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Post Tsunami Status of Coral Reef and Fish in Northern Aceh


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ABSTRACT

The coral reefs of northern Aceh, located in western Indonesia, are productive marine ecosystems that are important for the economies of local communities. The catastrophic tsunami in December 2004 affected local communities, and ways in which they utilized marine resources, while impacts on reef resources were patchy. Limited data and information on coral reef condition prior to the tsunami has highlighted the need for regular long-term coral reef monitoring to assess reef recovery from the tsunami and from previous damage caused by destructive fishing and overfishing. The objectives of this study were to provide reliable data and information on scleractinian corals and reef fish in the northern Aceh region of Weh and Aceh Islands. Using line intercept transects (LIT) and underwater visual census techniques at 21 sites, we found that the mean coral cover in Weh Island was significantly higher (30.0% - fair condition) compared to Aceh Island (10.8% - poor condition). Coral reefs at Rubiah Island, Sumur Tiga and Benteng that were protected under the management of Panglima Laot of Sabang were in the best condition. On Weh Island reef fish abundance (32,505 ha⁻¹) and biomass (748 kg.ha⁻¹) were also significantly higher than on Aceh Island (9539 ha⁻¹ and 396 kg.ha⁻¹, respectively). Pomacentridae (damsel fish) had the highest abundance and biomass among fish families. Macro-invertebrates numbers, in particular sea urchins, were highest on Aceh compared to Weh Island, and in open access areas compared to marine managed areas. The potential for sea urchins to influence coral recruitment and coral reef recovery of Aceh Islands requires further investigation.

INTRODUCTION

Northern Aceh on the Indonesian Island of Sumatra and the surrounding reefs in the Andaman Sea are well known for their extensive shallow reef flats that extend 200–500 m from the shore. The reefs are dominated by massive species (mainly Poritidae and Faviidae) intermingled by patches of branching Acropora and Montipora in sheltered areas, and mostly branching species at high-energy reefs (Brown 2005). Weh and Aceh Islands are the two main islands in northern Aceh, and also the westernmost reef areas in the Indonesian archipelago. The marine fauna and flora of north-west Sumatra, including the northern Acehnese reefs, are comprised of species from the Indian Ocean and the Pacific Ocean and make the region


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biogeographically distinct from the eastern Indonesian coral reef fauna and flora.

The reefs ecosystems support a local artisanal fishery as well as a primarily pelagic commercial fishery. The area also has a tourism industry with snorkeling and SCUBA diving and other recreational activities as the main attractions. These activities contribute income to coastal communities, in addition to farming, business, and government sectors (Baird et al. 2005b). Although these reefs were subject to substantial disturbance from the tsunami, initial claims that northern Aceh reefs were destroyed or greatly impacted by the tsunami in 2004 (Brown 2005, UNEP 2005) were unfounded (Baird et al. 2005a and b). Nevertheless damage to coastal communities was severe with five of thirteen villages in Weh Islands heavily affected by the earthquake and tsunami; mostly on the northern and southern coasts (Baird et al. 2005b). Damaged houses, boats, fishing gears, and loss of life paralyzed the northern Aceh fisheries, including destructive fishing activities. However, overfishing and destructive fishing practices had caused serious damage and significantly degraded coral reef ecosystems in Aceh Islands prior to the tsunami (Baird et al. 2005a and b).

The status of reefs of Northern Aceh range from very poor to good condition, but have overall been well documented only recently (Baird et al. 2005a and b, Campbell et al. 2005, Ardiwijaya et al. 2006). Coral reefs of Weh Island are known to be generally in better condition than those on Aceh Island because of past differences in management between these two. To improve the management of these reefs and prevent damaging, unsustainable, and illegal fishing practices from returning, investment is required to implement management practices and strategies that help rehabilitate, restore and protect marine resources through programs of regular monitoring, education, coastal management and establishment and maintenance of marine protected areas. The aim of this research project was to provide reliable data and information on coral reef resources of northern Aceh that will be useful for local people, scientists, tourists, and other stakeholders to evaluate the condition of the reefs, make recommendations for management and be used to evaluate the effectiveness of future coastal management. This report focuses on overall reef condition (benthic, invertebrates and fish) while a
more detailed analysis of fish biomass and fishery implications is presented in Campbell et al. (2007).

**METHODS**

**Survey Sites**
Coral reef and invertebrate surveys were conducted in 2006 and 2007 at 21 sites, 13 on Weh Island and 8 on Aceh Island (comprising Beras and Nasi islands), while coral reef fish surveys were conducted at 29 sites, 19 on Weh and 10 on Aceh Island (Fig. 1). Sites were selected to represent the reefs of the region and the types of site management, including Marine Protected Areas (consisting of Panglima Laot and Tourism Reserve) where fishing restrictions are in place, as well as areas with unrestricted fishing or open access areas, on both islands.

**Survey Techniques**
Methods used were manta tow and Line Intercept Transect (LIT) for benthic cover, under water visual census technique (UVC) for reef fishes, and belt transect for invertebrates (English et al. 1997, Hill and Wilkinson 2004). In order to obtain representative data of the reef, transects were laid at two depths at 2-3 m (shallow) and 6-8 m (deep).

**Coral reefs**
Manta tows were used to obtain general descriptions of reef areas, estimating percent cover of hard coral, soft coral, dead coral and sand. Two divers were towed along the reef edge using a boat at a constant speed of 2 knots, with regular stops every 2 minutes to record data on substrate cover. The number of tows varied between locations depending on reef and environmental conditions. Two replicate 30 meter LIT transects were recorded at each site and depth. Reef condition was assessed using percent cover of live hard coral was based on Gomez and Yap (1998): excellent, 75-100%; good, 50-74%; fair, 25-49%; and poor, 0-24%.

**Reef fish**
Abundance and biomass of reef fishes were recorded using Underwater Visual Census, recorded along the lines used for LIT, plus one additional transect. Data were collected at the species level and 9 size classes visually estimated (0 cm, 5-10cm, 10-15 cm, 15-20 cm, 20-25 cm, 25-30 cm, 30-35 cm, 35-40 cm and >40 cm). Transect size was 5x50 m for fish >10cm and 2x50 m for fish <10cm. Data is presented as abundance, in number of individuals per hectare (ha⁻¹), as well as biomass, in kilograms per hectare (kg·ha⁻¹), estimated using standard length-weight relationships for fish species (FishBase 2000).

**Macro-invertebrates**
Macro-invertebrates were counted using 2 replicate 5x50 m² belt transects at the two depths. Invertebrates were identified to species level and data converted to numbers per unit area (ha⁻¹). Species diversity was analyzed using the Shannon-Wiener index (H') (Krebs 1989), with the following ranges used in this study: H'<=1 : low diversity; 1<H'<3 : medium diversity; H'>3 : high diversity.

**Statistical Analyses**
Using two-way nested analysis of variance (ANOVA) we examined the effect of time (fixed; 2 levels, 2006 and 2007) and management areas (fixed; 4 levels, Panglima Laot, Kawasan Wisata, open access Aceh Islands, open access Weh Islands) on mean fish biomass and mean fish abundance. In order to understand the variation in reef fish biomass and abundance between sites within management areas (among sites) and among management areas data from each transect (n = 6) at each site (random; 6 - 9 levels) were nested within each management area. Significance among factors was tested at the P<0.05 levels. Biomass and abundance of reef fish were log transformed prior to ANOVA analyses to improve homogeneity and normality. All analyses were completed using SPSS v11.5.
The health of coral reefs varied considerably within the region, with the hard coral cover (pooled for both depths) being higher on Weh Island (30.0% ± 2.4 (standard error of the mean)) than Aceh Island, (10.8% ± 4.0). Hard coral cover on Weh Island was higher on reefs with fishing restrictions, i.e. Panglima Laot (51.5% ± 10.8), and inside the Tourism Reserve (31.7% ± 2.3) compared with open access areas on Weh Islands (22.8% ± 4.8). Using Gomes and Yap (1998) categories to estimate reef condition one site managed by the Panglima Laot authority, Sumur Tiga (site no 19) was categorized as in “good condition” with a coral cover of 62.3% ± 4.4, while 8 sites were in “fair condition” (25.3-47.2%) and 12 sites in “poor condition” (3.6-22.5%, Fig. 2). Results from manta tow surveys indicate that the benthic habitats are dominated by sand and that mean live coral on Weh Islands, ranging between 12 and 52%, was higher than in Aceh Islands (12%) (Fig. 3). These results support findings from the LIT surveys that reefs inside managed areas were in better condition than those in open access areas.

The Aceh Islands suffered severe catastrophic damage from the tsunami in 2004, but many reefs were already dead or in poor condition prior to the tsunami because of a history of destructive fishing including dynamite and cyanide use (Baird et al. 2005, Campbell et al. 2005). Enormous dead colonies of coral and rubble beds covered with a thick growth of filamentous algae remain common on Aceh Island reefs. However, there was also little evidence of recent coral mortality. To the contrary, an increase in the mean coral cover in Aceh Island from 2006 (8.2% ± 1.8: Ardiwijaya et al. 2006) to 2007 (10.8% ± 2.4: this study), suggests that recruitment of corals is occurring, as has been previously suggested (Ardiwijaya et al. 2006, Fig. 4). It is possible that the reduction in fishing effort, and particularly destructive fishing, following the tsunami have allowed reefs on Aceh Island to start to recover from the misuse of the past. Nevertheless, recent reports in 2007 of the use of cyanide fishing show it is starting up again. Sediment run-off from inappropriate and poor agricultural practices also highlights the need for an ecosystem-based approach to these problems, where land use and
In 2006, reef fish abundance was highest in Sumur Tiga (133,050 ha⁻¹), while in 2007 it was highest at Benteng (83,770 ha⁻¹) (Fig. 5). Both sites are located within Panglima Laot management areas, and in an ANOVA (Table 1) both abundance and biomass of fish were significantly highest in management areas in both 2006 and 2007 (Fig. 6). After management type, the abundance and biomass of reef fishes was significantly affected by site characteristics (nested within management) shown by the lower F ratio for site as a factor (Table 1). There was a considerable decrease in fish biomass and abundance in protected areas from 2006 to 2007, while both were stable in fished areas (Fig. 6). However this did not result in a statistically significant difference in the ANOVA of management type by site interaction. There was no significant difference in reef fish biomass between 2006 and 2007 (F= 0.160 p=0.716).

Biomass of reef fishes in 2006 ranged from 118 to 2399 kg.ha⁻¹, with the highest biomass recorded at Sumur Tiga and the lowest at Pasi Janeng. In 2007, biomass of reef fishes ranged from 149 to 1562 kg.ha⁻¹, the highest recorded in Canyon and the lowest in...
Lapeng (Fig. 7). Sumur Tiga and Canyon are protected under the authority of Panglima Laot while Pasi Janeng 2 and Lapeng are open access areas on Aceh Islands.

However there were changes in the size class structure of fish from 2006 to 2007 (Table 2) and in the family composition. The number of small size fishes (5-10 cm) decreased, mainly due to a decrease in the number of Pomacentrids, in parallel with an increase in mid-sized fish (15-25 cm), mostly of groupers, snappers, and jacks. Pomacentrids are the most abundant reef fish present on these reefs and a major prey item for large carnivorous fishes, such as groupers, snappers, and jacks. This suggests that predation may be responsible for the overall decrease in fish abundance. Reduced fishing pressure following the tsunami in 2005 may have contributed to the increased numbers of large carnivorous fish. Excluding Pomacentrids, Acanthuridae comprised the largest portion of the biomass (Fig. 7), at all sites.

The cause of the decline in fish in management areas is unknown. As it was found at many sites it is possible that non-anthropogenic factors such as migration, seasonal variation or predation may be responsible to any apparent decline in fish numbers. The decrease in Pomacentrid abundance and increase in the numbers of large fishes requires further investigation.

**Macro-Invertebrates**

A high density of macro-invertebrates was found at almost all survey sites (Table 3). Sea urchins, worms, and ascidians comprised more than 86% of all invertebrates (Fig. 8). The density of shells, shrimps, cephalopods, and jellyfishes were surprisingly low. The low density of cephalopods and jellyfishes was most likely due to seasonal variation. High economic value species, such as clams, oysters, and sea cucumbers were abundant, which contrasts with the depletion associated with high levels of exploitation of these species in other parts of Indonesia.

The number of species and the abundance of macro-invertebrates were highly variable among sites (Table 3). The highest number of species was found at Lhong Angin 3 and Ba Kopra (25 species), while the highest abundance of invertebrates was found at Paloh (25,600 individuals.ha⁻¹). The diversity index of invertebrates (Krebs 1989) in Rubiah Channel, located within the government tourism reserve, was highest of all sites, (H'=3.89). Most sites within marine managed areas had a medium level of diversity (1<H'<3). In contrast to corals and fish, sites within the open access areas both in Weh Islands and Aceh Islands generally...
Table 1. Nested two-way ANOVA of the effects of time, management and site (nested within management) on fish abundance (no. ha$^{-1}$) and fish biomass (kg ha$^{-1}$). Data was log transformed prior to analysis.

<table>
<thead>
<tr>
<th>Source</th>
<th>df</th>
<th>MS</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
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<td><strong>Abundance</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Time</td>
<td>1</td>
<td>9.730</td>
<td>0.335</td>
<td>0.603</td>
</tr>
<tr>
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<td>3</td>
<td>29.021</td>
<td>10.630</td>
<td>0.042</td>
</tr>
<tr>
<td>Management*Time</td>
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<td>2.730</td>
<td>1.426</td>
<td>0.261</td>
</tr>
<tr>
<td>Site (within Management Type)</td>
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<td>1.915</td>
<td>2.736</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Error</td>
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<td>0.700</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Biomass</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Time</td>
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<td>2.970</td>
<td>0.160</td>
<td>0.7159</td>
</tr>
<tr>
<td>Management</td>
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<td>13.572</td>
<td>0.030</td>
</tr>
<tr>
<td>Management*Time</td>
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<td>0.676</td>
<td>0.576</td>
</tr>
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<td>Site (within Management Type)</td>
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<td>2.763</td>
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<tr>
<td>Error</td>
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<td>0.731</td>
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</tr>
</tbody>
</table>

Figure 7. Mean biomass (± se) (kg ha$^{-1}$) of the five major reef fish families excluding Pomacentridae, in each of the 4 management areas, in 2006 and 2007.
had higher abundance of macro-invertebrates than sites within managed areas. These large numbers of invertebrates consisted mainly of sea urchins which are often an indicator of nutrient enrichment. Agricultural runoff is known to contribute nutrients to some areas in the Aceh Islands, although the tsunami would also have contributed high amounts of nutrients that led to the proliferation of turf algae on these reefs and a source of food for the sea urchin population. The high abundance of sea urchins may also be due to low predatory pressure from carnivorous fish. It would appear that on Acehnese reefs both urchins and herbivorous fish are highly abundant and impose high grazing pressure, reducing algal cover and currently creating space for coral recruits. Factors that reduce competition for food between urchins and fish (e.g. fishing) or increase the food supply for urchins (e.g. nutrient enrichment) may change this existing dynamic and enhance urchin population growth. In areas where coral cover and diversity are already at critically low levels, such as the Aceh Island, increases in urchin grazing pressure and space limitations arising from population growth of these and other invertebrates (e.g. Acanthaster plancii, Drupella spp.) may inhibit coral settlement and have negative consequences for coral recovery.

CONCLUSIONS

Overall the condition of coral reefs in terms of coral cover and reef fish abundance in marine managed areas was considerably better than in open access areas. These areas have been protected from blasting and cyanide fishing while other reefs around Aceh Islands have been subject to unregulated fishing and destructive fishing. The low live coral cover in the Aceh Islands and high abundance of macro-invertebrates, in particular urchin populations, indicates that these reefs have been heavily impacted by a range of anthropogenic factors including overfishing, destructive fishing and eutrophication, undermining the recovery of the reefs. However, the reduction in dynamite and cyanide fishing since 2005

<table>
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<th>Parameter</th>
<th>Category</th>
<th>2006</th>
<th>2007</th>
</tr>
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<td>Trophic group</td>
<td>Benthic invert.</td>
<td>1,071</td>
<td>1,534</td>
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<tr>
<td></td>
<td>Carnivore</td>
<td>1,861</td>
<td>3,653</td>
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<td></td>
<td>Corallivore</td>
<td>712</td>
<td>605</td>
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<td></td>
<td>Detrivore</td>
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<td></td>
<td>Herbivore</td>
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<td>1,675</td>
</tr>
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<td>Omnivore</td>
<td>25,852</td>
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</tr>
<tr>
<td></td>
<td>Planktivore</td>
<td>1,864</td>
<td>1,687</td>
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<tr>
<td>Size</td>
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<tr>
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<td>5-10 cm</td>
<td>23,196</td>
<td>10,947</td>
</tr>
<tr>
<td></td>
<td>10-15 cm</td>
<td>2,050</td>
<td>1,652</td>
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<tr>
<td></td>
<td>15-20 cm</td>
<td>1,834</td>
<td>1,582</td>
</tr>
<tr>
<td></td>
<td>20-25 cm</td>
<td>290</td>
<td>827</td>
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<td>25-30 cm</td>
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<td>17</td>
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<td>35-40 cm</td>
<td>9</td>
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<tr>
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<td>&gt;40 cm</td>
<td>20</td>
<td>17</td>
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**Figure 8.** Composition of the macro-invertebrate community on reefs in Northern Aceh. Relative abundance of the top three is shown in parentheses.)
has allowed the reefs in Aceh Islands to begin to recover, with recent reports of high coral recruitment (Ardiwijaya et al. 2007). A reduction in destructive fishing activities was indicated by an increase in the number of carnivorous fish from 2006 to 2007, such as groupers, snappers, and jacks, which are target species for local fisheries. Management controls that reduce pressure particularly on carnivorous and herbivorous reef fish species are urgently required to maintain reef diversity, the existing balance between grazing pressure and coral recruitment, and to prevent sea urchin population growth from having negative impacts on coral reefs. Further studies and monitoring are required to examine if recovery of Aceh Islands reefs continues and compare these trends with nearby reefs where existing management controls on fisheries also require support and strengthening.

Table 3. Number of species, abundance and diversity index of macro-invertebrates at study sites.

<table>
<thead>
<tr>
<th>Island</th>
<th>Site Number</th>
<th>Site Name</th>
<th>Number of Species (S)</th>
<th>Abundance (ha(^{-1}))</th>
<th>Diversity Index (H')</th>
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</thead>
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<tr>
<td>Weh (MPA)</td>
<td>26</td>
<td>Bate Meuronon</td>
<td>19</td>
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<td>Rubiah Channel</td>
<td>21</td>
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<td>Rubiah Sea Garden</td>
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<td>Ujung Seurawan</td>
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<td>Gapang</td>
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<td>11</td>
<td>Luen Balee 2</td>
<td>9</td>
<td>11,260</td>
<td>1.66</td>
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RECOMMENDATIONS

- Gear restrictions on the use of netting, and enforcement of prohibited and/or destructive fishing methods, in particular blast fishing and cyanide, need consideration in the context of designing marine management areas in northern Aceh.

- Continued monitoring of reef fish, sea urchins and coral reef recovery within and outside of management zones is required for informing management options for the area and evaluating management success.

- The abundance and biomass of carnivorous fish needs to be maintained through effective management controls that reduce pressure and protect target species from overfishing.

- Identifying reef fish spawning aggregation sites of high economic value fish, such as groupers and
napoleon wrasse, is required to design MPAs and assist in marine conservation planning.
- Strong support from stakeholders, including the government, private sector, and local communities are required to maintain and strengthen existing Marine Protected Areas and build a network of marine managed areas that represents marine habitats, processes and functions of the region.

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REFERENCES


Fishing Controls, Habitat Protection and Reef Fish Conservation in Aceh

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⁴BAPPEDA, Sabang, Aceh

ABSTRACT

In 2006 and 2007 we conducted coral reef and socio-economic surveys in the northern Sumatra islands of Weh and Aceh, to evaluate existing fisheries management practices and the influence of management on coral reefs. Two types of marine resource management were found to exist on Weh Island. A community management system known as the Panglima Laot was being implemented in at least one region of the island and a government tourism reserve or Kawasan Wisata was in place in another region. Both areas had prohibited the use of netting for reef fish over the past 10 years. Areas open to unregulated fishing, except for prohibitions on blast fishing and use of cyanide, also exist on Weh island and a group of islands to the west known as Aceh. In April 2006 and 2007 we examined the structure of coral reef fish populations in each of these 4 areas, the Panglima Laot and Kawasan Wisata and the open access areas of Weh and Aceh islands. The overall biomass of reef fish was greater in marine managed areas than in unmanaged areas but did not differ between years. A separation was also found in the trophic structure of reef fish between managed and unmanaged areas. In managed areas, where cover and diversity of corals was highest, coral dependant reef fish (e.g. Chaetodontidae, Pomacentridae) had higher biomass than in unmanaged areas. For fish that are targeted by local fisheries, both carnivorous and herbivorous species showed no difference in biomass among the management areas but both Labrids and omnivores had greater biomass in managed areas where netting was prohibited. The trends indicate a positive influence of management controls on the biomass of some trophic groups. Higher biomass of small size class fish (5-15 cm) were found in the managed areas compared with unmanaged areas, and as such recruitment of fishes is possibly more successful within these areas. Although it is clear that habitat and food availability regulate fish biomass and structure, the prohibition of netting practices and relatively low fishing pressure in managed areas may explain some of the differences in the structure of reef fish between managed and unmanaged areas. Reductions in destructive fishing that occurred in some places years before the introduction of regulations on netting also protected some coral habitats from damage and is likely to have contributed to differences in reef fish population structure. These data together with information on critical habitats, socio-economic conditions and stakeholder perspectives are being used to design a network of marine protected areas for the region.

INTRODUCTION

In response to worldwide degradation and collapse of marine resources a growing interest toward more effective management in marine resources is occurring (Dayton et al. 1998, Friedlander et al. 2003). Management agencies are applying concepts of marine resource protection by recommending and implementing marine protected areas that include a variety of regulations aimed at reducing negative impacts from human and natural causes. These areas form an important component of marine conservation whereby certain areas are off limits to extraction of marine resources and provide long term stability of marine ecosystems. They can also halt the decline in marine biodiversity and changes in species and functional groups of marine taxa. The adoption of marine protected areas is a precautionary approach to management that reduces the effects of exploitation and applies an ecosystem based approach to allow ecosystems to function naturally and provide fisheries enhancements. The design and expansion of marine protected areas (MPA’s) in response to the continued exploitation of marine resources is considered a necessary management tool for protection of fish populations (Sladek Nowlis and Roberts 1999, Halpern 2003, Sale et al. 2005) and for areas of biodiversity and ecosystem function (Bellwood et al. 2004).

On coral reefs, the intensity and frequency of overfishing contributes to extreme spatial and temporal variability in the biological structure of shallow-water marine communities (Karlson and Hurd 1993, Hughes and Connell 1999). In the extreme, synergistic effects of multiple chronic disturbances such as over-fishing and nutrient inputs can lead to irreversible and fundamental shifts in biological structure from coral-dominated to algal-dominated benthos (McCook, 1999). This in turn may have significant repercussions for the long-term survival of coral associated reef fishes (Wilson et al., 2006). The benefits of MPA protection in providing increased fish stocks and other improvements in resources have been reported globally (Russ 2002) and MPAs are also advocated widely as a management tool to conserve reefs in Indonesia (Mous et al. 2005). Yet although coastal marine habitats in Indonesia have been subject to a long history of disturbance from destructive fishing practices (Tomascik et al. 1997, Edinger et al., 1998, Pet-Soede et al. 1999) few reports describe ecological benefits that have been attributed directly to MPAs in Indonesia (Christie 2004, McClanahan et al. 2006, Campbell et al. 2007). Such paucity of data in Indonesian MPAs can be attributed to lack of implementation of MPA regulations and lack of enforcement within these areas.

Coral reef fish populations are highly variable in space and time as they exhibit high movement, diel changes, migrations, high spatial variability with changes in habitats, and observer biases. This inherent and near-instant variation in coral reef fish communities makes it difficult to detect temporal changes in these communities and to attribute changes to a given anthropogenic stress. It is therefore important to measure features of reef fish communities that are capable of showing change over relatively long time scales. This may be possible for a few species, yet pooled or aggregated community metrics such as species numbers or densities and biomass at the family, community and functional level have greater potential for detecting change at the sample sizes possible in coral reef studies (McClanahan et al. 2007a).

Coral reefs in northern Aceh gained prominence following the 2004 tsunami where initial reports of widespread damage were grossly unfounded (Baird et al. 2005). It was shown that past management and the misuse of coral reefs was likely to have been the dominant factor structuring coral reef communities (Campbell et al. 2007). Coral reefs of Aceh have been subject to destructive fishing practices, such as cyanide fishing and bombing, over many years with devastating effects on fish stocks as well as the benthic reef habitats. Yet existing conservation management practices have largely been unreported. In this study we examined the status of coral reef communities
(both coral and fish communities) inside and outside marine managed areas in Aceh, against a background of considerable prior disturbance from destructive fishing practices. To evaluate if current management practices were working in Aceh we surveyed 29 sites located within 4 management areas; a government gazetted tourism conservation area (Kawasan Wisata) (6 sites), a traditional Achenese management area (Panglima Laot) (6 sites) and inside unmanaged areas with open access to fishing on Pulau Weh island (8 sites) and Pulau Aceh islands (9 sites) with open access to fishing (Fig. 1). In both managed areas regulations prohibiting the use of nets and other destructive practices had been enforced for 10 years while fishing with lines, spears, traps and other artisanal gears occurred. In open access areas fishing was unrestricted and reports of destructive fishing practices from 1970 through to the late 1990’s were common. We also examine the level of community compliance and existing fishing effort with existing marine resource regulations.

METHODS

Reefs of Weh and Aceh islands situated off the northern coast of Sumatra, Indonesia for the most part grow on bedrock and in unconsolidated sediments. Reefs surround the islands and are subject to prevailing winds and currents dependent on geographic location and season. A selection of sites that represented the diversity and geography of coral reefs of the region were surveyed in April 2006 (27 sites) and February 2007 (29 sites). The objective was to examine the structure of coral reef fish assemblages within replicate sites inside marine managed areas at 2 locations and outside managed areas at 2 locations. The areas included a government managed tourism conservation area (Kawasan Wisata) (207 ha) (6 sites), a traditional Achenese management area (Panglima Laot) (206 ha) (6 sites) and outside managed areas (>1000 ha) on Pulau Weh island (8 sites) and Pulau Aceh islands (9 sites) (Fig. 1).

Estimates of Coral Abundance

Coral cover was estimated at each site using three 50m
transects, and at each 0.5 m interval the scleractinian coral genus under the transect was recorded. Two of these transects were the same as transects used to estimate fish abundance. Cover was then expressed as the number of times a coral genus was recorded along a transect divided by the total number of intercept points (n=100) per transect.

**Estimates of Fish Abundance**

Visual estimates of reef fish species abundance and size at each site was measured at a depth of 0-2 m and 6-8m along three 50m x 5m belt transects per depth in April 2006 and February 2007. All fish were recorded from a total of 41 families except for sediment dwelling families Gobidae, Blenidae and Tripterigidae. Surveys were conducted during daytime to reduce possible temporal effects on fish abundance among sites. Fish were visually assigned to one of 9 size classes (<5 cm, 5-10cm, 10-15 cm, 15-20 cm, 20-25 cm, 25-30 cm, 30-35 cm, 35-40 cm and >40 cm). Fish less than 10cm and those greater than 10cm were surveyed separately on the same transects. Abundance was expressed as the mean abundance of fish per hectare.

**Fishing Intensity**

To obtain a standard measure of fishing effort within each management area (except in open access Aceh island where data was not available) the mean number of fishing trips per fisher within 7 lhoks (sub-district) in each management area (identified from household surveys of 143 fishers) was multiplied by the total number of fishers who fished on coral reefs in the lhok and divided by reef area adjacent to each of the lhoks.

**Statistical Analyses**

Using two-way analysis of variance (ANOVA) we examined the effect of management areas (fixed; 4 levels, Panglima Laot, Kawasan Wisata, open access Pulau Aceh, open access Pulau Weh) and time (fixed; 2 levels, 2006 and 2007) on mean fish biomass, mean biomass of 6 trophic groups at each site, mean biomass of 6 major fish families at each site, mean biomass of 9 size classes of all reef fish pooled and mean coral cover at each site (random; 6 - 9 levels). In order to understand if the variation in reef fish biomass was highest between sites within management areas (among sites) or among management areas we performed separate analyses for each year using nested ANOVA. For these analyses reef fish biomass data from each transect (n = 6) at each site (random; 6 - 9 levels) were nested within each management area (fixed; 4 levels). Significance among factors was tested at the p<0.05 level.

**RESULTS**

**Reef Fish Biomass**

The mean biomass of reef fishes in 2006 and 2007 (averaged across all families) varied by an order of magnitude among sites. In 2006 the mean (± standard error) biomass ranged from 118.2± 72.3 kg ha-1 at Pasi Janeng (site 23), up to 1193.7± 332.2 kg ha-1 at Ujung Kaureung (site 16). In 2007, biomass ranged from 149.2± 49.2 kg ha-1 at Lapeng (site 9), up to 1561.9± 554.8) kg ha-1 at the Canyon (site 29) (Fig. 2). The families with the highest biomass were the Pomacentridae and Acanthuridae which accounted for 17% and 14% of all fishes counted respectively. The next most abundant families of fishes were the Scaridae, Chaetodontidae, Labridae and Pomacanthidae, although families comprising mostly small or cryptic fishes (e.g., Apogonidae or Blennidae), which comprise a significant component of the ichthyofauna on coral reefs (Munday and Jones 1998), were not surveyed.

Nested ANOVA (site (management area)) were performed to examine variation in reef fish biomass among sites within management areas and among management areas, separately for 2006 and 2007. In this case an effect of management was found in 2006 (df 3,20, F=7.310, P<0.002) and 2007 (df 3,22, F=3.378, P<0.038) suggesting that management had an influence on reef fish biomass. Significant variation among sites nested within management zones, in 2006 (df 22,148, F = 2.746, P < 0.001) and 2007 (df 20,84, F = 2.216, P = 0.006) was also found indicating high
Figure 2. Mean (±Standard Error) reef fish biomass (kg ha⁻¹) of all fish species at 29 sites in 4 management areas of Aceh in 2006 and 2007.

Figure 3. Mean (±Standard Error) reef fish biomass (kg ha⁻¹) of six major families in 4 management areas of Aceh in 2007. Different superscript letters denote significant differences (P<0.05) in biomass among management zones for a given reef fish family.
Figure 4. Mean (±Standard Error) reef fish biomass (kg ha⁻¹) of trophic groups in 4 management areas of Aceh in 2007. Different superscript letters denote significant differences (P<0.05) in biomass among management zones for a given reef fish trophic group.

Figure 5. Mean (±Standard Error) reef fish biomass (kg ha⁻¹) of size classes in 4 management areas of Aceh in 2007.
variation in reef fish biomass among sites within a given management area.

**Reef Fish Structure**

At sites within the Kawasan Wisata and Panglima Laot on Weh island there was higher biomass of Chaetodontidae, Labridae and Pomacentridae compared with open access areas of Pulau Aceh (Fig. 3). This was also reflected in the trophic structure with higher biomass of corallivores (mostly Chaetodontidae) and omnivores (mostly Balistidae and Pomacentridae) in Kawasan Wisata and Panglima Laot areas compared with open access areas of Pulau Aceh (Fig. 4). The size structure of coral reef fish assemblages was characterized by a higher biomass of 5-10 cm, 10-15 cm and 15-20 cm size classes in Kawasan Wisata and Panglima Laot compared with the open access area of Pulau Aceh (Fig. 5).

**Coral Cover**

There was a significant effect of management area (df 3,48, F = 22.273, P < 0.001) on percentage coral cover with post-hoc tests (Bonferoni) revealing significantly lower coral cover in open access Pulau Aceh (9.16 ± 1.50) compared to open access Pulau Weh (26.0 ± 2.21), Kawasan Wisata (38.8 ± 2.67) and Panglima Laot (53.3 ± 2.82) areas. Coral cover in the Panglima Laot area was also significantly higher than in the open access area of Pulau Weh.

**Fishing Pressure**

Highest fishing pressure was found in the lhoks of Balohan and Keunekai within open access areas and lowest in marine protected areas in Panglima Laot and Kawasan Wisata lhoks (Fig. 6).

**DISCUSSION**

Teasing apart the effects of habitat condition and management regulations on reef fish populations is difficult given the complex associations between low coral cover, low fish densities and high fishing pressure and the influence of coral habitat type and area on coral reef fish assemblages (Russ and Alcala 1996, Bellwood and Hughes 2001). Nonetheless, the lower fish abundance in open access fishing areas of both Aceh and Weh, irrespective of the lower coral cover and diversity, suggests that past fishing practices, including bombing and cyanide poisoning that were common throughout Indonesia (Pet Soede et al. 1999, Tomascik et al. 1997, Hopley and Suharsono 2002) impacted coral cover and fish biomass in Aceh. Because of the national laws prohibiting these illegal practices it is unlikely that fishers will admit to their use. In contrast, local management regulations such as bans of netting are rarely enforced so information from fishers on their use of legal gears are likely to reflect accurate estimates of fishing pressure. The relatively lower fishing pressure inside marine protected areas compared with areas where fishing is unregulated has almost certainly benefited reef fish biomass, despite regulations limiting netting not being fully complied with, enforced or monitored. Alternative income opportunities (ie. tourism, agriculture) in the lhoks adjacent to the marine protected areas have more than likely reduced
dependency on fishing in these areas and allowed communities to comply to some degree with fishing regulations.

The enhancement of fish numbers inside marine protected areas is likely to be closely coupled with habitat availability which promotes growth and maturation of fish in post-recruitment phases (Jones and McCormick 2002). Coral dwelling species including Pomacentridae, Chaetodontidae and Labridae showed higher biomass in protected areas where coral cover and diversity was higher than in open access areas (see also Baird et al. 2005). Both Pomacentridae and Chaetodontidae are heavily reliant on coral habitat (Bellwood and Hughes 2001, Jones et al. 2004) which provide shelter from predatory fish species and are likely to improve survival (Jones and McCormick 2002). Coral habitat is also likely to assist in recruitment of reef fishes by providing cues for active habitat selection by reef fish larvae (Montgomery et al. 2001). These coral dwelling families are not targeted by fishing and hence differences in biomass would appear directly related to the quality and quantity of coral habitat.

In Aceh, the first 10 years of marine management has seen Labrids and the faster growing coral dwelling Chaetodontids and Pomacentrids benefit in terms of biomass from the marine management in place, as they are clearly disadvantaged in areas where coral cover and diversity is low and algal growth is high. In contrast families such as Acanthuridae, Scaridae and Pomacanthidae showed no differences in biomass between protected and non-protected areas although both Acanthuridae and Scaridae are targeted by net and handline fisheries. This suggests that the trophic behaviour of reef fish may to some extent explain the observed differences in biomass among management areas. Both Acanthuridae and Scaridae are roaming herbivores that may be supported by high abundance of algae typically found in the open access reefs of Aceh and Weh (Campbell et al. 2007). Ecological succession may also explain differences in reef fish structure as increases in the biomass of some families such as Labridae and Scaridae have been reported during the first ten years of closure from fishing, while increases in the biomass of Balistids and Acanthurids have taken 10-20 years of closure (McClanahan et al. 2007b). In coral reef systems fish biomass may take many years or decades to recover to full diversity and biomass (Micheli et al. 2004, McClanahan et al. 2007b), as the influence of competitive interactions among functional groups alters the rate of change in the biomass of reef fish with time.

The paucity of small fish size classes (5-15 cm) in open access areas of Weh and Aceh islands may be due to the overuse of nets in open access areas and capture of small size fish. However, our observations suggest that exploitation of small sized fish is not a major concern except possibly the targeting in some areas by the reef fish aquarium trade. Interestingly we have noted large numbers of small sized cohorts in both 2006 and 2007 fish surveys in marine protected areas suggesting that recruitment to these reefs may be occurring. Mechanisms responsible for possible recruitment remain unclear although factors linked to habitat availability that affect recruitment and predation, including density dependent post-settlement mortality, food supply, larval growth and larval mortality (Jones and McCormick 2002), may influence reef fish structure in Aceh.

The overall higher reef fish biomass and low fishing effort in protected compared with open access areas, indicates a degree of compliance with fishing regulations and a sensitivity of reef fish populations to community or district based management controls. Yet the low biomass of carnivorous fish found in all management zones indicates that these populations remain in a state of severe depletion. Although fishing effort is low in protected areas and restrictions on netting may be working, a lack of restrictions on handline and speargun fishing and the illegal use of cyanide to target carnivores has undoubtedly contributed to the depletion in large fishes, typical of overfishing in coral reef systems (Russ and Alcala 1996, Micheli et al. 2004). Controls such as gear restrictions and periodic closures may be popular with local communities but have limits to preserving fishery
dependent species and ecological processes. On the other hand, permanent closures have been advocated as the only way to maintain coral reef ecosystems that are representative of unfished ecosystems with high fish biomass (McClanahan and Graham 2005, McClanahan et al. 2007b). Achieving this in Aceh is problematic when fishing dependency is often seasonally high, enforcement of regulations is low, limited monitoring occurs and poor management among responsible agencies persists.

In the absence of data on reef fish populations prior to restrictions on fishing we cannot conclusively attribute the higher biomass of reef fish in the Kawasan Wisata and Panglima Laot protected areas to regulations limiting netting in these areas. The differences may also be due to the history of blast fishing which ceased in the late 1980’s in the two managed areas of Pulau Weh but continued unabated until the mid 1990’s in some of the open access areas. Nonetheless, whether it be gear restrictions, limiting fishing pressure or the reduction in destructive fishing the conservation practices adhered to by communities would appear to have benefited coral reefs by reducing the harvest of some species of reef fish and protecting coral from damage. Responses of fishes to protection from fishing are influenced by many complex factors, including the size of reef, the structure of reef fish populations, the proximity of other reefs and the level of compliance with protection regulations (Babcock et al. 1999; McClanahan and Mangi 2001; Shears and Babcock 2003) and positive effects of protection may require many years to become manifest (Micheli et al. 2004, Russ et al. 2005, McClanahan et al. 2007b). Nonetheless, local MPAs such as the ones studied in the present study are gaining increasing acceptance among scientists as one of the few effective ways of managing fisheries of coral reef species (Russ 2002), and may be critical in making reefs more resilient to acute natural and anthropogenic disturbances (Bellwood et al. 2004).

The coral reefs of Weh Island more than a year after the tsunami are largely intact. The legacy of traditional and government based marine resource management appears to have protected some reefs from human misuse and afforded refuge for a diverse and abundant suite of reef fish. These reefs have depleted numbers of species targeted by artisanal coral reef fisheries most likely because permanent closures or no take zones have not been part of the management controls in Aceh and continue to lack community support. Nonetheless, the findings of existing community or government based management in reducing unsustainable fishing practices on coral reefs have rarely been documented in Indonesia (Christie 2004, McClanahan et al. 2006). On Aceh Island the condition of many reefs remains a cause for concern, yet the chronic mismanagement of marine resources appears to have abated in the wake of the tsunami and there is evidence that recruitment of corals and fish is occurring. This encouraging response is evidence of the resilience of coral reefs to severe disturbance and the determination of local communities to protect some of the most diverse coral reefs in Sumatra over the past 10 years. Pivotal to achieving sustainable fisheries and coastal livelihoods will be the strengthening of relationships between the communities and institutions involved in marine resource management.

ACKNOWLEDGEMENTS

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REFERENCES


Status of Earthquake and Tsunami Affected Coral Reefs in the Andaman and Nicobar Islands, India

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Reef Watch Marine Conservation, Mumbai, India

INTRODUCTION

The Andaman and Nicobar Islands, located in the south eastern part of the Bay of Bengal between latitudes 6° 45’ and 13° 41’ S and longitudes 92° 12’ and 93° 57’ E, are host to a rich biodiversity. The archipelago is one of the few key biodiversity regions in the world surrounded by fringing coral reefs characteristic of the Southeast Asian region, and is the most diverse among Indian subcontinent reef areas (Pillai, 1983, Davidar et al., 1994).

On 26th December 2004, an earthquake measuring 9.3 on the Richter scale hit the region. As a result of tectonic activity, low lying areas from South Andaman to the Nicobar Group of Islands were submerged by 1-2 meters, while large areas, including coral reefs, were uplifted in the northern group of the Andaman islands. The uplift resulted in permanent damage to shallow reefs in the northern group of the Andaman Islands (Kulkarni, 2005). The earthquake also generated tsunamis, the effect of which ranged from a temporary rise in sea level such as in South Andaman, and up to 15 m high waves in parts of the Nicobar Islands. This caused loss of human lives and destruction to infrastructure in the islands (Sankaran et. al., 2005). The environmental impacts of the tsunami were diverse, with damage to coral reefs and

Figure 1. The Andaman and Nicobar Islands. Numbered dots indicate survey sites. See also Table 1.


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other coastal habitats in the entire region (Malik and Murti 2005).

This study gives a detailed account of the impact and long-term implications of the earthquake and the tsunami on the reefs of the Andaman and Nicobar Islands.

**METHODOLOGY**

The study was carried out at 31 sites around 29 islands in the Andaman and Nicobar Islands (Table 1, Fig. 1), between February 2005 and June 2006. The extent of uplift was estimated based on the difference between the new and the old water level as indicated by high-water marks on rocks and structures. The presence of full-grown barnacles at deeper levels of jetties served as an indication of the submergence level. The height of the tsunami was estimated by newly formed high water marks on trees and structures along the coast, referred against previous indications.

Benthic cover was assessed using randomly laid Line Intercept Transects (LIT) (Loya, 1972). Five transects 20 meters long were run parallel to depth contours at each site at depths between 4 and 9 meters. Transects were separated by at least 10 to 30 meters, to cover approximately 150 to 250 m along the reefs. Benthic cover categories recorded were live coral, dead coral, broken corals and rubble, sand, algae, soft coral and other. Relative abundance of coral genera was recorded in permanent LIT and is presented by genera as percentage of total coral cover.

In addition, general visual observation of reefs and associated biota were made using SCUBA down to depths of 30 meters. Damage to the reef was categorized based on type/cause of the damage, and GPS readings of the damaged area were recorded. The areas were then demarcated on high-resolution post-tsunami satellite images (scale 1:2000) obtained from Google Earth Pro Inc. and areas of destruction were approximated by constructing polygons connecting the coordinates, and using Google Earth software for area calculations.

**Table 1. Sites where benthic assessment was carried out. Site numbers correspond to numbers in Figure 1.**

<table>
<thead>
<tr>
<th>Nr.</th>
<th>Site</th>
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<tr>
<td></td>
<td><strong>North Andaman</strong></td>
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<tr>
<td>1</td>
<td>Smith Island</td>
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<td>2</td>
<td>Lamia Bay</td>
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<td>3</td>
<td>North Reef Island</td>
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<td></td>
<td><strong>Middle Andaman</strong></td>
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<td>4</td>
<td>Interview Island</td>
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<td>5</td>
<td>South Reef Island</td>
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<td>6</td>
<td>Aves Island</td>
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<td>7</td>
<td>South Island</td>
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<td></td>
<td><strong>Ritchie’s Archipelago</strong></td>
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<td>8</td>
<td>Outram Island</td>
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<td>9</td>
<td>Henry Lawrence</td>
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<td>John Lawrence</td>
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<td>Havelock Island</td>
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<td>12</td>
<td>Neil Island</td>
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<td>13</td>
<td>South Button Island</td>
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<td><strong>South Andaman</strong></td>
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<td>14</td>
<td>North Bay</td>
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<td>15</td>
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<td>16</td>
<td>Redskin Island</td>
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<td><strong>Nicobar Group</strong></td>
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<td>Car Nicobar</td>
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<tr>
<td>23</td>
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<td>24</td>
<td>Camorta Island (southeastern)</td>
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<td>Pigeon Island</td>
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<td>29</td>
<td>Little Nicobar</td>
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<td>30</td>
<td>Menchal Island</td>
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<td>31</td>
<td>Kondul Island</td>
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The earthquake resulted in uplift of the west coast of North Andaman, which led to mortality of corals and associated life-forms in shallow waters due to aerial exposure and direct sunlight. Signs of sand deposition were also observed on these exposed reefs. Middle Andaman, including small islands from Landfall Island to South Reef was uplifted by more than one meter and the areas on the eastern coast such as Diglipur, Smith and Ross Islands, by less than one meter. Table 2 summarizes reef damage in the archipelago, including areas not surveyed with LIT. Almost 50 km² of reef was destroyed or severely damaged.

Smith island (1 in Fig. 1) has a length of 8.6 km and a width of 5.1 km. It has extensive mangroves on the northwestern side and narrow fringing reefs all around the island. The reef flat contains mainly rocks, sand and dead coral heads. Extensive coral growth starts at a depth of 4 meters and extends along a gradual slope down to 10 meters. The western reef is sheltered and dominated by Porites spp. while the eastern reef is dominated by Acropora spp.

The island appears to have been raised by half a meter, resulting in exposure of the reef flat on the eastern side. As coral growth on the reef flat was very moderate prior to the Tsunami, the impact, if any, has not been significant. The tsunami was not violent in this area and water only rose by around 2.5 meters. No physical damage, such as broken coral colonies, were seen.

In 2003, prior to the tsunami, live coral cover on the eastern side of the island was 54.0%, dominated by Acropora (44.6%) followed by Porites (16.1%), Favia (9.3%) and Montipora (8.3%). Coral cover in 2006 was 50.5%, with the relative abundance of most major coral genera unchanged (Acropora 41.5%; Porites 18.3%; encrusting Montipora 11.5%).

Lamia Bay (2 in Fig. 1) is situated on the east coast of North Andaman and at the base of Saddle Peak (the highest point in the Andaman and Nicobar Islands). The site is dominated by patch reefs. Eyewitness reports confirm that the tsunami came in the form of a rise in sea water of around 1.9m and did not cause any physical destruction on land. There are also no signs of reef damage. Live coral cover was 41.1% before the tsunami and 39% after. Since 2003 rubble cover has been reduced from 11.8% to 1.3% while the soft coral cover has increased from 0.5 to 9.8%.

Prior to the tsunami, North Reef island (3 in Fig. 1) had luxuriant coral growth all around, and was considered one of the most diverse reefs in the Andaman and Nicobar Islands. The northern side of the island had an inter-tidal reef flat dominated by mono-specific stands of Acropora formosa. The reef on the western side of the island was dominated by Acropora spp. and Montipora aequituberculata. Millepora dichotoma, Porites lutea and Acropora robusta dominated the southeastern bay.

The tectonic activity caused North Reef to be raised by more than 1.5 meters. This has resulted in exposure of the reef flat on the northern and western sides, resulting in increased turbulence in the sub-tidal reefs and deposition of sand. Destruction of coastal

<table>
<thead>
<tr>
<th>Island</th>
<th>Cause</th>
<th>Damaged Area (km²)</th>
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<tbody>
<tr>
<td>Landfall</td>
<td>UL</td>
<td>5.82</td>
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<tr>
<td>West</td>
<td>UL</td>
<td>2.07</td>
</tr>
<tr>
<td>White cliff</td>
<td>UL</td>
<td>0.16</td>
</tr>
<tr>
<td>Reef</td>
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 RESULTS

North Andaman
The earthquake resulted in uplift of the west coast of North Andaman, which led to mortality of corals and associated life-forms in shallow waters due to aerial exposure and direct sunlight. Signs of sand deposition were also observed on these exposed reefs. Middle Andaman, including small islands from Landfall Island to South Reef was uplifted by more than one meter and the areas on the eastern coast such as Diglipur, Smith and Ross Islands, by less than one meter. Table 2 summarizes reef damage in the archipelago, including areas not surveyed with LIT. Almost 50 km² of reef was destroyed or severely damaged.

Smith island (1 in Fig. 1) has a length of 8.6 km and a width of 5.1 km. It has extensive mangroves on the northwestern side and narrow fringing reefs all around the island. The reef flat contains mainly rocks, sand and dead coral heads. Extensive coral growth starts at a depth of 4 meters and extends along a gradual slope down to 10 meters. The western reef is sheltered and dominated by Porites spp. while the eastern reef is dominated by Acropora spp.

The island appears to have been raised by half a meter, resulting in exposure of the reef flat on the eastern side. As coral growth on the reef flat was very moderate prior to the Tsunami, the impact, if any, has not been significant. The tsunami was not violent in this area and water only rose by around 2.5 meters. No physical damage, such as broken coral colonies, were seen.

In 2003, prior to the tsunami, live coral cover on the eastern side of the island was 54.0%, dominated by Acropora (44.6%) followed by Porites (16.1 %), Favia (9.3%) and Montipora (8.3%). Coral cover in 2006 was 50.5%, with the relative abundance of most major coral genera unchanged (Acropora 41.5%; Porites 18.3%; encrusting Montipora 11.5%).

Lamia Bay (2 in Fig. 1) is situated on the east coast of North Andaman and at the base of Saddle Peak (the highest point in the Andaman and Nicobar Islands). The site is dominated by patch reefs. Eyewitness reports confirm that the tsunami came in the form of a rise in sea water of around 1.9m and did not cause any physical destruction on land. There are also no signs of reef damage. Live coral cover was 41.1% before the tsunami and 39% after. Since 2003 rubble cover has been reduced from 11.8% to 1.3% while the soft coral cover has increased from 0.5 to 9.8%.

Prior to the tsunami, North Reef island (3 in Fig. 1) had luxuriant coral growth all around, and was considered one of the most diverse reefs in the Andaman and Nicobar Islands. The northern side of the island had an inter-tidal reef flat dominated by mono-specific stands of Acropora formosa. The reef on the western side of the island was dominated by Acropora spp. and Montipora aequituberculata. Millepora dichotoma, Porites lutea and Acropora robusta dominated the southeastern bay.

The tectonic activity caused North Reef to be raised by more than 1.5 meters. This has resulted in exposure of the reef flat on the northern and western sides, resulting in increased turbulence in the sub-tidal reefs and deposition of sand. Destruction of coastal
vegetation indicates that the tsunami was severe with a height of about 3.5 meters on the eastern side of the island.

At the monitoring site (south-eastern), the average live coral cover pre-tsunami was 73.5%, dominated by large colonies of *Acropora* spp. at a relative abundance of 31.7%, followed by *Porites* (21.8%) and *Millepora* (10.7%). By 2006 the live coral cover had been reduced to 23%. The relative abundance of *Acropora* and *Millepora* was reduced to 8.8% and 2.3%.

**Figure 2.** Benthic cover at study sites in the Andaman and Nicobar Islands: North Andaman - North Reef; Middle Andaman - Interview Island, South Reef; Ritchie’s Archipelago - Henry Lawrence, John Lawrence, Neil Island; South Andaman - Grub Island, Twins Island, Cinque Island; Car Nicobar.
respectively. The percentage cover of rubble increased from 2% to 41% (Fig. 2). The area of coral reef destroyed by exposure from the uplift of the landmass and due to the tsunami is estimated to be about 13 km².

**Middle Andaman**

Reef damage in Middle Andaman is summarized in Table 3, including areas not surveyed with LIT. Over 22 km² of reef was destroyed or severely damaged.

Interview Island is situated to the south of North Reef Island and to the west of Middle Andaman. Prior to the earthquake and tsunami the western, northern and southern areas of this island had extensive reef flats mainly dominated by *Porites* micro-atolls. The sheltered eastern side showed patchy coral reefs and high turbidity.

Due to uplift caused by the earthquake, reef flats on the northern and western side have become exposed. Waves are breaking directly on the edge of reef flat, resulting in low visibility and deposition of sand mainly along the sub-tidal reefs on the western side.

Live coral cover in the sub-tidal region of the southern reef (4 in Fig. 1) has been reduced from 67.7% (dominated by *Porites* and *Acropora*) to 18.4%. The percentage cover of dead intact coral has increased from 25% to 65.5%, and that of rubble has increased from 1% to 8% (Fig. 2). High water temperature was also observed in the southern reef areas during the surveys in 2006. This is possibly due to localized warming of stagnant seawater in sub-tidal zones of the southern reef during low tide, causing coral mortality.

South Reef Island lies to the south of Interview Island. It is surrounded by a 30 to 70 meter wide reef flat, wider on the western than on the eastern side. This island was raised by 1 meter as a result of the earthquake, causing damage to coral reefs mainly on the western reef flat. The pre-tsunami coral cover on the eastern side (5 in Fig. 1) of the island, 63.2%, was reduced to 48% in 2006 (Fig. 2). The mortality caused a shift in the coral community with the dominance of *Acropora sp.* reduced from a relative abundance of 46.6% to 31.4% in 2006. An increase in relative abundance was recorded in *Porites* sp. (16.8% prior to the tsunami to 21.6% after), *Echinopora* (9.5% to 13.7%) and *Millepora* (7.1% to 9.3%).

Aves Island is situated on the eastern side of Mayabundar. Coral reefs occur all around the island from a depth of 4 meters to 14 meters, except at the southern end, where the seabed is covered mainly by rocks. On the eastern and northern sides, *Acropora* and *Porites* dominate the coral reefs while on the western side coral reefs are dominated by *Porites*. No significant change in live coral cover was recorded at the study site on the eastern side of the island (6 in Fig. 1.) between 2003 (61%) and 2006 (58%). *Acropora* (40.2%) dominated the live coral cover followed by *Porites* (23.1%), *Hydnophora* (8.9%) and *Echinopora* (6.4%).

Sound Island is situated to the northeast of Mayabundar and Aves Island. Coral reefs were surveyed on the eastern side (7 in Fig. 1.) of the island. Live coral cover was 47% in 2003 and 45% in 2006. *Porites* dominated the live coral cover with a relative abundance of 44.4% followed by *Montipora* (17.3%) and *Acropora* (12.6%).

**Ritchie’s Archipelago**

Ritchie’s Archipelago comprises 4 large islands, 7 small islands and several islets, extending in a roughly north-south chain, parallel to the main Great

<table>
<thead>
<tr>
<th>Island</th>
<th>Cause</th>
<th>Damaged Area (km²)</th>
</tr>
</thead>
<tbody>
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<td>Middle Andaman</td>
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<tr>
<td>Spike</td>
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<tr>
<td>South reef</td>
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<td>Anderson</td>
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</table>
Table 4: Estimated reef damage in Ritchie’s Archipelago, South Andaman and Little Andaman. The cause of damage is shown as uplift (UL) and/or tsunami (TS).

<table>
<thead>
<tr>
<th>Island</th>
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</tr>
</thead>
<tbody>
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<tr>
<td>South Andaman group</td>
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<tr>
<td>Ritchies Archipelago</td>
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<tr>
<td>Little Andaman</td>
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</tr>
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</table>

Andaman Group. Corals reefs at six islands were surveyed. Reef damage is summarized in Table 4.

Outram Island lies to the north of Henry Lawrence. Mangroves are present on the northern and southern side of the island. The island is surrounded by fringing reefs. The western reef flats are narrow, with widths ranging from 50-80 meters, and mainly dominated by Acropora. Porites and Acropora dominated the live coral cover on the southern and eastern sides with a relative abundance of 42.4% and 31.9% respectively. The maximum coral growth occurs down to 12 meters, beyond which sea fans, soft corals and some Faviids dominated the rocky bottom.

Coral cover and community composition appear unchanged by the tsunami, with live coral cover on the western side of the island (8 in Fig. 1.) recorded as 67% in 2004 and 64.7% in 2006. A marginal increase in dead intact coral cover was seen, with no increase in rubble.

Henry Lawrence Island is part of the Rani Jhansi Marine National Park. Coral reefs occur all around the island except on the western coast, where mangroves are found. The coral communities are mainly dominated by Porites lutea and P. nigrescens. On the eastern coast, reef flats range from 50 – 100 meters in width, with gentle reef slopes down to about 13 meters, beyond which the bottom is sandy. The western side of Henry Lawrence and the eastern side of John Lawrence form a narrow channel from south to north.

The tsunami wave’s kinetic energy increased in the narrow channels causing high-velocity currents. This caused damage to corals along the edges of reef flats. Large colonies of Porites lutea on reef edges were uprooted and some stretches of mangroves were destroyed. Accumulation of rubble was observed on the central-western side of the reef. Qualitative assessment carried out in the monitoring site on the western side (9 in Fig. 1.) revealed that live coral cover was reduced to 21.5% compared to 45.1% in 2003. The percentage cover of rubble has increased to more than 16% in 2006 compared to 2003 (Fig. 2). Porites is still the dominant genus, making up more than half of the live coral cover. Dense clusters of Turbinaria algae have occupied dead parts of Porites colonies.

John Lawrence Island is also part of Rani Jhansi Marine National Park. The entire north-western region is surrounded by extensive mangrove forests. Coral reefs are mainly patchy with narrow reef flats that slope steeply to a maximum depth of 12 meters. The reefs are dominated by large colonies of Porites.

The concentrated energy of the tsunami in the channels between John Lawrence, Wilson and Nicholson islands caused mangrove and coral reef destruction. Large patches of mangroves alongside the channel were destroyed, and boulders of Porites were uprooted and scattered all over the channel making navigation for dinghies difficult. Coral reefs were studied on the eastern side (10 in Fig. 1.) of the island. Large colonies of Porites were tilted and smaller ones were uprooted and smothered. Live coral cover declined from 65.5% in 2003 to 38.5% in 2006, and cover of rubble doubled (Fig. 2). The coral community structure suggests that Porites sustained damage with a reduction in relative abundance from 59.9% to 43.2%, with encrusting Montipora and foliose Echinopora increasing in relative abundance. Large clusters of Turbinaria algae were found on dead coral boulders.

Havelock Island is surrounded by fringing reefs with wide reef flats on the western side and narrow on the eastern side. Reef flats are dominated by large colonies of Porites, the top portions of which are mostly dead. This island experienced the tsunami mainly in the form of a rise in sea water level. Along
the north and north-west coasts the rise in sea water level was about one meter accompanied by strong currents. Corals, mainly in the channel and at the mouth of the channel sustained most of the damage. However, the damage was restricted to smaller colonies of *Porites* and *Acropora*.

The monitoring site, located on the north-western reef (11 in Fig. 1), close to the lighthouse, has a steep reef slope extending down to 21 meters, with a sandy bottom beyond. The average live coral cover at this site declined from 59.7 % in 2003 to 43.2% in 2006. Change in cover of intact dead coral was minimal but rubble cover increased somewhat.

Neil Island is situated to the south of Havelock Island. Fringing reefs occur on the eastern and western side of the island, with patch reefs to the north and south. Reefs studied on the northwestern side of the island (12 in Fig. 1) were *Porites* dominated, making up over two thirds of the live coral cover. A reduction in live coral from 27.8% to 18.6% in 2006 was observed, while the cover of dead standing coral remained unchanged and comparatively high (Fig. 2).

Coral reefs around South Button Island (13 in Fig. 1) studied in 2006 were healthy, with a live coral cover of 80% and high fish diversity and abundance.

**South Andaman**

In South Andaman, coral reefs around North Bay, Chidiyatapu and some of the islands of the Mahatma Gandhi Marine National Park were studied. Coral reef mortality due to the tsunami was visible in channels and edges of reef slopes. In recent years solar radiation has caused coral mortality in shallow areas (Kulkarni, 2004). The increase in depth due to subsidence may have a positive effect in promoting coral growth. Reef damage is summarized in Table 4.

North Bay is located near Port Blair Harbour, with fringing reefs to the north and south. The tsunami reached a height of around 2.5 meters in this area. Corals on the northern side of the bay were not affected as the community is dominated by massive boulders of *Porites*. The southern side (14 in Fig. 1) sustained around 10% damage that was restricted to primarily *Acropora* colonies, and an increase in coral rubble. *Porites* species, such as *P. solida* and *P. nigrescens*, remain dominant at the site with a relative abundance of 91.4%.

Grub Island is part of M. G. Marine National Park. The reef around this island has a gentle slope and coral growth occurs to a depth of 6 meters, with a higher live coral cover on the eastern than on the western side. The coral community is dominated by *Acropora*, followed by *Porites* and *Echinopora lamellose*; further north, where the reef stretches up to 300 m from the shoreline, by *Porites* and *Millepora dichotoma*. Other common coral genera include *Montipora* and *Hydnophora*. The southern part of this island is sandy with no coral growth.

On reefs studied in 2003 on the eastern side (15 in Fig. 1) the coral cover was about 57.7%, dominated by *Porites* (46%), *Acropora* (22.3 %) and *Echinopora* (22.3 %, mainly *E. lamellosa*). By 2006 coral cover had increased by 17 percentage points (Fig. 2), due to an increase especially in *Acropora*, apparently with aggressive competition from *Echinopora*. A reduction in intact dead coral suggests that new coral growth is covering dead standing reef structure. *Acropora* and *Porites* now dominate the reef, followed by *Echinopora*.

Redskin Island is surrounded by fringing reefs. Reefs to the north have a gentle slope with diverse coral growth that is comparatively healthier in deeper than in shallow water. The dominant species is *Acropora formosa*, unlike the other parts of the island which are dominated by *Porites*. The reef flat on the northeastern side of the island, protected from strong wave action, is wide, while in the rest of the area it is narrow and extends about 30 m from the shore. Reefs to the southeast are patchy and composed of *Porites*, *Favia*, *Favites*, *Acropora* and *Pectinia*. The southwestern and southern areas are mainly rocky with patchy coral growth. The depth of reef areas decreases towards the south. The western reef is narrow with a steep slope to a depth of 15 meters where the seabed is covered by calcareous sand.

Observations made in 2003 on coral reefs in the north, west and eastern sides of the island (16 in Fig.
1) showed an average coral cover 32.6 %, dominated by *Porites* (56.4%), followed by *Echinopora*, *Acropora*, *Montipora*, *Hydnophora*, *Favia*, *Lobophyllia*, *Turbinaria*, *Pocillopora*, *Galaxea*, *Pectinia*, *Montastrea* and *Symphillia*, with relative abundance ranging from 1 to 8%. Though there was an impact of the tsunami on corals, it is restricted to the outer edge of the reef crest, where some large colonies of *Porites* have been toppled and some of those on the edge of the steep reef slope have slid down into deeper waters. However, many of these colonies have survived in spite of the change in habitat. A moderate reduction in percentage cover was recorded, from 32.6% in 2003 to 25.6% in 2006.

On Jolly Buoy Island coral reefs were surveyed on the north, west and eastern sides of the island in 2003. The average coral cover was 37.4%, with *Porites* dominating the reef (62.3%), followed by *Hydnophora rigida* (11.8%) and *Echinopora* (5.9%). Other genera ranged from 1 to 4% of the coral cover. Post-tsunami surveys revealed an impact only on the eastern side of the island (17 in Fig. 1), with a reduction in coral cover to 28.67% in 2006. The percentage of rubble has increased to 12.8% in 2006 compared to 4.1% in 2003.

Chidiyatapu (18 in Fig. 1) is the southern end of South Andaman Islands. Fringing coral reefs occur mainly in the bay, from about 300m from the shore. The reef ends at Munda Pahad (barren hillock), after which the habitat becomes rocky. The width of the fringing reefs is about 20-30m, followed by a gradual slope down to a depth of 10m and sandy substrate. In 2003 the average coral cover in the area was 51.3%, mainly due to dominance of *Porites solida* and *Porites rus*, and growth of encrusting *Montipora* over dead *Porites* and *Acropora*. The relative abundance of *Porites* was 40.4 %, followed by *Montipora* (28.8%). The relative abundance of *Acropora* was 1.0 %. The survey in 2006 suggests that the tsunami and earthquake had minimal impact on the reefs, recording a coral cover of 47%.

Rutland Island is one of the largest islands in this archipelago. This island has extensive coral reefs, mangroves and turtle nesting beaches. The monitoring site lies on the eastern side of Rutland (19 in Fig. 1). Here, coral growth starts 50m from the shore, the reef slope is gentle and coral growth continues to a depth of 8m, beyond which the bottom is sandy. The coral cover was 26.9% in 2003, dominated by *Porites solida* and *P. lutea* (48.2%) followed by encrusting *Montipora* (10.2%), *Hydnophora rigida* and *H. microconos* (8.3%) and *Acropora* (7.8%). The tsunami had minimal impact at this site, with the coral cover of 23.2% recorded in 2006.

Twins Islands are the southernmost islands in M. G. Marine National Park. Corals occur at a depth of 2 to 12 m. Shallow areas mainly comprise of *Millepora* and *Heliopora*, with a live coral cover around 35-40%. *Porites* was found growing in patches off the southern rocky shoreline. *Acropora* colonies dominated the deeper (10-12m) parts of the reef, with a relative abundance of 10%. Reefs on the eastern side of West Twins Island (20 in Fig. 1) surveyed in 2003 had a coral cover of 35.6%, dominated by encrusting *Montipora* (relative abundance 38.2%), mainly growing over dead *Millepora*. Other major genera were *Millepora* (30.1 %) and *Porites* (19.5 %). Post-tsunami surveys indicated a significant increase in coral cover, to 67% in 2006 (Fig. 2). The reefs have also undergone a change in species composition, with *Acropora*, *Heliopora*, *Pocillopora* and *Porites* now

**Table 5** Estimated reef damage in the Nicobar Group of islands. The cause of damage is shown as tsunami (TS) and/or sedimentation (sed).

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<thead>
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<th>Island</th>
<th>Cause</th>
<th>Damaged Area (km²)</th>
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</thead>
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<td>Treis</td>
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<tr>
<td>Kondul</td>
<td>TS/Sed</td>
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<tr>
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<td>46.39</td>
</tr>
<tr>
<td>Total</td>
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<td>210.63</td>
</tr>
</tbody>
</table>
dominant. No major changes were evident in algae and soft coral cover.

Cinque Island is situated to the south of Rutland Island. The northern side has a rocky bottom and a steep slope down to 20 m, with sea fans, soft corals and some sporadic growth of sub-massive corals such as Goniastrea, Lobophyllia, Coeloseris and Goniopora. Currents are strong around the northern tip of the island. Towards the south the gradient of the slope decreases, with sandy bottoms, rubble and coral reefs in the southern portion of the island. Fish diversity is high in this area. Coral reefs were surveyed on the western side of this island (21 in Fig. 1). In 2003 the average coral cover was high (61.2 %), increasing with depth down to 15 m, and dominated by Acropora (relative abundance of 46.9 %), followed by Porites (21.5 %), Millepora (5.3 %) and Favia (5.0 %).

The 2006 assessment suggests that the reef was affected by the tsunami, with coral cover reduced by a fifth at the monitoring site (Fig. 2). No substantial change in intact dead coral was noticed, but rubble had increased from 7.0 % in 2003 to 20.2 % in 2006, indicating a direct impact of the tsunami.

**Nicobar Group**

Of the Andaman and Nicobar archipelago, the Nicobar group of Islands was closest to the epicenter of the 2004 earthquake. The resultant displacement of water had a severe impact on these islands, with a tsunami wave height reported at around 12 meters (the seawater first receded to a great extent), causing the death of thousands of people and wiping out coastal habitats. Coral reef destruction was highest in this group of islands, with the submergence of the islands by more than two meters triggering sedimentation that continued for more than 8 months due to the monsoon that followed. Table 5 provides detail on reef area impacted in the archipelago, including sites not surveyed using LIT. Over 200km² of coral reef is believed to have been damaged or destroyed.

Car Nicobar Island is surrounded by fringing reefs. Prior to the tsunami the average coral cover in Sawai Bay (22 in Fig. 1) was 72.7 %, dominated by Acropora spp. and with some large patches of Millepora dichotoma and Porites nigrescens. The reef profile was a drop to a depth of 4 meters and then a gradual slope to 8 meters over a distance of 60 meters, with the deeper sections dominated by Porites colonies.

The tsunami has all but wiped out the shallow reefs in Sawai Bay, with a reduction of coral cover to less than 5 % and an increase in rubble to almost 60 % of the benthic cover (Fig. 2). The impact was similar on shallow and deeper reefs on the east coast. The damage was restricted to 10 meters in the northern bay and to 25 meters on the east coast, with rubble observed in deeper waters on the eastern coast comprising mainly broken Acropora and Porites colonies. Debris deposited on the reef included wooden logs and tyres on the northern section, mainly of household materials, window panes and logs in the east.

Some shallow reefs on the eastern side of Car Nicobar survived the tsunami as the orientation of the bay sheltered them from the path of direct as well as refracted tsunami waves. These reefs are now showing signs of recovery from the damage sustained. However, the predominantly sub-massive and slow-growing corals such as Mussids, Faviids and Porites are now facing competition from soft corals. While Mussids and Faviids seem to be resisting this competition with some success, Porites appears not to be. However, a positive sign is the observation of settlement of juvenile corals on the reefs in Sawai Bay.

The islands of Central Nicobar comprise Nancowry, Camorta, Katchall and Trinket, while Southern Nicobar includes Great Nicobar, Little Nicobar and a few lesser islands. No pre-tsunami data was available for these areas, and due to cyclonic storms and turbulent seas during the post-tsunami surveys methods were limited to rapid assessment and visual estimates of the status of reefs. Results presented below are largely qualitative and indicative in nature and further quantitative studies of the area is recommended.

Live coral cover on the northeastern side of Camorta Island (23 in Fig. 1) was estimated to be
around 40%. Reefs in this area are dominated by *Porites*, *Millepora*, *Acropora*, *Pocillopora*, *Stylopora*, and *Hydnophora*. Physical damage to branching colonies of *Acropora* and *Pocillopora* was observed and evident in the coral rubble, and a few colonies of dead massive and sub-massive *Porites* were seen. This is likely to be an effect of the sedimentation that followed the earthquake and tsunami. An estimated 30% of live coral cover was seen on the southeastern side (24 in Fig. 1) of Camorta, with *Porites* dominating.

The reefs on the eastern side of Katchall Island (25 in Fig. 1) had a live coral cover of c. 40%, mainly comprising *Porites*, *Millepora*, *Heliopora Acropora* and *Seriatopora*. Dead massive *Porites* colonies and topped sub-massive colonies were observed, implying a direct impact of the tsunami waves.

Trinket Island residents reported that 10-12 m high tsunami waves had covered this entire island, causing major damage to the surrounding coral reefs. This was evident from the widespread distribution of rubble on the western side (26 in Fig. 1) dominated by fragmented *Acropora*, interspersed with small patches of live coral.

On Great Nicobar Island, the southernmost island of the Nicobar Group, the western side of Campbell Bay (27 in Fig. 1) was surveyed. The average coral cover was 10% dominated by *Porites*.

Pigeon Island is situated on the north-eastern side of Great Nicobar Island. Surveys on the eastern side of the island (28 in Fig. 1) indicate an average coral cover of 20%. Toppled colonies of massive and sub-massive *Porites* were seen.

Little Nicobar Island has a mountainous terrain and few beaches. Tsunami waves have ploughed into the mountainous terrain destroying vegetation all along the shore. The island has subsided by 2-3 m and land up to 50 m from the previous high tide mark has been engulfed by the sea. Coral reefs have been destroyed by direct wave impact as evidenced by the large amounts of coral rubble under water. Sand and silt deposition on dead corals was observed. Surviving coral species include *Heliopora* and *Millepora*, and *Acropora* and *Heliopora* showed signs of regeneration. Coral cover of approximately 30% was seen on the southeastern side of the island (29 in Fig. 1).

Menchal Island, on the southeastern side of Little Nicobar, is uninhabited and is covered by coconut and banana plantations belonging to the Nicobarese of Little Nicobar. Surveys of the western side of the island (30 in Fig. 1) indicated a coral cover of 20%.

Kondul Island is situated to the north of Great Nicobar Island. The island was inhabited by Nicobarese prior to the tsunami, but as the entire coastline has been destroyed the island has become unsuitable for habitation and people of this island have now permanently shifted to Great Nicobar Island. On the southeastern side of the island (31 in Fig. 1) the average coral cover was 10%. Large colonies of dead *Acropora clathrata* were seen.

**DISCUSSION AND CONCLUSIONS**

The impact of the tsunami differed between the Andaman and Nicobar groups of island. In the Andaman group, coral reefs in channels between small islands were most affected, while in the Nicobar group reefs all around the islands except those sheltered between islands have been affected. As a result of the tectonic activity, the northern islands of the Andaman group have been raised, causing the death of shallow coral reefs due to permanent exposure. Where coral reefs have been uplifted, shallow water reefs have been affected due to altered wave action and localized warming of stagnant sea water. Coral reef destruction due to the tsunami was restricted to shallow areas (up to a 5-meter depth) in the Andaman Islands. In the Nicobar Islands, destruction of coral reefs occurred up to a depth of 20 meters. Severe damage was caused by the impact of the tsunami and in particular the increased sedimentation that followed. Due to the subsidence of the Islands, changed beach profiles and the monsoon, erosion and sedimentation continued for more than 8 months.

Regeneration patterns also differed between the Andaman and Nicobar Islands. New settlement of
corals were recorded at several sites in the Andaman Islands but was negligible in the southern group of the Nicobar Islands. In some parts of the Nicobar Islands, hard corals are facing competition from soft corals.

Coral reef research priorities in the archipelago include: resilience of coral communities to changes in habitats and environmental conditions, coral regeneration patterns, emerging coral community structure, and implications for resource species and dependent communities. There is also a need for documentation of regeneration patterns in coral reef communities that have been completely destroyed. Management and conservation initiatives need to mitigate sedimentation effects caused by altered land-use patterns, deforestation and post-tsunami rehabilitation activities, building on available scientific research. There is also a need to create awareness among local communities, including Nicobari tribes, about the destruction of coral reefs and implications for their livelihoods.

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Assessment of Tsunami Impacts on the Marine Environment of the Seychelles

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ABSTRACT

The tsunami that hit the Seychelles islands on 26 December 2004 had traveled approximately 5000 km from the epicenter, offshore Sumatra, in less than seven hours, and had a wave-height of 2.5-4 m at first landfall. This study was conducted as part of the official UNEP Tsunami damage assessment to affected countries of the Indian Ocean, in February 2005. Two major patterns in coral reef damage were noted, dependent on the geographic location of each island, direction of exposure at each site, and reef substrate. The northern islands clustered around Praslin (including Curieuse, La Digue, Felicite and the rocks of Isle Coco and St. Pierre) showed very high levels of damage (approaching 100%) on carbonate reef substrates. By contrast, sites around Mahe showed much lower levels of impact, generally below 10%, due to the shelter provided by the outer northern islands and dissipation of wave energy as the tsunami traveled over the greater distance of shallow water from the outer edge of the banks to Mahe. Granitic surfaces and reefs suffered little damage due to their density and hardness. On solid carbonate reef surfaces attached corals showed little breakage and mechanical damage or overturning. However the majority of true coral reef sites in the granitic islands have a reef framework that was only loosely consolidated due to coral mortality during the 1998 El Niño and subsequent bioerosion. This reef matrix was not robust enough to resist the tsunami waves, either from direct impact of the force of water, or movement of rubble and rocks. In these areas significant reef rubble was moved by the wave and consequently associated live coral colonies were also displaced and damaged. Thus > 50% of substrate damage and >25% of direct damage to corals in northern and eastern-facing carbonate framework sites was recorded, <10% damage in shallow carbonate substrate sites in central, western and southern locations, and < 1% damage on all granitic substrate sites. Coral reefs are very important to the economy, society and infrastructure of the Seychelles – all the damaged northern sites are prime tourist locations for the country, and the most highly damaged terrestrial locations are adjacent to degraded reef areas. Though impacts from the tsunami was less than other threats, such as coral bleaching, they highlight the differential vulnerability of different locations and the need to implement strong measures for reef and coastal conservation.

INTRODUCTION

Background

The Seychelles comprises 115 islands covering a land area of 455 km² in the western Indian Ocean, between...
Forty-one islands comprise the inner granitic group of mountainous islands, within a radius of 50 km from the main island Mahe, with Mahe, Praslin and La Digue being the largest and most important for towns and settlement. The outer islands are all coralline and built of old reef carbonate growth, and rise to only a few meters above sea level.

The tsunami
The tsunami wave that hit the Seychelles islands on 26 December 2004 had traveled approximately 5000 km from the epicenter, offshore Sumatra, in less than seven hours. At 13.00 hours waves ranging from 2.5m to 4m in height hit the east coast of Praslin, La Digue and Mahé islands. The effects were felt all along the east coast of Mahé, propagating over a 30 minute period. Refracted waves hit the west coast of Praslin and Mahé 30 minutes to 1 hour after the respective east coasts were hit. Another wave occurred at 17.00 hours, followed by two smaller waves at 22.00 hours and 05.00 hours (on 27 December). The second wave had more or less the same effect as the first because, although smaller, it occurred at high tide. The two smaller waves caused damage only on the west coast of Praslin. The surges caused by the waves flooded the low lying areas of Mahé, Praslin and La Digue and caused widespread damage to beaches, coastal vegetation, roads, bridges, other infrastructure and houses. The flooding continued for a period of about 6 hours. Two people lost their lives.

The tsunami was followed on 27 December 2004 by extreme weather with rainfall reaching 250 mm in the northern and central areas of Mahé. Torrential rains continued for several days. Runoff from the hills formed virtual rivers that swept across the country, causing widespread landslides, tree and rock fall in the northern and central part of Mahé and in other areas, with associated further damage to infrastructure, dwellings and the vegetation on slopes. The rainfall caused more widespread damage to land areas of the Seychelles, thus hampered immediate mitigation and focus on tsunami impacts. Together, these almost simultaneous incidents caused serious damage to the infrastructure of the Seychelles.
The coastlines of the Seychelles granitic islands are of two types: 1) granitic rock which is often steeply sloping or accidented with large boulders and rockfalls; 2) coralline coastlines backed by flat coastal plains and fronted by fringing coral reefs (of either old or recent construction). On the coralline coastlines, the fringing reef crests break waves sufficiently to enable the coastal plains to form from sediment accumulation. Between the reef crest and shoreline, sheltered lagoons may be present, backed by fine sand beaches. Channels in the fringing reefs allow the passage of water in and out of the lagoons with the tides. The flat land, calm lagoons and ocean access of the coralline shorelines have attracted settlement and development, supporting a large proportion of agriculture, urban development and tourism of the Seychelles.

Fringing coral reefs around the central granitic islands have allowed the development of fine sand beaches and lagoons, and settlement of the sheltered coastal plains behind them. Channels through the fringing reefs provide access to the ocean from safe harbours for artisanal fishing boats and pleasure craft. The same coral reefs provide the primary infrastructure that supports Seychelles’ tourism industry, providing beaches and sites for snorkeling and SCUBA diving. Due to stresses from development and overfishing, and then the mass coral bleaching of

Figure 2. Map of central-southern section of Mahe Island showing areas of maximum terrestrial impacts at Anse Royale and Anse la Mouche channels in the fringing reefs (reefs shown in blue) approach closest to land. Map source: Seychelles Ministry of Environment.
coral reefs in the Indian Ocean in 1998 that caused 80-90% mortality of corals, many of the granitic islands’ coral reefs are significantly degraded.

**Terrestrial Impacts**

Impacts of tsunami waves are strongly affected by the shape and bathymetry of reefs and channels to the open sea – reef crests, complex coral reef surfaces and granitic shorelines absorb and dissipate the wave energy, while deep channels allow focusing of the waves closer to land and lead to higher wave heights. The importance of the coral reefs is shown by the locations of major terrestrial and coastline damage, and the influence of these marine ecosystems on shoreline vulnerability (Fig. 2). The major locations of terrestrial damage, at Anse Royale and Anse Mouche, on Mahe, and to the seawall in Curieuse Marine Park, are located on fringing reef coastlines. Significantly, shoreline damage was focused where deep channels lead through or up to the fringing reefs, focusing and amplifying the wave energy to these points. Thus the combined shelter and ocean access that have allowed coastal development just above the high tide line adjacent to fringing reefs contributed to the high vulnerability of these to the tsunami. This vulnerability will also extend to other wave- and storm-related threats, and intensification of these threats through sea-level rise and changes in storm patterns.

**METHODOLOGY**

As part of the UNEP fact-finding mission, the IUCN Global Marine Program and CORDIO were requested to assess the tsunami impacts on the marine environments of the Seychelles, undertaken from February 3rd-13th, 2005. The study included stakeholder consultation and site visits to eight of the inner islands of the Seychelles (Fig. 1). It was not possible in the time available to include outer atoll islands.

Two survey methods were used. The first was developed as a rapid assessment tool by the SCMRT- MPA in conjunction with the Marine Unit in the Department of Environment, and conducted by staff and rangers at MPA and other sites on Mahe, Praslin and Curieuse (SCMRT 2005 a,b,c). Four observers conducted approximately 10-minute samples, each assigned some of 7 coral taxa/groups (*Acropora*, other branching corals, foliose, massive, encrusting, fungids and soft corals). Colonies were recorded as damaged or undamaged (broken or overturned) along with general observations on the status of the reef. Colonies completely missing due to wave damage were not possible to differentiate using this method. Because the main coral reef areas in the Seychelles were significantly affected by high mortality in 1998 and had weak eroded frameworks at the time of the tsunami, it is likely that many coral heads were completely removed from the study sites, and thus not recorded. Surveys were conducted in 3 periods covering Mahé (30 December 2004), Curieuse (5 February) and Praslin (5 February).

The second method used was an ICRI/ISRS (International Coral Reef Initiative/International Society for Reef Studies) methodology for assessment of damage from the tsunami to coral reefs, developed during January 2005 (ICRI/ISRS 2005), using a 0-5 semi-quantitative scale (Table 1). This method recorded a broader variety of variables including damage to live corals, damage to the substrate and debris from the terrestrial environment. The method is based on samples of 10m² areas of the bottom, selected haphazardly during swims across the sample area.

Table 1. Classes used for estimation of benthic cover and the incidence / abundance of tsunami damage indicators. Based on Australian Institute of Marine Science long term monitoring programme and English et al. 1997.

<table>
<thead>
<tr>
<th>Class</th>
<th>Range (%)</th>
<th>Desc</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0</td>
<td>None</td>
</tr>
<tr>
<td>1</td>
<td>1-10</td>
<td>Low</td>
</tr>
<tr>
<td>2</td>
<td>11-30</td>
<td>Medium</td>
</tr>
<tr>
<td>3</td>
<td>31-50</td>
<td>Common</td>
</tr>
<tr>
<td>4</td>
<td>51-75</td>
<td>High</td>
</tr>
<tr>
<td>5</td>
<td>76-100</td>
<td>Extreme</td>
</tr>
</tbody>
</table>
Details of the method can be obtained from the authors and the UNEP website (http://www.unep-wcmc.org/latestnews/emergency/tsunami_2004/coral_ass.htm).

**RESULTS**

In general, the extent of the damage caused by the tsunami will mainly depend on the slope and topography of the seabed. On gradually sloping shorelines, the energy of the wave appears to build up, sucking water away from the shore, followed by powerful flooding waves and surge transporting vast amounts of water and unconsolidated rubble. Direct damage of the tidal waves results from the massive water flows and associated kinetic energy while indirect effects include sediment deposition and land-based pollution (nutrients, pesticides, industrial and urban chemicals, biological material) brought by the backwash. Increases in turbidity and organic carbon, as a result of this pollution, may result in oxygen depletion, potentially detrimental to fish, corals and seagrasses.

**Primary Impacts to Coral Reefs**

Coral reef damage in the inner Seychelles islands was limited principally to physical breakage due to the tsunami waves, surge and, potentially, backwash. Damage was documented to reef substrates, mobilization of sand and rubble, and damage to live corals. Limited damage from siltation and debris was noted, and no evidence of coral diseases or other effects of pathogens or pollutants was seen. Types of damage are summarized below. The assessment focuses on damage to coral reef habitats, but also mentions associated habitats and species.

**Mechanical damage to corals (Fig. 3)**

Mechanical damage to corals was documented as breakage of branches and overturning. *Acropora* and *Pocillopora*, being the two main genera of branching corals on Seychelles’ reefs were the most frequently observed to be damaged. *Pocillopora* occurs as individual heads up to 30 cm in diameter, and damage was observed as broken branches off a parent colony, and loose branches in the rubble. By the end of these surveys (on February 12, some 48 days or 7 weeks after the tsunami), most broken sections of Pocillopora had not fully healed with incomplete tissue growth over the break. *Acropora* was present as
individual colonies and as fields or thickets of staghorn morphologies. The former suffered breaks similar to *Pocillopora*, while for the latter a field could be entirely flattened, with scattered branches in the rubble in all directions, or in a consistent direction.

**Mechanical damage to substrate (Fig. 4)**

No damage occurred on any granite substrates, nor was there any clear indication of movement of granite rocks and boulders larger than 50 cm or so. Carbonate reef substrates showed considerable signs of damage. In areas of hard old reef framework, minor damage was noted by the presence of scars where rocks and perhaps corals were torn off, but the intensity of damage was low and restricted to areas shallower than 50 cm (e.g. Anse Royale). Coral reefs that were healthy before the 1998 coral bleaching event but suffered high mortality have shown only partial recovery since then, with the result that the reef framework is mostly made up of loosely consolidated coral skeletons and branches. Just before the tsunami, these had a varying degree of live coral attached to the reef or growing on loose rubble pieces of different sizes. These reefs showed severe physical damage by the tsunami waves with widespread rubble, loose rocks, overturned corals and eroded craters showing evidence of movement. Without definitive data before the tsunami it was hard to determine absolute levels of mechanical damage to substrates, however in general it appeared high, and in some cases (e.g. the northerly-exposed sites of I. Coco and St. Pierre) rubble movement and total damage may have been as high as 100%. Before the tsunami many of these areas had low coral cover so damage to coral was minor; however damage to the reef matrix was very high.

**Movement of substrate (Fig. 5)**

Movement of loose rocks and rubble was a major factor in exacerbating damage to reef substrates and to corals. Granite rocks were too dense and rounded, and showed no evidence of having been moved. Carbonate rocks were extensively moved, in all sizes from small rubble, through large dead *Acropora* tables to massive *Porites* heads over 1 m across. The low density of carbonate skeletons and the often irregular shape of rubble fragments contributed to their ease of movement by waves. In some cases, massive *Porites* heads 2 m in diameter and greater were toppled, though this was likely due to sediment movement, another form of damage described below. In some areas, such as Grand Anse, Curieuse, whole areas of the bottom looked whiter (observation by MPA ranger, Paul Lavigne), due to the overturning of rubble revealing their whiter undersides (with darker algal growth on the upper surfaces).
The tsunami waves, compounded by heavy rainfall and rough seas in the following week, mobilized extensive amounts of marine and terrestrial sediment. Missing sediment was commonly noted in many reef habitats, where old rubble that had likely been buried in sediment for many years was exposed. These areas were distinguishable because they lacked mature algal communities of filamentous, turf or coralline algae. At distinct reef channels, such as in Baie Ternay, erosion of sediment from the channel edges was noted, up to an estimated 70 cm of sediment lost.

Silt deposition on rock surfaces was noted, in layers up to 2-3 mm thick on surfaces that often had a cover of thin algal filaments. However, because of the time since the tsunami waves, it is possible that more silt had built up because of subsequent factors, or some had been lost. Heavy sedimentation on a seagrass bed was also noted (see above). Interestingly, high siltation was noted only for white carbonate silt, not darker terrigenous soil, suggesting little impact of the heavy rains following the tsunami.

**Curieuse Wall and Mangrove Forest**

Extensive damage was done to the causeway/wall enclosing a shallow lagoon previously used for turtle farming, and a mangrove forest area in the Marine Park at Curieuse Island. The mangrove forest developed over the approximately 100 years that the causeway has been in place, and is one of the largest in...
the Seychelles, containing 7 of the 9 species found in the islands. More than one half of the wall was knocked inwards by the tsunami waves, with the principal damage occurring where a channel leads up close to the wall and from this point east to the Park HQ beach. At the time of this study, no damage had been noted to the mangrove forest, as it is sheltered from the winds of the northwest monsoon. However, a wide channel on the beach and near shore was created by the large volume of draining water. This may further develop into erosion of the leading edge of the mangroves, with consequent loss of habitat area and species. The loss of mangrove forests could have major consequences on local marine biodiversity as these areas provide habitats for many juvenile and adult crustacean and fish species. The mangrove forest is one of the primary attractions for visitors to Curieuse Marine Park, accessed using a boardwalk that was also damaged by the tsunami. Without repair to the wall, further damage to the boardwalk will occur during the southeast monsoon and the combined loss of mangroves and boardwalk may significantly reduce financial income from the Marine Park, which subsidizes other protected areas that cannot support themselves.

Site Damage Summaries
Northern islands, north-east exposure
Several sites were surveyed around Curieuse Island, in the Marine Park, including Grand Anse, Baie Launay and sites to the north and west. Overall 8.1% of coral colonies showed signs of tsunami damage and extensive rubble movements were noted on shorelines facing east, south and north. On a deeper site at 8 m east of Curieuse (Coral Gardens), many massive corals were overturned and exposed due to their eroded bases and weak framework. Many live coral colonies (*Acropora*, *Pocillopora* and *Tubipora*) were washed up on the beach. Other damage included broken *Acropora* stands in Resort bay, and damaged turtle nests (see later section).

The coral reef at I. Coco was the farthest-east reef surveyed, and faced directly the path of the oncoming tsunami. St. Pierre is more sheltered, but both sites share a morphology of exposed granite rocks on their seaward side, and an extensive development of reef corals and carbonate framework in the shallows and in the lee of the islands. In both areas, corals on granite substrates showed little damage. However the reef frameworks of dead staghorn *Acropora* exhibited a near-total devastation. Signs of damage included: mobile rubble pieces and broken coral fragments, the accumulation of large amounts of carbonate rubble in drifts up the sides of granite boulders and in depressions, loose dead *Acropora* tables (their large surface area making them easy to move) and craters/depressions in the branching framework where back and forth movement of such pieces by the waves caused erosion of circular depressions. There were also erosion gullies through the framework where large sections of rubble framework may have been transported to deeper water. Damage to the reef framework was consistently estimated at > 50%. Corals close to the bottom on granitic surfaces showed evidence of breakage, likely due to rubble movement along the bottom.

The bay at La Reserve/Anse Petit Cours is west-facing. It was surveyed for two reasons: first, the shoreline and hotel suffered extensive damage, and second, this reef area suffered some of the lowest mortality of coral during 1998. Reef structure is slightly similar to Baie Ternay, with an extensive area of shallows leading out from the beach, and a sloping reef with high coral cover to a sand base at 6 m leading into deeper water. The island shoreline leading west from the bay is steeply sloping, with a fringe of coral growth at 1-10 m depth. Coral diversity was observed to be higher than other locations. Extensive rubble damage was found in the shallows, and because of the higher abundance and diversity of corals, higher levels of breakage of live coral. In particular, flattened areas of staghorn *Acropora* were common (e.g. *A. austera*), and damaged stands of the extensive columnar growth forms of *Goniopora*. Because of the sloping sand base, many *Porites* colonies in waters > 6 m were toppled, due most likely to erosion of sand
from under one side and tumbling of the colony/boulder.

**Mahe, north-east exposure**
Among all sites on Mahe, damage to coral reefs was highest at Anse Cimetiere with at least 27% of colonies showing signs of physical and mechanical damage. The damage to this site is likely underestimated as most of the coral colonies that were damaged were completely destroyed and therefore were not included in the sampling methodology employed by SCMRT. Historical data of this site show that the reef slope has experienced an 80% reduction of coral cover as a result of the tsunami, from 20% to < 5%.

The coral reef of Baie Ternay Marine National Park, on the northwest tip of Mahe Island, was among the most damaged sites on Mahe (Fig. 7), and illustrated the different types of damage (above) based on habitat and depth. It is a highly enclosed bay, with a reef crest dividing the inner seagrass/beach area from the outer deep bay, the reef crest being just below the surface and reef growth down to 8-10 m. Damage to corals was negligible below about 3 m, but > 10% of vulnerable branching corals at the reef crest were broken. A large proportion of the reef crest is dead branching corals from 1998, however the sheltered bay has enabled complete consolidation of branches and rubble by coralline algae, which prevented re-breaking of the framework by the tsunami. See ‘seagrass’ section, below, for a description of sedimentation impacts, and beaches around the east and west boundaries of Baie Ternay were built up by sand deposition. Overall, compared to long term damage caused by coral bleaching during the El Niño of 1998, damage from the tsunami event was minor.

The reef at Anse Royale is an old carbonate platform dominated by fleshy algae (*Sargassum, Turbinaria*) due to its highly exposed position to waves from the east and long term degradation from coastal land use. Tsunami damage was surveyed from a depth of 6 m, but was limited to the shallowest 50 cm at the reef crest where scars on the framework show where rocks (or perhaps corals) were ripped off.

**Mahé, south-west exposure**
Damage to granite reefs at the southern-most point of Mahe and in Port Launay was negligible. In the large bay of Anse la Mouche/Anse Copra corals grow in typical sheltered backreef areas, dominated by opportunistic species on eroding substrates in the shallows, and deeper reefs dominated by large massive corals. The area is impacted by eutrophication from land and overfishing, with large sea urchin populations. Damage was patchy, with some areas showing no damage. However in some locations on the deeper reefs below 5 m staghorn *Acropora* coral heads were broken by the waves. In the shallows large massive corals were toppled as their bases are highly bioeroded and likely also be sediment displacement from underneath. In the shallows small branching corals were completely undamaged. Overall, coral damage was less than 5%. A layer of sediment appeared to have been removed from the reef, with extensive fields of fine rubble visible in the channels between coral heads.
Damage to Seagrass Beds

Damage to seagrass beds in the Seychelles was low, with only one definite case of damage recorded at Baie Ternay Marine Park (above). Suspension of sediment and erosion of the reef channel resulted in the burial and smothering of the shallow seagrass area between the reef crest and beach inside the bay. Some of the seagrass areas appeared to be recovering as the excess sediment is being removed by normal tidal and wave action, exposing smothered seagrasses (though some still living) and dead pen shells. Mortality of pen shells (*Pinna* sp.) living in the seagrass beds was high, with many of the shells now exposed 1-2 cm above the substrate. This may indicate a minimum depth of newly deposited sand, and (unsuccessful) attempts by the bivalves to burrow upwards to avoid smothering. At the boundary between seagrass beds and the channel, undercutting of the seagrass bed and exposure of roots occurred.

Marine Turtles

The impact of the tsunami on nesting sea turtles in the Seychelles seems to have been relatively minor and what impact there was appears to have been restricted to the inner islands. No obvious damage to nesting beaches was reported from any of the following sites in the outer islands (pers. comm. Jeanne Mortimer): Aldabra (Terence Mahoune), Farquhar
atoll (Antonio “Mazarin” Constance), and D’Arros/St. Joseph (Jean-Claude Camille; pers. obs., J.A. Mortimer). Bird Island reported “large tides” but no apparent damage to any monitored turtle nests (Margaret Norah). Aride Island reported two nests destroyed by the tsunami (Dylan Evans). Within the Marine Parks, no apparent damage was reported on the beaches of Ste. Anne Island (Jude Bijoux), but at Curieuse nests were lost at Anse Cimitiere but not at the most important nesting beach Grand Anse (Alain Cedras). At Curieuse, erosion at Grande Anse is the norm at this time of year, but the problem appears to have been exacerbated by the tsunami. At Intendence beach on Mahe no nest damage was recorded (Anders Dimblad).

DISCUSSION

Two major patterns in coral reef damage were noted, controlled by the geographic location of each island and exposure direction of each site, and reef substrate. The northern islands clustered around Praslin (including Curieuse, La Digue, Felicite and the rocks of Isle Coco and St. Pierre) showed very high levels of damage (approaching 100%) on carbonate reef substrates. By contrast, sites around Mahe showed much lower levels of impact. The limited damage on Mahe is due to the shelter provided by the outer northern islands, and energy dissipation of the tsunami traveling over the greater distance of shallow water from the outer edge of the banks to Mahe.

Granitic reefs suffered less damage than reefs with a calcium carbonate substrate (Fig. 8). Granitic surfaces were either immovable as they form the bedrock of the islands, or in the case of boulders and rocks, are too dense and of a compact shape to be displaced by the force of the tsunami. Even on carbonate rock surfaces that were consolidated and firm, attached corals showed little breakage and mechanical damage or overturning. However the majority of true ‘coral reef’ sites in the granitic islands have a reef framework that is loosely consolidated due to mortality during the 1998 El Niño and subsequent bioerosion. This reef matrix was not robust enough to resist the tsunami waves, either from direct impact of the force of water, or movement of rubble and rocks. In these areas significant reef rubble was moved by the wave and consequently associated live coral colonies were also displaced and damaged. We documented > 50% of substrate damage and >25% of direct damage to corals in northern and eastern-facing carbonate framework sites), <10% damage in shallow carbonate substrate sites in central, western and southern locations, and < 1% damage on all granitic substrate sites. Given the importance of coral reefs to the economy and social structure of the Seychelles (e.g. all the damaged northern sites are prime tourist locations for the country) this provides a strong threat to the country and requires action for mitigation.

An important correlation between coral reef location (coastal geomorphology) and shoreline damage was noted. Most damage to shorelines occurred where fringing reefs and bays with extensive coral development occur – e.g. Anse Petit Cours (Praslin), the causeway (Curieuse), Anse Royale (Mahe) and Anse la Mouche (Mahe). At these locations, development immediately above the high tide line was made possible by the protection offered by fringing reefs. However the reefs offered only limited protection from a wave the size of the tsunami, and maximum damage occurred where reef channels cut in closest to land (the causeway at Curieuse, Anse Royale, Anse la Mouche). Thus the vulnerability of the low coastal plains to wave damage was clearly shown by the tsunami. While fringing coral reefs protect these shorelines during regular conditions, their protection was limited during this extreme event.

Along with the high vulnerability of coral reefs in the northern islands, the Curieuse Marine Park suffered damage to its infrastructure (UNEP 2005). The wall protecting the mangrove forest and artificial lagoon was damaged, which will expose the high-diversity mangrove stand to erosion during the southeast monsoon. Additionally, infrastructure of the MPA was damaged, including boat engines, electrical equipment and physical facilities on land.

Tsunami damage to coral reefs in the Seychelles was severe on the northern carbonate-framework reefs,
but minor elsewhere. These damages, occurring while reefs were still recovering from 80-90% mortality of corals during 1998, point to a critical vulnerability of the coral reefs of the Seychelles. The El Niño in 1998 created extensive rubble fields from death and breakage of the fast growing branching corals (*Acropora* and *Pocillopora*) that dominated the shallow waters of Seychelles reefs (Jennings et al. 2000). At the time of the tsunami, the primary reef carbonate frameworks in the granitic islands were relatively weak physical structures, consisting of attached and loose calcium carbonate pieces of varying sizes. These may become strongly consolidated by coralline algae growth over 5-10 years under good conditions (e.g. observation from Baie Ternay). The chemical and biological consolidation into a rigid reef framework, such as that found on some fringing reef sites (e.g. Anse Royale) may take hundreds to thousands of years. Only 6 years after the bleaching, the loosely consolidated reef frameworks were not able to resist the force of the tsunami, resulting in severe movement of rubble and breakage.

In the short to medium term, any mitigation activities will have to deal with the problem of loose reef frameworks and the long time needed for reef matrix consolidation, in order to promote coral reef recovery and growth. In the medium to long term, damage from the tsunami should be considered in the context of Seychelles as a Small Island Developing State. As such, it has a particular vulnerability to shocks and threats due to its small size, from natural disasters to economic and global political influences. While damage from the tsunami was not catastrophic to coral reefs, it significantly worsened the catastrophic impact of coral bleaching 6 years previously, with impacts focused on the most vulnerable, and most valuable, coral reef areas. On these reefs, the tsunami set back biological recovery of corals by 6 years. Because of the extensive physical damage to the reef matrices, however, the set back to overall reef recovery may be much longer than that.

The interaction of these two types of threats in the medium to long term will be particularly important for the Seychelles – physical exposure to extreme waves events, and their increasing severity due to climate change – rising sea level, northwards migration of the cyclone belt in the southern Indian Ocean, and increasing severity and frequency of major storms. While the occurrence of another tsunami cannot be predicted, the increasing severity of the threat from waves to the Seychelles is clear. Broad principles reflecting the importance of coral reefs to Seychelles were used to develop recommendations for mitigation:

I. Improve capacity for assessment of coastal health and vulnerability to waves and storms, using on bathymetry, coastal topography and coral reef status;

II. Improve watershed management that minimizes downstream and marine impacts of water use and treatment, to maximize the recovery potential of coral reefs impacted from multiple threats including eutrophication, overfishing and coral bleaching;

III. Integrate ICZM and Marine Protected Area management frameworks covering all of Seychelles’ coastal and EEZ waters, recognizing the coastal protection benefits of healthy coral reef ecosystems.

As part of the UNEP assessment mission, a number of short to long term recommendations were made to respond to the tsunami damage. Briefly, these included:

1. Mitigation of tsunami damage and enhancement of coral reef recovery - rehabilitation and restoration technologies for coral reefs are in their infancy, but studies should be initiated to address the key factors of substrate stability, water quality improvements to enhance coral survival and enhancement of natural recruitment and survival of small corals.

2. Assess and replace lost infrastructure of Curieuse Island Marine Park, in particular to protect the mangrove forest.

3. Development of coral reef and environmental monitoring capacity at SCMRT-MPA and strengthening of the Seychelles Coral Reef Network to ensure complementarity among
monitoring programmes in the Seychelles.

4. Development of a shoreline vulnerability model and planning capacity.

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Part 4 – Biological Research

The Effects of Habitat on Coral Resistance and Resilience to Bleaching

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ABSTRACT

This study examines the bleaching responses of scleractinian corals at four sites in Kenya (Kanamai, Vipingo, Mombasa Marine Park and Nyali) representing two distinct lagoon habitats (relatively shallow and relatively deep). Bleaching responses were monitored for the general coral community and zooxanthella densities and chlorophyll levels were monitored for target species (Pocillopora damicornis, Porites lutea and Porites cylindrica) during a non-bleaching year (2006) and a year of mild bleaching (2007). The objective of this study is to determine whether corals in different habitats display varying resistance and resilience to bleaching stress and to indicate which environmental characteristics are responsible for the variation in response. Considerable differences in bleaching responses between shallower and deeper lagoon sites were observed, with shallower sites Kanamai and Vipingo exhibiting much lower bleaching and paling incidence than deeper sites Nyali and Mombasa Marine Park. These shallower lagoons display much more fluctuating thermal and light environments than the deeper sites, with higher

maxima, lower minima, higher standard deviations and higher diel variation. These results suggest that corals in the shallower lagoons have acclimatized and/or adapted to the fluctuating environmental conditions they endure on a daily basis and have become more resistant to bleaching stress. Furthermore, in deeper sites that did exhibit significant bleaching (Mombasa Marine Park and Nyali), it was found that coral recovery occurred more quickly in the protected area compared to the non-protected area.

INTRODUCTION

Climatic changes are causing increased frequency of coral bleaching events worldwide. Coral bleaching events are usually associated with above-average water temperatures and high irradiance levels that cause the disruption of the obligate symbiosis between corals and micro-algal zooxanthellae. The zooxanthellae are expelled and the coral is left in a weakened state that can eventually lead to mortality of colonies and even a phase shift from a coral-dominated to an algal-dominated reef and a subsequent loss of biodiversity.

However, it is possible for corals to either resist conditions that cause coral bleaching or to recover from bleaching events (resilience). Resistance can be defined as the ability of an ecosystem or individual to withstand disturbance without undergoing a phase shift or losing neither structure nor function (Odum, 1989). Resilience can be defined as the ability of a system or an individual to absorb or recover from disturbance and change, while maintaining its functions (Carpenter et al., 2001). Several ecological and environmental factors can affect a coral reef’s resistance and resilience to bleaching events. These include the coral reef’s community species composition, herbivory levels, zooxanthellae population, temperature and irradiance history, connectivity to other reefs, water movement, and shading or screening properties (Grimsditch and Salm, 2006; Hoegh-Guldberg, 1999; West and Salm, 2003).

Furthermore, it has been shown that it is possible for colonies to acclimatize to increased temperatures and high irradiance levels so that they are able to resist bleaching events when they occur. Acclimatization can be defined as the ability of an organism to undergo phenotypic changes in response to stress in the natural environment that result in the readjustment of the organism’s tolerance levels to that stress (Coles and Brown, 2003). Threshold temperatures that induce coral bleaching-related mortality vary worldwide (from 27°C in Easter Island; Wellington et al., 2001, to 36°C in the Arabian Gulf; Riegl, 1999) according to the maximum water temperatures that are normal in the area, implying a capacity of corals and/or zooxanthellae to survive higher temperatures for a certain period of time. Furthermore, corals that are regularly exposed to stressful environmental conditions have been shown to acclimatize and exhibit physiological tolerance to elevated temperatures and UV-radiation that exceed normal thresholds (Brown et al., 2000; Brown et al., 2002a; Brown et al., 2002b, Coles and Brown, 2003).

In order to combat the worst effects of climate change and to conserve this valuable ecosystem, it is important to determine which factors affect coral reef bleaching resistance and resilience and to apply this knowledge in management plans. Although bleaching events cannot be prevented by managers, by implementing appropriate management responses it is possible to help a coral reef recover from bleaching and to mitigate the worst effects.

This study examines the bleaching responses of corals at four sites (Nyali, Mombasa Marine Park, Kanamai and Vipingo, Fig. 1) representing two distinct lagoonal habitats on the Kenyan coast (deeper and shallow lagoons). The sites were characterized using environmental parameters such as depth, water flow, light and temperature. Bleaching responses were monitored for the general coral community and zooxanthellae densities and chlorophyll levels were

Figure 1. Map of study sites.
monitored for target species (*Pocillopora damicornis, Porites lutea* and *Porites cylindrica*) during a non-bleaching year (2006) and a mild bleaching year (2007). The objective of this study is thus to determine whether corals in different habitats display different bleaching responses (i.e. resistance and resilience) and to suggest which environmental characteristics are responsible for the variation in response.

**MATERIALS AND METHODS**

Five large and healthy coral colonies for each target species (*Pocillopora damicornis, Porites cylindrica* and *Porites lutea*) were mapped at two shallow lagoons (Kanamai and Vipingo, 0.4m depth and 0.6m at low tide respectively) and two deeper lagoons (Mombasa Marine Park and Nyali, 1.4m and 1.8m depth at low tide respectively) along the Kenyan coast (Fig. 1). Target species were chosen according to general abundance and bleaching susceptibility, with one highly susceptible (*Pocillopora damicornis*), one moderate (*Porites cylindrica*) and one tolerant (*Porites lutea*) target species. Coral size class and line intercept transect data were also collected for two fixed 25x2m transects at each site.

The bleaching season in Kenya occurs during the late northeast monsoon from mid March when doldrum conditions become most intense to late April when the first storms of the southeast monsoon induce cooling of surface waters. In the two years of the study, sampling began before bleaching during the northeast monsoon (February and early March), continued during bleaching season of the northeast monsoon (late March and April) and finished during the recovery period of the southeast monsoon (May, June and July).

Coral fragments were collected from the mapped colonies using a chisel. The fragments were transported to the laboratory submerged in seawater in small plastic bottles and were held in an aerated seawater tank. Coral tissue was removed from the skeleton using a water jet. Seawater was then added to the tissue to make up a practical volume, usually 500 ml. The tissue slurry was homogenised and 1.0 ml of homogenate was loaded into a Sedgwick-Rafter chamber. Using a compound binocular microscope and a magnification of x400, the number of zooxanthellae in 10 random quadrats was recorded. The counting chamber was then reloaded and another 10 random quadrats were counted. The area of the coral skeleton was measured using the foil method; the area/weight ratio of aluminium foil was determined and coral skeletons were then wrapped in this foil. The foil was then trimmed to fit the skeleton area and weighed. Using the weight of the trimmed foil and the area/weight ration for the foil, the skeleton area could be determined.

To measure chlorophyll *a* levels, a known volume of the sample was filtered through a GF/F filter paper and the filtrate was dissolved in 10ml of 90% acetone. It was then centrifuged for 10 minutes at 2000rpm and the chlorophyll absorbance was read using a spectrophotometer.

If it was observed, bleaching incidence was recorded in the water using a random swim method. This consists of noting the size class, genus and bleached/pale/dead percentages of each bleached colony in 20 haphazard 2m² areas at each site.

Water temperature was monitored using underwater temperature loggers that were left at each site during the entire duration of the study. Light was monitored using a light meter for only 24-hour periods on the same day at each site to avoid theft and overgrowth of the light meter. Water flow was measured using a clod card method (McClanahan et al, 2005). Plaster-of-Paris clod cards were made in an ice-cube tray, dried and weighed before being deployed in the field in small net bags that were tied to the substrate. They were collected 24 hours later, dried and weighed again in order to determine how much had dissolved.
The shallow lagoons Kanamai and Vipingo displayed the most extreme temperature conditions, with higher standard deviations (±1.9°C and ±1.5°C respectively), higher maximum temperatures (36.2°C and 33.8°C respectively), lower minimum temperatures (24.0°C and 23.7°C respectively) and bigger differences between diurnal and nocturnal temperatures (+0.9°C and +0.4°C respectively) than the deeper lagoons. Between the two, Kanamai displayed the more extreme fluctuations, as well as the highest average temperature overall (27.8°C) (Fig. 2).

The deeper lagoons of MMP and Nyali displayed less pronounced temperature variations with lower standard deviations (both ±1.4°C) as well as smaller

### RESULTS

#### Temperature

The shallow lagoons Kanamai and Vipingo displayed the most extreme temperature conditions, with higher standard deviations (±1.9°C and ±1.5°C respectively), higher maximum temperatures (36.2°C and 33.8°C respectively), lower minimum temperatures (24.0°C and 23.7°C respectively) and bigger differences between diurnal and nocturnal temperatures (+0.9°C and +0.4°C respectively) than the deeper lagoons. Between the two, Kanamai displayed the more extreme fluctuations, as well as the highest average temperature overall (27.8°C) (Fig. 2).

The deeper lagoons of MMP and Nyali displayed less pronounced temperature variations with lower standard deviations (both ±1.4°C) as well as smaller
maximum-minimum (+8.9°C and +8.1°C respectively) and diurnal-nocturnal (both +0.2°C) differences than the shallower lagoons. However, the deeper lagoons displayed bigger temperature increases from the 2006 to 2007 northeast monsoon bleaching seasons, with a +0.9°C increase in Nyali and a +0.8°C increase in MMP compared to a +0.7°C increase in both Kanamai and Vipingo.

**Light**
As expected, the shallow lagoons Kanamai and Vipingo also displayed more extreme light regimes than the deeper lagoon MMP. Unfortunately, light data for Nyali was not available due to technical problems. Kanamai (4913±14,315 lux) and Vipingo (4758±10,489 lux) both displayed much higher average light levels, higher standard deviations and higher maxima (70,846 lux and 110,223 lux respectively) than MMP (average of 518±879 lux, maximum of 5167 lux) (Fig. 3). Although light data is unavailable for Nyali, it can be assumed that the light regime is similar (with perhaps even lower averages) to MMP due to the depth and sediment levels of the site.

**Water Flow**
There was no clear trend between water flow at deeper and shallower lagoons. MMP consistently displayed the highest average water flow during both neap (6.4 cm/s) and spring tide cycles (9.0 cm/s). Kanamai (5.4 cm/s) displayed similar water flow to Nyali (5.5 cm/s) and Vipingo (5.6 cm/s) during neap tide and, although unfortunately data for spring tide at Nyali is unavailable, it appears that MMP stands out as the site displaying highest water flow. Overall, Kanamai displayed the lowest water flow during both neap and spring tides (8.1 cm/s) (Fig. 4).

**Substrate Cover**
MMP and Kanamai displayed higher hard coral cover (28.4% and 23.0% respectively) while Nyali and Vipingo displayed lower hard coral cover (14.7% and 10.6% respectively), showing no trend with depth. Macroalgal cover was higher in deeper sites MMP (25.5%) and Nyali (21.0%) than in shallower sites Vipingo (8.3%) and Kanamai (3.7%) (Fig. 5).

**Coral Cover**
All sites are dominated by massive and branching *Porites* colonies. 90.5% of MMP, 80.3% of Kanamai, 59.9% of Nyali and 42.2% of Vipingo coral cover was massive and branching *Porites*. Vipingo (10.2%) and Nyali (8.3%) displayed the highest relative coral cover of the more bleaching-susceptible genera *Acropora* and *Pocillopora*. *Acropora* and *Pocillopora* only accounted for 3.7% of Kanamai’s coral cover and 0.7% of MMP’s coral cover (Fig. 6).
Bleaching was first observed during late April of 2007. The shallow lagoons Kanamai and Vipingo exhibited lower levels of combined paling, bleaching and mortality (4.4% and 6.3% of total coral area respectively) than the deeper lagoons Nyali and MMP (35.1% and 26.9% of total coral area respectively). During the recovery period, MMP still exhibited relatively high bleaching incidence (9.8%) but low mortality (0.9%). Nyali exhibited the highest mortality (11.7%) and also relatively high bleaching (4.5%). In July, mortality increased to 13.1% in Nyali and 2% in MMP, but bleaching incidence in MMP decreased to 6.0%, indicating recovery (Fig. 7). Overall, Pocillopora and Porites accounted for the highest proportion of bleached colonies, but differed greatly in their mortality rates. In MMP, Nyali and Vipingo, Pocillopora accounted for 37%, 47.1% and 33.3% of bleached colonies respectively. In Kanamai, MMP and Nyali, Porites accounted for 47.2%, 28.3% and 13.8% of bleached colonies respectively (Fig. 8). However, during the recovery period of June and July, Porites experienced low bleaching-related mortality while Pocillopora experienced high mortality. Porites colonies only experienced mortality in Vipingo, where they accounted for 12.5% of total bleaching-related mortality.

**Figure 6.** Percentage of coral area covered by each genus.

**Bleaching and Mortality Incidence**

Bleaching was first observed during late April of 2007. The shallow lagoons Kanamai and Vipingo exhibited lower levels of combined paling, bleaching and mortality (4.4% and 6.3% of total coral area respectively) than the deeper lagoons Nyali and MMP (35.1% and 26.9% of total coral area respectively). During the recovery period, MMP still exhibited relatively high bleaching incidence (9.8%) but low mortality (0.9%). Nyali exhibited the highest mortality (11.7%) and also relatively high bleaching (4.5%). In July, mortality increased to 13.1% in Nyali and 2% in MMP, but bleaching incidence in MMP decreased to 6.0%, indicating recovery (Fig. 7). Overall, Pocillopora and Porites accounted for the highest proportion of bleached colonies, but differed greatly in their mortality rates. In MMP, Nyali and Vipingo, Pocillopora accounted for 37%, 47.1% and 33.3% of bleached colonies respectively. In Kanamai, MMP and Nyali, Porites accounted for 47.2%, 28.3% and 13.8% of bleached colonies respectively (Fig. 8). However, during the recovery period of June and July, Porites experienced low bleaching-related mortality while Pocillopora experienced high mortality. Porites colonies only experienced mortality in Vipingo, where they accounted for 12.5% of total bleaching-related mortality.

**Figure 7.** Bleaching, paling and mortality incidence during 2007.
mortality, whereas *Pocillopora* accounted for the highest proportion of bleaching-related mortality in all sites (from 37.5% in Vipingo to 90% in MMP; Fig. 9). Furthermore, of the three species only *Pocillopora damicornis* suffered significant mortality in both 2006 (50% overall mortality) and 2007 (47% overall mortality) with colonies in Kanamai experiencing the lowest mortality rate in both years.

**Zooxanthellae Densities**

Average zooxanthellae densities for marked colonies (*Pocillopora damicornis, Porites lutea* and *Porites cylindrica*) were higher for all species at all sites in 2006 than in 2007. Average zooxanthellae densities were also higher in shallower sites Kanamai and

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*Figure 8.* Proportion of bleached corals by genera.

*Figure 9.* Proportion of corals exhibiting bleaching-related mortality by genera.

*Figure 10.* Average zooxanthellae densities for mapped corals (*Pocillopora damicornis, Porites lutea* and *Porites cylindrica*) during 2006 and 2007.
Vipingo that in deeper sites MMP and Nyali for all species. In general, Kanamai exhibited the highest average zooxanthellae densities while MMP displayed the lowest. *Porites lutea* colonies exhibited the highest zooxanthellae densities while *Pocillopora damicornis* displayed the lowest densities, with *Porites cylindrica* in between the two (Fig. 10).

In 2006, colonies in MMP, Nyali and Kanamai displayed similar zooxanthellae density trends in most species with decreases in the warm months of February and March, and recovery in the following months as temperature decreased. Colonies in Vipingo, on the other hand, exhibited increasing densities for all species during February and March followed by decreases from April to June. In 2007, average zooxanthellae densities in *Pocillopora damicornis* remained very low (<1x10^6 cells cm⁻²) in all sites with recovering densities in July. *Porites lutea* and *Porites cylindrica* colonies showed a general increasing trend in average zooxanthellae densities from March to July (Fig. 10).

**Chlorophyll a Levels**

Chlorophyll *a* concentrations were generally higher at shallower sites Kanamai and Vipingo than at deeper sites Nyali and MMP, and Kanamai exhibited by far the highest chlorophyll *a* concentrations while MMP exhibited the lowest. *Porites cylindrica* consistently exhibited the highest chlorophyll *a* concentrations at all sites, while *Pocillopora damicornis* consistently displayed the lowest concentrations, with *Porites lutea* in between the two (Fig. 11).

Chlorophyll *a* concentrations in *Pocillopora damicornis* colonies generally decreased progressively at all sites from March to July as colonies bleached and failed to recover. *Porites lutea* and *Porites cylindrica* colonies in the shallow site Kanamai displayed peaks during March and then decreasing...
concentrations to July. In MMP and Nyali, the concentration peaks for these two species were not as pronounced and generally occurred in May or June (Fig. 11).

**DISCUSSION AND CONCLUSION**

Considerable differences in bleaching responses during 2007 between shallow and deeper lagoonal sites were observed, with shallower sites Kanamai and Vipingo exhibiting much lower bleaching and paling incidence than deeper sites Nyali and MMP in both haphazard sampling and marked *Pocillopora damicornis* colonies. These results suggest that corals in the shallower lagoons have acclimatized to the more extreme environmental conditions they endure on a daily basis and have become more resistant to thermal stress than corals in the deeper lagoons.

Shallow sites Kanamai and Vipingo displayed more extreme temperature and light regimes than deeper sites Nyali and MMP with higher maxima, lower minima, larger standard deviations and larger diel variations. The smaller volumes of water in the shallower lagoons probably explain the difference in temperature and light regimes: shallower lagoons absorb but also lose heat more quickly and attenuate less light than in the deeper lagoons.

Moreover, differences in temperature and light regimes could explain differences in bleaching responses between similar-depth sites. Kanamai is a shallower site than Vipingo, displaying higher average temperatures, standard deviation in temperatures, difference between maximum and minimum temperature, diel temperature variation, maximum light level and standard deviation in light levels. This may explain why Kanamai also displayed lower bleaching incidence and bleaching-related mortality than Vipingo. Similarly, MMP is a shallower site than Nyali, displaying higher average temperatures and difference between maximum and minimum temperature. Nyali experienced increased warming during the 2007 northeast monsoon bleaching period (+0.9°C to 2006) compared to MMP (+0.8°C to 2006). These factors could explain why Nyali exhibited higher bleaching incidence than MMP.

On the other hand, water flow did not correlate well with depth and bleaching incidence, and although corals in Nyali suffered much higher bleaching incidence than Kanamai and Vipingo, all these sites displayed very similar water flow velocities. It therefore appears that light and temperature histories were more influential than water flow in determining the bleaching responses of corals.

Moreover, average zooxanthellae densities and chlorophyll *a* concentrations are higher in Kanamai and Vipingo than in Nyali and MMP, with highest densities in Kanamai. Studies in the region have shown that corals with higher zooxanthellae densities are more resistant to bleaching (Grimsditch et al, 2007), a hypothesis that is further confirmed by these results. In addition, *Porites lutea* displayed the highest densities and was the most tolerant to bleaching, while *Pocillopora damicornis* exhibited the lowest densities and was the most susceptible to bleaching. Zooxanthellae population dynamics could thus also partly explain bleaching responses at different sites.

Interestingly, species that suffered less bleaching (*Porites lutea* and *Porites cylindrica*) did not exhibit decreases in chlorophyll *a* concentrations through time, and actually exhibited increases in chlorophyll *a* during bleaching and recovery months in some cases. However, *Pocillopora damicornis*, which was more susceptible to bleaching, displayed decreasing chlorophyll *a* concentrations with time, indicating that bleaching events cause disruptions in pigments as well as symbionts.

The genus most affected by bleaching at Nyali, MMP and Vipingo was *Pocillopora*, and it accounted for most of the bleaching-related mortality in all sites. *Porites* also made up significant proportions of bleached corals in MMP, Nyali and Kanamai. However, *Porites* colonies hardly exhibited any bleaching-related mortality, showing that although this genus does bleach and pale it is tolerant to bleaching. Despite their susceptibility to bleaching, *Acropora* corals form a very small proportion of bleached corals in most sites because they are scarce, except for in Vipingo where they form a larger
proportion of the total coral area and subsequently also a larger proportion of bleached corals.

All sites were dominated by massive and branching Porites corals, discounting the possibility that large variations in coral community compositions determined bleaching responses. Bleaching-susceptible genera such as Pocillopora and Acropora did not necessarily form a larger proportion of the coral population in sites most affected by bleaching. The highest area percentage covered by these genera occurs in Vipingo, which is a shallow site exhibiting low bleaching incidence. However, differences in community composition could partially explain varying bleaching responses between similar-depth sites. Nyali exhibited higher bleaching incidence than MMP and also a higher proportion of area covered by bleaching-susceptible genera. The same applies to Vipingo compared to Kanamai.

During the recovery period, colonies in Nyali displayed higher mortality than those in MMP, while a higher proportion of corals in MMP remained bleached but did not die. It thus seems that colonies in Nyali were being overgrown by algae more quickly than those in MMP, which remained in a bleached state longer. This could be due to the protected status of MMP and the possibility that herbivory levels are higher than in Nyali. In fact, populations of important herbivores such as parrotfish and surgeonfish have been shown to be significantly higher in protected areas than in non-protected areas along the East African coast (McClanahan and Arthur, 2001).

In conclusion, the data from this study show that during the 2007 bleaching season, corals in the shallow lagoons of Kanamai and Vipingo were more resistant to bleaching stress than corals in the deeper lagoons of MMP and Nyali, probably due to a history of higher light and temperature extremes and variation at the shallower sites. Interesting differences were also found between the deeper lagoons at Nyali and MMP – on the one hand MMP reefs experience slightly larger diel variation in temperature, and on the other they experienced less warming in 2007. Whether this acclimatization is due to short-term phenotypic changes or due to a longer-term process of adaptation through natural selection is a question that could be further researched. Pocillopora was the genus most susceptible to bleaching-related mortality, especially at Nyali, and Porites branching, massive and columnar all displayed high tolerance to bleaching. Among all these patterns of bleaching susceptibility, it is not clear whether the acclimatization is host-based or symbiont-based, and monitoring of the zooxanthellae clades in corals at these sites could help in answering this question. Finally, bleached corals in MMP suffered lower levels of mortality than in Nyali, possibly associated with higher rates of herbivory due to MMP’s protected status. While herbivory is often related to resilience of reefs and the influence of herbivores assisting recovery by coral recruits, this result suggests herbivory may also mediate interactions between algae and bleached corals, and assist in the survival of bleached corals.

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Spawning Patterns of *Acropora* Species in the Mombasa Lagoon in Kenya

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*keywords: coral, sexual reproduction, spawning, equatorial reefs*

**ABSTRACT**

Seventeen species of *Acropora* were studied in the Mombasa lagoon in Kenya from 2004-2005 to determine the timing and frequency of spawning, and the level of synchrony within and among different species. Spawning in this genus occurred over an extended 7 month period from October-April, with the majority of marked colonies (77%) releasing gametes between January-March in the northeast monsoon. Individual species were capable of split-spawning over 2-4 lunar months, and the length of spawning varied between species. Spawning occurred over a range of lunar phases, but predominantly in the 2 weeks following the full moon. It is evident that *Acropora* species in Kenya display a greater degree of spawning asynchrony compared to similar assemblages in other parts of the world.

**INTRODUCTION**

Sexual reproduction of scleractinian corals has been fairly well documented in the Pacific, Atlantic and the Red Sea and studies have revealed a wide variety of reproductive patterns among species and geographic locations (reviewed by Harrison & Wallace 1990). Mass coral spawning events that occur annually on the central Great Barrier Reef (GBR) (Harrison et al. 1984; Babcock et al. 1986) contrast markedly with corals in the northern Red Sea, where spawning occurs in different seasons, months or at different lunar phases despite a high degree of synchrony within individual species (Shlesinger & Loya 1985; Shlesinger et al. 1998).

Only a few detailed studies have been done on coral reproduction in the Western Indian Ocean (WIO). *Pocillopora verrucosa* was found to be a simultaneous hermaphrodite that spawned gametes annually between March and April in the Maldives (Sier & Olive 1994) and around the new moon in January in South Africa (Kruger & Schleyer 1998). Synchronised spawning was also recorded in six soft coral species in South Africa (Schleyer et al. 1997), and an unusual form of asexual reproduction was described in *Goniopora stokesi* at Aldabra Atoll in the Seychelles (Rosen & Taylor 1969).

Some unpublished data and *in situ* observations of spawning are also known in the region. Daytime spawning was observed in more than 100 individuals of *Fungia danai* between 09:00-10:00 hrs in February 2006 (5 days after full moon) in the Chagos Archipelago (Mangubhai et al. 2007). Individuals
appeared to be males, releasing a cloud of sperm in short repeated bursts lasting for a few seconds, creating a distinct cloud along the reef. Spawning slicks were observed in March 1997 on Ari Atoll in the Maldives (Loch et al. 2002), and Acropora species were observed spawning in October 1994 on Chumbe Island reef in Tanzania (Franklin et al. 1998) and during full moon in November 1998 in the lagoon of Albion in Mauritius (R. Moothien Pillay, pers. comm.). A short study at Misali Island in Tanzania undertaken from December 2002-March 2003 found mature gametes present in 84% of A. tenuis and 7% of A. valida samples in February, 1-2 days prior to the full moon (A. Pharoah, unpubl. data).

Given the paucity of available information on coral reproduction in the WIO, a study was undertaken in 2003-2005 to provide data on the timing and patterns of scleractinian coral reproduction on Kenyan reefs. Results on the broad patterns in the timing of spawning of faviid and Acropora species in the first year were published in Mangubhai & Harrison (2006). This paper presents data on spawning patterns in Acropora species from the second study year, where the timing of spawning was narrowed to a finer level, and examines the level of synchrony within and among different species.

METHODS

A total of 209 colonies, comprising of 17 species, were marked and sampled at Kijembe and Mamba Reefs in the Mombasa lagoon (4º4'S, 39º43'E) over a 9-month period from September 2004–May 2005. A single branch measuring 5-10 cm was removed from marked colonies during each collection period, and the timing and frequency of collections was based on spawning periods recorded in the previous year for individual Acropora species (Mangubhai & Harrison 2006).

Reproductive condition of oocytes was assessed in the field as follows: ‘mature’ if oocytes were pigmented (Fig.1), ‘immature’ if oocytes were smaller and pale or white, and ‘empty’ if oocytes were too small to view with the naked eye (Oliver et al. 1988; Mangubhai & Harrison 2006). In the laboratory, tissue samples were fixed in 10% formalin in seawater, decalcified in weak HCl (0.5-5%) over a 2-3 week period and then preserved in 70% ethanol. Decalcified branch samples were placed on a dark-coloured petri dish and five polyps were randomly selected from the centre of the branch and removed under a dissecting microscope, and the presence or absence of gametes was recorded. Spawning was inferred from the disappearance of mature gametes between sampling periods, and the absence of brooded planulae or embryos.

Branches of 9 species (Acropora sp.1, A. divaricata, A. humilis, A. mossambica, A. samoensis, A. secale, A. sordiensis, A. tenuis and A. valida) were monitored in separate aquaria for 3-4 days from December 2004–March 2005. Coral branches were examined every half hour from 18:00 (just prior to sunset) to 23:00 hrs or until spawning ceased, using a torch covered with red cellophane to avoid causing stress to the corals which may cause colonies to release oocytes prematurely (P. Harrison, pers. comm.). Information was recorded on the timing of ‘setting’, (i.e. when the egg-sperm bundles are formed and then held in the oral cavity prior to release), the onset and duration of spawning and the method of gamete release. Where spawning in aquaria occurred, field samples were collected from the same colony the next day to confirm spawning in the field.
All Acropora species studied in Kenya were simultaneous hermaphrodites, with both mature oocytes and spermaries observed together in polyps prior to gamete release (Fig. 2).

Each polyp had eight gravid mesenteries with mature spermaries (two large and two small pairs) and oocytes (two long and two short strings) that develop on separate mesenteries, and this pattern was consistent in all polyps of Acropora species studied. Oocytes were generally oval though some became irregular in shape depending on their pawning was not observed directly in the field, observations of spawning in aquaria of 7 species of Acropora and the disappearance of mature oocytes and spermaries from consecutive samples taken a few days to a few weeks apart, indicates that these species are broadcast spawners. Furthermore, no planulae were recorded in any of the >1000 tissue samples examined.

Spawning
Mature pigmented oocytes were observed in the field on average 2-3 weeks prior to spawning, though in some colonies mature oocytes were observed for 4-6 weeks prior to spawning. Pink and orange were the most common oocyte colour observed at maturity, though not all oocytes within a colony became pigmented prior to spawning (Table 1). Cerise-coloured oocytes were less common and were only recorded in A. tenuis and A. subulata. Mature white or pale oocytes were recorded in 11 (64.7%) of the 17 species, and in 28 (13.4%) of the 209 colonies studied. Oocyte colour was generally inconsistent among different colonies within species, and therefore did not provide a useful tool for identifying morphologically similar species.

Spawning in Acropora species was asynchronous, and extended over 7 months from October–April, with the majority of colonies (77%) releasing gametes between the summer months of January–March in the northeast monsoon (Fig. 3). The main spawning

<table>
<thead>
<tr>
<th>Species</th>
<th>n</th>
<th>Colour</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acropora sp.1</td>
<td>17</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. digitifera</td>
<td>3</td>
<td>o</td>
</tr>
<tr>
<td>A. divaricata</td>
<td>33</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. gemmifera</td>
<td>5</td>
<td>p</td>
</tr>
<tr>
<td>A. humilis</td>
<td>17</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. kosurini</td>
<td>1</td>
<td>o, p</td>
</tr>
<tr>
<td>A. lutkeni</td>
<td>2</td>
<td>o, p</td>
</tr>
<tr>
<td>A. mossambica</td>
<td>23</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. nana</td>
<td>3</td>
<td>p</td>
</tr>
<tr>
<td>A. nasuta</td>
<td>6</td>
<td>o, w</td>
</tr>
<tr>
<td>A. samoensis</td>
<td>6</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. secale</td>
<td>9</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. sordiensis</td>
<td>20</td>
<td>o, p, w</td>
</tr>
<tr>
<td>A. subulata</td>
<td>12</td>
<td>o, p, w, c</td>
</tr>
<tr>
<td>A. tenuis</td>
<td>20</td>
<td>p, c</td>
</tr>
<tr>
<td>A. valida</td>
<td>25</td>
<td>o, p, w</td>
</tr>
</tbody>
</table>

Figure 2. Arrangement of mature oocytes (o) and spermaries (s) on mesenteries in a polyp of Acropora valida.

RESULTS
Reproductive Pattern
All Acropora species studied in Kenya were simultaneous hermaphrodites, with both mature oocytes and spermaries observed together in polyps prior to gamete release (Fig. 2).

Table 1. The colours of eggs observed in reproductively mature Acropora species. n = number of colonies sampled. Colours: w = white, o = orange, p = pink, c = cerise.
occurred in February with 42% of colonies, comprising 11 species, releasing gametes in this lunar month. In species represented by >5 colonies spawning was split over 2-4 months. Partial spawning in individual colonies was less common in *Acropora* compared to *faviid* species (unpubl. data), and was recorded in 1 *A. humilis*, 1 *A. valida* and 2 *A. tenuis* colonies (Fig. 3).

The length of the spawning period varied between species, with spawning occurring for shorter periods of 2 months in *A. subulata*, *A. divaricata* and *A. humilis* and longer periods of 4 months in *A. valida*, *A. sordiensis* and *A. mossambica* (Fig. 3). In species represented by ≥ 10 colonies, the main spawning period was in January, February or March, when 43-80% (mean = 51%) of the population of each species released gametes. No species had their main spawning period in the inter-monsoonal month of April, prior to the onset of the southeast monsoon season.

Data showing the lunar phases over which coral spawning occurred, for those colonies with spawning periods inferred from samples taken over 3 days or less, are shown in Table 2. Spawning in *Acropora* species in Kenya occurred over a range of lunar phases, but predominantly in the 3rd and 4th lunar quarters, that is, in the 2 weeks following full moon (Table 2). While *A. tenuis* spawned mainly in the 3rd and 4th lunar quarters, a small number of colonies spawned in the 1st/2nd quarter. *Acropora divaricata* and *A. subulata*...
spawned predominantly in the 1st lunar quarter, around new moon. Coral spawning was observed in 6 of the 9 Acropora species held in aquaria (Fig. 4). Three of the branches held in aquaria released gametes on the same night as the colonies in the field from which they were removed, and 7 branches released gametes in aquaria 1-2 weeks earlier than their corresponding colonies in the field (Table 3). In most branches, the setting of egg-sperm bundles under the oral cavity commenced between 19:10-19:45 hrs, with spawning commencing between 21:13-21:56 hrs. Acropora tenuis commenced setting around sunset, and spawned between 19:02-19:17 hrs. In Acropora sp.1 and A. valida setting

Table 2. The lunar phase when the last sample was observed (lunar day - mature) and the subsequent sample when gametes were absent (lunar day - absent) from coral colonies containing mature gametes. Data are shown for colonies with spawning periods inferred from samples taken over 3 days or less. Lunar days are divided into four lunar quarters: 1st = 0-7, 2nd = 8-14; 3rd = 15-21, 4th = 22-29, where lunar day 0 = new moon and lunar day 15 = full moon.

<table>
<thead>
<tr>
<th>Species</th>
<th>n</th>
<th>Lunar Day</th>
<th></th>
<th>Lunar quarter</th>
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</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mature</td>
<td>Absent</td>
<td></td>
</tr>
<tr>
<td>A. tenuis</td>
<td>1</td>
<td>18</td>
<td>21</td>
<td>3rd</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>24</td>
<td>27</td>
<td>4th</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>21</td>
<td>24</td>
<td>4th</td>
</tr>
<tr>
<td>A. valida</td>
<td>8</td>
<td>21</td>
<td>22</td>
<td>3rd</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>21</td>
<td>24</td>
<td>4th</td>
</tr>
<tr>
<td>Acropora sp.1</td>
<td>9</td>
<td>21</td>
<td>22</td>
<td>3rd</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>21</td>
<td>24</td>
<td>4th</td>
</tr>
<tr>
<td>A. secale</td>
<td>1</td>
<td>19</td>
<td>21</td>
<td>3rd</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>21</td>
<td>22</td>
<td>3rd</td>
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<tr>
<td></td>
<td>1</td>
<td>29</td>
<td>2</td>
<td>4th/1st</td>
</tr>
<tr>
<td>A. sordiensis</td>
<td>1</td>
<td>17</td>
<td>20</td>
<td>3rd</td>
</tr>
<tr>
<td>A. mossambica</td>
<td>6</td>
<td>21</td>
<td>24</td>
<td>4th</td>
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<tr>
<td></td>
<td>4</td>
<td>21</td>
<td>24</td>
<td>4th</td>
</tr>
<tr>
<td>A. divaricata</td>
<td>3</td>
<td>4</td>
<td>7</td>
<td>1st</td>
</tr>
<tr>
<td>A. humilis</td>
<td>3</td>
<td>21</td>
<td>24</td>
<td>4th</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>24</td>
<td>27</td>
<td>4th</td>
</tr>
<tr>
<td>A. gemmifera</td>
<td>1</td>
<td>3</td>
<td>5</td>
<td>1st</td>
</tr>
</tbody>
</table>

Figure 5. Spawned gametes of Acropora valida. Note the lighter pink areas are where sperm clouds are breaking up from the oocyte-sperm bundle.
commenced at 19:10 hrs, and spawning occurred between 20:55-21:00 hrs. Compact positively buoyant egg-sperm bundles were extruded slowly through the mouth (Type I spawning behaviour, Babcock et al. 1986), and floated immediately to the water surface where water movement generated by the airstone caused the bundles to slowly break apart, separating individual oocytes and sperm clouds (Fig. 5). All egg-sperm bundles were released rapidly from the branches, within 15-19 minutes of the commencement of spawning. Similar patterns of gamete released were recorded in A. sordiensis, A. divaricata, A. tenuis and A. humilis.

**DISCUSSION**

The 17 Acropora species studied in the Mombasa lagoon in 2004-2005 were simultaneous hermaphrodites and broadcast spawners. This sexual pattern and mode of reproduction and development predominate in Acropora (subgenus Acropora), and are consistent with studies from other regions (reviewed by Harrison and Wallace 1990).

In Kenya, Acropora species spawned over a 7-month period between October–April and interspecific spawning synchrony was less synchronous in Acropora species compared to faviids (Mangubhai

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### Table 3: The timing of setting and spawning in branches collected from marked colonies and held in aquaria between January–March 2005. Type: natural = spawning confirmed for the same night in colonies in the field; premature = branches in aquaria released gametes, but colonies in the field still had mature gametes; ‘-’: no samples were taken in the field to confirm spawning.

<table>
<thead>
<tr>
<th>Species</th>
<th>Date</th>
<th>Lunar day</th>
<th>Type</th>
<th>Start setting</th>
<th>End setting</th>
<th>Start spawn</th>
<th>End spawn</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>A. sordiensis</em></td>
<td>5-Jan-05</td>
<td>24</td>
<td>-</td>
<td>19:30</td>
<td>20:54</td>
<td>21:13</td>
<td>21:27</td>
</tr>
<tr>
<td><em>A. tenuis</em></td>
<td>7-Jan-05</td>
<td>26</td>
<td>-</td>
<td>19:30</td>
<td>21:16</td>
<td>21:22</td>
<td>21:40</td>
</tr>
<tr>
<td><em>A. sordiensis</em></td>
<td>10-Jan-05</td>
<td>0</td>
<td>premature</td>
<td>19:35</td>
<td>20:43</td>
<td>21:24</td>
<td>21:33</td>
</tr>
<tr>
<td><em>A. sordiensis</em></td>
<td>11-Jan-05</td>
<td>1</td>
<td>premature</td>
<td>17:47</td>
<td>21:08</td>
<td>21:35</td>
<td>21:38</td>
</tr>
<tr>
<td><em>A. sordiensis</em></td>
<td>12-Jan-05</td>
<td>2</td>
<td>premature</td>
<td>19:20</td>
<td>21:00</td>
<td>21:28</td>
<td>21:35</td>
</tr>
<tr>
<td><em>A. sordiensis</em></td>
<td>17-Jan-05</td>
<td>7</td>
<td>-</td>
<td>19:30</td>
<td>21:16</td>
<td>21:22</td>
<td>21:40</td>
</tr>
<tr>
<td><em>A. sordiensis</em></td>
<td>18-Jan-05</td>
<td>8</td>
<td>-</td>
<td>20:30</td>
<td>21:20</td>
<td>21:30</td>
<td>21:30</td>
</tr>
<tr>
<td><em>A. tenuis</em></td>
<td>21-Jan-05</td>
<td>11</td>
<td>premature</td>
<td>17:45</td>
<td>18:50</td>
<td>19:02</td>
<td>19:17</td>
</tr>
<tr>
<td><em>A. humilis</em></td>
<td>22-Jan-05</td>
<td>17</td>
<td>premature</td>
<td>19:45</td>
<td>22:00</td>
<td>20:04</td>
<td>20:14</td>
</tr>
<tr>
<td><em>Acropora sp.1</em></td>
<td>2-Mar-05</td>
<td>21</td>
<td>natural</td>
<td>19:10</td>
<td>20:38</td>
<td>20:55</td>
<td>21:05</td>
</tr>
<tr>
<td><em>A. valida</em></td>
<td>2-Mar-05</td>
<td>21</td>
<td>natural</td>
<td>19:10</td>
<td>20:38</td>
<td>21:00</td>
<td>21:19</td>
</tr>
<tr>
<td><em>A. valida</em></td>
<td>2-Mar-05</td>
<td>21</td>
<td>natural</td>
<td>19:10</td>
<td>20:38</td>
<td>20:55</td>
<td>21:05</td>
</tr>
</tbody>
</table>
& Harrison 2006), a pattern which is found on both equatorial (Oliver et al. 1988) and higher-latitude reefs (Hayashibara et al. 1993) in some locations. The spawning pattern emerging for Acropora species in lagoonal reefs in Kenya is one of asynchrony, and contrasts markedly with the central GBR and western Australia where high intra- and inter-specific spawning synchrony results in an annual mass spawning event where many species spawn over one or a few nights after full moon periods in the October/November (late spring/early summer) and March (summer), respectively (Harrison et al. 1984; Babcock et al. 1986, 1994). The results from this study also contrast with studies from Japan, Palau and Singapore, which display more synchronised multispecific spawning of Acropora assemblages (Hayashibara et al. 1993; Penland et al. 2004; Guest et al. 2005).

The spawning patterns recorded in Kenya share similarities with Acropora species on subtropical reefs in the Solitary Islands, Australia, where spawning was staggered among species and among colonies within some species, and occurred over 2-5 months from December to April, and over a range of lunar phases (Wilson & Harrison 2003). It was suggested that highly variable sea surface temperatures in the Solitary Islands, particularly during gamete maturation, may explain the extended and asynchronous spawning pattern observed in this location (Wilson & Harrison 2003). Sea surface temperatures in Kenya are generally less variable during maturation, but are more variable during the peak spawning period from January–April, with differences of 2.5°C recorded (see Fig. 5 in Mangubhai & Harrison 2006). The variability in sea surface temperatures is likely to be a result of the shallow depth of the lagoon, which makes it susceptible to temperature fluctuations associated with cloud cover and prevailing wind conditions. However, temperature variability in Kenya during peak spawning periods are still less than those at the Solitary Islands where 6-8.5 ºC differences can occur during the peak spawning period from December-March (A. Scott, pers. comm.).

The spawning patterns recorded in Kenya also share some similarities with scleractinian corals in the northern Gulf of Eilat in the Red Sea, which exhibit ‘temporal reproductive isolation’, where spawning is separated in time (seasons, months and lunar phases) for different species (Shlesinger & Loya 1985; Shlesinger et al. 1998). There are, however, a number of notable differences between the two regions. Firstly, in Kenya a smaller percentage of corals (~51%) participate in the main spawning month compared to the Red Sea (90-100%, Shlesinger et al. 1998), which means that intraspecific spawning is more asynchronous in Kenya. Secondly, Acropora species in Kenya may spawn in the month(s) immediately preceding and/or following the main spawning event, so that gamete release can occur over 2-5 consecutive lunar months in individual species. Broadcast spawning species in the Red Sea release gametes over 1 or 2 consecutive months (Shlesinger & Loya 1985; Shlesinger et al. 1998). Thirdly, in the Red Sea, species that spawn in the same lunar month will spawn in different lunar quarters from each other, which reduces the overlap between spawning species (see Table 1 and Fig. 1, in Shlesinger & Loya 1985). In Kenya, there is overlap between species during all lunar quarters, and colonies within some species may spawn over >1 lunar quarter. However, it is important to note that spawning times have only been described in 6 (Shlesinger and Loya 1985; Shlesinger et al. 1998) of the approximately 42 known species of Acropora in the Red Sea (Wallace 1999), and therefore there is a high probability that there would be overlap in spawning between some species in different lunar quarters. It is evident that Kenyan Acropora species display a greater degree of spawning asynchrony than in other regions. The level of reproductive plasticity recorded in this genus (cf. faviids), such as that recorded by this study, may explain the highly polymorphic nature of Acropora species, with high rates of hybridisation recorded between both similar and in some cases dissimilar morphological forms (Willis et al. 1997), their success in colonising reefs, and their dominance in most tropical reef habitats (Wallace 1999).
ACKNOWLEDGMENTS

This study was supported by CORDIO East Africa and the Kenya Wildlife Service. I would like to thank my supervisor P. Harrison for his guidance and support throughout my PhD research, and D. Obura for assisting and supporting my field work in Kenya. Hazel Pargeter helped with coral dissections and Juma Suleiman with laboratory work. This work was undertaken under Kenyan research permit (MOEST 13/001/32C-15).

REFERENCES


Spatial and Temporal Variation in Coral Recruitment and Mortality in Coastal Kenya

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ABSTRACT

Measuring recruitment patterns and mortality of corals is important for understanding mechanisms that regulate their populations and mediate species coexistence. However, there is limited data on coral recruitment dynamics in Eastern Africa and much of the WIO. We studied spatial and temporal patterns of coral recruitment and mortality in four lagoonal reefs in Kenya. The objectives of the study were to compare coral recruit densities and juvenile mortality between sites, months, seasons and years. Twelve 1m² permanent quadrats were sampled for the variables at each site on a monthly basis from February 2006 to June 2007. Recruit density in the protected Mombasa Marine Park was significantly higher (7.45 recruits/m²) than the other sites that are not protected. Recruit density was higher in SEM (Southeast Monsoon) than in NEM (Northeast Monsoon) seasons in both years with 2006 having higher recruitment than 2007. A total of 16 genera were recorded with Mombasa Marine Park having the highest number of genera (13) while a non protected site Kanamai had a significantly lower density (3.52 recruits/m²) with a low genera number (8). Other non-protected sites (Nyali and Vipingo) had intermediate recruit densities. Dominating genera were Favia, Porites, Favites, Pocillopora and Pavona in their order of overall abundance. Coral genera exhibited site specific abundance and mortality rates with Pocillopora having high abundance in Nyali (3.46 recruits/m²) and high mortality rate in Vipingo (85%). Benthic cover was dominated by Hard coral, turf algae, sand and rubble in all the sites. There was significant variation in seawater temperature levels with Kanamai recording the highest mean temperature (27.83°C), temperature range (12.27) and also recorded the highest maximum temperature (36.23°C). These findings suggest that there is spatial and temporal variation in recruit density, genera richness and survival of coral genera. Results also indicate that area protection and sea-water temperature influence recruitment between habitats but benthic substrate characteristics influence recruitment within a habitat.

INTRODUCTION

Coral recruitment can operationally be defined as the initial sighting of recently settled juveniles in the adult habitat (Caley et al., 1996). Successful recruitment is...
critical for the survival of a coral population. Measuring recruitment patterns and mortality of corals is of fundamental importance for understanding the mechanisms that regulate their populations and mediate species coexistence (Underwood and Fairweather 1989).

Recruitment in corals has been extensively studied, usually within the first weeks or months after initial settlement from the plankton (reviewed by Harrison and Wallace 1990), demonstrating considerable temporal variation between seasons and years, and spatial variation between sites (e.g., Birkeland 1977, Rogers et al., 1984, Wallace 1985a, b; Babcock 1988, Harriott and Fisk 1988). Spatial and temporal variation in recruitment has been shown to be an important component in the population dynamics of corals (Warner and Chesson, 1985). Differences in coral recruitment rates between and within sites has been measured in several sites in Kenya; (Obura et al., 2005, Maghribai et al., in review) and Tanzania (Muhando 2002) and has shown that seasonal variation in recruitment rates differ among taxa of corals, though a variety of locations show the general pattern of highest recruitment in the warmest season. Seasonality in recruitment has also been reported in the U.S Virgin Islands (Dustan 1998), Great Barrier Reef (Wallace and Bull 1982, Harriot 1992, Harriot and Fisk 1988) Okinawa (Sakai and Yamazato 1984) the northern Gulf of Mexico (Bagget and Bright 1985) and Hawaii (Fitzhardinge 1985). In all cases, most recruitment occurred in the spring and summer months, following the major spawning season. However, the interaction between recruitment and management regimes is little understood.

Settlement rates and taxonomic composition may be expected to vary spatially due to variations in geographic availability of larvae, prevailing hydrographic conditions and physical characteristics of sites. Most studies examining more than one site show significant differences in recruitment between sites, but replicated time-series show that many of these patterns are complicated by temporal variations (Baggett and Bright 1985, Wallace 1985). Taxonomic differences in recruitment patterns have also been found at the scale of individual reefs (Harriott and Fisk 1988). Some corals respond to specific settlement cues (Morse et al. 1988), which is likely to contribute to spatial variation in abundance. Similarly, preemption or overgrowth by established corals and algae is an important mechanism that can cause spatial variation in recruitment (e.g., Birkeland 1977, Sammarco 1980, and Hughes 1985). Studies of coral recruitment in the Caribbean have found variable rates of recruitment between species (Rogers et al., 1984 and Smith 1992, 1997) with large frame-building species showing low recruitment rates.

This study aimed at determining the spatial and temporal variation of genera recruitment and survival in the Kenyan coast. We asked 3 questions (1) whether there was site variability in coral recruitment and mortality rate (2) whether there was monthly,
seasonal and yearly variability in recruitment and mortality rate, and (3) whether area protection, temperature and coral cover area influenced coral recruitment and mortality rates.

MATERIALS AND METHODS

Study Area
The study area encompasses four reef sites; Mombasa Marine Park (MMP), Nyali, Vipingo and Kanamai on the North coast of Kenya (Fig. 1). MMP is characterized by water temperature range of 25-31°C throughout the year with stable salinity levels and moderately high nutrient level from terrestrial runoff and ground water. It has a reef height of 1.09m and the amplitude from neap tide to spring tide varies from 1.5 to 4m. This site experiences occasional water exchanges with the oceanic waters during high and low tides due to a small depression that forms a small channel through the reef crest. This site is protected from all extractive uses including fishing but is a primary site for glass bottom boat and snorkeling trips among tourists. Nyali resembles the MMP but traditional fishing is allowed. Kanamai and Vipingo are unprotected shallow sites characterized by extensive fishing and intensive exploitation of corals, shells and other marine organisms for commercial purposes. Their reef heights are 1.4 and 1.45m, respectively. All four sites are patch reefs within the shallow lagoon (0-7 m depth) that is formed by a 200 km fringing reef that extends from Shimoni (near the border with Tanzania) to Malindi.

Sampling Design
Two replicate stations were used at each site. The distance between stations ranged from 200m to 5m. A line transects of 25m long was used to permanently mark the area to be sampled at each station. Three 1m² permanent quadrats were sequentially located on each side of the line transect. The quadrats were randomly placed using a system of random numbers. Placement of the quadrats was such that they excluded at least 50% of coral cover, sand or other substrates that may inhibit settlement. Nails were driven into to the reef substrate at the corners of each square to delimit the quadrat area. A 1 m² PVC quadrat was used to help mark the quadrats for sampling (Fig. 2). Each station was then sampled once a month for a period of 16months.

Sampling for Recruits
Recruits were defined as corals that are less than 10cm in diameter. During the initial sampling, the number of recruits was recorded for each quadrat and recruits identified to genus level. The position of each recruit was recorded on an underwater paper for the purpose of sampling the same individual on subsequent sampling. The maximum diameter and the maximum perpendicular to that diameter of each recruit colony was measured to the nearest 1mm using a plastic caliper, from which the area of each coral was calculated. During subsequent sampling, new recruits (individuals not observed in the previous sampling) were recorded. Individuals observed in previous sampling period but not observed in immediate sampling period were considered to have died. Recruits that were overgrown with algae were also considered as dead.

The substrate characteristics at each station was described by characterizing the percentage intersection rate of the substrate type (e.g. hard coral, sand, rubble, algal turf, coralline algae, Halimeda, macroalgae, soft coral, sponge and zoanthids) along a 10m line transect. Seawater temperature was monitored throughout the study period using temperature loggers placed at each of the four sites.

Data Analysis
Monthly recruit density was derived from the number of recruits per 1m². Monthly mortality rate was derived from the number of dead recruits per 1m² per and was quantified as a percentage decrease. To determine site, monthly and yearly variations in recruit density and mortality rates t-tests were used.
There was a significant difference in recruit density within MMP and Kanamai while stations within Nyali and Vipingo did not vary significantly. Mean recruit density varied from 3.51 ± 2.14 (Kanamai 1) to a maximum of 7.45 ± 4.09 (MMP 1) as shown in Table 1.

The highest number of recruits recorded in a single quadrat was 20 in MMP 1. Monthly recruit densities varied significantly in Kanamai 1, MMP 2 and Nyali whereas the other sites did not have any significant variation in monthly recruit densities. Highest monthly recruit density occurred in Nyali in March 2006 (6.00 ± 1.23) and the lowest in Kanamai 1 in Jun 2007 (2.3 ± 2.31, Fig. 2). Southeast Monsoon and Northeast Monsoon seasons varied in their recruit densities with SEM 2006 recording the highest density (6.14 ± 3.48) and lowest density was in NEM 2007 (3.51 ± 2.52, Fig. 3). Both NEM and SEM showed high recruit densities in the year 2006 and low densities in year 2007.

RESULTS

Recruit Density

There was a significant difference in recruit density within MMP and Kanamai while stations within Nyali and Vipingo did not vary significantly. Mean recruit density varied from 3.51 ± 2.14 (Kanamai 1) to a maximum of 7.45 ± 4.09 (MMP 1) as shown in Table 1.

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Figure 2. Abundance of recruits per 1m² for the period from February 2006 to June 2007 (mean and standard deviation). Individual sites shown within MMP/Kanamai due to significant differences between them.

Figure 3. Abundance of recruits (mean and standard error) for the two major seasons (SEM and NEM) in the year 2006 and 2007.
**Table 1.** Mean, standard deviation, maximum and minimum of recruit density (no. per m²) and the overall mortality rate of coral recruits in 2006 and 2007. There were no significant differences between replicate sites at Nyali and Vipingo, so data for each reef were pooled.

<table>
<thead>
<tr>
<th></th>
<th>Kanamai 1</th>
<th>Kanamai 2</th>
<th>MMP 1</th>
<th>MMP 2</th>
<th>Nyali</th>
<th>Vipingo</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>3.52</td>
<td>7.37</td>
<td>7.45</td>
<td>4.81</td>
<td>4.03</td>
<td>4.85</td>
</tr>
<tr>
<td>N</td>
<td>77.00</td>
<td>81.00</td>
<td>69.00</td>
<td>72.00</td>
<td>152.00</td>
<td>157.00</td>
</tr>
<tr>
<td>StdDev</td>
<td>2.13</td>
<td>3.63</td>
<td>4.09</td>
<td>2.81</td>
<td>1.70</td>
<td>3.47</td>
</tr>
<tr>
<td>SE</td>
<td>0.24</td>
<td>0.40</td>
<td>0.49</td>
<td>0.33</td>
<td>0.14</td>
<td>0.28</td>
</tr>
<tr>
<td>Max</td>
<td>8.00</td>
<td>17.00</td>
<td>20.00</td>
<td>13.00</td>
<td>10.00</td>
<td>16.00</td>
</tr>
<tr>
<td>Min</td>
<td>1.00</td>
<td>2.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Mortality (m⁻²)</td>
<td>1.42</td>
<td>0.41</td>
<td>0.67</td>
<td>3.53</td>
<td>4.23</td>
<td>3.51</td>
</tr>
</tbody>
</table>

**Table 2.** Genus composition at each site and their overall abundance.

<table>
<thead>
<tr>
<th>Genera</th>
<th>Mombasa</th>
<th>Nyali</th>
<th>Kanamai</th>
<th>Vipingo</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Favia</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>20%</td>
</tr>
<tr>
<td>Porites</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>19%</td>
</tr>
<tr>
<td>Favites</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>19%</td>
</tr>
<tr>
<td>Pocillopora</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>13%</td>
</tr>
<tr>
<td>Pavona</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>8%</td>
</tr>
<tr>
<td>Platygyra</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td></td>
<td>6%</td>
</tr>
<tr>
<td>Galaxea</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td></td>
<td>3%</td>
</tr>
<tr>
<td>Acropora</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td>3%</td>
</tr>
<tr>
<td>Acanthastrea</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td>2%</td>
</tr>
<tr>
<td>Montipora</td>
<td>x</td>
<td></td>
<td>x</td>
<td></td>
<td>1%</td>
</tr>
<tr>
<td>Astreopora</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td>1%</td>
</tr>
<tr>
<td>Cyphastrea</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td>1%</td>
</tr>
<tr>
<td>Fungia</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td>1%</td>
</tr>
<tr>
<td>Hydnophora</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td>1%</td>
</tr>
<tr>
<td>Echinopora</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td>0%</td>
</tr>
<tr>
<td>Alveopora</td>
<td></td>
<td>x</td>
<td></td>
<td></td>
<td>0%</td>
</tr>
</tbody>
</table>

**Genera Abundance and Mortality Rate**

A total of 16 genera was recorded in all the sites. MMP recorded the highest number of genera (13, Table 2) while Vipingo showed the lowest number of genera (7) and the other sites showed intermediate numbers of genera. Recruitment was dominated by **Favia (20%)**, **Porites (19%)** and **Favites (19%)** whereas **Pocillopora** and **Pavona** occurred in all sites but had very low abundance (13% and 8% respectively, Fig. 4). Abundance of other genera was very low with majority having ≤ 2% abundance and was grouped together as others.
Favia was more abundant in Kanamai and MMP, Porites in Vipingo and Pocillopora in Nyali. There was monthly variation in the abundance of Favia and Favites with both having highest recruit densities in June 2006 (5.79 and 2.67 respectively), but Pocillopora and Porites did not have significant monthly variation in recruit densities. Genera also showed varying mortality rates between sites with Porites and Favites showing low mortality rates in Vipingo (38% and 50% respectively) and high mortality rates in Nyali (Fig. 5). Pocillopora recorded low mortality in Nyali (43%) and high mortality in Vipingo (85%).

**Habitat Characteristics**

Surveys on benthic cover reported high percentage cover of hard coral in Kanamai 1 (28%, Fig. 6), coralline algae and turf algae in MMP 2 (15%, 36% respectively) and macro algae in Nyali 2 (24%). Temperature varied significantly between sites with Kanamai having the highest mean temperature, 27.8 °C (Table 3).

Maximum temperature occurred in Kanamai (36.2 °C) and this site recorded the highest range in temperature over the entire study period (12.3 °C, Fig. 7), while the lowest range in temperature occurred in Nyali (8.1 °C).

**Table 3.** Overall mean and standard deviation, maximum, minimum and range of sea-water temperature in four sites in Kenya in 2006 and 2007.

<table>
<thead>
<tr>
<th>Site</th>
<th>MMP</th>
<th>Nyali</th>
<th>Vipingo</th>
<th>Kanamai</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>27.57</td>
<td>27.50</td>
<td>27.47</td>
<td>27.83</td>
</tr>
<tr>
<td>StDev</td>
<td>1.40</td>
<td>1.41</td>
<td>1.45</td>
<td>1.92</td>
</tr>
<tr>
<td>Max</td>
<td>32.90</td>
<td>32.79</td>
<td>33.84</td>
<td>36.23</td>
</tr>
<tr>
<td>Min</td>
<td>24.03</td>
<td>24.65</td>
<td>23.74</td>
<td>23.95</td>
</tr>
<tr>
<td>Range</td>
<td>8.87</td>
<td>8.14</td>
<td>10.10</td>
<td>12.27</td>
</tr>
</tbody>
</table>

Figure 4. Composition of recruits population in the four sites from February 2006 to June 2007.
The pattern exhibited in this study was of significant variability across sites, seasons and years. Within this pattern, Mombasa Marine Park (MMP) showed higher abundances compared to the other sites. Recruitment was high in the SEM season and the year 2006 recorded the highest recruitment. In addition, MMP recorded the highest number of genera and generally, *Favia, Porites, Favites, Pocillopora* and *Pavona* were the dominating genera. Substrate characteristics showed site specific abundances with hard coral having a higher percentage cover in station 1 of MMP (28%, Table 2). Kanamai recorded the highest temperature.
Figure 6. Benthic cover at all the stations.

Figure 7. Hourly seawater temperature (°C) on coral reefs in MMP, Kanamai, Nyali and Vipingo from March 2006 to June 2007.
levels and the broadest range between maximum and minimum temperatures (Table 3).

A number of factors co-occurred at the site with the highest recruit density, in the MMP. This site had the highest coral cover, which has been shown to be correlated with production of larvae (Miller et al., 2000), though it is not known to what extent study sites may be self-seeding. Additionally the greater structural complexity provided by high cover of varying coral morphologies, as found in MMP, contributes to greater juvenile survival. MMP is also located near a channel which brings a strong current flow. This provides for greater exposure to coral larvae from the reef front, and the strong current flow makes the water more aerated and substrate surfaces cleaner hence more conductive to settlement and post-settlement survival (Obura et al., 2005). Temperature range was low in MMP compared to that in Kanamai and Vipingo which is favourable for recruit survival (Shepherd et al., 2002). Similar results of high recruitment in MMP were found in Obura et al., 2005 and Tamalander, 2002.

Variation in recruitment within the MMP can be associated with the differences in coral cover. However, the site in Kanamai with the highest coral cover area recorded the lowest recruitment. This implies that apart from coral cover area; there are other factors that influence recruit density and also recruitment is patchy. In this case, the high temperature variation between MMP and Kanamai could be a possible factor that contributed to the variations in recruitment. This shows that coral cover can determine recruit densities within habitats but other broader scale factors such as temperature variation may be strong determinants of recruit densities between different habitats.

Results on recruit density between Nyali and MMP contrast those recorded on settlement tiles where Nyali recorded high settlement density (Mangubhai et al., 2007). Spat abundance is linked to larval availability and dispersal whereas the density of juvenile corals within an area reflects post-settlement mortality (Clark, 2002). This shows that there are many larvae that settle in Nyali but only a few make it to the visible juvenile stage. The high mortality rates recorded in this site (Table 1) explain this variability. However, a synchronized study on settlement plates and artificial substrate is recommended to make strong conclusions. The influence of mortality rates on recruit density is also experienced within Kanamai and MMP whereby the stations with high mortality rate recorded low recruit densities (Table 1).

Monthly variability in recruit density was observed in some stations only, with Nyali recording the highest monthly recruit density in March 2006 and Kanamai recording the lowest density in June 2007. The month of March is associated with calm and clear conditions. The month of June is a rainy season with high sedimentation, macroalgal dominance due to input of nutrients from terrestrial runoff and physical disturbance from waves make the substrate conditions less favorable for recruitment and may cause increased mortality of recruits. At a broader scale, the SEM season had high recruit density compared to NEM season in both the year 2006 and 2007. Studies on settlement plates in the East African region have shown a general pattern of high recruitment during the warmest months and low recruitment in the cool months (Maghubhai et al., in press; Obura et al., 2005; Muhando, 2002 and Nzali et al., 1998). Results of this study recorded high recruitment during the cool months and this is explained by the fact that there is a time laps before visible juveniles are observed in the natural substratum.

MMP and Nyali had a higher number of genera compared to Kanamai and Vipingo (Table 2). This reflects differences in both temperature conditions and protection between the sites. MMP and Nyali experience lower absolute and range in temperature, which in general are more favorable for survival of coral recruits. In addition, MMP is protected and Nyali has limited protection while Vipingo and Kanamai are not protected, which may also result in higher coral diversity at the former. The abundance of coral genera varied with Porites being more abundant in Vipingo and MMP, while Pocillopora was more abundant in Nyali, similar to findings of McClanahan and Maina (2003) and (Mangubhai et al., 2007).
In conclusion, protection level and sea-water temperature may be influencing coral recruitment at these study sites. However, variation of recruitment within a habitat may be mainly influenced by substrate characteristics. Further studies on temporal variation of benthic substrate characteristics are recommended to determine how the seasonal dynamics in benthic cover affect seasonal variability in recruit densities.

ACKNOWLEDGEMENTS

We would like to thank the Kenya Wildlife Service Rangers in Mombasa Marine National Park, Nyali Watersports and D Pope residents for their assistance with logistics. Thanks are due to Nassir Amiyo, Gabriel Grimsditch and Ahmed Mudasir for fieldwork support. This study was funded under Project 9 of the Coral Reef Targeted Research (CRTR) Program’s Bleaching Working Group.

REFERENCES


High Zooxanthellae Densities and Turnover Correlate with Low Bleaching Tolerance in Kenyan Corals

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When a coral bleaches, the obligate symbiosis between the coral polyp and the micro-algal zooxanthellae is disrupted and the zooxanthellae are expelled from the polyp. Although a bleached coral does not necessarily die, it is more vulnerable to disease, algal overgrowth, bioerosion and eventually mortality. Mass bleaching and mortality events in the last decade have prompted increased research into zooxanthellae, and it is possible that zooxanthellae population strategies affect a coral’s tolerance to bleaching stress.

Variation in zooxanthellae density and mitotic index was studied in eleven species of scleractinian coral (Acropora sp., Echinopora gemmacea, Favia sp., Galaxea fascicularis, Hydnophora microconos, Montipora aequituberculata, Pavona decussata, Pocillopora damicornis, Pocillopora eydouxii Porites cylindrica and Porites lutea) from the Mombasa Marine Park, from 1998 and 2006. This data was compiled with average and standard deviations of monthly SSTs and radiation levels in a Canonical Correlation Analysis (Fig. 1).

The Canonical axes shows that zooxanthellae density and mitotic index are nearly orthogonal, i.e. independent from one another, and neither align very strongly with any of the environment axes. Of the coral species, $Pocillopora damicornis$ and $Pocillopora eydouxii$ display the lowest zooxanthellae densities and the highest mitotic indices. By contrast, $Galaxea fascicularis$, $Porites cylindrica$ and $Porites lutea$ display the highest zooxanthellae densities but low mitotic indices. Looking(6,9),(995,996)
and high mitotic indices (high zooxanthellae turnover) are more susceptible. This apparent relationship between bleaching tolerance and zooxanthellae density may be explained by several hypotheses:

- higher densities of zooxanthellae could lead to higher self-shading and thus protection from light stress (Warner et al., 1999);
- higher zooxanthellae densities could mean higher concentrations of UV-absorbing compounds such as mycosporine-like amino acids (MAAs);
- higher zooxanthellae densities correlate with higher amounts of coral tissue per square centimeter of corallum surface, and coral tissue depth protects zooxanthellae from light stress (Hoegh-Guldberg, 1999);
- finally, turnover rates and zooxanthellae regulation may simply mean that high biomass/low turnover species bleaching slowly, while low biomass/high turnover species bleaching more rapidly, simple as a result of the turnover dynamics.

**REFERENCES**


Zooxanthellae Densities are Highest in Summer Months in Equatorial Corals in Kenya

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2 Vrije Universiteit Brussel, Pleinlaan 2, B-1050 Brussels, Belgium
3 Kenya Marine and Fisheries Research Institute, PO Box 81651 Mombasa, Kenya
4 Kenya Wildlife Service, Mombasa, Kenya
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Tropical reef-building corals contain micro-algae known as zooxanthellae (Symbiodinium sp.) within their tissue with which they exist in an obligate symbiosis. Zooxanthellae are crucial to coral polyps because they provide them with photosynthates, energy, oxygen and pigmentation.

Several studies from around the world have observed that zooxanthellae population densities can undergo marked seasonal fluctuations. Research in Israel (Shenkar et al., 2006) Thailand (Brown et al., 1999), Mauritius (Fagoonnee et al., 1999), the Bahamas (Fitt et al., 2000; Warner et al., 2002) and Hawaii (Stimson, 1997) has shown that zooxanthellae population densities are highest during colder months and lowest during warmer months, with intermediate densities in between. These fluctuations have mostly been explained in terms of temperature and solar irradiance that affect the zooxanthellae’s capacity to photosynthesize (Brown et al., 1999; Fitt et al., 2000; Shenkar et al., 2006; Stimson, 1997; Warner et al., 2002). However, these studies were conducted at higher latitude sites where seasonal environmental parameters are more variable and the difference between summer and winter conditions is more marked than at sites closer to the equator.

CORDIO collected zooxanthellae density and mitotic index data for eleven species of scleractinian coral (Acropora sp., Echinopora gemmacea, Favia sp., Galaxea fascicularis, Hydnophora microconos, Montipora equituberculata, Pavona decussata, Pocillopora damicornis, Pocillopora eydouxi Porites cylindrica and Porites lutea) in the Mombasa Marine Park from 1998 to 2006. Four main seasons are distinguished for analysis, the northeast monsoon (16 December – 15 March), the late northeast monsoon (16 March – 30 April), the southeast monsoon (1 May – 31 October), and the transitional period (1 November – 15 December). The late northeast monsoon and the transitional period are doldrum periods when warming of surface waters is most intense between the two winds. Temperatures (Fig. 1) and radiation levels (Fig. 2) are lowest during the southeast monsoon and highest during the late northeast monsoon.

Figure 1. Temperature data for the Mombasa Marine Park showing daily means and standard deviation for 1999-2005. Horizontal bars show seasons described in the text.

Figure 2. Radiation data for Mombasa showing monthly means and standard deviation for 1997-2003. Horizontal bars show seasons described in the text. Data courtesy of Mombasa Meteorological Office.
Figure 3. Average seasonal zooxanthellae densities and mitotic indices for all species. SEM = Southeast monsoon. Trans = Transitional period between monsoons. NEM = Northeast monsoon. NEMlate = Late Northeast monsoon. ANOVA values are also shown to illustrate significant differences between seasons. Den = Zooxanthellae density. Mi = Mitotic index.
Although different zooxanthellae densities in different species peaked in different months, all species displayed highest densities at some point during the overall northeast monsoon season (1st November to 30th April) and most displayed highest mitotic indices during the transitional period directly preceding the Northeast monsoon (1st November to 15th December) (Fig 3). The higher densities found during the northeast monsoon (when temperatures and radiation levels are higher) are surprising as they are contrary to trends found at higher latitudes. It is possible that at higher latitudes seasonal variability of temperature and light is so great that it dictates zooxanthellae density fluctuations, while corals closer to equator may be less influenced by seasonal variability of temperature and light, and other factors may have a greater influence on population dynamics.

This study thus highlights the degree of variability in zooxanthellae population dynamics there may be among coral species and between sites at widely different geographic locations.

REFERENCES


A Description of *Acropora* sp. 1 in the Mombasa Lagoon in Kenya – A New Species or a Potential Hybrid?

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INTRODUCTION

*Acropora* species are taxonomically complex because they are highly polymorphic and have the potential to cross-fertilize and form hybrids that may be competent to settle, undergo metamorphosis and survive for up to 3.5 years (Willis et al. 1997). In Kenya *A. valida*, *A. secale* and *A. lutkeni* are, at times, remarkably similar in the field and can be difficult to separate. Mangubhai & Harrison (2006) highlighted the difficulties in separating *A. valida* from *A. secale* in the Mombasa lagoon due to the presence of a large number of immediate ‘morphs’ with characteristics of both species. In 2004 skeletons were collected of *A. valida*, *A. secale*, *A. lutkeni* and any corals showing characteristics of these 3 species for comparison to specimens at the Museum of Tropical Queensland in Australia. *Acropora valida* and *A. secale* from the Mombasa lagoon were found to be generally more sturdy (i.e. thicker and longer branches), while *A. lutkeni* was less sturdy and radial corallites were more ‘organised’ compared to specimens from other parts of the Western Indian Ocean and the Asia-Pacific region.

A fourth putative ‘species’ was identified, *Acropora* sp.1, which has tentatively been placed in the *Acropora nasuta* group (Wallace 1999) because it appears to share skeletal characteristics with all 3 species. Field and skeletal characteristics are provided below. The potential for *Acropora* sp.1 to be a more ‘extreme’ form of *A. secale* or *A. valida* or a more ‘conservative’ form of *A. lutkeni* cannot be overlooked, and there is also the possibility that hybridisation may be occurring between the different species. Detailed morphometric and genetic studies, and the further narrowing of spawning times, may resolve the taxonomic status of this putative ‘species’ at a later date.

Field: Colonies are mostly brown with brown or cream axial corallites (Fig. 1a-b). Towards the base of branches radial corallites become white giving them a scale-like appearance. Branches are rarely tapering, with branch thickness mostly made up by the radial corallites. However, these skeletal characters also occur in a small number of *A. valida*, *A. secale* and in *A. lutkeni*, making field identification of *Acropora* sp.1 difficult.

Skeleton: Colonies are corymbose or caespitocorymbose with a central or side attachment, with branches of 12-22 mm diameter and up to 140 mm in length. Axial corallites outer diameter ranges from 2.3-3.0 mm and inner diameter from 0.6-1.0 mm. Radial corallites are touching and are a mixture of sizes with fairly thick walls. The branch tip (10-20 mm) has tubular radial corallites with round to nariform openings, sometimes alternating with smaller sub-immersed corallites so that it bears a strong...
**Figure 1**: *Acropora* sp.1. Live colony (a-b), portion of colony (c-d); electron micrograph showing axial and radial corallites (e-g) and coenosteum on radial corallites (h).
resemblance to *A. secale*. However, below about 20 mm, radial corallites can become strongly appressed so that the remaining portion of the branch more closely resembles *A. valida*. The presence of secondary sub-branches and long tubular radial corallites of different lengths gives the branches a sturdy robust appearance that resembles *A. lutkeni*.

**REFERENCES**


The Lallie Didham Coral and Shell Collection, at CORDIO East Africa

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The scleractinian coral fauna of the East African mainland coast has had little formal systematic study. H.J. Hamilton, at the University of Dar es Salaam in the 1970s completed a Masters degree focusing on the coral fauna of Dar es Salaam and nearby areas of Tanzania and Kenya (Hamilton 1975, Hamilton and Brakel 1984). Since then, field surveys for coral diversity have been conducted at many sites but until high quality in situ identification resources with global coverage were released after the turn of this century (Veron 2000, Wallace 2001), species identification was severely hampered. In a regional compilation, Sheppard (2002) reported from the literature coral species numbers of 112 for Kenya and Tanzania combined, compared to recorded numbers of 270+ and predicted numbers over 300 (see Obura, 2007). As a result of this under-representation, the East African coast has featured as a lower-diversity subregion within the overall Indo-Pacific province (Veron 2000).

At present coral collections are held at the University of Dar es Salaam in the Faculty of Aquatic Science and Technology, and in the National Museums of Kenya, in Nairobi. The Lallie Didham coral and shell collection will add a further reference collection to be based in Mombasa, Kenya, to improve training in coral taxonomy and identification, and as a reference for the updated diversity of corals in the region.

Lallie Didham was born in 1929 in London, England, and moved to East Africa in 1952. Living in Malindi since 1960 adjacent to Casuarina Point, she was instrumental in helping the then Wildlife Conservation and Management Department in selecting the Malindi and Watamu reefs as Africa’s first Marine Protected Areas, gazetted in 1968. Following that, Lallie was appointed an Honorary Warden, and maintained her support to the Malindi Warden throughout her life. Living by the reefs in Malindi, Lallie developed a passion for snorkeling and diving, which she pursued actively until 2002 with dive trips up and down the Kenya coast, and across the globe. With this passion came an interest in corals and shells, which she started to collect in the 1960s, recognizing the value of keeping voucher specimens, especially of rare and unusual species from limited localities. Lallie’s collecting contributed to many leading collections around the world, and she passed specimens to the collections of the Museum of Natural History (London), Museum of Tropical Queensland (Australia) and the Australian Institute of Marine Science, among others, to the leading coral taxonomists in the world.

With her sad passing in 2001, Lallie’s coral and shell collection has been loaned by her family under the care of CORDIO East Africa, to be curated and displayed for education and research. The coral collection was sorted and catalogued at CORDIO by Rose Machuku from Moi University, and Laurence Defrise, from Belgium. The collection numbers some

**Table 1.** Family composition of the Lallie Didham coral collection.

<table>
<thead>
<tr>
<th>Family</th>
<th>Specimens</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acroporidae</td>
<td>69</td>
</tr>
<tr>
<td>Agariciidae</td>
<td>54</td>
</tr>
<tr>
<td>Caryophyllidae</td>
<td>1</td>
</tr>
<tr>
<td>Dendrophyllidae</td>
<td>48</td>
</tr>
<tr>
<td>Euphyllidae</td>
<td>12</td>
</tr>
<tr>
<td>Faviidae</td>
<td>144</td>
</tr>
<tr>
<td>Fungiidae</td>
<td>140</td>
</tr>
<tr>
<td>Helioporidae</td>
<td>4</td>
</tr>
<tr>
<td>Meandrinidae</td>
<td>3</td>
</tr>
<tr>
<td>Merulinidae</td>
<td>27</td>
</tr>
<tr>
<td>Milleporidae</td>
<td>5</td>
</tr>
<tr>
<td>Mussidae</td>
<td>28</td>
</tr>
<tr>
<td>Oculinidae</td>
<td>12</td>
</tr>
<tr>
<td>Pectiniidae</td>
<td>38</td>
</tr>
<tr>
<td>Pocilloporidae</td>
<td>36</td>
</tr>
<tr>
<td>Poritidae</td>
<td>49</td>
</tr>
<tr>
<td>Siderastreidae</td>
<td>47</td>
</tr>
<tr>
<td>Trachyphyllidae</td>
<td>12</td>
</tr>
<tr>
<td>Tubiporidae</td>
<td>1</td>
</tr>
<tr>
<td>Grand Total</td>
<td>730</td>
</tr>
</tbody>
</table>

**Photo 1:** Lallie Didham, at home in Malindi (right) and out snorkeling with KWS Warden Janet Kaleha and coxswain Heri (above left).

**Figure 1.** Part of the Lallie Didham coral collection, on display at CORDIO East Africa, Mombasa.
730 specimens in 19 families and 64 genera. Cataloguing of specimens to the species level is in progress. The geographic range of specimens is not fully documented, but is known to extend along the whole coast of Kenya, with a small number of specimens likely from northern Tanzania. The shell collection has been grouped by family and photographed, and a malacologist will be sought to conduct further work on it.

In the long term CORDIO will seek a partnership with the National Museums of Kenya and Kenya Wildlife Service for the collection to form the heart of the national reference collections for corals and shells. In this capacity it will be open to the public for viewing, and to Kenyan students to teach them about corals and coral reefs, and train new generations of biologists and taxonomists.

ACKNOWLEDGEMENTS

We would like to acknowledge the family of Lallie Didham, for loaning the collection.

REFERENCES


Resilience-Integrating Science and Management in Coral Reefs Relevant to Climate Change

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ggrimsditch@cordioea.org

ABSTRACT

Climate change will inevitably continue to cause degradation of coral reefs over coming decades (Hughes et al. 2003). The amount of damage depends on not only the rate and extent of change, but also on the ability of coral reefs to cope with change. Many MPA managers are asking, “What can we do about a large-scale issue such as climate change?” Resilience is the ability of a system to absorb or recover from disturbance while maintaining its functions and services. Natural coral reef resilience is being undermined by anthropogenic stresses such as degraded water quality, unsustainable and destructive fishing, and coastal development. These local pressures act in synergy with climate change to functionally reduce the resilience of the system, undermining its ability to cope with climate change. It becomes critical then for scientists and managers to determine the range of threats affecting the ecosystem to manage its ability to cope with climate change. While science has clearly documented the effects of climate change on coral reefs, climate-conscious strategies for managing them are only just emerging (Marshall and Schuttenberg 2006). At the MPA scale, the primary approach is to reduce other stressors and to boost the resilience of the reef.

Resilience assessments provide a comprehensive overview of threats, as well as of the state of the system. With this information in hand, a manager can make sound decisions. For example, fishing may be closed for different herbivorous fish groups during or after a bleaching event to minimize algal competition with recovering corals. Or the manager may press for more stringent control of pollution or runoff to minimize stress to corals during bleaching events. Conducting resilience surveys before an MPA is zoned can also help identify critical sites and refugia from different threats.

The IUCN Climate Change and Coral Reefs Working Group (CCCR) was created with support from the MacArthur Foundation to bring together leading coral reef practitioners to expedite the development of management tools and strategies that boost coral reef resilience. This paper outlines the methodology developed by the CCCR for assessing the
resilience of coral reef sites to climate change (IUCN 2008). The paper focuses on one of the six components of the overall method, a suite of semi-quantitative ‘resilience indicators’ that are estimated on a 5-point scale.

**An Operational Framework for Assessing Resilience**

Coral reef resilience to climate change is determined by a range of oceanographic, environmental, ecological and anthropogenic factors (Grimsdith and Salm 2006). West and Salm (2003) and Obura (2005) provide a framework that relates these to thermal protection, thermal resistance, thermal tolerance and recovery potential. To operationalize these factors and variables for monitoring and management they can be distinguished by whether they are ecological, environmental (physical), or anthropogenic in origin, and by whether their primary influence is on the sensitivity of corals to thermal stress (protection, resistance and tolerance) or recovery ability following coral mortality, or both of these. Sensitivity is a combination of resistance (ability to experience exposure without bleaching) and tolerance (ability to survive once bleached); recovery potential is the capacity within the system following coral mortality for new corals to colonize and survive, thereby returning the reef to a coral-dominated state. Together, sensitivity and recovery potential determine the resilience of coral communities to rising sea temperatures.

The first step in practical application of resilience principles is to define resilience in operational terms: resilience of *what* to *what?* The IUCN methodology translates general resilience principles into a suite of variables that can be measured or estimated, on the resilience of corals and coral reefs to thermal stress. These can be used to derive an index for comparing the relative resilience of different coral reef areas. This operational definition for coral reef resilience provides the framework for selecting variables that can be measured, and for developing a semi-quantitative index of resilience.

**The IUCN Methodology, Resilience Indicators and the 5-point Scale**

The distinguishing features of the methodology compared to standard coral reef monitoring protocols are in more detailed measurements of coral population data (size classes, recruitment, condition etc), a functional group approach to fish surveys focusing on herbivores, and estimation of a variety of resilience indicators selected to quantify the main factors that affect reef resilience. Here we describe the resilience factors and semi-quantitative indicators quantified in the methodology.

Data collection on the resilience indicators is grouped by factors that affect resilience of a site to climate change. For example, screening and shading is a factor that affects the degree of bleaching of a coral (West and Salm 2003, Obura 2005), and can be estimated by multiple variables including turbidity, exposure to river plumes, reef slope, etc. With sufficient time, resources and expertise the quantification of these factors could be done through detailed quantitative measurements (e.g. turbidity readings throughout the year), however this is impractical for rapid assessment thus an estimation approach is used, based on a semi-quantitative 5-point scale that the observer must estimate based on conditions at the time of field surveys and general and published knowledge about the site or region.

The 5-point scale was selected to facilitate estimation of minimum (1), maximum (5) and moderate (3) levels for each indicator for the region of application, and intermediate levels of low (2) and high (4). In general scaling is done such that 1 designates low/poor/negative conditions for corals and 5 high/good/positive conditions. Where an indicator can be quantified directly (e.g. visibility in meters, slope in degrees) it is estimated directly, and converted to a 5–point scale during analysis.

Because the indicator scaling is semi-quantitative, observers must be fully comfortable with the rationale behind the scaling, and have a broad knowledge of local conditions. A criterion table containing a detailed description of each level of the 5-point scale
for each indicator is included in the manual for the method, and must be reviewed and customized for each region/area of application. For example the location in a region with maximum wave energy should define the maximum (5) on the scale for wave energy; in one region this might be a reef front that experiences 2 m wind-waves during storms, in another region this might be a reef front that experiences 4 m ocean swells during its winter season. Scaling for between-region comparisons will be dealt with later, based on the levels set in each region’s definition table.

**Toward a Resilience Index**

Resilience, even in its simplified operational form, is a multi-dimensional property of a system. By incorporating this complexity, resilience has emerged as a key concept for understanding and managing highly dynamic and interconnected systems such as coral reefs. Yet, for resilience to be useful in many applied settings it must be able to be represented on a single dimension. This enables decisions about the relative resilience of different parts of the system to be made, and facilitates the use of resilience in practical planning and policy decisions. For example, an assessment of the relative resilience of different reefs or reef sites within a planning area can help managers decide how to prioritize investment of limited management resources. With knowledge of the location of naturally resilient sites, or of reefs that have low resilience due to human activities (such as exposure to poor water quality), managers can strategize their efforts to optimize resilience outcomes. A simple resilience index can be derived from the multiple variables that are known to contribute to system resilience. Appropriately standardized (to generate a rating of 1 to 5, for example), such an index will enable managers to assess the relative resilience of different reefs. Further, by examining the values of the underlying resilience factors and variables, managers can delve into the basis for the resilience rating of any particular reef. This can be especially useful when multiple reefs have the same resilience rating, but for different reasons. Understanding the basis for a resilience rating can help managers decide whether action might be effective in restoring or maintaining the resilience of a site.

**REFERENCES**


RESILIENCE FACTORS & INDICATORS

a) Physical factors

<table>
<thead>
<tr>
<th>Factor</th>
<th>Description</th>
<th>Measurement Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>Primary stressor for bleaching related to climate change</td>
<td>Spot measurements with a thermometer allow basic comparisons among sites, but ideally need long term in situ records, and satellite data to infer differences among sites.</td>
</tr>
<tr>
<td>Depth</td>
<td>Basic zonation variable for coral reef and community structure, and for attenuation of temperature, light and other variables</td>
<td>In situ measurement, usually samples done in standard depth zones for analysis. Tidal variation important to be factored out, particularly where range &gt; 2 m.</td>
</tr>
<tr>
<td>Radiation</td>
<td>Primary stressor for high-light bleaching and during doldrum conditions.</td>
<td>Basic measurement with light loggers (eg. Hobo) possible, or better with light meter. Usually not possible, or can be done at a few locations to ground-truth visibility and depth factors.</td>
</tr>
<tr>
<td>Visibility</td>
<td>Proxy for turbidity and attenuation of light levels at a site, a primary and synergistic stressor with temperature.</td>
<td>Horizontal visibility at the sampling depth, or improved with use of secchi disc (though not possible in shallow water). Where possible suspended particulates/turbidity can be measured.</td>
</tr>
</tbody>
</table>

Temperature and light are two of the dominant factors that control coral growth and reef distribution from local to global scales, and both decrease with depth. They are also the primary variables involved in mass coral bleaching related to climate change, where increased temperatures and light levels during calm periods result in severe stress and bleaching. Physical measurements of both should be known for the study area over an annual cycle, such as by installing temperature loggers at key sites, and making light measurements during key seasons. These will help interpret the one-off estimates of temperature and light made during field surveys using the variables above.

Photos top to bottom: Coral reef depth zones on an outer atoll slope, showing influence of wave energy and light attenuation: surge zone impacted by high wave energy (0-5 m), shallow platform with high coral cover (5-15 m), reef edge above dropoff into deep water (15-20 m), and deep fore reef slope (25 m) showing coral plates determined by low light penetration.
b) Substrate and reef morphology factors

<table>
<thead>
<tr>
<th>Sediment layer texture</th>
<th>Sediment grain size and sorting affects benthic organisms.</th>
<th>Estimation on 5 point scale, from large-size/carbonate sand grains at one end (good) to fine silty sediment with high terrigenous content at the bad end.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sediment layer depth</td>
<td>Depth of sediment layers on hard substrata, particularly in association with algal filaments/turf.</td>
<td>Estimation on 5 point scale, from no sediment on hard substrata to drifts of sediment and/or entrapment of sediment in algal filaments/turf that inhibit settlement.</td>
</tr>
<tr>
<td>Topographic complexity – micro</td>
<td>The surface roughness and small-crevise space on reefs affects recruitment of corals.</td>
<td>Estimation on 5 point scale of surface roughness on &lt; 10 cm scale, from smooth to complex 3-D spaces allowing light penetration but shelter from predators and sedimentation (e.g. in complex branching frameworks)</td>
</tr>
<tr>
<td>Topographic complexity – macro</td>
<td>The large scale structure of a reef, providing habitats for large and higher-trophic level mobile organisms (e.g. fish)</td>
<td>Estimation on 5 point scale of structure on a 1-10 m scale, from a flat pavement to complex 3-D reef slopes with spur/ grooves, pillars, caves and large internal reef spaces.</td>
</tr>
</tbody>
</table>

Recruitment and growth of corals, and therefore recovery and resilience, are affected by the topographic complexity and substrate quality of a reef. Topographic complexity is important as it determines the amount of space available for fauna and flora to attach to, the types and numbers of micro-habitats and shelter provided for different reef residents, and the complexity of interaction between substratum and the water column. At the small scale (micro-topography) small crevices provide important habitats for coral recruits to grow. At the large scale (macro-topography) physical structures such as spurs, grooves, pillars and caves provides habitats for larger organisms crucial to resilience, such as herbivorous fish. Sediment quality and quantity strongly affect the survival of benthic organisms, and in particular settlement larvae to the benthos and their subsequent recovery. Sediment that is fine and of terrigenous origin (silt) is a stronger inhibitor than coarse sediment of calcareous origin, both for physical reasons and because it contains higher levels of organic matter and microbial community detrimental to corals.

A reef with complex micro- and macro-topography has the capacity for high recruitment and survival of young corals and provision of shelter for many other organisms. Photo credits to David Obura, unless indicated otherwise.
c) Cooling and flushing factors

<table>
<thead>
<tr>
<th>Currents</th>
<th>Currents cause vertical mixing that may reduce surface temperatures, and can reduce coral stress by reducing boundary layer effects on coral metabolism.</th>
<th>Estimation on 5 point scale, informed by local knowledge and/or by 'typical' expectations of particular reef structures such as linear reef fronts, channels, etc.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waves (Exposure)</td>
<td>Wave energy causes vertical mixing, can reduce boundary layer effects on coral metabolism and increases oxygenation of water, enhancing coral metabolism. Exposure to weather events is expressed as wave energy to corals.</td>
<td>Estimation on 5 point scale, from minimum waves on sheltered/leeward reefs to maximum waves on reef crests. Increasing depth reduces the influence of wave energy, so is quantified under ‘depth’ not in this indicator. Exposure and wave energy are related, so one may be sufficient for estimation.</td>
</tr>
<tr>
<td>Deep water</td>
<td>Proximity to deep water enables mixing with cold water by upwelling and waves, currents and exposure.</td>
<td>Estimation on 5 point scale, from immediate proximity at a vertical wall, to distant. Alternatively, distance to a deep contour (30/50 m) may be measured from charts.</td>
</tr>
<tr>
<td>Depth of reef base</td>
<td>The depth of the base of a reef slope affects the potential for mixing of deep cool waters.</td>
<td>Actual depth of base of main reef slope. Along with “deep water” gives an indication of potential for upwelling/mixing of cooler water.</td>
</tr>
</tbody>
</table>

Exposure to very warm surface waters is the primary cause of coral bleaching, where both the degree of warming (>1°C above normal during the summer months) and the duration of warming accumulate stress in the coral. These factors result in cooling of sea water temperatures by causing or facilitating mixing of deep cooler waters into the warmer surface layers. Currents mix the water column, especially when they are variable, such as tidal currents, and when interacting with a complex bottom structure. Breaking waves result in mixing within the wave/surge zone. Proximity of the coral community to deep water (e.g. the 50 m contour line), or a deep reef base facilitate mixing of deeper water.

Breaking waves cause water mixing, reducing the temperature of the warm surface layer.

Proximity to deep water and a deep reef slope enhance vertical mixing of cool deeper waters up into the shallows, by currents, tides, internal waves and surface waves.
d) Acclimatization/extreme condition factors

<table>
<thead>
<tr>
<th>Factor</th>
<th>Description</th>
<th>Estimation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low tide exposure</td>
<td>Shallow corals exposed to the air at low tide experience frequent stress, and may be more resistant to thermal stress.</td>
<td>Estimation on 5 point scale, relevant only to very shallow corals.</td>
</tr>
<tr>
<td>Ponding/pooling</td>
<td>Restricted bodies of water heat up more due to less mixing and greater residence times, and also enhance metabolic stress.</td>
<td>Estimation on 5 point scale, maximum for enclosed shallow bodies of water</td>
</tr>
<tr>
<td>Survival of past bleaching</td>
<td>Corals that have bleached in the past but not died may be acclimatized to bleaching conditions, and have higher tolerance for repeated bleaching events</td>
<td>Estimation on 5 point scale, based on time since, and scale of, bleaching event.</td>
</tr>
</tbody>
</table>

Acclimatisation is the learned adjustment to new conditions by an organism, i.e. phenotypic change in an organism due to stress that results in readjustment of the organism’s tolerance levels, and its continued survival. Corals that are regularly exposed to extreme or highly fluctuating environmental conditions, such as on reef flats or in shallow lagoons that are subject to intense solar radiation and high fluctuations in temperature, have acclimatised to survive in these environments. This is thought to give the corals some protection from high temperatures that cause bleaching and mortality in deeper corals less accustomed to high variability, even though they may display high levels of background stress due to the fluctuating conditions. This assessment method estimates the degree to which corals are exposed to the air during low tide or are found in restricted pools or bodies of water that heat due to solar insolation. Furthermore, corals on reefs which have previously been exposed to bleaching conditions and survived may be more resistant during future bleaching events.

An *Acropora* colony that survived a past bleaching event may have enhanced resistance for the next one.

Ponding of water is enhanced by enclosed bays and reticulate reef systems, inducing high temperature fluctuations.

Corals exposed at low tide and isolated in shallow pools may acclimatise to wide fluctuations in environmental conditions, enhancing resistance to bleaching. Photo: Andrew Porter.
exposure to high levels of solar radiation, both ultraviolet (UV, 280-400 nm) and photosynthetically active radiation (PAR, 400-700 nm), also results in coral bleaching, and strongly exacerbates bleaching due to high sea water temperatures. Physical features that provide shade, for example cliffs or overhead rocks and corals, or conditions that reduce water transparency and solar radiation, for example light scattering by turbidity or absorption by chromophoric dissolved organic matter, can protect corals from bleaching. The temporal/seasonal dynamics of variable factors that reduce light levels, such as turbidity or cloud cover, must be assessed carefully compared to those that are provided by permanent physical features, such as shading. Thus visibility at the time of surveys is used as a proxy for turbidity, but it must be further assessed in terms of seasonal variation and reliability during bleaching events. Areas that are reliably protected from solar radiation can provide crucial refugia for corals that re-seed surrounding reefs after bleaching events.

Coloured dissolved organic matter in low-visibility waters screen corals from UV and PAR, and river plumes during the warm season may even protect corals from bleaching.

Shading by overhanging structures can protect corals from bleaching. The macro-topography these structures offer also provide habitats for fish. Photo: Thomas Jundt.

Shading by steep slopes and walls, particularly if they point away from the mid-day sun during the local summer, shade corals from high light stress.
f) Coral population factors

<table>
<thead>
<tr>
<th>Category</th>
<th>Description</th>
<th>Indicator</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bleaching</td>
<td>Current levels of coral bleaching.</td>
<td>Percentage of corals bleached.</td>
</tr>
<tr>
<td>Mortality-recent</td>
<td>Current levels of coral mortality.</td>
<td>Percentage of corals showing partial/full mortality.</td>
</tr>
<tr>
<td>Mortality-old</td>
<td>Levels of mortality from the past.</td>
<td>Degree of historic mortality evidenced by appearance of dead coral skeletons. Directly quantified, or as 5 point scale depending on ease of estimation.</td>
</tr>
<tr>
<td>Recovery-old</td>
<td>Levels of recovery from the past mortality events.</td>
<td>Degree of recovery from old mortality, appearance of dead coral skeletons and regrowth/recolonization of corals since then, and knowledge on past mortality. Directly quantified, or as 5 point scale depending on ease of estimation.</td>
</tr>
<tr>
<td>Disease</td>
<td>Levels of coral disease</td>
<td>Percentage of corals showing disease conditions.</td>
</tr>
<tr>
<td>Recruitment</td>
<td>Recruitment of new corals is necessary for population recovery and injection of genetic variability.</td>
<td>Estimated number and genus of recruits/new corals &lt; 2-3 cm, per m2 of substrate.</td>
</tr>
<tr>
<td>Fragmentation</td>
<td>Asexual reproduction by fragmentation is an important strategy of propagation for many corals.</td>
<td>Estimated contribution of fragmentation in generating new colonies, and primary genera affected. 5 point scale based on evidence for partial mortality/fragmentation producing significant number of small to mid-sized corals (e.g. 5 – 20 cm)</td>
</tr>
<tr>
<td>Dominant size classes</td>
<td>The dominant size classes, by area, indicate the maturity and ecological stage of a community.</td>
<td>Estimation of dominance in the coral community by size class and genus of coral, indicating successional stage of the community.</td>
</tr>
<tr>
<td>Largest corals</td>
<td>The largest corals at a site indicate how long conditions have been suitable at the site, and the degree of environmental stability/community persistence</td>
<td>The size in meters, and genus/species of the three largest colonies at the site.</td>
</tr>
</tbody>
</table>

The state of a coral population is indicated by these factors, including aspects related to current stress and mortality, past mortality, recovery from past mortality, recruitment of young corals and what age/size group of corals dominates the population. Together with indicators of diversity and the dominant genera, these can give an indication of whether the population has already suffered impacts from bleaching, and potential resistance to future bleaching. These indicators provide indicators of coral health against which to assess the other indicators of resilience.

**Photos from left to right**: Extensive stands of large corals indicate stable conditions for the assemblage, low levels of past disturbance for those species and dominance by large colonies. Abundant small to mid-sized corals indicate high levels of recruitment and growth, and possible recovery from a relatively recent disturbance event, such as mass bleaching. A coral juvenile, having survived settlement and recruitment phases. Disease levels may increase with seawater warming and as stress levels in corals increase, making them less resistant to disease attack. Bleaching of corals can range from pale to fully white, and may affect only parts of a colony.
g) Coral associates

<table>
<thead>
<tr>
<th>Category</th>
<th>Description</th>
<th>Estimation Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Obligate feeders</td>
<td>The abundance and diversity of obligate coral feeders are indicative of the health of coral colonies and complexity of interactions at a site.</td>
<td>Estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
<tr>
<td>Branching residents</td>
<td>The abundance and diversity of fish and invertebrate residents in branching coral colonies are indicative of the health of coral colonies and complexity of interactions at a site.</td>
<td>Estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
<tr>
<td>Competitors</td>
<td>The abundance and diversity of coral competitors are indicative of inhibiting factors to coral growth and recovery.</td>
<td>Estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
<tr>
<td>Bioeroders – external</td>
<td>The abundance and diversity of nonfish external bioeroders are indicative of inhibiting factors to coral growth and recovery.</td>
<td>From transect/ quadrat counts or by estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
<tr>
<td>Bioeroders – internal</td>
<td>The abundance and diversity of internal bioeroders are indicative of inhibiting factors to coral growth and recovery.</td>
<td>Estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
<tr>
<td>Corallivores (negative impact)</td>
<td>The abundance and diversity of corallivores (e.g. COTs, Drupella) are indicative of additional mortality to coral colonies.</td>
<td>From transect/ quadrat counts or by estimation on 5 point scale, from absent to high abundance/ diversity.</td>
</tr>
</tbody>
</table>

Corals are affected by a broad variety of positive and negative interactions with other species. Obligate coral feeders (e.g. some butterflyfish) and residents in branching corals can be used as indicators of the health of corals and their ability to support dependent species. The presence of ‘negative’ associates can indicate levels of stress experienced by corals defending themselves against e.g. competitors or predators, and also of ecological/environmental conditions that might favour the competitors over corals, e.g. of high-nutrient conditions promoting competing sponges or internal bioeroders. While there are many positive and negative associates of corals, the ones identified for estimation in this assessment method should be well known and relevant to the study area.

Photos from left to right: Species resident in branching corals can give an indicator of the health and maturity of the coral community. Competitive interactions, such as with encrusting sponges and microbial mats indicate conditions suitable for competitors, and that corals are experiencing stress from these interactions. Boring organisms on coral skeletons may cause some stress to corals when at high densities. Coral predators such as the crown of thorns seaster (Acanthaster planci) can cause significant damage to coral communities.
**h) Connectivity factors**

<table>
<thead>
<tr>
<th>Capacity for self-seeding (autochthony)</th>
<th>Recruitment of new corals appears to be more strongly driven by self-seeding than previously thought.</th>
<th>Based on literature, proxy indicators – reef/island size, strength/linearity of currents, presence of eddies.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Capacity for external seeding (allochthony) – small scale</td>
<td>Larval density decreases with distance from healthy source reefs, thus inter-reef distances important for allochthonous larval seeding.</td>
<td>Distance from nearest healthy reef/system, on 10 km scale.</td>
</tr>
<tr>
<td>Capacity for external seeding (allochthony) – large scale</td>
<td>Larval density decreases with distance from the source, thus distances between major reef tracts important for allochthonous larval seeding.</td>
<td>Distance from nearest reef system/complexity of regional reef biome (in 100s of km)</td>
</tr>
<tr>
<td>Suitability of currents in maintaining connectivity among reefs</td>
<td>Locations within direct current flows will have enhanced capacity for external seeding of larvae, current systems maximizing flow among reefs and locations will maximize connectivity among sites.</td>
<td>Estimate degree of connection (cross-flow) vs. upstream/downstream flow.</td>
</tr>
<tr>
<td>Natural larval dispersal barrier</td>
<td>Natural dispersal barriers reduce the degree of external seeding of larvae</td>
<td>Distance to and size of nearest natural dispersal barrier.</td>
</tr>
</tbody>
</table>

Recovery of a coral community after mass mortality is strongly influenced by the supply of larvae from reproductive adult populations, i.e. connectivity between reefs. Further, since stressed corals may not produce larvae for over a year, and since susceptible species may be completely eliminated from a reef by bleaching, the recovery of a bleached reef may be initially dependent on coral larvae from neighbouring reefs. Reefs are therefore more resilient to disturbances such as bleaching if they are well connected to healthy ‘source’ reefs or ‘refugia’ by ocean currents that provide larvae to replenish the degraded reef. The availability of larvae is related to the proximity to healthy reefs and oceanographic features such as currents, eddies, and their interaction with bathymetric and topographic features such as bays and inlets. Localised hydrodynamic features that prolong water residence times increase the likelihood of settlement. Dispersal barriers prevent transport of larvae to downstream reefs and may be natural (for example river discharges) or anthropogenic (for example pollution sources or enhanced river outflows). In addition to these connectivity factors, recruitment success and post-recruitment survival of juvenile corals is affected by the quality and stability of the substrate and other factors such as coralline algae, herbivore grazing and sedimentation. Understanding connectivity patterns is essential to designing successful Marine Protected Area networks.

An *Acropora* recruit is likely seeded from a different reef, as the larvae of broadcast-spawning species spend some weeks in the water column feeding and growing.

The geomorphology of the coastline and adjacent reef systems, together with the direction and strength of currents, determine the capacity for source-sink relationships between reefs. In general, complex reef systems have higher levels of connectivity due to complex currents among them.

Dispersal barriers may be physical, such as land- or open ocean barriers that prevent connectivity between reef systems, or due to environmental barriers such as large estuarine or river systems that create unsuitable conditions in the water column and on the bottom for coral survival and growth.
### i) Anthropogenic factors

<table>
<thead>
<tr>
<th>Nutrient input</th>
<th>Nutrient enhancement or eutrophication alters many reef processes, enhancing algal and microbial growth, and metabolically stressing corals.</th>
<th>Estimate effect of anthropogenically derived nutrients on site, from zero to extreme.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pollution (chemical)</td>
<td>Chemical pollution causes metabolic stress to reef organisms, either causing mortality, or reducing their ability to withstand other stresses</td>
<td>Estimate effect of anthropogenic pollutants on site, from zero to extreme. Distance to pollution sources can be an alternative.</td>
</tr>
<tr>
<td>Pollution (solid)</td>
<td>Solid wastes foul the substrate and may make it unsuitable for coral recruitment and growth.</td>
<td>Presence of solid waste on site and/or distance to sources.</td>
</tr>
<tr>
<td>Turbidity/ Sedi-</td>
<td>Anthropogenically enhanced turbidity and sedimentation in general negatively affects corals, though see shading/screening factor.</td>
<td>Estimate effect of anthropogenic factors on turbidity/sedimentation at site</td>
</tr>
<tr>
<td>mentation</td>
<td>Physical damage to the site or to corals results in mortality and/or inhibits recovery.</td>
<td>Estimate effect of physical damage on site</td>
</tr>
<tr>
<td>Fishing pressure</td>
<td>Overfishing causes reef degradation by changing trophic web structures, altering top-down ecological controls and leading to phase shifts.</td>
<td>Estimate effect of fishing by observation underwater and/or using catch monitoring data, local knowledge and other sources.</td>
</tr>
<tr>
<td>Destructive fishing</td>
<td>Destructive fishing causes physical damage to the site, and/or alters fish population dynamics.</td>
<td>Estimate destructive fishing by observation underwater and/or using catch monitoring data, local knowledge and other sources.</td>
</tr>
<tr>
<td>Dispersal barriers</td>
<td>Anthropogenic factors that enhance natural barriers or create new barriers to external seeding of larvae</td>
<td>Distance to and degree of nearest anthropogenic dispersal barrier and/or enhancement of natural barriers.</td>
</tr>
<tr>
<td>MPA or other management</td>
<td>Management that reduces any of the above anthropogenic stressors enhances the natural ability of corals and reefs to resist bleaching and to recover.</td>
<td>Estimate effectiveness of management actions from none to high (as an additional factor to the reduced level of the stressors above).</td>
</tr>
</tbody>
</table>

Anthropogenic activities affect many ecological processes on coral reefs, and corals chronically affected by anthropogenic impacts must expend energy on resisting stress and therefore have a lower resilience to other disasters such as bleaching. Stressed corals bleach more easily, produce fewer larvae and grow more slowly, and this reduces recovery rates. This assessment method focuses on anthropogenic nutrient input, pollution, sedimentation, physical damage, fishing pressure and destructive fishing; and provides an estimation of how each of these stresses reduces natural resilience by degrading specific ecological processes. For example, over-fishing of herbivorous fish depletes the system of this key functional group and allows macro-algae to outcompete corals after a bleaching event. Effective management at the source of key anthropogenic stressors, for example through Marine Protected Areas or fisheries management that maintains populations of herbivores, can help promote recovery and thereby support the ability of corals and reefs to resist and recover from bleaching impacts.

Photos from left to right: Anthropogenic impacts on coral reefs are diverse and have led to the degradation of many reef systems worldwide. Fishing, even by small-scale commercial fishers can significantly damage reefs due to human population growth and increasing demand for marine products. Photo credit: Monica Zavagli. Examples of direct damage to coral reefs includes bottom-damage by nets and overfishing of juveniles, destruction of reef structures by dynamite and anchor damage from boats of all types. Development in coral reef areas, particularly for mining or industry, can result in high levels of pollution that reduce the resilience of reef communities. High density urban populations and over-developed coastlines can result in high levels of sewage pollution to coral reef waters, causing coral mortality and a switch to more nutrient-tolerant communities.
CORDIO Status Report 2008

Part 5 – Fish Spawning Aggregations

Reef Fish Spawning Aggregations in the Western Indian Ocean: Current Knowledge and Implications for Management

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keywords: spawning aggregations; local ecological knowledge; verification; periodicity; habitat, management implications

ABSTRACT

Studies of reef fish spawning aggregations are new to the Western Indian Ocean compared to other regions. This paper reviews the current state of knowledge of spawning aggregations in the region and assesses their implications for fisheries management and conservation. Fisher knowledge has identified more than 30 species of reef fish that aggregate to spawn, mainly belonging to the families Lutjanidae, Serranidae, Lethrinidae and Siganidae. Verification has been achieved for 25 spawning aggregations from 7 species, including five and six aggregations of Epinephelus fuscoguttatus and Siganus sutor, respectively. Reef fishes commonly spawn within the northeast (November-April) and inter-tropical monsoon periods. Serranid aggregation sites include reef passes, channels, reef slopes and pinnacles, while Siganus sutor spawns on patch reefs and granitic reefs. The status of spawning aggregations is poorly known and evidence of aggregation collapses are currently confined to Seychelles. Few spawning aggregations are protected in the region and their applicability to new approaches of managing for resilience will not be realised without considerable efforts in research and advocacy. The management of spawning aggregations through marine protected areas does not constitute a solution for fisheries management and must be viewed as complementary to tools such as catch and effort controls.

INTRODUCTION

In the Western Indian Ocean (WIO) scientific information on artisanal fisheries is insufficient and management regimes require substantial improvement (van der Elst et al., 2005). In areas of East Africa, overfishing may constitute the most important local threat to coral reefs (McClanahan et al., 2000). Combined with the impacts of coral bleaching, coastal pollution, development and other direct and indirect causes (Samoilys & Church, 2004; Obura, 2005;
problems in the region often appear intractable. Marine protected areas (MPAs) or reserves are increasingly viewed as a solution to a global fish crisis (Gell & Roberts, 2003; Halpern 2003). While the functioning of MPAs for conservation objectives is not in doubt, improving their use for fisheries management requires filling significant scientific gaps in their application and design (Sale et al., 2005).

In the context of coral reef degradation, concepts of managing for system resilience highlight the need for networks of MPAs (Obura, 2005, Schubert et al., 2006). The protection of reef fish spawning sites as sources of seed is central to marine reserve network models (Sala et al., 2002). However, this approach has progressed little as an applied conservation or reef fisheries management tool, even where scientific knowledge of aggregation sites is extensive. With few spawning aggregations effectively managed in no-take reserves (Sadovy & Domeier, 2005), let alone as part of networks, this element of managing for resilience lags far behind that of biodiversity and coral conservation (e.g. TNC, 2004, Grimsditch & Salm, 2006). Reef fishes are highly diverse in terms of their reproductive strategies and patterns (Sadovy, 1996) but spawning in aggregations at specific times and locations is common to several families of reef fishes (Domeier & Colin, 1997).

Dedicated initiatives to locate and study spawning aggregations in the WIO began in 2003 with a three-year programme in Seychelles to locate and verify sites reported by fishers (Robinson et al., 2004; Robinson et al., 2007). This was followed by an IUCN-led initiative to document local ecological knowledge of aggregations in Kenya, Mozambique and Tanzania and, later, to verify sites in Kenya (Samoilys et al., 2006; Samoilys et al., in prep). A more localised and fishery specific research project which also studied spawning aggregations was recently completed in southern Kenya (Kimani, in prep.). Fisher knowledge on fish spawning behaviour is often detailed (Johannes, 1981; Samoilys & Squire, 1994; Colin et al., 2003). However, fishers’ knowledge is also difficult information to analyse in order to distinguish reliable data on spawning aggregations (Daw, 2004). In summarising fishers’ information from the WIO (Table 1), some or all of the following criteria were met (Robinson et al., 2004, Samoilys et al., 2006):

i. descriptive information on spawning aggregation behaviour of species conforms to typical spawning behaviour such as courtship, territorial displays of males, release of gametes;

ii. fish were seen with fully ripe gonads (hydrated ovaries);

iii. information on species reported by >1 fisher;

iv. information on location reported by >1 fisher.

Slightly more species have been reported to form spawning aggregations in Kenya compared to Seychelles, while the lack of reports from Tanzania is probably due to less research on this topic. The

FISHER KNOWLEDGE OF SPAWNING AGGREGATIONS

The documentation of fisher knowledge on reproductive behaviour is widely recommended as a first step to locating spawning aggregations (Johannes, 1981; Samoilys & Squire, 1994; Colin et al., 2003). However, fishers’ knowledge is also difficult information to analyse in order to distinguish reliable data on spawning aggregations (Daw, 2004). In summarising fishers’ information from the WIO (Table 1), some or all of the following criteria were met (Robinson et al., 2004, Samoilys et al., 2006):

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iii. information on species reported by >1 fisher;

iv. information on location reported by >1 fisher.

Slightly more species have been reported to form spawning aggregations in Kenya compared to Seychelles, while the lack of reports from Tanzania is probably due to less research on this topic. The
rabbitfish *Siganus sutor*, a regional endemic and an important target species of reef fisheries, is well known as an aggregating species in all three countries. At the family level, the most species reported to form aggregations belong to the family Lutjanidae, followed by the Serranidae, the Lethrinidae and the Siganidae. Fisher observations of Carangidae spawning aggregations have largely been confined to Seychelles, Table 1. Species reported by fishers to form spawning aggregations in three countries of the WIO region (Robinson, et al., 2004; Samoilys, et al., 2006; Kimani, in prep.; Samoilys, et al., in prep.).

<table>
<thead>
<tr>
<th>Family</th>
<th>Species</th>
<th>Kenya</th>
<th>Seychelles</th>
<th>Tanzania</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acanthurida</td>
<td><em>Acanthurus mata</em></td>
<td></td>
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<tr>
<td></td>
<td><em>Naso brevirostris</em></td>
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<tr>
<td>Carangida</td>
<td><em>Carangoides gymnostethus</em></td>
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<tr>
<td></td>
<td><em>C. fulvoguttatus</em></td>
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<tr>
<td></td>
<td><em>Seral crumenophthalmus</em></td>
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<tr>
<td>Haemulida</td>
<td><em>Plectorhinchus flavomaculatus</em></td>
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<td></td>
<td><em>P. gaterinus</em></td>
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<td></td>
<td><em>P. schoraf</em></td>
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<tr>
<td>Lethrinida</td>
<td><em>Lethinus crocineus</em></td>
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<td></td>
<td><em>L. harak</em></td>
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<td></td>
<td><em>L. nebulosus</em></td>
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<td></td>
<td><em>L. obsoletus</em></td>
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<td></td>
<td><em>L. xanthonchilus</em></td>
<td></td>
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<tr>
<td>Lutjanida</td>
<td><em>Aprion virescens</em></td>
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<td></td>
<td><em>Lutjanus argentinamaculatus</em></td>
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<td></td>
<td><em>L. bohar</em></td>
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<td><em>L. chrenbergi</em></td>
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<tr>
<td></td>
<td><em>L. fulviamma</em></td>
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<td></td>
<td><em>L. gibbus</em></td>
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<tr>
<td></td>
<td><em>L. quinquelineatus</em></td>
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<td></td>
<td><em>L. rivulus</em></td>
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<td></td>
<td><em>L. sanguineus</em></td>
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<tr>
<td></td>
<td><em>L. sebae</em></td>
<td></td>
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<tr>
<td>Serranida</td>
<td><em>Cephalopholis miniata</em></td>
<td></td>
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<tr>
<td></td>
<td><em>Epinephelus fuscoguttatus</em></td>
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<tr>
<td></td>
<td><em>E. lanceolatus</em></td>
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<td></td>
<td><em>E. multinoctatus</em></td>
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<td></td>
<td><em>E. polyphkeadion</em></td>
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<td></td>
<td><em>Plectropomus laevis</em></td>
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<tr>
<td></td>
<td><em>P. punctatus</em></td>
<td></td>
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<tr>
<td>Scaridae</td>
<td><em>Scarus rubroiolaceus</em></td>
<td></td>
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<tr>
<td></td>
<td><em>Leptoscarus vaigiensis</em></td>
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</tr>
<tr>
<td>Siganidae</td>
<td><em>S. argenteus</em></td>
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<tr>
<td></td>
<td><em>S. sutor</em></td>
<td></td>
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<tr>
<td>Sphyraenida</td>
<td><em>Sphyraena jello</em></td>
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<tr>
<td>Mullidae</td>
<td><em>Mullloidichthys vanicolensis</em></td>
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</tr>
</tbody>
</table>

1: *Note: L. fulviamma incorrectly reported as L. kasmira in Samoilys et al., 2006.*
2: *Information on E. lanceolatus provided by N. Jiddawi, Institute of Marine Sciences, Zanzibar.*
3: *The same Creole name is often used for Epinephelus multinoctatus and E. flavoceruleus, leading to uncertainty in fisher reports (Robinson et al., 2007).*
where these species are important components of the artisanal catch.

Spawning in large aggregations is common to serranids (Domeier & Colin, 1997; Sadovy, 1996). Knowledge of reproductive behaviour is widespread in Seychelles, where at least five serranids and several key spawning sites have been consistently identified by fishers (Robinson et al., 2004). In contrast, knowledge in Kenya and Tanzania appears more fragmentary (Samoilys et al., 2006). Unlike the Seychelles where serranids are a target species of the artisanal fishery (Grandcourt, 2005), observations of fish catches and long-term data from catch monitoring systems suggest that serranids are unimportant to the Kenyan (see Waweru et al., this report; McClanahan et al., 1999; Kaunda-Arara et al., 2003) and Tanzanian (Anderson, 2004; Wells et al., 2007; Samoilys et al., in press) reef fisheries (but see below). It is therefore not surprising that fishers’ information on their spawning aggregations is scant in East Africa. E. polyphekadion aggregations were known to fishers in northern Kenya and northern Tanzania (Samoilys et al., 2006), however verification has only been obtained for E. fuscoguttatus spawning sites from southern Kenya, where large catches of this species with hydrated ovaries have also been observed (M.S. pers. obs.). Key informants and patriarchal fishers have proved invaluable in efforts to locate aggregations in the region.

VERIFIED SPAWNING AGGREGATIONS

A total of 25 spawning aggregations have been verified from seven species (Table 2). In Seychelles, 12 aggregations at 7 sites belonging to 4 species have been verified. E. polyphekadion and E. fuscoguttatus aggregations overlap spatially and temporally at three of the sites and are joined by P. punctatus at two sites. Multispecies sites are common amongst serranids, with E. polyphekadion, E. fuscoguttatus and Plectropomus areolatus commonly sharing sites in the Pacific (Sadovy, 2005). P. punctatus, being endemic to the WIO, replaces P. areolatus in this region. Since numerous families have been observed to spawn in close association with these serranids, their spawning sites may be considered key sites in reef systems (Johannes et al., 1999; Russell, 2001) and their protection may therefore be justified on grounds of biodiversity conservation in addition to fisheries management.

Aggregations of E. fuscoguttatus have been verified at two sites in Kenya through observations of behaviour and colour changes consistent with spawning in this species (Johannes et al., 1999; Rhodes & Sadovy, 2002; Robinson et al., 2007), and an increase in densities of fish at the two sites. These sites are also examples of multispecies sites, where spawning behaviour has been verified for several other species, using indirect behavioural signs (Table 2). Possible evidence that fishers have targeted E. fuscoguttatus aggregations was found at Msambweni, southern Kenya; fish with hydrated ovaries were observed in large catches of this species (M.S. pers. obs). Fishers also reported spawning behaviour at the sites of capture (P.K pers. obs.). Note that this information was not obtained through the structured fisher interviews in the area (Kimani, in prep.), highlighting the importance of including macroscopic staging of gonads during catch monitoring. Hydrated gonads are easily identified as the eggs spill out from the abdomen in a characteristic manner, from which the term “running ripe” was derived.

Peak aggregation abundances vary greatly within and between species and sites. For example, E. fuscoguttatus abundances vary from less than 100 (in sites in Kenya and Seychelles) to more than 1000 fish (at sites in Seychelles). For the serranids, E. polyphekadion formed the largest aggregations, with numbers at one site peaking at over 2000 fish (Robinson et al., in prep). No aggregations have been properly monitored in Kenya and therefore comparable data are not available. Aggregations of Mullolidichthys vanicolensis typically consisted of between 10 and 15 pairs (Samoilys et al., in press: Robinson et al., 2007). The largest reef fish spawning aggregations verified in the WIO to date belong to S.
Table 2. Spawning aggregations verified in Seychelles and Kenya through a) direct or indirect observations of aggregative spawning and/or b) increases in abundance (Robinson et al., 2007; Samoilys et al., in prep.; Kimani, in prep.).

<table>
<thead>
<tr>
<th>Country</th>
<th>Site</th>
<th>Species</th>
<th>Spawning(^1)</th>
<th>Abundance(^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Direct</td>
<td>Indirect</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S1</td>
<td><em>Epinephelus polyphekadion</em></td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S1</td>
<td><em>Epinephelus fuscoguttatus</em></td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S1</td>
<td><em>Plectropomus punctatus</em></td>
<td>Yes</td>
<td>No</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S2</td>
<td><em>Epinephelus polyphekadion</em></td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S2</td>
<td><em>Epinephelus fuscoguttatus</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S2</td>
<td><em>Plectropomus punctatus</em></td>
<td>Yes</td>
<td>No</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S3</td>
<td><em>Epinephelus polyphekadion</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S3</td>
<td><em>Epinephelus fuscoguttatus</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S4</td>
<td><em>Plectropomus punctatus</em></td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S5</td>
<td><em>Epinephelus polyphekadion</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S6</td>
<td><em>Siganus sutor</em></td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Seychelles</td>
<td>S7</td>
<td><em>Siganus sutor</em></td>
<td>Yes</td>
<td>No</td>
</tr>
<tr>
<td>Kenya</td>
<td>K1</td>
<td><em>Epinephelus fuscoguttatus</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Kenya</td>
<td>K1</td>
<td><em>Mullolidichthys vanicolensis</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Kenya</td>
<td>K1</td>
<td><em>Acanthus mata</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Kenya</td>
<td>K1</td>
<td><em>Naso brevirostris</em></td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Kenya</td>
<td>K2</td>
<td><em>Epinephelus fuscoguttatus</em></td>
<td>No</td>
<td>Yes</td>
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<td>K9</td>
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</tbody>
</table>

\(^1\) Direct signs of spawning include observations of gamete release or hydrated ovaries. Indirect signs include patterns in gonado-somatic index, colour changes and territorial/courtship behaviour associated with spawning, and observations of gravid females.

\(^2\) Text in bold indicates where increase in abundances has been determined quantitatively in both aggregating and non-aggregating periods. Otherwise, estimates are qualitative and/or lacking non-aggregation observations.

*sutor*, with more than 5000 fish aggregating at each of the two verified sites in Seychelles (Robinson et al., 2007).

The serranid aggregations verified in Seychelles conformed to the transient type often associated with these species (Domeier & Colin, 1997). Transient aggregations, which are common to the Serranidae, are usually short lived, are often large, comprising
hundreds to tens of thousands (Smith, 1972; Samoilys & Squire, 1994) of individuals, and participating fish may migrate considerable distances to and from the site (Bolden, 2000). Resident aggregations typically do not involve large distance migrations, form more regularly and occur close to or within the areas of residence for participating fish. Resident aggregations are common to the surgeonfishes (Acanthuridae) and some parrotfishes (Scaridae) (Domeier & Colin, 1997; Cornish, 2005). However, not all species conform to these two types, as seen in Plectropomus leopardus (Samoilys, 1997).

Aggregations formed by siganids, namely S. canaliculatus, have been characterised as belonging to the transient type (Domeier & Colin, 1997). S. sutor aggregations appear to conform to this definition. Four S. sutor aggregations have been verified in the Msambweni area in Kenya, and there are indications that another seven sites identified by fishers could also be spawning sites. Fishers have described S. sutor migrations to the sites within a spawning period that often lasts from 7 to 12 days (Kimani, in prep); a similar duration was observed in Seychelles (Robinson et al., 2007).

Few spawning aggregations have been verified in the WIO region compared to the western Pacific and tropical western Atlantic/Caribbean regions (Cornish, 2005). At the time of writing, a search of the online database of the Society for Conservation of Reef Fish Aggregations (www.scrfa.org) revealed that only two WIO aggregations (E. fuscoguttatus and E. polyphekadion, from Seychelles) have been reported. An E. polyphekadion aggregation from the Chagos archipelago and a Plectropomus areolatus aggregation from Maldives are also reported. The region remains inadequately represented compared to others where reports number in the hundreds. This situation most likely reflects a lack of research in the WIO rather than a rarity of occurrence (Cornish, 2005) since many species known to aggregate are common to the region and the reproductive strategy appears stable within species. The number of spawning aggregations that have been verified in the region during the past few years through the studies reviewed here suggests that more are likely to be documented if targeted research can be maintained.

**AGGREGATION SITE GEOMORPHOLOGY AND HABITATS**

Spawning aggregations often form in reef passes and channels, on reef promontories, shelves and drop-offs, and on patch and pinnacle reefs (Sadovy, 1996; Samoilys, 1997; Johannes et al., 1999; Russell 2001), though this information still remains largely anecdotal (Colin et al., 2003). The significance of site selection is still not fully understood (see Claydon, 2004). Reef passes or reef slopes close to major channels are common spawning sites for serranids in Seychelles, which are shallow (< 20 m) and defined by strong tidal currents. The serranid sites verified in Kenya are similar, being spurs on outer reef slopes. S. sutor aggregation sites may be more varied in terms of geomorphology and habitat. In Msambweni, Kenya, patch reefs inside the fringing reef lagoon are common spawning habitats for the verified spawning aggregations of S. sutor. These sites are characterised by generally high coral cover interspersed with coral rubble. Both S. sutor sites verified in Seychelles are granitic reefs on shallow bank habitats beyond the base of (carbonate) fringing reef slopes. Fishers in Seychelles also report that S. sutor and S. argenteus spawn on carbonate reefs, although this has not been verified. The degree of association with coral habitat appears stronger amongst serranids than siganids (Robinson et al., 2007).

Habitat degradation caused by destructive fishing practices, pollution and coastal development are considered threats to spawning aggregations (Sadovy & Domeier, 2005). Although the relationship between habitat variables such as rugosity and aggregation abundances is not well defined, habitat appears important for certain species (Beets & Friedlander, 1998). For example, E. polyphekadion, E. fuscoguttatus, P. leopardus and P. areolatus aggregate at coral dominated sites where territories are defended.
by males and refugia are occupied by large numbers of females (Samoilys & Squire, 1994; Johannes et al., 1999), which may indicate a degree of reliance on habitat complexity for spawning success. The impact of coral bleaching on spawning aggregations remains unstudied but may become more significant in a changing climate. In terms of acute stresses, habitat destruction resulting from coastal development may be important on local scales. For example, land reclamation has lead to the disappearance of *Epinephelus ongus* aggregations and the collapse of their fishery in Seychelles (Robinson et al., 2007).

**PERIODICITY OF SPAWNING AND AGGREGATION FORMATION**

Many reef fishes exhibit peaks of spawning activity nested within protracted spawning seasons (Munro et al., 1973; Sadovy, 1996). This pattern may be pronounced in the WIO due to the monsoon system (Nzioka, 1979; Kulmiye et al., 2002). Detailed information is available for very few species (Table 3). *S. sutor* spawning appears to peak within two periods in Kenya and Seychelles. While Ntiba & Jaccarini (1990) did not recognise a protracted season, recent studies indicate that spawning probably occurs across much of the northeast (NE) monsoon (Table 3). Spawning periodicity has not been described for many serranids in the WIO but appears to be concentrated in the NE monsoon. Species that form transient aggregations often spawn within a narrow season (Sadovy, 1996; Claydon, 2004), which appears to be the case for *E. fuscoguttatus* and *E. polyplekadion* in Seychelles where the seasons are typically 2-3 months long (Robinson et al., 2007). In Kenya, observations of *E. fuscoguttatus* aggregations vary from February to May, but more research is needed to determine the exact timing. The majority of other reef fish for which data exist also appear to spawn in the NE monsoon, including acanthurids and mullids (Table 3). Exceptions to this pattern include some signanids and lutjanids which spawn in the southeast monsoon (Robinson et al., 2004; Robinson et al., 2007; Samoilys et al., in prep).

**AGGREGATION STATUS AND FISHERIES**

Owing to their predictability in time and space, transient spawning aggregations are highly vulnerable to overexploitation (Johannes et al., 1999; Sala et al., 2001). Moreover, much of the annual reproductive output of participating fish may occur in a single aggregation (Shapiro et al., 1993; Samoilys, 1997), rendering populations of these species highly vulnerable to targeted aggregation fishing (Sadovy & Eklund, 1999; Sadovy & Domeier, 2005). The status of spawning aggregations, transient and resident, is poorly known in the WIO region. Fisheries-independent monitoring programmes are being developed for sites in Seychelles but are too recent to assess status. In the absence of scientific information, local knowledge indicates that several serranid aggregations have been depleted or have collapsed (Robinson et al., 2004), both in areas close to centres of population (*Epinephelus ongus*) and on the outer banks of the Mahé Plateau (e.g. *Epinephelus multirinotatus* and/or *Epinephelus flavocaeruleus*) (Robinson et al., 2007). Fisher knowledge of spawning aggregations in Kenya indicates that while the phenomenon has been widely observed, few have understood its significance as an important source of seed and fisheries recruitment, and no assessment of the status of spawning aggregations has been done.

Reef fisheries are generally considered overexploited in the WIO (McClanahan et al., 1999; Kaunda-Arara et al., 2003; FAO, 2006). For families characterised by the formation of transient aggregations in particular, aggregation fishing may be contributing to this situation. Where landings data exist and are disaggregated by family, it is apparent that fisheries targeting serranids are significant (Fig. 1). Given the large reported catches of serranids in Mauritius and Tanzania, aggregations may have been targeted, especially where species such as *E. polyplekadion* and *E. fuscoguttatus* are important constituents of the catch. It should be noted, however, that the relatively large serranid catches reported to FAO by Tanzania does not concur with the findings
Table 3. Comparison of spawning and aggregation periodicity for species known to form spawning aggregations. Information on some species is separated for different sites within the country. Spawning season information is from a) fishers’ knowledge, b) reproductive biology studies. Aggregation periods and lunar information are from research involving direct observation or gonad analyses. Lunar periods: NM = new moon, FM = full moon, Qtr = Quarter. Seasons in the WIO: NE monsoon (light trade winds) = Nov-Apr; SE Trades (strong trade winds) = Jun-Aug; inter-monsoon = May, Sep/Oct.

<table>
<thead>
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<th>Species</th>
<th>Country</th>
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<th>Spawning season: research</th>
<th>Aggregation: research</th>
<th>Lunar period: research</th>
<th>Reference(s)</th>
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<td></td>
<td></td>
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<td></td>
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<td>E. fuscoguttatus</td>
<td>Kenya, Tanzania</td>
<td>Nov-Jan</td>
<td></td>
<td></td>
<td>Nzioka (1979)</td>
<td></td>
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<tr>
<td></td>
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<td>Dec-Feb</td>
<td>NM</td>
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<tr>
<td></td>
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<td>Nov-Dec</td>
<td>Feb</td>
<td>NM</td>
<td>Robinson et al. (2004; 2007)</td>
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</tr>
<tr>
<td></td>
<td>Kenya</td>
<td>Feb</td>
<td></td>
<td></td>
<td>Kimani, P. (in prep)</td>
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<td>3rd Qtr, NM</td>
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<td>Nov-Jan, Dec-Feb</td>
<td>Dec-Feb</td>
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<td></td>
<td>Robinson et al. (2007)</td>
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<td>P. punctatus</td>
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<td>Dec, Jan, Feb</td>
<td></td>
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<tr>
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<td></td>
<td>1st Qtr</td>
<td>Robinson et al. (2007)</td>
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<tr>
<td>M. vanicolensis</td>
<td>Kenya</td>
<td>Apr</td>
<td></td>
<td>FM</td>
<td>Samoilys et al. (in prep)</td>
<td></td>
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<td>3rd Qtr</td>
<td>Samoilys et al. (in prep)</td>
<td></td>
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<td>Apr</td>
<td></td>
<td>1st Qtr - FM</td>
<td>Samoilys et al. (in prep)</td>
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</table>

of localised studies on artisanal fisheries, which indicate that this group are not particularly important constituents of the catch (McClanahan et al., 1999; Samoilys et al., in press a). This finding suggests that commercial rather than artisanal fisheries predominantly target serranids. In heavily exploited reef areas, sparse fisher knowledge relating to these phenomena may be because of low abundances or collapse of aggregations (Samoilys et al., 2006).

Siganids are key target species of trap, line and net fisheries in the region (Anderson, 2004; Samoilys et al., in press a) and aggregations are clearly known to
fishers and exploited (Kimani, in prep; Robinson et al., 2004; Robinson et al., 2007; Samoilys et al., 2006). Since Tanzanian fishers are aware that *S. sutor* spawns in transient aggregations, it is possible that aggregation fishing contributes to the large catches reported for this family in Tanzania (Fig. 2). Independent long-term monitoring since 1998 in Tanga, Tanzania, has shown a 5-6 fold decline in herbivore densities since 2003 which is attributed to the trap fishery that targets siganids (Samoilys et al., in press a, b).

In the Msambweni area of Kenya, four well known *S. sutor* spawning sites have been targeted by local fishers for generations, with anecdotal evidence indicating that the present day patriarch fishers began fishing the aggregations in the 1960s. Large catches of *S. sutor*, often close to 1 tonne, have been landed by ring-net fishers during a single fishing event (Fig. 3). While these catch data were recorded at the neighbouring landing site of Gazi, fishers ostensibly fished in Msambweni waters and are known to target the four prominent spawning sites located there.
A three month catch monitoring system at the Mkunguni landing site of Msambweni indicated that light artisanal gears like basket traps and fishing lines averaged up to 8 kg/fisher/day when fishing a *S. sutor* aggregation. Effort in the trap fishery is high (average 10 traps/fisher) and catches of up to 500 kg have been landed over a single lunar spawning period. In spite of the high levels of catch and effort, reports on the status of the fishery are contradictory, with many fishers reporting stable aggregation catches and others reporting a decline. By comparison, *S. sutor* has been heavily exploited, including its aggregations, for close to 100 years in Seychelles (Hornell, 1927), yet stocks have not collapsed, possibly due to the fact that only nearshore populations are targeted.

**MANAGEMENT IMPLICATIONS**

Fisheries management in the region is largely focused on industrial fisheries. Few artisanal fisheries have clearly defined harvesting strategies or management plans including controls on inputs and outputs. Overexploited coastal fisheries and ecosystem impacts of fishing are widespread (De Young, 2006). Consequently, the arguments for MPAs are well supported on both conservation and fisheries management grounds. For MPAs to complement fisheries management objectives, however, a greater emphasis on the protection of vulnerable life history stages is urgently required. MPAs are rarely designed to incorporate spawning aggregation sites and nursery habitats although some may have inadvertently received protection (e.g. in Seychelles). In Kenya, spawning aggregations were not considered in the design of MPAs. The early studies reported here have made some progress towards identifying spawning sites but much work remains in terms of verifying and studying the dynamics of aggregations.

A wide range of management tools for spawning aggregations could be considered depending on local circumstances. Appropriate responses to aggregation fisheries will likely differ between species, site, fishery and country. In Maldives, serranids form the basis of an economically important export-driven live reef food fishery (LRFF) (Sattar & Adam, 2005). This fishery is now considered heavily overexploited (Adam, 2006) and trade measures such as export restrictions or bans may be appropriate to limit effort. Seychelles recently prohibited the LRFF, in part to protect spawning...
aggregations (Aumeeruddy & Robinson, 2006). In many countries, the reliance on species that aggregate to spawn for food security may dominate management objectives, as exemplified by the siganids. In this case, gear restrictions, temporary area or seasonal closures and rights-based management approaches may be more appropriate than the formation of MPAs, and certainly more acceptable from political and socio-economic perspectives. However, these measures may only work if traditional and local compliance systems are in place. From a fisheries management perspective, the choice of tool to manage spawning aggregations should be part of a suite of measures to manage the overall fishery, which is relevant to both governmental and community-based regimes. A much greater emphasis on, and support for, research will be required in the region if spawning aggregation protection is to be part of regional fisheries management and conservation toolboxes.

Plans for a wider regional research and management programme on spawning aggregations are at an advanced stage. Studies will focus on further verification of reported aggregation sites. This will be followed by efforts to define the spatial and temporal dynamics of aggregations at key sites in order to provide information for management. In addition to research, components of the programme will focus on closed area design and application, reserve networking and awareness raising activities within fisheries and conservation management domains.

ACKNOWLEDGEMENTS

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Reef Fish Spawning Aggregations in South Asia and the Andaman Sea: Preliminary Findings from Local Knowledge

INTRODUCTION

Reef Fish Spawning Aggregations (FSA) are a phenomenon where reef fish species gather in large numbers at a specific time and site to spawn (for definitions see Domeier and Colin 1997). For coral reef fishes these sites are consistent over years. Spawning aggregations can be critical in the life cycle of the fishes that use this reproductive strategy. In particular, for those species that travel relatively large distances to aggregation sites and gather for a short period to spawn each year (termed “transient” spawning aggregations, Domeier and Colin 1997), such aggregations can represent 100 percent of the species’ reproductive output.

Fish may migrate over very large distances (10s of km) to an aggregation site, as exemplified by the Nassau grouper in the Caribbean (Bolden 2000). In addition, the pelagic fertilised eggs and larvae from such aggregations may travel far before settling out of the plankton to mature. Thus a single spawning aggregation may have an impact on fish populations over an area spanning several hundred kilometres. If these aggregations break down, for example through persistent fishing of the aggregations, the species’ population can decline dramatically to critically unsustainable levels (Sadovy 1999). Since many species from several families of reef fish spawn in aggregations careful management of this phenomenon is critical if the health of fish populations and hence...
the entire coral reef ecosystem is to be maintained (Sadovy and Domeier 2005). Accordingly, reef fish spawning aggregations are a central component to the coral reef resilience concept, as synthesised in The Nature Conservancy’s Reef Resilience (R2) Toolkit (Domeier et al 2002, R2 2004). This Toolkit addresses methods for minimising stress to coral reefs in order to maximise their ability to resist or recover from coral bleaching caused by climate change (e.g. Obura 2005).

The character of spawning aggregations leaves them highly vulnerable to over exploitation, and there are many examples where fishing has drastically reduced spawning aggregations (e.g. Sala et al 2001, Aguilar-Perera 2006). In view of this, the 3rd IUCN World Conservation Congress adopted Recommendation 3.100 on “Reef-fish spawning aggregations” (IUCN 2004). The recommendation expresses the concern of IUCN’s 1,072 member institutions about the increasing exploitation of reef-fish spawning aggregations in various parts of the world, and about the dramatic ecological and socio-economic effects that such exploitation can lead to. It further urges governments to establish sustainable management programmes for sustaining and protecting reef fish and their spawning aggregations, and also requests a number of organisations to take action to promote and facilitate the conservation and management of fish spawning aggregations, by raising awareness of the long term ecological, economic and societal values of spawning aggregations. The importance of this recommendation was further emphasised by the International Coral Reef Initiative (ICRI 2006).

Reef fish spawning aggregations have been described and/or documented in many locations, including the Caribbean (Sadovy 1999, Bolden 2000), South Pacific (Sadovy 2004), Micronesia (Johannes et al 1999), Australia (Samoilys 1997), the Seychelles (Robinson et al 2004), and East Africa (Samoilys et al 2006, Robinson et al, 2007). Studies have indicated some similarities between spawning aggregations. For example, aggregating fish tend to be of large species from the Serranidae, Lutjanidae, Siganidae, Labridae, Scaridae. The timing and behaviour of aggregations and the physical characteristics of the sites vary (Claydon 2004) and there is still much that we do not know or understand about this critical phenomenon (Domeier et al 2002).

Guidelines on the study of reef fish spawning aggregations have been prepared by Colin et al. (2003). However, it appears very little research has been focused on spawning aggregations in South Asia and the Andaman Sea, and while spawning aggregations are known to occur there seems to be little but anecdotal evidence available beyond some detailed work in Lamu atoll in the Maldives (Sluka 2001a,b,c). There is also little information available on the implications of reef fish reproductive biology for overall reef health as well as reef resource dependent human societies. Consequently, spawning aggregations have frequently not been considered in most aspects of coral reef and fisheries management in South Asia and the Andaman Sea.

**Interview Survey in South Asia and the Andaman Sea**

A project has been initiated by IUCN and CORDIO in collaboration with national and local institutions, aimed at gathering some of the first data on spawning aggregations in the South Asia and Andaman Sea region, with a view to providing information that can support further research on reef fish population dynamics and reef resilience, as well as strengthen management of coral reefs and reef resources.

The objectives of the study are to a) determine which reef fish species form spawning aggregations; b) determine the specific sites of aggregation formation; c) determine the seasonal patterns in spawning aggregations by species; d) determine the level of awareness of spawning aggregations and status of stocks of those species among fishers; e) sensitise fishers and marine resource personnel in south asia on reef fish spawning aggregations and their implications to conservation and sustainable fisheries; and f) provide recommendations for the protection and management of sites of spawning aggregations.
The study is being carried out through interview surveys with fishers, who often are aware of and are fishing spawning aggregations (Johannes 1981), following the guidelines by Colin et al (2003). The survey covers most of the key coral reef areas in the region, including Lakshadweep, Gulf of Mannar and Andaman and Nicobar Islands in India, the Thailand Andaman Sea coast, reef areas in the West and South of Sri Lanka, Aceh in Indonesia, and the Maldives. The interview surveys were initiated around the region between March and August 2007, and are still ongoing (September 2007).

**Preliminary Findings**

Based on discussions with fishers on the northern Sumatra islands of Weh and Aceh, Indonesia, as well as ecological assessments in the area, three potential spawning habitats for Giant Trevally (Caranx ignobilis) have been identified. A detailed survey of fishers knowledge gained from household and field surveys is currently in progress to identify biological and resource use characteristics of these areas and further potential spawning aggregation sites in the region.

In the Maldives surveys have been carried out at Vaavu and Baa Atolls. Fishermen on Vaavu are aware of reef fish spawning aggregations and have identified sites and times for groupers (Epinephelus fuscoguttatus, Plectropomus areolatus and P. pessuliferus). One island in the atoll has an established grouper fishery that targets spawning aggregations on a regular basis. However, reportedly the fishing pressure on these sites has decreased over the years. Preliminary results from islands in Baa atoll indicate most fishermen either do not know about spawning aggregations or are hesitant to provide information. However, this does not include results from some of the islands with a local reef fishery. Field verification and characterisation of some of the sites has been planned.

Interviews in the Union Territory of Lakshadweep, India, conducted in some depth on Minicoy and Agatti islands and opportunistically at Kavaratti and Kadmath islands, indicate limited knowledge of reef fish spawning aggregations. The phenomenon of spawning aggregations is known and has been observed in some tuna baitfish (a number of species are used for tuna bait, including Spratelloides gracilis, S. delicatulus, and Encrasicholina heteroloba, as well as some Caesionidae, Apogonidae, Pomacentridae etc.), but fishers in the area had never heard of or seen reef fish spawning aggregations in larger fish such as groupers and snappers. However, many seemed to know the spawning seasons of pelagic species. This is reflective of the relatively low commercial importance of reef fish in the islands, where the hook and line tuna fishery is the main export earner. The reefs in the archipelago do exhibit many of the characteristics associated with reef fish spawning aggregations elsewhere, and further field surveys as well as more intense interview surveys are underway. In view of the concerns with respect to a growing export fishery targeting high value reef fish, knowledge and management of potential reef fish spawning aggregations is viewed as a high priority.

In the North Andaman region of the Andaman and Nicobar Islands, India, a first preliminary survey provided indications of spawning aggregations including several grouper spawning aggregations (largely Plectropomus spp.) in South Andaman. These aggregations are apparently fished for the lucrative export trade to South East Asia.

Surveys of nine villages in the Gulf of Mannar, India have reported no spawning aggregations in the shallow (0.5-3m) reef areas around the near shore islands. However, four possible fish spawning aggregations have been identified in areas further off shore, 5-10 miles out and at depths between 10 and 20m, with opinions of species, timings, including lunar phase shared among several fishers from different villages. Species reported included primarily the Lethrinidae, but also Siganidae, Lutjanidae, and Scaridae. The area is heavily fished with gillnets and hook and line, and higher catches are reported at the times of aggregation.

Interviews with 190 small-scale fishers in the southern part of Phuket Island and Bulon Island,
Thailand provided some evidence of fish aggregations, although none of the respondents specifically mentioned spawning as a reason for fish aggregating, with some considering feeding the primary reason and others unable to provide an explanation. Sites were characterised as isolated underwater rock-outcrops or rock pinnacles on sandy bottom, and some located in channels between islands. Species observed aggregating included trevally (Carangidae) as well as certain serranids (Epinephelus coioides and E. lanceolatus), lethrinids (Lethrinus lentjan), and scombrids (Rastralliger brachysoma and R. kanagurta).

DISCUSSION

While several potential reef fish spawning aggregation sites have been identified through this study, the results are preliminary and unverified, and thus indicative only. It is clear that although many fishers are not immediately aware of spawning aggregations, others possess at least some knowledge of spawning areas, species and times. Further, as has been found in many other parts of the world, it appears that many aggregations are targeted by fishers in the area. The results are encouraging in the sense that they indicate functional spawning aggregations can still be found in the region.

In view of the intense fishing pressure in many parts of the region, particularly over the past two decades, it is expected that some aggregations have been diminished. As this trend is likely to continue, accurate and reliable information as well increased awareness among managers and policy makers of reef fish spawning aggregations, their ecological significance and vulnerability, are needed in order to design and implement suitable management responses.

Detailed and final results from the surveys will be published in national reports as well as a regional synthesis intended for presentation at the 11th International Coral Reef Symposium in 2008. Information on exact locations and timings of fish spawning aggregations will not be published in the public domain, but will be reported to the Society for the Conservation of Reef Fish Aggregations (SCRFA) database.

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The Coastal Oceans Research and Development in the Indian Ocean (CORDIO) programme was initiated in 1999 as a direct response to the El-Niño caused mass bleaching and mortality of corals in the Indian Ocean in 1998, focusing initially on Eastern Africa, Western Indian Ocean Islands and South Asia. Since the Indian Ocean tsunami in 2004 the programme also covers the Andaman Sea. The objectives of CORDIO are to sustain research on coastal and ocean ecosystems relevant to conserving and sustaining ecosystem function, goods and services; to strengthen social and economic assessment and research for integrated coastal management processes; to improve the livelihoods and well-being of coastal populations; to improve policies and the use of scientific and technical information in local to national and regional policy; to foster networking and integration of science, management and policy; and, finally, to build necessary capacity for meet these objectives.

This is the fifth in the series of CORDIO Status Reports, following ones published in 1999, 2000, 2002 and 2005. This publication reflects the evolution of the CORDIO programme in response to progressing threats from climate change as well as human activities. In all, the report includes 48 articles in sections covering overviews and regional summaries; reports on status, tsunami impact, biological research, fish spawning aggregations, artisanal fisheries, socioeconomics and livelihoods, and education and awareness.

We hope the publication will give the reader a sense of the immense scope of change that ecosystems and people are facing, and the urgent need to respond from local to global levels to assist positive responses and take steps to constrain and minimize the rate of climate change.

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