CORAL REEF DEGRADATION IN THE INDIAN OCEAN

Status Report 2000

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Foreword

The temperatures of the world's oceans are increasing at an accelerating rate. Recent estimates indicate that the magnitude of these increases might be as much as several degrees over the next century and undoubtedly, the impacts of these changes on the Earth's ecosystems are likely to become increasingly obvious. Coral reefs have already shown dramatic responses to the increasing ocean temperatures. Under normal temperature conditions, reef-building corals, which form the foundation of coral reefs, are living very near the maximum sea temperatures that they can tolerate. If they are exposed to even modest increases in sea temperatures, perhaps only 1° C - 2° C, they become stressed and often 'bleach'. This bleaching of corals is a response to stress, and it occurs when the symbiotic unicellular algae (zooxanthellae) that lives within the tissues of the coral polyp, are expelled or lost. The coral can survive for short periods without these zooxanthellae but unless the stress that caused the bleaching subsides and new zooxanthellae are incorporated into the tissue of the coral, the coral will die. For several months in early 1998, the temperature of surface waters (< 10 m) over much of the world's tropical oceans increased between 3° C and 5° C. As a result, corals on reefs throughout the world bleached and, unfortunately, many died. The mortality of corals was particularly serious in the central and western Indian Ocean, where as many as 50% to 95% of all corals died.

The impacts of the 1998 coral mortality are a matter of great concern. Coral reefs are one of the most diverse, productive and complex ecosystems on the planet. They are the home of hundreds of thousands of species, many of which are unknown to science. They are highly productive, providing food and shelter for most of the fish species caught in shallow tropical coastal waters. As a consequence, the impacts of such widespread coral mortality will have a direct bearing on critical marine and coastal biodiversity and fragile ecosystems that are affected. From a socio-economic perspective, the impacts of coral mortality are far reaching, affecting food security as well as local and national economies that are dependent on reef-based tourism and industry. It is particularly worrying that models, based on the IPCC Scenario A (doubling of atmospheric carbon dioxide levels by 2100), that forecast future climatic conditions predict that the temperature tolerances of reef-building corals will be exceeded permanently within the next few decades. Considering these grave predictions, it is a matter of great importance that we, as custodians of the planet, learn all that we can about the ecological and socio-economic implications of climate change. The CORDIO Program was launched in 1999 to obtain valuable data that will enable us to determine the consequences of increasing ocean temperatures. This report presents the results of work conducted within the first 18 months of the CORDIO Program and makes an important contribution to our knowledge of the problems caused by climate change. In gathering such data the CORDIO Program fulfils an important role and warrants full support.

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Address by the Global Coral Reef Monitoring Network (GCRMN)

During the severe El Niño and La Niña events of 1998, many coral reefs in the Indian Ocean and the Arabian region suffered unprecedented levels of coral bleaching and mortality. It was under El Niño conditions that severe bleaching occurred in the Indian Ocean countries of Comores, India, Kenya, the Maldives, Oman, Seychelles, Sri Lanka, and Tanzania, and the UK territories of the Chagos banks. During the La Niña events in the second half of the year, there was extensive bleaching in the Arabian Gulf and reefs of the southern Red Sea. Coral mortality exceeded 50% on many of these reefs, and in some countries like the Maldives and Seychelles, virtually all corals in shallow water died.

These events sounded the alarm for the future of coral reefs and particularly for the many communities of people in the Indian Ocean dependent on these reefs for their livelihoods. But there was a glaring problem; there was insufficient information on the extent and status of the reefs in the wider Indian Ocean and also very little understanding of how people of the region interacted with these reefs. Our knowledge of coral reefs of the Indian Ocean was much poorer than of those of Southeast Asia, the Pacific and the Caribbean, where considerable research and monitoring has been conducted.

This lack of knowledge was immediately recognized by Sweden and the World Bank and they reacted with amazing speed to implement the CORDIO project (Coral Reef Degradation in the Indian Ocean) with funding from the governments of Sweden and the Netherlands through a Trust Fund with the World Bank. CORDIO has five basic elements: to assess the extent of the bleaching; measure rates of recovery; determine whether rehabilitation is necessary and possible; assess the socio-economic impacts; and determine what alternative livelihood activities are possible for the people of the region. Thus, CORDIO became the first large-scale, ocean-wide project that took a holistic approach to assess coral reefs and interact with the people who depend on them.

The CORDIO Program is now being welcomed as an operational component of the International Coral Reef Initiative (ICRI), which was formed as a direct response to alarming reports of increasing degradation of coral reefs. Such concerns were a theme at the United Nations Conference on Environment and Development in 1992, and was again stressed by Small Island Developing States at a meeting in Barbados, 1994. The international community responded by forming ICRI with key countries being Australia, France, Jamaica, Japan, Philippines, Sweden, UK and USA along with the World Bank, UNEP, IOC/UNESCO, UNDP, IUCN and WWF. The first operational unit of ICRI was initiated in 1995 with the formation of the Global Coral Reef Monitoring Network (GCRMN). The GCRMN aims to determine the status of coral reefs around the world by networking existing monitoring activities, establishing new monitoring where necessary, and raising awareness amongst all stakeholders of the need for enhanced reef conservation. Thus, the goals of CORDIO and the GCRMN have considerable overlap and a strategic partnership was formed early to maximise efforts in the Indian Ocean.

The GCRMN operates through independent Nodes of neighbouring countries that co-ordinate training, monitoring, data analysis and report writing to determine the status of coral reefs. There are three functional Nodes in the Indian Ocean: South Asia, Western Indian Ocean, and Eastern Africa. While there had been considerable coral reef monitoring in Eastern Africa (especially Kenya and Tanzania), a co-ordinating Node had not been formed. This role was taken on by CORDIO which is funding and co-ordinating monitoring for both the GCRMN and CORDIO within Kenya, Mozambique and Tanzania, with South Africa assisting and hopefully Somalia to participate in the future.

Thus, the partnership between the CORDIO and GCRMN programmes has been established on the basis of mutual goals and methods. The GCRMN will assist CORDIO with the first two objectives: assessing reef status and observing recovery. What has been established under CORDIO should serve as an example for other parts of the world. This is a project that simultaneously tackles coral reef ecosystem problems over oceanic scales and integrates this with socio-economic assessment of the user communities. A search for solutions is an integral feature within the project. I was invited to participate in the meeting held in Lamu, Kenya in February 2000 and was impressed with the strong sense of teamwork and co-operation amongst participants from all Indian Ocean countries, external agencies and donors. Thus, I can predict that the CORDIO project will serve as a new model to tackle global environmental and social problems such as those caused by climate change, by sharing knowledge and experience amongst the concerned scientific and policy-making community over much larger scales than many current projects that work only with small areas within a single country.

Clive Wilkinson
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Executive Summary

The World Conservation Strategy (IUCN/UNEP/WWF) identifies coral reefs as one of the essential global life support systems necessary for food production, health and other aspects of human survival and sustainable development. However, their future is being threatened by overexploitation, coastal development, land-based pollution and moreover, by global climate change. The last two decades has been the warmest period ever recorded during which, 1998 was the hottest single year. As a consequence, the mean temperatures of the world's oceans are increasing at an accelerating rate which, at present, is approximately 2°C per century. In addition, 1998 witnessed the strongest El Niño ever recorded. The long period for which this El Niño prevailed and its extreme magnitude exacerbated already increased sea-surface temperatures. In 1998, as a consequence of these El Niño driven increases in sea temperature, corals throughout the world's tropical oceans suffered the most severe mass bleaching event ever reported. As a direct response to the massive degradation caused by this bleaching event, the CORDIO Program (Coral Degradation in the Indian Ocean) was initiated early in 1999. CORDIO's objectives are to determine the long-term impacts of degradation of coral reefs on coastal communities and national economies. Research under CORDIO focuses on:

- the bio-physical impacts of coral degradation as a result of bleaching and other disturbances, and the long term prospects of recovery,
- the socio-economic impacts of coral mortality and options for mitigating these through management and development of alternative livelihoods,
- the prospects of restoration and rehabilitation of reefs to accelerate their ecological and economic recovery

This second annual status report follows on from the first report that was based on initial assessments of coral mortality and presents the results of CORDIO's first full year of monitoring and research.

BIOPHYSICAL EFFECTS

Monitoring of the biophysical effects of the El Niño was conducted in 11 countries in East Africa (Kenya, Tanzania, Mozambique), South Asia (Sri Lanka, Maldives, India), and the central Indian Ocean islands (Seychelles, Mauritius, Reunion, Comoros, Madagascar). Reef degradation was most pronounced on the reefs of mainland East Africa and on the islands of Socotra, Comoros, Mayotte, Seychelles, and the islands of South Asia. In these areas coral mortality generally exceeded 50% and reached 95% or more in many locations. Large areas of coral reefs in the central and western Indian Ocean have undergone a total transformation where the former coral reef is now covered by thick carpets of filamentous algae.

Areas that suffered minimal bleaching and mortality included the more southerly islands of Mauritius and Reunion, where cyclonic activity retarded increases in sea-surface temperature. Also, in other areas, such as western Unguja and Pemba Islands and on some reefs of the Seychelles Bank, upwelling of cooler waters from off the continental shelf and bank slopes moderated increased sea surface temperatures and prevented the subsequent bleaching of corals.

No sites were completely unaffected by bleaching. Although bleaching and mortality among corals was observed at depths between 40 m and 50 m, generally the damage was much greater in shallow waters. Corals growing in shallow areas with restricted water circulation are generally considered to be more tolerant of high sea temperatures than corals situated in deeper waters. However, the magnitude and duration of increases in sea temperature caused by the 1998 El Niño event overwhelmed any capacity possessed by corals growing in shallow water to tolerate such conditions. Subsequently, it was these corals that suffered disproportionately high mortality.

Soon after, coral competitors (algae and some zoanthids (“false corals”)) began to occupy space made available by the massive coral mortality. Initial investigations of the fish communities associated with the reefs indicated that herbivorous species increased in abundance while the abundance of corallivorous species declined.

In several areas (Maldives, Seychelles, Chagos, Socotra) the dead coral skeletons have been broken down into rubble and no longer prevent oceanic waves from penetrating into sheltered waters in the lee of the reef. Subsequently, erosion of the beaches in these areas appears to have increased.

RECOVERY

Coral recovery is expected to occur through re-growth and/or fragmentation of colonies that survived the mortality event and also through recruitment of planula larvae from the water column. With the exception of some reefs of Maldives, recovery through the recruitment of larvae has been minimal. Re-growth of surviving corals has occurred on some reefs, but generally has not been significant. The degree to which recovery of reefs will be mediated by the influx of coral larvae from more distant reefs is currently unknown largely because so few reefs have escaped the mass bleaching event relatively unscathed and could potentially serve as sources of larvae.

Priority areas for biophysical research for the year 2000 include: identification of source reefs (reefs from which a supply of coral larvae could originate) and identification of threats to these reefs, assessment of processes of recovery and degradation of reefs in affected areas, the affect that local species extinction exerts on the structure of recovering reef communities with respect to dispersal and influx of larvae, and the promotion of regional synoptic research projects.

SOCIO-ECONOMIC EFFECTS

Socio-economic monitoring was conducted in seven countries (Madagascar, Comoros, Tanzania, Kenya, Maldives, Sri Lanka, India). Reef fisheries have not yet shown any obvious declines attributable to the El Niño induced coral bleaching. This is probably attributable to the fact that, at present, the structural integrity of many of these reefs remains largely intact and continues to provide the structure needed by fish communities. However, as soon as the reef framework becomes degraded by erosion, a decline in fish catches is expected. Because a substantial proportion of the coastal fishery is subsistence fishing, the impact of declining fish stocks may threaten food security of a large proportion of the coastal population in each of these countries.

Tourist divers interviewed at Zanzibar, Mombasa and Maldives indicated the state of the reef would affect their choice of destination and whether to dive, resulting in significant estimated economic and financial losses. However, differences in the tourism market and product among destinations will greatly affect their ability to offer alternative activities to preserve income. The principal questions derived from these investigations identify the importance of links between the biophysical and socio-economic environments and the need to adapt both the fisheries and tourism industries accordingly.

PROGRAM FOR YEAR 2000

CORDIO held its second annual workshop, “Coral Reef Degradation in the Indian Ocean (CORDIO): Progress to Date and Directions for 2000 and Beyond” between February 10th and 12th, 2000, in Lamu, Kenya. Thirty-one participants attended the workshop to review status of coral reefs in the region, and the progress of research to date. Working group discussions were held in five areas:

1. Climate change and long term trends
2. Regional research
3. Socio-economic considerations
4. Management and awareness issues
5. Database requirements

The conclusions of the working groups are included in this report. Of particular importance were discussions held on the novel challenges to research and management posed by climate change and the false vision of a pristine historical reef state that has underpinned most current research. What is needed is almost an emerging science that develops innovative ways to measure and interpret the influence of climate change on ecological systems. Of utmost importance is the compilation of long-term records (e.g. meteorological data) to establish long-term trends, measurement of simple variables (such as beach width) over large spatial scales to establish synoptic baselines and greater dedication to monitoring key processes such as recruitment rather than gross variables such as coral cover.

The increasing frequency and severity of mass coral bleaching events is the single most dramatic and tangible demonstration of global climate change. Predictions of the consequences of gradual rises in temperature and sea level have been eclipsed by the effects of violent fluctuations of natural climatic cycles. El Niño events of a magnitude previously experienced only once every 100 years are now occurring at intervals ranging between three and 20 years, with severe impacts extending beyond their original limits within the Eastern Pacific to the equatorial and monsoon regions of the Indian Ocean.

The development of alternative livelihoods for subsistence fishermen and others dependent on reef resources is an important area for research and development. Aquaculture of commercially valuable organisms, expansion of marine resource use to include a wider variety of species thus alleviating the pressure on target species and the development of terrestrial opportunities to improve livelihoods in terms of income and/or food security is desirable. However, participation of all stakeholders in the process of development and implementation is paramount for the success of such schemes

Coral reefs are among the first ecosystems on the planet that are being dramatically affected by global climate change. Their high biodiversity and their fundamental importance to coastal populations raises the need for problem solving research and management. Bleaching of coral as a result of increased sea temperatures caused by global climate change offers a unique opportunity to investigate the effects of climate change on an ecosystem scale and within a relatively small time frame. Conseqvently, CORDIO is establishing a process for addressing this global challenge in the western Indian Ocean. Despite the growing weight of local and global threats, CORDIO, in 2000 and 2001, will attempt to improve the fate of coral reefs through research and development projects that support management and improves the livelihoods of people dependent on coral reef ecosystems.

Synthesis

PREAMBLE

The CORDIO Program (Coral Degradation in the Indian Ocean) was initiated early in 1999 as a direct response to massive coral reef degradation caused by the 1997-98 El Niño induced mass bleaching event, the severest such event on record. Its objectives are to assess the long-term impacts of the event on coral reefs, coastal communities and national economies. The results of initial assessments of coral mortality were presented in the first CORDIO Status Report published in the first few months of the project. The results of the first full year of monitoring and research are presented here in this second status report.

CORDIO is a collaborative program initiated to determine the implications of global climate change and other causes of coral reef degradation in the western Indian Ocean. The foundation upon which the program is built is the collaboration between regional institutions with additional input from external institutions. Research under CORDIO focuses on:

- the bio-physical impacts of coral degradation as a result of bleaching and other disturbances, and the long term prospects of recovery,
- the socio-economic impacts of coral mortality and options for mitigating these through management and development of alternative livelihoods,
- the prospects of restoration and rehabilitation of reefs to accelerate their ecological and economic recovery

Increases in global sea temperatures and El Niño induced coral mortality in the Indian Ocean

The last two decades has been the warmest period ever recorded during which 1998 was the hottest single year. As a consequence, the mean temperatures of the world's oceans are increasing at an accelerating rate which, at present, is approximately 2°C per century. Moreover, 1998 witnessed the strongest El Niño ever recorded and, as a result, corals throughout the world's tropical oceans suffered extensive bleaching and mortality. The El Niño Southern Oscillation is a global phenomenon, driven by evaporation of equatorial sea-surface waters and the dynamics of current and wind movements. During El Niño the normal pattern of equatorial trade winds is altered and their strength is reduced resulting in calmer tropical waters. As a consequence, solar radiation that would otherwise be dissipated by wind-driven water movement is absorbed increasing sea temperatures and further exacerbating background rises in global ocean temperatures. The long period for which the 1997-98 El Niño prevailed and its extreme magnitude resulted in these elevated temperatures persisting and subsequently causing massive mortality of corals on reefs throughout the Indo-Pacific and Caribbean. Some of the worst affected reefs were those situated along the coast of East Africa and in the central Indian Ocean from which estimates of coral mortality commonly exceeded 90%. Once dead, algae rapidly colonise the exposed coral skeletons and, in addition, bio-eroding organisms degrade skeletons reducing the reef to rubble beds.

RESULTS OF 1999 MONITORING

Monitoring of the biophysical effects of the El Niño was conducted in 11 countries around the western Indian Ocean namely, Kenya, Tanzania, Mozambique Sri Lanka, Maldives, India Seychelles, Mauritius, Reunion, Comoros, Madagascar. Socio-economic impacts were determined in Madagascar, Comoros, Tanzania, Kenya, Maldives, Sri Lanka, India.

Biophysical

Generally, region-wide monitoring of coral reefs during 1999 recorded high levels of coral mortality and reef degradation as a result of El Niño induced elevated sea temperatures (Box 1). Indeed, no sites were completely unaffected by bleaching. Reef degradation was most pronounced on the reefs of mainland East Africa, Socotra, Comoros, Mayotte, Seychelles and the islands of South Asia, where coral mortality was generally exceeded 50% and in some locations reached 95%.

Lower levels of mortality were recorded from corals growing on fore-reef slopes or on the periphery of channels where localised upwellings or influxes of cooler water from the deeper ocean decreased the absolute magnitude and duration of local high sea temperatures. This occurred on some reefs on the Seychelles Bank and also along the west coast of Zanzibar.

The influence of local conditions on coral mortality was also evident in Mauritius/Reunion. A cyclone, present during a two or three week period when dramatically increased sea temperatures prevailed elsewhere, generated high winds and waves that mixed the water column and reduced sea surface temperatures to a tolerable level. Furthermore, the high cloud cover associated with the cyclone reduced the exposure of corals high levels of solar radiation further moderating the overall effect of elevated sea temperatures on coral mortality.

Corals situated in shallow water, particularly those in enclosed lagoons with restricted water circulation, are generally considered to be more tolerant of high temperatures than corals growing in deeper water. However, it was these corals that suffered greatest mortality. It appears that the magnitude of the stress overwhelmed any adaptation these corals may possess whereas corals growing in deeper or more exposed locations may have received some respite through occasional influxes of cooler water.

Surveys of associated reef biota, such as fish, revealed an increased number of herbivorous species on the reefs that suffered mass mortality of coral. However, although it is certain that the degradation of the reef framework will result in changes to community structure and abundance of higher trophic groups, the long-term effects of this event on these groups are not yet clear.

Box 1. Summary of results from the initial assessments made during the first half of 1999

East Africa:

- Several areas suffered extremely high mortality of corals. For example, surveys in Kiunga and Malindi (Kenya), Misali and Mafia (Tanzania), and Pemba and Inhacca (Mozambique) determined that between 90% and 100% of corals died as a result of exposure to water temperatures that exceeded 32° C, mainly in March and April 1998. Mortality of corals culminated approximately mid-May. However, some corals continued to die until October.
- Following the bleaching event in 1998, it appears that, in general, coral cover in most areas along the coast of Kenya, Tanzania and northern Mozambique was reduced to between 10% and 50% of previous levels. In some areas the reduction of the live coral cover was significantly greater.
- Initial investigations indicate that the fish communities associated with reefs were affected by the coral mortality and that typically, herbivorous species increased while corallivorous species decreased.
- Cover of algal turf on the bleached and dead reefs increased significantly in most affected areas. For example, on Kenyan reefs, algal cover in many areas increased 200% as a result of the newly available substrate.

Indian Ocean Islands:

- Coral mortality ranged from 50% to 90% over extensive areas of shallow reefs in Seychelles, Comoros, Madagascar and Chagos. In some areas (around Mahe, Seychelles) the mortality was almost 100%.
- Algal turf covered coral reefs throughout much of the region by the end of 1998.
- Monitoring of potentially toxic epiphytic dinoflagellates showed drastically increased concentrations in areas where the corals had died.
- By early 1999, much of the dead coral in Chagos was reported to be eroded to rubble preventing recolonisation. At Socotra, much of the coral rubble was washed ashore and could be found in piles on the beach.
- Preliminary assessments of the reef fish communities in Chagos indicated that abundance and diversity was less than 25% of their former levels.

South Asia:

- Bleaching was reported to depths of 40 m in Sri Lanka and 30 m in Maldives resulting from water temperatures of approximately 35° C during the period between April and June 1998.
- In many areas in Sri Lanka and Maldives, nearly 90% of all corals died. At Hikaduwa and Bar Reefs in Sri Lanka, close to 100% of corals died and at the end of 1998, these reefs were then covered by thick algal turf. In India, surveys indicated mortality between 50% and 90% on the reefs in the Gulf of Mannar, Andaman and Lakshadweep Islands.
- Assessments of the reef fish communities showed drastic reductions in butterfly fish numbers on Sri Lankan reefs.

Recovery

Recovery of corals and indeed coral reefs is dependent on re-growth of whole or fragments of surviving colonies or through settlement and recruitment of planula larvae from the water column. More than 18 months after the bleaching event, there is little evidence of recovery or coral recruitment on the majority of reefs surveyed. Despite small numbers of recruits being recorded on a few reefs, recruitment on most reefs surveyed was generally low. The influx of planula larvae and subsequent recruitment of corals to reefs degraded by coral bleaching will depend largely on the spatial distribution of reproducing adult colonies on less affected reefs (source reefs).

The following questions arose from the first year of studies and are the targets of research projects proposed for 2000:

- *Source reefs and their ecology* – Where are the reefs that survived the bleaching event situated? What localised environmental conditions or physiological factors enabled the corals on these reefs to survive? Are these reefs in a position to assist recovery of heavily impacted reefs through the provision of larvae?
- *Recovery versus degradation* – How long is recovery likely to take and what will be the contribution of re-growth vs. recruitment towards recovery? How is the rate of bioerosion and other degrading processes affected by mass mortality of corals? In view of the widespread destruction of the major contributors to reef growth, will the rate of reef degradation overwhelm processes of calcium carbonate ($CaCO_3$) accretion?
- *Extinction and dispersal* – Are particular species of coral and/or reef associated organisms locally or regionally extinct? How will widespread mortality of corals affect rates and taxonomic patterns of recruitment at local and regional scales?
- *Genetic implications* – Has there been a decrease in the genetic diversity of corals as a result of the widespread mortality? Do different colonies of the same

species exhibit different tolerances to factors related to climate change?
- *Regional synoptic patterns* – How will the outcomes of questions posed above vary according to local and regional scales?

Socio-economic

In the central and western Indian Ocean healthy coral reefs are essential for coastal fisheries and tourism. Subsequently, initial assessments socio-economic impacts of the mass mortality of corals focussed on these activities. Reef fisheries have not shown any obvious declines attributable to the El Niño induced bleaching and mortality of coral. This is because, at present, the structural integrity and topographic complexity of the reefs that attracts such a diverse fish assemblage remains largely intact. However, as soon as the reef framework becomes degraded by erosion a decline in fish catches is predicted. Moreover, because a substantial proportion of the coastal fishery is conducted at a subsistence level, declining fish stocks will undoubtedly threaten food security among large portions of the coastal population.

Surveys of tourist divers determined that there is already a potential negative impact of reef degradation on the tourist industry. Divers interviewed at Zanzibar, Mombasa and Maldives indicated the state of the reef would affect their choice of destination and activities, resulting in economic and financial losses for tour operators and countries that were significantly affected by bleaching. However, local differences in tourism markets will affect long-term impacts. For example, Maldives, which at present is a popular destination for tourists, has high occupancy rates and, as a result, has not suffered excessively from the effects of the bleaching event because reduced numbers of dive tourists have been compensated for by other types of visitors (e.g. Honeymooners). However, the tourism industry in East Africa, unlike Maldives, is not saturated and therefore, slack caused by declines in the number of tourists seeking reef related activities is not compensated for by an influx of tourists from other target groups resulting in a greater impact for the national economies of these countries.

The principal questions posed by the socio-economics projects include:

- *Links between biophysical and socio-economic impacts* – What are the links between reef health, fisheries production and fishermen's welfare? In this context, what types of fisheries and reef management might best address the problem of reef degradation?
- *Impacts of reef degradation on tourism*– How does the perception of reef health by tourists affect their choice of destination and willingness to pay for protection? In particular, what features of reefs attract tourists? Can changes in marketing strategies reduce losses in tourism caused by reef degradation?
- *National economic impacts* – How will potential losses in fisheries and tourism affect the national economies of countries affected by coral mortality?

CORDIO FUTURE DIRECTIONS

The CORDIO program held its second annual workshop, "Coral Reef Degradation in the Indian Ocean (CORDIO): Progress to Date and Directions for 2000 and Beyond" between February 10th and 12th, 2000, in Lamu, Kenya. Thirty-one participants attended the workshop representing CORDIO projects from Sri Lanka, Maldives, Mauritius, Seychelles, Reunion, Mozambique, Tanzania and Kenya, and included participating scientists from Sweden, Finland and the UK, and regional and international organisations such as UNEP, ICRI, IUCN and WWF.

The principal objectives of the workshop were to:

a) Review the status of coral reefs in the western Indian Ocean, particularly with regard to recovery or further degradation since the El Niño-related coral bleaching and mortality that occurred during the first half of 1998,
b) Present preliminary findings on the current and potential socio-economic impacts of reef degradation on tourism and fisheries, and
c) Outline future directions for productive research.

Objectives a) and b) are summarised in the previous sections and form the basis of contributions in the remainder of this volume.

Objective c) led to discussions in a number of thematic areas, representing key areas for expansion and development of CORDIO foci. Five working groups were held:

- *Climate change and long term trends* - issues of climate change and the need for innovative ways to measure and interpret its effects on ecological systems; the need to identify, describe and disseminate the common methodologies and databases already in use in the region to assist integration.
- *Regional research proposals* - the need for regional integration among research projects, with new emphasis on coral recruitment, bioerosion and impacts on fish community structure.
- *Socio-economic considerations* - the need to closely link socio-economic and biophysical research to understand fully the effect of changes in the status of reefs on resource use and economic activity.
- *Management and awareness* - the need for clear recommendations for management of bleached and degraded reefs, to assist and assure coral reef managers that recovery is possible and indicate practises that might assist in rehabilitation.
- *Database needs* – the need to develop use of practical database techniques for data archiving and analysis, to promote regional assessments in biophysical and fishery areas.

Climate change and recognition of "shifting baselines"

Climate change undermines previously held notions of stability and constancy in global environments and past practises in monitoring recent change against some presumed pristine state. Consequently, a clear need for innovative ways to measure and interpret the influence of climate change on ecological systems exists.

The increasing frequency and severity of mass coral bleaching events is the single most dramatic and tangible demonstration of global climate change. Predictions of the consequences of gradual rises in temperature and sea level have been eclipsed by the effects of violent fluctuations of natural climatic cycles. El Niño events of a magnitude previously experienced only once every 100 years are now occurring at intervals ranging between three and 20 years, with severe impacts extending beyond their original limits within the Eastern Pacific to the equatorial and monsoon regions of the Indian Ocean. Fuelled by global climate change, on the smaller scale of sites and ecosystems, dramatically different fluctuations in local climate have a fundamental influence on survival of their biota.

The El Niño induced bleaching and mortality of corals throughout the Indian Ocean was an event with both regional and local dimensions. Although massive degradation of coral reefs was reported throughout the areas monitored by CORDIO, at least three local factors appeared to affect the magnitude and duration of prevailing environmental conditions causing variations in the extent of coral mortality namely, local upwelling of deeper waters on fore-reef and platform slopes, high-current effects of water mixing in channel areas and shading of the sea-surface and water column mixing during cyclonic activity.

It is possible that local features that modify larger scale climate phenomena, could be used as one of the key tools for practical conservation of biodiversity and resources in the future. Research into the types and modes of action of local features that may mitigate the fluctuations induced by climate change is an immediate need for the region. The information derived will be critical to the development of management interventions for other threats acting at both regional and local scales and in the use of protection measures for genetic and biodiversity source reefs.

Emerging issues in the research and management of coral reefs under the threat of global climate change include the following:

- Trends recorded during the small 30-year window of coral reef research are a short component of longer-term, "shifting baselines" in environmental conditions. Thus, it is critical to measure appropriate variables using larger temporal (historical climate) and spatial scales (regional) against which to interpret field data.

- Understanding ecological change requires greater attention to ecological processes (recruitment, herbivory, bioerosion), not just static measures such as cover and biomass.
- Connectivity of coral reefs through ocean currents and larval dispersal will fundamentally affect recovery and survival of coral reefs. Knowledge of genetic source/sink relationships is essential to effective management.
- What role will protected areas play in local-regional contexts? Can they adequately protect source reefs and provide stepping-stones for larvae? What area or network of protected areas is needed?

A regional framework for research

The widespread impacts of the 1998 El Niño on reefs of the Indian Ocean have emphasised the need for a regional scale strategy and implementation network for research concerning environmental change. While research is generally implemented on a small scale at individual study sites, to obtain a broad-scale perspective projects must be replicated throughout the region. Thus, CORDIO is building a network of collaborating scientists and institutions along the following lines:

- Development of a research program that builds on the capacities and expertise available at local and international levels, addressing information needs with respect to global climate change.
- Development of local and national research capacity of institutes and scientists to conduct investigations in their areas of expertise and training.
- Collaboration and co-ordination with external scientists with historical and developing interests in the region, co-ordinated with appropriate local and national institutes.
- Specific regional research topics identified for proposal development for the year 2000 include coral recruitment, bioerosion, and impacts on fish community structure.

Socio-economics and alternative livelihoods

Socio-economic research within CORDIO is based on a model explicitly linking environmental and resource quality (measured using biophysical variables) with economic use and value. For example, in the case of fisheries, the model links coral bleaching habitat complexity fish community structure fisheries livelihoods. This mechanistic analysis of the effects of degradation on resource use will provide clearer analyses of peoples' dependence on the environment, and options for mitigation. Similar investigations on the reliance of tourism on environmental quality will help address practical questions on what can be done to minimise economic losses following from reef degradation.

As a proactive activity in the nature of research and development, CORDIO is establishing research projects on alternative livelihoods for subsistence fishermen. Multiple threats have already reduced fisheries productivity and ecological health of many reefs in the region, with El Niño induced degradation adding another unpredictable component. Against this background, project proposals have been written in a number of areas, such as investigating fishing community household production systems and accessible options for alternatives and development of small-scale mariculture of fish and crabs. A fundamental approach in these projects will be to develop participatory trials with the eventual users, that is, incorporating fishermen as research partners, to ensure local accessibility of the methods.

Management and awareness

The feasibility and success of management interventions on bleached and degraded reefs will become increasingly important as more reefs become affected. In contrast to historical protection of pristine reefs for conservation, the establishment of protected areas on degraded reefs will become a necessary strategy to promote recovery. In addition, many management agencies may feel disillusioned by the growing awareness and predictions of coral reef decline and subsequently, need practical advice on options for management. CORDIO will take actions to develop a short practical manual for managers to guide them in protecting degraded reefs and also to heighten public awareness and education of coastal populations throughout the region.

Databases

The collection of large volumes of monitoring and research data from a region such as the western Indian Ocean requires careful attention to comparability and standardisation. Currently the island nations are using a consistent set of databases for reef monitoring data (ARMDES), under the auspices of the Indian Ocean Commission (COI). The other CORDIO countries and regions will investigate existing options for suitable databases, in co-ordination with GCRMN and ICRI developments (e.g. ReefBase). An additional need for fisheries catch databases relevant to artisanal fisheries has also been identified with steps being taken to develop a basic version of such a database in Kenya.

CORAL REEFS IN A GLOBAL CONTEXT

Coral reefs are among the first ecosystems on the planet that are being dramatically affected by global climate change. Their high biodiversity and their fundamental importance to coastal populations raises the need for problem solving research and management. Bleaching of coral as a result of increased sea temperatures caused by global climate change offers a unique opportunity to investigate the effects of climate change on an ecosystem scale and within a relatively small time frame. Furthermore, it enables us to predict future impacts on reefs and other ecosystems and devise interventions that prevent dramatic reef degradation and the consequent human suffering.

In addition, marine ecosystems are fundamentally structured at regional scales by the movement of ocean currents and the genetic linkages among locations through the dispersal of planktonic seedlings and larvae. The regional approach being built by CORDIO offers a unique opportunity to address regional issues in conservation and research. Therefore, the following priorities will be incorporated in developing the future activities under CORDIO:

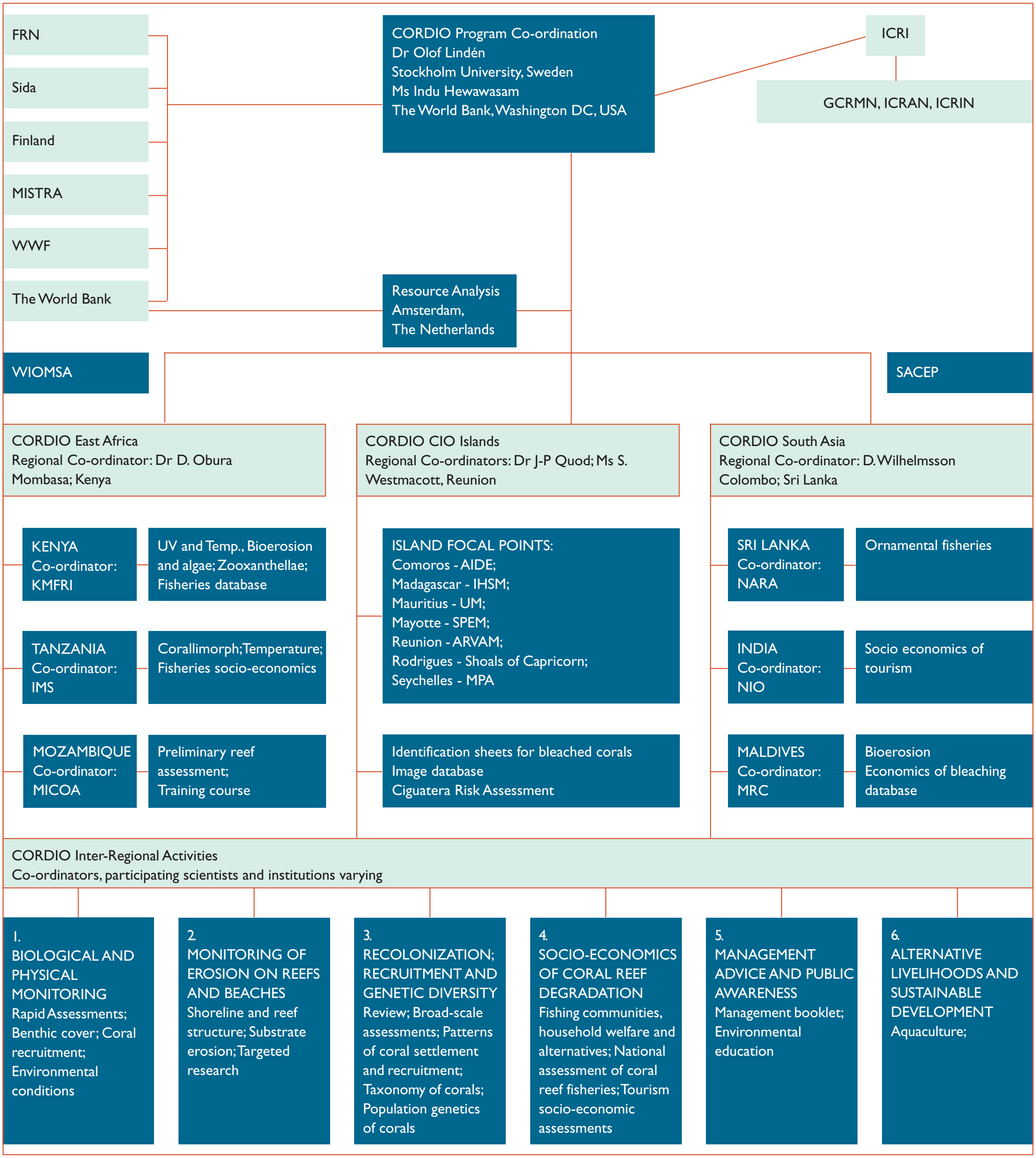
- Identification of the links between global and local phenomena with respect to climate change.
- Development of regionalized research programs in the key areas of coral reef monitoring, degradation and recovery processes, management and environmental awareness.
- Support for developing nation institutions and scientists and their ability to generate results and information of national and global significance.

THE CORDIO PROGRAM

CORDIO was started as a 3-year program, to run from 1999 to 2001. In one year of operation, activity has grown to include 12 national monitoring programs and approximately 25 research projects. It has served as a node for the crystallisation of a number of existing projects and proposals, it's strengths being in its flexibility and promotion of open collaborative activities. CORDIO has been successful so far in the solicitation of research proposals and in channelling funds from donors to the recipient researchers and institutions. The future of CORDIO will be determined along three major lines:

- The fate of coral reefs under the growing weight of local and global threats, particularly those caused by global climate change.
- The identification of emerging national priorities and capacity needs in coral reef research and protection in member countries.
- The construction of new links and collaboration between countries and donors and also with other organisations.

The conceptual structure of CORDIO is illustrated in a flow diagram presented below. CORDIO collaborates closely with regional and international programs such as Global Coral Reef Monitoring Network (GCRMN) and International Coral Reef Initiative (ICRI). CORDIO is implementing GCRMNs local program in the western Indian Ocean and has been invited to become a program under the ICRI framework. Furthermore, in the implementation of the research program, CORDIO is collaborating with organisations such as IUCN, WWF, WIOMSA (in the western Indian Ocean), COI (in the central Indian Ocean Islands) and with SACEP (in the South Asia Region)


FRN
Sida
Finland
MISTRA
WWF
The World Bank
CORDIO Program Co-ordination
Dr Olof Lindén
Stockholm University, Sweden
Ms Indu Hewawasam
The World Bank, Washington DC, USA
ICRI
GCRMN, ICRAN, ICRIN
Resource Analysis
Amsterdam,
The Netherlands
WIOMSA
SACEP
CORDIO East Africa
Regional Co-ordinator: Dr D. Obura
Mombasa; Kenya
CORDIO CIO Islands
Regional Co-ordinators: Dr J-P Quod; Ms S. Westmacott, Reunion
CORDIO South Asia
Regional Co-ordinator: D. Wilhelmsson
Colombo; Sri Lanka
KENYA
Co-ordinator:
KMFRI
UV and Temp., Bioerosion and algae; Zooxanthellae; Fisheries database
TANZANIA
Co-ordinator:
IMS
Corallimorph; Temperature; Fisheries socio-economics
MOZAMBIQUE
Co-ordinator:
MICOA
Preliminary reef assessment;
Training course
ISLAND FOCAL POINTS:
Comoros - AIDE;
Madagascar - IHSM;
Mauritius - UM;
Mayotte - SPEM;
Reunion - ARVAM;
Rodrigues - Shoals of Capricorn;
Seychelles - MPA
Identification sheets for bleached corals
Image database
Ciguatera Risk Assessment
SRI LANKA
Co-ordinator:
NARA
Ornamental fisheries
INDIA
Co-ordinator:
NIO
Socio economics of tourism
MALDIVES
Co-ordinator:
MRC
Bioerosion
Economics of bleaching database
CORDIO Inter-Regional Activities
Co-ordinators, participating scientists and institutions varying
1.
BIOLOGICAL AND PHYSICAL MONITORING
Rapid Assessments; Benthic cover; Coral recruitment; Environmental conditions
2.
MONITORING OF EROSION ON REEFS AND BEACHES
Shoreline and reef structure; Substrate erosion; Targeted research
3.
RECOLONIZATION; RECRUITMENT AND GENETIC DIVERSITY
Review; Broad-scale assessments; Patterns of coral settlement and recruitment; Taxonomy of corals; Population genetics of corals
4.
SOCIO-ECONOMICS OF CORAL REEF DEGRADATION
Fishing communities, household welfare and alternatives; National assessment of coral reef fisheries; Tourism socio-economic assessments
5.
MANAGEMENT ADVICE AND PUBLIC AWARENESS
Management booklet; Environmental education
6.
ALTERNATIVE LIVELIHOODS AND SUSTAINABLE DEVELOPMENT
Aquaculture;

SECTION I
Status Reports

East Africa – Summary

DAVID OBURA
CORDIO-East Africa, Mombasa Kenya

East Africa, comprising Kenya, Tanzania, Mozambique and South Africa, was greatly affected by coral bleaching and mortality as a result of the 1997-98 El Niño. Bleaching most probably started in East Africa in late February – early March, in the south, and finishing in May in the north, following the movement of the sun and the Inter-Tropical Convergence Zone. At any single location, surface water temperatures were raised for about 2 months, with severe bleaching for 6-8 weeks progressing into mortality up to 4 months later. While many shallow reefs suffered mortality levels of > 50%, some exceeding 95%, there were significant areas with low coral mortality of less than 20%. Recovery of affected reefs to the beginning of 2000 has been slow, and principally through growth of surviving colonies as coral recruitment rates have been low.

The country summaries presented here are extracted from the chapters in this volume.

SOUTH AFRICA

South Africa's reefs lie between 26° S and 27° S and are the southernmost reefs in the western Indian Ocean. They are generally deep (> 8 m) and on offshore, high energy banks, with normal sea temperatures ranging between 22° C and 26° C, and not exceeding 29.5° C. Forty-four genera and 132 species of hard corals have been reported, though soft corals dominate the reefs. Bleaching due to the El Niño was very low, at < 1%, and full recovery was reported. The depth and high wave energy of the reefs is likely to have protected them from greater impacts.

MOZAMBIQUE

Mozambique's coral reefs are most highly developed in the north (from 17° S northwards), with isolated reef areas and coral communities on the southern coast from Bazaruto (21° S) to the South African border. Together with southern Tanzania, the northern Mozambican coast is likely to be the center of diversity of the East African reef fauna. Coral bleaching and mortality due to the El Niño was highly variable among sites surveyed, with most sites in the north registering 30-80% mortality of corals, and in the south, 0-20%. Coral cover following mortality is noticeably higher inside Marine Protected Areas than outside. Algal cover now dominates most reef areas, with no coral recruitment noted indicating slow recovery. Baseline data on invertebrate and fish populations have been recorded in 1999 for monitoring of long term effects. Coral reef monitoring in Mozambique is being conducted in the context of a National Coastal Zone Management Program.

TANZANIA

The entire Tanzanian coastline supports coral reefs, form 5° S - 11° S. Coral bleaching and mortality were recorded along the entire length of the coastline, though at varying levels. Reefs that suffered the highest mortality levels of about 80% included Mafia and Pemba Islands. The majority of reefs were recorded with mortality levels of about 50%, including islands south of Zanzibar town and in southern Tanzania. A number of reefs showed very low mortality, and even increases of coral cover over previous years. Most of these reefs were on the west coast of Zanzibar Island and the mainland around Dar es Salaam, where lower temperatures were recorded due to upwelling of deeper water. Many of these reefs were already highly degraded because of anthropogenic threats prior to the El Niño. Baseline data on invertebrate and fish populations were recorded in 1999 for monitoring of long term effects. Increases in coral competitors that thrive on degraded reefs (Corallimorpharia) have been documented, which may significantly retard reef recovery. Monitoring of fisheries and tourism have shown the potential for losses in income following reef degradation, however these impacts are not yet being felt.

KENYA

The southern coast of Kenya is fringed by a continuous 200 km fringing reef, while reefs on the northern coast are less developed due to the influence of cold water from the Somali current system. Coral cover decreased from pre-bleaching levels of 30-50%, to post-bleaching levels of 5-10% on most reefs, representing losses of 60-90%. Shallow reefs on the southern coast were most affected, with less impact at depth and northwards, though some individual reefs along the coast were found with close to 100% mortality. Recovery of some lagoon patch reefs in the south has been noted, primarily by regrowth of surviving colonies, and partially by recruitment of some opportunistic coral species (e.g. *Pocillopora damicornis*). However, generally, coral recruitment has been low. Monitoring of zooxanthellae and chlorophyll concentrations in five coral species has continued since the El Niño, to provide baseline data for future events. Studies of components of the benthic community are investigating the responses of macroalgae, microalgae and bioeroder communities following the coral mortality. Fisheries catch monitoring is being conducted by a number of groups, however no response to the El Niño has yet been noted. As in Tanzania, tourism may suffer in the long term due to the loss of coral.

Kenya, reef status and ecology

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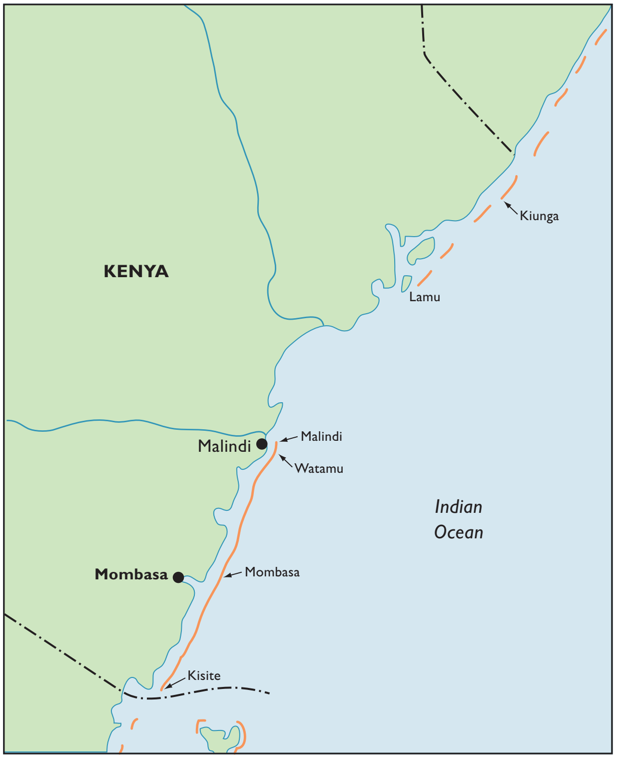
INTRODUCTION

Coral reefs along the entire coast of Kenya suffered widespread bleaching and mortality of corals during the first half of 1998 (Wilkinson, 1998; Obura, 1999; McClanahan *et al*., 1999). This status report summarises findings relating specifically to coral bleaching, mortality and effects on benthic community structure. In addition, preliminary results from a number of research projects investigating different aspects of the bleaching event are reported here. In shallow waters, on a per-area basis, most of Kenya's southern fringing reefs have lost between 66% and 80% of their live corals. Coral reefs in deeper water suffered less mortality due to lesser exposure to higher temperatures. Reefs on the northern part of the coast, influenced by cold water currents from Somalia, also suffered less mortality of corals. Coral recruitment appears to be low on Kenyan reefs except for some minor shallow patch reefs. It is likely that alterations in reef ecology due to overfishing (McClanahan & Muthiga, 1988) and consequent rapid growth of fleshy algae may have delaying effects on coral and reef recovery. Effects of the coral mortality on other components of the reef community, such as on algae, invertebrates and fish are only just beginning to be noticed, 1.5 years after the bleaching event. These components are the subject of ongoing research in an integrated study of the Mombasa Marine Park by scientists at the Kenya Marine and Fisheries Research Institute.



Kenya's coral reefs are divided between two main areas, a fringing reef system in the south (from Malindi to Kisite) and patch reefs and fore reef slopes in the north. After the 1998 bleaching event, the living coral reef cover has decreased on all known reefs in Kenya.

Table 1. Coral species and other groups that were observed both during and after (up to 1.5 years) the El Niño event showing the proportion observed of normal, bleached and dead colonies (0: = not observed; 1: < 25%; 2: 25-75%; 3: > 75%. The taxonomic group is by family for corals, and higher taxonomic groupings for other cnidarians

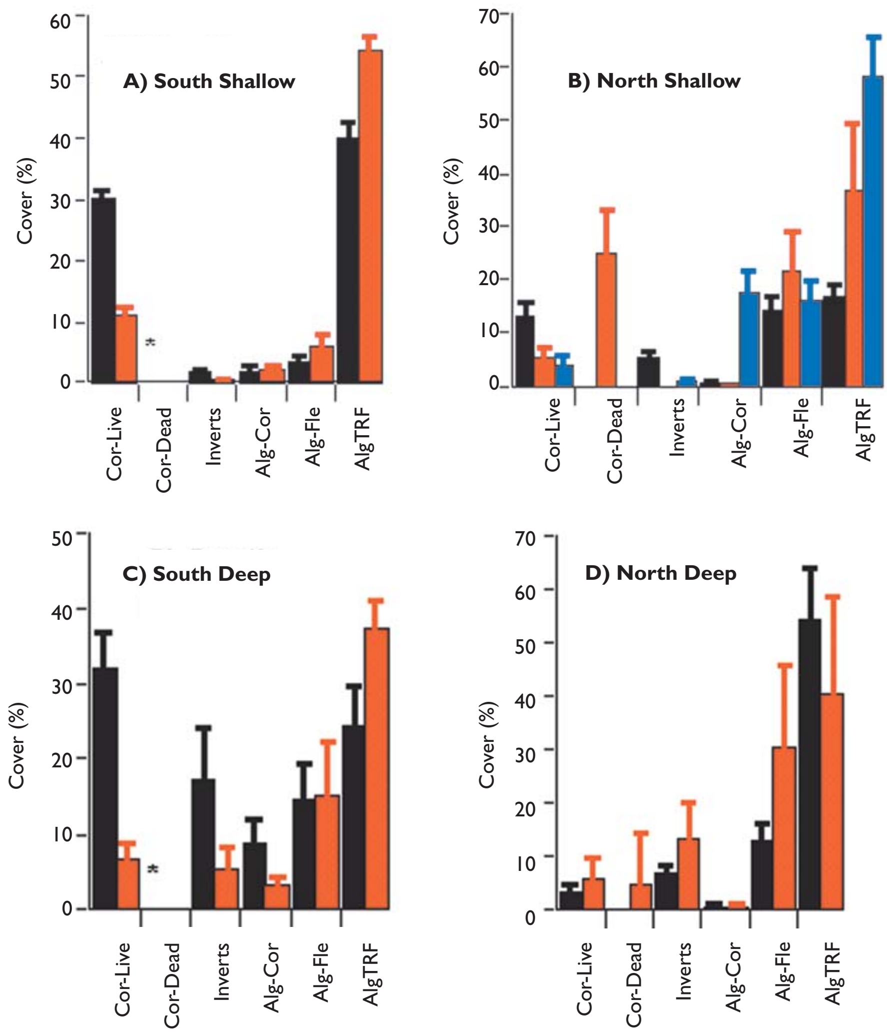
Coral species	Tax.	During			After		
		n	bl	d	n	bl	d
Immediate > 75% mortality							
Acropora f. tabular	acr	0	0	3	0	0	3
Acropora robusta	acr	0	0	3	1	0	3
Acropora f. corymbose	acr	0	1	3	1	0	3
Acropora solitaryensis	acr	0	1	3	0	0	3
Immediate > 75% bleaching							
Galaxea astreata	oci	1	3	1	1	1	3
Acropora spicifera	acr	0	3	1	2	0	2
Montipora tuberculosa	por	1	3	1	2	1	2
Porites compressa	por	0	3	1	3	1	1
Coscinarea monile	sid	1	3	1	3	1	1
Anemone species	zoa	0	3	0	1	2	0
Fungia danae	fun	0	3	0	3	1	0
Variable bleaching, > 75% mortality							
Pocillopora verrucosa	poc	1	2	2	1	0	3
Pocillopora damicornis	poc	2	2	1	1	0	3
Pocillopora eydouxi	poc	2	2	1	1	0	3
Acropora eurystoma	acr	2	2	2	1	0	3
Acropora formosa	acr	2	1	2	0	0	3
Acropora spp.	acr	2	1	2	1	0	3
Acropora humilis	acr	2	2	1	0	0	3
Porites nigrescens	por	1	2	1	1	0	3
Millepora tenella	mil	3	1	1	1	0	3
Variable bleaching and mortality							
Platygyra sinensis	fav	1	2	2	2	0	2
Goniopora lobata	por	1	2	1	2	0	2
Goniastrea retiformis	fav	1	2	1	2	1	2
Lobophyllia hemprichii	mus	2	2	1	2	1	2
Astreopora moretonensis	acr	1	2	1	2	0	2
Montipora informis	acr	1	2	1	2	1	2
Montipora aequituberculata	*acr*	*2*	*2*	*1*	*2*	*1*	*2*
Porites lutea	por	2	2	1	2	1	2
Echinophyllia aspera	ast	2	2	1	2	1	2
Platygyra daedelea	fav	2	2	1	2	1	2
Leptoria phrygia	fav	0	2	0	2	1	2
Variable bleaching, high recovery							
Montipora efflorescens	acr	2	2	0	3	0	1
Pavona varians	aga	2	2	0	3	0	1
Porites spp.	por	2	2	1	2	2	0
Favia pallida	fav	2	2	1	3	0	1
Favia speciosa	fav	2	2	0	2	0	1
Plesiastrea versipora	fav	2	2	1	2	0	1
Low bleaching, variable mortality							
Millepora platyphylla	mil	3	0	1	1	0	3
Acropora cerealis	acr	2	1	2	2	0	2
Astreopora myriophthalma	acr	2	1	1	2	0	2
Leptastrea purpurea	fav	3	1	1	2	0	2
Favia favus	fav	2	1	1	2	1	2
Goniastrea australensis	fav	3	1	1	2	0	2
Echinopora gemmifera	fav	2	1	2	2	1	2
Echinopora lamellosa	fav	0	0	0	2	1	2
Psammocora contigua	sid	3	1	0	2	0	2
Porites rus	por	2	0	2	2	1	2
Platygyra pini	fav	3	0	0	2	0	2
Goniopora stutchburyi	por	3	0	0	2	0	2
Hydnophora exesa	mer	2	1	1	2	0	2
Hydnophora microconos	mer	3	0	0	2	0	2
Low bleaching, low mortality							
Favites abdita	fav	2	1	1	2	0	1
Favites pentagona	fav	2	1	1	2	0	1
Goniastrea favulus	fav	2	1	2	3	0	1
Galaxea fascicularis	ocu	2	1	1	3	1	1
Acanthastrea echinata	mus	2	1	1	3	0	1
Pavona decussata	aga	3	1	1	3	0	1
Sinularia spp.	oct	2	1	1	3	1	1
Goniastrea edwardsi	fav	3	1	1	3	0	1
Porites annae	por	3	1	1	3	1	1
Sarcophyton spp.	oct	2	1	0	3	0	1
Palythoa spp.	zoa	3	1	0	3	0	1
Millepora exesa	mil	2	1	1	3	0	0

BENTHIC COMMUNITIES

Benthic community structure has been recorded by a number of methods with varying comparability. They include a rapid assessment method (IUCN/UNEP in Obura *et al*., 1998) used in northern Kenya in 1998, a variety of line transect methods (McClanahan & Shafir, 1990; Obura, 1995; Obura, in prep), and video transects. Results from all these methods are reported here, with reasonable comparability amongst the line transect and video methods. The rapid assessment method was used for a particular site in 1998 and provided the first documentation of bleaching in Kenya and northern Tanzania (Obura *et al*., 1998).

Kenyan reefs were some of the worst hit of the region, by El Niño bleaching and mortality, with many reefs estimated to have suffered levels of 50% to 90% coral mortality (Wilkinson, 1998; Obura, 1999; McClanahan *et al*., 1999). Transect-based monitoring of reefs following the first estimations has confirmed the general loss of corals countrywide (Figure 1). McClanahan (this volume) reports an average decline in coral cover from 30% to 11% (Figure 1a) on lagoon patch reefs < 1m deep at MLW along the southern Kenya

Figure 1. Benthic cover on Kenyan reefs before, 12 months after and 18 months after the 1998 bleaching event. Data shown for shallow and deep reefs in northern and southern Kenya. A) southern shallow reef records, source: McClanahan/CRCP, this volume and records from 1992 – press. B, C and D – shorter term records of deeper sites and northern reefs, sources: Obura, 1995; Obura *et al*., 1998, CORDIO.



coast. Coral cover decreased on both protected and unprotected reefs, with concomitant increases in algal cover depending on the status of herbivore communities (see McClanahan & Mangi, this volume).

Monitoring of reefs in northern parts of Kenya (in the Kiunga Marine Reserve) and in a few fore reef sites in both northern and southern parts of Kenya has been established more recently. Monitoring in the remote northern sites has involved collaborations among many organisations, including, Kenya Wildlife Service, WWF, UNEP, FAO, CRCP, and recently, CORDIO (reported in Obura *et al*., 1998). Transect surveys of deeper reef sites in the Malindi and Watamu areas are based on sites established in Obura (1995). Coral mortality shows the same patterns at these additional sites (Figure 1b, c, d), with coral cover falling from levels of 30% (southern, deep reefs) and approximately 10% (northern reefs) to about 5% at all sites. Lower pre-bleaching coral cover on northern Kenya reefs is thought to be due to a gradual transition from the coral-reef dominated East African fauna in the south, to the cold-water influence of the Somali upwelling system further north (Samoilys, 1988; McClanahan, 1988; 1990). While coral bleaching and mortality in shallow waters in this northern region were as dramatic as southern areas, deeper corals suffered less mortality (though > 50% bleaching was observed, pers. obs., J. Church, pers. comm.), and the small losses in coral cover were not distinguishable due to the differences in methods used in the before/after surveys.

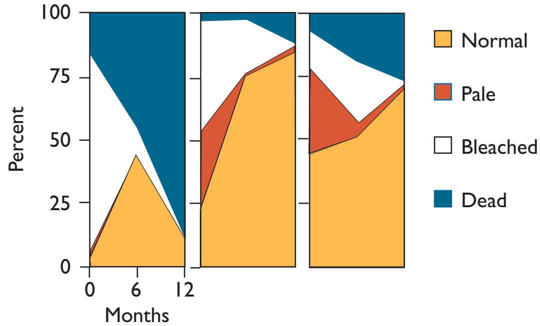
Kenya's network of Marine Protected Areas has offered a large-scale experiment for studies of the effects of protection on reef ecology (e.g. McClanahan *et al*., 1999) and has been the focus of over 15 years of coral reef monitoring and analysis of a variety of threats (Coral Reef Conservation Project, McClanahan & Obura, 1995). Analysis of the El Niño effects in relation to other stresses, and in the context of protection status (McClanahan & Mangi, this volume) will give critical support to calls for placing increased areas of coral reefs under protective management (Introduction, this volume).

CORALS

The condition of corals was observed during and following the El Niño, to record species-specific differences in response to the stress. The frequency of normal, bleached and dead corals for species that were observed during and after the El Niño are given in Table 1. The table partitions species first by the severity and rapidity of their bleaching response, then by mortality levels. The Acroporidae were the most affected group of cnidarians, showing rapid bleaching and mortality at levels approaching or at 100% even before the end of the El Niño event. The second group that showed high levels of bleaching included some species that also suffered close to 100% mortality (*Galaxea astreata*), other acroporids that suffered high but not complete mortality, and a number of other groups (*Fungia* spp., *Coscinaraea* sp., anemones) that had high levels of bleaching, but low mortality levels. Other acroporids, all pocilloporids sampled and most *Millepora* spp. showed variable bleaching during the event and high mortality subsequently. The remaining groups in the table exhibited variable and moderate to low bleaching and mortality levels, and included predominantly faviids, acroporids in the genera *Montipora* and *Astreopora*, agariciids, poritids and most of the octocorals and zoanthids.

Almost all of the coral species that suffered high bleaching and mortality rates are fast growing with branching morphologies, which enable them to dominate reef communities through sexual and asexual reproduction and competitive overgrowth. Many of the surviving species are massive or sub-massive in morphology and slower growing, or only attain small sizes and don't tend to dominate reef communities. These patterns in the bleaching response of coral species are reflected in cluster analysis groupings from a subset of 16 coral species and one soft coral genus (*Sinularia*) from the same data set. The primary division separates the fast growing, low resistance species from the others (A, Figure 2), with the main distinction between groups B and C being in higher mortality but less tendency to bleach, and also higher levels of variation in response, in

Figure 2. Three strategies for bleaching and mortality obtained from cluster analysis of 17 species of corals, showing tendency for normal, pale, bleached or dead condition during, 6 months after, and 1 year after the El Niño. Membership in the groups is: Group A - *Acropora eurystoma, Galaxea astreata, Pocillopora damicornis, Porites nigrescens, Pocillopora verrucosa*. Group B: *Favites pentagona, Favia favus, Galaxea fascicularis, Platygyra daedelea, Porites lutea, Echinopora gemmifera, Hydnophora exesa*. Group C: *Favia pallida, Pavona varians, Sinularia* spp., *Montipora informis, Montipora tuberculosa*. Source. Obura (2000).



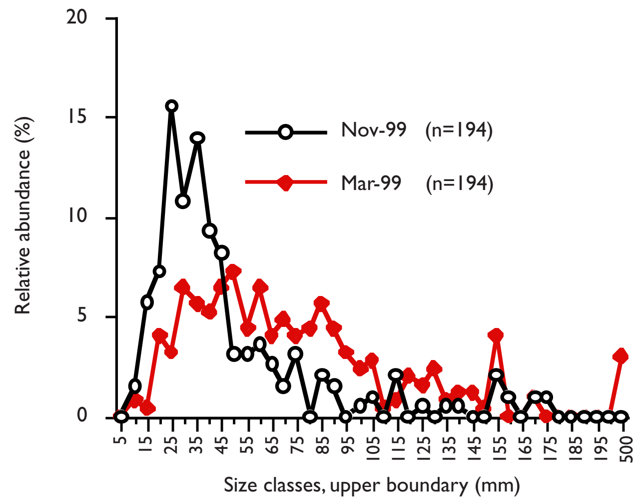


Figure 3. *Pocillopora damicornis* size class structure, Kanamai, November 1999 and March 2000. The number of colonies sampled is shown in brackets.

group C. The divisions among the groups are somewhat consistent with differences in coral species resistance to other stresses, such as sedimentation (Obura, 1995) and in studies of coral life history strategies (Hughes & Jackson, 1985; Kojis & Quinn, 1991). These patterns offer hypotheses for attempts to understand the ecological and evolutionary effects of El Niño-related and other stresses on the long-term prospects for coral reef survival. It also offers the possibility of using a suite of species with known responses as indicators for reef status either through observation of natural corals, as analysed here, or observation of transplants as bioassays (Obura, 2000; this volume).

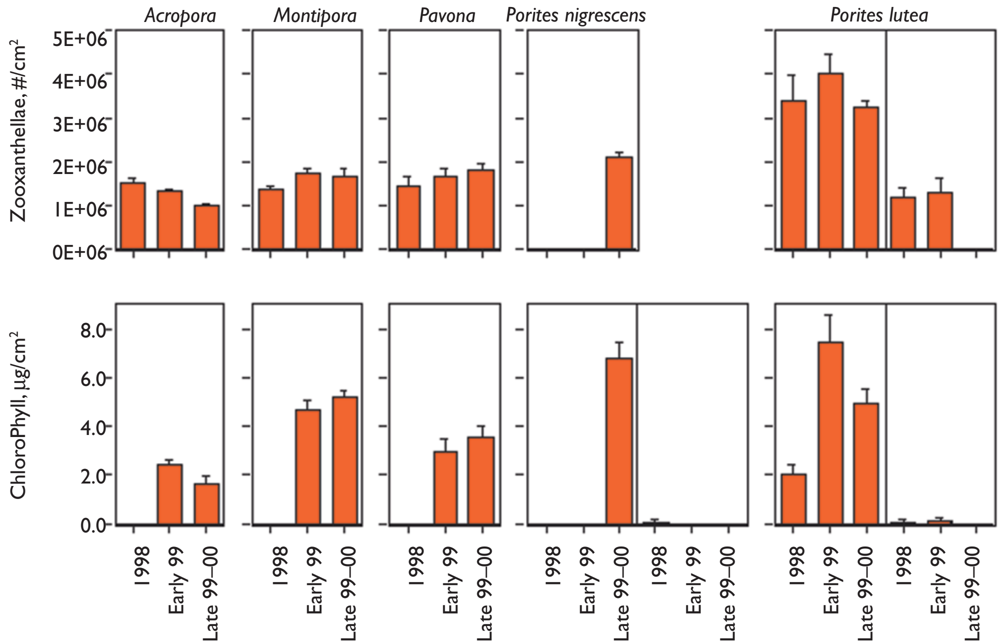
Coral recruitment in 1999 was low at almost all sites surveyed, with few or no recruits less than 2 cm to 3 cm in diameter seen at most reefs (Obura, personal observation). One exception is an extremely shallow back reef coral area at Kanamai, approximately 20 km north of Mombasa, where recruitment of *Poillopora damicornis*, one of the species that 'disappeared' for more than a year following the El Niño, and other opportunistic species such as *Porites nigrescens* and *Pavona* spp. was high.

The population of *P. damicornis* at this site was sampled in November 1999 and March 2000 (Figure 3) and showed a higher peak of smaller classes in November suggestive of recruitment since the El Niño. While this species suffered 100% mortality at all reefs surveyed, there is clearly a source population somewhere that seeded the Kanamai reef. The size and distribution of these remnant populations is likely to have a great impact on the recovery of coral communities throughout Kenya.

ZOOXANTHELLAE AND CHLOROPHYLL IN CORALS

Research on zooxanthellae and chlorophyll concentrations in Kenyan corals started in late 1997 just prior to the El Niño (Mdodo, 1999; Mdodo & Obura, 1999), and sampling has continued almost continuously since then. Five coral species were selected for monitoring (Figure 4), with 'normal' zooxanthellae and chlorophyll concentrations of 1-5 million/cm^2 and 1.5-8.0 µg/cm^2 respec-

Figure 4. Zooxanthellae density and chlorophyll concentrations of normal and bleached colonies of selected coral species during and immediately after the El Niño (1998) and in two time periods in 1999 – 2000.



tively, with consistent species specific differences for zooxanthellae numbers. Sampling of bleached colonies of two of the species showed significant decreases in both the number of zooxanthellae and the concentration of chlorophyll in both species. Ongoing monitoring of these variables is being conducted to establish a good baseline of data and to maintain capacity to sample in a subsequent bleaching event.

MACROALGAE

Degraded reefs are often colonised by different types of algae which may undergo a successive sequence ending in a climax community that might differ from the original community. Thus, the overall aim of this sub-project was to monitor changes in the algal composition of selected reefs affected by the El Niño bleaching event and to study the algal succession patterns in these areas. Three 10 m line transects were used to obtain the percent cover of each substrate type in the study areas. Settlement tiles to study algal recruitment and primary succession were prepared using bathroom ceramic tiles. The tiles were covered with mixture of sand and waterproof cement in order to provide a rough substrate for the settlement of algae. The tiles were set out in the two sites. A total of 45 tiles were set in each site in November 1999, with three tiles being collected at monthly intervals. In the laboratory, macroalgae attached to the tiles were identified and their wet weight obtained, while for small filamentous turf algae, 2 cm x 2 cm sections were scraped off, weighed and the wet weight extrapolated for the tile area.

There were more taxa and higher cover of macroalgae at Ras Iwatine compared to Starfish, though the cover of most species remained below 10% (Figure 5). *Padina, Sargassum,* and *Dictyota* were the most prominent algal genera, in common with other Kenyan reefs (McClanahan, 1997). Also, macroalgal populations were higher on the unprotected reef, Ras Iwatine, where herbivory by sea urchins would be higher (see Bioeroders section), also indicated by the high cover of herbivore-

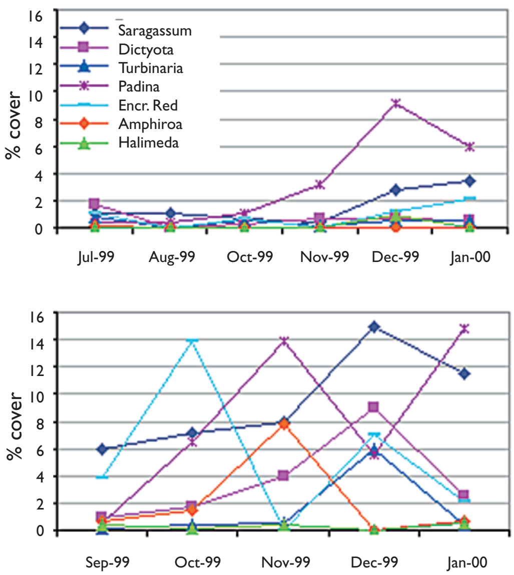


Figure 5. Macroalgae community at Starfish (above) and Ras Iwatine (below) in the Mombasa Marine Park Reserve.

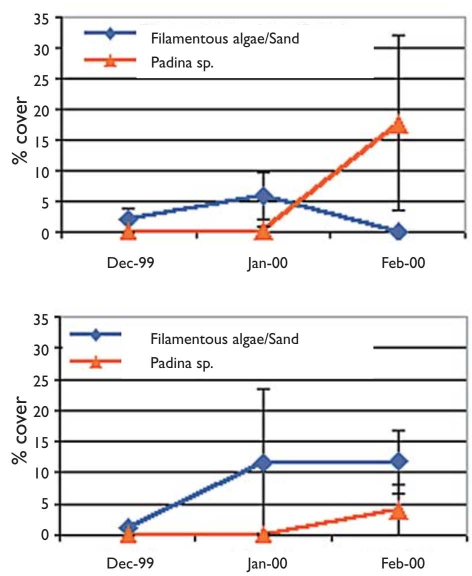


Figure 6. Algal succession on tiles at Starfish (above) and Ras Iwatine (below) in the Mombasa Marine Park Reserve (in g/225cm^2). Low settlement of brown algae *Hypnea cornuta, Dictyota adnata* and sponges was also recorded.

resistant encrusting coralline algae (Figure 5, McClanahan, 1997). At the Starfish site, all macroalgae remained low till November 1999 where *Padina* began to increase at the onset of the north-east monsoon (Figure 5).

Although results obtained from settlement tiles are still preliminary, information describing the initial stages of succession was obtained. The first colonisers were filamentous, composed of blue green algae (Cyanophyta), among which sand became trapped. At both sites *Padina* was the second colonising taxon, reducing the cover of filamentous blue-green algae. The colonisation of *Padina* occurred one month earlier at the Starfish site (January) than at Ras Iwatine (February) (Figure 6). Other algae present on the tiles included *Hypnea cornuta*, *Dictyota adnata* and *Chondria* sp.

MICROALGAE

Research has shown that macroalgal turf that colonises the surface of dead corals provides a rich substratum for

Table 2: Relative abundance of microalgal species encountered in the study sites. Relative abundance scale: XXXX very abundant, XXX abundant, XX significant presence, X rare.

Species	Relative Abundance	Potential Toxicity	Abundace peak
Navicula sp.	XXXX		
Oscillatoria sp.	XXX	yes	
Chaetoceros sp.	XX		February
Coscinodiscus sp.	XX		June
Licmophora sp.	XX		
Pleurosigma sp.	XX		
Prorocentrum cf. *mexicanum*	XX	yes	
Prorocentrum lima	XX	yes	February
Amphora sp.	X		
Campylodiscus balatonis	X		
Dinophysis sp.	X	yes	
Gambierdiscus sp.	X	yes	
Ostreopsis sp.	X	yes	
Pseudonitzchia cf. *pungens*	X	yes	
Pseudonitzchia sp.	X	yes	
Striatella interrupta	X		
Synedra sp.	X		

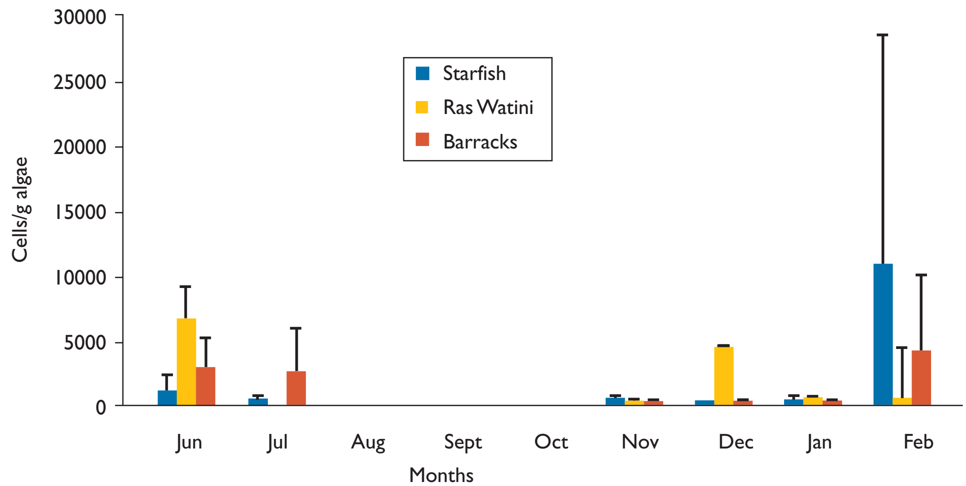


Figure 7. Microalgal population densities at three sampling stations in Mombasa Marine Park from June 1999 to February 2000 (mean, standard deviation).

diverse dinoflagellate assemblages, some of which produce toxins dangerous to human health. Surveys were carried out in the Mombasa Marine Park starting in June 1999 to study the distribution of benthic microalgae attached to the algal turfs on dead corals, document variations in qualitative and quantitative composition and provide information on the biodiversity of benthic microalgal taxa. Dead corals pieces covered by algal turfs were collected at 2 week intervals, scrubbed in filtered seawater and the suspension filtered through screens of 250 µm, 125 µm, 63 µm and 38 µm, and the microalgae counted using an inverted microscope.

Fifteen genera of benthic microalgae were identified (Table 2), eight of which are known to be potential producers of toxins harmful to human health. Mean abundance of macroalgal cells varied greatly between 283 ± 174 (± standard deviation) cells/g fresh algae in November to 10 545 ± 17 742 cells/g fresh algae in February. (Figure 7). The mean cell abundance from June 1999 to February 2000 was 3 174 ± 9 80, 2 039 ± 5 039 and 1 529 ± 3 009 cells/g fresh algae at the three sampling stations. Non-parametric ANOVA indicated that the abundance of microalgal cells at each station was similar. The study will be expanded to include the setting up of algal tiles in order to monitor succession and rate of colonisation by microalgae.

BIOERODERS

External and internal bioeroder communities were censused. Sea urchins within quadrats of 1 m in diameter were identified and counted Three species of urchins (*Echinometra mathaei, Diadema savignyi and D. setosum*) were then selected for analysis of gut sediment content by hydrochloric acid hydrolysis to remove the carbonate sediment fraction. For the examination of bioeroders, dead coral fragments of *Porites* spp. (massive and branching) and *Acropora* spp. were collected. Internal bioeroders were then identified and counted. An experiment is currently underway, in which cut coral blocks (*Porites* massive) and pieces of corals of known weight have been deployed at the study sites and will be collected at different intervals to determine the succession of internal bioeroders and their rates of bioerosion.

The study sites show typical sea urchins communities for Kenyan reef lagoons dominated by diadematids (especially *D. savignyi*), *E. mathaei* (Echinometridae) and *Tripnuestes gratilla* (Toxopneustidae) (Table 3). Sea urchin population densities in the marine park show con-

Table 3: Population densities and standard deviation (SD) per 10m² of six common sea urchin species at Starfish Point and Ras Iwatine

Sites/No. of quadrats	Starfish 3 / 30 Density	SD	Ras Iwatine 3 / 15 Density	SD
Diadema savignyi	5	14	5.5	1.7
Diadema setosum	0.67	0.16	1.67	1.37
Echinothrix diadema	8.66	8.1	1.5	0.7
Echinometra mathaei	2.3	1.2	2.5	2.0
Tripnuestes gratilla	1.76	0.6	4.67	1.8
Echinostrephus molaris	0.53	0.1	0.43	0.3

siderable reduction compared to past years (McClanahan & Shafir, 1990), possibly due to increases in predation as fish populations within the marine park mature following 10 years of protection. These preliminary results show that *D. savignyi* makes the greatest contribution to overall bierosion, with the highest gut sediment content (37%) followed by *D. setosum* (28.1%) and *E. mathaei* (25.6%) (Table 4). However, the method used in this study did not differentiate between sources of carbonate sediment, which included both corals and coralline algae. The internal bioeroder community was dominated by sponges (32.3%), polychaetes (14.5%), bivalves (22.8%) and sipunclid worms (16.8%) (Figure 8). Risk *et al*. (1995) contend that morphology of corals may be an important determining factor in total bioerosion levels. More holes were recorded in *Porites* skeletons, however the bioeroder community was similar for both *Acropora* and *Porites*.

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We would like to thank S. Ndirangu, C. Muthama, Kimantha, M. Zamu, J. Kilonzo, J. Tamelander, and L. Tamara for assistance in the field. The Mombasa Marine Park rangers and warden have also greatly assisted with transport and access to the study sites.

Table 4. Composition of gut contents (coral sediment; plant matter; invertebrates g/1000g) and contribution to bioerosion (given as gut content of coral sediment*density) from five sea urchin species (n=20).

	Coral Sed. m	SD	Plant m	SD	Invert. m	SD	Bioerosion contrib. Starfish	Ras Iwatine
Diadema savignyi	366	62	550	70	85	17	1825	2012
Diadema setosum	281	58	643	82	76	13	188	1074
Echinothrix diadema	51	12	ND[1]	-	ND	-	439	659
Echinometra mathaei	256	58	692	90	51	10	589	1473
Tripnuestes gratilla	14	8.3	ND	-	ND	-	25	118

[1] ND = Not determined

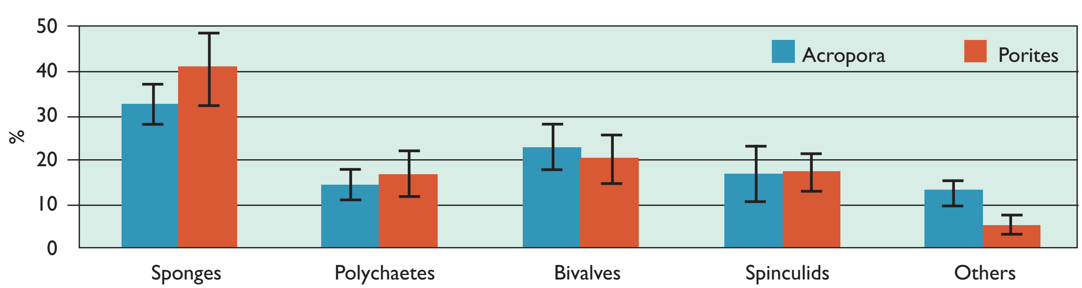

Figure 8. Proportion (%) of internal bioeroders in *Acropora* and *Porites* branches at Ras Iwatine.

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Assessment of coral reef degradation in Tanzania: Results of coral reef monitoring 1999.

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INTRODUCTION

Coral reefs play a crucial role in the well being of coastal communities in Tanzania (Johnstone *et al*., 1998; Muhando, 1999). However, despite their usefulness, coral reefs are being degraded by destructive anthropogenic activities (Salm *et al.*, 1998) and natural causes (e.g., competition, predation, diseases, bleaching, etc.). The coral bleaching and mortality event of March - June 1998 was the most serious natural calamity ever recorded in the Indian Ocean (Wilkinson, *et al.*, 1999). Several areas along the coast of Tanzania were affected. The degree of coral mortality varied between sites, from 60% - 90% at Tutia Reef in Mafia Island Marine Park and Misali Reef on the west coast of Pemba, to approximately 10% on reefs around Unguja Island, Zanzibar (Muhando, 1999). After the bleaching and coral mortality, the status of Tanzanian reefs became unclear and it was apparent that there was a need to assess and monitor the extent of coral mortality and its effects on reef ecosystems, as well as socio-economic effects (fisheries and tourism). Three teams were formed. The first dealt with the assessment and monitoring of coral reefs, the second with socio-economic effects and the third team investigated specific issues relevant to coral bleaching, mortality and recovery.

The objectives of the coral reef assessment and monitoring team were to assess and monitor a) benthic cover, b) the distribution and density of ecologically important macrobenthos, and c) reef fish distribution and composition. This presentation is a summary of reef assessment and monitoring results for 1999.

METHODS

Monitoring was carried out at reefs at Unguja (Zanzibar) Island (Chapwani, Changuu, Bawe, Chumbe and Kwale), Misali Island (West of Pemba Island), off the Kunduchi coast in Dar es Salaam (Mbudya and Bongoyo), at Mafia Island Marine Park (Tutia, Msumbiji and Utumbi) and at Mnazi Bay (Chamba cha Matenga, Chamba cha Kati and Kitelele)(Figure 1). The sites had variable exposure to waves (protected or semi-exposed) and tidal current strength (strong, medium, weak) (Table 1).

At each site, a mixture of random and permanent transects were used to sample benthic cover. Permanent transects were recorded using the Line-Intercept Transect (LIT) method (English *et al.*, 1994), with varying lengths of 10 m or 20 m recording a total length of 20 m – 40 m per site. Random transects were recorded using a Line-Point Transect (LPT) method, dividing a 50 m transect into two hundred points at intervals of 25 cm (Allison, 1996). With the exception of sites at Mafia that were sampled using nine and 10 LPT's, most sites

Figure 1. Map of the distribution of coral reefs in Tanzania illustrating each site at which monitoring was conducted.

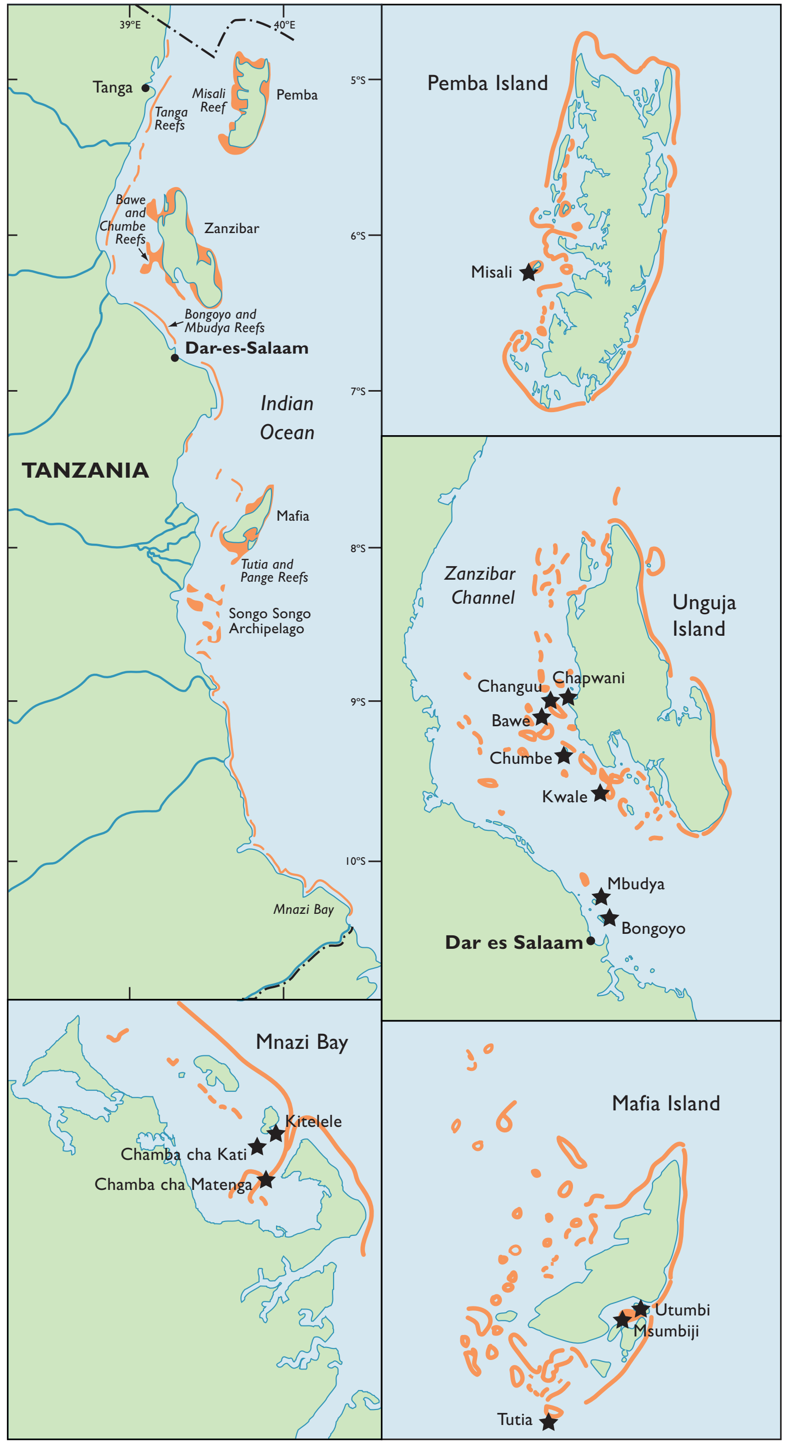


Table 1. Reef stations surveyed in the IMS/CORDIO monitoring programme. The table indicates exposure to waves (protected or exposed) and tidal current strength (strong (S), medium (M), weak (W)), and sampling effort for each station as the total length of permanent and random transects recorded.

Area	Station	# of sites	Exposure	Currents	Total transect samples (m) Permanent	Random
Pemba	Misali	4	Semi-exp.	M/W	140	200
Unguja -	Chapwani	3	Protected	W	90	300
Zanzibar	Changuu	3	Protected	W	90	300
town	Bawe	3	Protected	W	90	300
Unguja – south	Chumbe	3	Protected	W	0	300
	Kwale	3	Protected	M	90	300
Dar es Salaam	Bongoyo	2	Protected	M	80	250
	Mbudya	2	Protected	M	80	250
Mafia Island	Tutia	1	Semi-exp.	S	40	500
	Msumbiji	2	Protected	S	40	150
	Utumbi	2	Protected	S	40	450
Mnazi Bay	Matenga	1	Protected	S	40	100
	Kati	1	Protected	S	40	100
	Kitelele	1	Semi-exp.	S	40	100

were surveyed using two or three transects. A plumbline was used to reduce observer parallax errors (Allison, 1996). At all sites, transects were set on the reef flat (1 m - 5 m) and reef slope (5 m - 15 m) parallel to the reef front. Standard reef benthic cover classes were used following English *et al.* (1994), and recording corals in life form categories.

Belt transects were used to sample invertebrates and fish (English *et al.*, 1994). Invertebrates that were relatively easy to see and could indicate the health of the coral reef were counted in belt transects of 50 m x 2 m (100 m^2) placed randomly on the reef. A T-stick was used to maintain the belt width. The macrobenthos counted included crown-of-thorns starfish (COTS, *Acanthaster planci*), sea cucumbers (Holothuroidea), sea urchins (Echinoidea), sea stars (Asteroidea), giant clams (*Tridacna*), gastropods (commercial species), and lobsters (Palinuridae). Fish were counted in 50 m x 5 m belt (250 m^2) transects. On each occasion, fish were counted on reef flat and on the adjacent reef slope. The counting started between five and 10 minutes after placing the transect line to allow fish to resume their normal behaviour. Attention was given to 16 fish families that were considered to be commercially or ecologically important (Acanthuridae, Balistidae, Caesionidae, Chaetodontidae, Haemulidae, Kyphosidae, Labridae, Lethrinidae, Lutjanidae, Mulidae, Nemipteridae, Pomacanthidae, Scaridae, Serranidae, Siganidae, and Tetraodontidae), organized into seven trophic groups (corallivores, piscivores, herbivores, invertivores, omnivores, planktivores and spongivores) for analysis.

All sites were referenced using GPS, and key physical parameters such as turbidity, sedimentation, exposure to currents and waves and type of substrate of adjacent habitats were recorded. Visibility of the water was measured using a secchi disc set horizontally, and sedimentation was evaluated by visually estimating the amount of suspended particles in the water column, the type and size of bottom sediment, and sediment stress signals on live corals.

RESULTS

Benthic cover

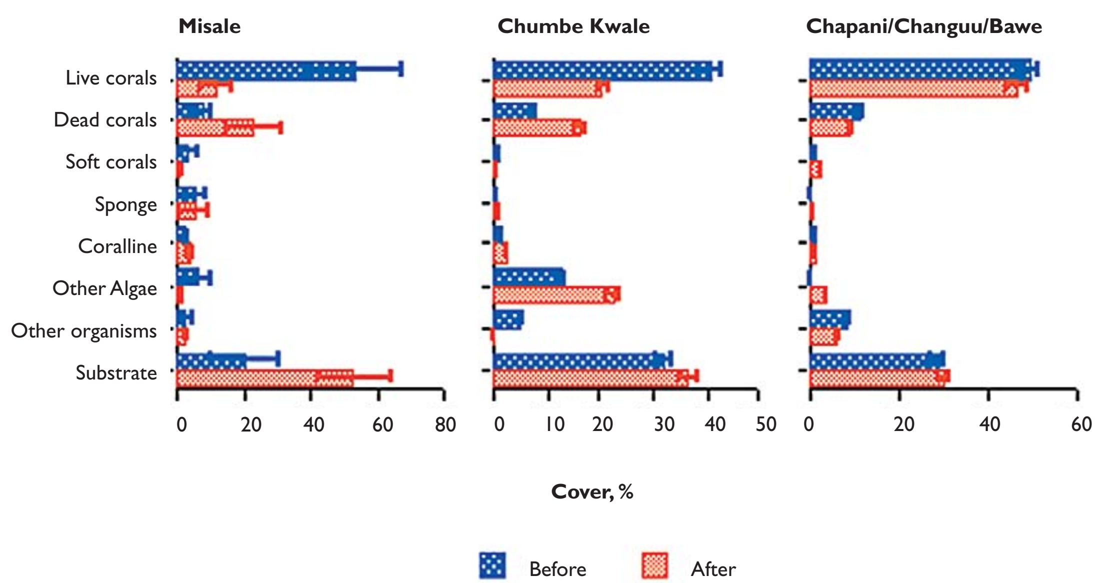
The benthic cover reported here is calculated from the random transects only. The observations from permanent transects will be considered together with results of next year (2000) to give a stronger indication of changes over time. This report only presents only summaries of the results, detailed reports for each monitored site are being prepared (as IMS - Internal Reports) for use by local area coastal zone managers and researchers.

In 1994 Misali Reef was dominated by live hard coral (51% & 74%, see Horril *et al.,* 1994) However, this decreased to 17% and 7% respectively in March 1999, after the bleaching and mortality event in 1998 (Figure 2). Conversely, the amount of dead coral increased from 11% and 15% to 10% and 40% respectively. The category "substrate" (rubble, rock or sand) increased significantly from an average of 20% in 1994 to 53% in 1999. The relative cover of other benthic categories such as fleshy algae, coralline algae, sponges and soft corals did not change appreciably.

Unguja reefs did not appear to have any serious negative effects from the coral bleaching, except on Chumbe Reef located south of Zanzibar town where coral cover decreased from 51.9% (1997) to 27.5% (1999) and Kwale where coral cover declined to 29.7% in 1997 and 13.3% in 1999 (Figure 2). A slight increase of live coral cover was observed at Bawe, off Zanzibar town, from 53% in 1997 to 57.7% in 1999. The cover of fleshy algae was lower on Chapwani, Bawe, and Changuu than at Chumbe and Kwale, where two genera (*Sargassum* & *Turbinaria*) occupied 8.8% and 16.6% of the substrate in 1997 and 18.4% and 26.2% in 1999 respectively.

The cover of live coral on reefs near Dar es Salaam appears to have increased in 1999. Coral cover at Bongoyo Marine Reserve was estimated at 55% in 1999, compared with 49% in 1997 (Kamkuru, 1998) and 70% in 1974 (Hamilton, 1975). Similarly, available data on

Figure 2. Summary of percentage coral reef benthic cover before and after bleaching event at Misali Island and reef stations on Zanzibar Island.

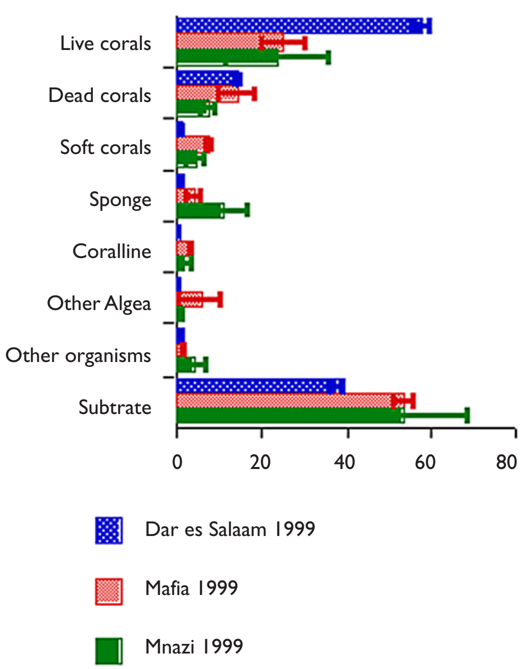


live coral cover for Mbudya Marine Reserve suggests an increase from 37% in 1997 (Kamkuru, 1998) to 59% in 1999. Dead coral comprised approximately 14% of the total cover of substrate on these reefs (Figure 3). Algal cover was relatively low in both Mbudya and Bongoyo reserves. Similarly, other organisms contributed less compared with substrate category and live coral cover.

Live coral cover at Tutia Reef decreased from 80% in 1991 (Ngoile & Horrill, 1991) to 15.1% in October 1999, (Figure 2). On Msumbiji and Utumbi Reefs (located in Chole Bay) the percentage of live coral cover was higher (30%) than at Tutia Reef. Fleshy algae had increased substantially on Tutia Reef from 1% in 1996 (pers. obs.) to (15%) in 1999. The most abundant species of algae was *Stypopodium zonale*, a brown alga found only at Tutia and Kwale Reefs.

At Mnazi Bay, live coral cover on Matenga (Chamba cha Matenga) and Kati (Chamba cha Kati) decreased from 55% and 60% in 1997 (Guard *et al.*, 1998) to 28%

Figure 3. Summary of percentage coral reef benthic cover at sites without comparable before-bleaching data.



and 42% in 1999, respectively. The channel reef at Kitelele had very low coral cover at 1%, however the high cover of rock substrate has high potential for coral settlement. The amount of dead coral was estimated to range between 9.3% at Matenga Reef and 9.5% at Kati Reef. The highest cover of sponges (22%) was observed at Matenga Reef. The cover of coralline and fleshy algae was relatively low at all sites monitored in Mnazi Bay.

Macro-invertebrates and fish

A summary of the macro-invertebrate counts for each site is shown in Table 2. High densities of sea urchins were recorded on reefs at Bawe ($2.6 \cdot m^{-2}$) and Chapwani ($2.2 \cdot m^{-2}$) in Zanzibar, while intermediate densities were recorded at Changuu in Zanzibar ($0.4 \cdot m^{-2}$), Mbudya and Bongoyo Marine Reserves in Dar es Salaam ($0.2 \cdot m^{-2}$ and $0.5 \cdot m^{-2}$ respectively) and Misali Island ($0.6 \cdot m^{-2}$). Sea urchin densities at other sites, including those with greater abundance of fleshy algae were low. Gastropods of commercial value and sea cucumber densities were generally low. Sea star densities were highest in Mnazi Bay with an average density of $0.5 \cdot m^{-2}$. COTS were found at all sites of Zanzibar but, because of their low densities, were not recorded in any of the transects sampled. COTS were recorded in transects at Bawe, at a density of $0.06 \cdot m^{-2}$. At Mafia, COTS were observed at all sites surveyed (pers. obs) though they were recorded in transects at Msumbiji Reef only. In Mnazi Bay COTS were recorded at all sites.

A total of 119 species of reef fish were recorded at all monitoring sites (Table 3) with the overall average density of 88 individuals per 250 m^2. Reefs at Misali and Mafia had comparatively higher densities and species richness of fish while Zanzibar reefs had the lowest density of fish. Reefs of Mtwara and those of Dar es Salaam have had lowest fish species richness (Table 3). Fourteen families of commercially important reef fish and three other reef indicator families were represented in the survey.

The most common trophic group was herbivores comprising more than 50% of the fish population at

Table 2. The number of invertebrates per 50 m x 2 m belt transect on the monitoring sites (mean and standard error). The number of stations sampled indicated in brackets (and see Table 1).

	Misali (1)		Zanzibar town (3)		Zanzibar south (2)		Dar es Salaam (2)		Mafia (3)		Mnazi Bay (3)	
	m	se	m	se	m	se	m	se	m	se	m	se
G. Clams	6.0	.	2.7	0.9	0.5	0.5	0.0	0.0	1.0	0.6	1.0	0.6
S.urchin	55.0	.	173.0	67.4	2.5	0.5	37.5	15.5	1.3	0.9	6.7	3.8
Gastropod.	0.0	.	0.0	0.0	0.0	0.0	1.0	0.0	0.3	0.3	1.3	0.9
S.cucumber	2.0	.	1.7	0.9	4.0	1.0	0.0	0.0	0.0	0.0	0.7	0.3
Star fish	0.0	.	0.3	0.3	2.0	1.0	0.0	0.0	2.0	1.0	4.7	2.3
COTs	0.0	.	0.7	0.7	0.0	0.0	0.0	0.0	0.3	0.3	1.0	0.0
Lobster	0.0	.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.7

Table 3. Average density and number of fish species counted on the monitoring sites.

	Misali	Zanzibar	Dar es Salaam	Mafia	Mnazi Bay	All sites
Average density/ 250 m^2	186	48	62	119	61	88
Total # of species	102	63	30	78	26	119

most sites except Chapwani, Changuu and Bawe in Zanzibar (Table 4). Invertivores and corallivores were fairly well represented at all sites. However, at most sites invertivores exceeded corallivores, except at Matenga, Mbudya, Bongoyo, Bawe and Chumbe. Planktivores, represented by the family Caesionidae, generally occurred at low densities except at Bongoyo and Misali Reefs. Spongivores were present at low densities at most sites.

Environmental conditions

Measurements of environmental parameters at the monitoring sites are shown in table 5. Low visibility was observed on reefs of Dar es Salaam (Bongoyo and Mbudya) and reefs close to Zanzibar town (Chapwani and Changuu). Reefs located away from the city and in shallow coastal areas such as Tutia and Misali Reefs had very clear water. Bongoyo Reef has the highest sedimentation and lowest secchi disc measurement.

Table 4. The number of fish per 50 m x 5 m belt transect on the monitoring sites (mean and standard error). The number of stations sampled indicated in brackets (and see Table 1).

	Misali (1)		Zanzibar town (3)		Zanzibar south (2)		Dar es Salaam (2)		Mafia (3)		Mnazi Bay (3)	
	m	se	m	se	m	se	m	se	m	se	m	se
Corallivores	1.8	.	14.0	6.4	3.4	2.8	5.2	1.6	5.2	0.8	5.9	2.1
Herbivores	60.8	.	27.2	1.9	85.4	1.2	59.6	4.8	71.3	2.4	63.5	3.1
Invertivores	6.0	.	13.8	0.8	5.6	1.6	3.4	1.1	9.7	0.4	8.7	4.2
Omnivores	2.7	.	3.7	1.8	1.4	0.5	1.0	0.2	4.9	2.1	3.6	2.7
Piscivores	0.2	.	5.4	2.5	1.2	0.0	0.5	0.1	1.4	0.8	1.7	1.0
Planktivores	15.2	.	0.2	0.2	0.0	0.0	20.5	14.9	0.8	0.5	0.0	0.0
Spongivores	0.5	.	0.9	0.9	1.0	0.4	0.6	0.6	0.5	0.2	0.2	0.2
Unidentified	12.7	.	34.7	11.2	2.0	0.2	9.3	6.5	6.2	1.1	16.4	5.3

Table 5. Some environmental parameters measured in the study sites

Site	Secchi disk (m)	Sedimentation
Misali	19	Low
Chapwani/Changuu	3.3	Medium
Bawe/Chumbe/Kwale	10.5	Low
Bongoyo	3	High
Mbudya	4.8	Low
Tutia	20.7	Low
Chole Bay	12	Medium
Mtwara	8.0	Medium

DISCUSSION

The 1999 coral reef assessment and monitoring shows that all reefs surveyed were affected by the major bleaching event. Several reefs suffered severe or catastrophic mortality: about 80% of the corals in Tutia, Mafia Island and reefs of Misali Island in Pemba suffered mortality. At Chumbe and Kwale in Zanzibar and Mnazi Bay in Mtwara coral mortality was intermediate, with 30% to 55% of hard corals left dead. On the reefs of Bawe, Changuu and Chapwani in Zanzibar and Bongoyo and Mbudya in Dar es Salaam 11% to 20% coral mortality was recorded.

High mortality occurred on reefs receiving direct oceanic waters (Tutia in Mafia and Misali in Pemba). Reefs in bays (Chole Bay in Mafia Island and Mnazi Bay in Mtwara) or shallow areas (Zanzibar and Dar es Salaam) had lower levels of bleaching and mortality. Shallower waters are influenced by night cooling, which reduces the stress level. It is also possible that the intrusion of cold water masses in Zanzibar (Muhando, 1999) and Dar es Salaam, and high cloud cover and rainfall both contributed to reductions in solar radiation and warming of the surface water. All of these factors would have reduced maximum seawater temperatures and, thereby, lowered the levels of bleaching stress.

There was an increase of coral cover from 37% in 1997 to 59% in 1999 at Mbudya Island. Differences in sampling techniques and study plots between surveys conducted in the area (Hamilton 1975; Kamkuru, 1998; IMS/CORDIO survey, 1999) could be responsible for this discrepancy illustrating the need for standardised techniques and study plots.

It is postulated that death of hard corals may give a chance for fleshy algae to proliferate (Bell, 1992; Done, 1992). The pronounced increase of macroalgae in Tutia Reef, Chumbe and Kwale could hinder coral settlement and retard the processes of reef recovery. Further stress of the reefs could lead to algal dominance. Nevertheless, at many reefs increased algal abundance was not very pronounced. On the contrary, at Misali there was a reduction in cover of macroalgae after the bleaching event, giving high chances for reef recovery.

Macro-invertebrate densities were generally low, except for sea urchins on some reefs close to urban areas. Chapwani and Bawe in Zanzibar had the highest numbers and Misali, Changuu, Mbudya and Bongoyo had moderate numbers. The densities of gastropods of commercial value and sea cucumbers were low in all sites. This is likely to be a result of over-exploitation rather than an effect of the coral bleaching event.

In general, the densities of COTS were low, but have been observed to cause substantial damage on a local scale. For example, the live coral cover in areas where COTS have aggregated on Changuu (permanent transects 12 and 13) was reduced from from 58% in 1996 to 25% in 1997 when the density of COTS was between $0.8 \cdot m^{-2}$ and $1 \cdot m^{-2}$. In 1999 no COTS were recorded at Changuu. However, high densities were recorded at Bawe, with aggregations observed in areas dominated by *Acropora*.

Misali and Mafia Islands had the highest density and species diversity of fish. These islands had the highest hard coral cover before the bleaching event, but the high coral mortality in 1998 has not yet had a noticeable effect on the populations of reef fish. Herbivores are the dominant trophic group on the reefs, and also dominated the artisanal fisheries catch at landing sites, at least in Zanzibar (Jiddawi, pers. comm.) both before and after the bleaching event. Therefore, it is too early to draw conclusion on the effect of bleaching on the reef fish population.

Sedimentation estimates showed that reefs close to Dar es Salaam and Zanzibar town had higher sedimentation rates compared with those away from highly populated areas. Bongoyo Reef, for example, is close to the waterway where ferryboats frequently pass into Dar es Salaam harbour. The powerful engines might be responsible for stirring up fine sediment from the bottom and creating high sedimentation rates. Continued monitoring of the reefs will provide information that can be used in the management of the coastal environment and its resources.

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Coral reef monitoring and management in Mozambique

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INTRODUCTION

MICOA, in conjunction with a number of other institutions and donors, have recently initiated a project for the development of a National Coastal Zone Management Programme (NCZMP). This program encompasses the entire coastal zone and is multi-disciplinary in its approach. Further, it is envisaged that one of the components of this NCZMP will focus on the critical ecosystems that comprise the coastal environment such as coral reefs. Consequently, a management plan for coral reefs is being formulated concurrently with the National Coastal Zone Management Plan (NCZMP). In this programme, four activities stand out as being vital for the achievement of the main goal of sustainable management of coral reef resources:

- Capacity building within the relevant fields required for effective sustainable management;
- Collection and synthesis of relevant information and scientific data to support sound management;
- Development of an appropriate and effective network for the co-ordination and sustenance of coral reef management related activities; and
- Identification, characterisation and mitigation of current and eventual problems faced by coral reefs and their management.

It is within this framework that recent activities on bleaching on coral reefs in Mozambique as well as capacity building were undertaken. This paper presents the activities and respective summary results within the framework of the coral reef management programme during 1999. Technical support for those activities was provided by ORI (Oceanographic Research Institute, Durban, South Africa) and CORDIO and funding was provided by CORDIO and DANIDA.

Mozambique possesses the third longest coastline along the Western Indian Ocean, extending 2700 km, much of which adjoins areas of coral reefs. The northernmost section of the coast extends for 770 km from the Rovuma River in the north (10º 20' S) to Pebane in the south (17º 20' S). In this section coral reefs constitute an almost continuous fringing reef on the eastern shores of the islands and the more exposed sections of the mainland coast. The central section of the coast between Pebane (17º 20' S) and Bazaruto Island (21º 10' S), a distance of about 950 km, is classified as a swamp coast. Twenty-four rivers discharge into the Indian Ocean along this section, each with an estuary supporting well-established mangrove stands. The coastal waters are shallow and this, combined with the sediment loading from the rivers, causes high turbidity levels. As a consequence, coral reef formation in this area is severely limited. The southern section stretches for 850 km from Bazaruto Island southwards to Ponta do Ouro (26º 50' S). The coastline is characterized by high dunes, north facing bights and barrier lakes. The distribution of reefs along the coast and near-shore islands is patchy and the reefs are more sparsely inhabited by corals.

This ecosystem constitutes an important biological resource in terms of their complex biodiversity and is also the basis for tropical fisheries and marine eco-tourism industries. Today, about 6.6 million people live within Mozambique's 48 coastal administrative districts, and the number is expected to grow at 3% p.a. (INE, 1999). Although this represents 42% of the current population of Mozambique (15.7 million), only 2% - 3% are fishermen or collectors. Nevertheless, Mozambique's economy is largely dependent on fisheries as shrimp exports contribute significantly to the GDP (~USD 100 million p.a.). Tourism, on the other hand, is a growing industry and most of its development occurs in coastal areas and in activities such as diving and snorkelling.

CORAL REEF RELATED ACTIVITIES IN MOZAMBIQUE DURING 1999

Preliminary assessment of coral bleaching

The survey of coral bleaching was undertaken between March 24 and April 8, 1999, at the end of summer. At each of 17 reefs, evidence of past and present bleaching was sought and visual estimates made of reef type, cover of benthic organisms and the extent of damage caused to the reef by bleaching and Crown-of-thorns starfish (COTS)

The effects of El Niño bleaching in Mozambique were most extensive on exposed reefs in the north and diminished further south except at Inhaca Island where serious recent bleaching was encountered. Extensive COTS damage was also found at Bazaruto and Inhambane. The consequences of the El Niño bleaching are going to be even more serious as coral mortality on the northern reefs was as high as 99% and eventual collapse of reef structure on these reefs is anticipated. The biodiversity of these sites will be impaired as coral recruitment was essentially absent and only observed at the Bazaruto COTS-affected site.

Fish populations on the damaged reefs, the basis of many of Mozambique's valuable artisanal fisheries, were also poor. Affected reefs had proportionately more herbivorous fish, correlated with heavy colonization by algae. Both the fisheries and the tourism value of these sites will be affected, the extent of which will have to be quantified.

The reefs least affected by bleaching were those in sheltered embayments. Such bays are characterized by a level of nutrient enrichment and turbidity from terrestrial runoff, as well as natural heating from insulation in their shallower reaches. Thus, considerable bleaching in embayments should be expected, particularly if the rate of water exchange is low. However, the coral communities which survived on these reefs generally consisted of species that are tolerant of these parameters. The reef in Pemba Bay was most typical of this environment (Schleyer *et al.*, 1999).

Training Course

A training course was held in August 1999 at the Center for the Sustainable Development of Coastal Zones, Xai-Xai (MICOA) under the auspices of CORDIO and MICOA. The course was attended by a number of participants from MICOA itself, the Institute of Fisheries Research and Eduardo Mondlane University. The lecturers were from ORI and CORDIO and the material presented covered the taxonomy of fish and invertebrates as well as survey and monitoring methods. Some of the participants were later integrated into the team that started the monitoring programme.

Monitoring and monitoring station installation

Sites were selected for permanent monitoring during a preliminary survey (Schleyer *et al.*, 1999) according to a number of criteria, the sites being:

1. Representative of Mozambican coral reefs, (i.e. typical of exposed Mozambican fringing reefs or of sheltered, specialized coral communities in sheltered embayments adapted to high nutrient levels, turbidity and thermal and saline stress).
2. Evenly distributed along the extensive Mozambican coastline in areas in which corals occur.
3. Reasonably accessible.

Figure 1. Map of the coast of Mozambique illustrating the location of coral reefs and each study site.

The field work was conducted during 22 days between August and September, 1999. For the first year of monitoring, nine "core" reefs were selected for annual survey. These reefs were widely distributed throughout the coast and represent different reef types (Figure 1).

METHODS

Reef surveys

Surveys were carried out using the GCRMN-recommended strategy of recording benthos, invertebrates and fish off the same transects (English *et al.*, 1994). A major modification was to use video transects to sample benthic cover. These were done using a Sony Hi-8 Handycam video camera in a housing using S.C.U.B.A. or snorkel. The underwater housing had a spacer bar fitted to maintain a working distance of 110 cm from the reef, thus ensuring that a frame size of 0.5m x 0.5 m was filmed. A minimum of 150 m of reef was recorded during snorkel transects and 5 m x > 20 m transects surveyed during S.C.U.B.A. transects. The photography was undertaken perpendicular to the reef and the transects were filmed in a straight line within a depth contour or zone of the reef at a velocity of approximately 0.25 $m \cdot second^{-1}$. Simultaneous with the transect, an observer would conduct a general survey to establish a species and cover-type list to assist later analysis of video images.

The Hi-8 footage was transferred to VHS tape for onscreen analysis. This was undertaken by stopping the video randomly every four seconds, or when a new field was on screen if a surge had slowed the transect progress below the desired speed. The life form category (English *et al.*, 1994) was recorded at each freeze-frame under four random spots placed within each quadrant of the television screen. Estimates of percent cover were determined by the proportion of the total number of sampling points for each category. Means and standard deviations were calculated based on the number of transects at each site.

Fish and reef invertebrates

The fish surveys were conducted using the method described by English *et al.* (1994). Transects 50 m long by 5 m wide were laid (250 m^2) and weighted at both ends. The observer started at one end of the transect, recording target species from a previous determined species list. The total size of each individual was estimated and grouped under one of each of three size classes: 0-10 cm; 10-20 cm and >20 cm. After the first pass of the transect, the observer made a second pass in the reverse direction to record species that were not seen in the first run. Swimming speed was kept at approximately five minutes per 50 m pass. Target fish species included commercial species, useful indicator species and visually and numerically dominant non-cryptic species. If conditions were too rough to lay a transect line, a point count method was used. Circles with a radius of 7 m (area = 153 m^2) were used, sampling two point counts for each transect to a total area of 306 m^2 instead of 250 m^2. On one occasion, due to very poor visibility and reef fish community, a 30 minute random survey was undertaken instead of the two methods described above.

Data was analysed focusing on abundance (total number of individuals per family), diversity (total number of species per family), trophic groups (total number of individuals belonging to a specific trophic

Table 1. Summary of status and condition of each reef surveyed based on benthic cover and fish trophic groups.

Locality	Reef	Status & Condition
Quirimbas Archipelago	Sencar channel	The reef was severely affected by the 1997-1998 bleaching event. The poor status of the reef is given by the high percentage of dead coral and algae as well as the relative abundance of herbivorous fish. Most of the fish were of small size reflecting the high pressure of fishing.
Pemba	Ponta Maunhane	Reef with a good cover of hard coral and recovering from bleaching, however still significant amount of dead coral. Good reef fish community represented by all size classes of fish.
Mozambique Is.	Sete Paus Island	Presence of considerable amount of dead coral probably due to the effects of bleaching and storm damage. A poor fish community – one dominant family of herbivores and mostly small sized fish indicate high pressure from fishing.
	Goa Island	Serious damage from a cyclone and bleaching. The presence of colonizing coralline algae indicates recent death of coral. Dominance of herbivore fishes of mostly small sizes suggest heavy fishing pressure.
Bazaruto Archipelago	Lighthouse reef *Protected area*.	Reef in good condition. However, substantial amount of dead coral colonized by coralline algae may indicate damage from sedimentation and aerial exposure at low tide. Carnivorous fishes dominant and all size classes were well represented with fish larger than 20 cm very common.
Inhambane	Anchor Bay	Typical rocky reef of southern Mozambique, with a fairly low cover of hard coral. Fish were mainly herbivorous and of small and medium size classes, indicating high fishing pressure.
	Mike's Cupboard	Similarly to Anchor Bay, this rocky reef shows a richer community of soft corals. Small and medium sized fishes indicate some pressure from fishing.
Inhaca	Barreira Vermelha *Protected area*	Reef in relatively good condition, with some physical damage shown by the amount of dead coral, probably from destructive fishing practices, though fishing is banned. All fish size classes were represented and fish over 30 cm were common.
Inhaca	Ponta Torres *Protected area*	Subject to aerial exposure, shown by the percentage of dead coral and algae on the top of typical bommies. All families of fish were well represented. Carnivores of large sizes dominated.

(Source: Rodrigues *et al.*, 1999)

category) and size classes (total number of individuals from a specific size class for each family).

Mobile invertebrates included echinoderms (sea urchins, sea stars and holothurians), molluscs (giant clams), lobsters, and sea anemones. The visual census was conducted following the fish using a 2.5 m wide band along the 50 m transect, giving a total area of 125 m^2. If conditions were too rough to lay a transect, the point count method was used.

RESULTS AND DISCUSSION

The condition of reefs surveyed varied between healthy to heavily impacted by natural and anthropogenic factors (Table 1). Many reefs are degraded from bleaching and the ravages of crown-of-thorns starfish. Coral cover was highest on the reefs of northern Mozambique and in marine protected areas (Figure 2). The high cover of rock and algal surfaces reflects mortality that was reported at these sites in earlier surveys (Figure 2, and

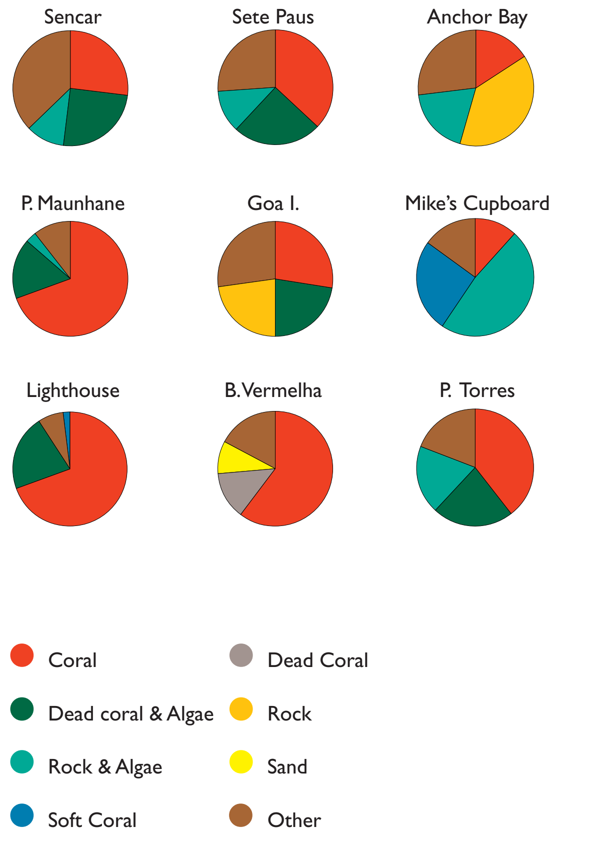

Figure 2. Benthic cover categories of permanent monitoring sites. The top three cover categories are shown for each site, the remainder grouped as "other". Lighthouse Reef, Barreira Vermelha and Punta Torres are in Marine Protected Areas.

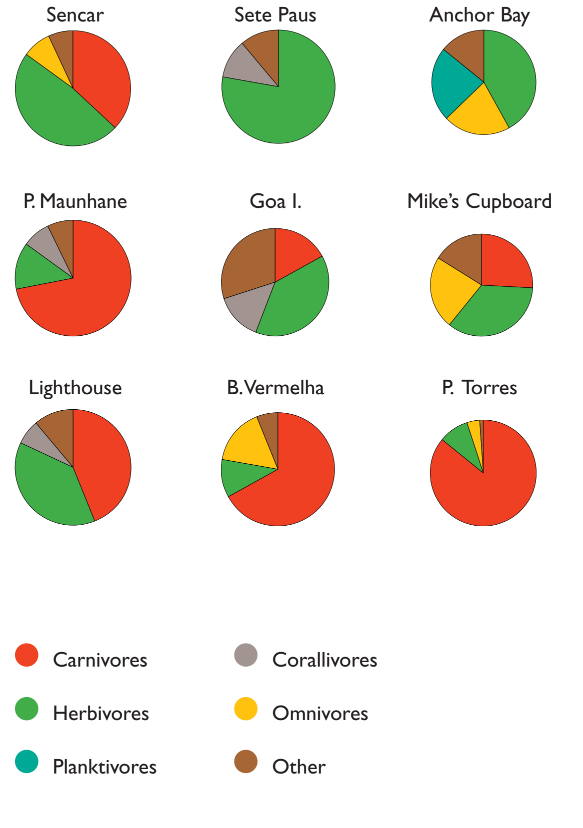

Figure 3. Dominant fish trophic groups of permanent monitoring sites. The top three trophic groups are shown for each site, the remainder grouped as "other". Lighthouse Reef, Barreira Vermelha and Punta Torres are in Marine Protected Areas.

Schleyer *et al.*, 1999). There is evidence of recovery on some reefs on which soft corals are the primary colonisers. Fish populations in the north and in protected areas were dominated by carnivores (Figure 3), following a similar pattern to that of coral cover. High fishing pressure on the other reefs was shown by the small size classes of fish and the dominance of herbivores, which are least preferred by fishermen.

The results show that reefs in protected areas are in better condition than unprotected reefs. However, considering the size of the Mozambique coastline, very little is protected in only three protected areas. Protected areas are important sources of invertebrates and fish larvae to adjacent areas. In addition, the fact that almost no turtles and few large fish were seen on the reefs surveyed is indicative that management measures need to be taken. Also, it should be taken into consideration that tourism and diving are growing industries in this country. Thus, there is an urgent need for establishment of more protected areas and sanctuaries to conserve biodiversity and provide breeding reservoirs. To ensure the successful management of Mozambique's coral reefs monitoring, of the type reported here, must continue and be expanded to include additional sites and also more detailed studies conducted on priority issues.

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The status of South African coral reefs

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The major coral-inhabited reefs in KwaZulu-Natal occur adjacent to the coast between 26° 50' S and 27° 55' S. They are thus some of the southernmost coral reefs in the world but are not typical of coral reefs resulting from biogenic accretion. Corals in South Africa grow rather as a veneer on late Pleistocene sandstone, which originates from submerged coastal sand dunes. The reefs run parallel to the coastline and are confined to the narrow continental shelf, which is between 2 km and 7 km wide in their vicinity. They can be conveniently grouped into a northern, central and southern complex, these being found respectively at Kosi Bay, between Sodwana Bay and Lake Sibaya, and north of Lake St Lucia.

The reefs range in depth from 8 m to just over 35 m, with only a few peaks approaching the surface. The coastline is straight and exposed and, as the prevailing northeasterly and southerly to southwesterly winds blow parallel to the coast, they give rise to substantial swells. The warm Agulhas Current, which has a mean peak velocity of 1.4 $m.s^{-1}$, generates the sub-tropical conditions in the area. The mean seasonal sea-surface temperatures range between 22° C in winter to 26° C in summer (SADCO data, 1960-1995), with the salinity varying between 35.0‰ and 35.5‰. Recent temperature records at a fixed station on one of the reefs are presented in Fig. 1. These manifest an upward trend that will be discussed later. The maximum tidal flux is 2 m during spring tides, dropping to 1 m during neap tides.

A consequence of the conditions described above is

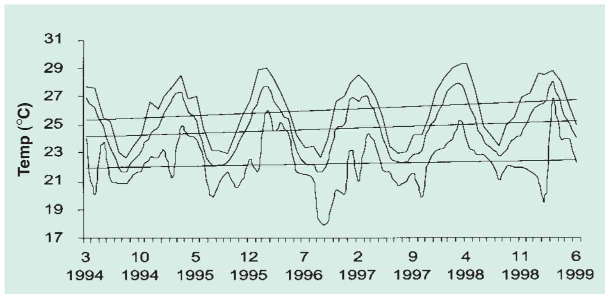

Figure 1. Mean, minimum and maximum sea temperatures at a fixed long-term monitoring site on the central reef complex at Sodwana Bay.

substantial water movement on the reefs, both in terms of a current, usually north to south (Schleyer, unpub. data), and considerable surge from the swell.

The reefs are all basically the same in structure. They lack the well-defined zonation of true coral reefs and conform to the topography of the base substratum. They thus tend to be flat with relatively few features, comprising low pinnacles and shallow drop-offs and gullies. In community structure, they can be separated into reef top and gully communities. The former are dominated by soft corals while hard corals attain a greater abundance in the latter.

Corals are the dominant life form on the reefs and 43 scleractinian (hard coral) genera have been found in the area as well as one member of the fire coral genus *Millepora*. Among the alcyonacean soft corals, 11 genera have been found, with the family Nephtheidae, comprising at least four additional genera, still under examination. The combined checklist of material identified thus far constitutes 132 species of mainly Indo-Pacific corals but includes both new and endemic species. Sponges and tunicates are also prominent in the sessile fauna on the reefs and include over 20 species of the former and 29 species of the latter.

The species richness of KwaZulu-Natal coral reefs is thus quite remarkable, constituting a biodiversity peak south of the equator rendered unique by its community structure viz., the frequent prevalence of soft corals. Overall, the three most abundant genera are, in order, the soft corals *Lobophytum* and *Sinularia* and the hard coral *Acropora*. The separation of the coral communities into those which are dominated by soft or hard corals has provided a simple and functional approach to their classification for their management.

The Oceanographic Research Institute (ORI) established a long-term monitoring site in the central complex in 1993 at which it is measuring annual changes in the reef community structure. The site is being used as a control in studying the effects of intensive diving on reefs elsewhere in the complex and to measure the consequences of climate change. The monitoring is being undertaken by photographing 80 fixed quadrats at the study site for subsequent image analysis. Subtle changes in the reef community structure have been detected and the study is ongoing. A temperature recorder was also installed at the site in 1994 at a depth of 17 m and an analysis of hourly mean data is presented in Figure 1. It is evident that the mean sea temperatures at the site are increasing at a rate of 0.25° C p.a. but it remains to be seen whether this is part of a long-term trend or merely a macro-cyclical variation. However, the greatest temperature maxima of 29.3° C and 29.4° C were attained in March and April 1998, the year of the most extreme ENSO event recorded thus far.

The ORI team undertakes fieldwork at regular intervals in the central complex and worked in the area over this period. Very limited, partial bleaching was observed of both hard and soft corals (particularly of the latter <1%) and it would appear that the threshold for this phenomenon was attained but not exceeded. The affected corals appeared to recover as no evidence of mortality could be found during subsequent visits.

The significance of these observations is two-fold, the first being that the bleaching threshold for South African corals was attained and it affected both types of coral. Secondly, and possibly more important, is the fact that the South African coral-inhabited reefs are generally found at depths of 12 m or deeper. Reefs shallower than this are found in very few places. While the turbulence of the water over the reefs generally promotes good mixing of the surface water, much warm superficial water may have passed over the corals. If the reefs were shallower, more corals may have been affected as was encountered elsewhere. As the soft corals are more prevalent on the shallow reef tops, the incidence of bleaching was more conspicuous in this group.

Crown-of-thorns starfish (*Acanthaster planci*) are also a cause of reef degradation in South Africa. The first recorded sighting of crown-of-thorns starfish (COTS) on South African reefs was made during the early seventies by ORI scientists. A spot outbreak of COTS was subsequently recorded on the outer perimeter of one of the reefs of the central complex during 1990-1998. The significance and consequences of this outbreak are currently under investigation. The marginal nature of the South African reefs makes them particularly vulnerable to disturbance-driven change in community structure. The deeper reef area infested by COTS was previously dominated by hard corals but is presently being recolonized by soft corals. The present disturbance regime suggests that the South African reefs constitute a dynamic successional reef system or one in which the hard corals are declining. The reefs are being monitored for further COTS outbreaks.

Finally, the degradation of the reefs caused by diver damage has been assessed and, while this is limited, recommendations have been made regarding their diver carrying capacity. This was estimated using a novel application of statistical techniques adapted for the purpose.

South Asia - Summary

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This summary is a synthesis of reports from Pet-Soede *et.al*., Zahir and Rajasuriya & Karunarathna presented in this volume. All numbers in the text refer to the project list below.

The reefs of South Asia, including Maldives, Sri Lanka and India were severely affected by the bleaching event of 1998, with subsequent mortality of ranging between 50% and 100%. Surveys of these reefs conducted during 1999 and at the beginning of 2000 recorded some coral recruitment, but many areas still show no signs of recovery.

INDIA

The reefs of the Gulf of Mannar were severely affected by mortality of coral during 1998. Post bleaching surveys on the coral reefs of the 21 islands in the Gulf show the mean cover of coral is approximately 26%. However, there is considerable variation between reefs with the cover of live coral ranging between 0% and 74%. In addition to reef-building corals, sea anemones and octocorals (soft corals) also bleached as a result of the increased sea temperatures that prevailed during 1998. Subsequently, a decrease in biodiversity of these reefs has been reported. Furthermore, extensive beach erosion on some islands was reported.

Initial assessments of recovery processes in Lackshadweep Islands during 1999 indicate the cover of live coral has increased to 15% to 20 % compared with the 5% to 10% reported immediately after the bleaching event. Infrastructure and capacity needed for continuation of the work are being created (1).

Early reports from the Andaman and Nicobar Islands indicated 80% of all corals bleached. However, the extent of post bleaching mortality has not been established.

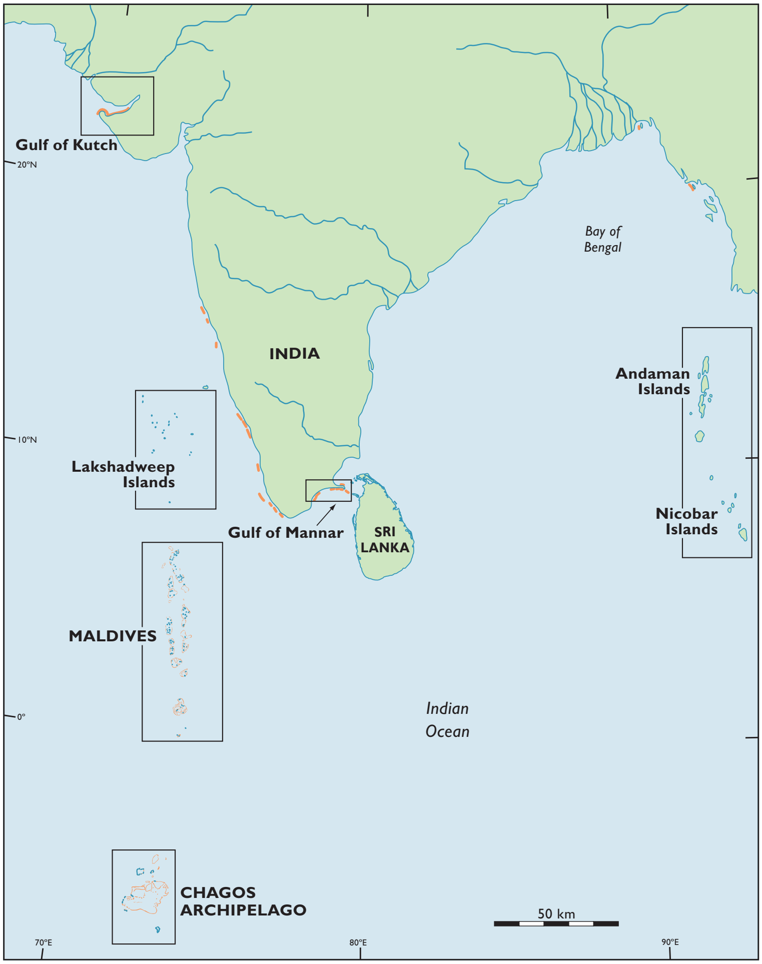
Results of a study of the socio-economic importance of reef fisheries in India (3), suggested that, despite the relative unimportance of India's reef fisheries at present, the increasing demands for food caused by burgeoning coastal populations and overexploitation of coastal shelf areas will increase the pressure on India's coral reefs to provide food and economic well-being to these coastal and island populations.

MALDIVES

Results of monitoring of the reef tops conducted during 1999 show that the cover of live coral has not increased since the post bleaching surveys conducted in 1998 and remains at approximately 2%. At present, the cover of live coral is 20 times lower than that recorded before the bleaching event. However, re-colonisation of fast growing branching corals has been recorded, indicating that reef recovery processes are underway. Furthermore, on some reefs coralline algae are abundant providing potential areas for coral recruitment. Nevertheless, despite these reasons for hope, it is clear that the reefs of Maldives were seriously affected by bleaching and subsequent mortality of coral and will require many years to recover.

In addition to biophysical monitoring of the reefs, studies were conducted to determine the spatial and temporal patterns of coral recruitment in Maldives (4).

The distribution of coral reefs in the South Asia region of the Indian Ocean. More detailed maps are provided illustrating areas of particular interest.



Initial results suggest that there is potential for the degraded reefs of Maldives to recover through the influx of coral planulae from surviving colonies elsewhere. Also, the degree of erosion and changes in the topographic complexity of these reefs are being assessed following the extensive coral mortality in Maldives (5).

The CORDIO program has also trained staff at the Marine Research Centre in Maldives responsible for conducting the CORDIO activities. The training focussed on general survey methods including taxonomic identification of major reef biota and on specific protocols for conducting assessments of recruitment and erosion of reefs.

Considering the importance of reef related tourism in Maldives, a survey was conducted to evaluate the potential economic impacts on the national economy of Maldives from declines in tourism resulting from bleaching induced degradation of coral reefs (6). Considering the importance of reef related tourism in Maldives, a survey was conducted to evaluate the potential economic impacts in the Maldives from tourism due to degradation of coral reefs (6). Results suggest that, although not severely, the tourism industry was adversely affected by the coral mortality in the Maldives (see Westmacott *et al.*, this volume).

SRI LANKA

Most shallow coral reef habitats in Sri Lanka were severely damaged as a result of coral bleaching in 1998. Surveys conducted between June 1998 and January 2000 (7) revealed that many of the dominant forms of reef building corals in the shallow coral habitats have been destroyed. Invasive organisms such as tunicates, corallimorpharians and algae now dominate the dead coral reefs. Furthermore, the dead coral patches were rapidly inundated by sediment thus preventing re-colonization of coral larvae. Also, in every area surveyed thus far, except Trincomalee in the northeast, the hydrocoral, *Millepora* spp., which was once common, appears to be completely absent. However, despite the destruction of corals in shallow water (< 8 m), corals growing in deeper waters (> 10 m) have recovered from bleaching almost completely providing a source for new recruits and reef recovery.

Recovery of bleached corals in shallow reef habitats has been extremely low and has been hindered by further damage to the reef structures by uncontrolled and destructive human activities. Even the marine protected areas in Sri Lanka are largely unmanaged and increasing human activities within these protected areas continue to degrade their condition. Considering the present condition of the reefs and the inevitability of future anthropogenic impacts, the prospects for reef recovery are poor.

The impact on fishes by the loss of live hard corals is clearly visible in the decreased abundance of several species of fish that depend on live corals for food (e.g. Chaetodonts). However, it is less obvious in fish populations in deeper water that are less dependent on live corals for their survival. Nevertheless, it is expected that the reduction in cover of live hard coral will directly affect the fishery potential of reefs through habitat degradation and loss thus, having an adverse impact on the income of coastal communities.

At present, a study is being conducted to determine the socio-economic impacts of degradation of the coral reefs of Sri Lanka, focusing on the demersal and ornamental fisheries and reef related tourism (8). Building on these results, another project is planned to investigate the prospects of providing alternative livelihoods for dependants on coral reefs (9).

In addition, CORDIO, with assistance from National Aquatic Resources Research & Development Agency (NARA), has begun training students of Eastern University in the basic techniques of coral reef monitoring, with the aim of developing the local capacity to undertake such monitoring. The training includes coral and fish taxonomy, LIT (Line Intercept Technique) and data analysis.

PROJECT LIST

India

Country co-ordinator: M.V.M. Wafar
Institute: National Institute of Oceanography

1. Recovery and monitoring of the reefs.
2. Assessment of the effects of coral mortality on reef communities.
3. Socio-economic effects on local populations and tourism.

Maldives

Country co-ordinator: Hussein Zahir
Institute: Marine Research Centre

1. Reef recovery processes: Evaluation of succession and coral recruitment in the Maldives
2. Assessing bioerosion and its effects on reef structure following a bleaching event in the Maldives.
3. The economics of coral reef deterioration with special reference to bleaching.

Sri Lanka

Country co-ordinator: Arjan Rajasuriya
Institute: National Aquatic Resources Research & Development Agency

1. Impacts of coral bleaching on reef communities.
2. Monitoring and assessment of socio-economic aspects of coral bleaching and degradation in Sri Lanka.
3. Develop alternative livelihoods for people dependent on coral reef resources.

Post-bleaching status of the coral reefs of Sri Lanka

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National Aquatic Resources Research and Development Agency, Colombo, Sri Lanka

ABSTRACT
Most shallow coral reef habitats in Sri Lanka were severely damaged as a result of coral bleaching in 1998. Surveys conducted between June 1998 and January 2000 revealed that many of the dominant forms of reef building corals such as *Acropora* spp., *Pocillopora* spp. and *Echinopora lamellosa* in shallow habitats have been destroyed. *Montipora aequituberculata* and *Porites rus* were only marginally affected. Overall, recovery of bleached corals among shallow reef habitats has been poor. Invasive organisms such as tunicates, corallimorpharians and algae now dominate these dead coral reefs. Despite some mortality attributable to bleaching, corals growing in deeper water (> 10 m) have recovered successfully.

Uncontrolled and destructive human activities are widespread and continue to cause damage to reef habitats. Even the marine protected areas in Sri Lanka are unmanaged and increasing human activities within these protected areas continue to degrade their condition. This may directly impact fish populations and the fishery potential of these reefs thus, having an adverse impact on the income of coastal communities. Furthermore, infestations of Crown-of-thorns starfish are still a major problem for recovering coral reefs on the north-west and east coasts of Sri Lanka. Considering the present condition of the reefs and the inevitability of future anthropogenic impacts, the prospects for reef recovery are poor. This report presents the current status of reef habitats at selected locations around Sri Lanka two years after the bleaching event of 1998 and discusses the implications of their lack of management in the context of reef recovery.

INTRODUCTION

In Sri Lanka three major types of reef habitats have been classified based on the dominant forms of substrate. These are coral, sandstone and rock habitats (Swan, 1983; Rajasuriya & De Silva, 1988; Rajasuriya *et al.*, 1995). The most extensive coral reefs in Sri Lanka occur in the north-western coastal and offshore waters. These are patch reefs dominated by *Acropora, Montipora* and *Echinopora*. The majority of other coral reef habitats in Sri Lanka are situated close to the shore. Abundant coral growth and large mono-specific stands of common reef building corals are limited to a depth of about 10 m. In these shallow waters the dominant reef building corals belong to the Acroporidae, Pocilloporidae, Faviidae and Poritidae. Also, offshore from both the east and west coasts coralline habitats occur at depths of approximately 20 m. However, because of the depth at which these habitats occur, they support only sparse populations of corals. In addition, sandstone and rocky habitats are extensive and support a diverse coral fauna and many associated species (Rajasuriya & De Silva, 1988; Rajasuriya *et al.*, 1995). The diversity of hard corals in Sri Lanka is relatively high, with 183 species from 68 genera recorded (Rajasuriya & De Silva, 1988; Rajasuriya, 1994).

In Sri Lanka coral reefs are important for the fisheries industry, coastal protection and tourism. However, the overall condition of reefs is poor due to damage caused by various human activities and natural causes such as the Crown-of-thorns starfish (De Bruin, 1972; De Silva, 1985; Rajasuriya & De Silva, 1988; Öhman *et al.*, 1993; Rajasuriya *et al.*, 1995). The primary anthropogenic activities causing degradation of coral reefs are mining of coral from nearshore reefs, use of destructive fishing practices, uncontrolled harvesting of reef biota, pollution and siltation resulting from deforestation and other improper land-use activities. Also, the Crown-of-thorns starfish (*Acanthaster planci*) has been one of the major natural causes of reef damage on the north-western and eastern coasts of Sri Lanka (De Bruin, 1972; Rajasuriya & Rathnapriya, 1994; Rajasuriya *et al.*, 1995; Rajasuriya & Wood, 1997). In addition, the bleaching event in 1998 caused severe mortality among corals in shallow water (Rajasuriya & White, 1998; Rajasuriya *et al.*, 1999).

The history of coral reef management in Sri Lanka is poor. Management actions have been initiated from time to time depending on the importance of management issues. In the past, such issues have been coral mining in the sea, exploitation of species and destructive fishing methods. However, actions taken to control these have been limited mainly to the introduction of new regulations under existing ordinances or parliamentary acts. Ground level action to actually protect coral reef resources have been largely unsuccessful (De Silva, 1985; Rajasuriya *et al.*, 1995; De Silva, 1997a; Dayaratne *et al.*, 1997).

The reef monitoring programme of the National Aquatic Resources Research and Development Agency (NARA) has documented the status of reefs in Sri Lanka since 1985 and has contributed to national level action plans for the protection of coral reefs and their resources. At present, funding from the national government, Sida/SAREC Marine Science Programme and CORDIO (Coral Reef Degradation in the Indian Ocean) support coral reef monitoring and conservation activities of NARA. As one of the activities within the CORDIO programme, NARA has initiated a study to determine the impact of coral mortality on the marine ornamental fishes of Sri Lanka. Data describing collections and exports are being collected from the marine ornamental fish traders and the Sri Lanka Customs Department in order to identify any major changes in the ornamental fishery as a result of the coral bleaching event in 1998.

SITE SELECTION AND SURVEY METHODS

Survey sites were selected from reef sites monitored periodically by NARA. Information provided by ornamental fish divers concerning specific sites on the east coast was also taken into consideration when selecting sites. In addition, the depth of water was considered important as corals and other zooxanthellate organisms close to the surface were affected more by bleaching than organisms in deeper waters. The depth of all shallow coral reef habitats selected ranged between 0 m and 10 m and deeper coral and other habitats were situated between 10 m and 40 m. A total of 21 locations were surveyed between June 1998 and February 2000 (Table 1 and Figure 1).

Two sites (Hikkaduwa and Weligama) were selected on the southern coast to obtain photographic records of bleaching and to determine the post-bleaching status of selected reef building corals. Numbered plastic tags were attached to these colonies and each colony was photographed once every two weeks. The status of each reef was determined by recording the benthic organisms on 50 m Line Intercept Transects (LIT) and within quadrats of 1 m^2 (English *et al.,* 1997). Belt transects measuring 50 m x 5 m were used to determine the abundance of reef organisms and associated fish communities (English *et al.,* 1997). Surveys were conducted using snorkel and S.C.U.B.A.. In addition, the manta-tow technique was also used for rapid surveys in order to ob-

Table 1. Locations and reef sites surveyed as part of NARA's /CORDIO national monitoring programme.

Site No.	Location and name of reef (Reef site)	App. distances from shore (Km)	Depth range (m)	Type of habitat
1	**Bar Reef Marine Sanctuary** (Shallow reefs)	2 –8	0 – 10	coral
2	**Kandakuliya** (Kudawa reef)	1 – 1.5	0 - 3	coral
3	**Kandakuliya** (Diyamba reef)	2 – 4	10 - 22	sandstone & coralline
4	**Negombo** (Negombo Suda)	19	18 - 22	coral
5	**Colombo** (Pitagala)	10	20 - 25	coral
6	**Colombo** (Gigiripita)	15	20 - 30	coral
7	**Colombo** (Palagala)	1	9 – 11	sandstone & coralline
8	**Colombo** (Vatiya reef)	4	25 - 30	sandstone & coralline
9	**Moratuwa** (Itipandama)	1.5	8 - 12	sandstone & coralline
10	**Moratuwa** (Bodhigala)	2.5	20 - 23	sandstone & coralline
11	**Panadura** (Rudigegala)	3	13 - 17	sandstone & coralline
12	**Hikkaduwa N. R.** (reef lagoon)	fringing reef	0 – 3	coral
13	**Hikkaduwa** (Supercoral reef)	0.3 – 0.4	7 - 13	coral & rock
14	**Hikkaduwa** (Kiralagala)	5	27 - 35	rock
15	**Rumassala** (Galle)	fringing reef	0 - 5	coral
16	**Unawatuna** (Welladewale reef & Diyamba reef)	fringing reef	0 - 6	coral
17	**Weligama** (Kapparatota reef)	fringing reef	0 – 3	coral
18	Great Basses Reef	11	2 – 30	rock, sandstone & coralline
19	**Kalmunai,** Batticoloa	1 – 8	2 - 42	coral & sandstone
20	**Trincomalee** (Elephant island & Chapel rocks)	0.5 – 0.8	18 - 30	rock
21	**Trincomalee** (Pigeon Islands)	1.5	0 – 10	coral & rock

tain information such as the presence of live corals, Crown-of-thorns starfish and damaged areas.

Neither the LIT nor quadrat method was used on reefs deeper than 25 m due to the time constraints imposed by the no-decompression limits of S.C.U.B.A. diving. These deeper reef sites were surveyed mainly to ascertain the status of individual coral colonies and determine the diversity of coral, fish and other reef species.

Plate 1. Corallimorpharians overgrowing live coral.
Photo: Nishan Perera, Sri Lanka Sub Aqua Club.

Plate 2. Infestations of corallimorpharians at Pigeon Island, Trincomalee.
Photo: Nishan Perera, Sri Lanka Sub Aqua Club.

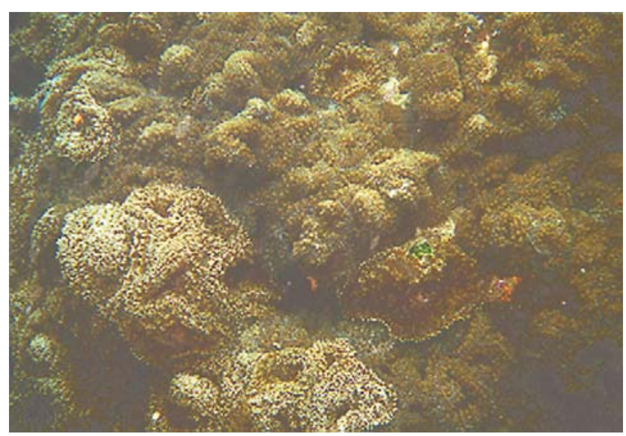
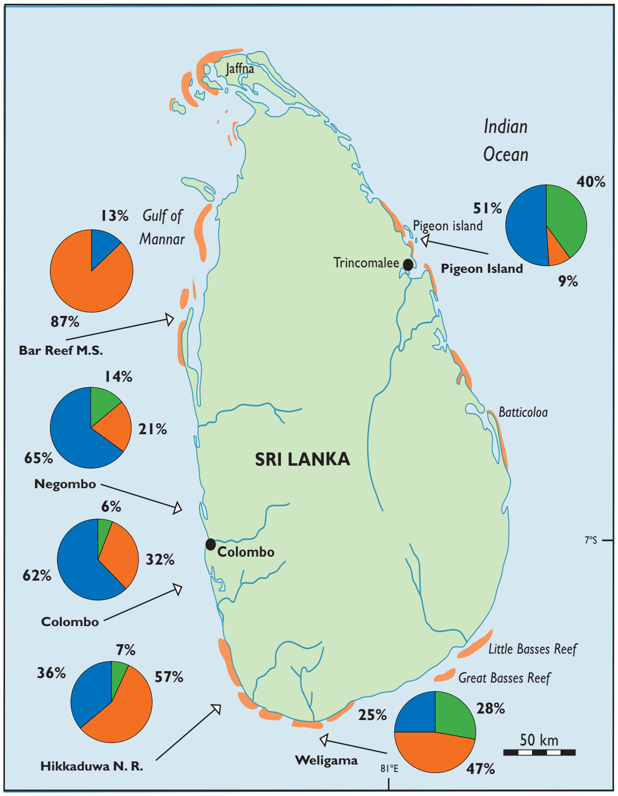


Figure 1. Distribution of coral reefs and post bleaching status of selected coral habitats in Sri Lanka.
Source: Coral Reef Research Programme, NARA



Source: Coral Reef Research Programme, NARA

Live coral
Dead coral
Other (Refer Table 2)

RESULTS

Status of investigated reef sites

Bar Reef Marine Sanctuary

Areas investigated in the Bar Reef Marine Sanctuary were limited to the coral habitats and intermediate habitats with sandstone and limestone. These were the shallow reef flats, shallow patch reefs and the adjacent areas to a depth of approximately 12 m. Live coral cover amongst the shallow reefs has been reduced to nearly 0%, while the amount of dead coral has increased to 87.5% (Table 2). In previous studies conducted at the same locations (1993 & 1994) the percentage of live coral was 78.5% (Table 3). Coral bleaching in 1998 has resulted in near complete mortality of branching and tabulate species of *Acropora* and the foliose coral *Echinopora lamellosa,* which were the dominant species in shallow areas previously.

Surveys conducted in late 1998 and 1999 revealed that shallow coral patches that were completely overgrown by two species of filamentous algae (*Bryopsis* spp). Also, a number of specimens of *A. planci* were recorded in the survey area. Although their abundance was low, the few starfish present can cause serious damage to the few surviving corals and new coral recruits. The surviving species of coral within the shallow areas were *Echinopora lamellosa, Galaxea fascicularis*, *Favia* spp., *Favites* spp., *Pavona varians*, *Pocillopora verrucosa* and *Acropora* spp.. Corals below 10 m had recovered completely by the end of 1998.

In shallow coral habitats there was a drastic reduction in the number of reef fish that were present prior to bleaching. This was particularly evident in the population of butterfly fish (Chaetodontidae). In recent surveys only nine specimens of *Chaetodon trifasciatus,* which was

Table 2. Percentage cover of substrate recorded from surveys conducted during 1999 at selected coral reef habitats in Sri Lanka.

Location	Depth	Live	Dead	*Other
Bar Reef	0–8 m	0	87.5	12.5
Colombo+	15–22 m	6.0	32.0	62.0
Hikkaduwa	0–4 m	7.0	57.0	36.0
Negombo+	15–22 m	14.4	21.0	64.6
Weligama	0–4 m	28.0	47.0	25.0
Pigeon Island	0–5 m	40.0	9.0	51.0

Source: NARA. 1999
* Soft coral; Sand; Rock; Algae & Coral rubble
+ Offshore habitats deeper than 10 m

Table 3. Percentage cover of live coral before and after bleaching at selected shallow coral reef sites in Sri Lanka.

Location	Pre-bleaching (Year)	Post-bleaching (Year)	Depth (m)
Bar Reef	78.5% (1993 & 1994)	0% (1999)	0–5
Hikkaduwa	47.2% (1997)	7% (1999)	0–3
Unawatuna	47.1% (1997)	0% (2000)	0–7
Weligama	92% (1997)	28% (1999)	0–2

Source: Rajasuriya *et al.* (1998 a & b)

the most abundant species of butterfly fish prior to bleaching, were recorded from a total of seven transects. Furthermore, none of the rare butterfly fishes (*Chaetodon bennetti*, *C. semeion* and *C. unimaculatus*) were observed and only one specimen of *Chaetodon octofasciatus* was recorded. Öhman *et al.* (1998) stated that more than 90% of all chaetodontids in the shallow coral areas of the Bar Reef were corallivores. Therefore, the loss butterfly fish can be directly attributed to the loss of living corals. An increase in the numbers of herbivores was not evident although turf and filamentous algae were abundant on these reefs. Feeding on *Bryopsis* spp. by any of the herbivores was not observed.

Although large areas of living corals have been lost, much of the reef structure was intact even in late 1999 and, as a result, some of the predators and larger herbivores that use the reef structure for ambush and refuge remained. However, an increase in coral rubble was evident in 1999 indicating that some of the reef structure has been weakened and is gradually deteriorating.

Table 4. Status of reef sites surveyed between June 1998 and January 2000.

Site No.	Location and name of reef (Reef site)	Depth range (m)	Status of habitats
1	**Bar Reef Marine Sanctuary** (Shallow reefs)	0–10	Highly degraded, now covered by algae Live coral less than 1%
2	**Kandakuliya** (Kudawa reef)	0–3	*Acropora* branching corals recovering
3	**Kandakuliya** (Diyamba reef)	10–22	Most bleached corals have recovered
4	**Negombo** (Negombo Suda)	18–22	Most bleached corals have recovered
5	**Colombo** (Pitagala)	20–25	Most bleached corals have recovered
6	**Colombo** (Gigiripita)	20–30	Most bleached corals have recovered
7	**Colombo** (Palagala)	9–11	Most bleached corals have recovered
8	**Colombo** (Vatiya reef)	25–30	Most bleached corals have recovered
9	**Moratuwa** (Itipandama)	8–12	Most bleached corals have recovered
10	**Moratuwa** (Bodhigala)	20–23	Most bleached corals have recovered
11	**Panadura** (Rudigegala)	13–17	Most bleached corals have recovered
12	**Hikkaduwa N. R.** (reef lagoon)	0–3	Highly degraded, Live coral is 7%. Heavily silted
13	**Hikkaduwa** (Supercoral reef)	7–13	Most bleached corals have recovered
14	**Hikkaduwa** (Kiralagala)	27–35	Mainly Ahermatypic corals
15	**Rumassala** (Galle)	0–5	Highly degraded, Live coral about 10%
16	**Unawatuna** (Welladewale reef & Diyamba reef)	0–6	Highly degraded, Live coral is almost 0%
17	**Weligama** (Kapparatota reef)	0–3	Partially recovered live coral
18	Great Basses Reef	2–30	No bleaching was observed
19	**Kalmunai,** Batticoloa	2–42	Last surveyed in Sep. 98, highly bleached, present condition not known
20	**Trincomalee** (Elephant Island & Chapel rocks)	18–30	Mainly Ahermatypic corals
21	**Trincomalee** (Pigeon Islands)	0–10	No bleaching was observed

Kandakuliya

Most of the nearshore reef at Kandakuliya was buried by sand due to beach erosion along the northern shore of the Kandakuliya village. The offshore reef (Kudawa reef), located approximately 1 km from the shore, possessed living branching *Acropora formosa* and *Acropora* spp. in late 1998 indicating that these corals had recovered relatively rapidly following bleaching. Also recorded from this reef were large *Porites* domes and other massive corals of the family Faviidae. Dead patches were observed on some of the large *Porites* domes, but it was difficult to determine whether this mortality was a consequence of bleaching in 1998 or from other causes such as feeding by *A. planci*. Corals on the sandstone habitats in deeper water (~18 m) were in good condition.

The shallow coral reefs were degraded even prior to bleaching due to the use of destructive fishing techniques such as bottom set nets (Öhman *et al.*, 1993). As a result, fish abundance was relatively low compared with Bar Reef. However, fish were abundant below 12 m in the sandstone and limestone habitats. A reduction of the fish population in these habitats was not apparent. This is probably because the majority of fish in sandstone-limestone habitats are not highly dependent on live corals and therefore, were not affected by the bleaching.

Negombo

The main reef site at Negombo is located approximately 19 km west of the lagoon outfall. The shallowest part of the reef is at a depth of 15 m and consists primarily of *Porites* domes that attain a diameter of approximately 5 m. Many of these domes had patches of dead coral but it was not possible to ascertain the cause of these dead patches. No bleached corals were seen in early 1999 and live corals were in good condition. However, an increase in the amount of filamentous algae on coral rubble and dead coral was recorded in early 1999.

A reduction in the abundance of fish was not evident although the use of damaging fishing gear such as bottom set nets is a major threat to the corals and other reef building organisms (Rajasuriya & Wood, 1997). The most abundant species were Acanthurids (*Naso hexacanthus* and *Acanthurus mata*). The Negombo reef also supports many less abundant ornamental fish species, particularly the long-nose butterfly fish (*Forcipiger longirostris*) and *Chaetodon madagascariensis* which are relatively rare in other areas of the West Coast.

Colombo

Four reef sites (Pitagala, Gigiripita, Vatiya and Palagala) were investigated. 'Pitagala' and 'Gigiripita' Reefs have coral habitats similar to Negombo Reef but they are smaller in size and are located approximately 10 km to 15 km offshore at depths between 18 m and 35 m. Both reefs contain similar habitats and support primarily medium sized *Porites* domes (1.5 m - 2 m diameter), and colonies of *Symphyllia* sp., *Pocillopora* sp., encrusting species of *Montipora, Echinophyllia*, *Acanthastrea* and small Faviid colonies. Also present are small (0.5 m), scattered colonies of *Acropora* spp. All living coral colonies were healthy in late 1999.

Vatiya Reef is located approximately 4 km offshore at a depth of 27 m to 30 m. This reef consists of sandstone and old limestone slabs. Living corals are mainly encrusting and laminar growth forms. Bleaching was observed in 1998 but recent surveys indicated that all corals had recovered completely.

Palagala Reef lies approximately 1 km offshore. The reef structure is similar to that of Vatiya Reef where live coral is sparse. Almost all large colonies were bleached in 1998, but have since recovered. Recent investigations in early 2000 revealed that most soft corals (*Dendronephthya* sp.) have bleached, probably indicating the influence of other stresses since this reef is closer to shore and may be affected by the influx of fresh water and pollution during rainy periods. However, similar bleaching of *Dendronephthya* sp. was observed in 1997, raising the question of whether bleaching of soft corals is an indicator of overall changes in the environment?

Moratuwa and Panadura

Moratuwa and Panadura Reefs are mainly sandstone and old limestone mixed among rocky habitats. The average cover of live coral ranged between 10% and 15% although small areas (< 100 m^2) supported coral cover approaching 40%. The dominant corals were encrusting, massive and laminar forms. Almost all corals were bleached in 1998 but have since recovered.

Hikkaduwa Nature Reserve

This shallow coral reef once supported extensive areas of branching *Acropora* spp. but was completely destroyed by bleaching. Live coral cover was 7% in 1999 due to the recovery of several colonies of *Porites*, *Goniastrea*, *Favites* and *Favia.* Colonies of *Montipora aequituberculata* were not affected by the bleaching in 1998.

The abundance of butterfly fish has been reduced considerably due to loss of live branching and tabulate *Acropora* spp. The most common species *Chaetodon trifasciatus* has declined from five individuals per transect (Öhman *et al.*, 1998) to less than three individuals in post bleaching studies. At present, the Hikkaduwa Reef lagoon is highly degraded by sedimentation and the loss of live corals. Many corals are now buried under sand and the exchange of water has also been restricted by the accumulation of silt in one of the passages at the southern end of the coral patch. Dead branching corals were beginning to break up into coral rubble by late 1999. De Silva (1997b) reported that the abundance of calcareous alga *Halimeda* sp. had increased to a level where it was impeding the growth of live coral and was also contributing to the increased sedimentation. This species continues to be a problem within the reef lagoon.

A coral and rock habitat just outside the southern boundary of the protected area supports coral in good condition. However, this area is relatively deep (~10 m) and therefore, the recovery of these corals is similar to those on other reefs at similar depths. The dominant coral genera were *Favia, Favites, Plesiastrea, Goniastrea* and *Echinophyllia*.

Rumassala Reef

The condition of the Rummassala Reef was similar to Hikkaduwa Reef but more colonies of coral survived following the bleaching event because of their location in deeper water (6 m – 8 m). Colonies of *Porites rus,* which was the dominant species in shallow water, was not bleached in 1998.

Sedimentation at Rummassala Reef was high and the general appearance of corals was poor and comparable to those corals in the Hikkaduwa Reef lagoon. However, healthy corals belonging to the families of Mussidae and Pectiniidae were found on the rocks close to the watering point on the Rumassala headland.

Unawatuna

Coral bleaching in 1998 has destroyed almost all living corals at Unawatuna. Surveys carried out in January 2000 determined that only colonies smaller than 1 m in diameter are alive. Species of coral recorded were *Porites* sp. *Pocillopora eydouxi, Acropora* sp. and *Favites abdita* but generally the cover of live coral was negligible. This strongly contrasts against investigations conducted in 1997 that reported cover of live coral of 47.1% (Table 3). Fish abundance was extremely low. Only one specimen of the butterfly fishes *Chaetodon citrinellus, Chaetodon decussatus* and *Chaetodon trifascialis* was recorded.

Local tourism is a major threat to the reef at Unawatuna. Trampling of the coral by reef-walkers, removal of corals for souvenirs and increasing numbers of glass bottom boats threaten the recovery process of this reef.

Weligama

All corals at Weligama were severely bleached except *Montipora aequituberculata* and *Psammacora contigua* which were only marginally affected and had recovered completely by the end of 1998. In some parts of the reef, colonies of *Acropora formosa* recovered after being bleached for up to five months. All other corals including *Pocillopora verrucosa*, *P. eydouxi P. damicornis,* branching and tabulate *Acropora* spp. and the hydrocoral *Millepora* were totally destroyed. Parts of the reef that died after the bleaching are now covered in *Halimeda* sp. and filamentous algae. Despite this, corals located in shallow water (0 m – 3 m) on the reef at Weligama exhibited the greatest recovery of any corals surveyed in shallow water anywhere in Sri Lanka (28%)(Table 2). Prior to bleaching the cover of live coral was as high as 92% (Table 3). With the loss of live corals there was a marked reduction in the numbers of fish, particularly butterfly fish. Also, extensive damage was being caused to the reef by the indiscriminate catching of ornamental species by snorkel divers using crow bars to chase fish into 'moxy nets' which are similar to small cast nets (Öhman *et al.*, 1993).

Great Basses Reef

Great Basses Reef is not a true coral reef but a combination of rock, sandstone and old limestone on which isolated colonies of coral grow. Coral bleaching was not recorded from Great Basses Reef. Recent surveys indicated that hermatypic corals were healthy.

The recent increase in the harvesting of chanks and ornamental fish may have an adverse impact on the fauna of this area. Great and Little Basses Reefs have also been identified for protection within a marine protected area.

Kalmunai

Widespread bleaching was recorded from the east coast around Batticoloa. The areas surveyed were located around Kalmunai, which is about 20 km south of Batticoloa. Surveys conducted in September 1998 revealed that the entire shallow coral reef was bleached and also that colonies of *Gonipora stokesi* and *Leptoseris papyracea* situated at a depth of 42 m were bleached. The present condition of these reefs is not known due to inaccessibility.

Trincomalee

The sites investigated at Trincomalee included the Pigeon Islands and Elephant Island. Corals at both locations were not bleached. Investigations were carried out in late 1998 and on five occasions during 1999. The coral reefs of Pigeon Island were completely destroyed by outbreaks of *A. planci* during the 1970's and early 1980's but have since recovered. Branching and tabulate species of *Acropora* were the dominant corals. However, some parts of the reef where coral rubble was present have been invaded by corallimorpharians that have subsequently begun to encroach upon an area supporting live colonies of *Acropora* (Plates 1 & 2).

OBSERVATIONS OF CORAL RECRUITMENT

Settlement of new corals was observed in most sites. However, these colonies were very small and may take several years to grow into larger colonies that can sustain the structure of the reef. The most common newly recruited species were *Galaxea fascicularis, Acropora formosa, A. valida, Acropora* spp.*, Pavona varians, Pocillopora eydouxi, P. damicornis, P. verrucosa, Echinopora lamellosa, Favia* spp., *Favites* spp. and *Goniopora* spp..

DISCUSSION

These studies have revealed that whilst living corals within the top 5 m of water were damaged severely, corals deeper than 8 m to 10 m have recovered well. Also, in some locations, colonies of the branching coral *Acropora formosa* recovered approximately five months after they initially bleached. The hydrocoral *Millepora* spp. has been almost completely lost from the areas that were affected by the bleaching. The initial bleaching response of soft corals and some sponges was similar to that of hard corals. However, soft corals and sponges recovered more rapidly than hard corals.

Bleaching of hard corals has been observed in the past (1980's and early 1990's), particularly in tabulate *Acropora hyacinthus* and *Acropora cytherea*. However, such bleaching did not draw significant attention as it usually occurred only in single colonies within large areas of healthy reef. Similarly, bleaching of soft corals (*Dendronephthya* sp.) was observed in 1997 on an offshore reef off Colombo.

The impact on fishes caused by the loss of live hard corals was clearly visible in the reduction in abundance of several species of fish associated with live corals, especially butterfly fishes (Chaetodontidae). However, changes in fish communities were less obvious in habitats below 8 m to 10 m where live coral cover is normally low and the species of fish are not totally dependent on live corals in their immediate habitat. Nevertheless, links in the food chain and also recruitment of juveniles between shallow water coral habitats and offshore deeper habitats could result in a reduction in fish abundance and species diversity. Immediately after bleaching some species, such as the *Gobiodon citrinus*, were completely absent from habitats that were once dominated by branching *Acropora* spp.. However, *G. citrinus* was recorded at Weligama Reef after some of the bleached branching corals had recovered in late 1998.

Although recovery of damaged reefs was observed, it is clear that good management practices are needed in order to improve the condition of reef habitats in Sri Lanka. As reported by Rajasuriya *et al.* (1995) and De Silva (1985; 1997a; b) management of human activities is poor. As a result, destructive fishing methods such as blast fishing and the use of bottom set nets and moxy nets to catch ornamental species and uncontrolled harvesting of resources is widespread. Within the last few years, initiatives in management actions have attempted to control coral mining in the sea, protect individual species from export and have declared several new regulations and a licensing system under the fisheries ordinance. Furthermore, education and awareness building has been a priority of the Special Area Management (SAM) of Coral Reefs at selected locations. Unfortunately, such initiatives have not produced the desired results because of difficulties and problems in management and the situation in marine protected areas (De Silva, 1985; 1997a; Rajasuriya *et al.*, 1995; 1999).

Lack of proper management will have serious long-term implications for the recovery of reef resources, particularly when reefs have been highly degraded by bleaching. It is important to think in terms of ecosystem conservation rather than protection of individual species. Furthermore, if the reefs are going to recover and survive in the long-term, it is vital that existing laws and regulations are implemented impartially within the present framework of laws and regulations.

ACKNOWLEDGEMENTS

We wish to thank the Chairman and the Director General of NARA, Dr. Olof Linden of Sida/SAREC Marine Science Programme and Coordinator of CORDIO programme and Mr. Dan Wilhelmsson, Regional Coordinator for South Asia CORDIO programme for their support for coral reef research and monitoring. We also wish to thank Dr. Malik Fernando, Mr. Nishan Perera and other members of the Sri Lanka Sub-Aqua Club for participating in the surveys, Reef Check and for providing information. The support given by Mr. Darrel Fryer and the ornamental fish collectors is also acknowledged in providing logistical support in the field as well as information on coral bleaching and impact on marine life.

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Status of the coral reefs of Maldives after the bleaching event in 1998

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INTRODUCTION

A pilot reef monitoring study was conducted in 1998 to assess the extent of coral bleaching in the Maldives. The aims of this monitoring exercise were:

1. To quantitatively document the post-bleaching status of the shallow-water coral communities on the reefs of the north, central and southern regions of Maldives.
2. To estimate bleaching-induced coral mortality by comparing data yielded by the pilot survey with data from previous surveys, especially those sites for which historical data are available.

METHODS AND SURVEY LOCATIONS

Site Selection

The reef area of Maldives is enormous and the resources available for monitoring it are small. Reefs that would provide a description of the current post-bleaching status of the coral communities throughout Maldives were selected. The sampling sites were chosen in the following region (Figure 1):

- *Haa Dhaal* (north and a regional development target)
- *Male* (east central with intensive tourism and other commercial activities)
- *Ari* (east central with intensive existing tourism development)
- *Vaavu* (south central with a community-based integrated island resource management project underway).
- *Addu-Gaaf Alif* (south and a regional development target)

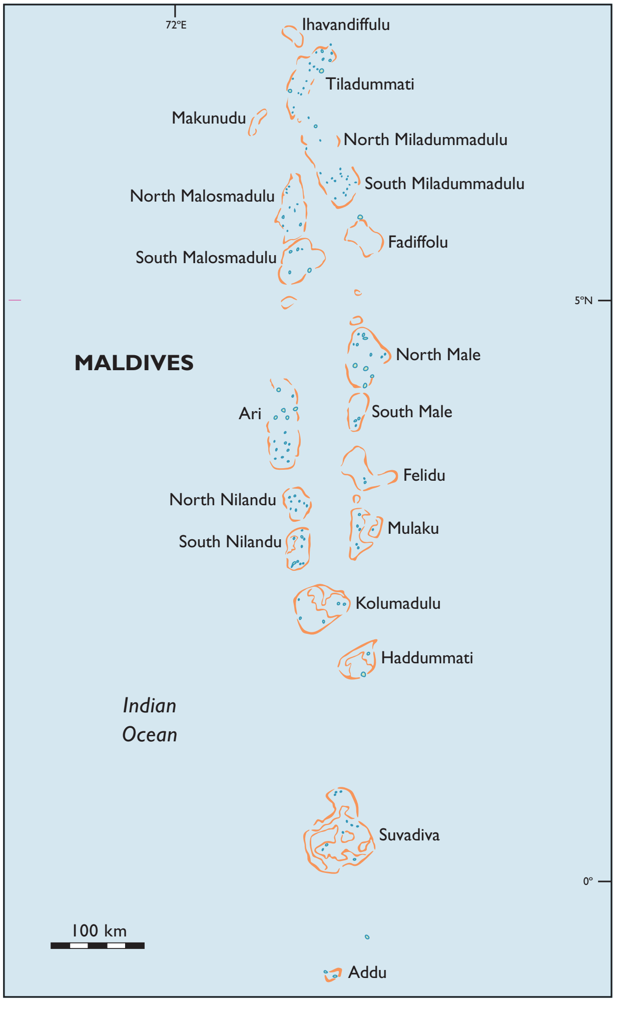

Figure 1. The reef area of Maldives with each survey site illustrated.

In each of these regions three reefs were selected ensuring that reefs that had been surveyed in the recent past were included thus providing baseline data against which data obtained from this study could be compared.

Because virtually all of the previous studies were conducted on the reef top and for logistical efficiency, all of the quantitative surveys conducted in this pilot study were conducted on the reef top also. Surveys were also confined to inner reefs within the atolls because this is where past surveys had been conducted and also because the surge caused by oceanic swells ensures that working in shallow water on outer reefs is usually impossible.

SURVEY METHOD

On each surveyed reef, data from three line intercept transects of 50 m (English *et al.*, 1997) were recorded in areas near the location of past survey sites and where physical conditions such as wave action permitted. Occasionally, when it was judged efficient to do so, a 50 m long line point insect transect was used. Surveys of the same sites were repeated in 1999 as part of an ongoing monitoring programme to assess the status of the reefs and to compare the results with those of the post bleaching study. It is anticipated that these sites will become permanent monitoring sites and will be surveyed annually to provide an insight to the processes of reef recovery especially after the bleaching in 1998.

RESULTS

A summary of post-bleaching data shows that the mean cover of live coral was 2.1% and ranged between 1.0% and 3.1% among the different atolls surveyed (Table 1) which is comparable to MRS Reef Check estimates of 1.7% (Table 2). This is in stark contrast with pre-bleaching levels of 46.5% (Table 2) and 37.4% (Table 3) (Figure 2). Although the cover of live coral is uniformly low, there is a suggestion of slight difference among atolls. Members of the family Acroporidae, excluding *Astreopora*, were rarely seen on the reef top, whereas poritids and agariciids, despite suffering high mortality, have survived best.

Table 1. Summary data from transects surveyed between August and October, 1998. Estimates for each reef comprises three transect surveys which are pooled to calculate the aggregate estimates.

Regions surveyed	Reef Number			Aggregate
	1	2	3	
Vaavu Atoll				
Mean % cover	2.8	1.3	4.7	2.9
Standard deviation	0.92	0.42	1.69	1.82
No. of transects	3	3	3	9
Ari Atoll				
Mean % cover	0.5	2.1	0.2	1.0
Standard deviation	0.38	2.03	0.28	1.36
No. of transects	3	3	3	9
Haa Dhal Atoll				
Mean % cover	0.4	1.6	0.8	1.0
Standard deviation	0.49	0.19	0.94	0.75
No. of transects	3	3	3	9
Addu & Gaaf Atolls				
Mean % cover	3.9	4.3	1.0	3.1
Standard deviation	1.81	2.54	0.24	2.21
No. of transects	3	3	3	9
North & South Male Atolls				
Mean % cover	1.4	5.3	1.0	2.6
Standard deviation	0.82	3.21	1.09	2.71
No. of transects	3	3	3	9
All Transects,				
All Atolls				2.1
Mean % cover				2.06
Standard deviation				45
No. of transects				n=45

Results from the 1999 surveys show that the mean cover of live coral is 1.9% and ranges between 0.33% and 3.04% among the atolls surveyed (Table 4). These results are compared with those of the post bleaching study and are shown in figure 3.

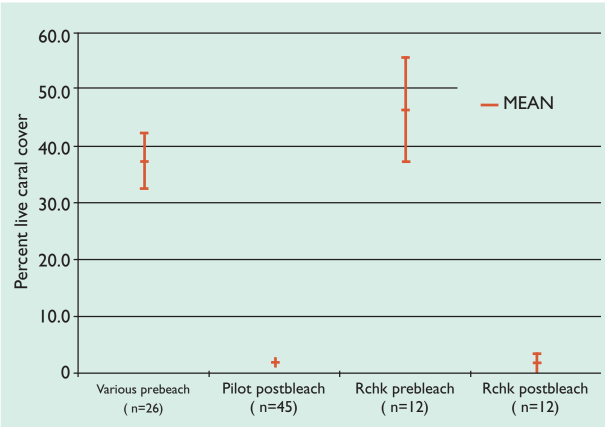

Figure 2. Estimates of live hard coral cover prior to and after the bleaching event plotted as mean bounded by upper and lower 95% confidence intervals. Pilot post-bleaching data from pilot project field work. Various pre-bleaching data from Coral Reef Research Unit, Riyaz *et al.* (1998). Rchk = Reef Check data (Hussein, *et al.*, 1998) from MRC Reef Check surveys in August 1997 (pre-bleaching) and August 1998 (post-bleaching).

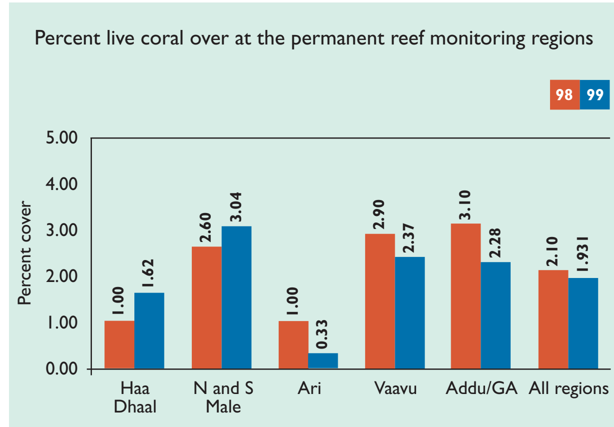

Figure 3. Comparison of estimates of the cover of live coral obtained immediately following the bleaching event.

Table 2. Reef Check data showing 1997 and 1998 coral cover at three permanent transect sites. Source: Marine Research Centre.

Reefs	1997 Transect Estimates						1998 Transect Estimates					
	Ta	Tb	Tc	Td	Mean	SD	Ta	Tb	Tc	Td	Mean	SD
Patch Reef, Vaavu	60.0	57.5	65.0	47.5	57.5	7.36	0.0	0.0	0.0	0.0	0.0	0.0
Thuvaru	22.5	22.5	22.5	45.0	28.1	11.25	0.0	0.0	0.0	0.0	0.0	0.0
Maduvvaree	70.0	50.0	50.0	45.0	53.8	11.09	10.0	7.5	0.0	2.5	5.0	4.56
Grand Mean	46.5						1.7					
SD	16.39						3.43					

Table 3. Estimates of live hard coral cover prior to bleaching. Source: Marine Research Centre and Naeem *et al.*, 1998.

Reef	Atoll	Date	Transects 1	2	3	4	5	6	7	8
Gan	Addu	29/09/97	60.0	45.0	57.5	30.0	35.0	22.5		
Khoodhoo	Gaaf Alifu	15/04/98	22.5	42.5	35.0	50.0	12.5	22.5	22.5	45.0
Bandos	N. Male	05/05/98	28.4	45.1	38.8	36.7				
Udhafushi	N. Male	29/06/98	51.5	26.4						
Kudahaa	N. Male	30/06/98	45.1	30.0						
Rasfari	N. Male	01/07/98	44.5	48.5						
Embudhu Finolhu Far	S. Male	05/07/98	31.0	44.3						
Grand Mean		37.4								
SD		12.05								

Table 4. Summary data from transects surveyed from April to June 1999. Estimates for each reef comprise three transect surveys which are aggregated to calculate the aggregate estimates.

Regions Surveyed	Aggregate
Vaavu atoll	
Mean percent cover	2.37
standard deviation	1.29
Number of transects	8
Haadhaal atoll	
Mean percent cover	0.33
standard deviation	0.41
Number of transects	9
Ari atoll	
Mean percent cover	1.62
standard deviation	2.18
Number of transects	9
Addu/Ga. Atoll	
Mean percent cover	2.28
standard deviation	1.92
Number of transects	6
N/S Male atoll	
Mean percent cover	3.04
standard deviation	2.67
Number of transects	9
All Transects (5 regions)	
Mean percent cover	1.931
standard deviation	2.047
Number of transects	41

DISCUSSION

The post bleaching study data show that only a small amount of live coral cover (~ 2%) remains on the reef tops surveyed. Qualitative observations made by many other people in other parts of the country are consistent with these quantitative surveys and lead to the conclusion that this is the general condition of reef tops throughout Maldives. Surveys conducted before and during the bleaching event indicate that live coral cover was approximately 20 times greater prior to the event. Although quantitative data describing the abundance of *Acropora* and *Pocillopora* prior to bleaching are unavailable, it is well known that they were common. Indeed, *Acropora* was often the dominant coral on many reefs.

Repeated surveys of the same sites six months later indicated the cover of live coral remained very low at all sites. Indeed, each site surveyed, with the exception of Ari Atoll and North / South Male, possessed less live coral one year after the bleaching event than it did immediately after indicating subsequent mortality of corals and negligible recovery. Furthermore, it is suspected that Ari and Haa Dhal Atolls were affected more than the other regions surveyed and the low level of coral cover was consistent with consecutive sampling.

Despite the grim picture painted by these data, the survey team has observed new coral recruits at all sites. Re-colonisation of fast growing branching growth forms were observed ten months after the bleaching event, indicating that reef recovery processes were already underway (Clark *et al.*, 1999). Several observations bode well for the recovery of these reefs. For example, many of the new recruits belonged to the genus *Acropora* which was the genus most seriously affected by the bleaching in 1998 (Figure 4-5). In addition, on some reefs encrusting coralline algae are abundant (Figure 6) providing potential areas for coral settlement and recruitment and in some regions (e.g. Haa Dhaal) large

Figure 4. Settlement and recruitment of fast growing acroporid species indicates that natural recovery is occurring.

Figure 5. *Acropora* was the genus of coral that was perhaps the most severely affected by the bleaching event of 1998.

Figure 6. Coralline algae (pink areas) have been implicated as being an important promoter of settlement of coral larvae. The abundance of coralline algae on the reefs of Maldives indicates that settlement and recovery of these reefs will not be hindered by the unavailability of suitable substrate.

Acropora tables that were believed to be dead are regenerating live tissue indicating prolonged recovery of some species of coral.

The impacts of the 1998 bleaching event will not be fully understood for some time. However, it is clear that reefs will be modified as a result of this bleaching event. In the short term (< 5 years), reefs formerly dominated by branching species will be dominated by non-living

Figure 7. An increase in the size of populations of grazing herbivores brought about by greater abundance of algae following the 1998 bleaching event could increase the rate of erosion of the reef framework.

substrate supporting only a low percentage cover of living corals of which the majority will be massive species. The consequences of bleaching for the reef framework will largely depend on the transport and fate of calcium carbonate (CaCO3) fragments. Where reef disturbance is severe, boring and grazing organisms may remove CaCO3 faster than primary frame-builders can add to it (Figure 7). Such biogenic processes will determine whether the integrity of the reef structure will be compromised.

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The status of the coral reefs of India following the bleaching event of 1998

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INTRODUCTION

The major reef formations in India are restricted to the Gulf of Mannar, Palk Bay, Gulf of Kutch, Andaman and Nicobar Islands and the Lakshadweep Islands (Hoon, 1997; Wilkinson, 1998) (Figure 1). Each area, particularly the Gulf of Mannar, faces specific problems resulting from anthropogenic influences such as high fishing pressure (trawls), high sedimentation from poor upland and coastal agriculture practices, and high levels of pollution. Furthermore, the reefs of Lakshadweep and Nicobar Islands are considered the most polluted in the Indian Ocean because the seas around them serve as major routes for oil tankers (Bakus, 1994).

STATUS OF CORAL REEFS

The bleaching event in 1998 affected the coral reefs of India to various extents, with Andaman and Nicobar Islands suffering the greatest mortality of coral (up to 80%) followed by the Lakshadweep (43% - 87%) and the reefs of the Gulf of Mannar (an average of 60%). The corals of the Gulf of Kutch were less affected (<30 %), which could be due to a greater tolerance of higher sea temperatures resulting from their occurrence in the extreme arid conditions in the north-west of India (Wafar, 1999).

Surveys carried out by Marine Biological Station, Zoological Survey of India, on the coral reefs of the 21 islands of the Gulf of Mannar during 1998 and 1999 (Venkataraman, unpublished report), revealed that the area was severely affected by post bleaching mortality and a number of reefs are considered to be in a critical condition. Results show that the mean cover of coral was approximately 26% (Figure 2) and ranged between 0% and 74%. Also, sea anemones and octocorals were affected by the bleaching and a decrease in the biodiversity of these reefs was reported. Further, several areas were covered with algae to a large extent.

Initial assessments of recovery of corals on six of the Lakshadweep Islands conducted during October and November 1999 indicated an increase in cover of live coral from between 5% and 10% immediately after bleaching to between 15% and 20%. Additional estimates of recovery will be obtained from permanent transects laid at Kadamat Island in February and March, 2000.

The condition of the reefs of Andaman and Nicobar Islands ranged from poor (~10% live coral cover) at Little Andaman to good (~70% live coral cover) at Middle Andaman (Sastry, 1998). Early anecdotal reports indicated 80% of corals in Andaman and Nicobar Islands

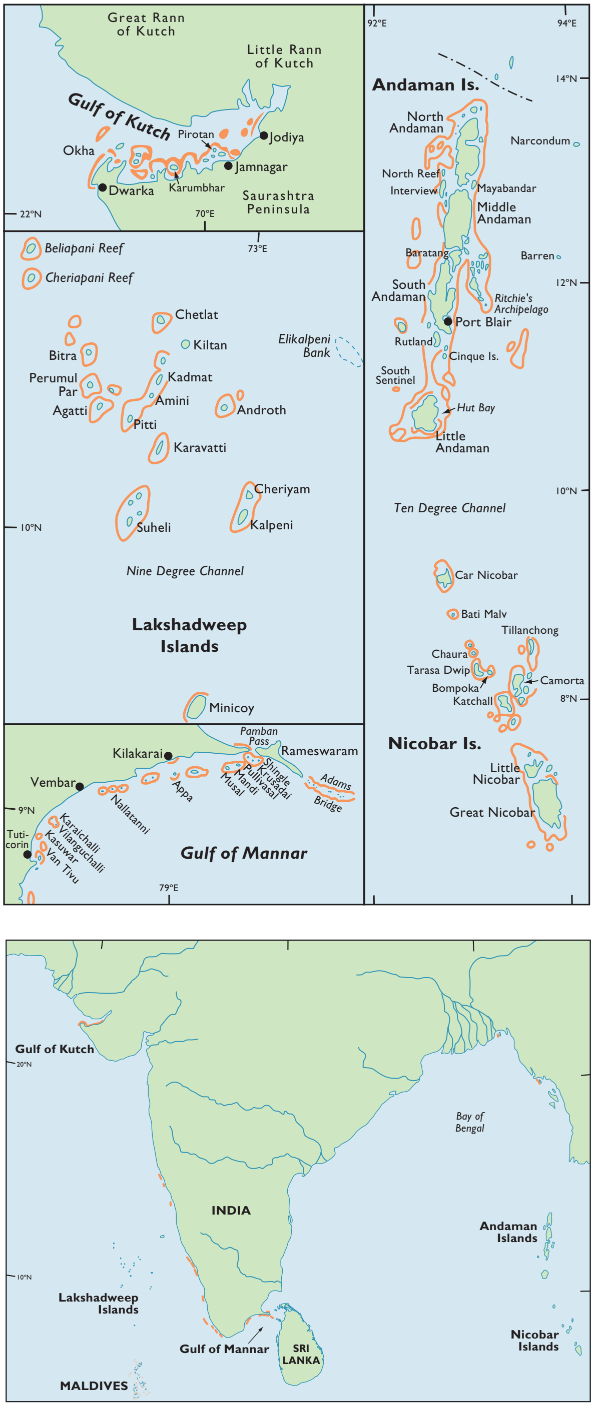

Figure 1. Coral reef areas of India.

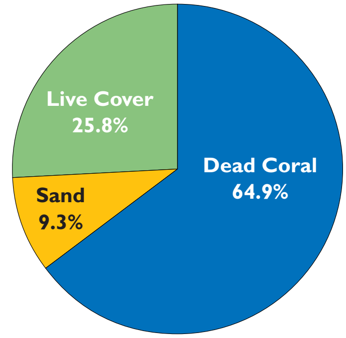

Figure 2. Benthic cover types recorded from the reefs of the Gulf of Mannar. Sorce: Venkataraman, unpublished report.

bleached, but the extent of post bleaching mortality has not been established. Although some qualitative observations report bleaching and some coral mortality at several reefs, quantitative data are not available primarily because line transects performed last season did not specifically record bleached areas.

THE IMPORTANCE OF REEF FISHERIES IN INDIA

When discussing the importance of reef fisheries for a country, it is important to distinguish between its importance in providing food, foreign currency and employment. In India, the reef fishery is not very important in terms of total fish landings or earnings when compared to other fishing sectors such as the demersal fishery in the shallow coastal areas or the tuna fishery around islands of Lakshadweep. Most Indian fishers make their living from either the pelagic fisheries, the prawn trawl fishery or the small-scale demersal fishery. However, the focus of India's fisheries are likely to change due to increasing demand from foreign markets for high quality reef fish such as grouper and snapper and also because of over capitalisation and exploitation of coastal shelf areas. So whereas reefs are hardly considered important for the fishery at present they may become so.

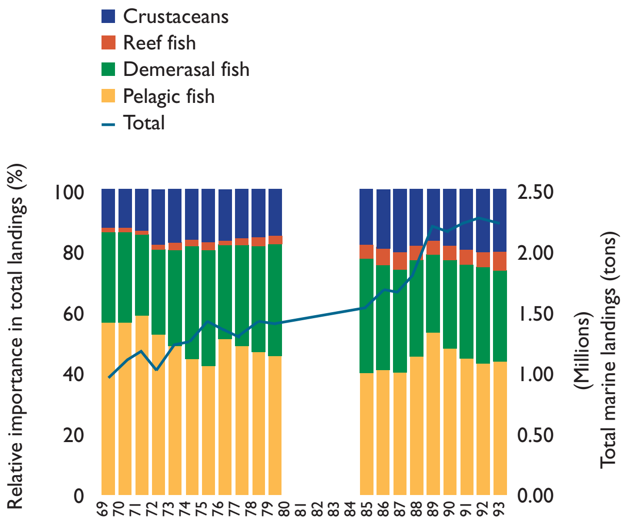

Figure 3. Trends in total fish landings and composition for India between 1969 and 1993.

The low relative contribution of reef fish to the total marine fish landings (Figure 3) illustrates how India, with its limited area of coral reefs, does not exploit its reef fish resources as much as other densely populated coastal areas of the world (e.g. Indonesia). Reasons for this are:

1. Most reefs are found in regions that are lightly populated such as Andaman and Nicobar Islands.
2. There is a high demand for large pelagic fish such as mackerel and tuna at both domestic and export markets and, as a consequence, the fishery in large reef areas such as Lakshadweep focuses on these large pelagic species rather than reef fish.

However, it is feared by the fishers Andaman and Nicobar Islands that fishers from the Gulf of Mannar, who are faced with declining fish catches in their own waters, will increasingly turn their attention toward the waters surrounding these remote islands. Furthermore, since 1969 there has been a significant increase in the relative importance of reef fish to the total landings indicating that the pressure on reef fish resources is likely to increase.

Mainland India

The total value of fish exported from India was estimated at more than 500 million US$ in the early1990's. This is an enormous increase from the 5 million US$ that was exported in 1951 (Anonymous, 1995). Although this can be explained, in part, by the greater number of fish landings, the most significant factor is the drastic increase in the value of fish. Between 1990 and 1995, the value of 1 ton of fish at Goa state has increased three fold from 0.3 million Rs to 1.1 million Rs (CMFRI, 1995). Of India's coastal states, Tamil Nadu and Kerala produce the largest landings of perches which, according to CMFRI, includes true reef fish species such as rock cods, snapper, emperors ("pig-face breams"), threadfin breams and other perches (Sivaraj *et al.*, 1992). For India, threadfin breams contribute most to the overall catch (Kumaran & Gopakumar, 1986). Of the reef areas, the largest catches, in terms of absolute biomass, are reported from the Gulf of Mannar and are more than an order of magnitude higher than the next most productive region, the Lakshadweep Islands. Andaman and Nicobar Islands produce the least of all Indian reef areas in absolute figures. The coastal shelf fishery in Gulf of Mannar produces mostly small pelagic fish such as sardines (25%) and few true reef species. In monetary terms, the prawn trawl fishery is most important. However, its damaging effects to the coastal habitats constitute a major threat to sustained exploitation of demersal resources in this area. Moreover, the large by-catch of these trawl fisheries has caused a decline in catches in other fisheries. Whereas the total landings for India increased by a factor of 1.5 between 1969 and 1979, landings for the oceanic states increased by a factor of 3.2 (Lakshadweep) and 4.2 (Andaman & Nicobar) during the same period.

Lakshadweep Islands

Of the 36 islands that comprise Lakshadweep only 10 are inhabited by nearly 11 000 people. The largest contributions to annual fish landings of this region come from Agatti (1400 tons) and Suheli (800 tons), mostly because of their tuna pole and line fishery (Pillai *et al.*, 1986). A census conducted during 1987 determined that approximately 235 mechanised and 488 non-mechanised boats operated by a total of 3750 fishers (56% full-time, 39% occasional and 5% part-time) worked the waters surrounding Lakshadweep (Sivaraj *et al.*, 1992). Of the mechanised boats, 49% were used for pole and line fishing, 45% for troll line fishing and 6% for long line fishing. In Lakshadweep Islands, tunas that were caught with pole and line gear contributed between 50% and 65% to the total landings between 1969 and 1979, followed by sharks and rays (10%) which were mainly caught with long lines. Total landings of fish almost doubled between 1979 and 1988 from 3831 tons to 6809 tons (Sivaraj *et al.*, 1992) and the relative contribution of tunas increased to 86% between 1980 and 1988 due to progressive mechanisation of the fishery and an increased number of pole and line operations. Gears that are used inshore in lagoons include shore seines (11% of total landings), gill nets (3% of total landings), harpoons (2% of total landings) and cast nets (1% of total landings). Reef fishing with hand lines (5% of total landings) takes place mostly from June - September when the tuna fishing is poor. Nevertheless, catches of perch ranges second in importance (5%) for the period between 1980 and 1986. These catches are mostly locally consumed (Kumaran & Gopakumar, 1986). During the tuna season, some demersal species are caught for live bait for the tuna fishery, but their importance in the live bait catches is small (5% - 10%) as they are less preferred as bait than Spratelloides (45%)(Pillai *et al.*, 1986).

Andaman and Nicobar Islands

The entire area of the Andaman and Nicobar Islands was estimated to be 8249 km^2 (Anonymous, 1996a), of which 78% falls under the Andamans District. Andaman and Nicobar Islands are comprised of 348 islands situated in the Bay of Bengal and a channel of 150 km wide separates the two districts. Only 27 of the 324 islands of the Andaman district are inhabited and 13 of the 24 islands in the Nicobar district are inhabited (Alfred *et al.*, 1998). The chain of reefs along the west side of Andaman and Nicobar Islands are mostly barrier reefs and are richer than those along the east side where there are mostly fringing reefs (Anonymous, 1996b).

There are 45 fishing villages and 57 fish landing centres that fall under the Andaman and Nicobar administration (Anonymous, 1996b). The maximum sustainable yield of the region has been estimated at 161 000 tons of fish per year. Approximately11 000 people depend upon the fishery for their income, most live at South Andaman (38%) and Car Nicobar (19%) (Table 1). Around 2240 fishers have a licence. There are 140 mechanised boats, 102 motorised boats and 1568 non-motorised boats. Fish landings increased 12% between 1991 and 1995 from 22 674 tons to 25 477 tons with a total value of some 16.7 million US$ (Figure 4). Most fishers operate gillnets (1038 units of gear) and hook and line (900 units). Less important are cast nets (612 units), shore nets (45 units) and anchor nets (35 units). Seventy percent of the total production is sold fresh and 9% is frozen. Sardines are most important in the catch biomass (12%) followed by perches (7%).

At present, only a small proportion of the total fishing effort focuses on reefs. Most fishing occurs in the coastal shallows or in open water (driftnets). The most

Table 1 Number of fishers in Andaman and Nicobar islands in 1995. Source: Directorate of Fisheries Andaman and Nicobar islands.

Region	Fishers
South Andaman	4 237
Car Nicobar	2 111
Diglipur	901
Mayabunder	827
Hutbay	800
Rangat	792
Katchal	415
Havelock	330
Campbellbay	322
Billiground	190
Neil Island	77
Kamorta	60
Total	11 062

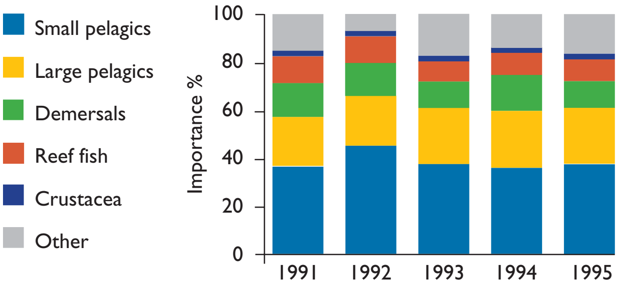

Figure 4. Trends in commercial catch composition for Andaman & Nicobar Islands.

valued fish in the commercial fishery is Spanish Mackerel (Scomberidae), which sells at 80 Rs. per kg (1 US$ = 42 Rs). Reef fish such as *Epinephelus* spp. are caught with hook and line but sold cheaply (~ 40 Rs. per kg). The majority of the fishers are involved in small-scale hook and line or gill net fishing and there are eight larger scale boats engaging mainly in bottom trawling. Average monthly incomes for fishers at Andaman Islands are low and fluctuate between 816-1225 RS and 7866-10200 Rs with expenditures ranging between 845 and 2800 Rs per month (Dam Roy & Dorairaj, 1998). The average catch per unit effort (CpUE) per day during the peak seasons varies between 8 kg and 75 kg. Women also participate in fishery related activities and their average incomes depend on the type of activity conducted. Those involved in the collection of clam shells earn three times as much as those involved in net making, fish drying or prawn peeling (Sivaraj *et al.*, 1992).

Although coral reef fisheries are not yet very important to Andaman and Nicobar Islands, they are being threatened by poaching of marine organisms by Burmese, Indonesians and Thais. This is a serious problem that is not easily controlled due to the large geographic extent of the islands. However, the most serious threats to the reef environment, as perceived by Andaman and Nicobar environmentalists, concern proposed developments for the near future. The airstrip is being extended to enable international planes to land, tourism is being developed and forest cut to accommodate a growing population of mainly mainland Indian people that are looking for jobs and cleaner environments.

Other man induced threats to the reefs include the level of siltation during the monsoon period and the level of pollution from agricultural chemicals. Although dive tourism is seen as a threat by most of the Andaman and Nicobar environmentalists, currently it occurs only during the calm weather season from October to February and very locally. There are only two dive schools that certify PADI open water divers at their home reef. At present, most diving on the reefs of Andaman and Nicobar is done from live aboard vessels. It is not clear how frequently these operations visit the reefs and, as yet, neither the local people nor local government benefits from this type of tourism expenditure. The Andaman and Nicobar government has been reluctant to allow any development especially when there is fear of negative effects to the environment. This basic conservation intent is strong yet the question is whether it is strong enough to keep outsiders out and maintain the near pristine condition of the reefs.

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Indian Ocean Islands – Summary

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CORDIO activities were conducted in seven of the Indian Ocean Islands namely, Comores, Madagascar, Mauritius, Mayotte, Réunion, Rodregues, and Seychelles. On each island biophysical and socio-economic assessments were conducted which have strengthened the annual monitoring of the reefs and collected baseline data for the socio-economic monitoring. In addition, a rapid assessment of the risk posed to island communities of ciguatera poisoning caused by potential increases in abundance of ciguatoxin producing dinoflagellates following bleaching was carried out. This study will eventually be expanded to encompass sites throughout the CORDIO region. Finally, as a record, a digital photographic database of the monitoring sites has been developed for the Island region. The results of these studies are summarised here and included in this document as separate more detailed reports.

COMOROS

Coral reefs surveyed in Comoros exhibited only around 10% coral mortality from the bleaching. Live coral cover remained high (36% - 40%) and dead coral comprised approximatley 47% of the substrate (Quod & Bigot, this volume). The reef supports both fisheries and tourism. The fisheries remains a small supplier of food to the country but provides employment for a large proportion of the population. In Comoros, reef-based tourism is an important component of a rapidly expanding tourism industry (Westmacott *et al.*, this volume).

MADAGASCAR

The coral reefs of Madagascar were not particularly affected by the increased sea temperatures recorded in 1998. At the survey site, live coral cover was over 40% and only 14% of the 55% dead coral cover was thought to be due to coral bleaching (Quod & Bigot, this volume). Overfishing is a critical issue affecting these reefs. Reef fisheries contribute 43% of Madagascar's total fish catch and are an important source of food and also foreign earnings (Westmacott *et al.*, this volume).

MAURITIUS

The reefs of Mauritius were not adversely affected by the bleaching event in 1998. The reefs are heavily utilised and are facing other threats at present. They support an extensive fishing industry with the lagoonal catch increasing. Most tourists who visit the island will be involved at some stage of the vacation with reef-related activities. (Westmacott *et al.*, this volume).

MAYOTTE

Most coral communities in Mayotte suffered from the bleaching. Live coral cover on the reef flats was low (4% - 6%) while on the deeper slopes it ranged between 20% and 28% (Quod & Bigot, this volume). As with corals in the lagoons of reef in the southern Seychelles (Teleki *et al.*, this volume), corals in the lagoon of Mayotte were less affected than those on adjoining reef flats and appeared to be adapted to fluctuations in water temperature. Tourism on Mayotte is a rapidly expanding industry and most tourists come from France and Réunion Island (Westmacott *et al.*, this volume)

RÉUNION, FRANCE

Réunion was not severely affected by the bleaching and those colonies affected are now showing signs of recovery. Live coral cover at the survey sites is reported to be in the region of 30% - 40% (Quod & Bigot, this volume). Tourism in Réunion contributes 8% to the national economy. However, only a small proportion of this can be directly related to the reefs. Demersal fish landings form the major part of the fisheries at 68% (Westmacott *et al.*, this volume).

RODREGUES

Little is known at present about the state of the reefs of Rodregues. The demersal fisheries provide little overall food for the island but are an important source of employment. Tourism at present is small scale with only two dive facilities on the island and few hotels. However, this is expanding rapidly (Westmacott *et al.*, this volume).

SEYCHELLES

The coral reefs of Seychelles were possibly the worst affected by the 1998 bleaching event. Live coral cover of the Seychelles granitic islands have been reduced to less than 10% on most reefs and signs of recovery are slight with low recruitment and 35% of the sites showing no recruitment at all (Turner *et al.*, this volume). This has led to the breakdown of the reef infrastructure and is likely to result in gradual erosion of the beaches. From the socio-economic point of view, these areas are the most heavily utilised by tourists. Although only 7% of the tourists visiting Seychelles dive, all participate in some sort of reef-based activity, whether it be snorkelling or only utilising the beaches. Fishing in Seychelles, is an important industry, the major investment being in the pelagic fisheries (Westmacott *et al.*, this volume). The lagoonal reefs in the outer islands appeared to have adapted to fluctuations in temperatures and faired relatively well, while the branching colonies of the shallow reefs were severely affected (Teleki *et al.*, this volume). Recovery from the bleaching in Aldabra is underway as many corals suffered only partial rather than total mortality (Teleki *et al.*, this volume)

At all the sites affected by the coral bleaching, the gradual breakdown of the reef is being seen. This will have a negative affect on the invertebrates and fish that utilise the reef structure for shelter. Careful monitoring of this breakdown and recovery as well as changes in the anthropogenic impacts will enable managers to make better informed decisions on how to cope with the consequences of the coral bleaching. The state of the reefs and the importance of reef fisheries and reef-based tourism can be seen in more detail in the reports indicated in this summary.

The reefs of the granitic islands of the Seychelles

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INTRODUCTION

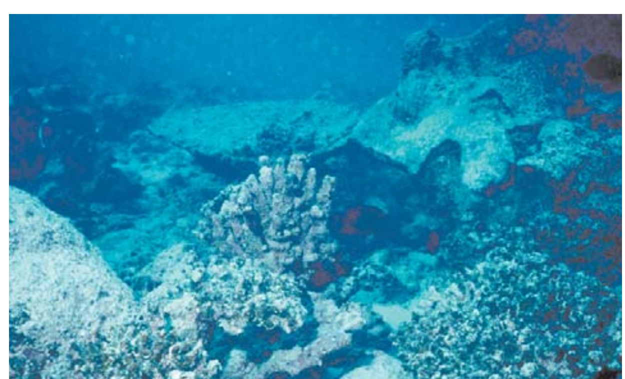
The status of coral reefs in the granitic islands of the Seychelles archipelago has been assessed by two independent surveys following the mass mortality caused by the 1997/98 bleaching event. Engelhardt (2000), working in collaboration with the Seychelles Department of Conservation surveyed 15 sites located mainly on the north west coast of Mahe during November and December 1999. During January 2000, Turner, Klaus, Hardman and West, working in collaboration with the Seychelles Marine Park Authority, surveyed 46 reef sites mainly to the east of Mahe, including Ste Anne, Ile Moyenne, Ile Cerf, Cousine, Praslin, Curieuse, La Digue, Grand Soeur and Felicite. Reefs around the granitic islands are shallow and rarely exceed 15 m depth. Both surveys aimed to assess reef structure over the full depth range, with corals identified to genus and species where possible, and assessed reef recovery by recording new colonies believed to have established since the bleaching event.

METHODS

Engelhardt surveyed 11 sites at a fine scale along the north west coast of Mahe, from Lighthouse to Baie Beau Vallon, with three additional sites at Conception Island, Moyenne Island, Beacon Island and Marianne Island to the east (Figure 3, open stars). Two 50 m x 5 m belt transects were employed at each site, at an oblique angle from a depth of 1-2 m to 15 m. Coral community structure was sampled in two 10 m x 1.2 m sub-transects by

Figure 1. Reef slope at Baie Ternay 1994.

Figure 2. High coral mortality on reef slope, Baie Ternay, 2000.

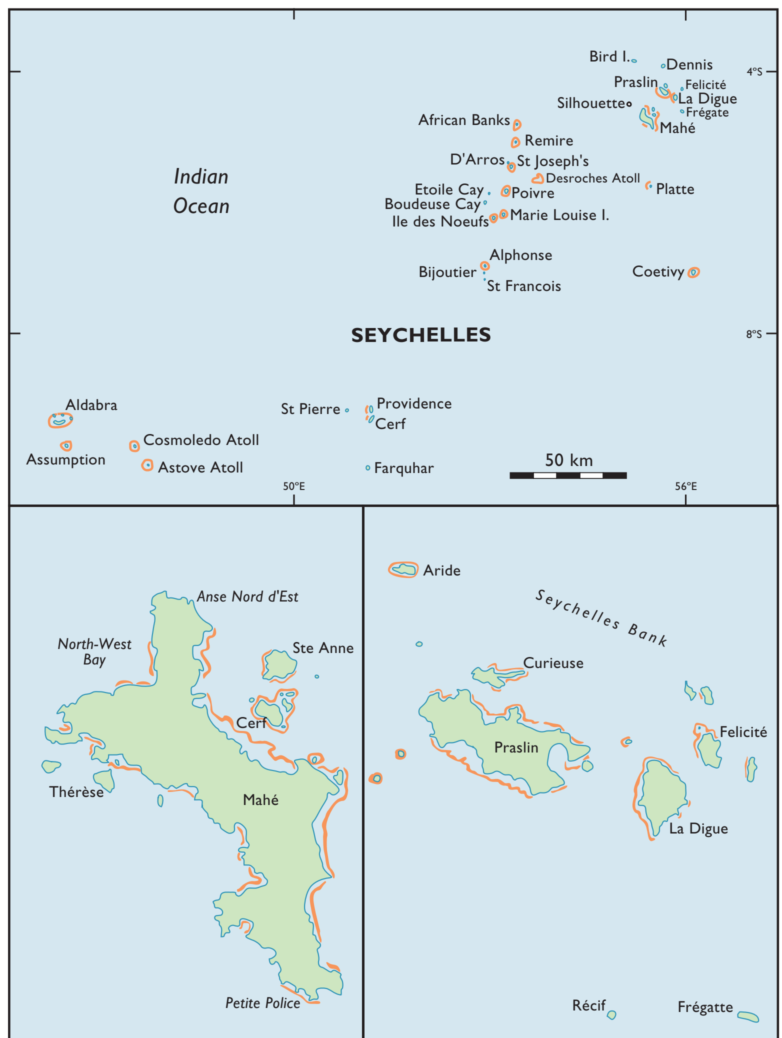


recording coral taxa in size classes 2-15 cm, 16-30 cm, 31-50 cm, 51-75 cm and >75 cm diameter. Coral cover was assigned to 10% range categories. Estimates of sea urchin densities within 250 m^2, level of sedimentation by a visual technique on disturbed sediment, and percentage cover of coralline algae were also recorded.

Turner, Klaus, Hardman and West used a rapid site assessment technique to assess reef composition and health over approximately one-hectare (100 m x 100 m) areas at 46 sites across the granitic islands (Figure 3, closed stars). It was not possible to survey northwestern exposed coasts during this period due to the monsoon. A two-tier approach was employed using a visual assessment of reef development and benthic cover combined with a taxonomic inventory. Sixty to 120 minutes were allowed for each survey, depending on whether snorkelling or S.C.U.B.A. was required. A team of four surveyors swam in a 'zig-zag' pattern across the area, each with a specific: visual recording, video recording, and photography.

The visual recorder described the state of reef development as (a) reefs with extensive flats (b) reefs with moderate flats (c) reefs with no flats but with carbonate accretion (incipient reefs) (d) coral communities developed on non-reefal rock, sand or rubble. A six point index was used for percentage cover and abundance and was applied to live cover, substrate type and abundance of organisms (0 = Absent 0% 0 individuals; 1 = Rare <1% cover or 1 individual; 2 = Occasional 1-10% or 2-10 individuals; 3 = Frequent 11-30% or 11-20 individuals; 4 = Common 31-50% or 21-50 individuals; 5 = Abundant 51-75% or 50-100 individuals; and 6 = Superabundant 76-100% or >100 individuals). Live cover was categorized as hard coral, dead standing coral, soft coral, turf algae, macro algae and other. Substratum categories used were hard substrate, continuous pavement, substratum in blocks >1 m, blocks <1 m, unconsolidated rubble, sand and silt. Organisms were identified to genus or species wherever possible. Coral colonies were classed by size into 1-10 cm, 11-25 cm, 26-50 cm, >50 cm classes, and the percentage of col-

Figure 3. Sites surveyed in Seychelles granitic islands, by Engelhardt in November December 1999 (open star), and Turner *et al.*, January 2000 (closed star).



onies damaged was recorded using the six-point scale. The number of coral recruits (colonies 1-10 cm in diameter or height) in the 100 m x 100 m area was recorded. Video records were made for archive and for cover analysis by random point counts, with frequent 360 degree scans to record reef development. Photography was used to confirm identification and record habitat structure.

PRELIMINARY RESULTS

Since the surveys have only recently been completed, preliminary results providing an overview are given, rather than site-specific information.

Living hard coral cover

Both surveys reported low percent cover of live hard corals. Engelhardt recorded 0% - 5% live coral cover at 80% of sites (n = 15) and a maximum of 0% - 10% at 20% of sites. Turner *et al.* recorded 1-10% living coral cover at 81% of the sites (n = 46) with a maximum of 31-50% at just two sites (Figure 4a). Many of the living corals displayed partial mortality. Dead standing coral was present at all sites with 100% cover at some (Figure 4b), and unconsolidated rubble was present at most, occasionally reaching 51% - 75% cover (Figure 4c). Live hard coral was mostly massive and sub-massive colonies of the coral genera *Porites, Acanthastrea, Goniopora, Diploastrea*, and *Physogyra,* usually occurring towards the bottom of the reef slopes. Branching and tabular *Acropora* and branching *Pocillopora* were either dead standing or reduced to rubble at most sites, especially along the upper reef slopes (Figure 5-6). The zoanthids

Figure 4. The proportion of the 46 reef sites surveyed in the Seychelles granitic islands during January 2000, which exhibited (a) percent cover classes of living hard coral, (b) dead standing coral and (c) unconsolidated rubble.

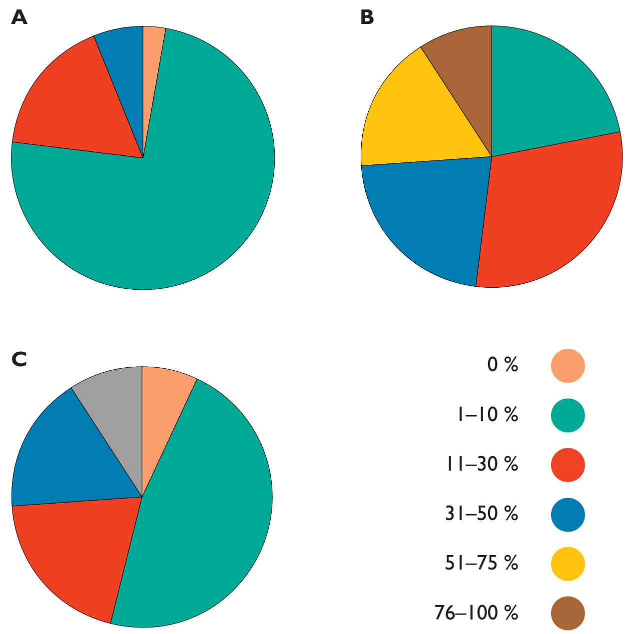


Figure 5. Reef reduced to dead standing coral and rubble at Cousine.

Figure 6. Unconsolidated branching *Acropora* rubble at St Pierre, Curieuse.

Zoanthus, *Palythoa* and *Protopalythoa*, and corallimorpharian anemones grew on dead standing coral and consolidated rubble at the top of reef slopes and in lagoons, together with red and green calcareous algae. The soft corals *Sarcophyton, Lobophyton* and *Sinularia* grew on hard substrates on the lower reef slopes.

Coral diversity

Species diversity of hard corals was low at all reef sites surveyed across the granitic islands. Engelhardt recorded a maximum of 10 families and 15 genera of hard corals (at Lighthouse), while Turner *et al.*, recorded a maximum of 13 families, 30 genera, and 49 species (at Port Launay Marine National Park) and a median of 5 families, 8 genera and 10 species per site. Other marine National Parks faired less well, although Turner *et al.*, found 9 families, 21 genera and 31 species at Coral Gardens Curieuse, 10 families, 15 genera and 28 species at Baie Ternay, and just 2 families 2 genera and 2 species living at Ile Coco Marine National Park. Some species were represented by just one colony in all surveys. Corals species recorded as dead standing, surviving, and recruiting are summarised in Figure 7.

Figure 7: Coral families, genus and species recorded as dead standing, surviving and recruiting recorded during January 2000 survey of the granitic island reefs of the Seychelles.

SCLERACTINAN CORALS

FAMILY	GENUS	SPECIES	DEAD STANDING CORAL	SURVIVING & REMNANT SPECIES	RECRUITS & JUVENILES
POCILLOPORIDAE	*POCILLOPORA*	*Pocillopora damicornis*	✓	✓	✓
		Pocillopora eydouxi	✓	✓	
		Pocillopora meandrina	✓	✓	
		Pocillopora verrucosa	✓	✓	✓
	STYLOPHORA	*Stylophora pistillata*		✓	✓
ACROPORIDAE	*MONTIPORA*	*Montipora aequituberculata*		✓	
		Montipora danae		✓	
		Montipora efflorescens		✓	
		Montipora informis		✓	
		Montipora tuberculosa		✓	
		Montipora venosa		✓	
		Montipora verrucosa		✓	
	ACROPORA	*Acropora sp.*	✓	✓	✓
		Acropora cf. cerealis			✓
		Acropora clathrata	✓		
		Acropora cytherea	✓		
		Acropora cf. danai	✓	✓	✓
		Acropora cf. digitifera			✓
		Acropora cf. divericata			✓
		Acropora cf. formosa	✓	✓	
		Acropora cf. humilis			✓

FAMILY	GENUS	SPECIES	DEAD STANDING	SURVIVING & REMNANT	RECRUITS & JUVENILES CORAL
		Acropora cf. hyacinthus			✓
		Acropora cf. longicyathus			✓
		Acropora cf. nasuta			✓
		Acropora cf. valida			✓
	ASTREOPORA	*Astreopora gracilis*		✓	
		Astreopora myriophthalma		✓	
PORITIDAE	*PORITES*	*Porites sp.*	✓	✓	✓
		Porites australensis		✓	
		Porites cylindrica		✓	
		Porites lobata		✓	
		Porites lutea		✓	
		Porites solida		✓	
		Porites rus		✓	
	GONIOPORA	*Goniopora sp.*	✓	✓	✓
		Goniopora columna		✓	
		Goniopora lobata		✓	
		Goniopora minor		✓	
		Goniopora savignyi		✓	
		Goniopora stokesi		✓	
	ALVEOPORA	*Alveopora allingi*		✓	
		Alveopora fenestrata		✓	
		Alveopora verilliana		✓	
SIDERASTREIDAE	*SIDERSATREA*	*Siderastrea savignyana*		✓	✓
	PSAMMOCORA	*Psammocora contigua*		✓	✓
		Psammocora digitata		✓	
AGARICIIDAE	*PAVONA*	*Pavona cactus*		✓	
		Pavona clavus		✓	
		Pavona explanulata		✓	
		Pavona maldivensis		✓	
		Pavona varians		✓	✓
		Pavona venosa		✓	✓
	LEPTOSERIS	*Leptoseris fragilis*		✓	
		Leptoseris explanata		✓	
		Leptoseris mycetoseroides		✓	

FAMILY	GENUS	SPECIES	DEAD STANDING	SURVIVING & REMNANT	RECRUITS & JUVENILES CORAL
	GARDINEROSERIS	*Gardineroseris planulata*		✓	✓
FUNGIIDAE	*CYCLOSERIS*	*Cycloseris sp.*	✓	✓	✓
	DIASERIS	*Diaseris sp.*	✓	✓	✓
	FUNGIA	*Fungia sp.*	✓	✓	✓
		Fungia fungites	✓	✓	✓
	HERPOLITHA	*Herpolitha limax*	✓	✓	
	HALOMITRA	*Halomitra pileus*	✓	✓	
OCULINIDAE	*GALAXEA*	*Galaxea astreata*	✓	✓	
		Galaxea fasicularis	✓	✓	
PECTINIIDAE	*ECHINOPHYLLIA*	*Echinophyllia aspera*		✓	
	OXYPORA	*Oxypora lacera*		✓	
MUSSIDAE	*ACANTHASTREA*	*Acanthastrea echinata*	✓	✓	
	LOBOPHYLLIA	*Lobophyllia corymbosa*	✓	✓	
		Lobophyllia hemprichii	✓	✓	
	SYMPHYLLIA	*Symphyllia recta*		✓	
MERULINIDAE	*HYDNOPHORA*	*Hydnophora microconos*	✓	✓	
	MERULINA	*Merulina ampliata*		✓	
FAVIIDAE	*FAVIA*	*Favia sp.*	✓	✓	
		Favia favus		✓	
		Favia pallida		✓	
		Favia stelligera		✓	
	FAVITES	*Favites sp.*	✓	✓	✓
		Favites abdita		✓	
		Favites complanata		✓	
		Favites flexuosa		✓	
		Favites halicora		✓	
		Favites pentagona		✓	
	GONIASTREA	*Goniastrea australensis*		✓	
		Goniastrea aspera		✓	
		Goniastrea edwardsi		✓	
		Goniastrea pectinata		✓	
		Goniastrea retiformis	✓	✓	
	PLATYGYRA	*Platygyra daedalea*	✓	✓	

FAMILY	GENUS	SPECIES	DEAD STANDING	SURVIVING & REMNANT	RECRUITS & JUVENILES CORAL
	LEPTORIA	*Leptoria phrygia*		✓	
	OUOPHYLLIA	*Oulophyllia crispa*		✓	
	MONTASTREA	*Montastrea sp.*		✓	
		Montastrea annuligera		✓	
		Montastrea magnistellata		✓	
		Montastrea cf. curta		✓	
		Montastrea cf. valenciencis		✓	
	PLESIASTREA	*Plesiastrea versipora*		✓	
	DIPLOASTREA	*Diploastrea heliopora*	✓	✓	
	LEPASTREA	*Leptastrea inaequalis*		✓	
		Leptastrea purpurea		✓	
	CYPHASTREA	*Cyphastrea chalcidicum*		✓	
		Cyphastrea microphthalma		✓	
		Cyphastrea serialia		✓	
	ECHINOPORA	*Echinopora gemmacea*		✓	
		Echinopora cf. hirsutisima (?)		✓	
		Echinopora lamellosa	✓	✓	✓
CARYOPHILLIDAE	*PHYSOGYRA*	*Physogyra lichtensteini*		✓	
DENDROPHYLLIDAE	*TURBINARIA*	*Turbinaria patula*		✓	
	DENDROPHYLLIA	*Dendrophyllia sp*		✓	
	TUBASTREA	*Tubastrea sp*		✓	
NON-SCLERACTINAN CORALS					
MILLEPORIDAE	*MILLEPORA*	*Millepora sp.*		✓	
HELIOPORIDAE	*HELIOPORA*	*Heliopora coerulea*		✓	
TUBIPORIDAE	*TUBIPORA*	*Tubipora musica*		✓	
STYLASTERIDAE	*DISTICHOPORA*	*Distichpora sp.*		✓	

42 GENERA OF SCLERACTINIAN CORALS (FROM A PREVIOUSLY REPORTED 55 GENERA)

109 SPECIES (PRELIMINARY LIST) PREVIOUSLY REPORTED 143 (Land 1994)

4 GENERA NON-SCLERACTINIAN CORALS

Coral recruitment

Engelhardt and Turner *et al.*, observed variable levels of coral recruitment. Based on mean densities of size classes of corals, Engelhardt reported more small (2 cm - 15 cm) corals than larger corals at all sites. Highest densities of over 200 small corals per 24 m^2 area were recorded at two sites (Conception Island and Marianne Island), and low densities were recorded on the central reefs of the north west coast of Mahe (e.g. 6 small corals per 24 m^2). Branching *Acropora* and *Pocillopora* recruits showed a more patchy distribution with none at 46% (7 of 15) of sites, 175 per 24 m^2 at Marianne Island and an average of <20 per 24 m^2 at other sites. Turner *et al.*, recorded no branching *Acropora* or *Pocillopora* recruits (classed as being 1-10 cm in size) at 35% (16 of 46) sites, and only 17% (8) of sites had >20 recruits per 100 m x 100 m area. Those sites with highest recruitment were the more sheltered sites, while exposed sites showed low or no recruitment (Figure 8). Recruits were observed settled on limestone pavement, dead standing coral and rubble. Many recruits exhibited damage and scars from abrasion, breakage, fish and urchin grazing (Figures 9-11).

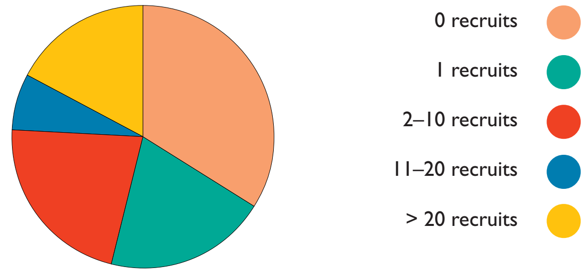


Figure 8. The proportion of sites with recruits of branching *Acropora* and *Pocillopora* (1-10cms colony size) in 100 m x 100 m areas, recorded during January 2000.

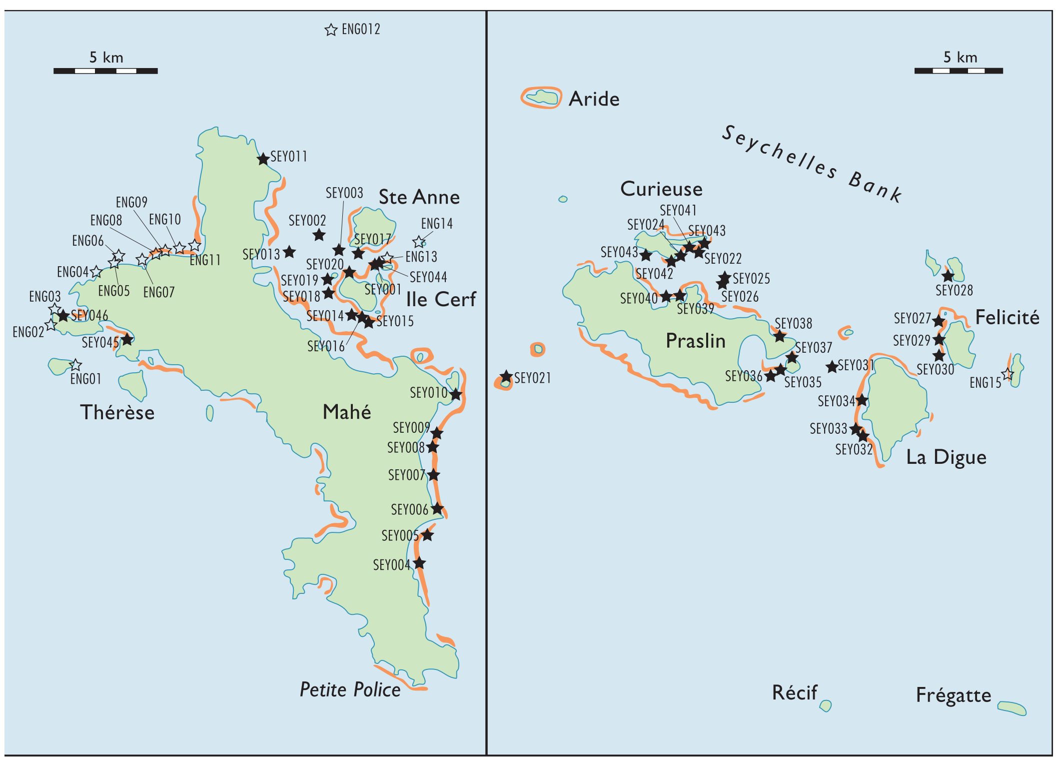


Figure 9. Recruitment of branching corals (*Acropora* and *Pocillopora sp.*) at 46 sites around the Seychelles granitic islands recorded during January 2000.

Figure 10. Recruit on consolidated substrate.

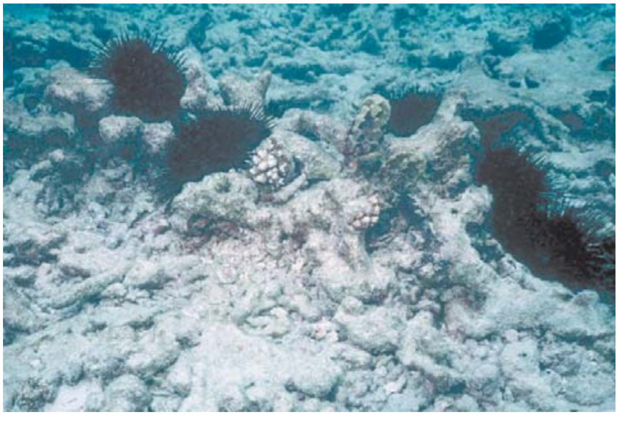


Figure 11. Recruit vulnerable to predation by urchins.

Figure12. Recruit on unconsolidated substrate and vulnerable to abrasion.

DISCUSSION

The coral reefs of the Seychelles granitic islands have suffered severe degradation during the two-year period following the 1997/98 mass coral bleaching event, and signs of recovery are slight. Live coral cover has been reduced to less than 10% on most reefs around the inner islands, and partial mortality of colonies is high. Although hard coral cover was rarely 100% before mass mortality, this still represents a major reduction from the 35-80% hard coral cover typical of reef slopes e.g. Baie Ternay, (TMRU, 1996) and 25-40% hard coral cover typical of patch reefs e.g. Coral Garden, Curieuse (TMRU, 1997). In common with many other reefs throughout the Indian Ocean region, the massive and sub-massive corals survived the mass mortality, particularly *Porites*, *Goniopora*, *Acanthastrea* and *Diploastrea*. Branching and tabular *Acropora* and branching *Pocillopora* were mostly dead and are now either standing or reduced to rubble. However, care is required in assessing rubble, since broken *Acropora* is a common component of these reefs (Braithwaite, 1971; Rosen, 1971). There is no indication that reefs further away from the main islands of Mahe and Praslin faired any better, and conversely more remnant corals were recorded from lower reef slopes in turbid water, especially around the main island of Mahe (such as Beau Vallon Bay and adjacent to the harbour on Mahe). Dead standing coral is brittle and is reduced to rubble at exposed sites where it remains largely unconsolidated, becoming mobile in storm waves causing abrasion. Calcareous algae, zoanthids and corallimorpharians are binding rubble in shallow waters in sheltered sites.

A total of 109 species and 42 genera of scleractinian corals were recorded from granitic island sites during the January 2000 survey (although some species were represented by only one colony), compared to 143 species in 55 genera listed for the whole Seychelles archipelago (granitic and coralline islands combined, including deep sites) by Sheppard (1998). The 13 genera not found during the January 2000 survey were *Heteropsammia, Trachiphyllia, Euphyllia, Catalaphyllia, Seriatopora, Ana-*

cropora, Stylaraea, Anomastrea, Cosinarea, Pachyseris, Mycedium, Cynarina, and *Caulastrea*. The coral diversity for the granitics is comparable with those of surveys conducted prior to the mass mortality event. TMRU (1996) recorded 31 genera and 69 species from the Baie Ternay area of Mahe, and 36 genera and 84 species from Curieuse (TMRU, 1997). The Dutch Tyro expedition recorded 37 genera and 109 species from the Seychelles Plateau around Mahe (van der Land, 1994). Thus, the preliminary results presented here indicate that most species of corals have survived somewhere in the region, but that diversity at most sites is low (median 8 genera, 10 species). This finding has important implications on the probability of future recruitment from within the region.

To date, two years after the mass mortality event, recruitment to the degraded reefs is patchy and low, with 35% of the sites surveyed showing no recruitment. In particular, recruitment is low for fast growing *Acropora* and *Pocillopora* (<20 recruits per hectare at most sites) that used to dominate the reefs. Recruitment that has occurred is greatest on more sheltered reefs in bays and may be related to suitable consolidated substrate. Recruits are vulnerable to fish and urchin predation and to breakage, abrasion and removal during storms. There is an urgent requirement to monitor recruitment and survival and to protect surviving species that may be reproducing sexually and providing new sources of coral larvae.

The Seychelles granitic island reefs appear to have experienced one of the greater mass mortalities of corals in the Indian Ocean following the 1997/98 bleaching event, probably because they are mostly shallow (<15 m depth). Reef structure is breaking down, and this may expose island shores to erosion during storms, as has occurred at La Digue. The Marine National Parks of the Seychelles, once the most diverse reefs, have been badly degraded and will require even greater protection if they are to recover. Fishing gear, boat anchors and especially sedimentation and siltation from coastal development activities continue to place many of these recovering reefs at risk. There may need to be a reassessment of the management of Marine National Parks to afford greater protection to larger areas.

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Reef systems of the islands of the southern Seychelles

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INTRODUCTION

There are 74 coralline islands in the Southern Seychelles, from which representative reefs have been selected for this report, comprising an atoll (Alphonse), a raised platform island (St Pierre), a carbonate bank (Providence-Cerf) and a drowned atoll (Cöetivy) (Figure 1). Aldabra Atoll is treated in a separate section in this publication (see Teleki *et al.*, this volume).

Qualitative observations of reef morphology, coral community composition and reef health in the southern Seychelles were made between March and May 1998 (Southern Seychelles Atoll Research Programme - SSARP), February and March 1999 (Thalassi/Shoals of Capricorn Expedition) and November 1999 (Aldabra Marine Programme – AMP). These observations were supplemented by quantitative descriptions of coral communities at 48 sites at four study locations. Twenty-five meter long transects were set out at water depths, where possible, of 5 m, 10 m, 15 m and 20 m. Transects were surveyed using both a line point intercept method and digital videographic imagery which was subsequently analysed using point sampling to generate estimates of benthic cover. Digital stills of individual coral species were obtained from each site for taxonomic inventory purposes.

ALPHONSE ATOLL

Alphonse Atoll (9°0' S, 52°45' E), situated at the southern end of the Amirantes Ridge, is a small (24 km^2) atoll located 415 km south of Mahé (Figure 1). The peripheral reefs enclose a simple dish-like lagoon, reaching depths of approximately 10 m at its centre. Alphonse has recently undergone extensive development to establish a hotel complex comprising 24 guest chalets and a number of staff quarters. Previously the island supported a small number of Seychellois (<10) involved with the production of copra from the coconut trees which until recently formed the dominant land cover.

The fore-reef is characterised by a rocky pavement, with low relief spur-and groove topography immediately seaward of the reef flat margin. The fore-reef slope ranges in width between 50 m and 150 m and extends from a depth of 5 m to 15 m. Below this a steep slope or drop-off with exposed rock surfaces and accumulations of coral rubble extends below 15 m to 20 m. Coral cover rises from less than 30% in shallow water to over 40% at intermediate depths before declining again at 20 m water depth. The reef community was dominated by *Pocillopora* spp., *Acropora* spp. and *Stylophora* spp. with widely spaced large colonies of *Pavona* spp. and *Porites* spp. In some areas of the outer reef evidence of monospecific stands of the blue octocoral *Heliopora coerulea* were present. Of the 39% coral cover on the outer reef slopes at Alphonse Atoll, 74% was recorded as being bleached or recently dead (Figure 2).

Bleaching and mortality patterns were observed to be different according to depth, reef zone and exposure. In March, 65% of corals at 10 m depth were bleached or recently dead whereas the equivalent figure at 15 m was 73% (Figure 3). In the subsequent five weeks prior to the second repeat survey, the percentage of normal corals at 10 m declined by over one third, to 22%, and the

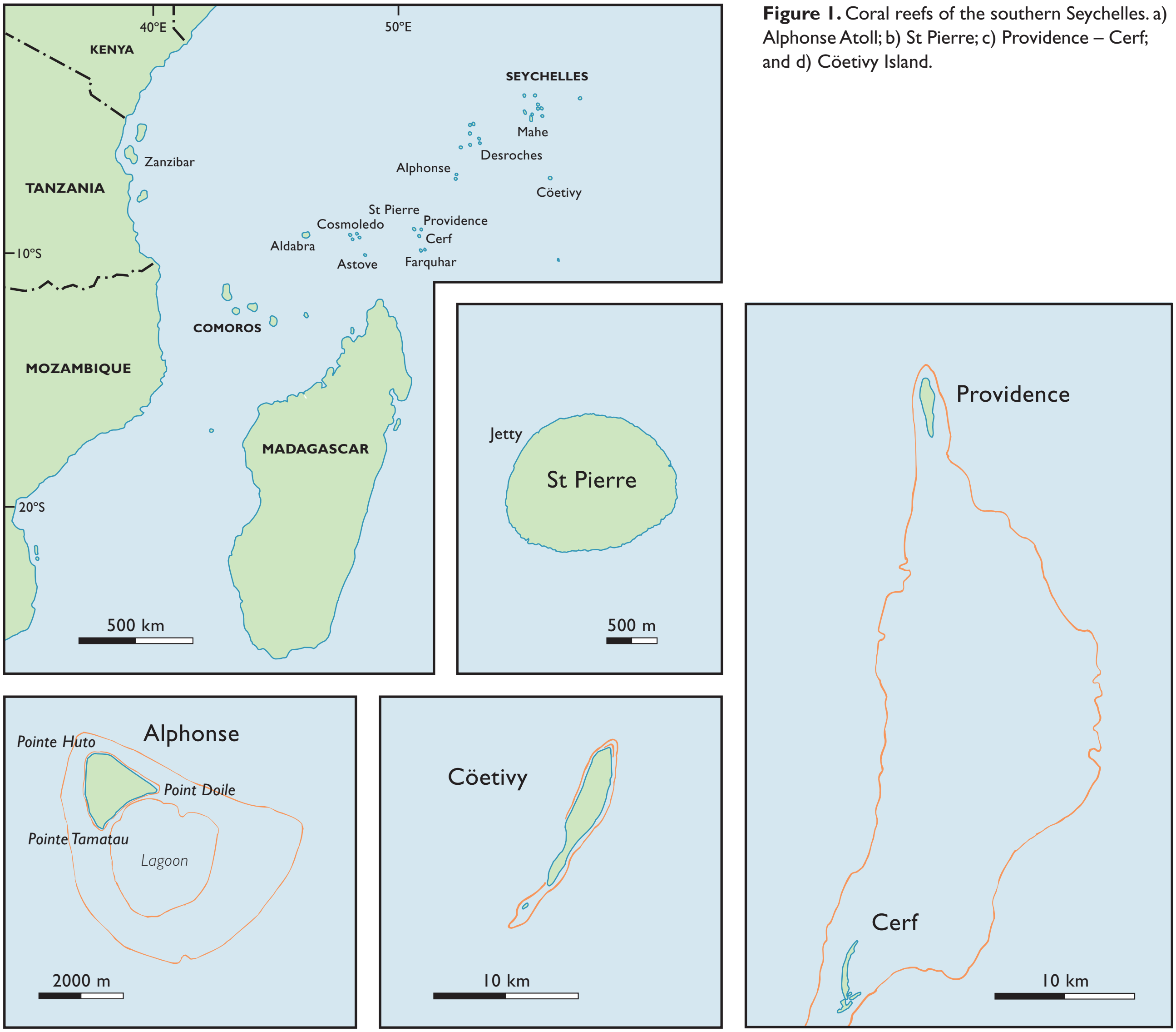


Figure 1. Coral reefs of the southern Seychelles. a) Alphonse Atoll; b) St Pierre; c) Providence – Cerf; and d) Cöetivy Island.

percentage of dead corals almost doubled, to 64%. By contrast, at 15 m water depth, the bleaching episode appeared resolved by May, leaving either dead corals or corals which appeared to have recovered from bleaching (Figure 3). In the long term, the overall impact of this bleaching episode is likely to be greater in shallow water, with a proportion of the 14% of bleached corals at 10 m in May 1998 being added subsequently to the stock of dead coral substrate.

Many of the corals in the lagoon would be accustomed to regular inundation from abnormally high sea surface temperatures and less likely to have suffered a bleaching event. Localised upwelling along the eastern (windward) side of the Atoll was observed in 1999 and

Figure 2. Coral condition by location.

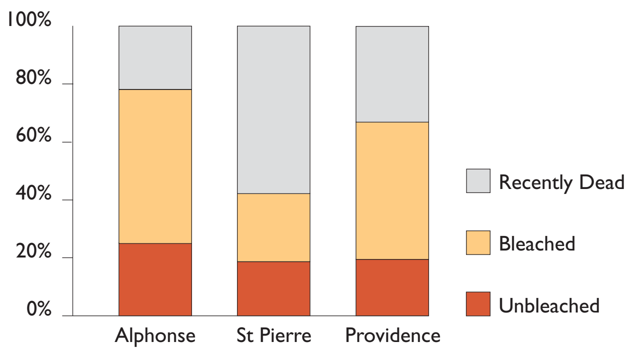
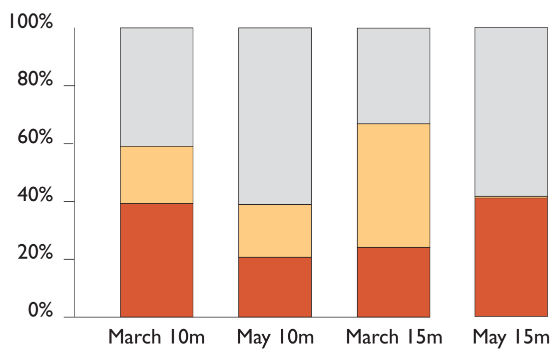


Figure 3. Coral condition over time at Alphonse Atoll at a) 10 m and b) 15 m water depth for March and May 1998 survey periods.



appeared to be a common occurrence creating a temperature differentiation of approximately 3° C - 6° C. This phenomenon could have mitigated the impact of the anomalous 1998 sea-surface temperatures which caused the widespread coral bleaching.

A year after the bleaching event (February - March 1999) the shallow (8-15m) outer slope, was dominated by dead coral and coral rubble composed of substantial quantities of dead branching coral. The dead coral surfaces on the NW and SW slopes were dominated by the reticulate, foliaceous, green alga *Microdictyon*. Observations in November 1999 showed that this cover has persisted although the alga has been substantially grazed. Pocilloporid and acroporid recruits (3-5 cm diameter) were observed in November 1999 but these were very few. The largest cover percentage of live coral was found in the NE and SE spur and groove zone (<5 m depth) where extensive monospecific stands of *Porites nigrescens* were dominant.

St. Pierre

St Pierre (9°19' S, 50°43' E) is a small (2 km^2) raised reef island, 250 km south of Alphonse Atoll (Figure 1). It consists of Late Quaternary reef limestone, approximately 5 m above present sea level. St. Pierre was previously extensively quarried for island phosphates (Gardiner, 1926-36; Stoddart, 1967), but is currently uninhabited.

Coral communities on the western/northwestern, leeward side of the island are found in water depths of less than 20 m and within 100 m - 200 m of the shoreline; offshore of this point, substrate angle increases rapidly, forming a steep slope to 90m in places. Before the bleaching event, leeward reefs were characterised by a typical coral cover of ca. 60% in which *Pocillopora* spp., *Acropora* spp. and *Millepora* spp. were dominant. By contrast, the south-eastern, windward coast has a low angle slope which only steepens at 20 m to 30 m water depth, 1 km - 2 km offshore. Coral cover was lower (50%) and dominated by *Millepora* spp. (particularly *Millepora tenella*) with low percentage cover of *Acropora* spp. and *Pocillopora* spp. Substrate cover by coral rubble (23%), bare rock (19%) and *Halimeda* sands (12%) was correspondingly greater than on leeward coasts.

At a within-island scale, there is some evidence to suggest that levels of bleaching in 1998 were mediated by aspect. From the data assembled for St. Pierre at 15 m water depth, bleaching impacts were lower (77% bleached or recently dead) on windward coasts than on leeward coasts (87% bleached or recently dead). This may be due to greater water movement on more exposed reefs, perhaps operating through localised wind-driven upwelling of deeper, cooler waters along windward island margins. A similar pattern to that of Alphonse was seen at St. Pierre where coral colonies found in the 10 m - 20 m depth range appeared to be in a more advanced stage of bleaching-related mortality than those in the shallower 3 m to 10 m depth range. In addition, at windward sites, many of the smaller massive (e.g. *Favia pallida*) and encrusting coral colonies displayed no evidence of bleaching in spite of the high general levels of incidence at these locations (Figure 4).

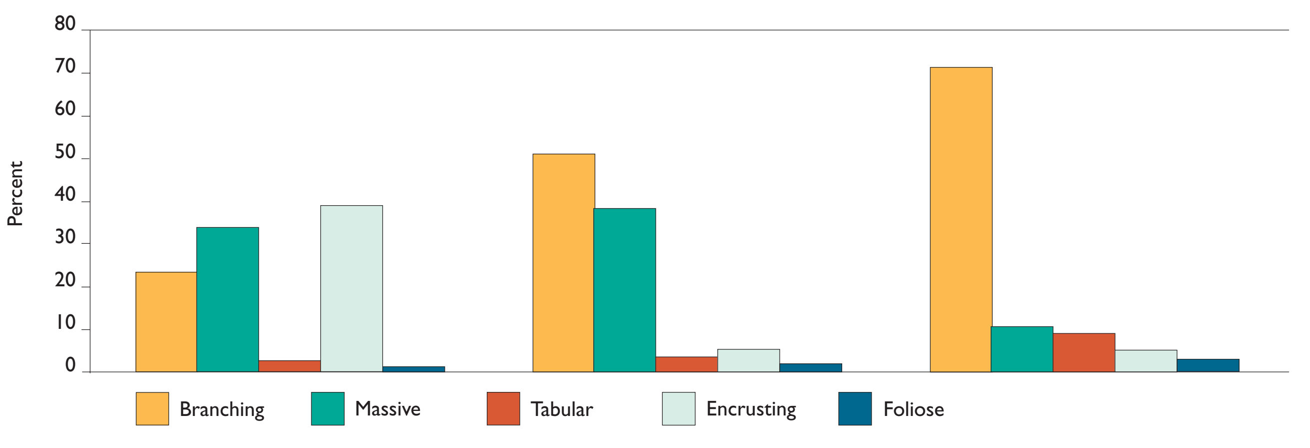
In February 1999 the shallow slopes (5 m - 15 m) of St. Pierre on the leeward side (NW) with a low gradient (10º) recorded up to 95% mortality in branching corals (*Acropora* spp. and *Pocillopora* spp.). Mortality was less on the windward side where coral rubble, probably a result of high levels of hydrodynamic activity, was common. Massive, sub-massive and encrusting species displayed partial and sporadic mortality in all sites surveyed here. The deep slope substrate (20 m - 25 m) in the N and NE, was mainly composed of sand and rubble, but 1 m tall *Tubastrea micrantha* trees grow every 1 m - 2 m. Also at this depth were live and recovered colonies of *Pachyseris*, *Pectinia*, *Lobophyllia*, *Pocillopora, Millepora*, *Acropora* and *Porites*.

Coral recruits (<2 cm diameter) were evident in February 1999 on both the windward and leeward coasts, sitting within and on the dead, but intact, branching coral matrix. This matrix, in the intervening year following the bleaching event, has been rapidly colonised by fleshy macroalgae, encrusting coralline algae and other invertebrate species such as bryozoans, tunicates, hydroids and zooanthids. The structural integrity and three-dimensional complexity of the reef has been maintained. However, it is vulnerable to weather induced physical forces. If physical degradation processes do give rise to unconsolidated rubble fields, then coral larval settlement, growth and survival, and recovery of the reef system will be impaired.

Providence-Cerf

The Providence - Cerf (P-C) Bank lies 40 km to the east of St. Pierre and occupies an area of 300 km^2 (Figure 1). The Bank is 40 km long and varies between 1 km and 10 km wide and is orientated in a N-S direction. The surface of the Bank is characterised by extensive seagrass beds, subtidal sand channels and, in places, tidal flats and sandbanks which dry at low tide. The land area of the bank covers 2.3 km^2 (Stoddart, 1984). The island of Providence (9°15' S, 51° E) occupies the northern extremity of the bank and supports small-scale fishing operations and copra production, while a series of smaller sand cays form the uninhabited Cerf islands at its southern margin.

Figure 4. Susceptibility to bleaching by different coral growth forms: summary of all survey sites in the Southern Seychelles.



At the Bank's northern and southern limits, cliffed and bio-eroded bedrock surfaces with minimal living coral cover are characteristic in water depths between 5 m and 25 m. At intervening sites along the western side of the Bank more gently shelving sub-tidal slopes are typical. Sand, bedrock and rubble substrates account for over 60% of the bottom type on the Bank and Bank margins. A further 20% of substrates are characterised by dense beds of *Thalassodendron ciliatum* (Selin *et al.*, 1992) within which small colonies of *Stylophora pistillata* are common. Coral communities are localised and account, on average, for only 6% of substrate cover. On both the Bank margin and within the deeper channels on the Bank itself, species of the coral genera *Pocillopora*, *Acropora, Porites* and *Favia* are present. In 1998, 81% of the coral cover was recorded as being bleached or recently dead (Figure 2). Soft corals accounted for approximately 2% of substrate cover, of which 76% showed tissue necrosis. Soft coral communities that had already died were marked by large bare patches which remained following their disintegration.

Coëtivy

Cöetivy is a narrow, elongate island (9 km long and 0.25-1.6 km wide) located 300 km southeast of Mahé and 50 km south of the Seychelles Bank, making it the eastern most island in the Seychelles territory and its EEZ (Figure 1). Anthropogenic presence on Cöetivy has increased following the construction of a prawn farm in 1992-93, though impact is localised to the vicinity of its outfall pipes. In 1996, a small-scale fin-fishery was initiated to meet the needs of the inhabitants and to export fish products to Mahé. It is not known whether present fish populations and the degraded reef can support current levels of fishing.

The coral communities of Cöetivy once displayed a high degree of structural and species variability on each side of the island, forming a fringing reef 100 m - 500 m offshore. The southern extent of the reef continues to approximately 3.5 km beyond the end of the island producing a wide shallow (2 m - 4 m) body of water where the reef is dominated by a few remaining live colonies of *Heliopora* and dead *Acropora* spp. (primarily of a tabular morphology). The substrate varied from one dominated by sand/rubble to what was once 80-100% coral coverage.

In February 1999, coral mortality was close to 95% at all sites on the western (leeward) side of the island. Large, and once luxuriant, monospecific stands of branching and tabular *Acropora* spp. are now dead, with *Platygyra* spp. the dominant live coral. Most of the coral matrix was still structurally intact and even the most heavily encrusted and overgrown table corals showed a low incidence of microbioerosion. Massive corals had patches of live tissue, though these were being overgrown by adjacent algal communities.

In February 1999, coral recruits of 5 cm - 10 cm diameter were rare on the dead coral structure. Availability of suitable substrates for coral settlement and growth is limited as a result of high surface coverage of macroalgae.

DISCUSSION

The coral bleaching event was extensive on all the southern Seychelles reefs in 1998. Bleaching was generally worse in shallower waters (≤10m). Mortality was particularly high in the branching corals *Acropora*, *Pocillopora*, *Millepora* (fire coral) and *Heliopora* (blue coral), with live *Millepora* being rare at all locations. Death in the massive corals such as *Porites*, *Favia*, *Pavona, Platygyra* and *Diploastrea* was in most cases partial and spatially patchy at both the colony and reef scale. Areas which were least impacted were those influenced by cooler currents, such as in the upwellings on the windward side of Alphonse and St. Pierre, and in lagoon channels where water fluxes are high.

Corals subjected to frequent high temperatures, such as in lagoons, also faired well. Based on previous studies undertaken elsewhere (e.g. Cook *et al.*, 1990; Hoeksma, 1991; Sheppard, 1999), it can be hypothesised that corals in shallow (3 m – 10 m) waters were more tolerant to temperature fluctuations than were corals in deeper water (10 m – 20 m) which usually experience a more constant temperature regime. On the two visits to Alphonse Atoll between 29 March and 4 May 1998, corals in deeper water bleached first, followed later by shallow water coral colonies. Many of the corals survived in Alphonse lagoon, suggesting an adaptation of corals in the lagoon to periodic exposure by high sea-surface temperatures.

Bleaching was not exclusive to hermatypic corals. Incidences of bleaching and mortality were widespread in alcyonaceans, non-scleractinian coelenterates (*Stichodactyla* and *Heteractis*) and bivalves (*Tridacna*). Some alcyonaceans, which were also completely bleached (*Lobophytum* and *Sinularia*), had evidence of recent mortality with subsequent necrosis and disintegration of their growth form in an advanced state. Recovery did not occur in the soft corals and anemones with a conspicuous absence at all reef sites visited in 1999 following the 1998 bleaching event.

Consistent in all locations was the decimation of acroporid populations, with the exception of individual colonies at St. Pierre and Alphonse. 'Refuge' areas of local live coral species may be important to reef recovery. The sites surveyed are isolated oceanic reefs where alternative sources of larvae are very distant and local opportunities for recruitment limited. Whether the long-term reproductive success of a limited number of live colonies is sufficient to encourage significant reef growth is unknown.

At two sites (Cöetivy and St. Pierre) coral recruits were observed on dead coral structures and coralline algal encrusted substrates. The source of these recruits is currently unknown. However, areas in which the coral have faired well such as the Alphonse Channel, deep reef slopes and areas of localised upwelling on all reef sites may play a key role in seeding the dead reef.

Signs of decay within the reef architecture were apparent in all sites with breakage of branching corals and high incidences of corals no longer in life form position (i.e. branching *Pocillopora* and *Acropora*, tabular *Acropora*, also noticeably *Heliopora* and *Millepora*). Those dead corals that were intact had considerable evidence of microerosive activity, especially from clionid sponges, which have a severe weakening effect on coral skeletons. The relative rates of microerosion versus cementation by coralline encrusting algae will greatly affect provision of substrate upon which other organisms, such as corals, can settle and grow. Excessive erosion and formation of coral rubble can also be damaging to marine life through direct physical contact and abrasion.

The architectural complexity of a reef is one of the important features providing shelter and niches for invertebrates and fish, and can be affected on a small scale by the growth of epiphytes, and to a lesser degree epifauna. Spaces between dead coral branches were observed to be filled in by algal turf and calcareous crusts or covered over by algae such as *Lobophora*. At Cöetivy, the number of commensal crabs were found to be very low amongst dead *Pocillopora* branches that were overgrown by *Lobophora*, whereas in locations with live colonies there was an obviously higher abundance of invertebrates (crabs, shrimps) and fish (*Dascyllus* spp.). The continued decay of the reef will impact the invertebrate and fish life as the post disturbance reef communities evolve.

The vertical relief and three dimensional complexity of the reef habitat provided by both live coral and erect dead coral structures is not only crucial for fish survival, but is also as an aggregation attractant for reef fishes. Degradation and reorganisation of the reef structure following a bleaching related mortality might, therefore, be expected to have effects on both reef dwelling and non-reef dwelling fish communities. However, detailed quantitative descriptions of reef fish communities suggest that in 1998 (SSARP) and 1999 (Thalassi/Shoals and AMP) there was no major impact of the bleaching and related coral mortality on fish communities. Fish communities may be more robust than the coral communities to major disturbances but there is often a lag in the response of reef fishes to the loss of live coral habitat.

Recovery of these reef systems perhaps has better prospects than those of the granitic islands of the Seychelles (Turner *et al.,* this volume) where mortality was more comprehensive and anthropogenic influences on the marine system are greater, thereby complicating he recovery process.

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The reefs of Mauritius

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ABSTRACT

The study investigated whether the coral reefs of Mauritius had suffered a mass bleaching event during 1998 as had been reported for other Indian Ocean reefs. Sea-surface temperature (SST) anomaly charts produced by NOAA show that SST was raised 1°C - 1.25°C above the climatological maximum for this region during February 1998, but the extent of bleaching around Mauritius was thought not to be severe, but was not recorded. A rapid assessment of the degree of coral bleaching on reefs around the whole coast of Mauritius was made during April 1999. Surveys were conducted while snorkelling and SCUBA diving and assessments made by direct observation, underwater video transects and underwater photography. Video was analysed to confirm the results from the field surveys. Results were displayed within a Geographical Information System (GIS). Meteorological data for the period between January 1997 and April 1999 were also analysed. The results indicate that the coral reefs in Mauritius were still healthy, but that all sites showed some signs of degradation particularly from boat and anchor damage and cyclone damage. The coral reefs of Mauritius do appear to have escaped the mass bleaching event of 1998. There were no large areas of dead standing coral other than on Barrier Reef off Mahebourg. Mean bleaching was <10% at all sites and in all cases represented only partial bleaching of colonies. It is suggested that Mauritius escaped the mass bleaching event due to the effect of cyclone Anacelle, which produced wet and cloudy unsettled weather during February 1998. The minor bleaching episode observed during this survey is thought to be a frequent and normal event relating to large environmental fluctuations experienced within the lagoons. With the potential threat of increasing mass coral bleaching events, it is suggested that Mauritius needs to act quickly to protect its coral reefs from further degradation.

INTRODUCTION

The status of the coral reefs of Mauritius was assessed during April 1999 by rapid site survey at reef and lagoon locations around the mainland island of Mauritius. SST anomaly charts produced by NOAA show that SST was raised 1°C - 1.25°C above the climatological maximum for the region by end of January 1998. SST rose to 1.25°C – 1.5°C above normal in mid February and remained 1°C - 1.25°C above normal until the end of February. The conditions indicated that extensive bleaching was likely during this period, however the extent of bleaching in Mauritius has not been confirmed to date. Turner (1999b) reported bleaching beginning in April 1998, when up to 25% of *Acropora formosa* thickets were partially bleached but still alive. Some *Acropora cytherea*, *Porites,* small faviids and anemones were also partially bleached. The Albion Fisheries Research Centre, Mauritius reported observations from two sites, in the north west and south east of the island, and concluded that bleaching affected 39% and 31% of live corals in the shallow bays of Balaclava Marine Park and Blue Bay Marine Park respectively. Partial bleaching was 27% in both areas and total bleaching in Balaclava and Blue Bay was 12% and 4% respectively (Goorah *et al.*, unpubl.). It thus seems unlikely that Mauritius escaped the mass coral bleaching that was so severe elsewhere in the Indian Ocean, even though Mauritius is located further south than the severely affected areas, and is regularly subjected to fluctuating conditions due to cyclones.

Mauritius has a subtropical climate, and normal sea surface temperatures vary seasonally between 23°C in the winter (September) and 27°C during the summer (February) with a mean of 25.7°C. Tides are semidiurnal and have a very small tidal range of 0.6 m at springs and 0.5 m at neaps. The south east trade winds blow most of the year, especially during the cooler season of May to November. Mean wave heights on the south coast range between 1.67 m in the summer to 2.86 m in the winter (Fagoonee, 1990). Annual rainfall varies from 1200 mm on the north coast to 3600 mm on the central plateau. Mauritius is affected by cyclones each year. Cyclones originate in the lower latitudes of the South western Indian Ocean between November and March, with the highest frequency occurring in January and February. Strong winds (up to 200 km•hr^{-1}), high rainfall and heavy swell accompany cyclones.

One hundred and fifty kilometres of fringing coral reefs surround most of Mauritius protecting a series of lagoons cut by surge channels and river mouths. A section of barrier reef exists in the south east off Mahebourg. Most of the reefs are well-established spur and groove reefs with an algal ridge, although the spur and groove zone is sometimes replaced by dead coral flagstone (Fagoonee, 1990). The reef flat is usually less than 25 m wide and is exposed at low tide. The width of the lagoons varies greatly from a few kilometres to a few hundred metres, with wider lagoons generally occurring on the east coast (up to 4 nautical miles). The lagoons are usually only 1 m – 2 m deep, but reach depths of up to 6 m in the north.

The reefs of Mauritius are degraded by agricultural, industrial and urban run off, eutrophication, overfishing and sand mining, and are used heavily by a coast based tourism industry (Turner *et al.*, in press). The reefs are regularly impacted by cyclones, and there is good evidence that the lagoon patch reefs seasonally exhibit partial bleaching during the summer months (Fagoonee *et al.*, 1999). If Mauritius was affected by the mass coral bleaching event of 1998, then coral mortality could result in socio-economic impacts, including a further decrease in lagoon fish stocks, greater coastal erosion due to the islands exposed oceanic position and a possible decline in tourism. It is important that the degree of bleaching and the current health of the coral reefs in Mauritius are determined, in order that adequate management measures can be taken to encourage rapid recovery of the coral reefs and offer greater protection in the future.

METHODS

Field surveys

A rapid assessment of the degree of coral bleaching was carried out at 34 coral reef and lagoon sites around the coast of Mauritius (Figure 1) between the 8th and 18th of April 1999. Sites were selected using a Landsat 4 TM image (Figure 2) classified by Klaus (1995), and groundtruthed by survey teams from University of Wales Bangor (Daby, 1990; 1999; Dykes, 1996; Orme, 1997; Taylor, 1998; Walley, 1997) (Figure 1). Surveys were conducted from local fishing boats in lagoons, and in collaboration with dive operators outside reefs. Timed surveys of between 15 and 70 minutes were carried out across areas of reef by snorkelling in shallow areas (<5 m depth) and by SCUBA in deeper areas. Assessments of coral bleaching were carried out by direct observation, underwater video and underwater photography over areas approximately 100 m x 100 m.

Two levels of visual survey were made by between four and eight observers who swam across the reef 5 m apart so that the same section of reef was surveyed only once. Broad scale physical and biological features of the reef were recorded using a six point semi-quantitative

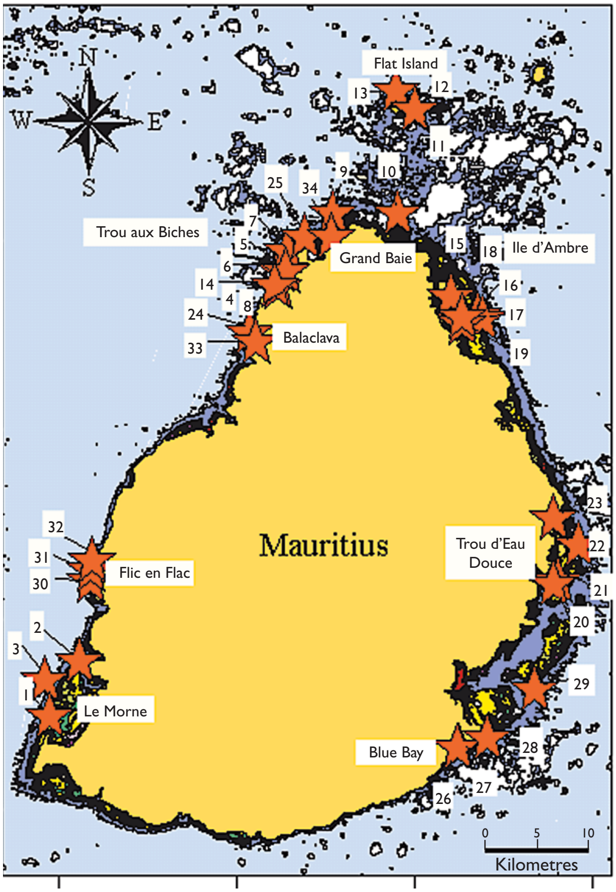


Figure 1. The 34 rapid site assessment locations surveyed around the coast of Mauritius in April, 1999 overlaid on the reef habitat classification within the Geographical Information System.

Figure 2. A rectified true colour composite Landsat 4TM satellite image of Mauritius.

scale (see Turner *et al.*, this volume for full description). These included substrate structure, cover by benthic organisms, and severity of impacts to the reef such as bleaching, *Acanthaster planci*, anchor and storm damage. A second level description of the species composition of the reef was also made in which all hard and soft corals and macro algae were identified to genus or species where possible. The size class of each colony of hard and soft coral was recorded (1-10 cm; 11-25 cm; 26-50 cm; >50 cm) and the abundance of all species was recorded on the same semi-quantitative scale of 0-5 as for the biological and physical attributes. Percentage bleaching observed in each species was also recorded on a scale of 0-5. At the end of the timed swim the results of all observers were combined.

Underwater video was recorded at an angle of 45^{o} to the substrate and the operator swam slowly across the reef following the contour of the substratum. Videotapes were analysed by pausing the tape at randomly spaced intervals and placing sample points at random locations on the monitor screen. Wide angle habitat photographs were taken using Nikonos underwater cameras equipped with a 28 mm lens and flash. The 35 mm slides were used to confirm species identification, and to provide a permanent archival record of the health of the coral reefs and the degree of bleaching observed on these reefs.

Development of a geographical information system (GIS)

Results from the original survey and video analysis were displayed within a Geographical Information System (GIS) using MapInfo Professional. A supervised and rectified Landsat 4 TM image was used to produce a base map displaying 17 classes of reef habitat The positions of the 34 survey sites, acquired using a GPS were added to the base map and a grid of longitude and latitude created (Figure 1). The biological, physical and bleaching data were displayed within a thematic overlay as interactive pie charts. The GIS was used to identify geographical patterns in the degree of bleaching and the general health of the coral reefs around the coast of Mauritius by determining criteria for the percentage cover of particular key biological attributes or species diversity and richness. A query-select function was then used to highlight those sites with a higher or lower percentage cover, species diversity or species richness than specified in the query. Thirty-five millimeter slides showing a representative image of particular sites, bleaching and impacts to the reef were digitally captured and the image files were linked to relevant survey sites.

Analysis of meteorological data

Meteorological data for the period January 1997 to April 1999 were provided by the Mauritius Meteorological Office. Sea-surface temperature and mean significant wave height data were obtained from a Waverider Buoy, located off Blue Bay in the south east of the island and cloud, sunshine and rainfall data were obtained from sites around the island. The data were analysed to investigate whether climatic conditions during the period of sea warming were unusual.

RESULTS

Outline results are presented here. Full details are in the Geographical Information System described by Hardman (1999).

Reef composition

The coral reefs of Mauritius were generally healthy (Figures 3 - 6), even though most sites showed some sign of degradation. Hard coral cover was >50% at 16 of the 34 sites studied and >75% at four of the sites. Soft coral cover was <10% at 31 sites. Dead coral cover was <10% at 16 sites but was >30% at six sites and was >50% at the Barrier Reef. Macroalgal cover was <10% at 25 sites, turf algal cover was <10% at 12 sites, but >50% at Calodyne Reef in the north (site 10) and Ile d'Ambre (north-east, site 16). Unconsolidated rubble was <10% at 23 sites and >30% at two sites: Flat Island (north, 12) and Blue Bay (south-east, 27).

Coral bleaching

The coral reefs surrounding mainland Mauritius had escaped the mass bleaching event of 1998. There were no large areas of dead standing coral that could be attributed to the bleaching event other than corals just behind the shallow reef flat of the Barrier Reef. This site was dominated by dead coral (>50%) covered in turf algae and some small colonies of regenerating corals with healthy tips (Figure 7). Bleaching was observed at 29 of the sites surveyed but in all cases was only partial, and occurred in all zones of lagoon patch reefs, reef crests and on the deeper (<10m) fore reefs in some species. Mean bleaching was <10% at all sites surveyed and was absent at five sites (lagoons at Le Morne (south-west, site 2), Blue Bay (south-east, 27), Flat Island (north, 13), Trou d'Eau Douce (south-east, 23) and on the reef crest at Ile d'Ambre (north-east, 16)). There was no pattern in the sites bleached and no significant difference in median bleaching value between the 34 sites studied (Kruskall-Wallis: $H = 9.00$, $df = 33$, $p>0.05$) or between geographical areas when the sites were combined (Kruskall-Wallis: $H = 3.05$, $df = 9$, $p>0.05$), although Grand Baie (north west, 25) exhibited the highest mean bleaching value (0.79) on a 0 – 5 point scale.

Nineteen species were observed to show bleaching and in all cases only part of the colony was bleached (Figure 8). The number of species bleached at each site

Figure 3. Healthy diverse coral reef community on reef slope on the west coast of Mauritius.

Figure 4. Reef crest on exposed east coast, with abundant *Echinometra mathae*i

Figure 5. Lagoon patch reef, Balaclava Marine Park

Figure 6. Healthy monospecific lagoon patch reef, Blue Bay Marine Park

Figure 7. Algal colonised corals of the barrier reef Mahebourg, which may have bleached

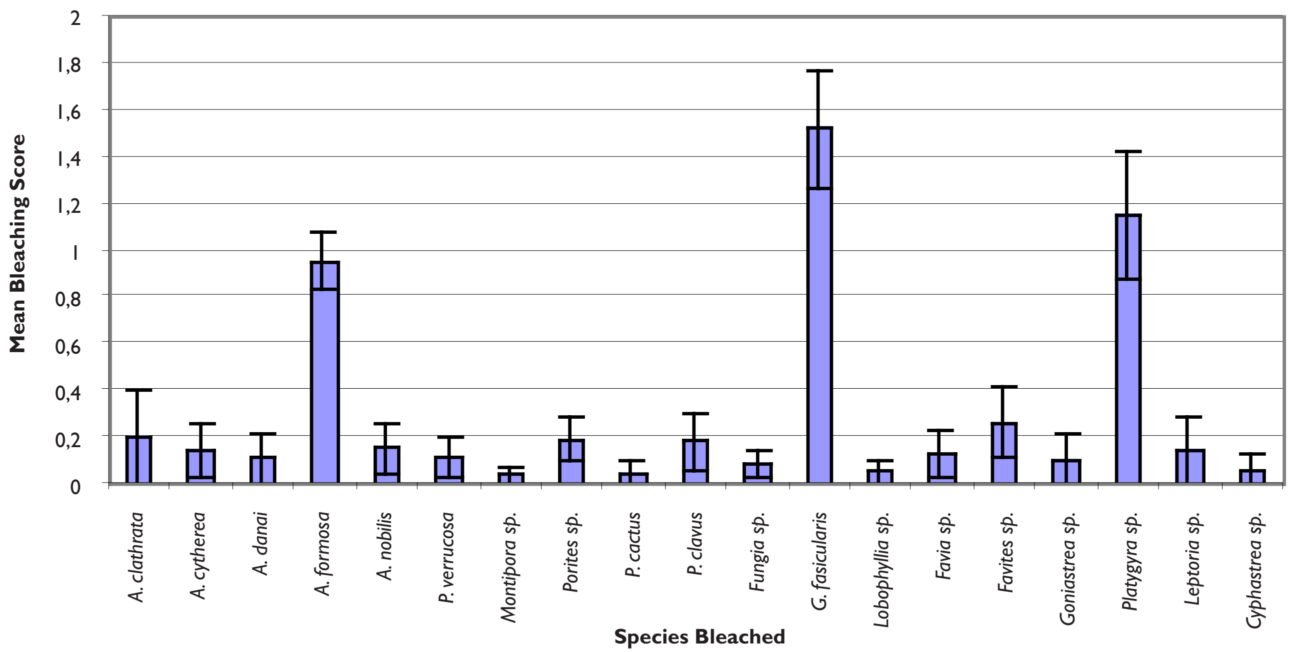


Figure 8. The mean bleaching score (on a scale of 0-5) ± SE of the 19 species of coral observed to be bleached around the coast of Mauritius (n = 365). The values are based on the results obtained in the initial surveys combined with observations from the video analysis.

ranged from 0 (at Le Morne, site 2) to 6 at Le Morne, site 1 (both in south-west). Of the 19 coral species, *Galaxea fascicularis*, *Acropora formosa* and *Platygyra* sp. were the species most often observed to be bleached (figure 8). *G. fascicularis* was bleached at 21 of the sites studied; *A. formosa* at 19 of the sites and *Platygyra* sp. at 14 of the sites. Bleaching in *G. fascicularis* and *Platygyra* sp. occurred mostly in shallow water (<10 m deep) and ranged from a value of 1 (<10%) to 4 (51% - 75%). Bleaching of *A. formosa* ranged from a value of 1 (<10%) to 2 (11% - 30%). Bleaching of *A. formosa* occurred only on the upper surfaces of horizontal branches (Figure 9) while *G. fascicularis* and *Platygyra* sp. exhibited bleaching on the upper surfaces of massive and sub-massive colonies (e.g. colony of *Galaxea* in Figure 10). Bleaching was patchy and bleached colonies were often observed adjacent to unbleached colonies of the same species. There was no significant difference in the median bleaching value of different sized colonies (Kruskall-Wallis: $H = 6.43$, $df = 3$, $p>0.05$).

Other impacts

Human impacts were common and observed at 26 sites including the two Marine Parks at Blue Bay (south-east) and Balaclava (north-west). Fishermen and tourists anchored their boats on the lagoon patch reefs causing destruction of the corals at 21 sites. Trampling damage caused by tourists snorkelling in shallow water and fishermen walking across the reef searching for reef fish and octopus was recorded in the lagoons at seven sites. Fourteen sites displayed fish traps on top of recently broken corals (Figure 11). Other human impacts to the reefs included sand mining at east coast sites and nutrient enrichment in lagoons and on fore-reefs in the north west. Macro algae and soft coral dominated many north west sites. Damage from tourist developments included a bulldozed section of reef at Balaclava Marine Park opposite the construction site of a new hotel, concrete moorings on top of coral and coral heads arranged in a circle, possibly for dive training at Blue Bay Marine Park.

Figure 9. *Acropora formosa* exhibiting bleaching of upper surfaces only.
Figure 10. A large *Galaxea fascicularis* colony partially bleached across its upper surface.
Figure 11. Fish trap on top of corals in Blue Bay Marine Park
Figure 12. *Acropora* tables overturned by cyclone

Natural impacts were also observed at many of the reefs studied. Supposed storm/cyclone damage was observed at 15 sites, where tabular *Acropora* colonies were overturned (Figure 12) and there was a high percent cover of unconsolidated rubble (>30%) on the reef crest at Flat Island (site12) (north) and in the lagoon at Blue Bay (site 27) (south-east). Cyclone Davina passed Mauritius just one month before the survey on 4th March 1999, producing gusts up to 173 km•hr^{-1}. Crown-of-thorns starfish (*Acanthaster planci*) were observed at 12 sites and were particularly abundant in the lagoon at Trou d'Eau Douce (site22) (south-east). The sea urchin *Echinometra mathaei* was abundant in the lagoon at Ile d'Ambre (site 15) (north-east) and on the reef crest at Ile d'Ambre (site 17) (north-east).

Meteorological data from the period of the 1998 sea warming event

Sea-surface temperature (SST)

In situ SST data obtained from the Waverider Buoy between March and July 1997 and February and July 1998 indicated that the mean SST reached its highest values during February (28.5° C) and March (28.1° C) 1998 (Figure 13). The long-term mean sea temperature at this time of year is 27° C, thus temperatures were 1° C - 1.5° C higher than normal. SST was higher in 1998 than in 1997 in both March (1998: 28.1° C; 1997: 27.5° C) and April (1998: 27.4° C; 1997: 26.8° C). No data were available for February 1997. SSTs were significantly higher in February and March 1998 than in April-July 1998 (student's t-test: $T = 5.19$, $df = 7$, $p<0.05$) and were also significantly higher in 1998 than in 1997 (paired t-test: $T = -3.47$, $p<0.05$). These data were confirmed by the SST anomaly charts produced by NOAA.

Cyclone

During the period of high sea surface temperature, Mauritius experienced unstable weather conditions. Between the 6th and 10th of February a trough crossed the region producing thunder and showers over the whole island. This tropical depression was named Anacelle by the Mauritius Meteorological Services on the 8th of February 1998 when it was located at 13.2° S, 61.0° E. As it moved south south west, it intensified and became a tropical cyclone while passing off the west coast of St. Brandon. The cyclone passed approximately 60 km off Belle Mare on the east coast of Mauritius on the 11th of February 1998, and the air mass remained unstable until the 15th of February 1998. During the second half of February, wet and unstable weather persisted due to another low-pressure system located off the east coast of Madagascar. Clouds associated with this low continued to influence the weather until the 24th of February 1998 when it started to move south eastward.

Rainfall

As a consequence of cyclone Anacelle, rainfall during the first half of February was above average and during the second half the west coast of Mauritius received 422% of the long-term mean rainfall (Figure 14). Significantly more rain fell in February 1998 than in February 1997 and February 1999 (Wilcoxon: $H = 6.3$, $df = 2$, $p<0.05$) (546 mm of rain fell during February 1998 compared to 95 mm in 1997 and 111 mm in 1999 in the north).

Figure 13. Mean sea-surface temperature (°C) in Mauritius between March 1997 and July 1998. Data were obtained from the Waverider Buoy, located off Blue Bay. Courtesy of the Meteorological Office, Mauritius.

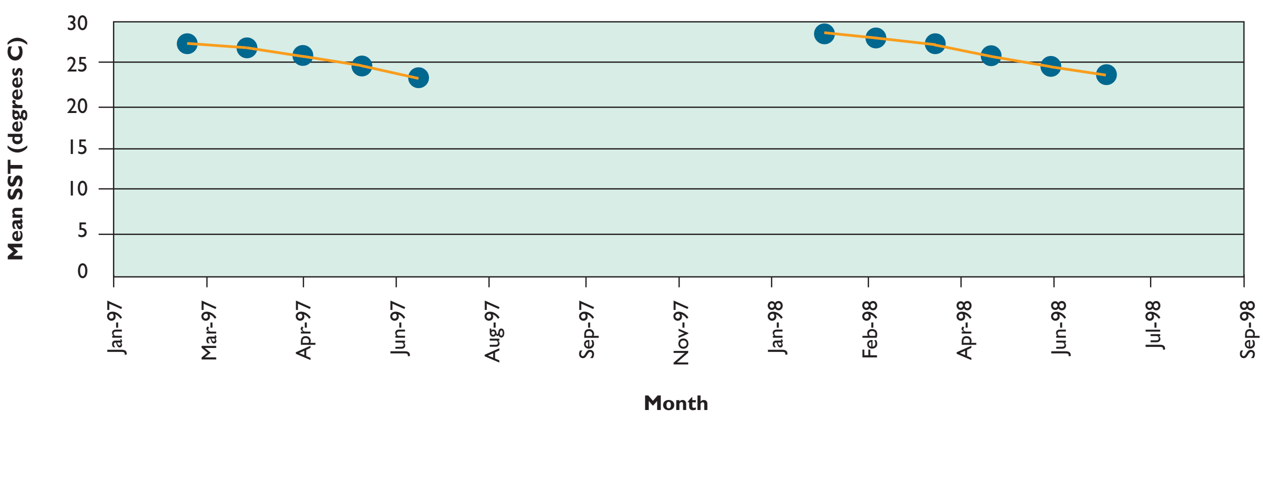
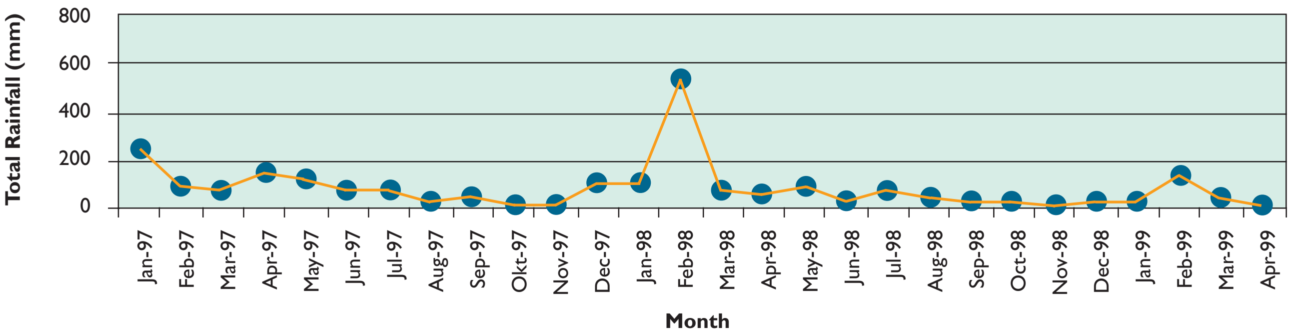


Figure 14. Total rainfall (mm) per month between January 1997 and April 1999 for sites in the north west of Mauritius. Data were provided by the Mauritius Meteorological Office.



Cloud cover and sunshine

February 1998 had the highest daily mean cloud coverage (6.7 hours, recorded at Vacoas) during the whole period between January 1997 and March 1999 (Figure 15) due to cyclone Anacelle and the subsequent low-pressure system. Total hours of sunshine in February 1998 were the lowest recorded during the period between January 1997 and April 1999, due to the high cloud coverage (Figure 16). Analysis of variance indicated that there were significantly lower hours of sunshine in February 1998 than in February 1997 ($F = 4.68$, $df = 17$, $p<0.05$) (only 126 hours of sunshine in February 1998 compared with 235 hours in February 1997 and 190 hours in February 1999 on the west coast).

Tides

Neap tides occurred during the first and third weeks of February 1998 and low tide occurred in the middle of the day between 1000 hrs and 1600 hrs. The tidal range was small (20 cm) and low tide was not extreme (170 cm above Chart Datum). Spring tides occurred during the second and final weeks of February with a range of 40

Figure 15. Mean cloud cover per month between January 1997 and March 1999. Data were provided by the Mauritius Meteorological Office.

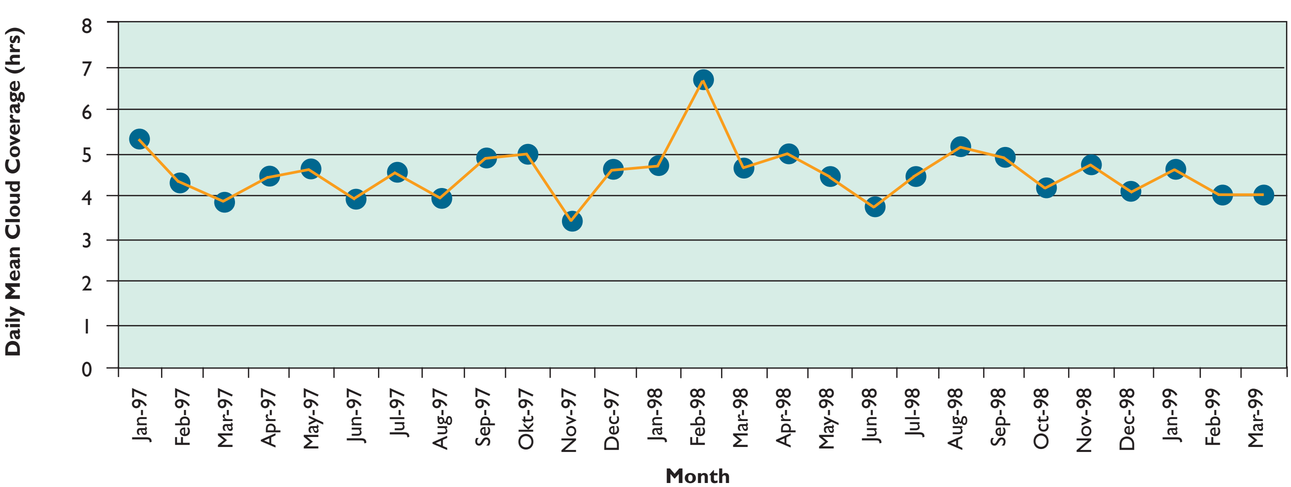
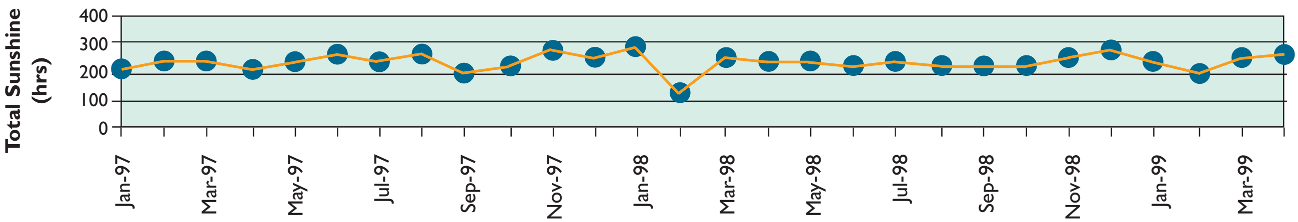


Figure 16. Total sunshine (hrs) per month between January 1997 and April 1999 for west coast. Data were provided by the Mauritius Meteorological Office.



cm – 50 cm and lowest low tides (150 cm - 160 cm above CD) occurred between 0600 hrs and 0800 hrs and between 1800 hrs and 2000 hrs.

Conditions during the 1999 survey
SST did not exceed its normal monthly maximum during the April 1999 survey. SST was normal throughout January 1999, rose to 0.5°C - 0.75°C above normal during February and March and returned to normal at the end of March (NOAA SST charts). Two tropical depressions and one cyclone caused unsettled conditions during the three months preceding the 1999 survey. Unusually, very little rain was associated with these tropical depressions and during the first four months of 1999, rainfall was significantly less than in the first four months of both 1997 and 1998 (Kruskall Wallis: $H = 10.97$, $df = 2$, $p<0.05$). Sunshine levels during the first four months of 1999 were normal or above normal in all parts of Mauritius.

DISCUSSION

The results show that the coral reefs of Mauritius escaped the mass bleaching event, which severely affected other areas of the Indian Ocean in 1998. Sea-surface temperature (SST) anomaly charts produced by NOAA indicate that SST in Mauritius was raised 1°C - 1.25°C above the climatological maximum for the region for over one month. *In situ* data from the Waverider Buoy confirmed the satellite data, showing that sea surface temperatures offshore from Blue Bay were 1°C - 1.5°C higher than normal. The temperature rise was not as high as in other regions of the Indian Ocean (Turner *et al.*, this volume), but such temperature increases have been shown to induce bleaching in corals in the laboratory (Jokiel & Coles, 1977) and the field (Glynn, 1984). A mass bleaching event should therefore have been expected in Mauritius during 1998.

One year later there were no indications that Mauritius had suffered a mass bleaching event. Of the sites studied, 47% were healthy with >50% coral cover, which compares favourably with the status of reefs elsewhere (Hodgson, 1999). Using the ASEAN-Australia Living Resources criteria (Chou, 1998), 47% of the sites surveyed can be described as 'Excellent' and 12% can be described as 'Good'. Sites with low hard coral cover were the reef crest sites, which have a naturally low cover of hard coral colonies due to wave and air exposure. These sites displayed a high species diversity and low dominance of corals (*Porites* sp., *Pavona cactus, Platygyra* sp., *Favia* sp. and *Goniastrea* sp.). Fore reef sites showed high similarity with *Porites* sp., *Lobophyllia* sp., *Platygyra* sp., *Pocillopora damicornis* and *Favites* sp. Being common to all sites. In contrast, 57% of lagoon patch reefs had >50% coral cover, but these sites had low species diversity and were dominated by large colonies of *Acropora formosa* and *Acropora cytherea*.

Monitoring of previous bleaching events elsewhere (Brown & Suharsono, 1990) suggest that the coral reefs in Mauritius could not have recovered from a mass bleaching event within one year. Indeed, reefs have generally deteriorated further one year on from the 1998 bleaching event (see Lindén & Sporrong, 1999 for country reports), many being covered in thick algal turf, and in Chagos (Sheppard, 1999a;b), Socotra (Turner, 1999a) and Seychelles (Turner *et al.*, this volume) dead reefs have been eroded to rubble. In Mauritius, no large areas of dead standing coral covered in macro and filamentous algae could be attributed to the bleaching event, other than at one site on the Barrier Reef.

The actual cause of bleaching at a particular reef may be a combination of factors other than high SST (Wilkinson *et al.*, 1999). Factors such as the level of storm activity, the number of days of cloud cover and the strength and direction of winds all affect how warm water is dissipated or builds up to lethal levels. Many workers have reported that mass coral bleaching follows extended periods of high temperatures, low wind velocity, clear skies, calm seas and low rainfall, where conditions favour localised heating and high penetration of UV radiation (e.g. Brown & Suharsono, 1990; Williams & Bunkley-Williams, 1990; Glynn, 1991; Goreau & Hayes, 1994).

In Mauritius, meteorological data indicated that during the period of elevated SST, unstable weather prevailed and cyclone Anacelle caused higher cloud coverage (mean of 6.7 hours per day), very high rainfall and lower hours of sunshine than normally experienced at that time of year. Also, low spring tides occurred during the early morning and evening and therefore reef flats were not exposed during the hottest part of the day. Further, the lagoons in Mauritius are often cooler than the sea outside because they receive large quantities of cooler freshwater from terrestrial run-off (Daby, 1994).

Site 29 on the Barrier Reef did show signs of possible mass bleaching. This site was dominated by dead standing coral (>50%) covered in turf algae and by small colonies of regenerating corals with healthy tips. The Barrier Reef is situated 3-5 km off the east coast and thus may be subjected to different climatic and oceanographic conditions than the inshore sites surveyed. The meteorological data do not give any indications as to why this site may have bleached when other sites did not. One possible explanation is that unlike mainland Mauritius, high cloud cover did not protect the Barrier Reef, allowing high levels of UV radiation to penetrate the shallow water. Unfortunately, no satellite image was available for Mauritius during this period to confirm offshore cloud cover. Another possible explanation is that localised oceanographic features may have caused the water around the Barrier Reef to heat up more than at the inshore sites. Further, being offshore could have meant that the reef was not affected by freshwater run-off, which tends to cool some parts of shallow lagoons (Daby, 1994).

During the 1999 survey, bleaching was observed at 85% of the sites visited, however, in all cases it was minor (<10%) and partial, rather than total bleaching. Fagoonee *et al.* (1999) have shown that there is a large variability in the zooxanthellae population of *Acropora formosa* with regular episodes of low densities occurring in the spring and summer. Minor bleaching episodes such as that observed during this survey may be a frequent and normal event relating to large environmental fluctuations experienced within the lagoon. Partial bleaching of *Acropora formosa* occurred only on the upper surfaces of horizontal branches. *Galaxea fascicularis* and *Platygyra* sp. bleached mostly in shallow water (<10 m) and only bleached on their upper surfaces. Others have reported greater bleaching on the upper or more light exposed surfaces and that portions of upper surfaces that fell in shadows of fixed objects were often not bleached (e.g. Williams & Bunkley-Williams, 1990). Harriott (1985) suggested that bleaching on the upper and unshaded surfaces of corals on the Great Barrier Reef indicated that the cause of the bleaching was high levels of solar irradiance. In Thailand, localised, naturally occurring bleaching at intertidal sites was also attributed to solar irradiance (Brown *et al.*, 1994). It is therefore, possible that the bleaching observed during this survey was a normal minor event caused by UV radiation. Sunshine levels in Mauritius during the first four months of 1999 were significantly higher than the two previous years, supporting this theory.

The number of coral species bleached per site varied from 0 to 6 and bleached colonies were often observed adjacent to unbleached colonies of the same species. Other workers have commented on the patchy spatial distribution of bleaching in coral colonies (e.g. Oliver, 1985; Glynn, 1990; Jokiel & Coles, 1990; Lang *et al.*, 1992). Edmunds (1994) suggests that this intraspecific variation in coral bleaching is due to the distribution of bleaching-susceptible genotypes. Rowan *et al.* (1997), explain the patchy distribution of bleaching by the preferential expulsion of symbionts associated with low irradiance, suggesting that some colonies are protected from bleaching by hosting an additional symbiont that is more tolerant of high irradiance and temperature. Fitt and Warner (1995) also explain interspecific variation in bleaching by different physiological tolerances of the specific symbiotic algae of the different coral species.

For many of the species that suffered bleaching, particularly *Platygyra* sp., *Galaxea fascicularis, Favia* sp., *Favites* sp. and *Porites* sp., the colonies appeared pale rather than fully bleached. This can cause confusion as to whether the colony is actually bleached as some corals that appear to have bleached may have been light-adapted (Falkowski *et al.*, 1990). Brown and Ogden (1993) state that some intertidal coral species have an adaptive behavioural response to reduce desiccation, known as blanching, in which they pull back their external tissues, leaving their skeletons exposed. The misidentification of bleached colonies could cause an overestimation of the extent of coral bleaching. It has therefore, been suggested that a standardised method is needed to assess the degree of coral bleaching (Glynn, 1993). There are a number of difficulties with standardising the degree of bleaching, such as the assessment of bleaching in colonies of the same species with different patterns of tissue colouration (Knowlton *et al.*, 1992) and the presence of genetically different zooxanthellae with different environmental tolerances (Rowan & Powers, 1991). In order to predict the future of coral reefs it is, however, important that bleaching events around the world are reported in a comparable manner.

Mauritius was not severely affected by the bleaching event of 1998, however, computer models predict an increase in climatic fluctuation and the severity and scale of coral reef bleaching. The coral reefs around the whole coast of Mauritius are degraded by human and natural impacts. Stressed corals succumb more readily to bleaching stress and are more likely to die. Those that do survive may produce fewer larvae to repopulate damaged areas. It is therefore, important that reef management acts to reduce anthropogenic impacts on these coral reefs. At present, most reefs in Mauritius are affected by trampling and anchor damage, overfishing, pollution and eutrophication. Even in the two Marine Parks fishing traps and boat damage were observed. If greater protection is not given to the coral reefs of Mauritius they may not survive the next bleaching event. In addition, Mauritius is one of the few areas in the Indian Ocean not affected by the bleaching event. Coral reefs in the Maldives suffered up to 90% mortality and in the Seychelles mortality was close to 80% - 100% (Lindén & Sporrong, 1999, Turner *et al.*, this volume). In both areas, diving and coastal tourism are a major source of income and it has been predicted that if recovery is slow, there may be a loss in tourism as tourists go elsewhere (Lindén & Sporrong, 1999). As information is spread throughout the diving community the number of tourists visiting Mauritius may increase, resulting in increased pressure on the reefs. Furthermore, reefs in the western Indian Ocean are closely linked by ocean currents of the sub-tropical anti-cyclonic gyre and the seasonally reversing monsoon gyres (Rao & Griffiths, 1998). The coral reefs in Mauritius, may therefore, act as a source of larvae for those coral reefs that suffered severe damage during the bleaching event. Little is known about the source-sink relationships in the Indian Ocean, and the destruction of this source of larvae could prevent the recovery of coral reefs in other parts of the Indian Ocean (Salm *et al.*, 1998).

Despite the fact that Mauritius appeared to have escaped the mass bleaching event of 1998, the island's coral reefs are not secure. There has been significant progress in the management of reef resources in Mauritius in recent years, but greater protection is needed. Stricter regulations and surveillance is required within the two marine parks to prevent boat damage and illegal fishing. In addition, an integrated coastal zone management (ICZM) plan (Fagoonee & Daby, 1993) needs to be implemented and further designations need to be made to limit human activities to certain areas of reef. With the potential threat of increasing mass coral bleaching events, Mauritius needs to act quickly to protect its coral reefs from further degradation from man's activities and give them the best chance to survive natural impacts. These reefs may prove to be an important source of larvae for the recovery of other reefs in the Indian Ocean.

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Coral bleaching in the Indian Ocean islands: Ecological consequences and recovery in Madagascar, Comoros, Mayotte and Reunion.

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INTRODUCTION

During the period from January to August 1998, the largest coral bleaching event and subsequent mortality ever recorded had a huge ecological impact on coral reefs throughout the Indian Ocean. This event corresponded to increased seawater temperatures due to an ENSO phenomenon (Wilkinson, 1998). The full extent of the socio-economic impacts will depend on the recovery capacity of corals which, in many locations, are seriously threatened by human activities. This study documents the ecological status and recovery of corals reefs from the Comoros archipelago (Comoros, Mayotte, Geyser), Madagascar and Réunion which were affected by the bleaching from January to August 1998. The impact of the bleaching at each location varies in its extent in time and severity.

METHODS

To survey the status of the coral reef communities, in late 1999, a simple and harmonised methodology was developed for all the islands of the Indian Ocean. For each participating country, CORDIO sites were defined according to available information on the 1998 bleaching event, but also to strengthen existing monitoring programmes such as GCRMN, Reef Check and COI programmes. This strategy was chosen to ensure improved long-term monitoring in future activities. Estimation of percentage cover by each major reef component was performed at one or more stations. Rapid assessment and/or line transects were used as described in the GCRMN adapted guidebook for the Indian Ocean islands (Conand *et al.,* 1997). Fish populations were sampled in belt transects.

RESULTS AND DISCUSSION

In the four islands covered in this report, a total of 10 monitoring sites with various morphological categories of reef habitats: flats, inner slopes, outer slopes and lagoons were studied (Table 1). Specific observations of benthos and fish populations conducted during the surveys are described and discussed in the follow sections.

Comoros

Grande Comore is characterised essentially by fringing reefs. The results of the survey in North-West Mitsamiouli Reef suggests that 10% bleaching had occurred by November 1999. The cover of live corals was 40%, recently dead coral 15%, and hard substrates with algae, 32%. Itsamia, located in Mohéli island displayed a simi-

Table 1. Location of CORDIO monitoring stations.

Country/Reef	Reef Type	Coordinates
COMOROS		
Mitsamiouli	Fringing reef	
Itsamia	Fringing reef	12°21'56" S 43°52'22" E
MADAGASCAR		
Ifaty	Barrier outer slope	23°09'41" S 43°34'66" E
Nosy Ve	Outer slope	23°38'35" S 43°36'16" E
MAYOTTE		
Geyser Bank	Offshore bank	12°20'39" S 46°26'29" E
Longogori pass	Outer slope	12°52'57" S 45°16'66" E
Surprise	Inner lagoonal reef	12°38'60" S 45°07'86" E
RÉUNION		
La Saline	Fringing reef	21°04'87" S 55°13'10" E
Saint Leu	Fringing reef	21°90'12" S 55°17'40" E

lar pattern with 36% live coral cover and 47% dead substrates of which 18% were recently dead colonies. The community composition of live corals was *Acropora* (22%), *Diploria* (20%), *Favia* (2%) *Pavona* (5%) and *Porites* (50%). The fish community was dominated by Scaridae, Serranidae and Pomacentridae. Specimens were small indicating the impacts of over-fishing. This was found in Grande Comore also.

Mitsamiouli was surveyed in 1998 as part of the COI Reef network (Bigot *et al.*, 1998) and displayed 47% live coral cover. No recovery seems to have occurred at this reef during the last year. Itsamia was not surveyed in 1998.

Madagascar

Two sites were surveyed in the south-western region of Toliara: Ifaty and Anakao/Nosy-Ve. Neighbouring Northern areas (Belo sur Mer) were strongly affected by bleaching in February/March 1998. According to satellite imagery analysis and questioning of the local population, it was concluded that the warm water mass did not seriously affect this area. At Ifaty, one transect was established on the outer slope. Live coral cover was 40.7% (20% *Acropora* and 20.7% non *Acropora* species). Approximately 14% of the 54.8% dead substrate was found to be recently dead and thought to be related to the bleaching event. Of the 36.8% dead coral cover at Nosy-Ve, 26% were reported as covered by recent algal turfs. These data support the idea that coral bleaching did occur in coral reefs from Toliara region but was not reported during the onset phase.

Overfishing was affecting the reefs of Ifaty (28 carnivorous individuals). Chaetodont fishes dominated the fish population (105 individuals.). Proliferation of crown-of-thorns starfish (*Acanthaster planci*, COTS) was seen in lagoonal locations during the survey while a few juveniles were responsible for whitening coral colonies on the outer slopes. At Anakao, the fish population was higher (73 carnivorous fishes and >160 Chaetodontidae).

Data from the 1998 survey confirm that algal communities are dominant on this outer slope. However, live coral coverage increased significantly (1998: 29%; 1999: 40.7%).

Mayotte

Most coral communities (reef flats, inner and outer slopes, inner reefs, fringing reefs) suffered of bleaching from April to August 1998 (Descamps *et al.*, 1998). Mayotte's reefs were previously affected by bleaching in 1982/83 and 1987/88 but to a lesser extent and severity than during the 1997/98 event. Two sites in Mayotte's coral ecosystem, both part of the Coral Reef Observatory (ORC) implemented yearly by local marine environment authorities, were monitored: Longogori Pass (S pass) in the West and Surprise inner reef in the northeast. Another site was monitored on the off-shore Geyser Bank, located 100 km from Mayotte.

As shown in table 2, the bleaching impact was severe on Mayotte's coral reefs. Live coral coverage in Surprise and Longogori reef flats was very low (4.6% and 6.2% respectively), while on the deeper reefs, coral cover was higher (28% and 21% respectively). Recovery of colonies varied greatly from place to place. Susceptibility of corals at the CORDIO sites (2% to 4.1% recovery rate) was of lower importance, compared to the great recovery capacity of fringing reefs. As Mayotte experienced bleach-

Table 2. Benthic cover of surveyed reefs. Notes : cover of bleached coral and dead coral are proportions of total cover, not of Live coral or Algae subate, respecitvely. Recovery was estimated from field studies (tips growth and recent settelment) and from COI 98/99 comparison.

Country	Site	Station	Depth, m		Coral, %		Substrate, %		Other, %	Recovery
			Outer	Inner	Live	Bleach*	algae	dead coral*		(cm/yr)
Comoros		Mitsamiouli		1	40.0	10.0	47.0	15.0	13.0	
		Itsamia		1	36.0	4.0	47.0	18.0	17.0	
Madagascar		Ifaty	8		40.7	1.0	54.8	14.1	4.5	
		Nosy Ve		x	42.3	10.0	36.2	26.0	21.5	
Mayotte	Surprise	Surprise		0.5	4.6	0.0	92.5	90.0	2.9	2.6
			-6		28.0	0.0	71.5	69.2	0.5	2
		Longogori	-6		21.6	0.0	77.4	32.8	1.0	4.1
				0.5	6.2	0.0	79.7	12.2	14.1	1.5
		Geyser	-6		63.1	0.0	15.8	9.3	21.1	0.5
				-6	34.3	0.0	47.3	25.0	18.4	
Réunion	La Saline	Planch'Alizé	-12		40.0	0.0	50.0	1.5	10.0	
				1	40.0	5.0	30.0	13.2	30.0	
		Trois Chameaux	12		50.0	<5,0	40.0	0.4	10.0	
				1	30.0	<10,0	10.0	< 5	60.0	
	Saint Leu	Corne Nord	-8		60.0	0.0	20.0	3.6	20.0	
				1	50.0	0.0	20.0	3.5	30.0	

ing events in 1983/84 and 1987/88, one can suggest that colonies situated in the lagoon are adapted to seawater heating and/or rapid growth process. In May 1999, Quod surveyed the outer slope of Longogori Pass and observed that most corals as deep as 35 m depth were dead and that the few remaining living colonies were bleached (Quod, unpublished). In the November survey, up to 100% of tabulate *Acropora hyacinthus and A. cytherea* that were found healthy in our previous study (Quod *et al.*, 1995) were dead.

In both locations the fish population was dominated by acanthurids (mainly *Ctenochaetus striatus*: 30-100 individuals. At Surprise; 49-51 individuals at Longogori). Damselfish (*Chromis viridis*) were observed only at Surprise (140-200 individuals). Butterfly fish (Chaetodontidae) occurred exclusively on the outer slope of Longogori (48 individuals).

Change in the live cover of coral from 1998 to 1999 was tightly associated with recruitment of *Goniastrea retiformis*, *Pocillopora* spp. and several other species of faviid on Surprise reef flat (Figure 1). On the outer slope, live coral cover increased 12%. At Longogori, the cover of live coral did not change appreciably on the reef flat (1998: 6.4%; 1999: 6.2%) while at deeper sites, an 11% increase was noticed (1998: 11.1%; 1999: 21.6%).

Due to its economic importance, Geyser bank was surveyed in November 1999. As information describing the impact of the coral bleaching event was not available, data collected suggest that reefs are healthy with 63.1% live coral cover at 6 m depth. Additional evidence of recent dead branching colonies (9.3% & 25%) suggests that Geyser Bank has been slightly affected by bleaching. Geyser Bank lies far away from human activities and displays high hydrodynamic conditions, which protects it from warming.

With a 10% increase in live coral coverage, recovery is underway in most locations around Mayotte except on some outer slopes. High degradation rate and slow recovery of outer slope species may result partially from stratification of the seawater mass in the lagoon during summer or unpredictable warming events. During the bleaching event, colonies on the outer slopes, which usu-

Figure 1. Recovery from settlement of *Pocillopora* and *Acropora* colonies at Surprise inner reef (Mayotte)(J. Turquet)

Figure 2. *Acropora* sp. colony from Trois Chameaux/Réunion showing bioerosion (lower part) and recovery from the tip, collected in November 99. Bleaching was observed a that time. (JP Quod)

ally experience good quality waters in terms of temperature, turbidity, salinity and nutrients were not able to tolerate the heat and suffered massively from the outflow of warm lagoonal waters, and displayed a slower growth rate. In other locations of the lagoon, fringing reef colonies were observed to be alive and displaying signs of recovery and health (Figure 2).

Réunion/France

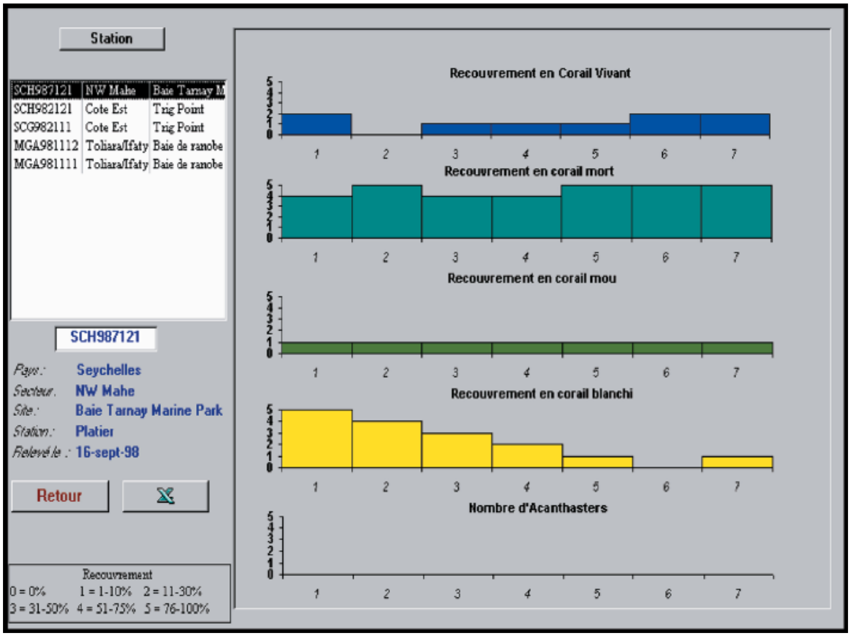
Transects of La Saline (Planch'Alizé and Trois Chameaux) were sites which have been previously surveyed in 1998 (Figure 2 & 3). Rapid assessments at 1m depth confirmed that the majority of bleached corals that suffered partial mortality in 1998 were showing signs of recovery. On the outer slopes and in the lagoons at both sites, live coral cover was good (40% - 50% and 30% - 40% respectively). During the survey period, Trois Chameaux was suffering from a localised bleaching process affecting *Acropora formosa* colonies (av. 10%). Necrosis seemed to be a result of a viral disease as water temperatures were normal (Naim, pers. comm.). Live species observed in the Trois Chameaux lagoon were *Acropora* (80%) and *Porites*, *Pavona*, *Pocillopora* (20%) but degradation was more prominent in the southern part. White colonies of *Pocillopora* on the outer slope was associated with COTS. At Planch'alizé station, live cover of coral was higher (40%) but colonies of *Acropora* were few and have been replaced by colonies of *Montipora*, *Porites*, *Pocillopora* and *Pavona*.

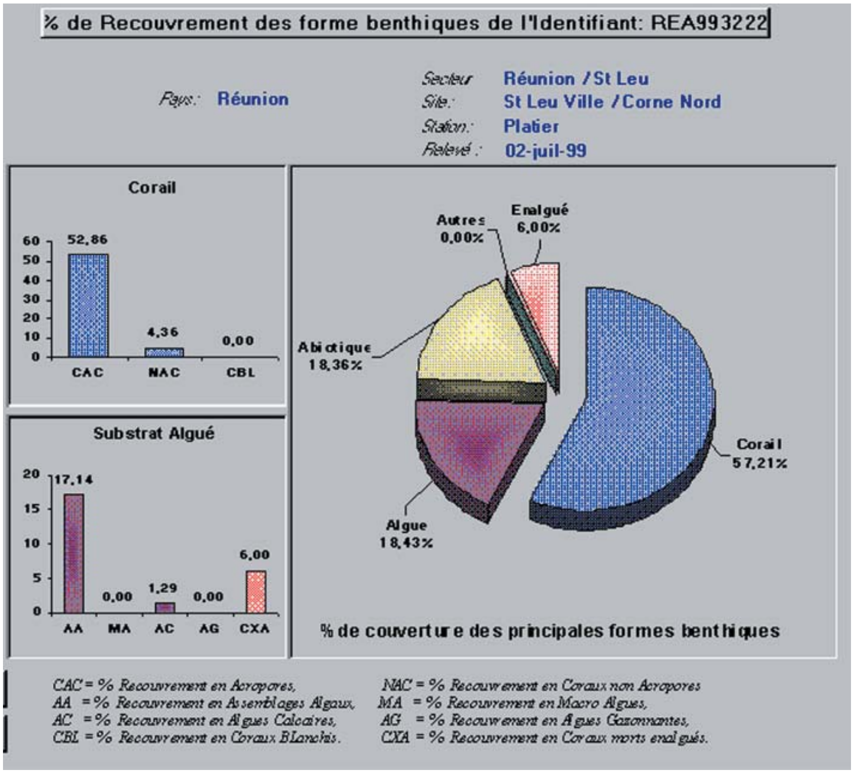
Recovery of corals from remaining tips of branching colonies was studied, with special interest on their capacity to grow during this one-year period. If good recovery (10 cm growth) was noticed, bioerosion had affected the structural integrity of branching forms and with the high incidence of cyclones, integrity is likely be compromised.

The abundance of juvenile fishes was recorded at all locations. *Stegastes nigricans* (Pomacentridae) were dominant where dead branching corals with algal turfs occurred (400 - 700 individuals) while in sub-massive colonies *Dascyllus aruanus* was the dominant species (Planch'alizé: 300 individuals). The population of the non-territorial grazing herbivorous fish *Ctenochaetus striatus* was significant in Planch'alizé (700 individuals).

In Saint Leu Cxorne (northern tip), live coral cover was 50% on the reef flat and 60% on the outer slope. This area suffered from bleaching in February 1998 but recovered in the following weeks. There was evidence

Figure 3. Processing result of LIT in the adapted ARMDES-COI database, icluding bleaching as a specific component (CBL). When monitoring reefs, white corals were first considered as dead colonies.





that due to its low impact, dead corals resulting from the bleaching were less than 3.5%. No monitoring was implemented in 1998.

A large recruitment of juvenile fish occurred at Saint Leu also. The dominant species were *Ctenochaetus striatus* (flat: 22 individuals, slope: 202 individuals) and *Plectrogliphydodon dickii* (flat: 16 individuals; slope: 68 individuals).

CONCLUSION

To provide valid and integrated data for future management and conservation of coral reefs, the CORDIO stations were established under the existing network of GCRMN in the Indian Ocean islands as complementary components of the operational Reef Network for COI and Coral Reef Observatory for Mayotte.

The ocean warming took place between February and August 1998 and the subsequent extent and severity of the mortality varies from place to place and with depth. Mascareignes islands (Réunion, Mauritius, Rodrigues) were slightly affected as they are located in open ocean and experienced protection by a cloudy season during the onset period.

The southeast coast of Madagascar was hit by the warm water mass early in 1998. Bleaching was not significant while in the northern location of Belo sur Mer, most corals turned white.

In the Comoros archipelago, all the islands suffered from bleaching. Geyser Bank located off-shore of Mayotte was also affected by bleaching. The worst affected site was Surprise inner reef flat of Mayotte with only 4.6% live corals.

After the 1997-1998 coral bleaching event, affected reefs from the Indian Ocean islands are at present undergoing regeneration either by recruitment or recovery from live tips with an average 10% rate for slopes. Evidence of 10 cm growth from tips was observed for branching corals of Réunion and Mayotte, but bioerosion of dead bottom skeletons may lead to breakdown of colonies with high water energy. Analysis of available data on benthos cover for all CORDIO sites confirmed that discrepancies exist. Reef flats still display high algal substrates dominance. The same pattern was observed for fringing reefs coral species, which are adapted and more tolerant to high temperatures. Outer slope and deep-water species usually experienced good water quality in terms of temperature. Exposure to long-lasting high temperatures had severe consequences.

Relationships between coral reef substrata and fish have been confirmed in Réunion island reef flats and outer slopes (Chabanet *et al.*, 1997). Associated fish species monitored in the CORDIO sites were mainly herbivorous species (*C. striatus*) and damsel fish Pomacentridae. No significant change was detected to others fish groups which are however subject to increasing human pressure.

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The status of the Aldabra Atoll coral reefs and fishes following the 1998 coral bleaching event

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INTRODUCTION

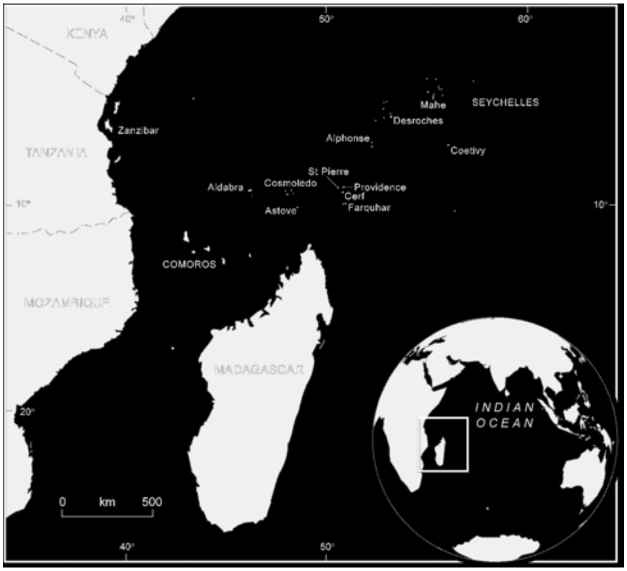
It is important to establish benchmark reef locations that are remote from centres of human activity and free from anthropogenic disturbances, against which human impacts elsewhere can be assessed and rates of recovery evaluated. Aldabra Atoll in the southern Seychelles, is free of anthropogenic disturbances and an ideal location in which to study reefs and adjacent ecosystems. It has further significance with it being in the middle of a region which has been classified as having a number of reefs at high risk (Bryant *et al.*, 1998) and has been designated as a UNESCO World Heritage site.

Aside from acting as a regional benchmark there are a number of practical reasons why Aldabra provides a key component to monitoring the effects and long-term impacts of the bleaching event. Many studies of coral bleaching have concentrated upon documenting the degradation of structural reefs into algal-dominated systems. Yet, the reverse set of processes are of considerable importance: the ability of a coral reef to replenish lost coral populations, reinstate framework growth and recover reef structural complexity. The recovery from high bleaching related coral mortality of these oceanic reefs, such as those around Aldabra, is of much regional interest. The systematic monitoring of the Aldabra reef environment will provide a greater understanding of the natural reef processes acting locally, with implications to understanding reef dynamics elsewhere in the region.

ALDABRA ATOLL

Aldabra Atoll (9°24' S, 46°20' E) is a large (34 km long, maximum 14.5 km wide) raised atoll 420 km north of Madagascar (Map, Figure 1). Raised reef limestones, averaging 2 km in width and up to 8 m above sea level, enclose a shallow central lagoon. A tidal range of 2 to 3 m results in large-scale hydrodynamic exchanges between the lagoon and open ocean through two main channels and a number of small ones. The climate is heavily influenced by NW monsoon winds from November to March bringing the heaviest rainfall with SE trades blowing throughout the remainder of the year.

Previous studies proposed that the reef front areas of Aldabra Atoll could be classified into six morphological categories, based on exposure to wave and storm action in the shallower depths, and on light attenuation in the deeper reef zones (Barnes *et al.*, 1971; Drew, 1977). The western reefs are characterised by a 460 m wide reef flat, a reef ridge margin and reef front slopes of 20° to 45°. The topography of reefs on the northern coast of Aldabra varies very little. Characteristically, there is a very short reef flat that slopes gently down to 10 m, below which the reef drops at an incline of approximately 35° - 45°. At between 20 m and 25 m the reef shelves off to



Map of the Western Indian Ocean with islands of the Granitic and Southern Seychelles.

form a sandy ledge with patches of coral growth and then drops off steeply again at between 35 m and 40 m. The east and southeast coasts are the most severely exposed and have neither a reef flat nor a reef ridge. No hermatypic corals are present. Finally, on the southern, less exposed shore the reef flat is present but not delimited by a prominent ridge. The reef front itself is characterised by large areas of dead coral which vary greatly in extent.

There is a long history of Aldabra reef fish studies extending back over 100 years (i.e. Jatzow & Lenz, 1899; Regan, 1912). The fishes of Aldabra were included in a description of 820 marine fish of the Seychelles (Smith & Smith, 1969). This list was expanded and revised by numerous studies during the period 1969-1979 to 883 species in the region (Polunin, 1984). Specific studies of the fishes at Aldabra found a high diversity, with 185 species recorded in a 300 m^2 section of reef habitat in 1973. However, there were substantial variations in species and abundance between habitats (reef-slope – 228 species, back-reef – 146 species; Polunin, 1984). Fish surveys conducted at Aldabra Atoll in April-May 1998 noted 287 species from 35 Families (M. Spalding, University of Cambridge, unpublished).

CORAL BLEACHING IN ALDABRA IN 1998

Localised sea-surface temperature (SST) records for Aldabra indicate that SSTs for 1998 were the highest of the previous three and a half decades (Figure 2). Anomalous temperatures began with a rapid increase in SSTs from November 1997 to a +1° C SST anomaly by January 1998. Peak SSTs (30.65° C) were reached in March,

Figure 1. Map of Aldabra Atoll with the locations of the seven permanent coral reef transects and six settlement plates sites (■) established by the Aldabra Marine Programme (AMP) 1999.

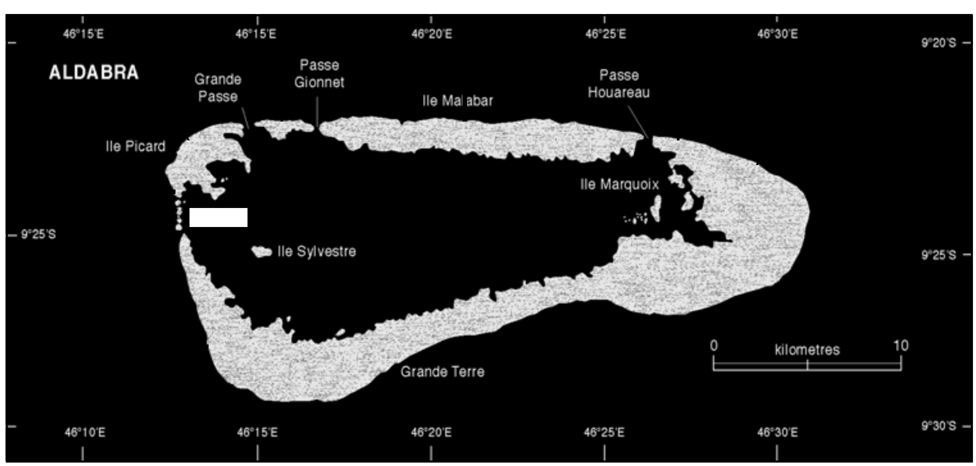
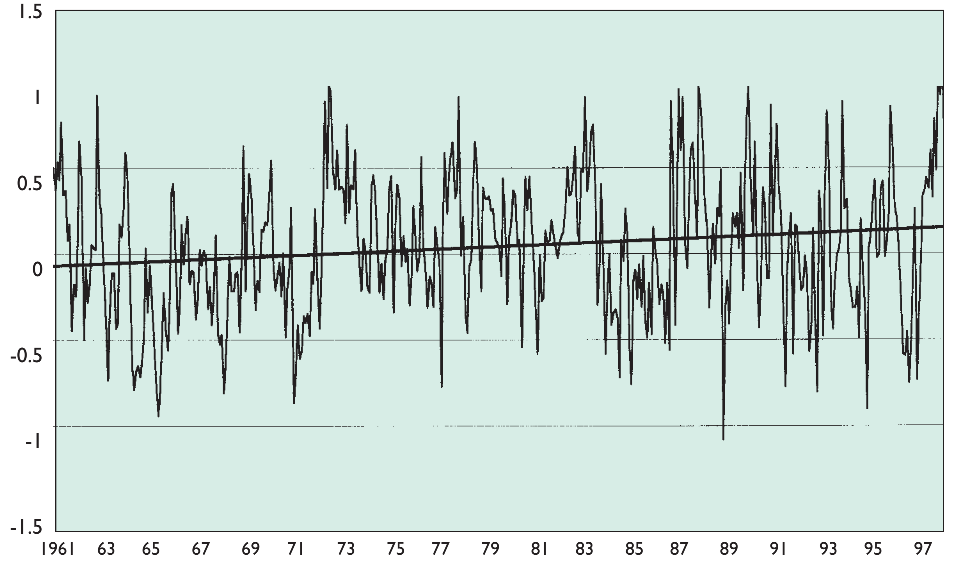
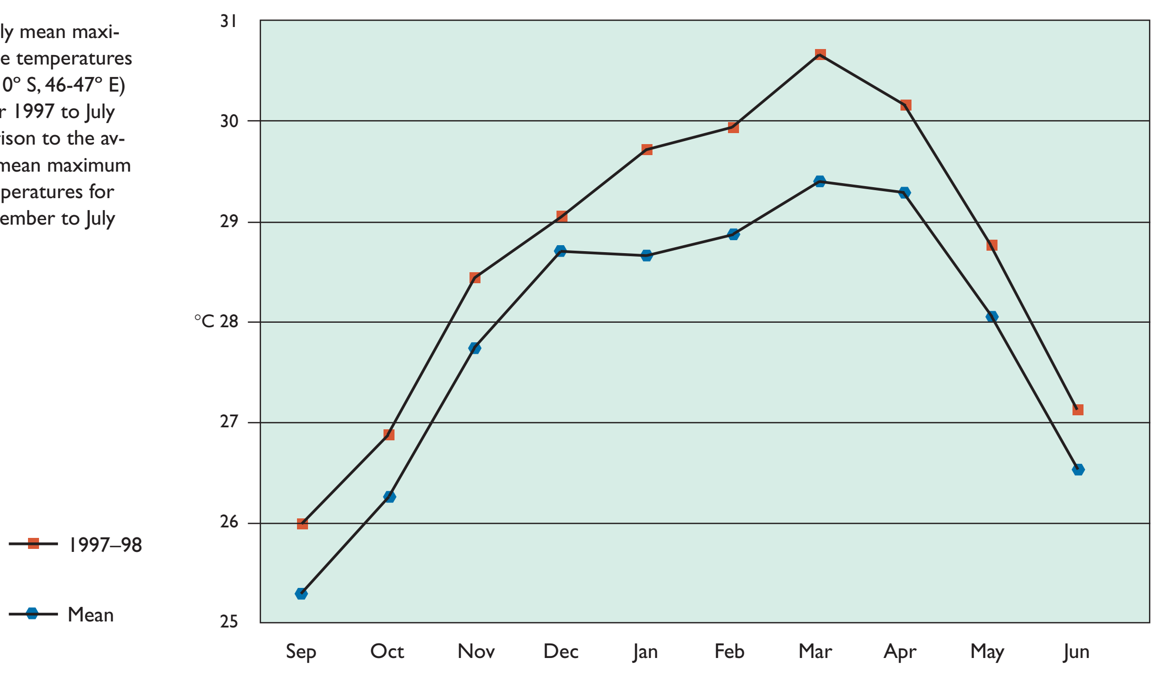


Figure 2. Monthly SST anomalies (°C) for Aldabra (5-10° S 45-50° E) 1961-1998, using 1961-1990 as the baseline.



representing a +1.31° C anomaly above the long-term mean maximum SST for that month. The +1° C anomaly persisted until April 1998 indicating a duration of almost four months (Figure 3). All temperatures recorded for the period leading up to the bleaching event and those following ranged from +0.5° C to +1° C higher than the long-term average of the monthly mean maximum temperatures (1961-1997).

Figure 3. Monthly mean maximum sea-surface temperatures for Aldabra (9-10° S, 46-47° E) from September 1997 to July 1998 in comparison to the average monthly mean maximum sea-surface temperatures for the period September to July 1961-1996.



In April 1998, close to the peak of the coral bleaching event, 41% of corals (coral coverage = 37%) were bleached or displayed recent mortality on the outer reef slopes (3 m -25 m) from the western to north-eastern sides of Aldabra (Cambridge Coastal Research Unit (CCRU), Southern Seychelles Atoll Research Programme – SSARP). Bleaching intensity in Aldabra was not as high as other areas in the southern Seychelles possibly because peak warming was 0.5° C lower than other areas in the region (Spencer *et al.*, in press). Bleaching and related mortality was primarily seen in the branching and tabular species of coral (*Pocillopora* and *Acropora*), and partial to patchy in most massive species (*Porites* and *Pavona*). Bleaching in some areas was confined to a single side of the coral colony. However, a high proportion of the massive species of corals displayed signs of previous mortality, as indicated by a thick layer of algal overgrowth and the presence of encrusting and boring invertebrates. As in other areas, soft corals showed high levels of bleaching and mortality. Although no quantitative data were gathered for the reef communities in the lagoon, extensive observations were made in all of the channels and in the western half of the lagoon. Most of the coral species found in the channels, with the exception of isolated incidences of branching corals, were observed to be alive and displaying no obvious signs of perturbation. Lagoonal patch reefs and individual heads of massive coral species displayed very limited bleaching. Distinctive species such as *Galaxea*, *Seriatopora*, *Acropora* and *Pocillopora* were completely bleached, particularly with increased distance from the flux of water within the channels.

METHODS

In November 1999, a series of seven permanent transects were established on the northern and western coasts of Aldabra Atoll (Figure 1). At each site quantitative baseline surveys of the corals and reef fishes were conducted along each transect. Primary transects were placed from a depth of 20 m to the reef crest at 3 m - 5 m. Secondary transects were established at 20 m and 10 m following the depth contour. The benthos at each site was surveyed using digital videography. Video imagery was analysed recording live and dead, hard and soft coral growth forms (branching, tabular, massive, encrusting and foliose). The occurrence of coral genera within each of these growth form categories was also noted, as well as incidences of mushroom corals, *Heliopora* (blue coral) and *Millepora* (fire coral). Additional observations of the substrate were noted in predefined categories: sand, rubble, rock, turf, *Halimeda* and other macro-algae.

Settlement plates were located at six sites, three on reefs around the outside of the atoll, and three within the atoll (Figure 1). Of the six tiles, two were oriented in the horizontal plane, two at approximately 45° to the horizontal, and two in the vertical plane.

Rapid visual fish surveys were conducted using species, number and size of fish recorded in a 2 m corridor extending out from either side of the transect, and vertically to the surface. Six total length categories were used: 0-5 cm, 6-10 cm, 11-20 cm, 21-30 cm, 31-40 cm and >40 cm. Each section was first surveyed for larger/conspicuous fishes, and then immediately re-surveyed for small/cryptic fishes. Fish size classes have been condensed to three grouped categories for this paper.

RESULTS

Coral and general benthic composition

There was a range of variation in percent cover of live coral (11% - 27% ± 6% (± 1 standard deviation)) and of dead coral (11% - 30% ± 7%). Live coral cover was strongly correlated with depth. From the shallow depths of 5 m - 10 m to >15 m there was an increase of 10% - 20% of live coral cover. Conversely, dead coral cover was higher in shallow waters (<10 m) at 44%, decreasing with depth (>20 m) to 10% - 20% of the coral cover.

There was a general increasing trend for cover of live and dead coral from east to west on the whole atoll (Figure 4), coinciding with decreasing levels of exposure to strong hydrodynamic conditions. The least exposed

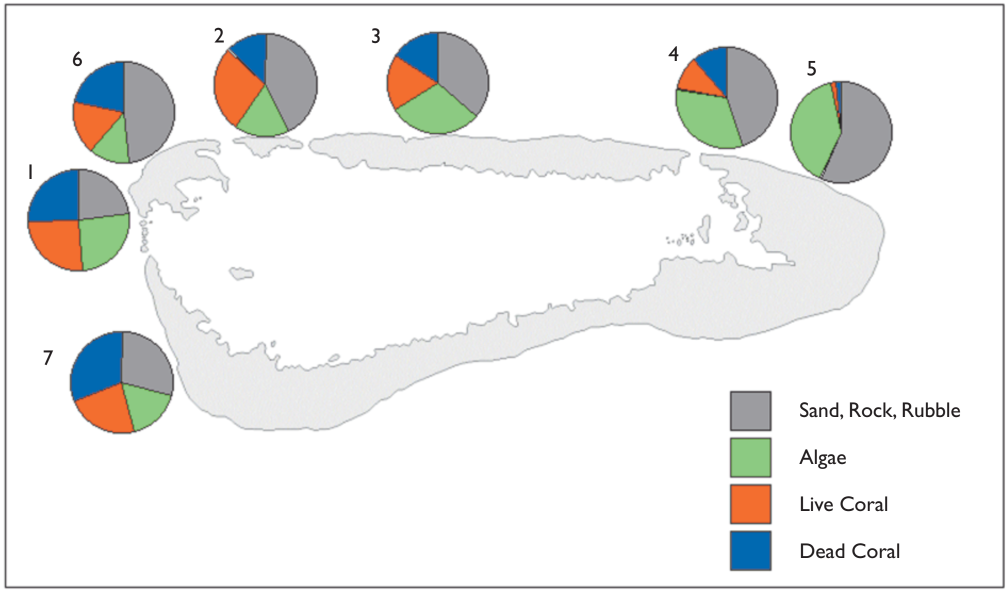


Figure 4. Calculated general substrate cover for the transect sites around Aldabra.

sites with the highest mean live coral cover were sites 1 (25%), 2 (24%), 6 (16%) and 7 (21%) (Figure 4). Note that site 5, at the highly exposed eastern corner of the atoll, was treated separately from other sites as it was distinctly different from the others surveyed, dominated by a high percentage of a sand, rubble and rock matrix (56%) and macro-algae, primarily *Halimeda* spp. (40%). Site 5 did have isolated coral heads but only 1.5% live coral and 2.0% dead coral of the total substrate cover.

Across the range of the sites examined *Halimeda* spp. was the dominant macro-alga, with localised high concentrations of *Dictyota* spp., *Caulerpa* spp. and *Lobophora* spp.. *Thalassodendron* spp. was present but was not considered to be a major component of the outer reef slopes. In most areas red encrusting coralline algae were observed especially where incidences of dead upright coral and coral rubble were present. On massive coral heads small patches of red encrusting algae were beginning to colonise in presumed bleaching related dead areas.

Soft corals (*Sinularia* spp., *Lobophytum* spp. and *Sarcophyton* spp.) were present as small isolated colonies and comprised no more than 5% of the substrate at any given location. Soft corals show a slight increase in abundance moving east along the northern coast.

The dominant live coral growth forms were massive, branching, encrusting and foliose at both shallow and deep sites. Massive corals were dominant at deep and shallow sites (63% and 45% of live coral cover, respectively). *Physogyra* sp. was the dominant massive coral on deeper transects (65% of massive spp.) but was virtually absent (3%) from 10 m depth sites. Other prominent massive genera were *Favia*, *Favites*, *Galaxea*, *Gardineroseris*, *Goniastrea*, *Leptoria*, *Lobophyllia* and *Porites*. The majority of branching corals found at both shallow and deep sites were *Pocillopora* and *Porites* spp., representing 20% of the total live coral cover. Live tabular coral species were not encountered at any of the sites surveyed but they were noted at the seaward edge of the western channels to the lagoon. Foliose (*Echinopora*, *Pachyseris*, *Turbinaria* spp.) and encrusting corals (*Montipora* spp.) were also common, but foliose corals were almost exclusively in deeper waters. *Millepora* and *Heliopora* were rare at all sites and depths, amounting to <4% of total coral cover. Extensive stands of dead *Millepora* were observed at transect 4 as deep as 23 m.

Although mortality was not as high in massive corals as in others, there was evidence of tissue death which was spatially patchy on individual coral heads. In some cases live tissue remained on the sides or underside of the colony. In sites where branching corals have survived it is often the case that the colony has suffered only partial mortality. It is noteworthy that plate and fine branching corals were not abundant at any of the sites visited, probably because of relatively high hydrodynamic activity present in the vicinity of the atoll. It is also possible that with the onset of mortality the structural integrity of these growth forms, would be rapidly compromised in the first incidence of high wave action.

Sites 1, 2, 4 and 7 were previously surveyed by the SSARP in April 1998 close to the peak of the coral bleaching event (Spencer *et al.*, in press). At 10 m depth, the majority of the bleached corals in 1998 had suffered subsequent mortality, translating into an increase of 22% in dead coral cover by November 1999, with minor increase of normal live coral cover (+7%, Figure 5a). At 20 m depth, the proportion of bleached corals in 1998 was higher than at 10 m, however rates of recovery were much better with an increase of 52% of live coral coverage, from 14% in 1998 to 66% in November 1999 (Figure 5b).

Figure 5a. Comparison of % coral cover in Aldabra at 10m depth at sites 1, 2, 4 and 7 in April 1998 and November 1999 (note no bleached coral in 1999).

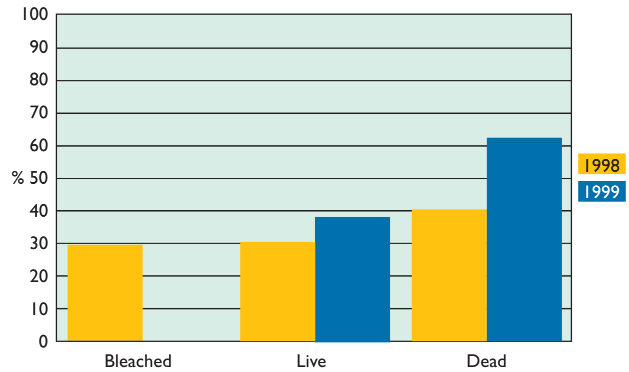
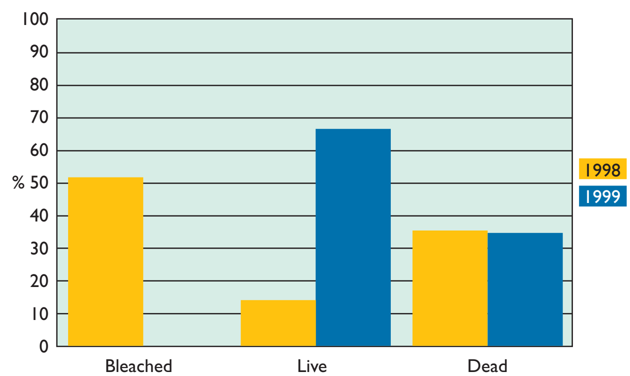


Figure 5b. Comparison of % coral cover in Aldabra at 20m depth at sites 1, 2, 4 and 7 in April 1998 and November 1999 (note no bleached coral in 1999).



Fish communities

Fish surveys gave quantitative information on 164 species representing 27 families. An additional 48 species and six families were recorded in qualitative surveys, giving a total of 212 species for November 1999 at Aldabra.

The densities of fishes ranged from 352 per 100 m^2 from 33 species and 16 families at Site 5, to 7162 per 100 m^2 from 90 species and 23 families at Site 6 (Figure 6).

The density and species of fish at Site 5, near the eastern end of Aldabra Atoll, were clearly different from the six other sites along the northern shoreline and at the western end of the atoll. Disregarding site 5, there was little variation in families of the fish in the surveys at the other sites. The lowest density of fishes at Sites 1-4, 6, and 7 was 645 per 100 m^2 from 70 species at Site 3. The large differences in the density of fish resulted from large schools of a few species from the families Serranidae (groupers and basslets), Apogonidae (cardinalfishes) and

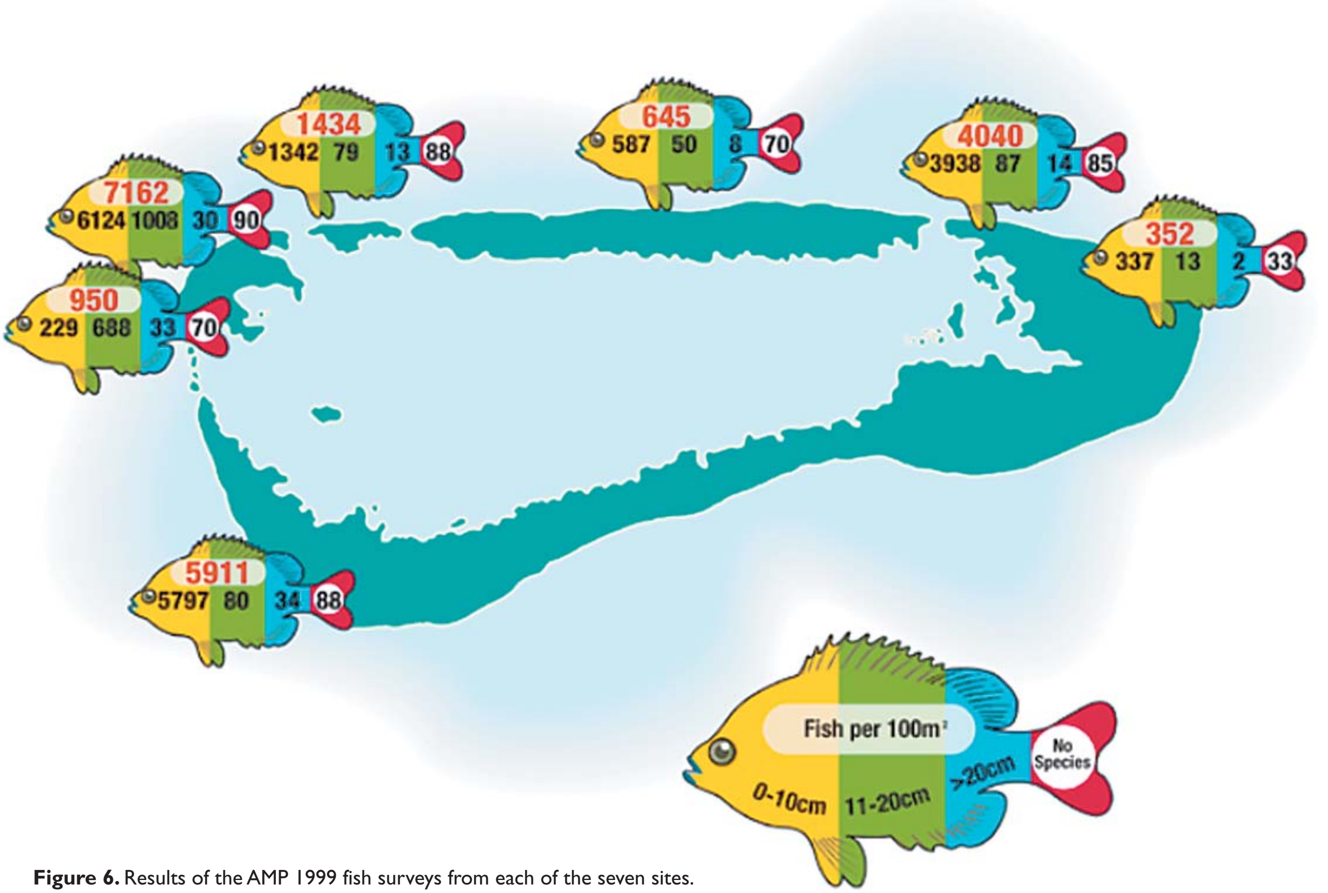


Figure 6. Results of the AMP 1999 fish surveys from each of the seven sites.

Pomacentridae (damselfishes). When the fish density exceeded 1000 per 100 m^2, these three families accounted for between 80% and 94% of the fishes.

The sizes of the fishes in the transects at Sites 2-7 were dominated by those in the 0 cm -10 cm total length category, which contained from 86% - 98% of the fishes surveyed (Figure 6). At Site 1, 72% of the fishes were in the 11 cm -20 cm total length group, due to an unusually large school of Caesionidae (fusiliers) that contributed 63% of the total fishes in this size group. The abundance of fishes in the 0 cm -10 cm category was caused primarily by large numbers of small-sized species, and secondarily by juvenile life-stages.

No significant correlations were found between overall measures of fish community structure (density and species number) and habitat variables (longitudinal gradient, live coral cover and dead coral cover). However, the families Chaetodontidae, Labridae, and Serranidae each have several species that are commonly associated with live corals and habitat structure formed by erect dead corals. Significant positive correlations were found for a) species richness and density of Chaetodontids with respect to live coral habitat (r^2 = 0.88 and 0.71, respectively), and b) density of Labrids (r^2 = 0.61) with live coral cover, and c) species richness of Serranids (r^2 = 0.59) with live coral cover.

Partial mortality of *Favites flexuosa* and *Acropora* spp. colonies at 20 m depth with blacktip grouper (*Epinephelus fasciatus*) and evidence of the macroalgae *Halimeda* spp. in the lower left corner. This photo illustrates well the variability of mortality on the reefs of Aldabra.

NASA space shuttle image (Atlantis - STS044-082-057) Aldabra Atoll, Indian Ocean - 29 November 1991.

DISCUSSION

The 1998 coral bleaching event had pronounced effects upon the reef complex. Shallow corals suffered the greatest mortality and lowest recovery, perhaps due to the long period of exposure to high and increasing SSTs as shown in figure 2. Although bleaching levels were greater for deeper corals in 1998, recovery was higher and mortality lower than for shallow corals. It may be that penetration of warmer SSTs to deep water sites took longer and may have had a shorter duration than in the shallows. Shorter duration and magnitude of anomalous temperatures may have led to high recovery at depth (see Brown, 1997), and the timing of the onset of bleaching may be important. The 1998 survey was conducted at the end of the anomalous period, recording high bleaching cover. This suggests that the bleaching may only have started at the end of the three-month period of anomalous SSTs.

Coral communities in Aldabra may have undergone a shift in species composition as a result of the bleaching, which is likely due to susceptibility to bleaching. *Physogyra* sp., abundant between 15 m and 25 m, and *Pachyseris,* abundant below 20 m - 25 m, are now dominant corals. The dominance of *Physogyra* sp. across all deep water sites suggests a particular robustness of this species to the perturbation of the ambient environment. In areas where currents were strong, particularly the channel exits, *Tubastraea micrantha* is abundant.

There is no evidence that the death of large numbers of corals has led to an explosion of macro-algae, as these were not abundant on dead coral surfaces. Dead corals had been covered or partially covered by red encrusting algae which not only cement and maintain the structural integrity of the coral, but may also create suitable substrate for coral larval settlement. It is essential that the substrate be fixed and/or consolidated for coral development to occur.

The recovery of reefs at Aldabra from the bleaching event is underway. In areas where coral had died, there was evidence of acroporid and pocilloporid recruits up to 3 cm in diameter. Many corals suffered only partial, not full, mortality, and the live patches will be monitored to assess whether these will re-grow over the dead areas. With live coral colonies on the deeper outer slopes, in the lagoon and channels local coral larval recharge may occur. Future recovery of settlement plates put out during the surveys will provide insight into coral recruitment levels in and around Aldabra, and to further monitor and understand the recovery process.

The diversity of fish species and families found in coral reef ecosystems is indicative of the productivity or health of the system (Sale, 1991). The vertical relief and three dimensional complexity of the reef habitat provided by both live coral and erect dead coral structures is not only crucial for fish survival, but is also an aggregation attractant for reef fishes. This habitat complexity is often positively correlated with the diversity of fishes on the reef (see Ebeling & Hixon, 1991; Sebens, 1991; Williams, 1991; Turner *et al.*, 1999). Alterations and reorganisations of the reef structure following a bleaching related mortality event may, in turn, have varied spatial affects through a range of temporal scales on non reef dwelling fishes in the system.

There is often a time-lag in the responses of reef fishes to the loss of live coral habitat (Turner *et al.,* 1999). The November 1999 surveys may have captured essentially pre-bleaching event diversity of the fishes at Aldabra Atoll. This makes the quantified baseline information from these surveys exceedingly valuable for long-term monitoring of the natural recovery of this remote coral and reef fish ecosystem. Future surveys will be critical to fully understand the responses of the ecosystem, and for developing a comprehensive management plan for this and similar coral reefs.

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SECTION II
Environmental Monitoring and Climate Change

Sea surface temperature in the western and central Indian Ocean

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CORDIO – East Africa, Mombasa, Kenya

DATA ACKNOWLEDGEMENTS:
Chagos - C. Sheppard; Kenya: 92-94, 97-98: D. Obura. 95-97: S. Mwangi; Maldives – S. Clark; South Africa – L. Celliers; Zanzibar – C. Muhando. SST Anomaly map: NOAA.

LONG TERM RECORDS
Annual peaks in sea temperature occur in the first half of each year, as the sun moves northwards after heating the sea-surface during the southern summer. Long term records in the central Indian Ocean indicate a distinct warming trend (Chagos, Maldives) of almost 1°C over the last 30 to 50 years, leading up to the highest recorded maximum during the El Niño Southern Oscillation in early 1998.

THE EFFECT OF EL NIÑO
The sea surface temperature anomaly map shows the position of the high in April 1998, having moved northwards from a position off Madagascar and Mozambique in February, and finishing in the Gulf of Aden in May.

SHORT TERM RECORDS
Monitoring of water temperatures on reefs started in East Africa in 1992 using hand-held thermometers and subsequently automated temperature loggers. Initially, records were used to document seasonal changes, extending now to daily and even hourly changes (Kirugara, this volume). Distinct daily patterns of warming during the day and cooling at night are the norm, superimposed on tidal and seasonally-influenced changes, and local topographic and current effects.

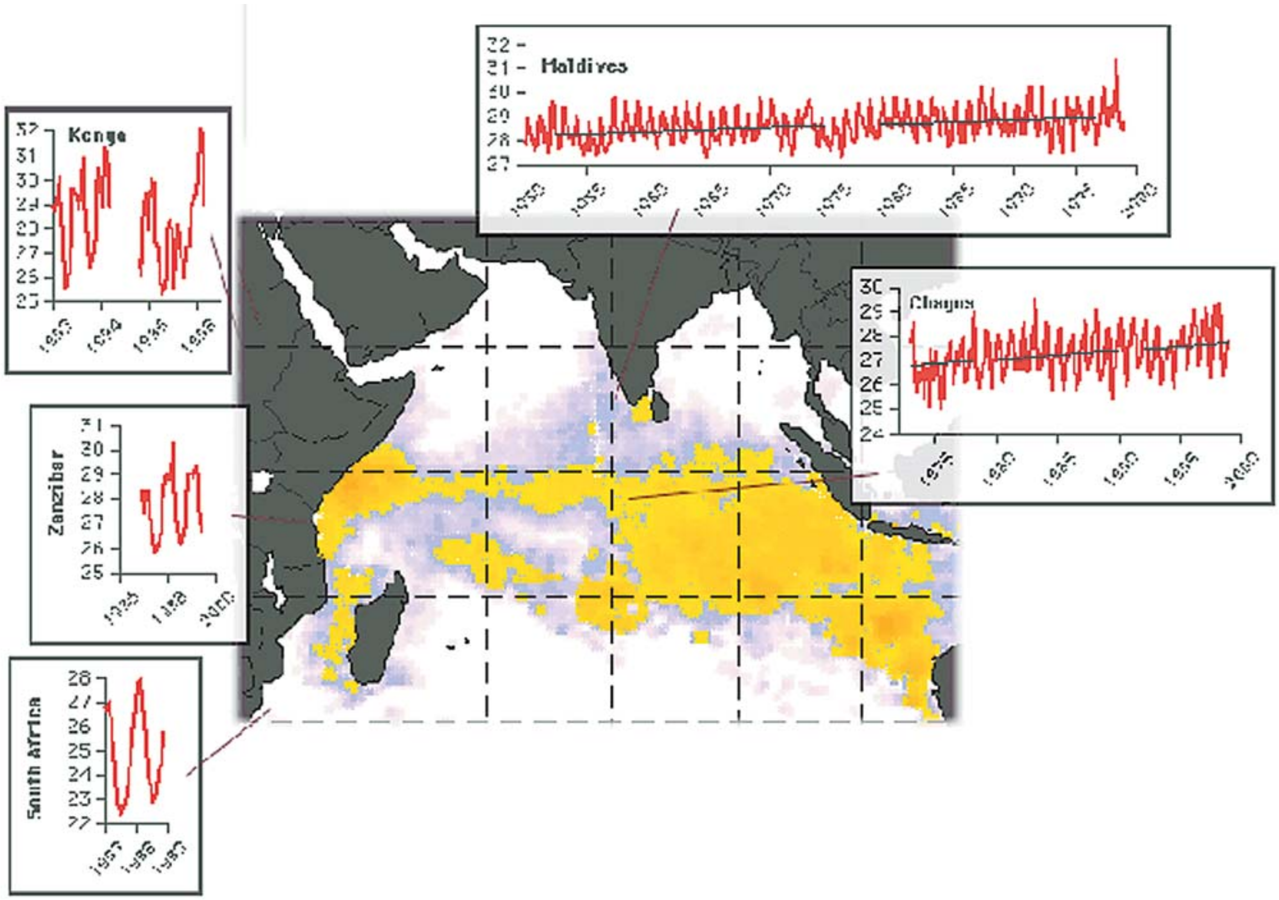
METHODS
Sea-surface temperature measurements are now being taken by a number of different methods, including spot measurements using thermometers during field visits, automated measurement by *in-situ* temperature loggers, ship-based temperature measurements and satellite remote-sensing of sea-surface temperature. With increasing variety of methods and number of locations monitored, standardization among methods will become more important, to account for differences in resolution in space and time, depth of measurements and daily variations (e.g. see McClanahan, this volume).

LOCAL FACTORS THAT INCREASE TEMPERATURE
Research on a semi-enclosed lagoon in Kenya (Kirugara, this volume) has shown that maximum heat transfer to lagoon waters occurs due to the coincidence of spring low tides with maximum sun height at midday. Exposed reef surfaces heated during spring lows transfer their heat to flooding water resulting in a distinct temperature peak during the flooding tide that persists for several days. The effects of this regular temperature increase on local coral adaptation, and conversely, on the absolute magnitude of an El Niño-related high temperature anomaly, are likely to be important.

LOCAL FACTORS THAT DECREASE TEMPERATURE

Several factors have been identified that might reduce the absolute magnitude of the El Niño-related high temperature anomaly. Understanding where and how these factors might protect sites from bleaching may be important for management and conservation.

1. Upwelling of cooler water due to continental shelf and/or reef bank topography
2. Temperature loss through exchange and mixing of water along fore-reefs and in lagoon channels, and potentially with the air in shallow bays.
3. Cyclones causing reductions in water heating through cloud cover, and mixing with deeper waters due to wind.



Temperature and water exchange in a semi-enclosed lagoon, Bamburi, Kenya.

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INTRODUCTION

In the context of high sea-surface temperature anomalies causing widespread coral mortality, this study investigates the heat balance of a semi-enclosed coral reef lagoon system and any additional contribution of UV radiation as a trigger for bleaching. The study lagoon is situated north of Mombasa Island, along the Nyali-Bamburi-Shanzu coastline, at 4°0'01" S, 39°0'44" E (Figure 1) and has a surface area of 3.75 km^2 and 12.5 km^2 during spring low and high tides respectively. It consists of three topographic features: the shallow back-reef lagoon, the 300 m wide, 7.5 km long reef crest that is exposed during low tide and shelters the lagoon from oceanic swells and the relatively deep central longitudinal channel that collects all lagoon water at spring low tide. The mean depth of the lagoon does not exceed 0.7 m and the width varies between 1.5 km and 2.0 km at MSL. The main channel system connects the lagoon southwards to Nyali lagoon through a 250 m wide, 5.8 m deep point. Towards the northern end of the lagoon is a shallower (2.5 m) channel system connecting the lagoon to the mouth of Mtwapa creek (Kirugara *et al.,* 1998).

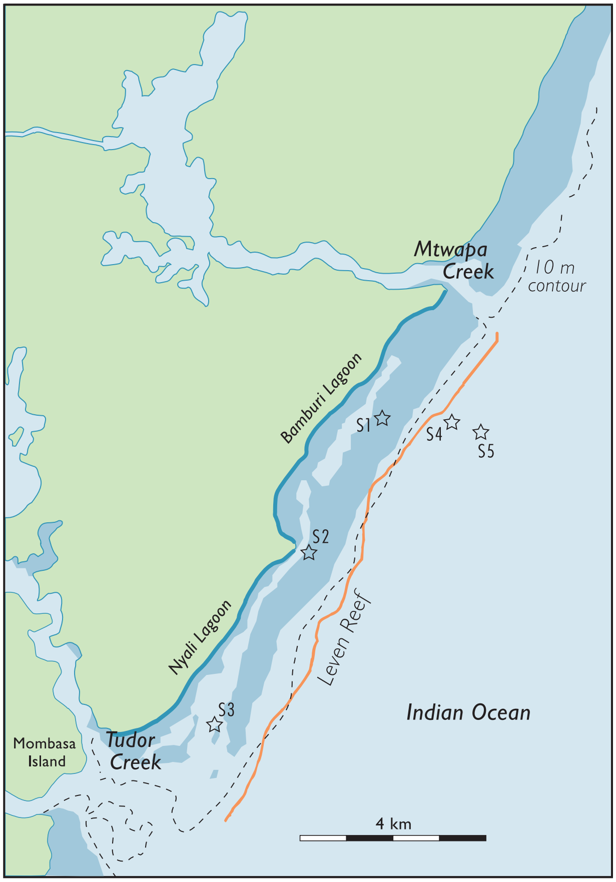
The lagoon water circulation has been described previously and modeled by Kirugara *et al.* (1998). Lagoon water exchange is driven by the wave-induced flow that is dependent on the degree of the reef submergence by tide and wave conditions, the characteristics of the incoming swell and the difference between the oceanic and lagoonal tidal levels. During spring tides, more than 80% of lagoon water is exchanged at each tidal cycle. In neap tides, unlike many other coastal marine areas, the continuous pumping of water over the reef (wave-induced flow) into the lagoon maintains a higher lagoon sea level compared with the typical oceanic level. This forces a simultaneous exit of lagoon water through the channels modulated by the tidal regime ensuring that more than 60% of lagoon water is exchanged during every tidal cycle. These mechanisms ensure the efficient flushing of lagoon waters during 1 - 2 tidal cycles.

The study monitors the following biophysical factors, integrating them over space and time: salinity and temperature inside the shallow lagoon and adjacent oceanic waters at 10 m to 30 m depth, solar radiative heat flux at the sea surface, and attenuation of this radiation into the lagoon and ocean, possibly differentiating UV and PAR. With this information an attempt will be made to estimate the heat balance for the lagoon on a long-term basis, taking into consideration the different monsoon periods.

METHODS

Three lagoon sites and two 'oceanic' sites at 10 m and 20 m outside the reef were used. Temperature and conductivity (salinity) measurements were recorded at 10 minute intervals for 90 consecutive days (18th September-14th December 1999) using four bottom-mounted conductivity- temperatures meters (Type Hugrun) (Figure 1). One bottom mounted pressure gauge was de-

Figure 1. Map showing the instrument deployment site and simplified bathymetry of Bamburi lagoon. The dotted area is more or less exposed during extreme low tide. The outside 10 m contour is indicated. Note the channel in the southern end, connecting the lagoon.



ployed at 10 m at the oceanic site to record pressure and temperature, logging data every 10 minutes. An Aanderaa Automatic Weather Station 2700 was securely erected at KMFRI, 2 km from the study site, equipped with the following sensors: short wave radiation (0.3µ - 2.5µ), reflected long wave back radiation (0.3µ - 60µ), sunshine duration, wind speed and direction, air pressure, temperature, relative humidity and rainfall. This equipment simultaneously logged data from all the sensors at 10 minutes interval also. The entire raw data set was processed into half-hourly means and later smoothed with a moving average filter to obtain average measurements over a 24 hourly window. Further data analysis will be carried out in future.

RESULTS

Figure 2a shows a short-term temperature series in two shallow lagoon sampling sites, showing strong diurnal and semi-diurnal signatures in temperature variability. The diurnal trends are in response to the daytime heating and night time cooling while the semi-diurnal trends respond to the tidal phase. The tidal effect including the spring-neap variability is masked in this figure but is important because flooding brings cooler waters into the lagoon while ebbing water removes the excess lagoon heat and maintains a thermal balance (Pugh & Rayner, 1988; Mwangi *et al.*, 1998). Generally, comparison of the two lagoon stations reveals that the waters are well mixed and homogenous with no pronounced salinity-temperature fields within the lagoon stations.

Figure 2b shows a different record of the half-hourly oceanic temperatures at 10 m and 20 m for two days. This figure highlights the possible existence of a shallow dynamic thermocline between 10 m and 20 m. The thermocline seems to be unsteady and the water becomes unstratified within short duration. It will be interesting to monitor this feature to establish whether the thermocline is maintained for long periods and its possible ecological consequence particularly as related to temperature induced bleaching.

Figure 3 shows a 20 day comparison of oceanic temperatures at 20 m depth with lagoon temperatures using a 30 minute average. The oceanic station showed a relatively stable cooler temperature profile, and no influence of tides. However, the lagoon temperatures increased from 28^{o}C to 30^{o}C during the same period. This difference between oceanic and lagoon temperatures, of the order of 2^{o}C is consistent with that suggested by Pugh & Rayner (1981) for tropical lagoons in cloudless, windless weather.

Figure 2a. Time series temperature records for 11 consecutive days from a) two lagoon stations showing strong diurnal and semi-diurnal variation.

Figure 2b. Time series temperature records for two consecutive days from two oceanic stations indicating the possible existence of a shallow dynamic thermocline.

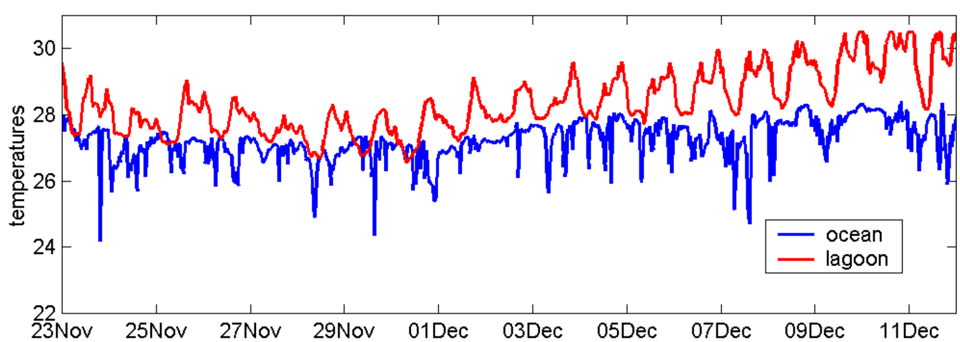


Figure 3. Twenty day half-hourly time series temperature record of oceanic temperatures (20 m) and lagoon temperatures (< 5 m) illustrating that oceanic temperatures respond to the prevailing coastal current and lagoon temperatures exhibit fairly strong diurnal and semidiurnal trends.

Figure 4. Ninety day consecutive record of water temperature, air temperature, and solar radiation collected from the lagoon beginning 18th September 1999.

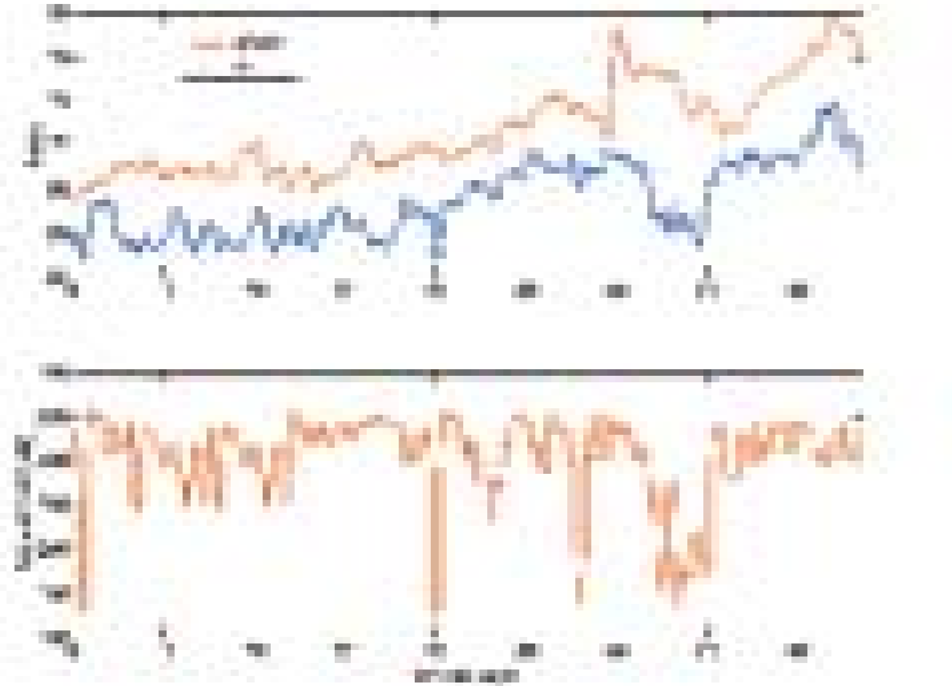


Figure 4 shows three month records of water and air temperature and solar radiation beginning on the 18th September 1999, the period of transition into the North East monsoon season. These data show a daily insolation of about 300 W/m^2 (clear skies) on most days and slowly increasing air and water temperatures throughout the study period. During the same period a steady decline in wind speed from daily means of 4 m•s^{-1} to 2 m•s^{-1} was recorded.

DISCUSSION

Substantial amounts of data are now available to begin heat flux calculations based on the work of Mahongo (1997) at Chwaka Bay, Zanzibar. These calculations will show long term trends of incoming radiation intensity (flux). An additional feature that could potentially be measured would be to distinguish between PAR and UV wavelengths, as these have different roles in coral biology and bleaching dynamics. Parameters to be determined will include whether UV(A, B or C) flux is a function of the total radiation or is constant, and their penetration characteristics at different depths. The extinction co-efficients for UV radiation are known for clear waters, but not for lagoon waters.

An important factor for the heat balance in this lagoon, was that spring low tides always coincide with maximum insolation between 09:00 hours and 12:00 hours (Kirugara, in press), resulting in maximal heating of exposed reef surfaces and interstitial waters. Influx of the flooding tide transfers much of this heat into the lagoon, such that lagoon temperature records show a peak during and immediately after spring low tides.

ACKNOWLEDGEMENTS

I would like to thank J. Kamau, Z. Masudi, J. Kilonzo and S. Ndirangu of KMFRI for assistance in the field, and F. Schollinger (Barracuda Diving Club) and K. Mbambanya (Kenya Wildlife Service) for assistance with transport and access to the study sites.

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Initial considerations on methods to survey reef status and recovery

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METHODS MUST MEET THE OBJECTIVES OF THE STUDY

Objectives. Any method applied to assess the status of impacts on, or recovery of coral reefs must meet the specific objectives of the study. For example, a survey of near shore reefs designed to assess the damage caused by sedimentation from a land reclamation project is unlikely to provide adequate data on the extent of coral bleaching. Further, a study of settlement of coral larvae should be designed so that groups of settlement plates are brought up periodically over a period of up to 1.5 years, and not all after 6 months, when little settlement may have taken place. Thus, the objectives should be SMART, i.e. Specific, Measurable, Area bound, Realistic and Time bound. There is a tendency in studying coral reef degradation to advocate 'keep it simple' or KIS methods, based on a general survey technique which is applied to all objectives. While such methods are valuable for reef assessment and comparison of reefs between nations or across wide areas (e.g. ReefCheck), they rarely have adequate rigour to answer specific problems relating to reef degradation and recovery at a location.

Scale. Methodologies must be applied at different scales to address objectives. For example, the effects of major environmental events such as the mass mortality following the 1997/98 bleaching, or monitoring recovery of degraded reefs. It is important that surveys across wide areas are conducted to give the 'big picture' of the impact of a natural environmental event. For example, a survey of the reefs around the main island of Mahe in the Seychelles may provide a false indication of impact from bleaching in the Seychelles, since the coastal waters are often turbid due to activities on land, compared with the clear waters around more isolated islands. Corals in turbid waters may not have been exposed to strong solar radiation at the same time as elevated sea temperature, unlike the corals in clear water. Therefore, surveys conducted on reefs around all islands in the archipelago are required to assess the real impact. Similarly, the number of coral recruits can be assessed over a large area if coral colonies ranging in diameter between

Figure 1. Three divers conducting rapid site survey video and visual census at Cousine Reef, Seychelles.

Figure 2. Settlement plates for corals, Harbour Patch Reef, Seychelles.

1 cm and 10 cm are being sought and recruitment is low. However, if assessing settlement on natural surfaces, detailed searches at small scales may be required for corals smaller than 1 cm in diameter, and settlement plates with artificial surfaces that can be brought up for detailed analysis will be required for the identification of settled species.

Rapid survey assessments (RSA) which provide a broad evaluation of the current status of many sites across many locations are suitable for large scale studies, and the methods allow large areas (e.g. one hectare) at each site to be surveyed. The large sample surveyed at each station overcomes the problems of smaller scale patchiness, zonation and need for replication at stations within sites. A different methodology is required when monitoring reefs for change, where surveys are repeated over time. Such methods require a finer scale of assessment over a smaller area and at selected sites only. Often, the impact being assessed is on-going and, if anthropogeneic (e.g. land infill), may have a point source, in which case control sites must be adopted. By using controls, the underlying natural changes can be assessed (e.g. suspension of sediments by a storm, or rise in sea-surface temperature).

METHODS SHOULD REFLECT REEF STRUCTURE AND TAXONOMIC SCALE

Methods to survey reef structure are often designed around surveying hard coral colonies even though these may only cover 30% - 40% of a reef. Carbonate and algal components are often neglected resulting in under estimations. There is a requirement to adapt methodologies on reefs where algae or even soft corals dominate. Thus, it is advisable to record reef development, benthic cover and composition of the substratum. DeVantier *et al.* (1998) have developed such methods, and adapted versions of these are used by Turner *et al.* (this volume).

Due to limited expertise, surveyors often collect data at the level of life form (e.g. hard coral, soft coral, macro-algae). Such data are valuable and can often indicate to a manager that a reef community is changing. However, more subtle changes, such as the shift in species composition from branching hard corals to massive and encrusting species following a bleaching event, may be hidden. Clearly, the hard coral record can be usefully subdivided into coral morphology (e.g. branching, tabular, massive). However, shifts in the diversity of families, genera and species require expertise to record at higher taxonomic levels. Managers may not immediately be interested in this data, but scientists will need information at this finer scale to understand the pattern of change and to communicate the causes to the manager. For instance, the responses of corals to an impact such as sedimentation (Rogers, 1990) or temperature (Jokiel & Coles, 1990), or solar radiation are often specific.

EXPERTISE AND TRAINING

A compromise is often required in designing methods that meet the survey objectives and meet the level of expertise of the surveyor. Again, there is a tendency to oversimplify the methods rather than increase expertise. Training should be provided, and it is often easier than perceived to increase identification skills from life form to genera and common species.

COMPILATION OF STANDARD METHODS

There are many proven methodologies compiled for the survey of coral reefs and associated ecosystems e.g. CARICOMP's *Level 1 Methods Manual*, (CARICOMP, 1994) and ASEAN Australia Living Coastal Resources *Project's Survey Manual for Tropical Marine Resources* (English *et al.*, 1997). There is certainly no need to return to first principles, but this does not mean that new methods should not be developed. There is a need to be flexible and adapt methods to suit the geographical areas and reef structures under study since, while widely applicable, the earlier examples were designed for the Caribbean and ASEAN regions respectively as standardised methods to allow international co-operation. An example of adapted methods are those of the COI Regional Environment Programme's *Methodology Manual for the South West Indian Ocean* (Conand *et al.*, 1997). New methods have been devised for addressing the specific problems of bleaching and reef degradation in the Indian Ocean by members of CORDIO (this volume), and there is much value in making these widely available to teams within the region. The compilation should include guidance on the use of methodologies in meeting objectives, and should be produced in a cheap hard copy format for dissemination at the ranger level.

INTER-CALIBRATION OF SITES

Different interpretations of observations can be misleading (e.g. varying the size used to classify a coral recruit). Greater standardisation and clarification in interpretation can be achieved through training, and by survey teams overlapping some sites (e.g. Turner *et al.*, survey and Engelhardt surveys of the Seychelles, this volume). Well geo-referenced and labelled underwater photographs, and especially video records represent good archive information allowing comparability between different surveys and times.

DATABASES AND GEOGRAPHICAL INFORMATION SYSTEMS

Databases are required to store the large amounts of survey results systematically and allow the efficient retrieval of selected data in response to specific questions. There are many existing databases (e.g. ReefBase, FishBase) and hence, there is usually no need to design a database from first principles. Although existing databases are often useful for assembling data on an international scale for use by networks and global policy makers, they may not be helpful in formatting data to answer the specific objectives of a study, or to meet specific management requirements in a locality. Thus, there is often the requirement to produce smaller databases from first principles, but thought should be given to how these can be merged with larger databases in the future.

Geographical Information Systems (GIS) are extremely valuable databases that allow the overlaying of geo-referenced information (e.g. reef biotope and fish community composition and fish catch). GIS are usually more complex than databases and require greater expertise, although a simple laptop computer, base map or chart, scanner and software is often all that is required. GIS has a greater value when dealing with specific management areas and issues. Particular strengths are the analytical and predictive capabilities, often thought of as 'what if scenarios' such as: *Which biotopes will be affected if the water volume from a river is doubled? Where can sand be mined from this lagoon given a 100 m buffer around all sensitive biotopes?* Further, GIS can incorporate spatially related photographs and video, so that clicking on a point on a computer presented map can reveal photographs of coral communities at that point, or a video sequence. Such visual information can be of great value to managers and can influence the decision making process. GIS has been used to effect in zoning the coastal zone of Socotra (Turner *et al.*, 1999), and in presenting coral bleaching results in Mauritius (Turner *et al.*, this volume).

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Measuring change and recovery in reef ecosystems

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INTRODUCTION

Assessing environmental impacts and monitoring ecosystem recovery requires an understanding of the underlying spatial and temporal changes that have occurred. There are three main questions to be answered with specific reference to the Indian Ocean and the current status and future of coral reef ecosystems: i) what are the effects of widespread coral mortality going to be in the Indian Ocean, ii) how long will it take for reef ecosystems to recover, and iii) will 'recovered' systems have the same structure and functional integrity? Research into these questions must bear in mind that most measurements to be made are points on a long-term trend. Although there may be another disturbance event similar to that of 1998, its occurrence (if it occurs) should not negate the work, but provide further data points which contribute to the understanding and establishment of long-term trends.

While direct changes in structure can be measured relatively simply, more advanced study will be required to assess changes in functional relationships. For instance, does the 'recovered' reef with its different coral community support a different fish community, and will the productivity of this reef be different? One-off measurements must be put into context. Several examples exist of measures which are difficult to interpret, for example, measures of total fish catches, coral cover or population recovery: As a fish species is depleted, even commercially extinguished, it may be replaced by other species, the *total* catch remaining the same. A 'pristine' coral cover value of many systems is about 50%, which rises with increasing long-term stress, then decreases through the 50% value again before reaching very low values (Figure 1). Thus, only very low percentages might be indicative of degradation, by which stage the quantification of coral cover may be of little use. The problem of recovery is to ask at what level a population is considered to be 'recovered'? Figure 2 shows a hypothetical decline of a population that has progressed over 25 years. The sag following the 1998 event is recovered from, but to which long-term trend, (a) or (b)? This is a complication from the 'shifting baseline syndrome'.

Sample size is one of the main difficulties in measuring change and recovery. It should be noted that if significant differences occur between two samples (e.g. by

Figure 1. Coral cover (from Red Sea and Arabian Gulf examples) of changing coral cover with increasing stress (Sheppard *et al.*, 1992).

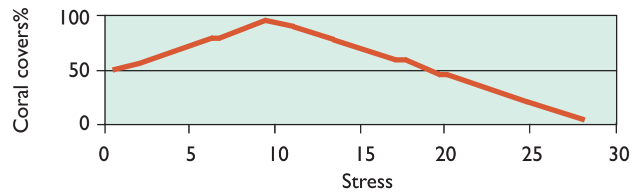
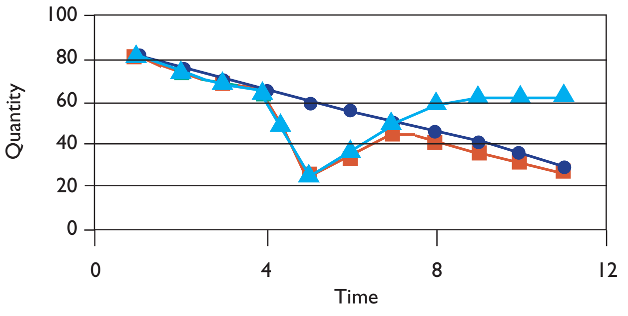


Figure 2. Two possible recoveries from a crash, e.g. a fisheries or coral cover crash, occurring at time marked by arrow.
Straight line: continuous downward trend over many years (i.e. with no biological crash).
Red line: recovery from crash to reduced and declining level.
Light blue line: assumed recovery to level at pre-crash state, which might be assumed if the overall downward trend is not known. Example from coral cover in Chagos archipelago (Sheppard, 1999).



t-test) then the sample size to determine this is more or less irrelevant. However, if the objective is to prove no difference, then samples may need to be quite large – typically 80 – 100 replicates. Large samples can be achieved (and possibly can *only* be achieved) by pooling data from several sites, i.e. by working at a regional level. Thus, this regional perspective and approach is critical.

The following proposed general projects may be largely immune from the first three complications, and all are contingent on adequate sample sizes. It is important to note that the majority of these objectives can be achieved at low cost, and that the expertise to carry them out can be attained through straightforward basic training exercises. The multi-scale approach, both spatial and temporal, to construct long-term trends will provide a background against which future disturbances, changes to the environment, and indeed recovery, can be assessed and monitored.

A. TIME SERIES DATA

The objective is to establish long-term trends over as large as possible spatial scales. These projects access data and methods common to other sciences, and in many cases may only need informal contacts and collaborations with appropriate researchers and/or institutions.

1. Remotely sensed images

Images need to be processed for the extraction of biological information (quantified habitat cover data) to make synoptic cover measurements which not only include the reef, but the adjacent systems as well (macro-reef system, i.e. integral seagrasses, mangroves and sand). Archived remotely sensed imagery provides measurements from years past to establish longer time series.

2. Meteorological data

Acquisition of long time series of *in situ* data from a variety of national and international sources, and analysis of meteorological data to establish trends.

3. Fisheries

Assessing broad scale ocean productivity through the use of remote sensing.

4. Land use changes

Establish trends in development, tourism and reclamation to assess long-term effects on coastal ecosystems.

B. SPECIFIC REEF MEASUREMENTS

The objective is to record and monitor *changes* in reef composition through a range of techniques. These methods are consistent with many existing monitoring schemes, and can be added or revised to be incorporated with minimal additional work while maximizing capture of long-term trend data and regionalization of research and monitoring.

1. Cover and damage

Recording cover by major biotic and abiotic groups (i.e. total coral, soft coral, unconsolidated rubble, sand etc.) to document ecological and substratum characteristics. Identification to genus/species level of reef constituents, measurements of size class frequencies of coral colonies and recording the proportion of damaged colonies.

2. Erosion of reefs and reef-protected shores

Measurement of beach widths to LWST from fixed datum points with a monthly sampling interval as well as measurement of algal ridge widths, percent cover of red coloured algae and the down-slope extent.

3. Coral recruitment and monitoring

Quantification of new coral recruits (colonies between 1 cm and 10 cm diameter/long) using standardised counting methods and photography of recruits at 3 m depth. Monitoring of tagged coral colonies for growth, damage and disappearance.

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SECTION III
Thematic/Research Reports

Socio-economic assessment of the impacts of the 1998 coral reef bleaching in the Indian Ocean: A summary

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INTRODUCTION

Coral reefs are a vital resource to many areas of the Indian Ocean. Coastal populations are continuously increasing (Table 1) and relying on this resource as the basis of the economy. Across the region, the two common socio-economic reef based activities are fisheries and tourism. For local subsistence fishermen, reef fisheries often represent their only livelihood. Degradation of coral reefs will first impact the reef fishery and subsequently, the local fishing community. Tourism also is often heavily dependent on coral reefs as the main attraction.

The countries of the Indian Ocean vary both physically and socio-economically (Table 1). The size of a country, the area of coral reefs, the coastal population utilising the reefs and the wealth of the country are all

Table 1 Physical and socio-economic indicators of each country participating in CORDIO.

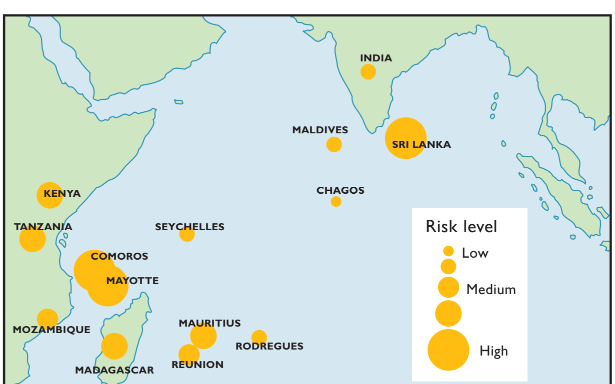
Country	Length of coastline Km	Coastal pop density est 2000	Population growth rate (% 1997-2015)	Real 1997 GDP/ cap (PPP$)
India	7516	811	1.3	1670
Madagascar	4000	130	2.6	930
Mozambique	2500	351	1.8	740
Sri Lanka	1739	344	1	2490
Tanzania	800	412	2.3	580
Maldives	644	1648	2.6	3690
Seychelles	600	281	1	8171
Kenya	500	250	1.6	1190
Comoros	350	878	2.5	1530
Mauritius	200	697	0.8	9310
Mayotte	185	1177	—	—
Reunion	160	326	1.3	—
Rodregues	37	—	—	—

Sources: Delft Hydraulics, 1993; Gaudian, *et al.*, 1998; National Aquatic Resources Research and Development Agency, 1998; Semesi, 1998; United Nations Development Programme, 1998; Central Intelligence Agency, 1999; Linden & Sporrong, 1999; Mirault, 1999.

indicators of pressure and dependence on reef resources and their ability to cope with impacts such as coral bleaching. CORDIO was initiated in response to degradation of coral reefs caused by the 1998 coral bleaching event. However, other factors, such as rapidly expanding coastal populations or poor planning and management, may also cause reef degradation. Recently, Bryant *et al.* (1998) estimated that 9 000 km^2 of coral reef in the Indian Ocean were at high risk, 10 500 km^2 at medium risk and 16 600 km^2 at low risk of degradation from coastal development, marine based pollution, overexploitation of marine resources and inland pollution, including sedimentation. Within the CORDIO countries, the level of risk of reef degradation ranges from low in areas like the Chagos archipelago where there is negligible human activity, to high in areas such as Comoros and Mayotte where high population growth rates are exerting increasing pressure on these reefs (Figure 1).

This report is a summary of the complete project report assessing the socio-economic impacts of the coral bleaching in the Indian Ocean (Westmacott *et al.*, 2000) and presents the main approaches adopted to determine the importance of fisheries, particularly reef fisheries, and reef based tourism to countries and local communities in this region. Also, this report presents the results of specific case studies of the reef fishery of Kenya and of the tourism sectors of Maldives, Sri Lanka, Tanzania and Kenya. In addition, the assessment also highlights the need to account for other threats to coral reefs and the capacity to manage these resources.

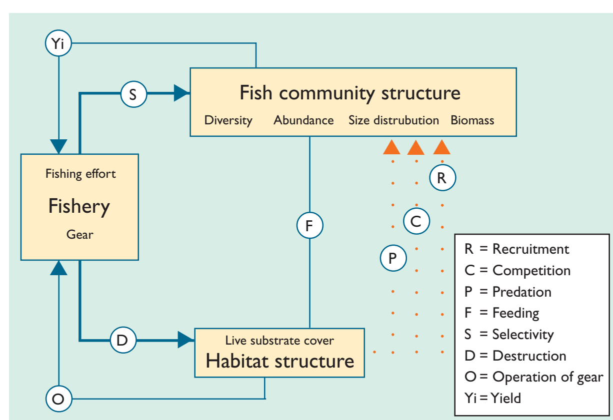
Figure 1. Level of risk to reefs from coastal and marine activities in the Indian Ocean.



POTENTIAL IMPACTS OF CORAL BLEACHING ON REEF FISHERIES

The effects of coral bleaching on reef fisheries are likely to be observed in the long-term through changes in the habitat complexity. Although controversy surrounds the proposed mechanisms by which reef fish communities are structured (Richards & Lindeman, 1987; Sale, 1991; Sadovy, 1996), it is generally thought that three ecological processes are involved. First, competition for food and space determines fish diversity and density (Robertson & Gaines, 1986). Second, patterns of recruitment of juveniles determine adult fish community structures (Eckert, 1984; 1987; Medley *et al.,* 1993; Lewis, 1997). Third, predation determines patterns of survival and consequently, the density of adult fish (Eggleston, 1995). The structural complexity of a reef habitat influences all three of these ecological processes (Figure 2). The reef provides niches for various species to coexist on a coral reef, suitable substrate for reproductive activities and larval settlement (Roberts, 1996) and also shelter for fish

Figure 2. Relationships between the reef habitat, the fish community structure and the fishery.



to escape predation (Williams, 1991; Polunin, 1996). Population structure, species diversity, density and biomass of the fish community can be related to the state of the coral reef which can be measured using various parameters (e.g. rugosity, live coral cover, algal cover) (McClanahan, 1994).

The way reef habitats affect a coastal zone fishery takes three forms. First, maximum yields are limited by the status of the habitat through habitat-fish interactions as described above. Second, the characteristics of a reef habitat (e.g. high coral cover, sandy lagoon floor) and the risk of damage to gear they pose will determine the type of gear used and, to a degree, the species of fish caught. Third, the spatial distribution of physical features that are perceived by fisherman to be attractive to their target species, such as large coral heads or converging currents, will determine the areas in which fishing effort is concentrated.

In a fishery that is entirely dependent on reef fish, catch rates may decrease and the catch composition may shift more towards the herbivorous species. These fish are often lower in value so, as a result, the economic position of fishers may deteriorate. Fisher communities that live on islands with few alternative sources of income will have difficulty sustaining their livelihoods. A fishery that targets large predatory pelagic species that forage on reef fish may also experience lower catches when these fish are forced to move to other less destroyed areas to hunt for prey. A fishery that targets small pelagic species that occupy a reef area or lagoon during certain phases of their life cycle may also experience lower catches when reefs disappear.

METHODS

Fishery assessment

A study of the effects of coral bleaching and mortality on reef or coastal zone fish resources preferably includes historic data (Type I) and spatial data (Type II) enabling:

- An assessment of the qualitative and quantitative impact on the perspective of the total fishery performance – nation-wide.
- An assessment of the social/economic impact based on cases – coastal provinces/districts.
- Predictions of future developments in social/economic conditions of fishers, in response to the event based on past trends in the fishery performance.

Official marine fisheries statistics (Type I) were used to characterise the importance of reef fisheries in each country. Although the quality of official fisheries statistics is often weak, usually they remain the only information used by policymakers to assess the status of a country's fisheries. However, an analysis of data that were available previously and of those that were collected by contributors from each country in the region identifies weaknesses in the information needed to assess the importance of reef fisheries on both a nation-wide and region-wide scale. Information describing resource utilisation and fishery performance (Type II) was collected during a case study of the reef fishery of Kenya. This information, when combined with Type I data, enables an economic valuation of a reef fishery that includes both a financial analysis at the individual household level and, where possible, an economic analysis at the society level (Cesar & Pet-Soede, in prep).

RESULTS

Trends in marine fisheries in the Indian Ocean

Developments in the national demography and social-economy of most countries in the Indian Ocean suggest a continuously increasing pressure on fish resources (McClanahan, in press). In some countries, total fish production is declining (Table 2). Most of the catch from coastal fisheries is used for local consumption, as it is the most affordable source of protein (FAO, 1999b). Shrimp and tuna are the main export commodities. In most countries, fishing gears used in coastal areas include traps, spears, gillnets, seine nets, hook and line, and cast nets.

The large number of small fishing vessels from which the millions of Indian Ocean fishers operate makes monitoring of stock status and implementation

Table 2. Marine fisheries catch in 1997 (tons).

Country	Marine Fish	Crustaceans	Molluscs	Trend since 1990 (%)
India	2,455,947	298,313	121,896	+28
Sri Lanka	208,350	11,000	300	+58
Maldives	107,087	—	271	+35
Madagascar	71,596	14,622	850	?
Tanzania	45,530	2,500	653	-29
Mozambique	14,500	12,906	659	-53
Mauritius	13,397	40	309	+16
Comoros	12,480	20	—	+8
Reunion	5,581	301	—	+213
Kenya	4,382	950	726	-54
Seychelles	4,052	604	19	-27

Source: (Food and Agriculture Organisation, 1999a)

of fisheries management measures difficult (Table 3). Methods used to sample marine fisheries and the way the collected information is processed and presented in reports differs greatly between countries. Little regulation of fishing effort exists, except in a number of marine protected areas around the region and a closed season for the large net fishery off Mauritius.

The number of fishers may be small compared to the number of people engaged in other economic activities (Table 3). These fishers often have few other opportunities to make a living and the fish they catch is a vital source of protein. These factors make it relevant to study trends in fish catches to prepare for alternatives if capture fisheries collapse.

The importance of reef fisheries in the Indian Ocean

The percentage of the demersal fish landings compared to the total fish landings can be seen in table 4. However, when discussing the importance of reef fisheries per country, it is important to distinguish between its importance in providing food, foreign currency and em-

Table 3. Number of marine fishers in each country.

Country	Fishers (number)	Relative importance in Agriculture employment (%)	Full time (%)	Survey
India	5,958,744	2.3	40	1994
Sri Lanka	83,776	2.7	100	1996
Madagascar	67,566	1.3	69	1996
Kenya	43,488	0.4	?	1994
Maldives	22,109	78.6	100	1996
Mozambique	18,000	0.3	100	1990
Tanzania	12,564	0.7	100	1996
Mauritius	10,713	14.3	49	1996
Comoros	9,000	4.4	44	1994
Seychelles	1,960	—	100	1996
Reunion	500	3.3	100	1990

Source: (Food and Agriculture Organisation, 1999b)

Table 4 Demersal fish landings in 1996 per country in the Indian Ocean sorted by relative contribution to total marine landings.

Country	Demersal landings (tons)	Importance in total marine landings (%)	Trend in importance of demersal landings
Reunion	2,970	68	+/-
Mauritius	8,664	52	+
Tanzania	18,939	49	+
Seychelles	1,704	39	+/-
Kenya	1,349	34	+/-
India	805,408	33	+/-
Sri Lanka	36,922	18	+/-
Mozambique	165	17	+[1]
Maldives	11,856	11	+
Comores	na	na	na

Source: (Food and Agriculture Organisation, 1999a)

ployment (Table 5). A short description of the fisheries of each region within the central and western Indian Ocean is presented.

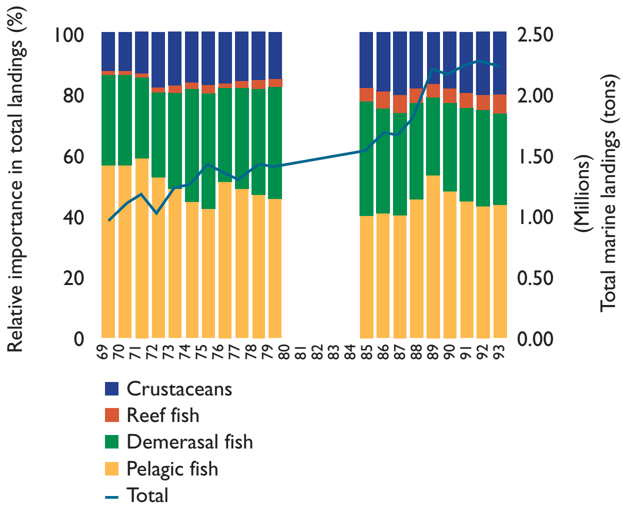
South Asia

In India, the relative contribution of the reef fishery to both earnings and total fish landings (1 790 702 tons in 1993) is low. This did increase from 1% in the early 1970's to 5% in the early 1990's. The importance of other demersal fish remained stable at 32% throughout the entire period (Figure 3) (CMFRI, 1980; 1995). This low importance is caused by the fact that most reefs are found in lightly populated regions such as the Andaman and Nicobar Islands (DOD & SAC, 1997; Bakus, 1994). In addition, there is a high demand for large pelagic fish, such as mackerel and tuna, at both domestic and export markets ensuring that the fishery in large reef areas, such as Lakshadweep, focuses on catching these large pelagics instead of demersal reef species (James *et al.*, 1984; Bakus, 1994). On the mainland, most of India's coastal fishers make their living from either the pelagic fisheries, the prawn trawl fishery or from small-scale

Table 5. Importance of marine and demersal fisheries in providing food (kg of fish), employment (part of overall population) and foreign earnings to Indian Ocean countries. ++++Very high; +++high; ++medium; +low.

	Food		Employment		US$	Other
Country	Marine	Demersal	Marine	Demersal		
India	++++	++++	+++	++	+	
Madagascar	+++	++++	++	+	+++	
Sri Lanka	+++	+++	+	+	-/+	Import equals export
Tanzania	++	+++		+++		
Kenya	+	++	+	+++		
Mauritius	++	++	+	++	++	Foreign fish licenses
Seychelles	+	++	+	++	+++	
Mozambique	++	+				
Maldives	+++		+++	+	+++	Baitfish supports tuna
Comores	++		+	+++		
Rodregues	+		+	+++	+	

Figure 3. Trends in landings and composition of landings in India.



demersal fisheries using beach seines. However, this is likely to change as a result of increasing demand (foreign markets) for high quality reef fish such as grouper and snapper and because of declining catch rates resulting from over-capitalisation and exploitation of coastal shelf areas (Devaraj, 1997).

The reef fishery in Sri Lanka provides an important part of the fish consumed in the country. The demersal fishery does not provide employment to a large portion of the population. The coastal fish production is highly variable but increased five fold from 36 865 tons in 1951 to 157 500 tons in 1995 with highest production in 1982/83 of approximately180 000 tons (Figure 4). Coastal fisheries contributed almost 90% to the nations total fish production in the early 1970's but this importance decreased to 65% in 1996 with increasing importance of offshore fish landings of some 25% in 1996. Inland fish production was relatively important (18% - 19%) in the mid 1980's but decreased to approximately 9% in 1996.

In Maldives reef fisheries contribute least to total fish production, although this is increasing. However, the indirect importance of reef fisheries to the entire fish production is much higher because bait fish for the tuna fishery are caught in the lagoons and near the reefs. Total fish catches have increased dramatically in recent years from 39 000 tons in 1983 (Anon., 1998a) to 118 115 tons in 1998 (Anon., 1998b) (Figure 5). With its vast area of coral reefs, it is remarkable that reef-associated de-

Figure 4. Trends in coastal fish landings in Sri Lanka.

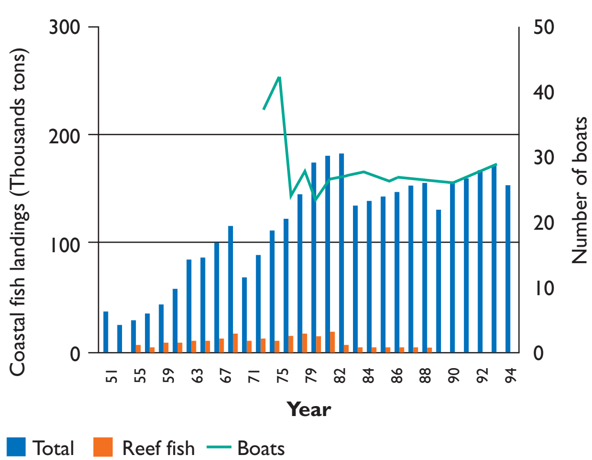
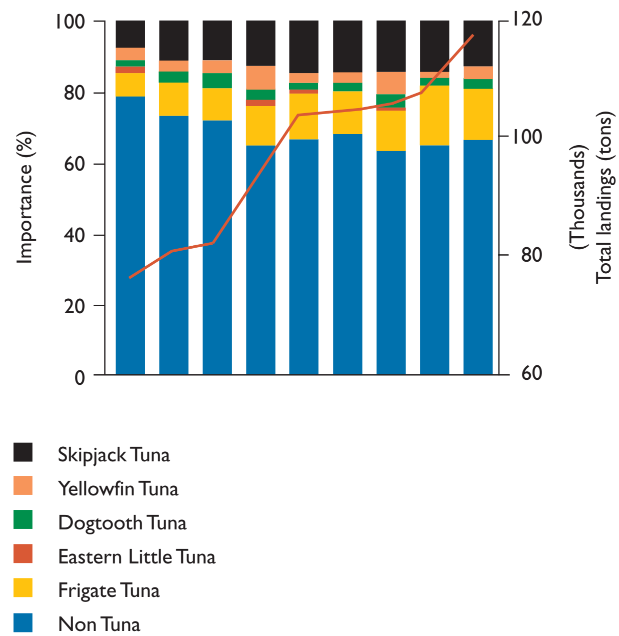


Figure 5. Trends in landings and composition of landings in Maldives.



mersal species have not been heavily exploited in the past. The majority of Maldivians have a high preference for tuna. Some demersal fish were caught, mostly with a single hook hand-line, to supply the tourist resorts and the Sri Lankan market for salt-dried low value reef fish (Anderson *et al.*, 1992).

The Ministry of Fisheries and Agriculture collects data from every inhabited island. However, the Catch Effort Data Recording System (CEDRS) focuses on tuna catches and fisheries. The importance of reef fish has increased following increased demands from new export and domestic markets. It is believed that the current expansion creates overexploitation of the resources and conflicts among resource users (Shakeel & Ahmed, 1997). Reef fishing is most important in the atolls where tuna fishing is poor. In the other atolls reef fishing remains the second most important fishing activity. Catches increased significantly (nearly 5 fold) between 1988 and 1994, but CpUE seems to have declined. Grouper fishing is increasingly important and caters to the international market.

East Africa

The catches of reef fish increased during the 1970's but at the beginning of the 1990's catches decreased to levels recorded in the 1950's. In Tanzania, especially in the northern districts and Zanzibar, the demersal fisheries provide employment to a large portion of the coastal population. Overall, marine landings are decreasing, mainly because of declining pelagic catches. This results in an increase in the relative importance of the demersal fish production (Table 4). The majority of Tanzanian fisheries (96%) are small-scale and exploit the reef-associated habitats (Darwall & Guard, in press). There are 3 232 registered vessels and their fish production varied between 36 000 tons and 56 000 tons between 1990 and 1995.

Reef fish in Kenya is less important in the total fish production. Nevertheless, the high numbers of coastal fishers have few alternatives to make a living. The total marine fish catch seems to have collapsed in the early

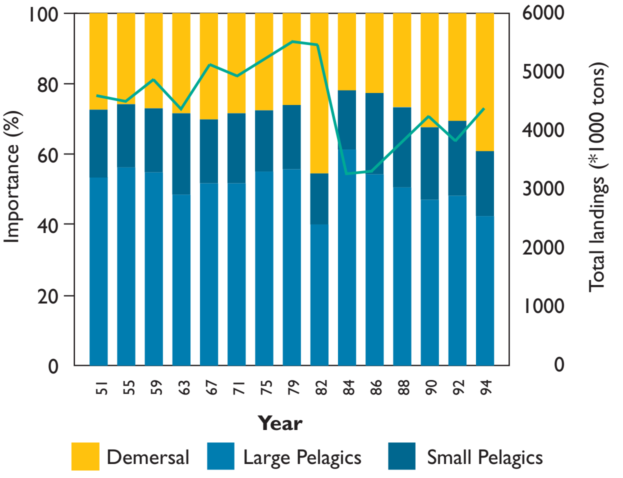


Figure 6. Trends in landings and composition of landings in Kenya.

1990's and, although catches have increased since then, only by the late 1990's was production back to 5 000 tons, the level that was produced in the early 1980's (Dept. of Fisheries, Mombasa) (Figure 6). Some 4 700 fishers are active in coastal regions of Kenya. The contribution of demersal fish to the total marine fish production has been quite stable during recent years at approximately 30% - 40%.

Indian Ocean Islands

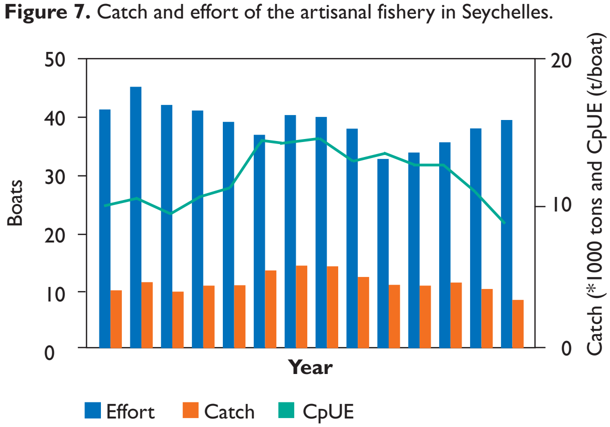
In Madagascar, 43% of the fishery is based on the coral reefs (65 090 tons). Of the total fish production, 20% is exported, which means an important amount of foreign earnings is derived from the demersal fishery (Table 4). Export of fish almost doubled between 1986 and 1989 to 24 264 tons, of which 8 000 tons were shrimp. In 1994, the value of the exported shrimp catches was 80 million US$ and landed by-catch represented 45% of the total fish landings in that year. This was a 27% increase during 1994, therefore shrimp trawling poses a growing threat to the sustainability of Madagascar's demersal fisheries. In 1994, 117 500 tons of fish were produced of which 55% was captured in small-scale fisheries. Approximately 50 000 people are involved in this fishery using 22 000 boats and living in 1 250 villages. Total fishing effort increased five fold between 1977 and 1994.

In the island states of Mauritius, Reunion and Rodrigues the relative contribution of demersal and reef related fisheries to total fish production is high (Table 4). In 1999, there were between 2 500 and 3 000 professional fishers in Mauritius. Total landings in the artisanal lagoon fishery (traps) in Mauritius have been relatively stable and have increased only slightly from approximately 1 600 tons in 1991 to nearly 2 000 tons in 1999 (Naim *et al.,* in press). An important fishery in Mauritius is the Banks fishery along the Mauritius-Seychelles Ridge with 15 vessels that produced another 4 424 tons of fish in 1996. The vessels are large (200-430 GRT) each employing between 50 and 60 fishers and 20 crew.

Fisheries in Seychelles include an industrial fishery of foreign licensed tuna purse seiners and longliners, a semi-industrial fishery of longliners for swordfish, and the reef fishery. In 1992, handlining and traps set on sandy or sea-grass substrate that target rabbit fish contributed most of the total fishing effort (Jennings *et al*., 1995). Handlines caught 78.3% of the 5 718 tons of fish landed in that year (Figure 7). Seychelles relies largely on fish exports and tourism for foreign revenue. In 1992, 200 tons of fresh fish and 839 tons of frozen fish were exported. In 1995, artisanal fish production was 4 313 tons of which 420 was exported for 10 million SR.

In Comores there are approximately 8 000 fishers. All are artisanal fishers and live in approximately110 fishing villages.Total fish catch in 1995 was13 000 tons of which 72% was pelagics. (8 000 tons were caught at Grande Comores). The monetary value of the catch in that same year was estimated at 9 million francs of which 6 million was contributed by Grande Comores alone.

Figure 7. Catch and effort of the artisanal fishery in Seychelles.



Assessment of the effect of bleaching and coral reef degradation on coral reef fish and fisheries in Kenya: A Case Study

The Coral Reef Conservation Project (CRCP) is a U.S. based NGO of The Wildlife Conservation Society that has monitored Kenyan coral reefs since 1986 and fish catches associated with coral reefs since 1995. The project includes a study of fish populations in Kenya's older (>25 years) fully protected marine parks (Malindi and Watamu MNP), a more recently created park Mombasa MNP (1991), and four sites on heavily fished unprotected reefs (Vipingo, Kanamai, Ras Iwatine and Diani). This study was conducted in late 1997 and repeated in early 1999, around four months before and 10 months after the coral bleaching event. For the purpose of assessing possible effects of the 1998 bleaching event, abundance and composition of the reef fish community was determined, together with biomass and composition of individual fish catches.

The underwater visual census data showed no clear changes in fish community structure that can be attributed solely to the bleaching and mortality of corals. Only the increase in abundance of surgeon fish, which are grazers that feed on algae on the surface of the dead coral, may be related to coral mortality. It appears that there is a strong relationship between management (marine park versus exploited reefs) and fish abundance for many of the studied fish families (McClanahan & Arthur, in press). The catch assessment data show a significant decline in catch between 1995 and 1999, whereas the total fishing effort, measured in numbers of fishers or boats remained constant. There is no significant deviation from this trend after the 1998 bleaching event.

Therefore, it must be concluded that, at this stage, the fishery has not been significantly affected by the bleaching and mortality of corals. Nevertheless, the declining catches may be a result of overall environmental degradation. Therefore, it is expected that the effects of the recent bleaching and coral mortality may become more evident once the reefs are further eroded in the future.

ASSESSING THE IMPACTS OF THE CORAL BLEACHING ON REEF BASED TOURISM

The second major socio-economic impact of coral bleaching would be expected on the tourism industry. Tourism will be affected by bleaching in those areas where a substantial proportion of the industry is based on reef activities and where there are few other attractions or activities for tourists to enjoy. Tourism varies throughout the countries of the Indian Ocean and the diversity of the tourism product ensures a greater or lesser dependence on coral reefs. Table 6 indicates the level to which each of the countries is dependant on coral reefs, and the national growth rate in tourism seen over the past five years.

Reef based tourism is a major industry in both Maldives and Seychelles, although they are marketed quite differently. Maldives caters for the diving market (45% of all tourists dive) and the honeymoon market. Seychelles, on the other hand, offers a variety of activities and people may snorkel and dive as a small part of their vacation (only 7% of all tourists dive). Similar patterns were seen in Zanzibar where people spend, on average, less than 40% of their vacation time diving and snorkelling. In Kenya and mainland Tanzania, wildlife parks and safaris are probably the main attraction for visitors. However, visitors may often spend a week on safari and then a week at the coast where the reef based attractions form an important component of their vacation. Island states, such as Comoros and Rodregues have small-scale tourism industries. In Comoros, tourism employs 600 people and in Rodregues 254 are employed, of which only five are employed directly in the dive industry. India supports a huge tourism market, although relative to the size of the country and its economy it is of lesser importance compared with some of the smaller island states. The reefs of India tend to be remote and difficult to access so reef based tourism is limited. Sri Lanka has some reef based tourism, but has also many other attractions. There has been enormous overuse of certain areas, such as Hikkadua where over 90 glass bottom boats operate. Visitors to Reunion and Mayotte are generally friends and family of residents and those visitors that are genuine tourists are usually from France or, in the case of Mayotte, from Reunion.

Table 6. Relative importance of reef based tourism to the economy and 5 year trends in national tourism for countries of the Indian Ocean. ++++Very high; +++high; ++medium; +low; -negative.

Country	Contribution of reef-based tourism to the gross domestic product, GDP	National tourism trend
Maldives	+++++	++
Mauritius	++++	++
Comoros	+++	++
Seychelles	+++	+/-
Zanzibar, Tanzania	++	+++
Madagascar	+	++
Kenya	+	+/-
India	+	+
Reunion	+	+
Sri Lanka	+	-
Mozambique	+	No data
Rodregues	No data	++++
Mayotte	No data	+++

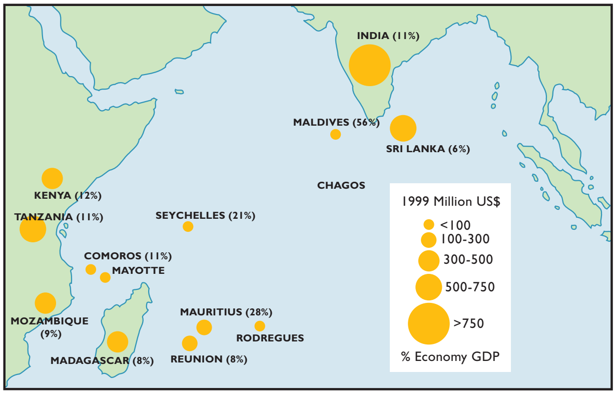


Figure 8. Revenue generated from travel and tourism and the contribution of this to the national GDP within the Indian Ocean. Source: World Travel and Tourism Council, 1999.

Perhaps more importantly than the total arrivals, is the actual financial gain a country or region might receive from tourism. The World Travel and Tourism Council produce simulated forecasts of world tourism. The revenue generated from travel and tourism and the contribution of this to the national economy (GDP) is shown in Figure 8. Maldives shows that 55.9% of the national economy is based on travel and tourism. The next highest is Mauritius at 27.9%, followed by the Seychelles at 20.7%. All these countries are small island states and most of this tourism will be all or partly based on the reefs. Those countries with lower revenue from travel and tourism depend heavily on industry for their national economies.

Two specific case studies were carried out to examine the financial and economic impacts of the coral bleaching on tourism. The first was conducted in Tanzania and Kenya and the second in Maldives and Sri Lanka. The following sections give a brief synthesis of these two studies.

Assessing the impacts of coral bleaching on tourism in Tanzania and Kenya

One of the specific case studies initiated as part of the socio-economic assessment of the impacts of the coral bleaching within the CORDIO programme was carried out in Tanzania (Zanzibar) and Kenya (Mombasa). The aims of the research were to:

- Establish whether tourists are familiar with coral bleaching
- Estimate the financial and economic cost of coral bleaching to tourism in Zanzibar and Kenya
- Compare the recreational value of the reef before and after the bleaching event.

Methods

This research is based on a questionnaire survey of tourist divers in Zanzibar and Mombasa. The economic analysis is based on the contingent valuation methodology (CVM). Financial costs are based on expenditure data given by the respondents. The questionnaire was initially developed for use in Zanzibar and Mafia Island, Tanzania and had been through pre-testing and a full survey (Andersson, 1997). Although a few questions were omitted and a few added, it was not felt that it was necessary to pre-test the survey again. In Zanzibar, 199 divers were surveyed, the sample being split evenly between the two sites. Initially, in Mombasa, a total of 105 divers were interviewed. Surveys were carried out at the dive shops of Zanzibar and Mombasa.

Results

The divers visiting Mombasa were found to be on average older and more experienced divers than those in Zanzibar. However, the respondents in Zanzibar had a higher level of education than those in Mombasa. In Zanzibar, it was estimated that divers spent approximatley 42% of their vacation participating in reef related activities compared to 50% in Mombasa. The importance of the reef can be seen in the diver's ranking of the various attractions in Figure 9.

Diver awareness of coral bleaching

The study found that only a limited number of tourists surveyed at the two case study sites were actually aware of coral bleaching. In Zanzibar, this was 28% and in

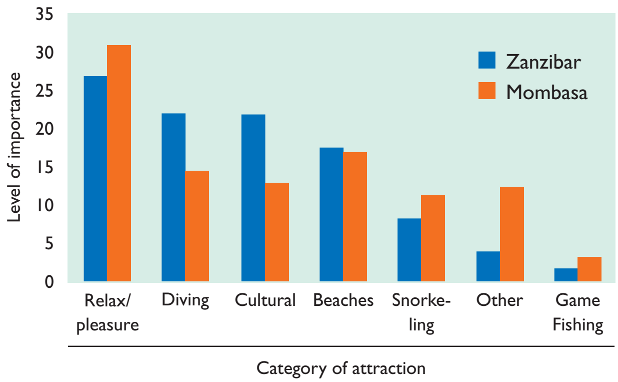


Figure 9. Ranking of the importance of the various attractions.

Mombasa this was 45% (Figure 10). This low awareness could be related to their country of origin, level interest in the marine environment and dive experience. These links were explored but the sample size of those aware of the bleaching was too small to make any significant conclusions. However, of those who were aware of the bleaching, over 80% stated that knowledge that an area was bleached would affect their decision to either visit that area or to dive and snorkel in that area (Figure 10). This enabled estimations of the financial and economic costs of the coral bleaching to be made.

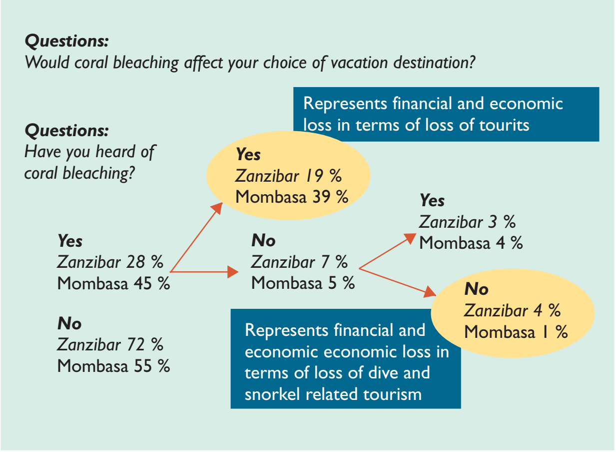


Figure 10. Responses to the questions regarding the knowledge of coral bleaching and its effect on choice of diving destination.

Valuation of the reef resources

In estimating the financial and economic costs of the coral bleaching, the survey techniques and the valuation methods developed by Andersson (1997) for the previous survey in Zanzibar were used. The financial cost of the bleaching are the losses to the local community and tourism economy resulting from those tourists deciding not to visit or simply not to dive in the locations because of the bleaching. This is calculated using the diver's and snorkeller's expenditure data collected during the survey. The economic cost of the bleaching represents the loss of value to the same group of tourists, either not visiting because of the bleaching, or visiting but not diving. This loss affects the divers and snorkellers for not having access to healthy reefs. The economic cost is calculated from the diver's and snorkeller's stated willingness to pay. There are two components to the willingness to pay. The first is the consumer surplus, which is the additional money the tourists would be prepared to pay to still visit the place. This reflects the value of the benefits they gain from recreation exceeding the total cost they have spent on visiting the place. The second is the willingness to accept compensation for the fact that they are unable to dive because of the degraded reefs. The first value is used as the cost when divers do not visit the area and the second is used when divers visit but choose not to dive. When aggregating the results, a range (% of tourists diving) is used if the exact figure is not known. This is thought to be in the region of 20% - 30%. In Maldives, a diving destination, 45% of the tourists were recorded as divers. These aggregated costs can be seen in Table 7.

Table 7. Financial and economic cost of the coral bleaching on Zanzibar and Mombasa (range based on % of total tourists who dive).

	Financial cost million US$	Economic cost million US$
Zanzibar	3.08- 4.62	1.88-2.82
Mombasa	13.33- 19.99	10.06-15.09

Comparison of economic value between 1996 and 1999
One of the main components of this research was the ability to compare diver and snorkeller valuations of the reefs of Zanzibar before and after the coral bleaching by incorporating the data collected in 1996/7 by Andersson (1997) into analyses. Compared with the 1999 results, the overall consumer surplus was unchanged indicating that a complete holiday to Zanzibar was valued the same in 1996 as it was in 1999. The level of reef use was also comparable. However, the willingness of the divers and snorkellers to accept compensation for non-access to the reefs had increased 20% between 1996 and 1999. This indicates that the reef remains an important component of the visit and the value placed on having access to reef related activities has actually increased.

The willingness to pay for reef conservation can be related directly to the state of the reef in 1996 and in 1999. In 1996, the average willingness to pay to maintain the reef in the same state was $30. In 1999, this had dropped to $22, a 27% decline from 1996 to 1999. This reflects either a decline in the perceived state of the reef or a change in the type of tourist and their willingness to pay for reef conservation. However, comparison of the socio-economic data obtained in 1996 with those obtained in 1999 determined that the only difference between the two groups was that divers surveyed in 1996 were generally more experienced than those surveyed in 1999. In 1996, the average number of dives each diver had completed was 83, compared with only 33 in 1999. This may be an indicator that the more experienced divers are aware of reef conditions and their decision has already been affected by stories of reef degradation or that these divers are travelling elsewhere, where they can get more adventure and extreme conditions of diving.

Management implications of the results
One of the major findings of this research was the fact that, although only a limited number of tourists were aware of coral bleaching, or of reef degradation generally, their decision to visit may well be affected. From a management perspective, this has implications for the type of information that the tourists are receiving on the state of the reefs. Should bleaching adversely affect the reefs, tourists may still visit the area if alternatives are supplied. These may be marine based or even land based. Planning for a change in tourism activity may need to take place sooner rather than later.

The decrease in willingness to pay for the conservation of the reefs may be related to the state of the reefs but also could be related to the level of visible management. To gain support for reef conservation from visitors, management efforts need to be visible through public information, brochures, active rangers and patrols. What is useful from the data collected is the approximation of a willingness to pay being approximately 2% - 3% of the total vacation expenditure. This type of data can be utilised when establishing protected areas and generating revenue through user fees.

Limitations of the study and further research
There were several limitations of the study imposed by time and financial constraints. For full analysis and comparison of results obtained in 1996 and 1999, the survey needed to cover the higher-price hotels along the east coast also. In addition, Zanzibar was only mildly affected by the coral bleaching whereas Mafia Island was heavily affected. The 1996 survey was also carried out on Mafia Island and a re-survey of this area could provide some useful insights into financial and economic costs of the bleaching. Broadening the survey to cover all tourists both at home and abroad would also increase the understanding of tourist behaviour with respect to coral bleaching and reef degradation.

Assessing impacts of coral bleaching on tourism in Maldives and Sri Lanka
This study focuses on impacts of coral bleaching and subsequent mortality on tourism in the Maldives and Sri Lanka. In Maldives, with 430 000 tourists in 1999 (Ministry of Tourism, 2000), diving and other reef-related tourism are the main income generating activity in the country. Sri Lanka has a similar number of tourists but

very few come specifically for reefs, even though they are attracted in general to the coastal areas. The current study addresses socio-economic questions related to coral bleaching and tourism primarily by recording tourists' perceptions of coral bleaching. Also, estimates of the financial and associated welfare losses resulting from the 1998 coral bleaching event are provided.

Methods

This research was based on both questionnaire surveys and secondary data sources. Four different surveys were carried out: (i) one for tourists departing from Male airport in Maldives and from coastal tourist locations in Sri Lanka; (ii) one for key informants such as dive operators and glass bottom boat captains in Sri Lanka; and tour operators in Italy; (iii) one for tourists at the airports of Amsterdam, Duesseldorf and Milan on their way to Maldives and Sri Lanka; and (iv) dive tourists were asked via the internet about their knowledge of coral bleaching in Maldives and whether bleaching and coral mortality was a factor that influenced their decision to go there. The secondary data sources were the official tourism statistics of Maldives and Sri Lanka.

Results

Interest in the marine environment

In Maldives, there seemed to be three main categories of tourists: (i) divers; (ii) honeymooners; and (iii) 'relaxers'. Around 45% of all tourists going to Maldives were divers. In Sri Lanka, only approximately 8% were divers. Italians tend to visit Maldives for their honeymoon while Germans go to dive. The number of dives made while visiting each country also varied considerably. In Sri Lanka, of the 8% that went to dive, 50% did only one or two dives, while in the Maldives, 69% of divers did more than five dives. With respect to their interest in the marine environment, 52% of the tourists at Male airport responded that the importance of marine life was very high, 34% answered that is was rather important and only 13% said that it was not important. In Sri Lanka, the results were quite the opposite. Only 18% stated that marine life was very important, while 32% and 51% said that marine life was rather important and not important respectively.

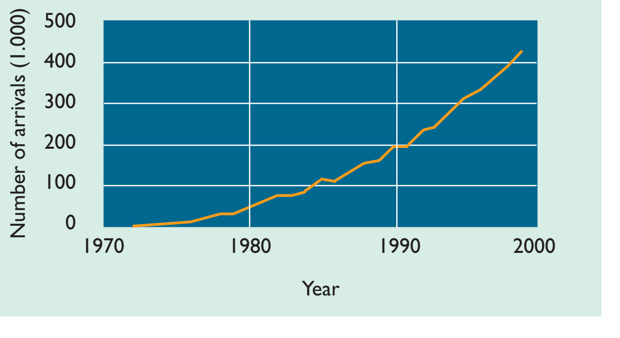
Divers' and snorkellers' knowledge of coral bleaching

The media coverage of the coral bleaching episode of 1998 has been substantial. Dive journals have given considerable attention to the bleaching event and to reactions of divers. Yet, interviews at the European airports showed that many tourists on their way to Maldives did not know of the episode. Fifty percent of Germans surveyed had heard of the coral bleaching event in Maldives, compared with 30% of the Italians and 16% of the Dutch. This can be explained partly by the exceptionally large media coverage in Germany and by the large percentage of divers among German tourists. At Male airport, 68% of departing tourists had heard of coral bleaching, while in Sri Lanka, less than one third knew of this problem.

Losses in Tourism Revenues in Maldives

Possible losses to Maldives' economy were analysed based on the official tourism statistics up to December 1999. Figure 11 presents tourist arrivals since the 1972. Surprisingly, there was not a significant drop in tourist arrivals in 1998-1999. In fact, tourism arrivals have increased 8% in both 1998 and 1999.

Figure 11. Number of tourist arrivals in Maldives between 1970 and 1999.



Source: Maldives Ministry of Tourism (1997, 2000).

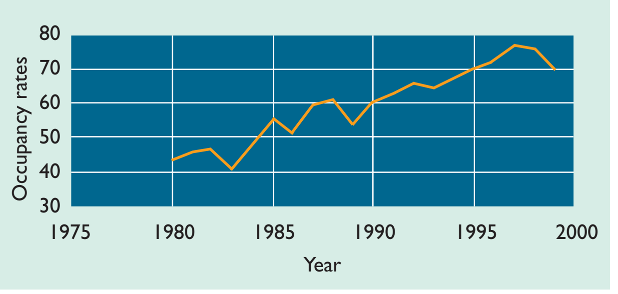


Figure 12. Bed occupancy rates in Maldives between 1980 and 1999. *Source:* Maldives Ministry of Tourism (2000).

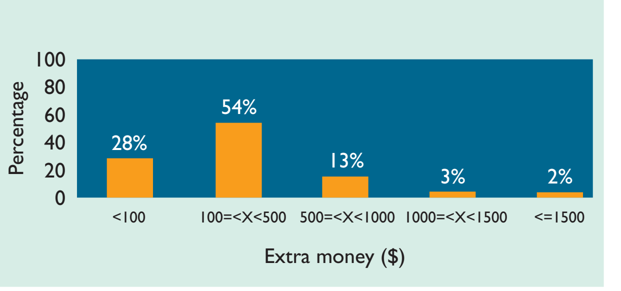


Figure 13. Willingness to pay for better reef quality in Maldives.

However, trends in bed occupancy rates since 1975 give a different picture (Figure 12). Given the time lag between the planning phase of expansion and the additional bed capacity, occupancy rates give a proxy for expected growth in tourism and the decrease in 1998/9 was substantial. However, the Asian crisis was also affecting tourist numbers. Another way of looking at expected growth of tourism arrivals is to check the official government tourism forecasts. In 1997, an annual growth of 10% was expected for the years of 1998 and 1999 (Ministry of Tourism, 1997), which was 2% higher than the realised figures. Here, we assume that half of this difference was due to coral bleaching.

Welfare losses from divers

Besides financial losses to the local economies, coral bleaching can also affect tourists' holiday satisfaction and thereby create a loss in their welfare. In order to calculate these welfare losses, the surveys at Male airport focused on tourists' willingness to pay for 'better reef quality'. In order to ensure the tourists value the same change in reef conditions, two pictures were shown, one of a reef that had completely died because of bleaching and another that was still intact. The question asked of tourists was how much extra were they willing to pay to go to hypothetical remote areas in Maldives where reefs were not affected by coral bleaching and which were, in all other respects, the same. Figure 13 shows the distribution of this willingness to pay (WTP) and illustrates that the tourists surveyed were willing to pay an average of US$ 284 more to visit these hypothetical reefs. Divers were prepared to pay more than other tourists, though the difference was surprisingly small. The mean WTP for divers was US$ 319 while for non-divers it was US$ 261. The aggregated losses can be seen in Table 8.

Finally, tourists were asked about the most disappointing part of their Maldives holiday. The possible answers were: (i) the price of food and beverages; (ii) the weather (humidity, clouds, etc.); (iii) the fact that a lot of the corals were dead; (iv) the mosquitoes; (v) the resort accommodation; (iv) others. Figure 14 summarises the responses, showing that 47% considered the dead corals the most disappointing experience, while the price of food and beverages was second with 28%.

Table 8. Losses in tourism revenues and welfare in Maldives and Sri Lanka for 1998/9.

	Financial costs million US$	economic costs million US$
Maldives	3	63
Sri Lanka	0.2	2.2

This last result is interesting, because nearly all resorts are based on half or full board, so that the actual amount of money spent on additional food and beverages is quite low, though beer is expensive at around US$ 5 per bottle. The interesting aspect of these responses is

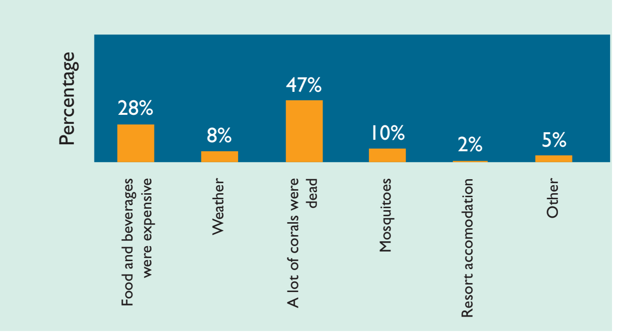


Figure 14. Question about the most disappointing part of the Maldives holidays.

that they allow us to compare and therefore scale the willingness to pay WTP values. Surprisingly, the average WTP for better reef quality was not statistically different for those who found coral mortality most disappointing and those who found other parts of their holiday most disappointing. Note that one could buy more than 50 bottles of beer for the average WTP for improved corals, which might either suggest an inconsistency in the way people respond to the various questions or, alternatively, there are quite a few very hefty drinkers among the tourists. Unfortunately, it might also mean that many tourists do not really care about the death of coral reefs.

Assessing the future tourism impacts

The two case studies show a number of interesting similarities as well as differences.

- Awareness of the 1998 coral bleaching event among tourists going to destinations with coral reefs is generally rather limited.
- Current losses in tourism revenue due to coral bleaching event have, so far, been rather low. In Maldives, it is estimated that only US$ 3 million was lost during 1998 and 1999 combined. In Mombasa, the losses were estimated to be much higher (US$ 13-20 million), but these were hypothetical losses assuming permanent disappearance of tourists.
- A key determinant of losses in tourism revenues was the ability to attract other types of tourists who, despite being interested in coral reefs and reef based activities, were not interested only in diving. This flexibility could help explain the lower losses in Maldives compared with Zanzibar and Mombasa.
- Divers seemed willing to pay considerable sums for better reef quality. In Maldives alone, the total welfare loss for 1998/99 was estimated at US$ 19 million.

Future tourism losses remain uncertain. Key determinants are the long-term impact of relatively slow word-of-mouth reports or TV documentaries on bleaching. Despite the loss of some avid divers who appear to be going to areas that have not been impacted by bleaching, they are easily replaced by the hundreds of new divers that appear on the market. The key uncertainties are related to the impacts of coral mortality on fish populations and on beach erosion.

The impacts of the coral bleaching on tourism should be seen in the wider picture of reef degradation which, in itself, is not the only issue affecting tourism. Mombasa has seen a huge decrease in tourism relating to public opinion on personal safety. Much of this impression is created in national newspapers indicating the power of the media in altering public perception.

A further aspect of analysing the impacts of events such as the bleaching, is to look carefully at who is being impacted. The tourist has a variety of alternative locations and may not be affected, whereas the local dive guide may be unemployed as the dive industry adjusts or is impacted.

DISCUSSION

The 1998 El Niño event has so far not affected socioeconomic indicators dramatically. Reef fisheries in many areas in the region have been showing a general decline over the last decade and data collected can not yet tell what the added negative impact of coral bleaching is. On the other hand, diving tourism has been growing rapidly all over the world (except in East Africa). Again, the added influence of coral bleaching on these trends is uncertain. Tourism studies show however, considerable financial costs ranging between US$ 3.1 and US$ 4.6 million in Zanzibar and US$ 13.3 and US$ 20.0 million in Mombasa. In Maldives, financial costs were estimated at US$ 3.0 million, while economic costs over the last two years were roughly US$ 63 million.

In the long run, the impacts may be rather more dramatic if increased erosion of the reef and a loss of reef complexity occurs, which would be expected to take between two and 10 years. Given the lack of other global coral bleaching events, the likelihood of this scenario is uncertain. Yet, major declines in fisheries and tourism can not be excluded, with corresponding impacts on marginal populations in coastal areas.

Furthermore, coral bleaching has re-opened the discussion about effective coral reef management. Reducing the pressure on coral reefs from their over-usage has never been an easy task. However, it will be essential if reefs recover from the bleaching and survive future threats. The bleaching of corals is more difficult to control. If this is a natural event, there is little man can do to manage it. We can only assist in recovery through appropriate protection of the reefs and vital sources of larvae. If, on the other hand, coral bleaching is caused by world-wide pollution and the consequences of climate change and global warming, it will take a massive global effort to reduce impacts in the future.

If continued coral reef degradation is going to be a widespread phenomenon in the Indian Ocean, the following questions need answering:

- To what extent will reef fish stocks be affected?
- Will a decline in reef fisheries or change in population composition affect pelagic fisheries?
- Will reef based tourism be replaced by other forms of tourism?
- What will happen to the Marine Protected Areas dependent on tourists visiting the reefs for their income?
- Can we maintain the tourism industry and utilise the tourism market for basic monitoring of reef fish and habitats?
- What are the links between reef usage and the bleaching?

ACKNOWLEDGEMENTS:

This report is the summary of a series of studies to which many different people have contributed. It has been made possible with funding from the African Environment Department of the World Bank co-ordinated by Indu Hewawasam. In addition, Sida and WWF-Sweden have funded various vital components of the fieldwork, which has been co-ordinated by Olof Lindén. The in-country, CORDIO teams collected a substantial amounts of data for the Islands. Namely, ARVAM in Reunion, Marine Parks Authority, Seychelles Fishing Authority and Shoals of Capricorn Programme in Seychelles, University of Mauritius in Mauritius, Shoals of Capricorn in Rodregues, SPEM in Mayotte, AIDE in Comoros and University of Toliara in Madagascar. More detailed studies were possible in India with the support of Dr. Venkataraman and Mr. Rajan and in Kenya with Tim McClanahan from CRCP. The fieldwork for the case study in Tanzania and Kenya was carried out by Irene Ngugi, supported in Zanzibar by the staff of the Institute of Marine Science and in Mombasa by the CORDIO office. Support in the analysis and access to the 1996 data was given by Jessica Andersson. The fieldwork for the Maldives case study was carried out by Ali Waheed and Marie Saleem and in Sri Lanka by Dan Wilhelmsson. The fieldwork in the airports in Europe was carried out by Bas Rabelling and Ludovica Reina. Computational assistance was provided by Clement Roos at the IvM in the Netherlands.

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Coral and algal response to the 1998 El Niño coral bleaching and mortality on Kenya's southern reef lagoons

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INTRODUCTION

The 1998 interaction between the El Niño and the Indian Ocean dipole produced one of the warmest years in recent records (McPhaden, 1999; Saji, 1999; Webster *et al.*, 1999) and is reported to have caused extensive coral bleaching and mortality throughout the western Indian Ocean (Strong *et al.*, 1998; Goreau *et al.*, 1999; Wilkinson *et al.*, 1999). Previous observations of coral bleaching in Kenya were recorded in 1987 and 1994 with the 1987 event causing significant mortality in corals and other benthic invertebrates (McClanahan, unpublished data and observations). The East African coast has a strong seasonal cycle and these bleaching events occurred at the end of the warm north-east monsoon, usually beginning in March, during the local annual peak of solar irradiance and water temperature (McClanahan, 1988). The recent 1998 coral bleaching event was the most severe in terms of the mortality of benthic organisms, particularly corals, and, therefore, efforts were made to document this event and to determine the sensitivity of coral genera to this disturbance, the role of reef management and in particular, the role of herbivory, on the ecological outcome of this coral mortality.

Coral mortality from bleaching or other factors has been shown to produce a variety of responses in coral reef benthic communities (Brown, 1997). These range from quick recovery of coral cover and species composition (Brown, 1997), switches in coral species dominance (Aronson & Precht, 1997; Greenstein *et al.*, 1998), overgrowth of bare substrate by erect fleshy algae (Shulman & Robertson, 1997; McClanahan *et al.*, 1999), near extirpation of species (Glynn & Feingold, 1992), to the destruction of reef framework by bioeroding organisms (Glynn, 1988; Eakin, 1996). The factors that determine reef changes by bleaching after mass mortalities are, therefore, of considerable interest to understanding reef ecology and for reef management. This study compares four un-fished coral reef parks of Kenya and a monitoring study of herbivorous sea urchins and fishes in and out of these marine parks to determine how these two factors influenced the mortality and the benthic response to this mortality approximately one year after the bleaching mortality. We hypothesised that coral mortality and herbivory would interact to influence the response of benthic algae and that management of herbivores would determine the response to this mortality.

METHODS

Study sites included nine sites in the four MPAs, Malindi, Watamu, Mombasa and Kisite Marine National Parks (MNPs) and seven sites in four unprotected reefs, Vipingo, Kanamai, Ras Iwatine, and Diani. Sites within a reef are often separated by 20 m to 100 m and the reefs are distributed along 190 km of coastline with distance of 3 km to 50 km between reefs. Sites were in back reef lagoons with shallow water (< 3 m) at low tides (Kenya has a 4 m tidal range) dominated by hard substrate colonised by corals and other benthic invertebrates and algae (McClanahan & Shafir, 1990). Benthic cover, sea urchins, and major fish groups have been monitored in these sites since 1987. Artisanal fishing, which dominates the fishery on the Kenyan coast, is mostly restricted to these shallow reef lagoons in Kenya.

A water temperature logger (Onset Corporation Hobo Temperature Loggers) that recorded hourly measurements was deployed in a shaded lagoonal area of the Mombasa MNP under a massive coral at about 1 m water depth (at low tide) for a period between August 1996 and May 1998. In addition, NOAA satellite sea water temperature data for Malindi, from 1982 to 1998 was obtained from NOAA's electronic archives. To compare these two data sets we used and present only the midday temperatures from the *in situ* temperature logger and performed a least-squares regression on the monthly averages for the two data sources for the period during which the logger was deployed.

In each of the above 16 study sites, benthic line transects were completed before (November 1997 to January 1998) and after the bleaching (August 1999 for some sites to determine mortality and again between January and March 1998 for another group of sites to determine changes in benthic cover). In each site, nine to 12 haphazardly placed and loosely draped 10 m line transects were used to describe the benthic cover (McClanahan & Shafir, 1990). We classified and measured the length of all benthic organisms > 3 cm into the following nine gross substrate categories, hard coral and soft coral; fleshy, turf, red coralline and green calcareous (*Halimeda*) algae; sponge, seagrass and sand.

Sea urchins, surgeonfish and parrotfish were censused in each of the study sites both before and after the bleaching event to estimate their wet weights and rates of consumption. Sea urchins were identified to species and counted in nine to 12 10 m^2 circular quadrats in each study site. The biomass of each species was calculated by multiplying the average wet weight (McClanahan, unpublished data) with the average density of each species, and the total weights of each species were summed to estimate the total sea urchin wet weight. Consumption rate studies of the four most common species of sea urchin undertaken on Kenyan reefs indicate that the average daily consumption for these four species is ~1.6% of their wet wets (McClanahan & Kurtis, 1991; Silva, 1999). Consequently, we multiplied the total sea urchin wet weight (kg/ha) by 0.016 to estimate sea urchin daily consumption in kg/ha/day.

Herbivorous fish were counted in three to five 5 m x 100 m belt transects per site, identified to the family and standard body lengths estimated in 10 cm intervals, with a 3 cm minimum. Count and body length data were converted to wet weight estimates per family from length-weight relationships established from fish catches at a landing site adjacent to the Mombasa MNP (McClanahan & Kaunda-Arara, 1996). We estimated an average daily consumption rate for these fishes as 16% of their body weight per day based on a summary of literature studies (McClanahan, 1992; Bruggeman *et al.*, 1994), so consumption rates of fish were estimated by multiplying their estimated average wet weights by 0.16.

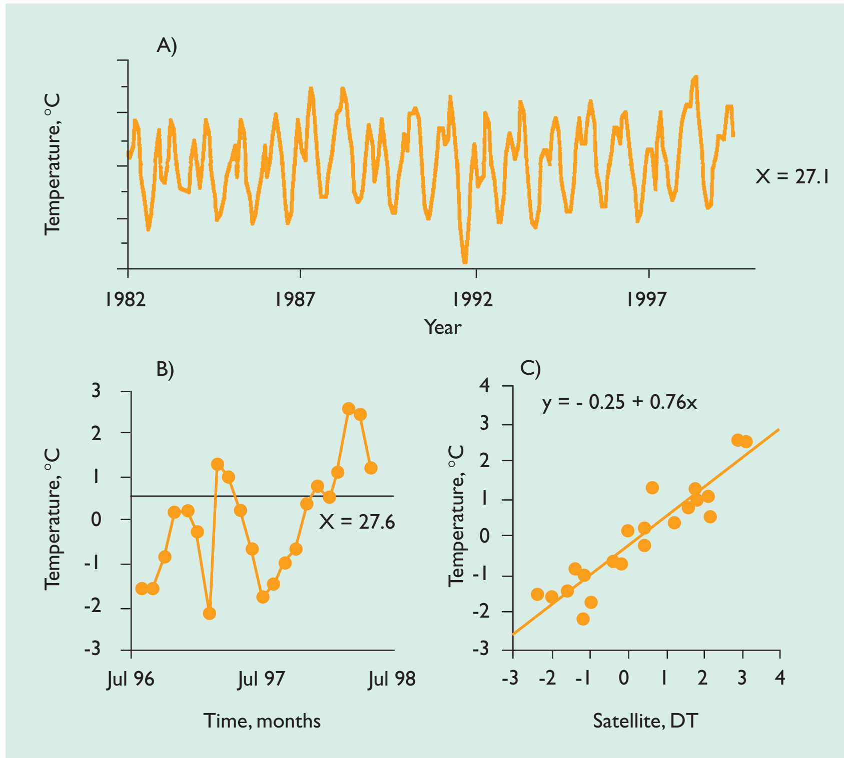
Sea urchin and herbivorous fish consumption rates were summed to estimate the total herbivore consumption rate for each site on a kg/ha/day basis. In order to determine the ability of herbivore consumption rates to predict changes in fleshy algal abundance due to coral mortality, regressions and multiple factor ANOVA tests (Sall & Lehman, 1996) were used on pooled herbivore consumption rates before and after bleaching with the change in coral and fleshy erect algae cover.

RESULTS

Sea water temperatures

Sea water temperature data indicate that average midday water temperature during March 1998 was between 30° C and 31° C (Figure 1). The NOAA satellite data suggest that this was the warmest month on record for the Malindi site with an elevation of 1° C to 1.5° C for the month of March and April. 1998 was also unique in that there was only a very small drop in water tempera-

Figure 1. Water temperature time series for a) NOAA satellite data for Malindi, b) monthly average midday temperature collected by the Hobo temperature data logger (Onset Co.) deployed in the Mombasa MNP and c) regression analysis of the two data sets for the period between August 1996 and May 1998



ture during the onset of the north-east monsoon winds, usually between January and February, prior to the maximum temperatures in March. The satellite and logger data were well correlated, although the intercept was not zero, nor was the slope 1, which suggests that the logger was less responsive than the satellite data to temperature changes.

Post-bleaching benthic surveys

Gross substrate categories derived from line transects (Table 1) indicate that hard coral was reduced to 11% of the benthic cover in both the protected and unprotected reefs. Because the protected reefs had a higher pre-bleaching abundance of coral than the unprotected reefs, this resulted in a 71% and 44% reduction in hard coral in the protected and unprotected reef sites, respectively. There was also a 65% and 85% loss in soft coral cover in the protected and unprotected sites, respectively. Increases in turf and fleshy algae were also statistically significant. The protected reefs experienced 88% and 115% increases in turf and fleshy algae, respectively, while the unprotected reefs largely experienced a 220% increase in fleshy algae with no appreciable change in turf algae.

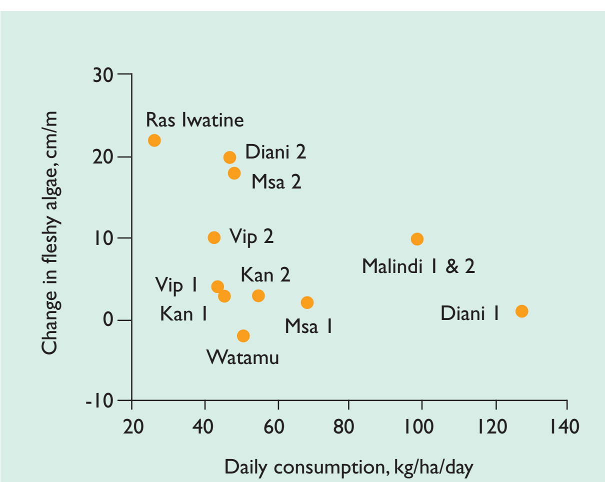
Analysis of the benthic cover approximately one year after the bleaching event comparing 12 sites for the relationships between the loss of coral cover, the estimated daily consumption rates of algae by herbivores and the increase in fleshy algae indicate that there was no clear relationship between the change in fleshy algae with the change in coral, but a weaker relationship with the estimate of herbivory (Figure 2). Some reefs appeared to fit

Table 1. Comparison of the gross benthic substrate in protected and unprotected reefs before and after the 1998 bleaching and mortality event.

	Protected					Unprotected					Kruskal-Wallis test			
	Before		After		Chng	Before		After		Chng	Mngmt		Bleaching	
	x	sem	x	sem	%	x	sem	x	sem	%	F	p	F	p
Hard Coral	39.6	2.9	11.4	1.4	-71.3	20.6	0.7	11.4	2.2	-44.4	1.7	NS	21.9	0.000
Algal Turf	31.0	3.7	58.5	3.6	88.3	49.0	3.6	50.4	2.3	2.9	1.1	NS	9.6	0.002
Calcareous	4.6	2.3	1.8	1.3	-61.0	0.6	0.2	0.5	0.2	-17.8	0.2	NS	0.1	NS
Macroalgae	4.5	1.6	9.8	2.3	114.9	4.0	1.2	12.9	4.3	222.0	0.0	NS	6.7	0.01
Coralline	2.2	0.6	2.6	0.4	16.9	1.9	1.0	2.2	0.9	14.6	11.7	0.001	0.1	NS
Seagrass	6.3	2.2	7.5	2.9	18.9	6.5	1.2	13.1	2.1	103.7	21.9	0.000	0.9	NS
Softcoral	3.8	1.0	1.3	0.3	-64.7	3.1	0.6	0.4	0.2	-85.8	1.3	NS	12.3	0.001
Sand	7.6	1.3	6.8	1.2	-10.3	13.6	4.1	8.4	1.5	-38.6	1.8	NS	0.5	NS
Sponge	0.3	0.1	0.3	0.1	4.7	0.7	0.2	0.6	0.2	-19.1	4.4	0.04	0.0	NS

a clear negative relationship with the change in algae and consumption estimates (Ras Iwatine, Diani 2, Mombasa 2, Malindi 1 and 2, and Diani 1) while another group of sites that experienced only a small or no increase in fleshy algae had low to moderate levels of herbivory (Vipingo, Kanamai, Mombasa 1 and Watamu). Consequently, six of the 12 sites experienced an increase in fleshy algae equal to or more than 10 cm/m over the year, but there was no clear distinction between protected and unprotected reefs.

Figure 2. Relationships between the difference in the fleshy algal cover in the study sites as a function of estimates of herbivory by both fish and sea urchins in sites studied ~13 months after the 1998 bleaching event



DISCUSSION

The warm north-east monsoon of 1998 was unique in being one of the strongest El Niño events since 1877/78 (McPhaden, 1999). Both NOAA satellite data and our in situ logger suggest that the midday water temperatures were above 30° C, 2° C to 3° C degrees above the overall average temperatures, and 1° C to 1.5° C above average monthly temperatures for about two months in March and April. The lack of a significant drop in sea water temperatures during the north-east monsoon windy period in January, indicates that there was also poor water-column mixing during this monsoon as well. In addition, sea water temperature measurements haphazardly taken while diving to 20 m also found that this warm water extended to those depths (Muthiga, unpublished data). Consequently, although Kenyan reefs experience good tidal mixing, having a 4 m tidal range, there was probably little refuge from this warm water for a few months. Bleaching and coral mortality is likely due to this warm water, although other factors such as changes in water chemistry, low wind and water column mixing, and high light or UV penetration probably contributed to the response (Coles & Jokiel, 1978; Hoegh-Guldberg, & Smith, 1989; Gleason & Wellington, 1993; Berkelmans & Willis, 1999; McField, 1999).

Coral mortality became evident a little over a month after the first bleaching began, and the branching species of *Acropora*, *Pocillopora*, branching *Porites*, and *Stylophora* experienced the highest mortality. *Stylophora* appears to have been extirpated from many Kenyan reefs and may be one of the more susceptible genera to temperature fluctuations (Sheppard *et al.*, 2000). Coral community structure after the bleaching largely reflected survival from the bleaching with reefs being dominated by massive *Porites*, *Galaxea fascicularis*, species of *Pavona* and other massive and sub-massive genera in the Faviidae. *Montipora* was still common in the protected sites after the bleaching.

One year after the bleaching nearly half the study sites experienced an increase in fleshy algae of 10% or more, which is an unprecedented inter-annual change for Kenyan reefs in comparison with previous monitoring studies of algal cover (McClanahan & Obura, 1995; McClanahan, unpublished data). The increase in fleshy algae cover could not entirely be explained by fishing levels, the amount of coral loss or herbivory as separate variables. Herbivory was the best predictor of fleshy algae abundance, however, a number of reefs with low to moderate levels of herbivory experienced only small increases in fleshy algae. Fleshy algae in Kenyan reefs can be sensitive to other factors such as physical disturbances by waves and currents (McClanahan *et al.*, 1996; 1999; McClanahan, 1997). In addition, nutrient levels may have played a role, but phosphate and nitrate levels in most Kenyan reefs are above a suggested nutrient threshold (LaPointe, 1999; Obura *et al.*, 2000) and, therefore, algae is unlikely to be limited by these nutrients. The measured increase in fleshy algae is likely to be an interaction between herbivory and other disturbances to algae interacting with the newly created space opened up by coral mortality and high growth rates of algae. This suggests that coral bleaching could be a catalyst in the expansion of erect algae in reefs as reported in other regions that have not been severely influenced by fishing or nutrification (Shulman & Robertson, 1997; McClanahan *et al.*, 1999; Sheppard, 2000).

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Ciguatera risk assessment in the Indian Ocean following the 1998 coral bleaching event

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INTRODUCTION

During the last two decades, countries from the Indian Ocean region have experienced a variety of seafood poisonings involving coral reef fishes (ciguatera), turtles (chelonitoxism), sharks (carchatoxism) and sardines (clupeotoxism) (Quod & Turquet, 1996; Turquet *et al.*, 2000a). The ecotoxicological process of ciguatera fish poisoning (CFP) is caused by benthic dinoflagellates from coral reefs (*Gambierdiscus*, *Prorocentrum*, *Ostreopsis*) that are epiphytic on algal turfs, coral rubble and macro-algae. They produce potent neurotoxins that accumulate in herbivorous marine animals and are transferred to higher levels of the food chain by carnivorous fish. These microalgae are natural inhabitants of coral reefs and become problematic when densities reach critical levels.

There is currently a global increase of harmful algal blooms (HABs) with consequences for both human health and aquaculture. Further, the intensity, frequency and distribution of HABs has increased in relation to coastal marine environment degradation. Furthermore, increased media focuson HAB's has increased public awareness of the problem. HABs are also associated with natural phenomena such as cyclones, resulting in endemic hot spots for CFP. Unusual climatological conditions such as the El Niño-Southern Oscillation (ENSO) are also associated with blooms of toxic planktonic *Pyrodinium bahamense* in Philippines (Hallegraeff, 1995) or benthic *Gambierdiscus toxicus* in French Polynesia (Bagnis *et al.*, 1992). Benthic microalgae are particularly influenced by disburbances that result in mass coral mortality, due to the large increase of dead coral surfaces, which form a good substrate for recruitment of algal turfs and associated epiphytes. This paper is the preliminary report of a Ciguatera Risk Assessment (CRA) to document any increase in HABs resulting from the 1998 El Niño related mortality of corals.

Descriptions of the key-features of contamination levels of coral reef ecosystems serve numerous purposes. First, quantification of the toxic species abundance is the first step to identifying the potential risk of reef fishes after natural or anthropogenic disturbances (e.g. cyclones, bleaching, pollution). Second, micro-algae and particularly dinoflagellates are biological indicators of reef health and could form part of the reef monitoring activities and be included into the general monitoring database. Monitoring of coastal waters in relation to HABs is also promoted by the IOC of UNESCO.

A large scale, rapid assessment of CRA was undertaken in countries from East Africa, South Asia and Indian Ocean islands. CRA packages were sent to CORDIO national focal points in order to collect 10 samples, fix biodetritic fractions and send them back for analysis. As capacity of the countries stakeholders was unequal and adequate equipment not available, it was proposed to assess ciguatera risk by implementation of a semi-quantitative methodology. To avoid seasonal variability, collection of all samples was done in November 1999.

PRELIMINARY RESULTS AND DISCUSSION

CRA sampling on coral reefs in the Indian Ocean region showed that degraded reefs where dead corals are covered by algal turfs are contaminated by a multispecific assemblage of micro-algae (Table 1). Cyanobacteria, diatom and dinoflagellate densities vary from place to place due to environmental different conditions. Special attention was focused on the potentially toxinogenic dinoflagellate *Gambierdiscus cf toxicus* (Figure 1). Available information from CRA and other sources did not find a strong correlation between contamination levels and CFP endemicity (Figure 2). It is thought that in many places, an increase in population may not be sustainable in time. Confirmation of this pattern was confirmed for Mayotte where local authorities (SPEM) found a bloom in November 1998 on Surprise inner reef, followed by a drop in 1999.

Investigation of the impact of 1998 coral bleaching was also initiated as part of the REP/COI (Regional Environment Programme/Indian Ocean Commission with support of the EU) (Turquet *et al.*, 2000a). Two studies were carried out. First, the Vigitox project was started to survey the abundance of toxic micro-algae in bleached coral reefs and the potential toxicity of herbivorous fishes *Ctenochaetus striatus*. Degraded reefs were visited in January 1999. Second, a socio-economic survey was initiated to estimate the impact of ciguatera on human activities. Only one set of data was collected but no increase in seafood poisoning outbreaks was detected in 1999.

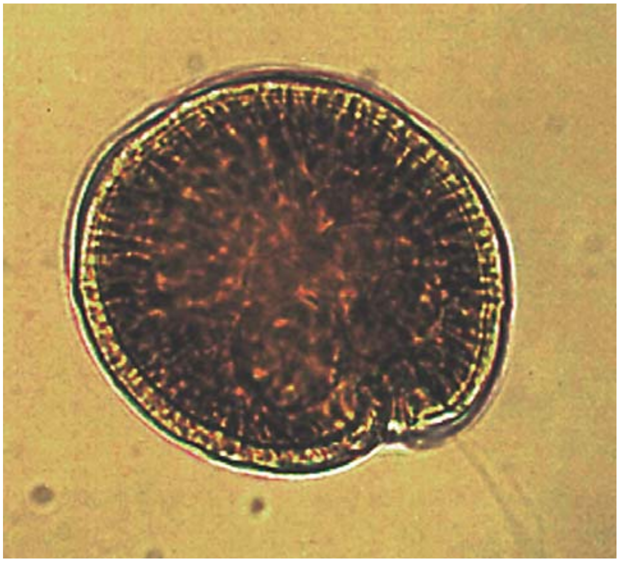


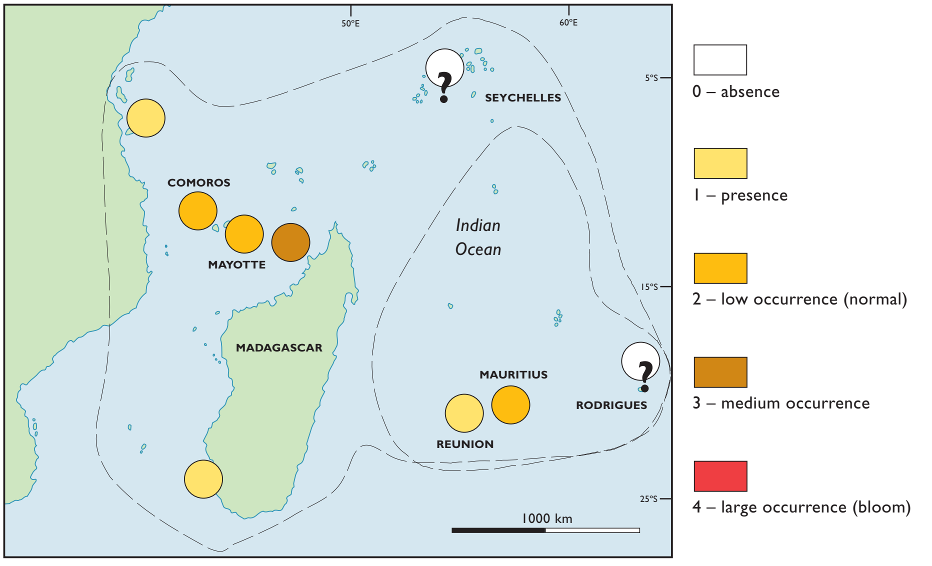
Figure 1. *Gambierdiscus toxicus*, the main progenitor of Ciguatera Fish Poisoning (CFP). Human poisoning may occur when densities of toxic strains are reached in degraded reefs.

The present-day distribution of *Gambierdiscus* and associated species is significantly larger that previous descriptions. Values observed during the first regional survey (Bagnis, 1980) were low. At present, we have learned that CFP progenitors are now widespread and that critical levels are reached in some places. For example, in Mayotte, iterative contamination of the reef ecosystem started in 1984 with circumstantial evidence for

Table 1. Abundance of micro-algae, with consideration of three toxinogenic Dinophyceae genera. (0) absence; (1) presence; (2) low (normal) density; (3) medium density; (4) large (bloom) density.

Country	Cyano-bacteria	Diatoms	Dino-flagellates	*Gambierdiscus* spp.	*Prorocentrum* spp.	*Ostreopsis* spp.
Comoros	1.9	1.6	2.3	0.4	1.5	2.4
Madagascar	2.3	1.7	1.8	0.3	1.7	1.7
Mauritius	2	1.4	1.8	0.8	0.8	2.2
Mayotte	1.7	1.4	1.7	1.6	0.9	1.4
Reunion	2.3	1.8	1.6	1	1	1.5
Tanzania	2	0.7	1	0.3	0.7	0.5

Figure 2. Map of the Indian Ocean with areas where *G. toxicus* has been reported and areas of ciguatera endemicity. ? indicates that the abundance of *G. toxicus* is unknowns.



coincidence with bleaching events in 1983/84, 1987/88 and 1997/98 (Turquet *et al.*, 2000b).

The question of whether the increase of HABs may affect marine biodiversity and human activities represents a growing problem. In the Indian Ocean, currently the main impacts of HABs are through human poisoning and the bans on potentially edible fishes from safe areas, but human activities such as aquaculture, artisanal fisheries, and tourism may be seriously affected.

Harmful algal blooms need management to mitigate their impact on society. Assessment of the toxic microalgae required capacity building, which is at present developed in the East Africa region by IOC-HAB of UNESCO with support from Sida. Most importantly, awareness of decision-makers in coastal management (aquaculture, pollutants loading, etc.) should be raised on increase of harmful algal bloom as a probable outcome.

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Evaluation of succession and coral recruitment in Maldives

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ABSTRACT

A SST anomaly occurred in April-May 1998 resulting in widespread bleaching of zooxanthellate reef invertebrates in Maldives. Subsequently, bleaching induced mortality was unprecedented. This study reports on the initial stages of reef recovery and describes initial results from integrated studies on coral settlement and recruitment. Preliminary results from the *in situ* recruitment surveys indicate that recovery of the reef at Feydhoo Finolhu is not recruitment limited. At the study site, mean coral recruitment ranged between 19 individuals per m^2 at a depth of 10 m and 26 individuals per m^2 at 5 m. Overall, the ratio of branching corals to massive corals was approximately 10:90. The most abundant recruit was *Pavona* (>57%). Despite the severity of the coral mortality in Maldives these data suggest that the capacity of this reef system to recolonise degraded reefs, through a supply of coral planulae from surviving colonies is high.

INTRODUCTION

In 1998, unprecedented bleaching and subsequent mortality of corals was reported in Maldives (Naaem *et al.,* 1999; Rajasuruya *et al.,*1999; Wilkinson *et al*., 1999) Bleaching of zooxanthellate reef invertebrates was widespread but most severe in shallow reef-flat areas, where up to 90% bleaching was observed. However, bleaching was also observed at depths greater than 30 m. Subsequently, a significant reduction in live coral cover has been recorded at all sites studied with average live coral cover decreasing from 42% to 2% on shallow reef flats (Zahir *et al.,* 1999). Maldives is composed of low-lying coral atolls and cays and is particularly vulnerable to the potential loss of protection formerly provided by healthy accreting reef flats. The consequences of the bleaching induced coral mortality may have a profound affect on the calcium carbonate structure that provides a foundation for the islands' survival.

Where mortality has been severe reef recovery will largely be dependent on the supply of larvae. However, recruitment processes are subject to high levels of natural variability (Hughes *et al.,* 1999). Thus, time scales for reef recovery processes are difficult to predict and largely depend on site-specific conditions. Factors that may influence the recovery processes include: the severity of bleaching and subsequent mortality of coral within a given site, the supply of coral larvae, the availability of consolidated reef framework for recolonisation by coral recruits and the life histories of the dominant corals in the surviving community.

Studies of coral recruitment fall into two categories: those that quantify the number of visible juvenile corals appearing on the natural reef, and those that use natural or artificial settlement tiles to estimate larval supply. The latter involve a fixed period during which the settlement tiles are deployed on the reef, before they are removed and examined microscopically in the laboratory to estimate non-visible settlement (i.e. coral spat). Therefore, these methods concentrate on different phases of the life history of corals and community development, with spat abundance being linked to larval availability and dispersal whereas the density of juvenile corals within an area reflects post-settlement mortality. Few studies have combined both approaches in one study (see Rogers, 1984). Consequently, there is a lack of information on the relationship between the relative measure of larval supply and the actual recruitment of corals to a site.

The purpose of the present investigation was two fold: 1) to investigate spatial and temporal patterns of coral recruitment in Maldives using permanent quadrats and experimental settlement tiles, and 2) to estimate post-settlement mortality by comparing the abundance and community composition of juvenile corals that settle on the reef with the abundance and community composition of coral spat that settle on experimental tiles. This paper reports preliminary data from the CORDIO funded research programme conducted in Maldives.

METHODS

Settlement tiles

An initial pilot experiment was conducted in November 1999 to test the suitability of terracotta-like tiles at Feydhoo Finolhu reef. After 3-4 weeks the terracotta-like tiles became soft and were breaking up in sea water and therefore, were considered inappropriate for this study. Subsequently, glazed ceramic tiles were found to be suitable for the settlement tile studies. When placed on the reef two tiles were placed together with the glazed surfaces facing each other ensuring the unglazed surfaces were exposed and available for potential settlement of coral larvae.

Tiles were deployed during mid December 1999 at three sites in North Malé atoll that were selected for sampling every three months. At each site two sets of tiles (10 pairs of tiles per set) were deployed at two depths (5 m and 10 m). Tiles were orientated vertically or near-vertically, as this has been shown to be the preferential orientation in other studies (Bak & Engel, 1979; Birkeland, 1977). The tiles were fixed to hard substrate such as raised buttresses on the natural reef using push mount plugs and cable ties (see Plate 1). Experimental tiles are only suitable for coral settlement once a fouling community has developed on the artificial substrates during a process called conditioning. This is achieved in the experimental design by deploying two sets of tiles several weeks prior to the period during which settlement of corals on the tiles will be recorded (i.e. overlapping tiles for settlement). Every three months, 10 pairs of tiles will be retrieved from each depth and examined under a binocular microscope for coral spat. Information will be collected describing the orientation, size and condition (live or dead) and, wherever possible, each coral spat will be identified to genus. At the same time, the 10 pairs of tiles that were collected will be replaced using the same mounts ensuring that tiles that have been conditioned are always available for coral settlement.

Permanent quadrats

Permanent quadrats were mapped *in-situ* to obtain data describing rates and taxonomic patterns of coral recruitment. Ten permanent quadrats ($0.25 m^2$) were marked

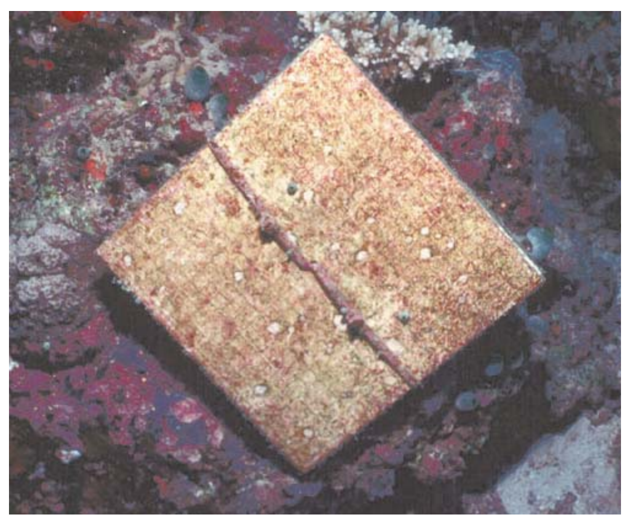


Plate 1. View of one of the ceramic settlement tiles deployed onto hard reef substrate on the reef slope. Note the cable ties that attach the tile via a push mount plug to the reef. Small colonies of the colonial ascidian *Didemnum molle* have settled onto the tile after 2 months.

within each depth zone (5 m and 10 m). Individual coral colonies within the quadrats (recruits, survivors and fragments of survivors) were identified to the lowest taxonomic level. Colony sizes (maximum and minimum diameters) were measured with vernier calipers and the condition of the colony (live, dead, partial mortality) noted. In this study, recruits were defined as the recently settled individuals that have survived for a period after settlement and are large enough to be detected in the field (usually around 5 mm). This represents a measure of new individuals entering a population. Observers were trained to recognise fragments of older colonies and surviving colonies. If observers could not distinguish between fragments, survivors or recruits they were omitted. Line intercept transect data were also collected describing the benthos at each site to determine the percentage cover of live coral and other major constituents of the reef benthos.

Sites

Sites were selected on the basis of 1) data from previous studies and 2) protection from interference from other reef users. Three sites were selected in North Malé Atoll (Figure 1): the reef at Bandos Island, (a tourist resort 2.5 km from Malé), Udhafushi (a faro to the north west of Bandos) and the reef at Feydhoo Finolhu (1 km north west of Malé). An additional three sites were selected in Vaavu Atoll (Figure 2): Wattaru, Vaavu (a patch reef), and Foththeyo. In North Male atoll sites will be sampled every three months and in Vaavu atoll the sampling frequency will be every six months to coincide with the GCRMN reef monitoring programme.

PRELIMINARY RESULTS

Coral settlement tiles

An initial examination of a sub-sample of settling tiles during February 2000 revealed no coral recruitment during the first two months of the study.

Initial evaluation of succession - The composition of the benthic community was examined under a binocular

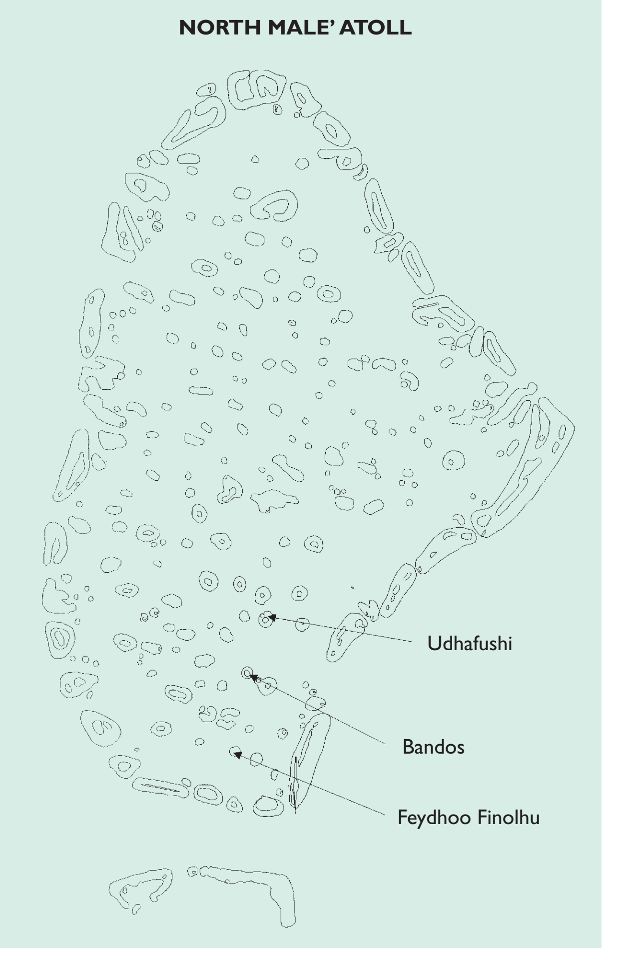


Figure 1. Map of the North Malé atoll showing the location of the study sites.

microscope and the percent cover of the main benthic colonisers estimated visually. Space on the tiles was rapidly colonised. The outer surfaces were dominated by green algae (40%) and the inner surfaces by bryozoans (50%), polycheates (10%) and ascidians (5%). Both inner and outer tiles had a covering of red coralline algae at the outer margins (5 mm –10 mm).

Next steps - The first set of tiles will be retrieved in mid March. Additional information will be collected on ben-

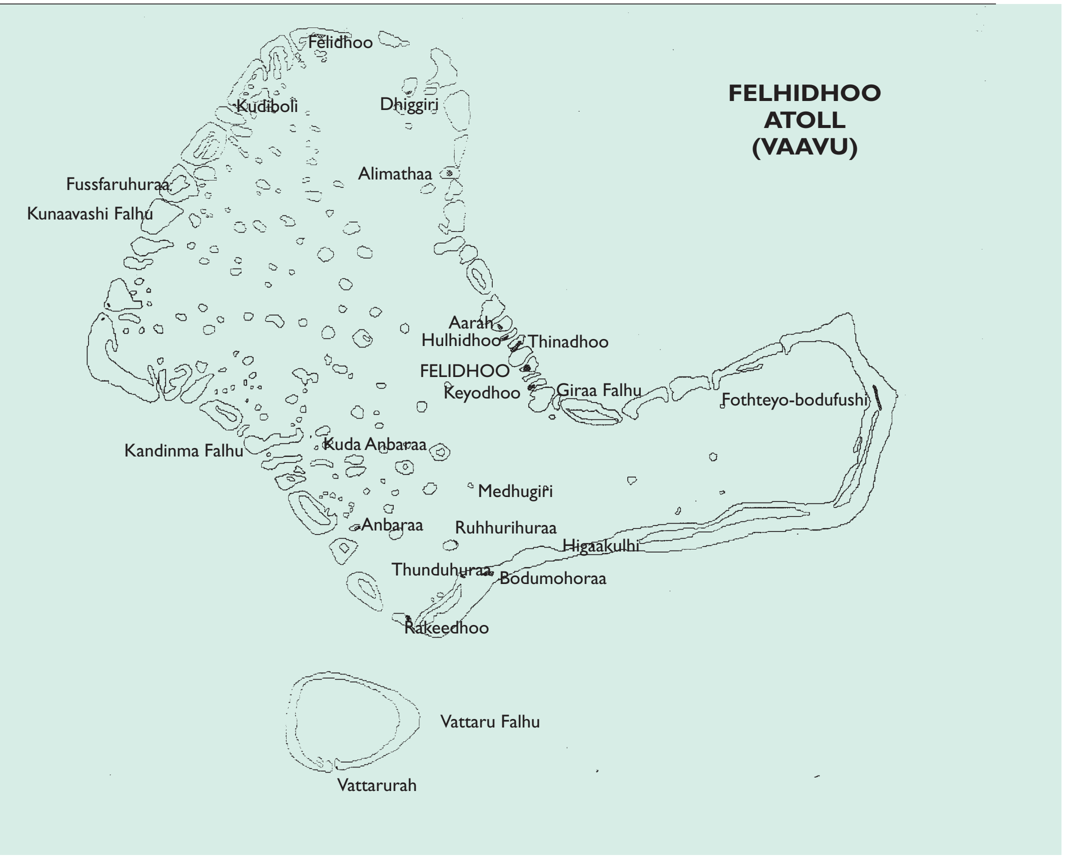


Figure 2. Map of the Vaavu atoll showing the location of the study sites.

thic community development to evaluate patterns of succession and competition. A grid point method will be adopted to estimate cover, species richness and community development (see Fairfull & Harriot, 1999).

In-situ mapping for coral recruitment

Initial results from Feydhoo Finolhu, North Malé atoll

A total of 88 coral recruits were found within 26 m^2 and 19 m^2 sampled at depths of 5 m and 10 m respectively. At 5 m recruits constituted 49% of all corals observed, fragments 35% and the remaining 16% were colonies that had survived the bleaching. At 10 m more adult colonies survived (28%) and persisted as fragments (42%). Only 30% of colonies observed were recruits (see Figure 3). Despite the sample size being too small to draw solid conclusions, recruitment was slightly higher at 5 m than at 10 m whereas fragments and survivors were more abundant at 10 m.

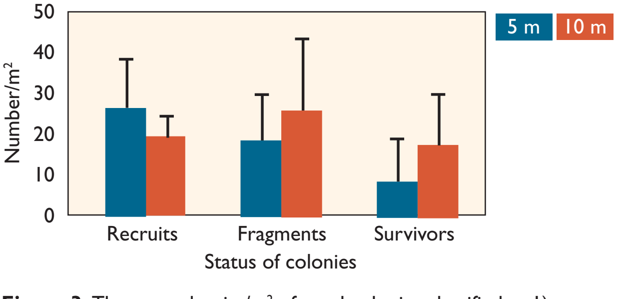
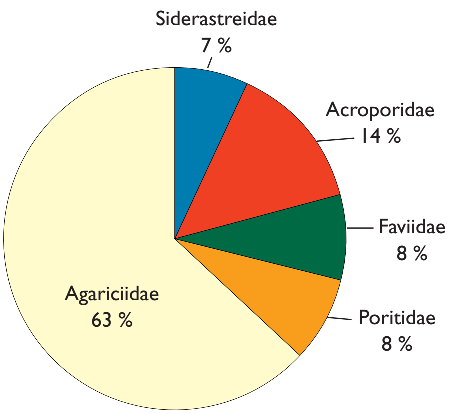


Figure 3. The mean density/m^2 of coral colonies classified as 1) recruits, 2) fragments and 3) survivors at two depths at Feydhoo Finolhu in February 2000. Error bars = +1 SD between replicate quadrats.

The taxonomic patterns of coral recruits at each depth are shown in figure 4. Overall, the ratio of branching to massive colonies was 14:86 and 10:90 at 5 m and 10 m respectively. Among branching corals, only Acroporiidae (14%) was recorded at 5 m whereas colonies of both Acroporidae (7%) and Pocilloporidae (3%) were observed at 10 m. At 5 m Agariciidae (*Pavona* spp.) was the dominant coloniser (63%) followed by Acroporiidae (14%), Poritidae (8%) and Faviidae (8%). At 10 m



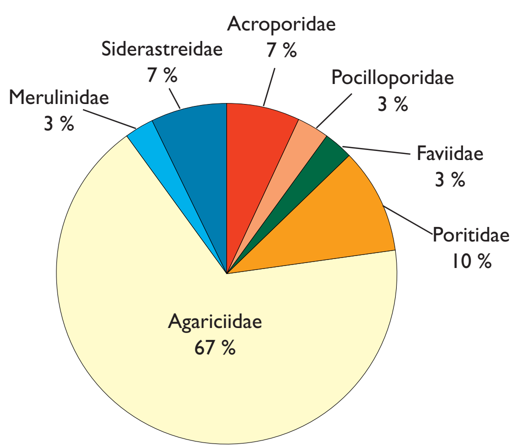


Figure 4. Pie charts showing the taxonomic patterns of coral recruitment at two depths at Feydhoo Finolhu recorded in February 2000 (21 months after the onset of the 1998 coral bleaching event).

the taxonomic pattern was similar with agariciids constituting 67%. However, the remaining 33% of corals were comprised of a more diverse assemblage than at 5 m with seven families represented.

The dominant massive corals within the category of fragments were Agariciidae (*Pavona* spp.) (56%) followed by Poritidae (22%), whereas among survivors Faviidae colonies were dominant (34%) followed equally by Poritidae and Agariciidae (22%). No fragments of *Acropora* were recorded within any of the quadrats sampled and no colonies with staghorn growth form were observed at this site. General observations around the reef found small numbers of *Acropora* colonies, with a digitate growth from, present below 10 m but quantitative surveys were not conducted. Observers noted that branching species of *Acropora* and *Pocillopora* tended to occur in crevices, over-hangs and under dead standing colonies. Thus, it is possible that this study may have under-estimated the density of these colonies due to difficulties in placing quadrats in such locations. To test this hypothesis a trial study will compare data obtained from belt transects (200 cm x 20 cm) and quadrats within the same location to determine the most appropriate survey protocol.

Size frequency distributions of the coral recruits at both depths are shown in figure 5. Ninety-six percent of all coral recruits had mean colony diameters less than 40 mm. Only two recruits, one *Acropora* and one *Pocillopo-*

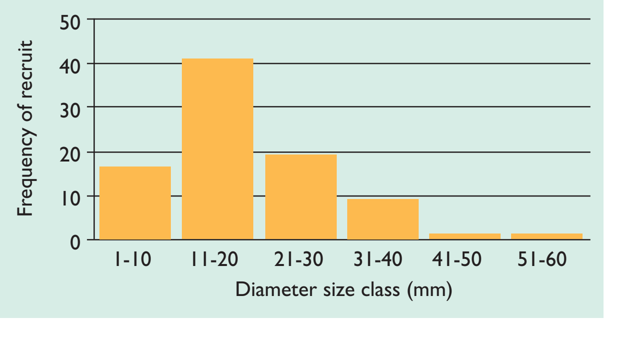


Figure 5. Size frequency distribution of coral recruits (n= 88) at Feydhoo Finolhu. Data for 5 m and 10m are combined.

ra, exhibited colony sizes greater than this. Overall, branching corals were not well represented in the surveys. Colony sizes of *Acropora* species ranged from 13 mm to 60 mm, which may represent wide variation in rates of growth between different growth forms (e.g. digitate versus tabular) or different cohorts.

Reef Structure

At a depth of 5 m the cover of live coral averaged 5.4% and 8.4% at 10 m while rugosity was 1.17 at 5 m and 1.34 at 10 m respectively. At both depths algae was the dominant living cover (> 30%) and coral rubble (> 30%) the dominant non-living cover.

DISCUSSION

Preliminary results from the *in situ* recruitment surveys indicate that the reef at Feydhoo Finolhu is not recruitment limited. Mean recruitment ranged between 19 individuals per m^2 at 10 m and 26 individuals per m^2 at 5 m to depth at the study site. Among recruits the overall ratio of branching corals to massive corals was approximately 10:90. The dominant coloniser was *Pavona* (> 57%). Branching species represented less than 10% of all recruits recorded (*Acropora* 8% and *Pocillopora* 1.2%). Broadscale assessments of other reefs in North Malé atoll indicate that similar levels of coral recruitment have occurred since the bleaching event. McClanahan (in press) found a total of 29 recruits per m^2, with *Pavona* being dominant (11.7 recruits per m^2), and the previously abundant branching and encrusting species (*Acropora*, *Pocillopora* and *Montipora*) had recruit densities less than 0.65 recruits per m^2. Clark (this volume) describes recolonisation of artifical reef structures and natural reefs following the bleaching event. Branching corals (*Acropora* and *Pocillopora*) were dominant colonisers constituting 67% of the post-bleaching coral community on the shallow artificial reef structures. The remaining 33% of colonies were massive corals. This represented a marked shift in community composition when these data are compared with surveys conducted prior to the bleaching (1990-1994) in which branching corals represented 95% of the coral community and massive corals only 5%. However, at this site, taxonomic patterns may be linked to site settlement preferences of the different taxa on the artificial reefs (Edwards *et al.,* in press).

Patterns of reproduction and spawning are important in understanding the subsequent recruitment of corals to the reef. The only information on spawning of corals in Maldives is that reported by Sier & Olive (1993) for *Pocillopora verrucosa*. Broadcast spawning was inferred from the disappearance of mature gametes between late March and April. Clark (unpublished data) detected small juveniles of *Acropora* (size range 5-20 mm) on the reef between February and April between 1991-1994. This suggests that coral spawning occurs during early summer when sea-surface temperatures are increasing towards their annual maximum. In Tanzania, a recent study (Nzali *et al.,* 1998) reported maximum coral recruitment to settlement tiles occurred during April which also coincided with the period of the annual maximum sea-surface temperature.

In this study, the initial surveys of recruits began 21 months after the 1998 bleaching event occurred. Therefore, it is important to distinguish between settlement that occurred immediately after the bleaching event (1998/1999) and settlement during the second recruitment season (1999/2000). Given the large colony diameters of some *Acropora* species (>15 cm) present on the reef slope, it is possible that settlement occurred immediately after the bleaching event or even before but was not visible during initial bleaching assessments. This suggests that juveniles and recruits may be less susceptible to bleaching induced mortality than adult colonies. Mumby (1999) reported greater resilience to bleaching amongst small recruits (2 mm – 20 mm) compared with the adult population during the 1998 bleaching incident at Glovers Reef in Belize. As further data become available information on colony sizes and analysis of size-frequencies will provide insights into population development following the bleaching event.

Despite the severity of the coral mortality in Maldives, this study has demonstrated that this reef system possesses the capacity to recolonise degraded reefs, through a supply of coral planulae from surviving colonies.

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Studies of bioerosion on coral reefs of Tanzania

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INTRODUCTION

Responses of coral reef ecosystems to disturbance

Historically, coral reefs have been subjected to a range of disturbances including major sedimentary events from flooded rivers (Robertson & Lee Long, 1990), storm events and changes related to long-term patterns such as El Niño (Glynn, 1985). Reef ecologists have accumulated a great deal of evidence implicating disturbance as a major influence on reef ecology (Connell, 1978). Increasing global temperatures, eutrophication in coastal areas and the physical damage that is becoming so widespread is potentially reducing the reefs intrinsic ability to cope with such perturbations. Primary production on a reef is largely dependent on a variety of algae including the symbiotic algae living in hard corals (zooxanthellae). The level of production that any of these components might attain is determined by a variety of physico-chemical (e.g. hydrodynamics and nutrients) and biological factors (e.g. grazer-communities).

Furthermore, the addition of nutrients such as nitrogen and phosphorous has the potential to increase the growth rates of some algae and heterotrophic invertebrates relative to corals (Littler *et al.*, 1991; Sammarco & Risk, 1990). As a consequence, many reef ecologists warn that eutrophication of waters surrounding coral reefs may alter patterns of dominance on these reefs from corals to various forms of fleshy and filamentous algae and bioeroding sponges (Rose & Risk, 1985; Sammarco & Risk, 1990). In practice, however, there are a number of factors that prevent this from occurring. Perhaps the most important is herbivory (Hatcher & Larkum, 1983). Vertebrates and invertebrates comprising the grazing community consume the plant production (Hatcher & Larkum, 1983) and often inadvertently ingest limestone from the skeletons of corals and epiphytic micro-invertebrates while grazing. Some investigators have argued that the abundance of grazers is the main limitation to primary production on coral reefs (Larkum & Steven, 1994). Indeed, the abundance of fleshy algae is generally very low on healthy reefs that support reasonable populations of herbivores. However, declines in grazing pressure through overexploitation of herbivorous fauna can shift the balance in favour of the algae allowing them to out-compete and eventually exclude corals from the reef community.

The products of photosynthesis by zooxanthellae are the primary source of energy used by corals for accretion of calcium carbonate ($CaCO_3$) and skeletal growth (Muscatine, 1990). Mass bleaching on the scale reported in 1998 can alter the balance between net accretion and net decay of $CaCO_3$ from the reef framework. Subsequently, the loss of zooxanthellae through bleaching can reduce the rate of accretion of $CaCO_3$ into the skeleton of the coral. This imbalance may also be mediated by grazers, bioeroders and organisms whose actions directly affect benthic community structure (Done *et al.*, 1996). The persistence of a coral reef, in the long-term, requires that its overall rate of calcium accretion equals or exceeds losses from biological and physical erosion and transport of sediment away from the reefs (Done *et al.*, 1996).

Bioerosive processes

An important aspect for the maintenance of coral reefs is that the skeletal structure is broken down by bioeroding organisms into sediment, which form sandy bottoms and beaches and is an important factor of the inorganic pathway. Bioerosion is divided into two types; external bioerosion, which is usually performed by different types of fish and sea urchins (McClanahan, 1994) and internal bioerosion by polychaetes, sipunculans, molluscs, barnacles and sponges (Sammarco & Risk, 1990).

Risk and Sammarco (1982) demonstrated a significant positive correlation between the degree of internal bioerosion in dead coral and reduced grazing pressure. This correlation can be explained in three ways. First, a decrease in the level of biological disturbance imposed by grazers on newly settled larvae of endolithic borers might reduce post-settlement mortality among these bioeroders allowing greater numbers to survive to adulthood (Sammarco, 1980). Second, reduced grazing pressure enables algae to flourish which, in turn, provides a refuge for bioeroders. Third, food and nutrient levels might be more abundant in the associated algal turf and sediment (Klumpp *et al.*, 1988). Risk and Sammarco (1982) also demonstrated that predation may influence the rate of internal bioerosion by affecting population dynamics within the endolithic bioeroder community. Sammarco *et al.* (1987) reported that boring by sponges with large exposed papillae (*Cliothosa hancocki*) increased significantly when grazing is reduced within damsel fish territories, while the abundance of more cryptic boring sponges decreased.

Further, on reefs across the central region of the Great Barrier Reef, Australia, Sammarco and Risk (1990) reported that total internal bioerosion decreased significantly with distance offshore. The greatest decreases in abundance occurred in bivalves and sponges. It was suggested that because bivalves and sponges are filter feeders they may gain more food in the more nutritious inshore waters.

Grazing by fish, particularly scarids and acanthurids, and sea urchins cause high levels of external bioerosion of dead foliose coral substratum. The microtopography of coral substratum is altered greatly by grazing, suggesting that these activities might hinder settlement of internal bioeroders. Although small fish and galatheid crabs contributed only negligibly to bioerosive activities, some of them appear to prey directly on cryptofauna such as boring sponges. Therefore, predation may have an important influence on the population dynamics of endolithic communities (Sammarco *et al.*, 1986).

Infestation by boring organisms also weakens the corals which may make them more susceptible to damage from catastrophic high-energy events such as storms and cyclones (Highsmith, 1982) causing them to become easily detached from the substratum (Highsmith *et al.*, 1980). The degree to which the skeleton has been eroded will also be an important factor determining the resistance of corals to wave shock (Highsmith, 1982; Tunicliffe, 1983). Further, bioerosion may contribute to control of nutrient and gas fluxes on coral reefs (Tudhope & Risk, 1985) and others have stated that it can affect the morphology of carbonate coastlines (Acker & Risk, 1985). On severely bleached reefs, bioerosion could contribute to the total erosion of corals such that entire reefs could be eliminated if rates of coral recruitment do not increase (Reaka-Kudla *et al.*, 1996).

A review of the response of reefs to eutrophication suggests that the maintenance of fish populations and their feeding responses as well as the physical structure of the reef may be important factors inhibiting overgrowth of coral reefs by algae and degradation by internal bioeroders (Sammarco & Risk, 1990). These studies suggest that it is often the combined effect of the loss of reef consumers, reef structure and eutrophication that cause drastic changes in reef ecology.

Aims

If environmental factors and the observed trends in bioerosion are functionally related, then eutrophication or increased productivity in the waters surrounding coral reefs may promote increased levels of internal bioerosion which, when combined with decreases in grazing pressure by fish, can further accelerate bioerosion and the degradation of reefs that are already seriously affected by bleaching. The aim of this study is to investigate how some of the main regulators of coral reef ecosystems affect rates of bioerosion of the reef structure and key coral species. Also, it is important to determine whether different parts of the reef such as reef flat, reef crest and slope have different erosion rates to be able to predict how disturbance affects the different parts.

EXPERIMENTS AND PRELIMINARY RESULTS

Bioerosion across a nutrient concentration gradient

Between February and April 1999, a study of the distribution and abundance of bioeroders was conducted across a nutrient gradient from the runoff of Zanzibar town situated on the west coast of Unguja Island. Four sites at progressively increasing distances from the sewage outflow of town were chosen: Chapwani Island nearest the runoff, Bawe Island 2 NM from town, Chumbe Marine Park 7 NM from town and Mnemba Reef off the east coast of Unguja Island. Coral-rubble from the bottom and dead standing corals of the genera, *Acropora*, *Porites* and the subgenus *Synaraea* were collected from each site. The percentage erosion from different eroding groups (i.e. sponges, polychaetes, sipunculans, bivalves and balanoides) was measured and the abundance of each taxon in the dead coral was determined to identify higher-level interactions between water quality and the composition of the bioeroding community. Preliminary results show that the variation of bioeroding organisms within and among sites was very high. This indicates that the common method of using corals for bioerosion studies where the time of mortality is unknown is questionable.

Succession of the bioeroder community

Although it is known that bioeroding organisms are more active in dead corals than in living corals, the relationship between the time since mortality and the rate of erosion and colonisation of the skeleton by various bioeroders is poorly documented. In order to determine temporal changes in the rate of bioerosion and also patterns of succession within the bioeroding community, skeletons of corals at Mafia Island killed by El Niño induced coral bleaching of 1998 were sampled after one year (March 99) and 1.5 years (October 1999). Additional samples will also be taken 2.5 years and 3 years after the bleaching event.

Pieces of dead *Acropora formosa* were collected from 2.5 m x 2.5 m quadrats. All corals in the quadrats were killed in March 1998. The corals were sectioned longitudinally and the percentage area eroded by different taxon (i.e. polychaetes, sponges, sipunculans, barnacles, bivalves) were calculated. All bioeroding organisms found in the pieces of corals were collected and identified. To date, only preliminary results are available but a small increase in bioerosion was recorded between March 1999 and October 1999.

Experimental gradient in grazing pressure

A gradient in fish grazing pressure was established experimentally by placing the dead coral substratum under three experimental conditions: 1) fully exposed to natural levels of fish grazing; 2) protected from grazing fish within cages; and 3) beneath cage sides without top and two sides to control for decreased light and current while allowing access to grazing fish and invertebrates. The coral samples will be exposed for 12 months before analysis.

Temporal patterns of bioerosion within habitats characterised by different fish communities

Two sites were selected outside Zanzibar, Bawe Island and Chumbe Island. Bawe is an island situated near Zanzibar town where fishing is permitted. Chumbe Island is a marine park situated 7 NM from Zanzibar town and is protected from fishing. Three different fish communities were identified at both sites. At both sites, living colonies of *Acropora formosa* were collected and killed before being placed within the habitats in which each of the three different fish communities resided. The coral samples will be exposed for 24 months before analysis.

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Transplantation of coral fragments

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INTRODUCTION

Coral reefs globally are increasingly under threat from environmental and anthropogenic factors, particularly the recent widespread bleaching and mortality of corals due to the temperature anomaly recorded during the 1997/98 El Niño. The active rehabilitation of reefs may be necessary in some locations. Different rehabilitation methods require development for use in different conditions according to the constraints of area, availability of funding and reasons for rehabilitation. A number of studies have involved transplantation of parts of adult corals at a variety of technical, financial and spatial scales. Methods have included placement of loose staghorn *Acropora* branches (Bowden-Kirby, 1997; Lindahl, 1998) on suitable substrates, cementing corals to natural substrates using cement or epoxy-type glues, and cementing corals to movable bases (Obura, unpublished data). Transplantation can be used for management purposes in the rehabilitation of reefs (Harriott, 1988), and in conjunction with transplants of wider reef communities (e.g. Muñoz-Chagin, 1997).

The primary objective of this study is to investigate the capacity of coral transplants, covering a range of genera with different growth and life history strategies, in the repair and rehabilitation of degraded reefs. A secondary objective is to develop a suitable (efficient, economical and practical) methodology for the transplant procedure. Higher level objectives can be investigated in the long term, including three-dimensional complexity and diversity in the vicinity of the transplants. The study is conducted in the Mombasa Marine National Park, Kenya.

METHODS

Coral species used were *Porites lutea*, *Pavona cactus*, *Montipora spongodes*, *Echinopora gemmacea*, *Acropora* sp. (f. corymbose), *Hydnopora microconos* and *Goniopora* sp. Small fragments were broken off parent colonies and immediately fixed using an epoxy "Quickset putty" as the cementing agent to a) natural reef substrate cleaned by scraping with a wire brush and b) small conical cement bases to enable movement of the fragments, held in place on the reef in holes on an elevated rubber rack (Figure 1). Coral fragments were left for >2 days to acclimatize to the manipulation. Size was measured at approximately 30 day intervals, recording height and base diameter for branching species, and maximum and a perpendicular diameter to compute projection area for sub-massive species. Losses and mortality were recorded and sample sizes made up by addition of new fragments. The results of three and four intervals of growth are reported here (number of intervals are varied due to the time of starting different species).

RESULTS

Positive growth was recorded only for *Echinopora*, *Hydnopora* and *Porites* on the racks, and for *Acropora* and *Montipora* on natural substrate (Figure 2). Negative growth was recorded in the remaining instances, with *Pavona* and *Goniopora* displaying negative growth rates for transplants on both natural substrate and on the racks. Therefore, transplants of branching corals appear to do better on natural substrates and transplants of submassive corals grow better on the racks, with the excep-

Figure 1. Coral fragments transplanted to natural substrate (below) and onto conical cement bases for fixing onto raised rubber racks (left).

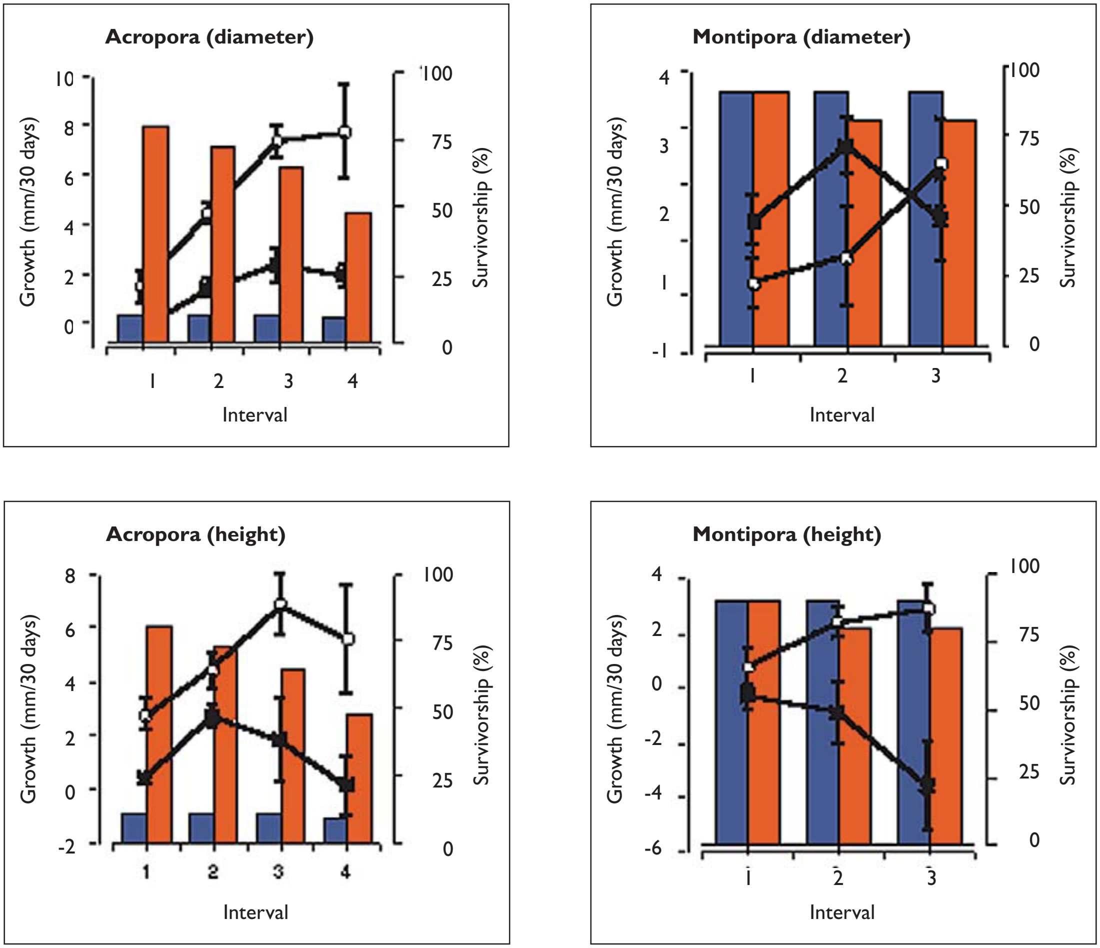
tion of *Goniopora* and *Pavona*. Growth of the corals, particularly on natural substrate, was better during rough-water conditions, decreasing during the calm transition between monsoons in November-December. During this time large amounts of fine silt accumulate on reef surfaces and are likely to stress benthic organisms.

Survivorship was high (> 80%) for all transplants except for *Acropora* on both racks and natural substrate and *Echinopora* on natural substrate. *Porites* suffered no losses or mortality on either the racks or natural substrate. *Echinopora* on the racks and *Hydnopora* on the substrate also exhibited a 100% survival rate. The lowest survival value was for *Acropora* on the racks, caused by predation by *Drupella* during the first interval. Replacement fragments survived at close to 90% for the remaining intervals presented here. *Acropora* and *Echinopora* showed significant long-term decline of substrate transplants.

DISCUSSION

The primary objective for this study was to investigate the capacity of transplants of different species of corals in the rehabilitation and repair of degraded reefs. One of the main findings so far was that sub-massive corals tend to fare better on elevated racks while branching species tend to do better when transplanted onto natural substrates. The difference is most likely due to algal competition and the accumulation of sediment in algae adjacent to the coral tissue margin (especially in calm conditions) and overgrowth of the coral by algae, that suppresses growth of non-erect corals. However, why branching corals should fare less well on racks is not

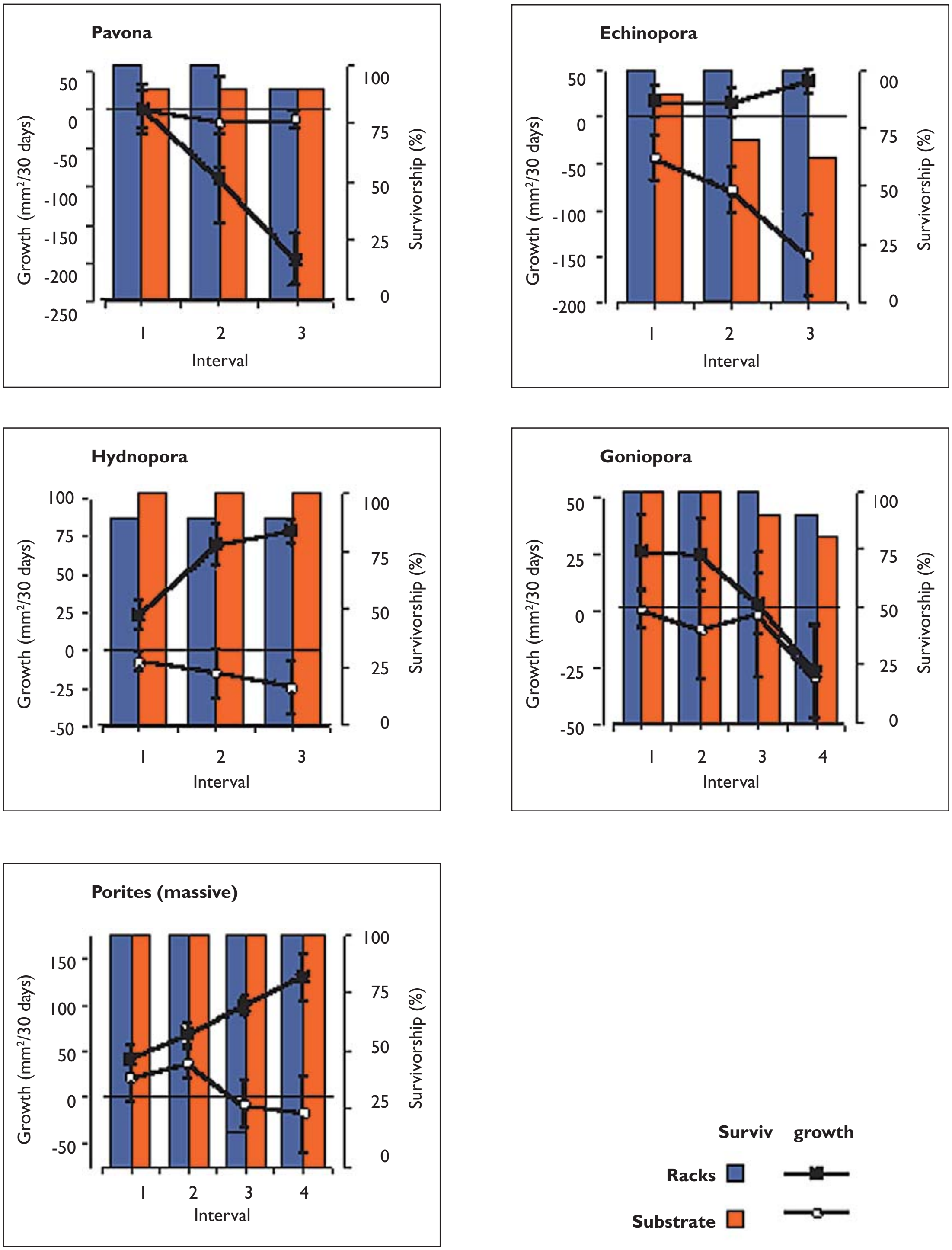
Figure 2. Growth and survivorship of coral fragments transplanted to elevated racks, and natural substrates. Branching corals (this page) in mm/30 days, sub-massive corals (facing page) in mm²/30 days.



clear. Future investigation will include the use of larger sized fragments to determine if the large amount of negative growth is a function of size.

Higher level objectives can be derived with the continued monitoring of transplants over the long term. The response of transplants to manipulations and transplanting to reefs with different environmental conditions enables research into species-specific differences in growth and survival. One possible outcome could be the development of a coral bio-assay in which the health and environmental conditions of varied reef systems could be assessed through the use of transplants of a species with a known and predictable response profile under well defined environmental conditions.

The two methods used for the study are relatively low cost with the following estimates per transplant (underwater materials only): approximately 6.50 Kshs (US$ 0.10) per coral fragment (on natural substrate), and


Pavona
Growth (mm²/30 days)
Survivorship (%)
Interval
Echinopora
Growth (mm²/30 days)
Survivorship (%)
Interval
Hydnopora
Growth (mm²/30 days)
Survivorship (%)
Interval
Goniopora
Growth (mm²/30 days)
Survivorship (%)
Interval
Porites (massive)
Growth (mm²/30 days)
Survivorship (%)
Interval
Surviv growth
Racks
Substrate

25-30 KShs (US$ 0.35-0.40) per coral fragment (on racks). However, further studies have to be performed before transplantation of corals by these methods can be considered a feasible rehabilitative technique for degraded reefs. Different transplant methodologies such as the use of cement will also be considered for evaluating the strengths and weaknesses of specific methods under different contexts, including economic.

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A low-tech method for reef rehabilitation by stabilisation of transplanted corals

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INTRODUCTION

Many coral reefs that have been subjected to severe coral mortality may erode into rubble before the reef framework has been stabilised by the growth of recruiting or surviving corals and other calcifying organisms. Since unconsolidated coral rubble provides a poor substrate for coral recruitment and subsequent growth, these damages can persist for a long time, even where there is ample supply of coral larvae. Rehabilitation of this type of habitat through coral transplantation has therefore been hampered since the substrate does not provide a base for attachment. Unattached corals that are moved by water movements may suffer severe damages through breakage and abrasion, and are also at risk of being buried in the shifting sediment. The aim of the present study was to further develop and evaluate a method to stabilise transplanted staghorn corals on unconsolidated substrate in a moderately exposed environment.

METHODS

The staghorn corals *Acropora formosa* and *A. vaughani* were transplanted in November 1998 to an area moderately exposed to waves at Tutia Reef, Tanzania. The seabed at the transplantation site is made up of a mix of coral rubble and sand at a depth of 3 m. Due to the unconsolidated and shifting substrate there were few corals growing at the study site, apart from some thickets of branching corals encroaching over the seabed. Coral branches were either dropped on the seabed without any attachment or attached to each other on 1.2 mm polythene strings, which were placed on the seabed using a method modified from Lindahl (1998). Groups of 10 coral branches were tied on pieces of string to form 1 m sections of "coral necklaces", which were stored under a tarpaulin onboard a boat. Each coral necklace was taken to the seabed at the transplantation site by a snorkler where the ends were tied to other sections to form an interconnected grid of 1 m x 1 m squares with each side made up of a coral necklace. Altogether 364 corals were placed, of which 260 were tied to strings to form a grid made up of 26 sections. The remaining 104 corals were dispersed in the surrounding area. Prior to the transplantation each coral was weighed and tagged, and the linear dimensions (length, width and height) were recorded. The average weight of the transplanted corals was 371 g ± 256 g (SD). One year after transplantation the corals were recovered, the measurements were repeated and the degree of tissue mortality was estimated.

RESULTS AND DISCUSSION

Twelve of the 104 loose corals were not recovered, and were presumed dead. The tied corals increased their live weight more and suffered less mortality than the loose corals (Figure 1). Among the tied corals, there was a sig-

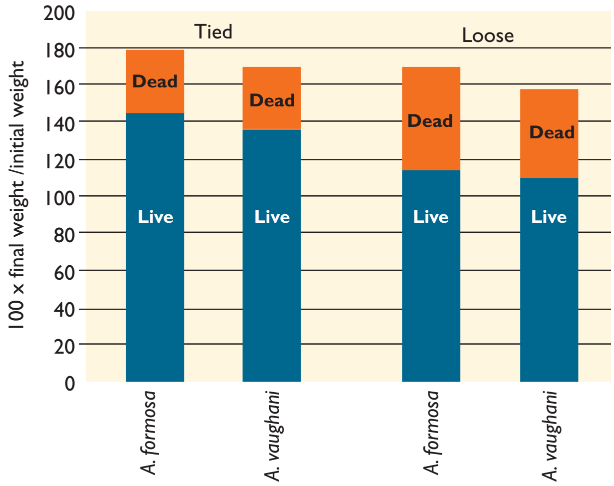

Figure 1 Final weight of coral colonies one year after transplantation expressed as a percentage of initial weight. Sample size = 130 (tied treatment); 52 (loose treatment).

nificant trend toward slower relative growth rate for larger colonies, as estimated by linear regression ($p < 0.05$), whereas the trend was the opposite, but not significant for the loosely placed corals.

The results show that the attachment method significantly increased the growth and survival of the corals. Most of the tissue mortality on the tied corals occurred on branches that were embedded in the sediment, whereas many of the loose colonies showed significant mortality also on exposed parts. Tumbling, abrasion and sequential burial of different parts of the colony had probably caused this mortality. All tied corals seemed to be fixed in their positions, remaining in place also after a slight tug by an inspecting diver. Most of the tied corals had fused with adjacent branches and were partially embedded in the seabed. The reduced relative rate of weight increase for heavier colonies in the tied treatment was probably related to the lower ratio of surface to volume with increasing size, and to some degree to self shading among branches in the same colony. This effect must have occurred for the loose corals as well, but was probably outweighed by the advantage of increased stability that comes with a larger size.

The described method is easy to apply, and requires very little material resources. The grid created by the coral necklaces will probably have a stabilising effect on the substrate, as noticed in a similar study by Lindahl (1998), and should thereby facilitate recruitment of other sessile biota as well as fish. The combined weight of the transplanted corals and the anchorage provided by the partially embedded branches was enough to stabilise the grid, and it may be resistant to water movements much stronger than those prevailing at the study site. The presence of naturally growing thickets on unconsolidated substrate at the study site shows that, once a certain density is achieved in a population, the stabilisation provided by the string will no longer be necessary. The applications of the described method should be further studied with regard to different exposure regimes, substrates and species. Unpublished results have showed that a third staghorn species, *Acropora nobilis,* exhibits responses very similar to those described here.

The most important limitation of this method, apart from wave protection and shallow depth, is the availability of suitable source populations. If corals are harvested in an uncontrolled manner, the damage to source populations may be greater than the beneficial effect on the receiving site. However, since the staghorn corals often grow in dense thickets, a selective, small scale harvesting of colonies in the interior of these thickets should result in a reduced intra-specific competition for space and facilitate encroachment over the harvested patches by neighbouring coral colonies.

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Impacts of bleaching on coral communities on artificial reef structures in Maldives

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ABSTRACT

A research programme to evaluate the feasibility of using artificial reef structures (ARS) to rehabilitate degraded reefs was conducted in Maldives between 1990-1994. Detailed monitoring and analysis of coral recruitment patterns on the ARS over a period of 3.5 years demonstrated that a diverse community of branching corals developed within three years with a similar composition to adjacent reef flats. A warm water anomaly of +3° C occurred in Maldives between late April and May 1998 resulting in extensive bleaching of corals and other zooxanthellate reef invertebrates. On the ARS bleaching was followed by full to partial mortality of certain corals within 4 - 6 weeks of the onset of bleaching. Branching species of the genera *Acropora* and *Pocillopora* were the most susceptible corals to bleaching. At the same time massive corals, such as *Porites, Favites, Pavona* and *Favia* spp. showed partial to full recovery on the ARS and on natural reefs

Quantitative surveys in March 1999 recorded 205 recently settled coral recruits on the three concrete SHED areas. Although branching corals were the dominant colonisers (67%), there was a significant increase in the abundance of massive corals (33%) when compared with the taxonomic patterns recorded between 1990-1994, in which massive recruits represented less than 2% of the population. On natural reefs rates of recruitment were 23.2 colonies per m^2. Settlement and subsequent growth of coral recruits with both branching and massive growth forms since the bleaching event indicates that a supply of viable coral larvae are available from reefs located upstream or from deeper areas of the same reef. Bioerosion and breakage of dead standing corals on the ARS was evident ten months after the bleaching event. As the reef framework breaks down the ultimate fate of the calcium carbonate fragments will have important consequences for the integrity of the reef framework, which provides natural protection for low lying islands against oceanic waves.

INTRODUCTION

In Maldives coral mining for the construction industry has resulted in widespread degradation of shallow reef flat areas (Brown & Dunne, 1988). The loss of vital coastal resources, the associated problems of coastal erosion and the slow rates of natural reef recovery have prompted interest in the potential of artificial reef structures to rehabilitate physically damaged reefs.

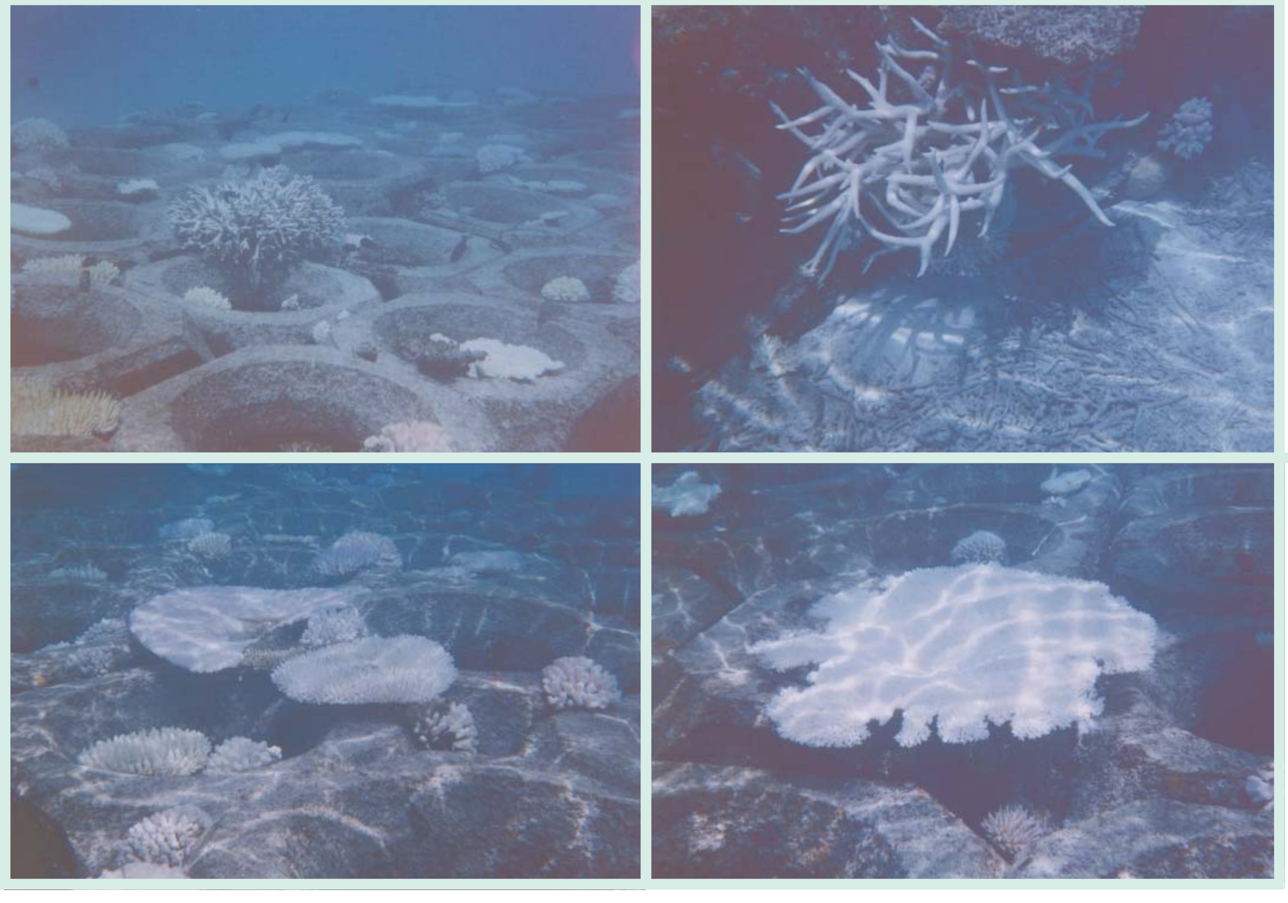
With funding from the UK Department for International Development (DFID - formerly ODA) an experimental artificial reef programme was initiated in 1990 to determine the feasibility of using a bio-engineering approach to kick-start natural reef recovery. The main objectives were to: i) restore reef fish and coral biodiversity, ii) test the hypothesis that lack of reef recovery is due to the lack of stable surfaces for settlement of coral planulae, iii) investigate the biological and physical conditions conducive to reef recovery and iv) determine which rehabilitation techniques are cost effective. Accordingly, 360 tonnes of concrete structures with varying levels of topographic complexity, stabilising effect and cost were deployed on a heavily mined study site close to the island upon which the capital Malé is situated.

Recruitment and subsequent growth and mortality of corals on the Artificial Reef Structures (ARS) were monitored over 4 years (Clark & Edwards, 1994; 1995; 1999). By late 1994, substantial coral recruitment had occurred on the larger reef structures, which were each supporting *circa*. 500 colonies. The coral community that developed on the reef structures was dominated by acroporids (53%) and pocilloporids (45%), with very few massive colonies (2%) present. Given the slow recruitment and growth of massive species, which are crucial for reef framework construction, it was predicted that a further 5-10 years are required to attain a community structure similar to pristine adjacent reefs.

Coral bleaching on artificial reef structures

In May 1998, scientists attending a training workshop supported by the South-east Asia Global Coral Reef Monitoring Network (GCRMN) visited the ARS at Galu Falhu and observed almost 100% bleaching of

Plate 1. In May 1998, virtually all corals on the ARS at Galu Falhu exhibited bleaching.



branching and massive corals (Plate 1). In June 1998, a brief inspection of the ARS revealed that the majority of massive corals had recovered from the bleaching event but almost all of the branching corals were dead and covered with filamentous algae (Plate 2). A follow up visit in February 1999 found evidence of new coral colonisers, particularly the opportunistic species *Pocillopora damicornis* had settled on both the SHED and Armorflex structures, indicating reef recovery processes were underway.

This report describes the results of field surveys conducted 11 months after the onset of bleaching to assess quantitatively the post-bleaching status of the coral communities developing on the ARS and evaluate initial stages of recovery.

Plate 2. By June 1998, almost all colonies of massive corals had recovered their zooxanthellae. However, mortality amongst colonies of branching corals was extensive.

METHODS AND SITE DESCRIPTION

The approach involved *in-situ* surveys of visual recruitment to document patterns of recruitment within different taxa on a variety of substrate types. Information was collected on the taxonomic patterns, density of colonies on different surface orientations and survivorship. Coral recruitment on the ARS was also compared to recovery processes on the adjacent reefs. Due to time constraints, attention focused on the larger SHED structures which had shown greater levels of coral recruitment during the initial monitoring programme. In the past, a lack of information describing the growth rates of juvenile corals has lead to confusion over the age at which corals become visible to the naked eye (usually around 5 mm). In this study, rates of growth measured from local juvenile corals (Clark, unpublished data) were used to define the size of coral recruits. The following are considered to be small juveniles that have settled within 12 months: i) branching colonies with a diameter of 7 cm or less and ii) massive colonies with a colony diameter not exceeding 3 cm.

Description of artificial reef structures

Four sets of artificial reef structures weighing a total of 360 tonnes were deployed on a 4 ha area of reef flat on a severely degraded faro called Galu Falhu, which is situated 2.4 km north-west of the capital Malé. The study area was 0.5-1.8 m below LAT (approx. tidal range 1m) and largely consisted of sand and loose coral rubble. The four sets of structures comprised: 1) One cubic metre SHED (Shephard Hill Energy Dissipator) hollow concrete blocks modified with infills to reduce internal void space, 2) Amorflex-220 flexible concrete mattresses, 3) Armorflex-220 onto which corals were transplanted and 4) Chain-link fencing anchored by concrete paving slabs.

Status of the coral community prior to bleaching

Two rapid assessments of the ARS conducted in June 1998 and August 1998 found that juvenile coral populations at the SHED sites had increased substantially since the last survey in 1994. Qualitative assessments suggested an increase in abundance of frame-building massive species, particularly on the internal surfaces of the SHED areas and the interstitial and void spaces of the Armorflex mats. These data indicate that good conditions for growth continued at the site prior to bleaching.

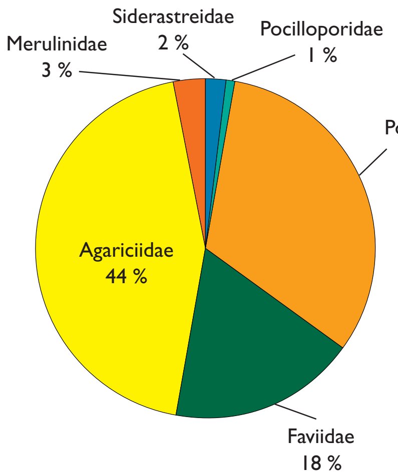


Figure 1. Taxonomic patterns exhibited by the coral community on the ARS which had recovered from the 1998 bleaching event. Note that colonies with massive growth forms constitute 99% of the coral community.

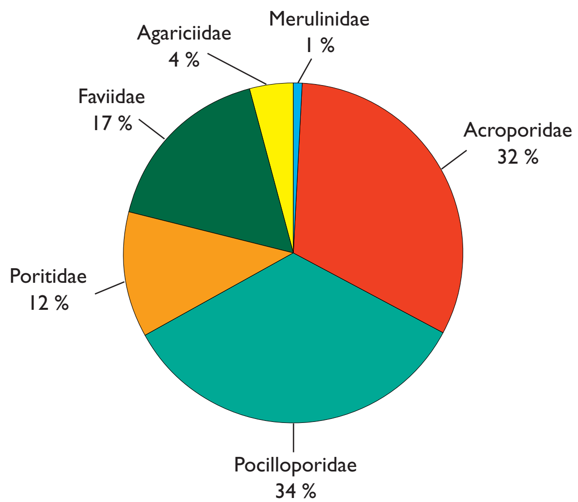


Figure 2. Taxonomic patterns exhibited by coral recruits recorded on the ARS 11months after the 1998 bleaching event. Note that colonies with branching growth form constitute 66% of the coral community.

RESULTS

Bleaching induced coral mortality

Ten months after the onset of bleaching it was difficult to distinguish between dead coral, which can be directly attributed to the bleaching event, and pre-bleaching mortality. Colonies of *Acropora* suffered early mortality within 6-8 weeks of the onset of bleaching and were rapidly colonised by filamentous algae (Plate 2). At the same time the majority of pocilloporid colonies were bleached but still alive. In 1994, there were 728 live colonies of *Acropora* and 497 colonies of *Pocillopora* on the three SHED areas. A thorough search of the three SHED areas in March 1999 revealed that no colonies of *Acropora* and only four colonies of *Pocillopora* survived indicating that branching acroporids and pocilloporids, which were the dominant colonisers of the SHED areas, suffered almost total mortality as a result of the bleaching event in April-May 1998.

Survivorship of the coral community on the SHED areas.

A total of 246 colonies of coral, classed as survivors, were found on the three SHED areas. Figure 1 shows the taxonomic composition of survivors with massive growth forms showing greater levels of resilience to bleaching. The greatest numbers of survivors belonged to the genera *Porites* (Poritidae) and *Pavona* (Agariciidae). The highest densities of survivors were found on internal surfaces (1.8 colonies per m^2) followed by outer vertical (0.8 colonies per m^2) and metal bar (0.7 colonies per m^2) surfaces. These patterns appear to be related to the settlement preferences of the massive corals *Porites* and *Pavona*.

Recolonisation of the artificial reef structures

A total of 205 new coral recruits were observed on the three SHED areas. Densities of coral recruits ranged from 0.6 colonies per m^2 to 2.8 colonies per m^2, with the highest density found on outer vertical surfaces. Lower densities of recruits were found on near-vertical surfaces, metal bars and PVC surfaces. No recruits were found on outer horizontal surfaces.

The taxonomic patterns of coral recruitment on the SHED areas is shown in figure 2. Overall, the ratio of branching to massive colonies was 67:33. This represents a greater abundance of massive species than the initial colonisation patterns recorded between 1990 and 1994, where massive colonies represented less than 2% of the entire population of corals. In total, nine genera of massive corals were represented on the three SHED areas from the following families: Poritidae, Faviidae and Agariciidae. The dominant massive corals on the three SHED areas were *Porites lutea* and *Pavona varians*.

There was considerable variation in the taxonomic patterns of recruits on different settlement surfaces. The highest percentage of recruits belonging to *Acropora* and *Pocillopora* were recorded on outer vertical surfaces, which concurred findings reported by Edwards *et al.* (1994). Recruits with massive growth forms belonging to Poritiidae and Faviidae were also found on both internal and vertical surfaces. In contrast, the genus *Pavona* was predominantly found on internal surfaces (91%).

Bioerosion and breakage

Although the dead stands of branching coral colonies largely remain *in-situ* they have been subject to breakage and erosion (Plate 3). Most of the dead coral surfaces have been colonised subsequently by opportunistic species including filamentous and macro-algae (Plate 4). Repeated measurements of dead colonies on the SHED and Armorflex areas have provided an early indication of the extent of erosion of the reef framework. Many of the large *Acropora* colonies with a tabular growth form have shown a 40% - 60% decrease in mean colony diameter in 6 months. However, it is not possible to determine the causative agents (i.e. physical breakage, grazing or bioerosion) and it is likely that a combination of factors is involved. The magnitude and direction of change in the reef framework and topographical complexity of shallow reef flats may be crucial in determining the time scales for reef recovery in Maldives.

Natural recovery processes

Coral recruitment patterns on the adjacent mined reef were very patchy. The reef consists of raised coral boul-

Plate 3. Although many dead branching corals remained intact, their skeletons were being degraded by biological and physical erosion and breakage.

Plate 4. The exposed skeletons of dead corals are colonised rapidly by filamentous algae and other opportunistic species.

ders interspersed amongst unconsolidated sediment and rubble. Coral recruits were found only on hard substrates such as raised boulders. Densities of recruits reaching 23.2 colonies per m^2 were recorded on patches of raised hard substrate, but were negligible over much of the degraded reef flat. Availability of suitable substrate for larval settlement appeared to be the most critical factor limiting recruitment in the adjacent reefs.

DISCUSSION

Extensive searching revealed a total of 246 adult colonies of coral on the three SHED areas that survived the bleaching event. Distinct taxonomic patterns were evident amongst the surviving coral population, which was dominated by massive corals. No colonies of *Acropora* survived on any of the ARS and less than 2% of the survivors on the three SHED areas belonged to the genus *Pocillopora*. In contrast, colonies with a massive growth form, with slower growth rates and low recruitment, showed greater resilience to the temperature anomaly and many colonies regained their pigmentation within four to six weeks. These findings concur with those of Fisk & Done (1985) and Brown & Suharsono (1990) who found that *Acropora* and *Pocillopora* were most affected by bleaching in the Indo-Pacific.

Quantitative surveys found 205 new coral recruits on the three SHED areas. Although branching corals were dominant among recruits, the ratio of branching to massive colonies (67:33) was significantly lower than that previously recorded during the initial colonisation period in 1990/1991. Between 1990 and 1994 the major reef frame-building species were rare or absent from the ARS indicating that reef recovery rates would be very slow. A comparison between the early colonising species in 1994 and those following the 1998 bleaching event indicates that although the same suite of massive coral colonies dominate the ARS there were greater densities present in 1999.

Results of this survey show that extensive mortality has occurred to the coral community of the ARS since the bleaching event. Despite the severity of the mortality, this evaluation has demonstrated that the capacity of the reef system to recolonise degraded areas, through an influx of coral planulae from undisturbed sites, is high as long as suitable substrate is available. *Acropora* recruits have colonised dead coral surfaces indicating that recovery of these reefs is not limited by coral recruitment and also that those colonies that survived bleaching were able to complete gametogenesis and spawn within one year of bleaching. Although there was a source of larvae in the local area, it was likely that larval supply was lower than in previous years. Therefore, the ARS provided a suitable tool to evaluate the potential availability of larvae within shallow reef flat areas in Maldives.

Implications for reef recovery processes

Corals recovering from sub-lethal bleaching may experience reduced skeletal growth, reproductive failure and have a lower capacity to resist disease, shed sediment and heal injuries (Jokiel & Coles, 1990; Glynn, 1993). Coral reefs are dynamic systems exhibiting high levels of natural variability. Thus, time-scales for reef recovery processes are difficult to predict and largely depend on site-specific conditions. Studies of coral reefs following major bleaching events have shown contrasting patterns of recovery. For example, in the Thousand Islands of Indonesia, Brown & Suharsono (1990) and Brown (1996) found that, although the community structure was markedly different, coral communities attained their pre-bleaching cover within five years. The differences in community structure were attributed to storm activity and an increase in anthropogenic influences during the recovery period.

Ecological consequences of the bleaching event

Coral diversity and community structure on both the ARS and natural reefs are different from the pre-bleaching state. In the short term (< 5 years), the ARSs, which were formerly populated by a preponderance of branching species, will be dominated by non-living substrate that supports a low percentage cover of living corals, of which most will be massive species. Recruitment processes, particularly the subsequent growth and survival of juvenile corals, will depend on the frequency and intensity of recurrent bleaching events.

ACKNOWLEDGEMENTS

This research was funded by the U.K. Department for International Development. I would like to thank Mr Hassan Maniku Maizan, Former Director of the Marine Research Centre and all of his staff who assisted with this work, particularly Mr Hussein Zahir. I am also grateful to Dr Alasdair Edwards and Dr John Tarbit for comments on an earlier draft of this report.

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Management of bleached and severely damaged coral reefs

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BACKGROUND

Coral reefs are now recognised as the most diverse ecosystems in the sea and of immense economic importance. In 1998/1999 a major coral bleaching event (linked to El Niño and almost certainly to climate change) caused extensive reef damage throughout the Indian Ocean, with coral mortality exceeding 90% in some areas. This can be likened to large areas of tropical rainforest being razed to the ground. There is an urgent need to take immediate action. We need to stimulate a response comparable to the international reaction to the forest fires in Indonesia, which generated intense international concern and rapid response planning.

Protection of the few remaining healthy reefs, as well as those that are largely damaged, is now critical if the reef ecosystems as a whole are to have any chance of recovery. The future livelihoods of human populations dependent on reefs will depend on reef recovery. Countries of the Indian Ocean are now at serious risk of losing this valuable ecosystem. The economy of Maldives, for example, has traditionally been based on fisheries and tourism. Both of these activities can be linked directly to the reefs, which have been severely affected by bleaching. This, combined with ongoing additional human impacts of over-fishing, pollution and coastal development, will potentially have major economic and ecological impacts.

Crucial management questions are now being raised in many areas. Managers are already asking how they should deal with this situation: whether the reefs will recover and - if they will - what actions they should take to aid and accelerate regeneration; how can they convince policy makers and government agencies of the value of maintaining marine parks and conservation efforts in the face of dead or dying reefs; should they be investing in what may be costly and risky reef rehabilitation projects; and what economic impact will degradation have and how can such impacts be mitigated?

ACTIONS

Manual on management of coral reefs in the western Indian Ocean

It is recognised that scientific information is not yet sufficient for precise recommendations to be made, but it is clear that the knowledge that is available must be transferred to those in a position to protect the remaining resources and stimulate recovery. A publication on management of coral reefs in the western Indian Ocean is proposed as a rapid response measure, which will provide guidance on precautionary measures to be taken, translate current scientific opinions on consequences and predicted outcomes of bleaching, and make suggestions on positive actions that might aid reef recovery.

The booklet will be short, about 30 pages, and target reef managers. It will focus on the management of degrading/degraded coral reefs, the possibility for recovery/restoration and what actions should be taken to aid regeneration and prevent further destruction. The booklet will contain sections on the value of maintaining parks, (protecting what's left, conservation efforts), the role of MPAs in reef recovery, the roles of fishing, tourism and coastal development in reef destruction as well as information on restoration techniques and monitoring. Management guidelines, information on economic impacts of reef destruction and mitigation of these as well as suggestions on actions will be included in each section.

The booklet will be translated into all major languages of the region and distributed free to governments, MPAs and NGOs as well as published on the Internet. A summary leaflet of 4-6 pages will also be produced for wider distribution.

This booklet is needed immediately. Following publication and dissemination, it may be possible to expand the concept into a more measured response. This could involve regular updating of managers on advances in understanding of recovery rates, economic impacts, etc. and linking existing and planned monitoring programmes with management to ensure appropriate feedback.

Environmental education

Environmental education efforts will be intensified. Education should be directed primarily at the general public and school children, in collaboration with organisations, such as NGOs, already established in the region. The aim should be to increase awareness of the functions of healthy coral reefs and their role in the ecosystem. Conservation and the need for protected areas, as well as the value of healthy coral reefs as a source for food and income for coastal populations, should be emphasized.

This will be achieved through the development of hard-copy materials (e.g. posters, booklets, photographic material and videos) that can be used by experienced educators and awareness groups. Existing material will be used as much as possible although new material will have to be produced to suit local conditions, target groups and issues. Local level collaborations to develop and ensure use of the materials and using local knowledge, language, artwork and concepts will help achieve impact at the local level. The regional context will be addressed through forming linkages across countries, to disseminate and assist development of parallel materials.

APPENDICES

Coral Reef Degradation in the Indian Ocean (CORDIO) -Progress to Date and Directions for 2000
February 10th - 12th, 2000, Lamu, Kenya

AGENDA

Thursday 10th Jan:

8.00 am
Arrival and short field trip

2.00 pm
Opening and Presentation of Regional Reports
East Africa – David Obura
South Asia – Dan Wilhelmsson
Island States – Lionel Bigot

2.30 pm
Presentation of Country Reports
Kenya (David Obura)
Tanzania (Christopher Muhando)
Mozambique (Helena Motta)

3.15 pm
Coffee Break

3.30 pm
Country Reports (cont.)
India (Dan Wilhelmsson)
Sri Lanka (Dan Wilhelmsson)
Maldives (Hussein Zahir)
Réunion (Lionel Bigot)
Seychelles (John Turner)
Mauritius (John Turner)

5.00 pm
Presentation of Other Regional Activities
WWF (Irene Kamu)
IUCN (Sue Wells)
GCRMN (Clive Wilkinson)
UNEP (Agnetha Nilsson)

7.00 pm
Cocktails & Snacks

Friday 11th Jan:

8.30 am
Presentation of Thematic Reports
Socio-economic effects on fisheries and tourism (Herman Cesar, Lida Pet-Soede, Susie Westmacott & Nariman Jiddawi)
Measuring change: The importance of long term trends (Charles Sheppard)
Southern Seychelles islands (Kristian Teleki)

10.30 am
Coffee Break

10.45 am
Temperature and water flow in a Kenyan fringing reef lagoon (David Kirugara)
Wildlife Conservation Society Coral Reef Programme activities in East Africa (Tim McClanahan)
Growth of Finnish co-operation in coastal management in Eastern Africa (Erkki Siirila)
Low-tech rehabilitation of coral reefs (Ulf Lindahl)
Recruitment of corals in Maldives (Susan Clark)

12.30 pm
Lunch

2.00 pm
Discussion Groups (All participants)
General themes:
I Socio-economics
II Databases, methodologies; monitoring.
III Alternative livelihoods, fisheries
IV Reef recovery processes, rehabilitation.

7.00 pm
Dinner

Saturday 12th Jan:
8.30 am
Discussion Groups (cont.)

12.30 pm
Lunch

2.00 pm
Presentation of outputs from discussion groups
Discussion of targets for 2000/2001

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