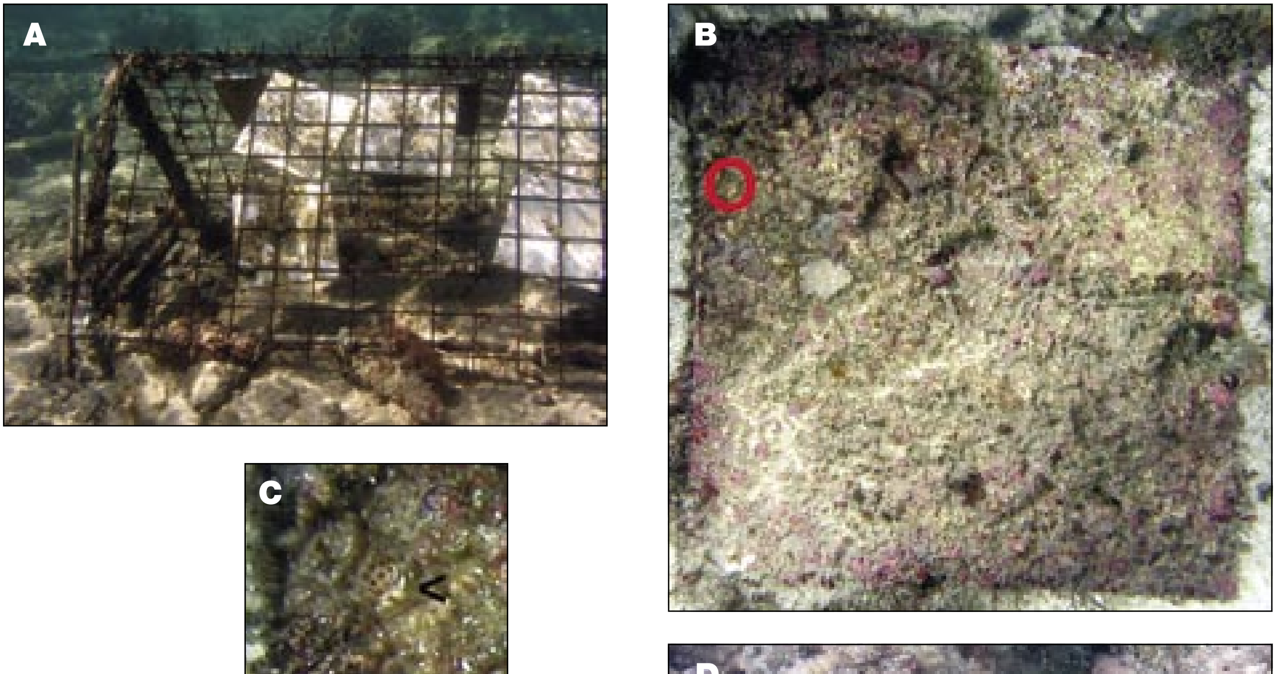

Figure 2. A) Settlement tile rack: tiles are angled at 45° with the rough surfaces facing downwards. B) Tile after approx. 3 months in submergence and growth of algal turf, coralline algae and various invertebrates. A coral spat is circled in red at top left, and shown in detail in (C) at approx. 2 mm diameter and 7 polyp stage. D) coral recruits on natural substrates at about 2 cm size, with Pocilloporidae (left, *Pocillopora damicornis*) and Poritidae (right, *Porites lichen*).

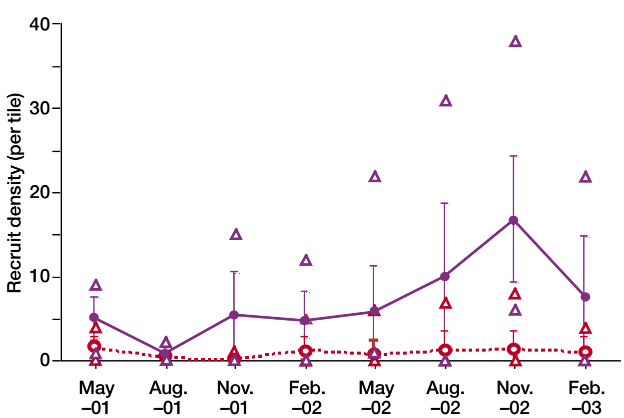

Figure 3. Abundance of recruits per tile (225 cm²) for the 2 year period from May 2001 to February 2003 (mean, standard deviation, minimum and maximum).

Table 2. One-way ANOVA results of settlement rates by year, season, site and family (data from figure 3). Fully factorial ANOVA could not be conducted due to missing data

Factor	F	p	Post-Hoc test (LSD)
Year	14.367	<0.001	2002 >> 2001, 2003
Season	8.498	<0.001	Dec-Feb >> Mar-May,Jun-Aug,Sep-Nov
Site	98.164	<0.001	Coral Gardens >> Starfish
Coral	69.3	<0.001	Pocilloporidae >> Poritidae>> Other

Table 3. One-way ANOVA results of coral spat diameter by year, season, site and family. Fully factorial ANOVA could not be conducted due to missing data

Factor	F	p	Post-Hoc test (LSD)
Year		ns	
Season	5.655	0.001	Jun-Aug > Mar-May, Sep-Nov > Dec-Feb
Site	11.044	<0.001	Coral Gardens >> Starfish
Coral	54.063	<0.001	Pocilloporidae >> Other > Poritidae

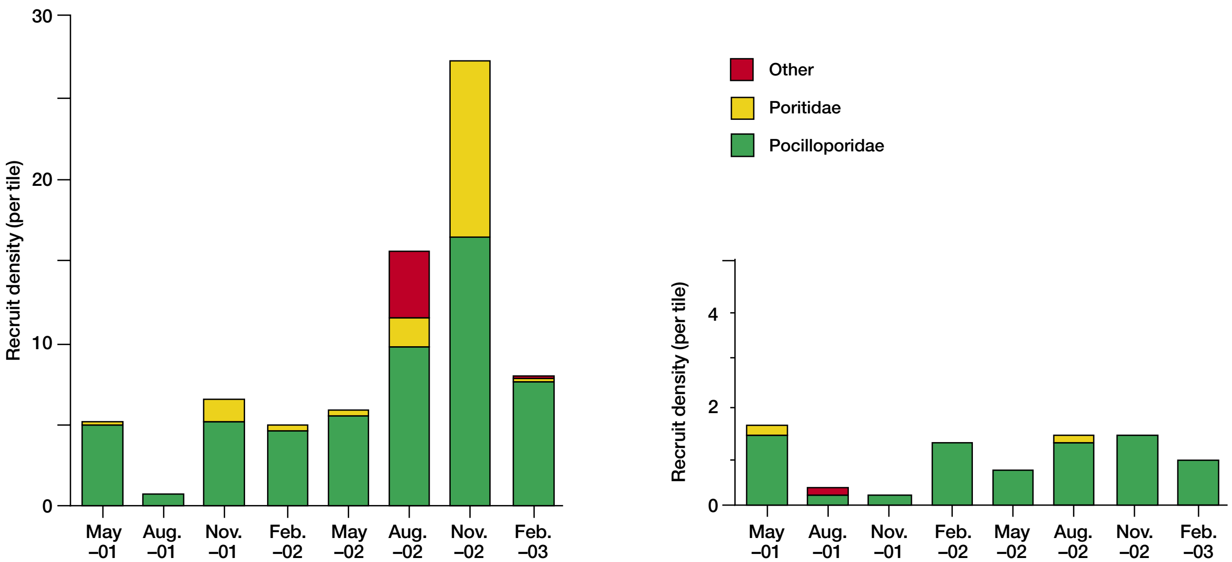

Figure 4. Composition of recruit populations at Coral Gardens and Starfish.

variation in the sizes of Pocilloporidae and a large degree of overlap with those of Poritidae (figure 5 on next page), however the Pocilloporidae showed larger maximum size colonies.

DISCUSSION

The dominant pattern in this dataset was of significant variability across sites, seasons and years (table 3). Within that, patterns were not statistically significant, but clearly

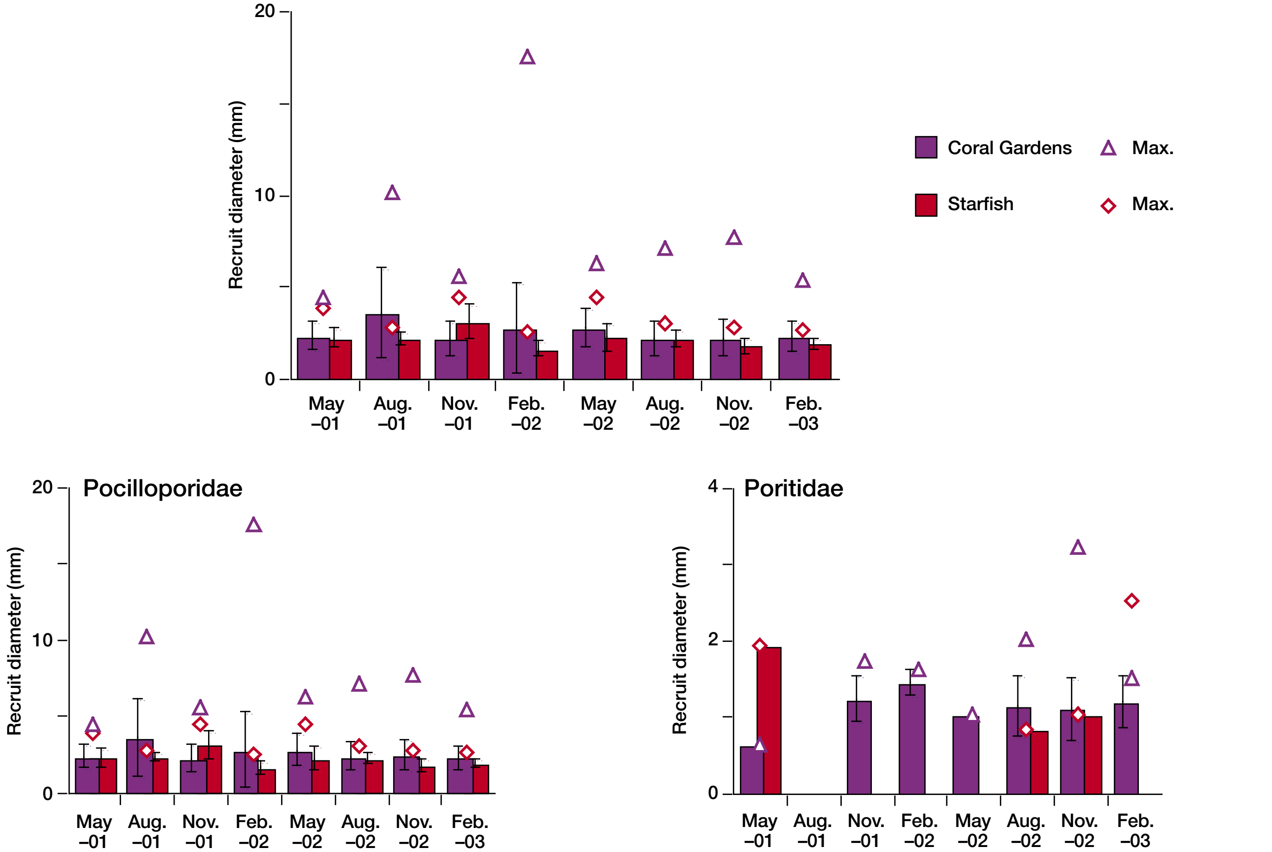

Figure 5. Mean, standard deviation and maximum sizes (mm) for all corals (above), Pocilloporidae (bottom left) and Poritidae (bottom right).

showed higher settlement at Coral Gardens compared with Starfish and during September–November relative to other times of the year (though additional years of sampling will be needed to confirm this). Additionally, pocilloporids (mostly *Pocillopora* spp.) were the dominant settlers throughout the year. Coral Gardens is located adjacent to the only cross-reef channel for over 3 km north and south, giving it the strongest current flow conditions and therefore most likely the greatest exposure to coral larvae carried in from the reef front. Both the level of settlement and diversity are indicative of greater exposure to potential recruits. Additionally, water and substrate conditions may be more conducive to successful settlement and post-settlement survival of corals, as the water may be more aerated and substrate surfaces cleaner than at Starfish, which experiences calmer conditions within the lagoon. Coral Gardens is also more topographically complex, and has more fish than Starfish, which may confer better conditions for settlement and post-settlement survival.

Corals of the family Pocilloporidae dominate the settlement plates, and are the most abundant recruits to natural surfaces in the area (Tamelander, 2002) and are among the most abundant recruits at other sites in Kenya (Obura, in review). Most of these are in the genus *Pocillopora*, and abundant *P. damicornis*, *P. verrucosa* and

P. eydouxi of multiple year classes were found adjacent to and on the settlement racks (Obura, pers. obs.). Corals in the genus *Pocillopora* are well known from other parts of the world for their prolific reproductive output, primarily of brooded planula larvae that often settle very close to the parent colony (Richmond & Jokiel, 1984; Richmond, 1985; Glynn *et al.*, 1991). The year-round presence of *Pocillopora* recruits recorded here (and pers. obs) suggest a similar year-round reproductive output in Kenya, with a strong peak, in September–November, particularly noted in 2002. This occurs when water temperatures are rising at their maximum rate (McClanahan, 1988; Obura, 2001), and prior to maximum temperatures experienced in March–April at the end of the northeast monsoon season. The presence of poritids was most strongly noted in September–November of both years, though low levels were present throughout the year. Other corals were not abundant enough to draw any strong conclusions concerning seasonality in their reproduction.

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Local Level Coral-Reef Fisheries Management in Diani-Chale, Southern Kenya:

Current Status and Future Directions

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key words: Diani-Chale, fisheries, local level management, co-management, local institutions

ABSTRACT

The current regime of fisheries management and issues concerning the achievement of a more locally oriented system of fisheries management in Diani-Chale, southern Kenya are examined. Fisheries management in the area is characterized by a lack of strong government capacity for regulation, weakened local institutions, and an absence in the ability to exert control over the use of the fishery. Local level management requires the development and use of local institutions that can govern the use of fishery resources. The landing sites and associated fishing grounds constitute a socio-ecological unit and were identified to be the appropriate level at which many fishery management issues could be resolved. A more formal role for these entities, together with clarification of tenure of fishing grounds and support for the development and enforcement of local rules for the use of the fishery are essential actions that should be taken by government to enable more local level fisheries management. The socio-economic condition of fishers, the fear fishers have over the loss of landing sites, and the continued perception of the imposition of a marine reserve in the area pose barriers to initiatives seeking to further local level management. The need for a coordinated approach among agencies working on fisheries issues in the area is essential so as to avoid the erosion of social trust with local fishing communities.

INTRODUCTION

The Diani-Chale area on the south coast of Kenya has been the focus of much of the participatory monitoring work undertaken by CORDIO East Africa (figure 1). The area represents one of the most degraded coral reef systems in East Africa (McClanahan & Obura, 1995) and is characterized by high fishing effort, high levels of conflict among fishers and other coastal resources users, with relatively poor enforcement of regulations. The efforts of CORDIO East Africa have focused on the use of participatory monitoring and research approaches as a strategy to create awareness among fishers and engage them to be more involved in resource management (Obura, 2001; Obura *et al.*, 2002). The development of a local system of resource management in Diani-Chale is an evolving process and is of interest to a number of organizations active in the area, including CORDIO East Africa. The aim of this paper is to provide a context to the current regime of fisheries management in Diani-Chale and to relate this to attaining a more locally oriented system of fisheries management. The paper examines local level institutions and initiatives relevant to fisheries management and highlights some of the issues relevant to the development of a local and collaborative system of fisheries management. It concludes with a number of recommendations that are pertinent to

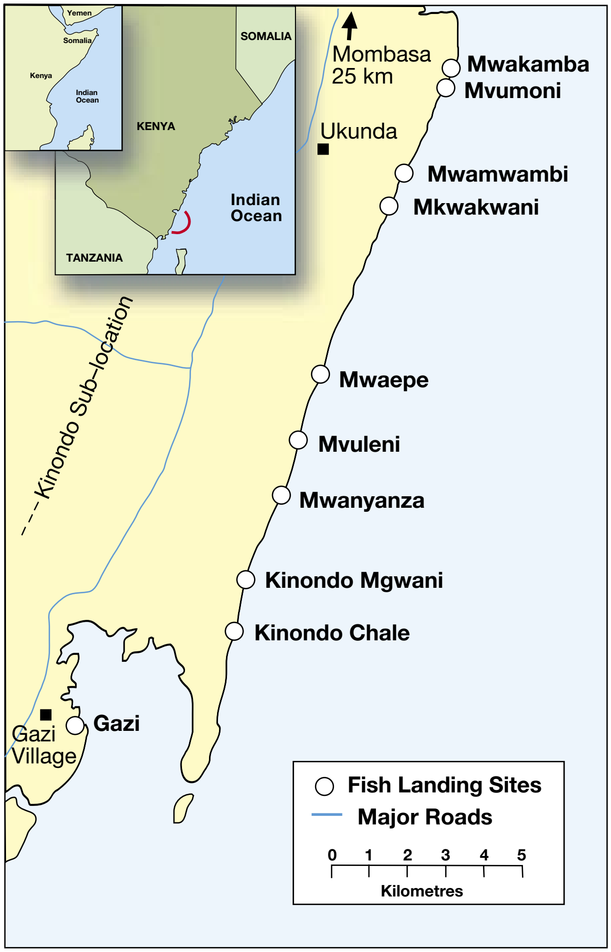

Figure 1. The general location of the Diani-Chale area on the south coast of Kenya (inset) and some the fish landing sites used by fishermen within the study area (outset).

improve local level resource management in Diani-Chale.

FISHERIES IN DIANI-CHALE: AN OVERVIEW

Fishing Activities

The near shore fisheries in Diani-Chale, as in most parts of East Africa, are largely artisanal in nature, serving both subsistence needs as well as local markets. Fishing activities occur within the coral reef lagoons and sea grass beds sheltered by the fringing reefs and extend beyond the reef in calmer conditions. An estimated 600 fishers fish in Diani-Chale (Malleret-King *et al.*, 2003), within an area that encompasses between 30–40 km^2, equivalent to a fisher density of 15–20 fishers·km^{-2} which is much higher than that reported elsewhere (Obura *et al.*, 2002). Fishing and fish trading related activities constitute between a fifth and a quarter of all activities of households in Diani-Chale (CORDIO, unpublished data). The catch primarily consists of demersal reef species and some pelagic species (King, 2000; McClanahan & Mangi, 2004). Fishing is undertaken using traditional vessels and traditional gears that include basket traps, hand lines and hand spears, as well as more modern gear types such as spear guns, gill nets and beach seines (Obura *et al.*, 2002). Fishers are each associated with a landing site at which catches are landed (figure 1). Each landing site in turn, is loosely associated with an adjacent fishing ground where fishers from a landing site fish. The fishing ground is generally considered as the area of sea in front of the landing site. It has been noted that boundaries between fishing grounds are maintained by the daily interaction of fishers (McClanahan & Mangi, 2001). However, fishers fish various recognizable fishing sites within the fishing grounds and many of these fishing sites are shared among fishers from adjacent landing sites (Obura, pers. comm.). Fishers from a landing site generally accommodate fishers from adjacent landing sites fishing in their fishing ground. Thus, there aren't strong tenure arrangements for fishing grounds at least between closely associated and adjacent landing sites.

State of Fish Stocks

The reef fisheries in Diani-Chale are considered to be heavily exploited and are fished at or above sustainable levels (McClanahan *et al.*, 1997; Obura *et al.*, 2002). Catch per unit effort (CPUE) for most species fished between the years 1995-1999 was reported to be declining (McClanahan & Mangi, 2001). The average catch per fisher per day is considered very low and varies between 2 and 6 kg·fisher^{-1}·day^{-1} depending on the season (Obura *et al.*, 2002). The coral reefs of Diani-Chale are also considered to be one of the most degraded, having low coral cover, low fish abundance and a high abundance of sea urchins resulting from overfishing (McClanahan & Muthiga, 1988; McClanahan & Obura, 1995). On Kenyan fringing reefs, fisher densities of between 8–9 fishers·km^{-2} have been estimated to induce an ecological shift from herbivorous fish dominated reef communities to sea urchin dominated communities (McClanahan, 1990).

Fishery Management Measures

The Fisheries Department is the primary agency responsible for fishery management and development in Kenya. The *Fisheries Act* (GOK, 1991; GOK, 2001) contains regulations that include measures for licensing fishers, restricting destructive gear types and protection of breeding areas. In Diani-Chale, interventions by the Fisheries Department have focused on restricting gear types by prohibiting the use of beach seining and spear guns. The lack of enforcement capacity within the Fisheries Department however, has limited the effectiveness of these interventions. While there has been widespread local resistance to beach seining by local fishers using traditional gear types, beach seiners have managed to carry on fishing in some areas and it is claimed that officials have been convinced to ignore their activities (Glaesel, 2000a). Beach seining utilizes fine mesh nets and has been associated with damaging bottom habitats, capturing juvenile fish and competing with other traditional gear types (Rubens, 1996; Glaesel, 2000a; McClanahan & Mangi, 2001). Spear guns, on the other hand, are cheap to construct and have become one of the most widely used gear types (Glaesel, 2000a; King, 2000; Obura *et al.*, 2002).

Local and Traditional Fishing Arrangements

Fishers in Diani-Chale have historically had various forms of traditional and customary practices that served in some ways as fishery management measures (McClanahan *et al.*, 1997; Glaesel, 2000b; and see Tunje, 2002). These practices were not explicitly associated with conservation concerns but rather with relieving anxieties of dangers at sea and of spirits directing fish away from the community (Glaesel, 2000b). A number of taboos such as using poison to capture fish, capturing juvenile fish, and catching more fish than could be used, are thought to have influenced the conservation of fish stocks (Glaesel, 2000b). Elders were respected for their authority and exerted a significant influence over where and how fishing was conducted on a daily basis. These elders were a central part of the social organization of a landing site and regulated access to fishing grounds by non-resident fishers who had to pay compensation in exchange for permission to fish in an area (McClanahan *et al.*, 1997).

Unfortunately, most of these measures have broken down for a variety of reasons (see Glaesel, 2000b; Tunje, 2002) and, at present, they do not appear to have a measurable impact on the conservation of fishery resources (McClanahan *et al.*, 1997). Relatively modern fishing methods of beach seining and spear guns are considered by elders to be unacceptable to ancestral sprits (Glaesel, 2000a). While elders have discouraged their use, younger fishers still make use of them causing an inter-generational conflict (Glaesel, 1997). Conflict among gear types has been most notable between beach seine users and other gear type users, and between younger fishers using spear guns and older fishers using traps (Obura *et al.*, 2002). A number of landing sites have sought to ban the use of beach seines in their waters through passive means and by disallowing the landing of fish caught by this gear at their landing sites but have only had moderate success (McClanahan *et al.*, 1997). In essence, traditional authority at a landing site can only control whether fish are landed at the site, but cannot control who fishes on the reefs in front of the landing site.

PROSPECTS FOR LOCAL LEVEL FISHERIES MANAGEMENT IN DIANI-CHALE

The continuing decline of fishery resources in Diani-Chale does beg the question as to who really exercises the right to manage the fishery resources. In the absence of a strong government capacity for regulation, weakened local institutions and traditional measures, there is a general absence in the ability to exert control and formally or informally define rules among fishers for the use of fishery resources. When those participating in resource extraction undermine rules without incurring a penalty, then the use of resources can be considered unregulated and open access. This is arguably the case in Diani-Chale, and those concerned with promoting local level management need to be concerned with strengthening or creating local institutions that can govern the use of fishery resources.

There are two sets of questions that are central to the development of local level fisheries management in Diani-Chale that are of importance to ask. First, does the government, primarily the Fisheries Department, recognise the need for involving fishers in the management of resources and hence the need for a more collaborative approach to fishery management (co-management)? Is there a genuine willingness to create and use local institutions and to share responsibility with the very people whom the Department has sought to regulate? Does current policy enable the development of a more locally-focused system of fisheries management? Second, are fishers and/or others at the local level capable of playing a role in shared management? Are fishers motivated and sufficiently organised to begin to undertake management functions and does their socio-economic condition allow them to participate in management actions?

These sets of questions are central to those concerned with local level management and the paradigm of fisheries co-management. While it is not in the scope of this paper to provide a detailed review of fisheries co-management, the issues discussed in the following sections describing the local conditions in Diani-Chale are examined with regard to fisheries co-management and interactions between different institutions in a co-management framework.

Government/Policy Environment

The guiding legislation for fisheries management in Kenya is the Fisheries Act (GOK, 1991) and its various subsidiary regulations (GOK, 2001). Current legislation vests significant authority in the Director of Fisheries for fisheries management and is heavy on various regulatory measures. Current legislation does not accord any role to local communities or groups to undertake tasks relating to fisheries management. The references made to fishers all relate to regulations associated with harvesting and trading fish. A policy that would enable co-management should seek to empower local institutions through a delegation of some resource management functions to local authorities, community groups or other capable organizations over time. As has been noted elsewhere, the government has the *de facto* control over fishery policy and therefore holds the power to determine the type of co-management arrangement that is developed (Pomeroy & Berkes, 1997; Charles, 2001).

Local Institutions for Fishery Management

A number of informal arrangements carried over from historical times, although weakened, still persist and play a role in day-to-day management of the fishery. For instance, many landing site leaders still hold authority to allow or disallow fishers to use a landing site. It is traditional practice for fishers to seek permission before fishing in an area that is considered the fishing ground of another landing site. However, there is no recourse fishers can take with those who have been disallowed from landing fish at a landing site from fishing in their fishing ground. In essence, there is no secure claim over the tenure of the fishing ground by the landing site. There are 12 landing sites and adjacent fishing grounds in Diani-Chale, with various degrees of overlap in the use of fishing sites between them. Most have some form of traditional authority or beach/landing site chairman present,

Figure 2. The landing site where fishers interact, and the fishing ground in front of it where fishers fish, collectively form a socio-ecological unit at which many local fishing arrangements can be negotiated.

but the authority of these chairmen and respect among fishers varies.

These collections of landing sites and the associated fishing grounds form an identifiable socio-ecological unit, both in terms of their membership and the spatial area over which they occur (figure 2). Strengthening these units can serve to address many fishery management related issues. This is the lowest level at which one can conceive the negotiation of fishery management measures since it collects together all those who utilize a fishing ground. Within the current regime of fisheries management, there is no apparent recognition of the role that such local entities can play in day-to-day management of fisheries.

Strengthening Local Institutions

The use of existing social organizations and the revitalizing of traditional forms of organization have been noted to be appropriate ways in which to initiate and engage into fisheries co-management (Pomeroy, 1995; Pomeroy & Carlos, 1997). Nonetheless, existing social organizations may in some cases not be appropriately structured or sufficiently represent local interests. Under a traditional landing site organization, the authority is vested in an elder or elected leader who reaches decisions that fishers were expected to abide by. With traditional authority and leadership being eroded and undermined by migrant and younger fishers, it is doubtful whether traditional landing site organizations can effectively function under an authority-based system. The fishers are clearly not a homogeneous group and some, despite sharing kinship, are divided across age and gear barriers, whereas others are divided by origin (Glaesel, 2000a). Any landing site organization is going to have to contend with these differences and seek to resolve the conflicts that occur. Effective conflict resolution would be crucial to the functioning of any local landing site institution. Landing site institutions that are more representative in their authority structure may be better suited to this. Gear conflicts among fishers appear to be the most intense and persistent issue and one amenable to be addressed at the level of a landing site through arrangements among fishers.

Local Information-Decision Making Loops

A challenge to any form of fisheries management is how to use information about the fishery and changing conditions to flexibly manage fishing activity on a day-to-day basis such that it ideally results in the conservation of fish stocks. Local knowledge and information from fishers could be used to guide how and where fishing activity occurs. Such information-knowledge-management loops would be essential to the local landing site institutions. Mechanisms for monitoring the fishery on a day-to-day basis as well as over the long term by the fishers themselves are developing through participatory monitoring initiatives of CORDIO East Africa. Nonetheless, fishers in Diani-Chale have not embraced the use of this information for decision-making on their fishing activities. This may be due, in part, to the fact that fishers and their landing site do not feel responsible for managing their own fishing activities and do not see this as part of their role. This is an area that would require awareness and a formal recognition of the responsibilities that fishers have to play in resource management.

In addition, the Fisheries Department and other organizations have a continued role to play with regards to undertaking research and creating awareness among local institutions. There is also a need to better target the results of their monitoring and research to local levels and strengthen the links between monitoring and management. Currently, data on fish catch is collected by at least four different organisations that also conduct research in the area. They include the Fisheries Department, Kenya Marine and Fisheries Research Institute (KMFRI), Wildlife Conservation Society/Coral Reef Conservation Project (WCS/CRCP) and CORDIO. The changes in fishery policy and management resulting from research and monitoring activities are not apparent.

Clarifying Resource Rights and Tenure

The success of any management measures negotiated by fishers and the continued investment of effort by fishers in the process of creating and adjudicating rules can only be expected if fishers have an incentive to do so. Securing the benefits of management interventions (i.e. healthier fish stocks) can only occur if the tenure of fishing grounds is secure and access to them can be regulated and allocated to the membership of fishers utilizing that fishing ground. The recognition of the tenure of the fishing grounds by government is critical in order to exclude outsiders and those not abiding by local rules. In particular, the ability to sanction and prosecute violators from outside the community with government support has been attributed to securing tenure and the revitalization of many local level institutions (Ostrom, 1990; Johannes, 2002).

In order for the management interventions and actions of fishers to be fully effective, the government has an enabling role to play. In particular, the government needs to recognize the tenure of the fishing grounds and clarify who holds access rights to them. A few landing sites in the area have attempted to exclude beach seines using a variety of measures, resulting in mixed success (McClanahan *et al.*, 1997). The lack of follow up support and enforcement from the Fisheries Department has limited the effectiveness of this measure. Legal prosecution of outsiders who violate or break local rules for fishing would do a lot to make measures at the landing site/fishing ground more effective. Without this sanctioning of violators the benefits of management measures implemented will be diluted and, at the very worst, may lead to similar rule breaking within fishers of the landing site. Support for local enforcement units from amongst the community with government assistance should be considered.

CURRENT AND EMERGING INITIATIVES WITH IMPLICATIONS ON FISHERIES MANAGEMENT

Diani-Chale Management Trust (DCMT)

A recent development has been the formation of the Diani-Chale Management Trust (DCMT). This is an umbrella group of representatives from local community-based organisations, community leaders, local administration and government agencies working in Diani-Chale. It consists of a membership of 33 people, a number of sub-committees and a five-member executive drawn from the membership. The DCMT is perceived to be the body through which community development initiatives are to be developed and the primary body through which Integrated Coastal Area Management (ICAM) activities and issues will be addressed and resolved. This forum represents a local institution with an emerging mandate to address resource management and development issues. While there are representatives from fishers groups on the DCMT, there have also been recent efforts to organize local landing site chairmen into a group that can report and bring forward issues requiring intervention. Participatory fisheries monitoring work that was being undertaken by CORDIO has now been included under the DCMT and its sub-committee on research and monitoring and represents an effort to integrate activities that can support fisheries management at a more local scale.

While the exact manner in which the DCMT engages fisheries management still needs to be articulated by its membership, it operates at the appropriate scale and has the critical mass to engage in a number of measures that could make fisheries co-management operational. The DCMT, together with the landing site chairmen and Fisheries Department represented within it, could be the starting point from which a fishery management plan could develop. Its initial actions could target issues on which there is broad agreement amongst fishers. The development of a community enforcement unit that initially targets beach seiners could be a step in this direction. Another could be the facilitation of the development of a vision and future direction of the fishery that can be used as a basis for further activities.

Immediate challenges with the DCMT concern its capacity to undertake activities and to sustain itself and develop into a stable community institution. Fisheries are only one of the resources that the DCMT is concerned with and the efforts of the DCMT are going to have to be distributed between other resources and competing mandates for enterprise development.

Fisheries Development Initiatives

The Coast Development Authority (CDA), a governmental development agency, and community development non-governmental organisations (NGOs), such as Eco-Ethics International Union Kenya chapter (EEIU) and PACT Kenya, have been involved in renovating and building landing site structures and encouraging enterprise development. Of notable mention is the ongoing acquisition of fishing vessels and gears for the offshore fishery for some fishers. While the development of an offshore fishery presents an opportunity to reduce fishing pressure inshore, it would not be prudent to develop it without any arrangements or institutions in place that can regulate its use. Integrating fishery development measures with other aspects of fishery management is essential. In the inshore where there is already excessive effort, development measures need to focus on increasing its value rather than catches. Government supported, co-operative societies that should have been active in fisheries development in the area have largely failed and have been abandoned by fishers (King, 2000) and fishermen have formed self-help groups to provide their own small credit facility to members.

BARRIERS TO LOCAL LEVEL FISHERIES MANAGEMENT IN DIANI-CHALE

The socio-economic conditions of fishers and several issues that currently affect them pose a number of barriers to attaining a more local level fisheries management. For fishers, livelihood security considerations would be expected to feature prominently in any actions and management measures they choose to engage in. The current socio-economic condition of fishers and the low level of catch, pose a grave challenge to their participation in measures that are perceived to reduce their catches in the short term. In a fishery that is characterized by excessive effort, the improvement of fish stocks is not likely without the relief of excessive fishing pressure. This presents the greatest challenge to any form of management of the fishery.

In addition, the continuing loss of lands on which landing sites and their access routes are located has been the primary issue that has preoccupied fishers. With the advent of tourism development in the past few decades, community owned lands have largely fallen into private hands and most of the 12 landing sites in Diani-Chale and the access routes to them now sit on privately owned land. Fishers need some security of existence through arrangements that can guarantee their continued use of the landing sites, beaches and access to them. Fishers have cited the lack of such security as the primary issue that needs to be addressed before they can engage in discussions relating to other aspects of fisheries management. The DCMT is perhaps the best suited institution to take the lead and facilitate the collective negotiation any such arrangements.

A third issue that is historical in nature yields from the failed efforts of the Kenya Wildlife Service (KWS) to institute a marine reserve in the area. The community rejection of the proposal in 1994 and the events surrounding it have left a strong suspicion that such a reserve may be still be imposed through other means. As a consequence, discussions on any aspect of management have been treated with suspicion and often rejection for fear of it being associated with KWS. This has effectively limited the progress of any management initiatives that government agencies and other external agents have tried to develop with the communities. The current sentiments of the community make any intervention by the KWS a virtual impossibility. Nonetheless, in legal terms, the KWS still holds the mandate to manage the area. A productive intervention of KWS could be to publicly disengage itself from the management of the area. The official and formal delegation of authority from KWS to a local authority, such as the Kwale County Council, another government agency or perhaps even the DCMT, could be a step in this direction.

FISHERIES MANAGEMENT IN DIANI-CHALE: MULTIPLE PLAYERS AND LINKS

This paper has touched on several key players relevant to fisheries management in Diani-Chale (table 1). It has been noted that co-management can be considered as a mechanism to bridge institutions at multiple scales to improve the management of natural resources (Berkes, 2002). Thus, one may expect that the internal capacity of these individual institutions and the links and interactions between them will determine how effective fisheries co-management will be. At present, there does not appear to be a coordinated approach among players involved. It is important that the players at upper levels intersect and coordinate their efforts in a manner that is consistent with the development of local level management. Misguided policy and actions can erode social trust between local level organisations and agencies working at higher levels. The history in Diani-Chale between the community and others has always been one of suspicion and scepticism compounded by the failed in-

Table 1. Multiple institutions and scales pertinent to fisheries management in Diani-Chale.

	Institutions | Players		
Scale | Level	***Government Institutions***	***Community Institutions***	***Other Institutions***
Country	National Government; Fisheries Department ↓		
Coast Province	Provincial Administration; Coast Development Authority (CDA) ↓		Research & Conservation NGO's ↕
Kwale District	Local Administration ↓	South Coast Fisherman's Association (SCOFEC)	
Divisions, Locations			Diani Chale Management Trust (DCMT)
Village		Village Organizations	Other Stakeholders
Landing Site		Landing Site Organizations	Community Development NGOs
Fishing Ground		Fishers Self-Help Groups (Fishers + Traders)	

stitution of a marine reserve in the area in 1994. Frustration among fishers arising from the lack of fishery development measures or measures that were improperly implemented and, as a result, were detrimental or yielded insufficient benefits to them is already evident.

CONCLUSIONS AND RECOMMENDATIONS

At present, coral reef fisheries management in Diani-Chale is characterised by a combination of limited capacity for government based management and the lack of sufficiently developed and empowered local institutions that can play an effective role in fisheries management. Government policy is currently inadequate and not conducive to the development of more local level management. Specifically, it needs to recognise the landing site and adjacent fishing grounds as a socio-ecological unit and the role that these can play in fishery management. These units need to be empowered to be able to make decisions on the basis of information and local knowledge and accorded a formal role to play in fisheries management. Encouraging the development of arrangements among fishers, however simple they may be, may be an initial step towards according greater management responsibility to such local institutions. The Government needs to further clarify the tenure of fishing grounds and provide support to the actions of fishers that restrict access to fishing grounds to outsiders and those who have violated local regulations. Several barriers stand in the way of achieving a more local level fisheries management and the ability of fishers to fully participate in such management. These include the socio-economic condition of fishers, the unresolved loss of landing sites and access routes and the continued suspicion by fishers of interventions from the KWS. Addressing these issues can remove some of the current obstacles that hinder the committed participation of fishers in management. A coordinated approach involving all the players relevant to fisheries management is necessary and its absence may result in misguided actions that may further erode the trust between players at local and upper levels. The DCMT could effectively make fishery co-management in the area operational and needs to be supported to that end.

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A Diver and Diving Survey in Southern Mozambique

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ABSTRACT

Recreational SCUBA diving has grown tremendously along most of the southern Mozambican coastline in the last eight years. This growth was not managed, largely due to a lack of baseline information on various aspects of the industry, including diving intensity and socio-demography of the SCUBA diving community. These aspects of the industry were thus covered in this study to redress this shortfall. Information was collected on divers and diving pressure in southern Mozambique using questionnaires and dive log sheets distributed through local dive centres. Divers visiting southern Mozambique were found to be mostly educated South Africans with high levels of experience. They were committed divers, satisfied with their diving experiences in the area and sensitive to reef conservation issues. The diving intensity was estimated at 42 500 dives in 2001 and 62 000 dives in 2002 occuring at about 20 dive sites, which is not as high as previously thought. Nevertheless, there is potential for overuse of favoured dive sites, and recommendations are made to disperse divers among a greater number of sites and to undertake awareness programmes on sustainable diving practises.

INTRODUCTION

The southern Mozambique coastline between Ponta do Ouro and Cabo Santa Maria has been a focus of coastal tourism development since the end of the civil war in 1992 (Hatton, 1995; Massinga & Hatton, 1996). Around 115 000 tourists visit southern Mozambique annually (Saia, A., 2003, Ministry of Tourism, pers. comm.). Of these, approximately 10 000–13 000 visit the Ponta do Ouro and Ponta Malongane region (Bjerner & Johansson, 2001) to dive, fish and camp, the majority (60–72%) being certified SCUBA divers (Bjerner & Johansson, 2001; Abrantes & Pereira, 2003).

The high fish diversity in the area (Robertson *et al.*, 1996; Pereira *et al.*, 2004) contributes to the beauty and attractiveness of the reefs. Further, the occurrence of large, resident fishes such as potato bass (*Epinephelus tukula*), several species of sharks and marine turtles has resulted in specific localities such as 'Bass City' and 'Pinnacles' near Ponta Malongane becoming popular with divers (Robertson *et al.*, 1996). The diving intensity in 1995 was estimated at 30 000–40 000 dives per year and Robertson *et al.* (1996) stated that this dive rate was high considering the size of the reefs. Unconfirmed reports claimed that in 1998 this number increased to around 80 000–90 000 dives per year and it was suggested that the diving intensity was approaching unsustainable levels (H. Motta, WWF Mozambique, pers. comm.). Later, Bjerner & Johansson (2001), estimated the diving intensity in the area to be approximately 63 000 dives per year.

Despite the remarkable growth and economic importance of recreational diving in southern Mozambique (Bjerner & Johansson, 2001), its regulation and management are deficient and the existing legislation is obsolete (dating back to the late 1960s) and poorly enforced. Important demographic and socio-economic information on divers visiting the southern Mozambique reefs is generally lacking and management actions have been partially hindered by the lack of knowledge on the current diving pressure in the area. This paper reports aspects of an MSc thesis study of the demography, participation and attitudes of recreational SCUBA divers visiting southern Mozambique. Estimates of the diving pressure during the study period (February 2001–December 2002) and recommendations for management are also provided.

MATERIALS AND METHODS

An 8-page, 30-question, self-administered questionnaire was developed to collect data from recreational divers. The questionnaires were distributed to dive centres operating in Ponta do Ouro and Ponta Malongane and collected throughout the study period. The questionnaire collected information in four areas:

- demographic profile of recreational scuba divers;
- diver experience, activities and qualifications;
- diving activities and experiences in southern Mozambique;
- opinions on the condition and management of diving in southern Mozambique.

The number of recreational dives done during the study period at individual dive sites was estimated using two methods:

- pre-prepared log sheets filled out by dive centres;
- boat launch data extracted from resort log sheets and dive centres.

RESULTS

Diver experience

A total of 108 questionnaires were filled in and returned. Most (57.9%) of the recreational SCUBA divers that answered and returned the questionnaires were South African males and only three were Mozambicans. A total of 9 nationalities were represented, most of them European. Altogether, the average age was 34.9 years (S.D. = 8.8), with most (73%) between 21 and 39 years of age. Divers were highly educated; all divers had completed their secondary level of education (high school) and the majority (36.4% of females and 50.0% of male divers) had completed their tertiary education at university (B.A./B.Sc.). A number of female (27.3%) and male divers (21.6%) had undergone post-graduate education (B.Sc. (Hons)/M.Sc./PhD.).

On average, divers had been certified for 5.8 years, ranging from 0–27 years. The majority had been certified for at least 4 years and more than half of them had completed 51 or more dives. 35% of divers were 'newcomers' ('have dived for one year or less in southern Mozambique'), 48.5% had been visiting southern Mozambique for the past 2 to 5 years and the rest for more than 5 years. 63% stated that they have also dived at other locations in Mozambique, most notably Inhambane (31.3%) and Inhaca Island (19.3%). The majority of divers (65.1%) considered SCUBA diving to be their most important or second most important outdoor activity. Many outdoor recreational activities are primarily family-orientated, and recreational SCUBA diving in southern Mozambique does seem to be one of these activities. Most divers practised their sport frequently with friends (47.1 %) or a combination of friends and family (42.4%).

The diving in southern Mozambique compared favourably with other dive sites, as 84.2% of the divers considered the diving in southern Mozambique slightly or much better than other diving sites they have visited. The perception that southern Mozambique offers good quality diving is also reflected in the overall diving satisfaction expressed by the respondents. The great majority (93.4%, average rank of 4.5 on a scale of 1–5) stated that they were very or extremely satisfied with their diving experience in southern Mozambique, with none of them dissatisfied or slightly satisfied.

Diver Attitudes

The great majority of divers (91.6%) rated 'look at fish and other marine life' as a very important or extremely important reason why they came to dive in southern Mozambique. This was the highest ranked reason (4.7 on a scale of 1–5). Another important reason, with an average rank of 4.4 (83.7% of the divers), was 'to experience unpolluted surroundings'. Marine megafauna were the highest ranked attraction, especially dolphins, whales and whale shark. Following these, divers valued tropical reef fish (i.e. damsels – Pomacentridae, angels – Pomacanthidae and butterflyfishes – Chaetodontidae; 77.2%), other reef fish (74.2%) and small reef fish (72.9%) at higher levels (average rank of 4.1) compared with hard and soft corals, and most of the other reef fish categories (average rank of 4.0 by 70.9% respondents). Furthermore, 10.7% of the divers considered hard and soft corals to be unimportant or slightly important as opposed to only 7% for large reef fish or 6.9 % for small reef fish, which declared that these reef fish categories were unimportant for their diving experience.

The questions on the condition and management of the reefs was aimed at more experienced divers that have dived in southern Mozambique before 1999 (divers that potentially witnessed the effects of the 1998 bleaching event; Schleyer *et al.*, 1999). More than half of these (55.3%) stated that the reefs appeared the same and no changes were noted in the reef environment. Most divers also considered that the coral cover and the abundance of small reef fish had not changed. When asked about the abundance of large reef fish (rock cods, kingfishes), responses were mixed; a large proportion (44.4%) considered that there were less rockcod and kingfish, while 36.1% felt that the abundance of these fish had not changed. Those that noted changes in the overall reef environment were more experienced divers (highest level of diving qualification and mean number of logged dives). They also declared that both the coral cover and abundance of small reef fishes had remained the same, but they were unsure if the large reef fishes had decreased in abundance.

Attitudes towards reef conservation and the management of diving activities varied according to the nature and context of statements in the questionnaire (table 1). For example, the majority of respondents disagreed (average rank 2.3 on a 1–5 scale; 1 being strong disagreement) with the deployment of mooring buoys or with the idea that there is excessive diving in southern Mozambique (54.5% of the divers strongly disagreed or disagreed with the statement 'The reefs in southern Mozambique are too crowded'). Most divers agreed that excessive diving might damage reef communities (64.5%) and that a pre-dive briefing should emphasize environmentally friendly

Table 1. Percent agreement or disagreement of recreational SCUBA divers visiting southern Mozambican reefs, with attitude statements concerning management of the reefs. 1=Strongly disagree; 2=disagree; 3=neutral; 4=agree; 5=Strongly agree. (3Ts = don't Touch; don't Tease; don't Take).

	1	2	3	4	5	Mean rank
Emphasize 3Ts on pre-dive briefings	2.0	0.0	2.0	17.2	78.8	4.7
Can be damaged by excessive diving	5.1	10.2	19.4	27.6	37.8	3.8
Designated for specific uses	8.2	14.4	16.5	23.7	37.1	3.7
Artificial reefs should be deployed	19.4	7.1	20.4	27.6	25.5	3.3
Number of dives should be restricted	10.1	18.2	29.3	22.2	20.2	3.2
Mooring buoys should be provided	34.4	15.6	26.0	7.3	16.7	2.6
The reefs are too crowded	22.2	32.3	36.4	7.1	2.0	2.3

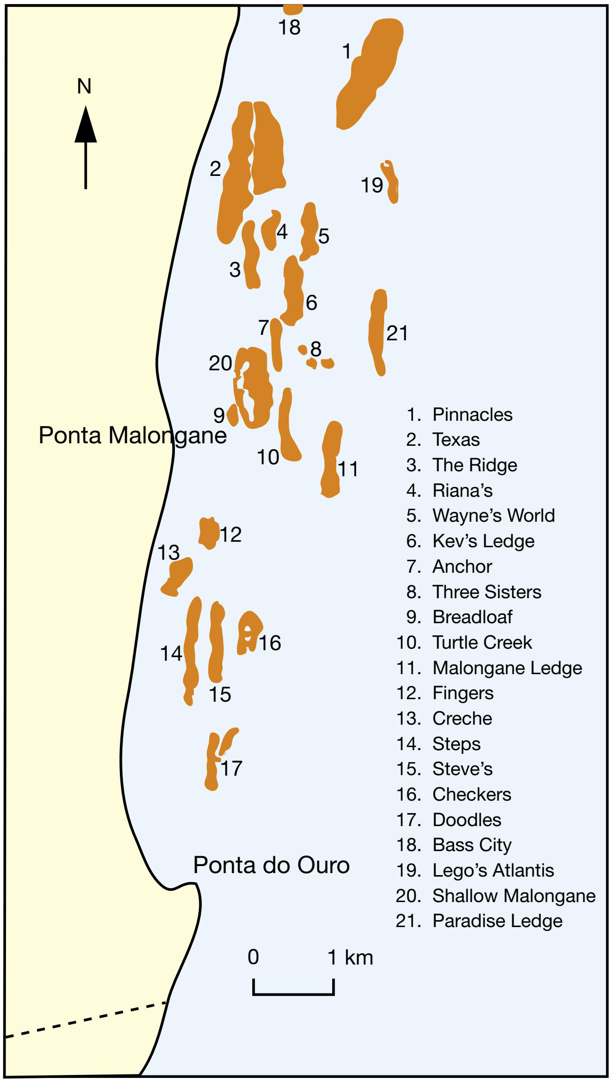

Figure 1. Schematic map of the most frequently dived reefs in southern Mozambique.

diving practices (96%), emphasising the 3T's ('do not Touch, do not Tease, and do not Take'). A clear-cut picture could not be drawn from responses regarding two management statements on restrictions on the number of dives per site and the deployment of artificial reefs, thus suggesting that these were not very popular among the respondents.

Diving Pressure

A total of 5 dive centres operated continuously throughout the study period (February 2001 to December 2002). Four of them were based in Ponta do Ouro and the fifth in Ponta Malongane. Two new operations initiated their activities in April (one based in Ponta do Ouro) and May 2002 (one based in Ponta Mamoli). The total number of dives for 2001 and 2002 was estimated at 104 500, with considerably more dives being executed in 2002 (62 000) when compared with 2001 (42 500).

A total of 1 526 launches (13 661 dives) were logged during the study period. Of the 23 reefs dived (see figure 1), 4 (Doodle, Creche, Kev's Ledge and Texas) were used the most, hosting more than 44% of the dives (table 2 on next page). This means that each of these 4 reefs was dived more than 7 000 times during the study period (February 2001 to December 2002) with over 12 000 dives at Doodles the most popular reef. Diving intensities are not presented for the reefs at Ponta Techobanine as they were not commercially dived for most of the study period due to the great distance from the dive centres at Ponta do Ouro and Ponta Malongane.

DISCUSSION

Virtually all the divers on the southern Mozambican reefs were South Africans. This is not surprising as the great majority of tourists (>95%) visiting the area originate from South Africa (Bjerner & Johansson, 2001; Abrantes & Pereira, 2003). Surprising, however, was the fact that a minimal number of Mozambique nationals participate in this recreational activity, probably due to levels of interest, and high cost.

Diving qualifications and experience were of a high standard and comparable to those previously found at Ponta do Ouro (Bjerner & Johansson, 2001) and in Texas, USA, (Ditton & Baker, 1999; Thailing & Ditton, 2001) but higher than those reported in Zanzibar and Mombasa (Westmacott *et al.*, 2000a) and Australia (Roberts & Harriot, 1995; Rouphael & Inglis, 1995; Harriot *et al.*, 1997). Divers were also committed to their sport, consid-

Table 2. Number of recreational dives conducted on southern Mozambican reefs during the study period. NA=refers to launches in which the number of divers was recorded but not the reef.

Reef	Dives logged	%	Estimated dives 2001	Estimated dives 2002	Total	Rank
Doodles	2 706	19.81	8 419	12 282	20 700	1
Creche	1 414	10.35	4 399	6 417	10 816	2
Kev's Ledge	980	7.17	3 047	4 445	7 497	3
Bass City	937	6.86	2 916	4 253	7 168	4
Texas	928	6.79	2 886	4 210	7 099	5
NA	818	5.99	2 546	3 714	6 257	6
Steps	817	5.98	2 542	3 708	6 250	7
Three Sisters	751	5.50	2 338	3 410	5 745	8
Anchor	699	5.12	2 176	3 174	5 347	9
Paradise Ledge	628	4.60	1 955	2 852	4 804	10
The Ridge	520	3.81	1 619	2 362	3 978	11
Breadloaf	461	3.37	1 432	2 089	3 526	12
Shallow Malongane	454	3.32	1 411	2 058	3 473	13
Checkers	419	3.07	1 305	1 903	3 205	14
Pinnacles	403	2.95	1 254	1 829	3 083	15
Malongane Ledge	244	1.79	761	1 110	1 866	16
Lego's Atlantis	215	1.57	667	973	1 645	17
Aquarium	92	0.67	285	415	704	18
Riana's Arch	88	0.64	272	397	673	19
Wayne's World	26	0.19	81	118	199	20
Padi	20	0.15	64	93	153	21
Fingers	15	0.11	47	68	115	22
Turtle Creek	14	0.10	43	62	107	23
Steve's	12	0.09	38	56	92	24
Total	13 661	100.00	42 500	62 000	104 500	

ering it an important outdoor activity. It might be argued that only experienced and more conscientious divers responded to the questionnaire, but similar proportions of experienced and novice divers responded to the questionnaire, suggesting that the survey data is representative of the recreational diving population of southern Mozambique. Diving experience in reef users is an important asset as far as the management and conservation of coral reefs is concerned. There is evidence that novice divers (<100 logged dives) cause more physical damage to corals than more experienced and conscientious ones (>100 logged dives; Davis *et al.*, 1995; Bjerner & Johansson, 2001).

'To look at fish and other marine life' was identified as the most important reason why divers chose to dive in southern Mozambique, with the most popular attractions being marine mammals (dolphins and whales), cartilaginous fish (sharks and rays) and marine turtles. These species enjoy worldwide popularity, being flag species for a number of marine conservation campaigns. Divers prefer reef fish (large or small tropical reef fish) when compared with benthic species (e.g. corals, sponges). Divers interviewed in Zanzibar and Mombasa by Westmacott *et al.* (2000a) and in the Caribbean by Williams & Polunin (2000) also regarded the variety and abundance of fish as the most important reef feature. This may be an impor-

Figure 2. Divers preparing to launch through the surf at Ponta Malongane, southern Mozambique. *Photo:* MARCOS A. M. PEREIRA.

Figure 3. Divers returning from a dive at Ponta do Ouro, southern Mozambique. *Photo:* ARTHUR FERREIRA/ÁFRICA IMAGENS.

tant issue to consider if diving pressure and fishing restrictions or zoning schemes are to be implemented.

Southern Mozambique seems to attract a loyal diver clientele. The percentage of respondents that have been visiting these reefs for more than four years totalled almost 40%. Despite the fact that tourism, and SCUBA diving in particular, were badly affected by the February 2000 floods in southern Mozambique, new divers are still attracted to this destination, with divers being 'very or extremely satisfied' with their diving experience.

This study also assessed divers' perceptions on reef condition and changes in coral cover and fish abundance, which can be used, with suitable care, as indicators of changes in reef condition and community structure (e.g. Hodgson, 1999; Uwate & Al-Meshkhas, 1999; Seaman *et al.*, 2003). Another important issue is that divers' personal perceptions as to whether reefs are changing or undergoing degradation may have an influence on the local economy. For example, Westmacott *et al.* (2000a, b) reported that coral bleaching influenced the choice of destination in 39% of the instances of divers visiting Mombasa who were aware of the 1998 bleaching event. They also noted that coral bleaching affected tourists' holiday satisfaction (with 47% of them considering dead corals the most disappointing experience), thus causing financial losses to the economies of Sri Lanka and the Maldives. Graham *et al.* (2001) recorded similar results in Palau, Micronesia.

In the present study, divers were in agreement regarding changes in the overall reef environment, with the majority of them noting no changes from before 1999 to the present. This concurs with findings that reefs in southern Mozambique were less impacted by coral bleaching in 1998 compared to reefs farther north (Schleyer *et al.* 1999; Motta *et al.*, 2000) and are likely to remain less vulnerable to climate change in the near future. It is quite worrying that the majority of divers (44%) stated that the large reef fishes (rock cods, kingfishes) had decreased since 1999. This could be attributed to fishing activity in the area, as bottom fishing or over-fishing on the reefs is widespread and illegal industrial vessels have frequently been seen around the reefs.

Some of the divers' attitudes towards the management of SCUBA diving and reef conservation in southern Mozambique were similar to those of divers in Texas (Ditton & Baker, 1999) and Bonaire Marine Park (Dixon *et al.*, 1993). They disagreed with the statement that the reefs were too crowded and agreed that the reefs should be designated for specific uses. The deployment of mooring buoys and artificial reefs (sunken ships) were not popular with divers in southern Mozambique, nor are they in South Africa, where rough sea conditions and relatively high costs preclude these interventions. Divers strongly agreed with the 3Ts (do not Touch, do not Tease and do not Take), suggesting they would accept and welcome awareness campaigns and pre-dive briefings (Medio *et al.*, 1997) on environmentally-friendly diving practices.

Previous estimates on the number of dives in southern Mozambique have varied greatly, from 30 000–40 000 in 1996 (Robertson *et al.*, 1996) to 80 000–90 000 in 1999 (H. Motta, WWF Mozambique, pers. comm.) and 50 000–63 000 dives in 2001 (Bjerner & Johansson, 2001). In the present study, the diving pressure was estimated at 42 500 dives in 2001 and 62 000 dives in 2002. The low number registered in 2001 is indicative of the impact of massive floods that occurred throughout southern Mozambique in February 2000, causing widespread destruction in the basic infrastructure with consequent bad publicity that resulted in a decline in tourism. In 2002, better marketing and a decline in the value of the South African Rand, causing more South Africans divers to dive 'locally' rather than travel overseas for their diving vacations, resulted in higher numbers. It thus appears that the diving pressure in southern Mozambique is not excessive, however it may have have reached its 'carrying capacity', as the present tourism facilities such as accommodation, roads, electricity and medical facilities appear to be saturated.

As noted above, most diving takes place on about 20 reefs with more than 50% of the dives being concentrated on 5 of them (see table 2). In nearby Sodwana Bay, South Africa, 85% of more than 100 000 dives per year (Schleyer, M., 2004, pers. comm.) are carried out on Two-mile Reef, the closest of four reefs used by divers, where significant diver damage has been noted (Schleyer & Tomalin, 2000). Reef features such as the abundance and diversity of fish, distance from the shoreline and depth are some of the most important factors in the selection of dive sites by both dive operators and divers, focusing most pressure on a small number of sites.

Interestingly, in spite of the high pressure at favoured sites, evidence of damage on similar reefs in South Africa, and the high levels of experience and awareness in the diver community, divers did not accept that southern Mozambican reefs are too crowded. As a result, they were reluctant to endorse management interventions to limit crowding and damage, such as placement of mooring buoys and artificial reefs, but were amenable to raising awareness about more environmentally friendly diving practices. The results of this study suggest there is a need to alleviate the diving pressure on the southern Mozambican reefs through a more balanced distribution of the diving intensity, better awareness programmes on sustainable diving practices and perhaps the deployment of artificial reefs (van Treeck & Schuhmacher, 1999; Wilhelmsson *et al.*, 1998).

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