CORDIO
Högskolan Kalmar University

Coral Reef Degradation in the Indian Ocean

Status Report 2002

Editors: Olof Lindén, David Souter,
Dan Wilhelmsson & David Obura

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The coral reefs of the Indian Ocean are extremely diverse ecosystems and play a key role in the productivity of the coastal areas. In addition, these reefs are essential for the food security of a large proportion of the human population, for the protection of coastlines and as a valuable tourist attraction. Unfortunately, a number of threats that have caused serious declines in the distribution and condition of these reefs during recent decades. In particular, the El Niño event of 1997/98 produced extremely high water temperatures that caused bleaching and mortality of corals on a global scale. CORDIO is a collaborative program involving research groups in 11 countries and was created in 1999 to help mitigate the widespread degradation of the coral reefs throughout the central and western Indian Ocean. This report presents a summary of the results obtained during 2001 and 2002.

CORDIO is an operational program under the International Coral Reef Initiative (ICRI).

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Contents

Foreword

In addition to being magnificent ecosystems full of life and colour, coral reefs are critical to the food security of coastal people in large parts of the world. It has been estimated that around 50% of the fish catch from several countries in the tropics are directly related to coral reefs. The productivity of these ecosystems is phenomenal. One square kilometre of coral reef can provide food for several hundred people per year. In addition the reefs provide shelter for larval stages of a number of different fish and other organisms. Furthermore we know that along coasts where coral reefs have been destroyed, floods and storms easily wash the coastline away. It is therefore with great concern that we learn about the destruction of coral reefs around the planet. It is obvious that a number of human activities are to blame, including fishing using destructive fishing methods including dynamite, and pollution and sedimentation from land. During the last decades it has becom obvious that climate change is a factor playing an increasing role in the stress on coral reefs. If the trend towards increasing temperatures on the planet continues, the background sea surface temperatures is likely to raise to levels where bleaching – a general sign of stress affecting corals – will become regular phenomena within a few decades.

A climate related phenomenon; the El Niño of 1998 caused massive coral bleaching, which killed a large portion of the corals on the planet. The Indian Ocean was particularly badly affected by this El Niño and most of the corals in East Africa and in central and northern Indian Ocean bleached and died during this event. Since the early 1990s Sida, IUCN and a few other organizations such as UNEP has supported capacity building in marine science and management in the East African region. Institutions that took part in this capacity building reported much of the early data showing the extent of the coral bleaching in 1998. It is interesting to note that the flow of data and information regarding the impacts of the El Niño of 1998 came from the East African region to the rest of the world and not the other way around. At the same time as the reported information is worrying indeed, we are convinced that is an encouraging sign of the growing capacity in the field of environmental science at local institutions in the region.

As a result of the region-wide bleaching of 1998, Sida decided to support the CORDIO program, which was launched early in 1999. The purpose of CORDIO, which also was supported by several other Swedish and international organizations, is to obtain data regarding the ecological and socio-economic consequences of the mortality of corals, particularly focusing on vulnerable groups of the coastal human population. This report presents the results of the work conducted under the CORDIO program during the last 18 months. The results make an important contribution to our knowledge of some of the very worrying problems caused by climate change.

The scientific assessments carried out under the CORDIO program is done by local scientists and experts in 11 countries around the Indian Ocean. CORDIO is therefore an example of a locally driven regional initiative to provide valuable data for not only the countries of the region but for the world at large. It is important that such networks are maintained as it is only through long-term data series that we are able to assess the state of the environment and the potential threat to the society from environmental change.

We are pleased to endorse this report and hope that it will stimulate national and international efforts to better manage coral reefs. Action is needed at all levels, from assisting local communities with co-management agreements, via national programs for capacity building and strengthening of institutions to improve management and enforcement, to truly global initiatives like managing the trade in coral reefs products or reducing the release of greenhouse gases that are linked to coral bleaching. We need to conserve coral reefs because of their magnificent biodiversity, unparalleled beauty, enormous tourism potential and particularly because well managed reefs can provide food and shoreline protection for hundreds of millions of people around the world.

Bo Göransson
Director General, Sida

Achim Steiner
Director General, IUCN

Executive Summary

Since the 1998 coral bleaching and mortality, which affected reefs throughout the tropics, a number of projects dealing with research, monitoring and management of coral reefs have been conducted. A number of these projects have been carried out under the CORDIO Program in the Indian Ocean where coral reefs have been severely impacted by bleaching related mortality and other anthropogenic influences. This, CORDIO's third status report, describes the condition of coral reefs throughout the central and western Indian Ocean and presents the results of several targeted research projects conducted within the last 18 months.

The Recovery of Coral Communities Including Fish

The covery of coral reefs is highly variable between sites in East Africa and South Asia. Generally, extensive mortality affected the islands of the northern and central Indian Ocean (Maldives, Seychelles, Aldabra, Chagos), the East African coast (Mozambique, Tanzania and Kenya), Sri Lanka and parts of India. Islands of the southern Indian Ocean (Reunion, Comoros, Mauritius, Madagascar) were not seriously affected. In most of the areas that suffered high coral mortality (i.e. more than 50%), recovery is slow, patchy or non-existent. In some areas, significant recovery appears to be underway. However, such areas are usually located far away from human influence or are well protected in marine reserves. It is obvious that in many places closer to human habitation other stresses from anthropogenic activities, such as destructive fishing, coral mining and sedimentation from land, continue to degrade the reefs, slowing down or preventing recovery from the 1998 bleaching event.

TANZANIA

Little recovery was observed on the reefs of Misali Island where 80% mortality was noted in 1998. Similar observations were made on Kwale and Mafia reefs where the cover of live coral remained low or continued to decline. Chumbe Island and the reefs off Zanzibar town were relatively unaffected by the 1998 bleaching event. However, destructive fishing and outbreaks of Crown of Thorns starfish are now affecting several of these reefs. Fish populations showed significant changes and several groups (especially invertivores, piscivores and plaktivores) decreased while other groups (particularly herbivores) increased. During 2001–02, harmful algal blooms and an unknown fungal coral disease have impacted reefs in northern Tanzania. These stresses may be climate-change related. Over-fishing, destructive fishing, pollution from human settlement and development, mining and shipping industry activities continue to cause degradation to the reefs of the country.

KENYA

The coral reefs were severely impacted in 1998, with bleaching and mortality levels of 80% and greater. Recovery of affected reefs to 2002 has been moderate and patchy. Reefs with high coral diversity and cover that were strongly impacted in 1998 have recovered to less than 50% of pre-bleaching coral cover. Some high diversity reefs that were protected from bleaching impacts have remained healthy. Degraded reefs outside of Marine Protected Areas that were strongly impacted by the El Niño have generally recovered to 50–100% of pre-bleaching coral cover. In 2001–02, multiple threats have impacted Kenyan reefs that may be climate-change related, including harmful algal blooms and an unknown fungal coral disease. Anthropogenic activities including over-fishing, destructive fishing, pollution, human settlement and development, mining and shipping industry activities continue to threaten East African reefs.

MOZAMBIQUE

In the two years after 1998, reefs that had been severely affected by the bleaching showed slow or no recovery. Reefs in northern remote areas and in protected areas showed increasing coral cover. In February 2000, severe floods affected the country. These floods brought large quantities of sediment to the reefs and caused significant reduction of the remaining coral cover: from 20 and 5% to 10 and 0.5% respectively for hard and soft corals in the Xai-Xai lagoon.

MADAGASCAR

Information is limited to a few areas in the northwest and southwest. In the northwest the corals were relatively unaffected by the bleaching and the ecosystem was in good condition. In the southwest the reefs were little impacted and recovery is underway.

SEYCHELLES

The reefs of the inner granitic islands suffered extensive and system-wide degradation in 1998 and hard coral abundance and diversity are now well below pre-1998 levels. However, small but significant increases in coral cover and diversity have been noted over the past 15 months. Recovery may be underway in the near- to mid-term. The rate at which local coral communities will recover will be largely dependent on the environmental conditions encountered by newly settled coral recruits.

REMOTE REEFS OF THE CENTRAL INDIAN OCEAN

The cover of live coral at Aldabra has changed little between 1999 and 2001 indicating that recovery is a slow process even in remote areas beyond the reach of anthropogenic influences. Nevertheless, recovery of the reef at Aldabra is unlikely to be limited by the influx of new corals as the average number of recruits per m^2 ranged between 4 and 7 with the greatest number recorded at approximately 10 m depth. The impacts of bleaching related coral mortality on Aldabra’s fish communities was difficult to discern. The abundance of fish at Aldabra varied both spatially and temporally but the diversity of fish species has remained high with 287 species from 35 families being recorded in 1998 prior to the bleaching event, 211 species from 35 families in 1999 and 205 species from 40 families in 2001. Also, in Chagos, where most corals died in 1998, the recovery is very slow and is being hampered by the breakdown of the dead coral thickets due to bio-erosion forming large areas of unstable substrates.

MALDIVES

The recruitment of corals in Maldives is well underway, but is patchy and does not resemble the pre-bleaching community structure suggesting that these reefs will experience a change in community structure.

SRI LANKA

The recovery of shallow reefs in Sri Lanka is variable between sites but has, in general, been slow and in some areas non-existent. Algae, tunicates and corallimorpharians largely dominate dead reefs. Fish populations have changed and several of the typical coral fish species have become rare or have disappeared. Destructive human activities such as coral mining, extensive fishing and collection of fish for the aquarium trade continue to degrade the reefs preventing recovery.

REEFS OF THE GULF OF MANNAR AND NICOBAR AND ANDAMAN ISLANDS

Recent reports from the Gulf of Mannar show variable results, which might indicate both significant recovery and that the impact in 1998 was patchy affecting certain areas while others did not suffer to the same extent. Recent surveys from the Nicobar and Andaman Islands show no signs of a mass mortality in 1998.

Socio-Economic Effects

Fish populations were affected by the bleaching both in terms of species composition and the total quantities of fish . However, more research will be needed to establish the detailed relationships and quantify the effects. Results from certain areas seriously affected by bleaching and subsequent coral mortality indicated an initial increase in catch figures caused by increased populations of herbivorous species that were able to utilize the overgrowth of fast-growing algae that colonised the reefs after the corals had died.

It is difficult to directly link decreasing fish catches to the bleaching event. The general trend in most of the shallow reef areas in the region is one of declining stocks due to over-fishing and habitat destruction. However, the data from different studies carried out in the region are not conclusive. It should be pointed out that observations of impacts on fish populations through catch data are often unreliable, as catch statistics do not usually reflect the situation in the environment. If traditional sites yield fewer fish, fishermen usually move to other areas and/or target other species or use different gear. In addition, the catch per unit effort may appear to be stable or even increasing giving the impression that the fish stocks are healthy and under-utilized. However, increased catches are often the result of the use of more efficient fishing gear and boats and not at all a true reflection of the condition of fish stocks. The general trend in most areas in the region is that the fishing pressure is increasing.

In addition, the official statistics often have little to do with the real catches, as much of what is caught is not recorded. What is obvious from several observations throughout the region is that there has been a shift in fish species composition in areas severely affected by the bleaching. Species such as butterfly fish and surgeon fish are decreasing in numbers in areas where the coral has died while some other species such as rabbit fish as well as small pelagic species such as clupeids appears to increase, at least in the short-term. Results from several areas indicate that, as long as the reef structure remains intact, the reefs are still able to provide protection and function as nursery areas for juvenile fish and other invertebrates. However, once the structure of the reefs collapses, these functions are likely to be lost. Observations from Chagos indicate that extensive areas of reef that died in 1998 have been eroded to rubble and are now forming a substrate too unstable to allow settlement of coral larvae.

The impacts of coral bleaching on tourism have also been variable across the western Indian Ocean region. In saturated tourist destinations such as the Maldives, only slight impacts have been observed because the industry was able to compensate for losses incurred in dive and other reef-related tourism sectors with gains in other areas such as the honeymoon market. In less popular destinations the impacts were more severe. According to surveys carried out in Sri Lanka in 2000–2001, the snorkelling and glass bottom boat industry have declined as a result of the mass bleaching of corals in 1998.

Detailed investigations of the impacts of bleaching on tourism in Seychelles, Kenya and Zanzibar indicated that the possibility to enjoy healthy reefs was an important component of the holidays of tourists at these destinations. On average, tourists were ‘willing to pay’ US$ 98.7 extra per holiday in the Seychelles, US$ 87.7 extra in Zanzibar and US$ 59.0 extra in Kenya in order to experience high quality reefs indicating that healthy reefs are important assets for tourism in the coastal regions of the Indian Ocean and that coral reef degradation could potentially cause a significant decline in tourism in these areas. Moreover, continued degradation of coral reefs will have a significant impact on the economies of those countries in the western Indian Ocean. Based on tourists willingness to pay, the number of tourists visiting Seychelles, Mombasa and Zanzibar and their interest in the marine environment, the economic losses were estimated to be US$ 71.5 million for the Seychelles, US$ 47.2 million for Mombasa and US$ 39.9 million for Zanzibar in net present values over a 20 year period with a 10% discount rate.

Public Awareness and Alternative Livelihoods

In addition to ecological and socio-economic research in the Indian Ocean, CORDIO, in collaboration with other organizations, is involved in projects to inform the public, staff of governmental agencies and politicians of the need for better management of costal marine ecosystems including coral reefs. For example, during 2001, CORDIO supported an awareness project titled 'A tomorrow for our reefs' implemented by IUCN Sri Lanka. In addition, on the Tuticorin Coast of India, and with WWF as a lead partner in Kiunga, Kenya, community based reef restoration has been initiated. Coral fragments are cultivated and transplanted to the reefs by the local community members thus restoring parts of the reefs and creating an understanding of the value and importance of the corals among the local communities in the area.

Raising awareness and building community capacity to participate in management of reefs and to improve local livelihoods is one goal of a CORDIO project in Diani, Kenya. Here, monitoring of reef fish catch, underwater fish populations and socioeconomic factors is being undertaken by fishermen and community members in partnership with scientists. While the data contributes to knowledge on reef health and status of the fishery, the process of monitoring has a strong educational and empowerment component, encouraging participating community groups to take a lead in caring for the coral reefs they depend on. The project is serving as a model for other community-based projects in East Africa working with coral reefs.

The results of a number of studies show the necessity to reduce the pressure from both legal and illegal human activities on the reefs, through better management and law enforcement. A high and growing population pressure characterizes many coastal areas. In order relieve the pressures on the ecosystem, there is a great need for alternative income sources for the people living in areas where the ecosystem is degrading. In Sri Lanka, a project has been started to assess the experiences of previous efforts and initiatives as well as experiences from other regions to generate alternative income sources. The outcome of this project will result in recommendations to relevant authorities in Sri Lanka, and for the basis for small-scale pilot projects under the CORDIO program starting 2003.

The Way Forward

The scientific assessments carried out under the CORDIO program are done by local scientists and experts in 11 countries around the Indian Ocean. Hence, CORDIO is a rare example of a locally driven, regional initiative that provides valuable data not only for the countries of the region but also for the world at large. It is important that such networks are maintained, because the collection of long-term data series is the only process that enables changes in the state of the environment to be determined and the potential threats to coastal populations and societies to be mitigated. The CORDIO program will continue to support and coordinate ecological and socio-economic monitoring of reefs in the Indian Ocean region with the aim of studying the processes of coral reef recovery from the 1998 coral bleaching, and reporting the state of the coastal environments. The long-term information obtained from continued monitoring will be used to develop projects investigating new and improved management options in several pilot areas in the region. This work will involve the regional nodes of CORDIO in Mombasa, Colombo and St. Denis. In addition, with a closer collaboration with the World Conservation Union (IUCN) agreements will be made between CORDIO regional offices and both IUCN national and regional marine programs to develop this type of activities.

Part I

East Africa

Status of Coral Reefs in East Africa

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ABSTRACT

East African coral reefs were severely impacted by the El Niño Southern Oscillation of 1997–98, with bleaching and mortality levels varying from <1% in South Africa to 80% and greater on reefs in northern Tanzania and Kenya. Recovery of affected reefs to 2002 has been moderate and patchy. Reefs with high coral diversity and cover that were strongly impacted by the El Niño have recovered to less than 50% of pre-bleaching coral cover. Some high diversity reefs that were protected from bleaching impacts have remained healthy, with high coral cover and diversity. Degraded reefs outside of MPAs that were strongly impacted by the El Niño have by and large recovered to 50–100% of pre-bleaching coral cover. Recruitment of corals to reefs has also been moderate, with highest levels recorded in protected reefs with high coral cover and diversity. In 2001–02, multiple threats have impacted East African reefs that may be climate-change related, including floods in Mozambique, Harmful Algal Blooms in Tanzania and Kenya, and an unknown fungal coral disease in Kenya and northern Tanzania. Anthropogenic threats to East African reefs continue since the global reports of 1998 and 2002, including over-fishing, destructive fishing, pollution, human settlement and development, mining and shipping industry activities. Socio-economic studies of coral reefs are becoming increasingly common in East Africa, including the development of socioeconomic monitoring under GCRMN. Importantly, while socioeconomic losses from coral mortality from the 1998 bleaching event have been predicted, particularly in fisheries and tourism, these have not yet been realized.

INTRODUCTION

The coast of Eastern Africa covers 40° of Latitude with the warmer temperatures in the central region and cool upwellings off the coast of Somalia (10°N) and cooler, temperate waters off South Africa (30°S). The major equatorial influence is the South Equatorial Current that hits the coastline near the Mozambique-Tanzania border to flow northward to Somalia and south to Mozambique. To the north and south, a gradient to cooler waters caps the distribution of coral reefs, due to the Somali current upwellings in the north, and the Agulhas current in the south. Starting with the coral bleaching event as a result of the El Nino in 1998 (Lindén & Sporrong, 1999; Souter *et al.*, 2000; Wilkinson *et al.*, 1999) to new potentially climate-related threats in 2001 and 2002, this biogeographic gradient on the East African coast appears to mitigate impacts to coral reefs. On top of this primary influence, the long term effects of local threats such as overfishing, sedimentation and pollution are still strong, and in many cases influence recovery from climate-related threats. This paper is drawn from the same sources as the East Africa GCRMN report for 2002 (Obura, 2002a).

MORTALITY AND RECOVER AFTER THE 1998 EL NINO

Coral bleaching and mortality during the El Niño event of 1998 was most severe in Kenya, northern Tanzania and parts of northern Mozambique, and diminished to virtually nothing in the south (Table 1). The bleaching started in the south in February–March 1998, and finished in May in the north, following the path of the solar zenith during the southern Summer. The most severely damaged reefs suffered levels of coral mortality between 50–90%. Coral cover on lagoon patch reefs and fore-reef slopes in southern Kenya dropped from an average of 30% to 5–11% within and outside MPAs (Obura *et al.*, 2000; Obura, 2001a; McClanahan *et al.*, 2002). There was similar bleaching in northern Kenya in shallow areas, but below 10m depth coral mortality was often less than 50%. There was high bleaching (60–90%) at Tutia reef in Mafia Island Marine Park, Tanzania and Misali reef on the west coast of Pemba, whereas there was 10% or less bleaching on Unguja Island, Zanzibar (Mohammed *et al.*, 2000, this volume). The most extensive bleaching in Mozambique was on exposed reefs in the north, with up to 99% mortality on some patch reefs (Motta *et al.*, 2000, this volume). Reefs in sheltered bays, where there are naturally higher levels of nutrients and turbidity, and more variance in water temperatures, were least affected. Bleaching of hard corals in 1998 was absent in South Africa because most reef corals grow at 12m or deeper (Schleyer & Celliers, 2000; Schleyer & Celliers, this volume).

Table 1. Coral cover during 1997/98 (prebleaching), 1999 and 2001/02 at monitoring sites in East Africa

	1997/98	1999	2001/02
Kenya			
Northern Kenya (>10 m)		5.7	1.2
(< 3 m)	13.2	5.1	9.9
Southern Kenya (Protected)	39.6	11.4	19.5
(Unprotected)	20.6	11.4	14.4
overall	26.3	8.1	11.25
Tanzania			
Tanga	53.0	33.3	
Pemba	53.7	12.3	16.3
Unguja	45.8	32.0	37.9
Kunduchi	43.0	35.0	
Mafia	73.3	19.4	24
Songosongo	35.0	37.5	
Mnazi Bay	60.0	20.0	
overall	52.0	27.1	26.1
Mozambique			
Quirimbas		48.4	35
Mozambique I.		32.5	30
Bazaruto		69.5	65
Inhambane		13.8	7
Inhaca		50.0	40
overall		42.8	35.4

Source: Obura 2002b. Data sourced from all citations in the reference section

Coral mortality during the 1998 El Niño event was more intense on the fast growing genera *Acropora*, *Pocillopora*, *Stylophora* and *Seriatopora*, as well as *Galaxea* and *Echinopora*, with up to 100% bleaching and mortality (Obura, 2001b; McClanahan *et al.*, 2001). In some other genera (e.g. *Fungia*, *Coscinaraea*, anemones) there was high bleaching, but high rates of immediate recovery. Many other coral groups (e.g. *Montipora*, *Astreopora*, faviids, agariciids, poritids, siderastreids, and most octocorals and zoanthids) bleached at variable levels, but the full impacts are difficult to determine in these relatively rare corals. Thus, there will be a shift in the structure of the coral community in the short to medium term away from species that grow and reproduce most rapidly to slower growing, massive species. Recovery will depend on available parent stock for new recruitment and no short-term repeats of these damaging El Niño events. Higher coral recovery rates in MPAs suggest that a healthy parent stock improves local coral recruitment and regrowth potential, though these factors are yet to be tested in East Africa.

Recovery since 2000 has been patchy in all countries. For example in northern Kenya, shallow reefs are generally recovering from the bleaching while deeper reefs have suffered stasis or further declines in coral cover

(Obura, this volume, Table 1). This variability in recovery also occurs between adjacent reefs, with some showing good recovery while their neighbours do not (Obura & Mangubhai, in prep.). In general, reefs within MPAs have shown higher recovery rates of coral cover, especially on Chumbe Island off Zanzibar (Mohammed *et al.*, this volume). This has also been true in Kenya, though the higher prebleaching coral cover on MPA reefs means the time to full recovery may be greater (McClanahan *et al.*, 2002). Recruitment has also been variable, with peak values of >20 per m^2 being recorded for Mafia (Tanzania, Mohammed, this volume) and Kiunga (Kenya, Obura, this volume), though average levels are much lower in the range of 1–3 per m^2. Recruitment is overwhelmingly dominated by *Pocillopora*.

In Mozambique, reefs affected by bleaching have shown little recovery to 2002, though inaccessible reefs (in the north) and those in MPAs showed greatest improvements in coral cover and had the most complex fish populations (Motta, this volume). Some recovery has been recorded, howver, in the form of primary colonisation by soft corals.

In Kenya and Mozambique in particular, recovery from the El Niño has been reversed on significant reef areas by new threats, which are outlined below.

ADDITIONAL THREATS AFFECTING CORAL REEFS

Over-fishing and destructive fishing, pollution, mining, deforestation and poor land management, and poorly managed and planned tourism are the major localized stresses damaging coral reefs on the Eastern African coast. Excessive and destructive fishing was the major anthropogenic problem for reefs in East Africa in the 1990s (Muthiga *et al.*, 1998) until the damaging 1998 El Niño bleaching event. The damaging fishing practices include the use of dynamite, pull-seine nets, poisons, over-exploitation of small fish in small mesh nets and traps, and over-harvesting of octopus, shellfish, sea cucumber and lobster. However new threats have emerged in 2001 and 2002, acting on scales intermediate between localized anthropogenic stresses, and regional scale climate change. Their direct causation is not yet known, though meso-scale climate changes are likely to be important factors, and the contribution of climate change to changes at this level are currently unknown.

Floods and Changing Rainfall Patterns

In February to May 2001, there were unprecedented rainfall and floods in southern Mozambique which increased sediment transport to the coastal zone (Motta *et al.*, this volume). Hard and soft corals in the Xai-Xai lagoon suffered declines of 60% and 95% (from 19% and 5% cover respectively), with minimal recovery by December 2001. This effect rivaled the impact of the El Niño at this latitude, and reversed recovery on reefs affected by the El Niño.

Harmful Algal Blooms

In December 2001–February 2002, a number of medium to large-scale Harmful Algal Blooms (HAB) impacted the northen coasts of East Africa, in Somalia, Kenya and Tanzania, as well as being reported in the Gulf of Aden, in Oman and Yemen (Table 2). The largest blooms appeared to impact Oman and Yemen, and Somalia and Kenya, in January–February 2002. Sampling of waters in the last stages of the Somali-Kenya bloom enabled identification of the toxic alga as *Karenia mikimoto* (formerly variously known as *Gymnodinium nagasakiense* and *G. aureolum*) by G. Pitcher (South Africa). The event coincided with stronger than normal upwelling in the Somali Current system and stronger onshore winds of the northeast monsoon, resulting in high phytoplankton concentrations onshore in southern Somalia and northern Kenya (D. Robins, Table 2 on next page). This coincidence of events may have caused the bloom, but to now no cause has been confirmed.

Extensive fish kills were reported for both blooms, including surgeon fish, parrot fish, snappers, puffers, triggerfish, eels, rays and more. The blooms were also associated with changes in billfish and tuna distributions. Human health effects were reported but unconfirmed, including stinging eyes and sore throats, but

Table 2. Area and date for reporting of Harmful Algal Blooms and coral disease in East Africa and the Gulf of Aden, December–February, 2002. Communications reported through an email discussion thread (January–March 2002) initiated by reporting of HAB in Kiunga, by J. Church

	Area	First report	Source
Harmful Algal Bloom	Zanzibar	Dec. 2001	J. Mmochi, N. Jiddawi and M. Kyewalyanga, Institute of Marine Science, Zanzibar; Winley Sichone/Salim Amar, Menai Bay Conservation Area
	Oman	Dec. 2001	S. Wilson
	Yemen	Jan 26, 2002	O. Portrat
	Somalia	Jan 8, 2002	UNDP
	Kiunga, northern Kenya	Jan 23, 2002	J. Church, World Wide Fund for Nature (WWF)
	Somalia-Kenya, Oceanography	End Jan, 2002	Oceanographic conditions (1mg/m^3 chlorophyll) reported from SeaWIFS (NASA) data by D. Robins, Marine Biological Laboratories, Plymouth, UK.
	Northern Kenya, plankton ID	Feb. 6, 2002	Identification of *Karenia mikimoto* by G. Pitcher, South Africa
Coral Disease	Kiunga, northern Kenya	mid-February, 2002	J. Church, World Wide Fund for Nature (WWF)
	Southern Kenya	Feb 25, 2002	T. McClanahan, Wildlife Conservation Society
	Tanga, northern Tanzania	mid-March, 2002	E. Verheij, Tanga Coastal Zone Conservation and Development Project

mortalities were not confirmed. As a precaution, however, reef fisheries were halted in northern Kenya for 3 weeks, with considerable economic consequences to families in the region.

Coral Disease

Shortly after the HAB dissipated in Kenya, an unknown coral disease was first reported in northern Kenya, and subsequently observed in southern Kenya and northern Tanzania (Table 2). The disease primarily impacted the genera *Montipora*, *Astreopora* and *Echinopora*, in some aras causing near 100% mortality (Obura, this volume). Many other coral genera were affected but to a lesser extent and more variably among regions, including *Pocillopora*, *Acropora* and *Platygyra*. The disease was subsequently identified by electron microscopy as a fungus (T. McClanahan, pers.comm.), though of unknown identity.

Climate Change at High Latitudes

A steady increase of 0.27^{o}C has been measured on S. African coral reefs since *in-situ* temperature recording began in 1994 (Schleyer & Celliers, this volume). During this time, hard coral cover has increased and soft coral cover remained stable on reef slopes at 12m depth. On reef tops, hard coral cover increased to a lesser extent, and soft coral cover decreased. While bleaching was absent in 1998, it did occur in 2000 associated with elevated temperatures over 28.8^{o}C and clear water conditions. In the short term it appears that warming temperatures may improve conditions for corals in high latitude reefs, especially at depth, however the long term increase in temperatures is likely to be detrimental here as well. Also important will be the change in water chemistry and pH that may significantly reduce calcification rates of corals.

In addition to the epidemics of planktonic algae and disease, increased occurrence of unusual invertebrate (mantis shrimp) and fish swarms have been reported from East Africa (Richmond *et al.*, in press). Whether these epidemics may be related to oceanographic and monsoon changes influenced by climate change is not yet possible to know, but if so, they are likely to increase in frequency and intensity in coming years, along with the multiple stresses to reef corals that they are associated with.

SOCIO-ECONOMIC MONITORING

An analysis of the socio-economic impacts of the El Niño found that there were subtle changes in fish biomass and fisheries yields in Kenya and the Seychelles, but these could not be attributed to bleaching impacts. Furthermore, while assessments of 'Willingness to Pay' by tourists has indicated potential bleaching-related losses of US$ 5–9 million in 2001 for Mombasa, Zanzibar and the Seychelles, these theoretical looses to tourism have not been demonstrated to have occurred in practice (Cesar *et al.*, 2002).

Socio-economic monitoring of coral reefs is in a trial phase in East Africa, building on a relatively strong base of socio-economic assessments over the last 5 years, and in parallel with developments in South Asia (GCRMN, 2002). This project by CORDIO is identifying indicators and methods that may be applied by a conservation or fisheries-related project to monitor progress towards its objectives, through use of local staff and/or community members alongside scientists in conducting the monitoring (Malleret-King & King, 2002). The project has identified three core areas for data collection, resource use patterns, household occupational structure and attitudes towards interacting groups. Initial tirals in Diani, Kenya have been conducted, with 2 further pilot sites in Kenya and Tanzania targetted. Monitoring is intended to be conducted twice yearly, during different seasons.

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Hodgson-initial(er)?

Coral Reef Degradation in Tanzania: Results of Monitoring 1999–2002

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Key words: coral bleaching, coral degradatio, coral recruitment, El Niño, Tanzania, western Indian Ocean

INTRODUCTION

Coral reefs play a crucial role in the well-being of coastal communities in Tanzania (Muhando, 1999; Johnstone *et al.*, 1998). Despite their usefulness, coral reefs are being degraded by destructive anthropogenic activities (Salm *et al.*, 1998) and natural causes (e.g., coral competition, predation, diseases, bleaching, etc.). The 1998 coral bleaching and mortality event 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. Highest coral mortality was recorded in Tutia reef in Mafia Island Marine Park and on Misali reef on the west coast of Pemba. Coral mortality was low on reefs around Unguja Island, Zanzibar (Muhando, 1999). After the bleaching and coral mortality event (in March–June 1998), the status of Tanzania 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 and the socio-economy of coastal populations (fisheries and tourism). Three teams of researchers were formed. The first dealt with the assessment and monitoring of coral reefs, the second dealt with socio-economic effects and the third team investigated specific issues relevant to coral bleaching, mortality and recovery.

The specific objectives of the coral reef assessment and monitoring team in Tanzania were as follows:

- To assess and monitor the benthic cover on coral reefs on selected sites on Tanzania mainland and the Islands.
- To assess and monitor the distribution and density of ecologically important macrobenthos.
- To assess and monitor the reef fish distribution and composition.
- To measure some of the environmental parameters at the study sites.
- To assess the distribution and density of coral recruits (new objective added in the last monitoring).

This presentation is an attempt to summarise reef assessment and monitoring results for the years 1999–2002.

MATERIALS AND METHODS

Study Sites

Coral reef assessment and monitoring were carried out around Unguja (Zanzibar) Island reefs (Chapwani, Changuu, Bawe, Chumbe and Kwale), and on Misali reefs (West of Pemba Island). The geographic positions (in UTM units), exposure to waves (protected or semi-exposed) and tidal current strength (strong, medium, weak), the number of random and permanent transects are given in Table 1 (next page).

Table 1. The geographic positions (in UTM units), exposure to waves (protected or exposed) and tidal current strength (strong (S), medium (M), weak (W), the number of random (RT) and permanent transects (PT) at each study site

Station	Position (UTM)	Exposure	Currents	Length of each point transect (m)	No. of point transect (m)
Chapwani	521205 9323362	Protected	W	50	6
Changuu	518524 9324192	Protected	W	50	4
Bawe	514784 9321110	Protected	W	50	9
Chumbe	519205 9306420	Protected	W	50	9
Kwale	531327 9294468	Protected	M	50	6
Misali*	566038 9421475	Semi-exp.	M	20	8
Tutia	571445 9102894	Semi-exp	S	50	20
Utumbi	586768 9121552	Protected	S	50	13
Msumbiji	587441 9120090	Protected	S	50	8

*permanent line intercept transects were used

Measurement of Coral Reef Benthic Cover

Benthic reef cover was estimated using permanent Line-Intercept Transect (LIT) method (English *et al.*, 1994) in Misali Island and random Line-Point Transect (LPT) (Allison, 1996) in Zanzibar reefs. Random transects were placed on the study site and benthic categories were recorded. A total of 200 points were recorded on each 50 m random transect. The interval between points was 25 cm. A plumb-line was used to reduce observer parallax errors (Allison, 1996). At all sites, transects were set on the reef flat (1–5 m) and reef slope (5–15 m) parallel to the reef front. The benthic categories on the reef were then categorised into seven groups; live hard corals, soft corals, coralline algae, fleshy algae (including filamentous algae), sponges, dead coral, substrate (rubble, sand, rock and silt) and others (*Rhodactis*, zoanthid, clam, seagrass and others). Live corals were recorded according to their life-forms (as in English *et al.*, 1994).

Reef Macrobenthos (macro-invertebrates)

Reef animals that are relatively easy to see and could indicate the health of the coral reef were counted in belt transects 50 m long and 2 m wide set randomly on the reef. A T-stick was used to maintain the belt width. The macrobenthos counted included crown-of-thorns starfish (*Acanthaster planci* – COTS), sea cucumbers (Holothuroidea), sea urchins (Echinoidea), sea stars

(Asteroidea), clams (*Tridacna*), gastropods (commercial species), and lobsters.

Fish Count

Fish were counted in a 50 x 5 m belt (250 m^2). On each occasion, fish were counted on reef flat and on the adjacent reef slope. The counting started 5 to 10 minutes after placing the transect line to allow fish to resume their normal behavior. Attention was given to fish families that were considered to be commercially or ecologically important (see Mohammed *et al.*, 2000). The reef fish were categorised into six trophic groups; corallivores, piscivores, herbivores, invertivores, omnivores, planktivores and spongivores.

Coral Recruitment Study

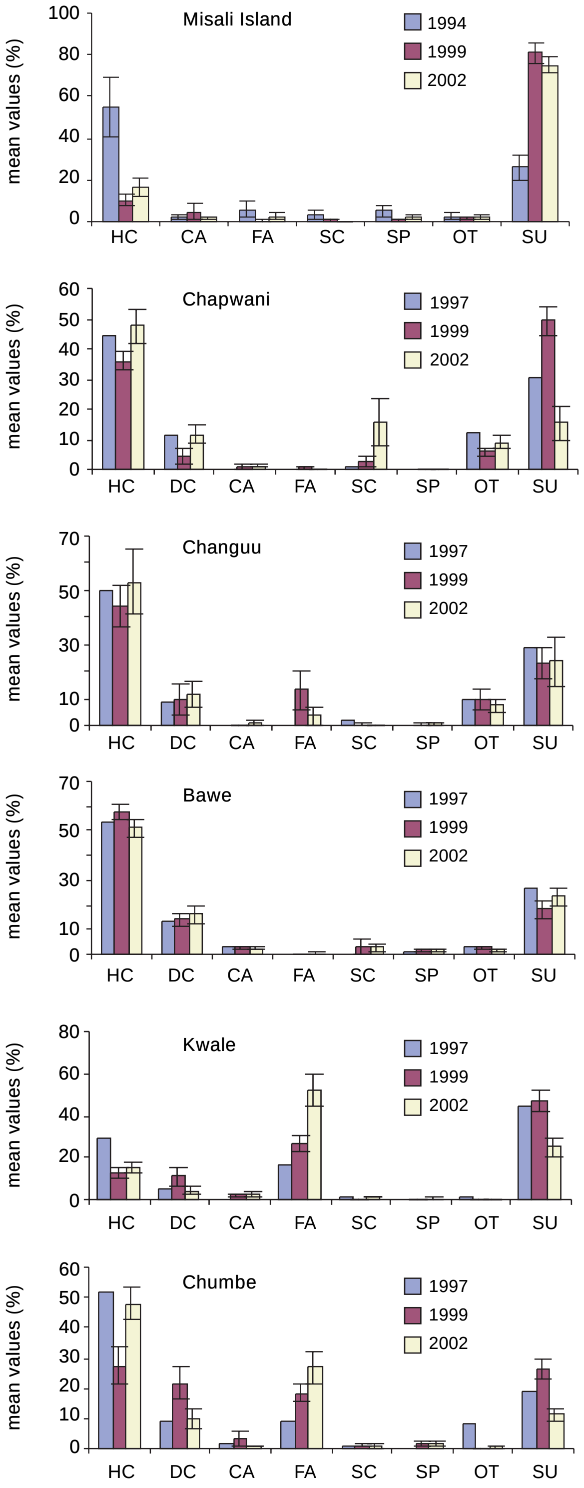
Coral recruitment count was done in Misali and Mafia only. Coral recruits included in the study were those about 5 cm or less. The recruits were counted in 1 m^2 quadrats spaced every 10 m along a transect 50 m long. Recruits were grouped into *Acropora*, *Pocillopora* and others (all remaining genera).

RESULTS

Benthic Cover

MISALI, PEMBA

Benthic cover of Misali showed minor changes since the first survey of 1999 made after the bleaching event. However, the changes are dramatic when compared to before the bleaching in 1998 (Fig. 1). A major decrease is observed in live coral between 1994 (Horrill *et al.*, 1994) and 1999 while between 1999 and 2001 the



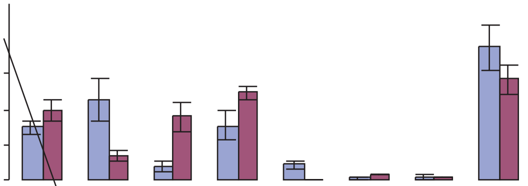
HC = hard coral, DC = dead coral, CA = coralline algae,
FA = fleshy algae, SC = soft coral, SP = sponge,
OT = other organisms, SU = substrate

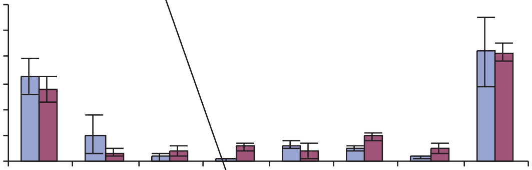
Figure 1. Benthic cover of the Misali and off Zanzibar town reefs showing changes from 1994/97 to 2001/02. (Bars indicate standard error.)

changes are minor and show slight increases of hard coral, fleshy algae and sponge. Soft coral and coralline algae are decreasing.

ZANZIBAR TOWN

Coral reefs off Zanzibar town have not suffered great loss of live corals between 1997 and 1999. A slight decrease of hard coral cover has been observed on Chapwani and Changuu while on Bawe there was an increase. Higher loss of live coral cover was on Chumbe and Kwale (Fig. 1). Between 1999 and 2002 hard coral cover increased in Chapwani, Changuu and significantly at Chumbe but hard coral cover remained the same in Kwale and decreased in Bawe. Beside the minor fluctuations of other categories (dead coral, coralline algae, fleshy algae, soft coral, sponge other organisms and substrate) between the pre and post bleaching period, there is a sequential increase of soft coral in Chapwani and fleshy algae in Chumbe and Kwale. Substrate cover has decreased significantly in Chapwani Kwale and Chumbe.





MAFIA ISLAND

In Mafia live coral cover has decreased slightly in Msumbiji and Utumbi reefs while increased slightly in Tutia reef (Fig. 2). Percentage of dead coral cover has been reduced in all three reefs monitored. Coralline algae coverage has increased significantly in Tutia and the differences are low in the other two reefs. Fleshy algae have increased in all three reefs while soft coral cover is lower compared to 1999. There are increases of other organisms and sponge cover in Msumbiji, but these categories have remained the same since 1999 in Tutia and Utumbi. Substrate cover has not changed much from the 1999 assessment.

MACROBENTHOS (MACRO-INVERTEBRATES)

A summary of the macroinvertebrate count results for each site between 1999 and 2001/2002 are shown in Fig. 3. High densities of sea urchins were recorded on reefs of Bawe and Chapwani with no significant changes between the sample periods. Medium levels of sea urchins were recorded in 2001 on the Misali island reefs, an increase since 1999. The lowest number of sea urchins was recorded in Chumbe and Kwale reefs for both sampling periods. However, these reefs did experience a relative increase in abundance of sea urchins from 1999 to 2002. Gastropods of commercial value and lobster were generally occuring at low densities and did not show fluctuations. The number of sea cucumbers and starfish were also low in all the sites, and decreased in Kwale and Chumbe. The density of other bivalves was high only in Chapwani. COTs were reported to occur in all sites of Zanzibar in 1999, and highest at Bawe. In 2002 few COTs, were observed in Bawe. Lobsters were not observed.

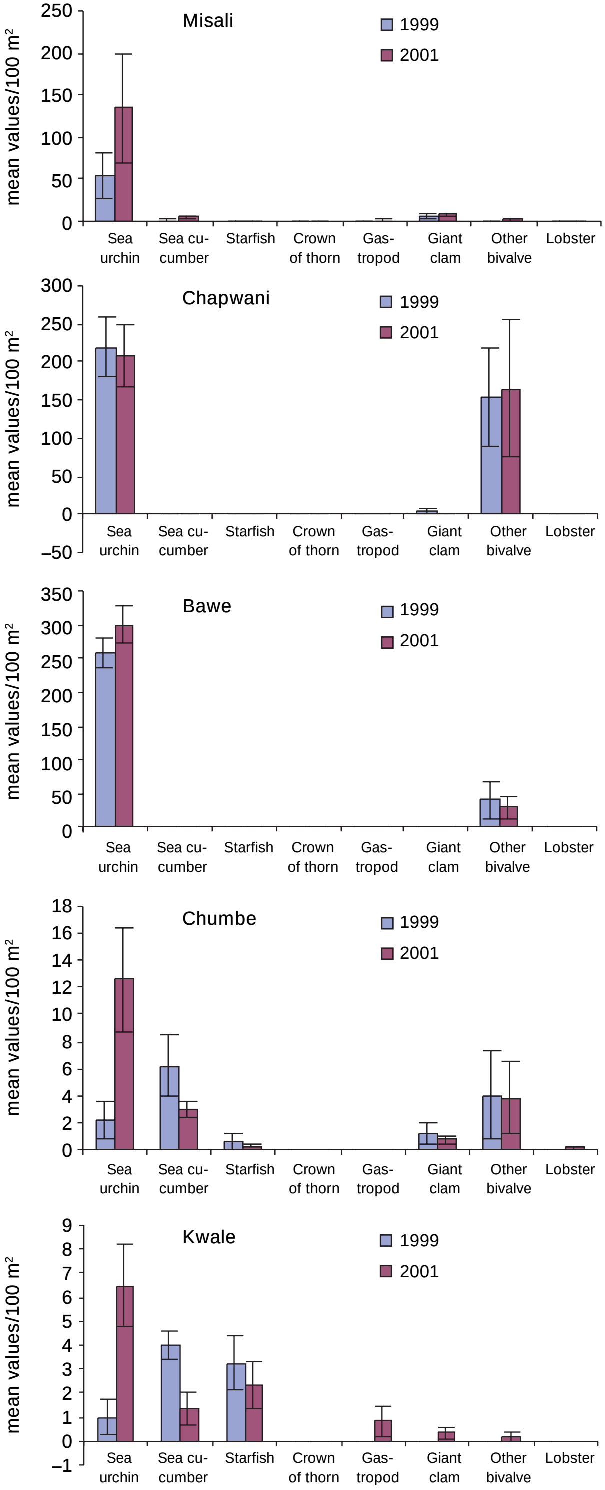


Figure 3. Macrobenthos (macro-invertebrates) of the Misali and off Zanzibar town reefs showing changes from 1999 to 2001/02. (Bars indicate standard error.)

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Figure 4. Macrobenthos (macro-invertebrates) of the Mafia Island reefs showing changes from 1999 to 2001. (Bars indicate standard error.)

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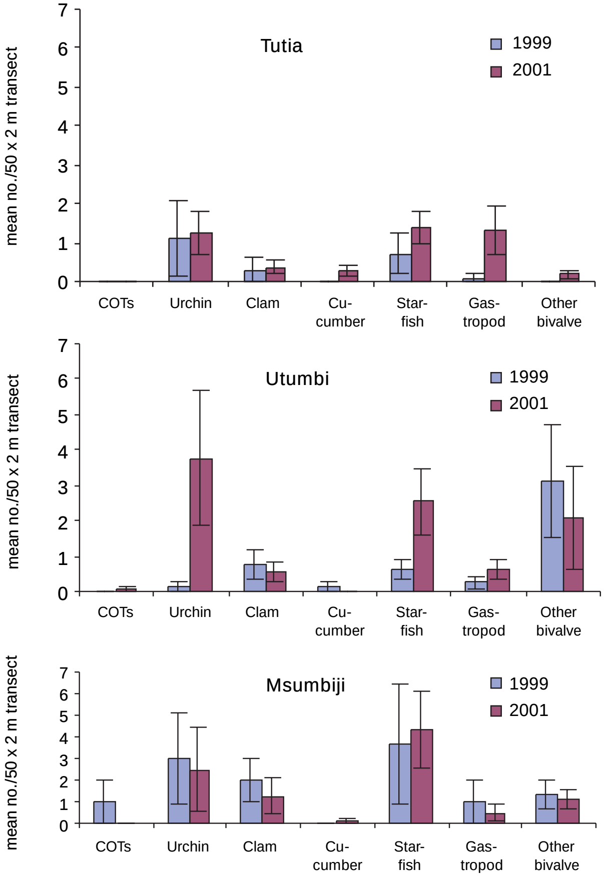


In Mafia a comparison of macroinvertebrates between the two monitoring sessions show a disappearance of COTs in Msumbiji (Fig. 4). In Tutia an increase is observed in sea cucumbers, starfish, gastropod and other bivalves. In Utumbi an increase in the number of urchin, starfish and gastropod was observed while the number of other bivalves number was reduced.

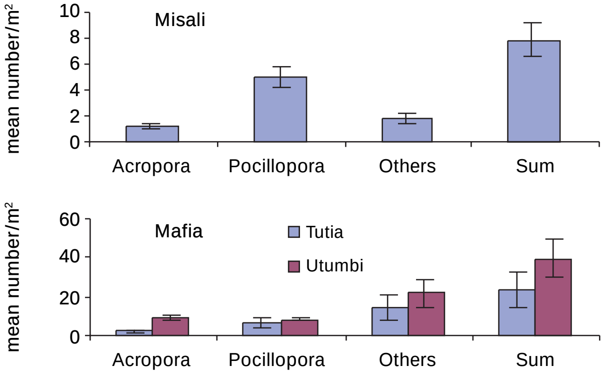


Figure 5. Number of coral recruits in Misali Island and Mafia Island for the year 2001. (Bars indicate standard error.)

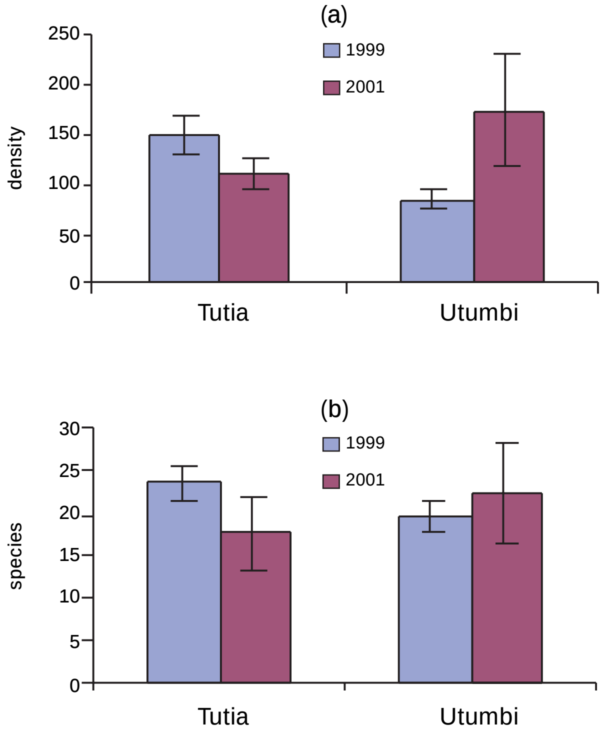


Figure 6. (a) Total fish density and (b) species of fish in all transects.
(Bars indicate standard error.)

CORAL RECRUITS

The numer of coral recruits are generally low on Misali Island, at 8 m^{-2} (Fig. 5). Density of *Pocillopora* recruits were highest followed by other groups collectively. Recruits of *Acropora* were the lowest in number. In Mafia the number of coral recruits at Tutia and Utumbi is low in general, with the former much lower in number than the latter.

UNDERWATER FISH CENSUS

Density Comparison

TUTIA

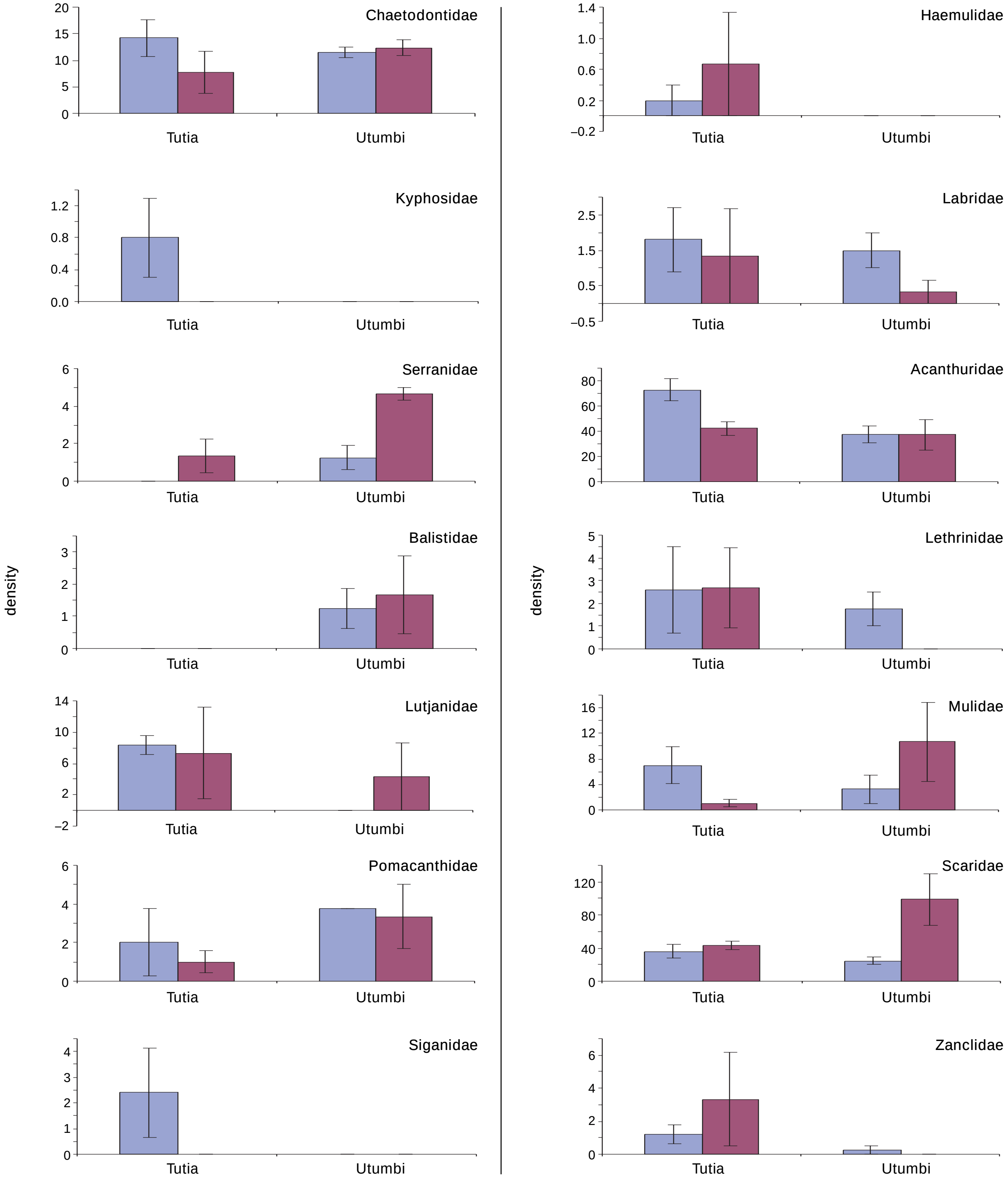
At Tutia reef, fish density had decreased from an average of 149.6 individuals per 250 m^2 in 1999 to 112 in 2001. The average species richness of fish in this site was 23.8 species per 250 m^2 in 1999 and 17.6 in 2001 (Fig. 6).

The change in fish densities by family was noticed in all families except Lethrinidae. Eight families showed a decreasing pattern, while only 4 families had higher densities in 2001 as compared to 1999. Considering higher variability associated with this method, the number of families that showed 'significant' change was only 5 out of 14 families. These were Kyphosidae, Serranidae, Acanthuridae, Siganidae and Mulidae. The densities of Kyphosidae, Acanthuridae, Siganidae and Mulidae decreased from 0.8 to 0 individuals, from 72.8 to 42.3 individuals, from 2.4 to 0 individuals, and from 7 to 1 individual, between 1999 and 2001, respectively. The only family that showed an increasing trend was Serranidae which had no individual in 1999 and 1.3 individual in 2001 (Fig. 7).

Trophic analysis of fish species in Tutia shows an increasing percentages of herbivores, piscivores, invertivores and the unidentified group, while corallivores, invertebrate and fish feeders and planktivores decreased. The percentage of omnivorous fishes remained almost unchanged. Of all these changes, a profound increase was

Figure 7. Comparison of different families over the years. (Bars indicate standard error.)
🟦 1999 🟥 2001

▶


density
Chaetodontidae
20
15
10
5
0
Tutia
Utumbi
Haemulidae
1.4
1.0
0.6
0.2
−0.2
Tutia
Utumbi
Kyphosidae
1.2
0.8
0.4
0.0
Tutia
Utumbi
Labridae
2.5
1.5
0.5
−0.5
Tutia
Utumbi
Serranidae
6
4
2
0
Tutia
Utumbi
Acanthuridae
80
60
40
20
0
Tutia
Utumbi
Balistidae
3
2
1
0
Tutia
Utumbi
Lethrinidae
5
4
3
2
1
0
Tutia
Utumbi
Lutjanidae
14
10
6
2
−2
Tutia
Utumbi
Mulidae
16
12
8
4
0
Tutia
Utumbi
Pomacanthidae
6
4
2
0
Tutia
Utumbi
Scaridae
120
80
40
0
Tutia
Utumbi
Siganidae
4
3
2
1
0
Tutia
Utumbi
Zanclidae
6
4
2
0
Tutia
Utumbi
density

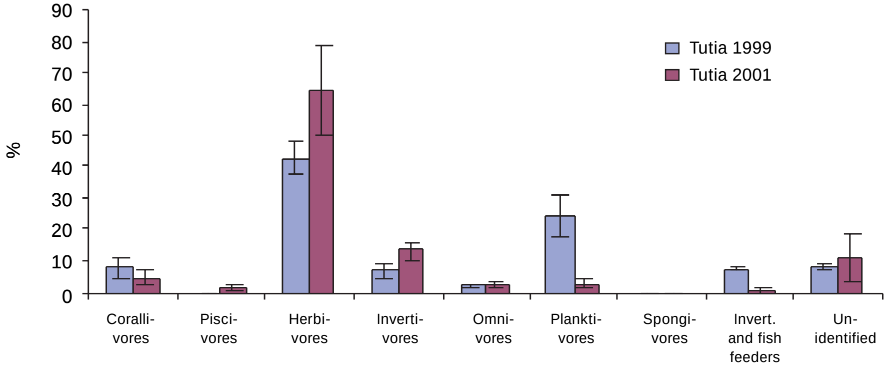


Figure 8. Fish community structure comparison at Tutia. (Bars indicate standard error.)

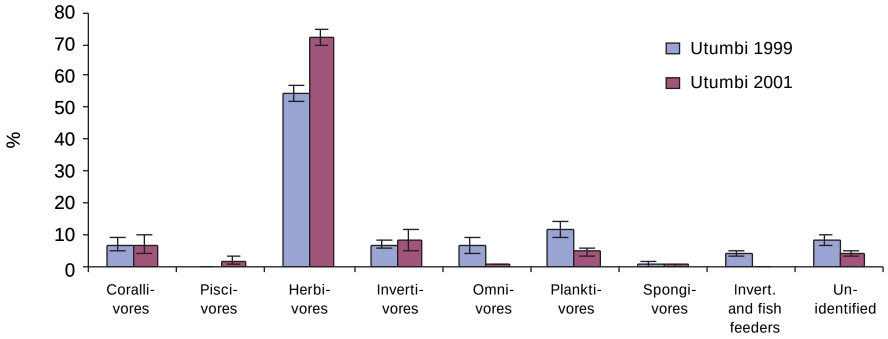


Figure 9. Fish community structure comparison at Utumbi. (Bars indicate standard error.)

found in herbivores and invertivores, 21.25% and 5.89% respectively. Planktivores decreased by 21.4%, while invertebrate and fish feeders decreased by 6.9% (Fig. 8).

UTUMBI

At Utumbi fish density increased from 86.75 to 173 indivuduals per transect (S.E. 9.53 and 56.26 respectively). There was an average of 19.5 species per transect in 1999 and 22.33 in 2001 (Fig. 6).

A comparison of densities by family indicates, as in the Tutia case, a mixed pattern with only 2 families having a 'significant' increase, (Serranidae: 1.25 to 4.67 individuals; and Scaridae: 24.75 to 99.0 individuals); 2 families with a significant decrease (Labridae: 1.5 to 0.33 individuals; Lethrinidae: 1.75 to 0 individuals). Seven families displayed no significant change, including the Chetodontidae, Balistidae, Acanthuridae, Lutjanidae, Pomacanthidae, Mulidae, and Zanclidae (Fig. 7).

In Utumbi there was significant increase in herbivores (17.6%) and piscivores (1.9%). There was also a slight, but not significant, increase in invertivores. The rest of the groups decreased with omnivores, planktivores and invertebrate and fish feeders and unidentified groups showing significant changes of –6.28%, –7.39%, and –3.67%. Corallivores and omnivores remained almost unchanged (Fig. 9).

DISCUSSION

Results obtained during the second monitoring after the bleaching event are being compared with the information collected in the first monitoring of 1999 and the monitoring done before the bleaching.

Misali Island, which had the most severe mortality of corals of up to 80% (Mohammed *et al.*, 2000) shows very little recovery. Chumbe, which experienced intermediate coral mortalityin 1999, showed some recovery. Both reefs are in protected areas. Kwale reef experienced a similar extent of reef damage as Chumbe, but its recovery has been much poorer. They both have similar environmental conditions, however Kwale reef is not protected. Thus Chumbe had higher coral cover before the bleaching, and has shown better recovery after the bleaching. This may be due to the presence of more corals that can produce more larvae, or regeneration of colonies that survived the bleaching.

The reefs of Bawe, Changuu and Chapwani in Zanzibar showed less mortality caused by the bleaching event. The recovery rate on Changuu and Chapwani is not high and is similar to that of Kwale. Fishermen using destructive fishing methods visit these reefs very frequently. Among the reefs monitored, Bawe is the only one showing a slight decrease of coral cover, from 1999 to 2002. Conditions prevailing in Bawe are the same as those of Changuu and Chapwani, therefore no threat which is specific to Bawe coral reefs have been observed recently except for the increased COTs in 1999 (Mohammed *et al.*, 2000) which probably are responsible for the reduced coral cover.

Death of hard corals may give a chance for fleshy algae to proliferate (see Bell, 1992; Done, 1992). In Chumbe and Kwale there is seasonal growth of dense macroalgae a condition that cannot be linked with coral death. However, thick macroalgal cover can pose a problem in coral settlement and reduce the process of reef recovery. In this instance, further destruction of the reef should be avoided otherwise a shift to an algal dominated reef may occur. The threat of increased algal cover was very clear in Kwale, where some coral colonies were completely covered by macro algae and showed signs of stress. Coral bleaching, a colour change to pink and growth of thick turf algae were seen (Mohammed personal observation). The decrease in substrate cover in Chumbe and Kwale was due to increased coverage of macroalgae over the bare substrate. In some of the reefs, however, algae cover decreased, such as at Misali, increasing the chances for reef recovery. Reduction of fleshy algal cover was also recorded in Changuu from 1999 to 2002.

Three years after the massive coral death in Tutia reef has passed, there is no signs of further degradation of the reef. Death of hard coral gave chance to the proliferation of fleshy algae (Mohammed *et al.*, 2000), which still continue to grow. However, fleshy algae are not the only organisms thar are increasing in numbers. Also hard coral and coralline algae are increasing. The very low growth rate of hard coral extends the time of reef recovery; two years is probably not enough to be able to forecast the direction of the new reef community.

Msumbiji reef has shown slight difference from the pattern observed on the other two reefs of Mafia Island. Here slight reduction of hard coral cover and an increase of the fleshy algae and soft coral is observed. This condition needs close monitoring of the reef.

Macroinvertebrate densities were generally low in 1999, except for sea urchins in some reefs close to Zanzibar towns. Chapwani and Bawe had the highest numbers, which did not show any significant increase. Changes in sea urchin densities are seen in reefs which over time had higher coral cover before bleaching eg. Misali, Chumbe and to some extent Kwale. Coral death and increase in fleshy algae can be the reason for the increase in sea urchin numbers. The gastropods of commercial value and sea cucumber densities were low in all sites. Low densities could be a result of over-exploitation on these reefs.

In general the densities of COTs are low, but were observed to cause substantial damage at a local scale in 1997 at Changuu (Mohammed *et al.*, 2000). However, COTs have disappeared in this area and have invaded Bawe coral reefs in 1999 where higher densities were found in areas dominated by *Acropora*. In 2002 very low numbers were observed.

The number of macroinvertebrates in Mafia reefs have not shown any major changes between 1999 and 2002. Few changes are seen in sea urchin and starfish increase in the Utumbi reef and gastropod increase in Utumbi.

Number of coral recruits in Misali and Mafia are still at a low level. The number is lower in Misali than in Tutia reef in Mafia Island.

In Misali there are very few reproducing adult coral colonies left because of the bleaching, especially *Acropora*. As a consequence there is a very low number of *Acropora* recruits. *Pocillopora* recruits are surviving relatively well and may in the future dominate the reef if this trend persists. Other coral genera recruits are higher but indicate a lower probability of Misali reefs to have high coral diversity in the future. In general the highest number of coral recruits were found on dead coral tabulates which are still standing (Mohammed personal observation). The future of these recruits is quite uncertain as it depends on how long these structures will remain standing up.

In Mafia 23 very few coral recruits were counted. A cause for thus was probably the massive death of adult coral colonies in 1998. The Songosongo reef complex, which is close to Tutia, was not severely affected by the 1998 bleaching (Mohammed Nur, personal communication). Thus reef could provide coral recruits to the affected reefs in Mafia. However, no or very few recruits from Songosongo reef apperas to be distributed to Mafia. It is not known if this is due to the prevailing current patterns during the phase when coral larvae are distributed. The observed low level of recruitment will lead to slow recovery of the reef system and hence increasing chances of competition from fast growing organisms like soft corals and fleshy algae.

Reproducing adult coral colonies have a better chance to provide more seeds in Utumbi. The bay is not as open as Tutia so the larvae will probably be retained within the bay.

The observed trends in fish densities indicate a general increase in Utumbi while a decrease was noticed in Tutia. This change in density does not correspond to the fish community structure. The community structure in these sites indicate a general similarity in trophic contributions. The difference in fish trophic structure between 1999 and 2001 show that three trophic groups contributing to up to 61% of the fish population had increased in both sites. The major contributor of this group is herbivores which forms the main part of the shallow reef fish community

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The Status of Corallimorpharia on Coral Reefs of Tanzania

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Key words: coral bleaching, corallimorpharians, El Niño, reef succession, Tanzania, western Indian Ocean

ABSTRACT

The distribution and abundance of corallimorpharians after the 1998 coral bleaching event in Tanzania was assessed in November 1999 and November 2001 on five reefs in Zanzibar and Dar es Salaam. Benthic cover of corallimorpharians differed between surveyed reef sites; it was higher on Changuu (11.8–14%), Bongoyo (11–13.7%), and Mbudya (5.1–6.4%), and lower on Bawe (3.4–3.8%) and Chumbe (2.4–3.3%) reefs. Higher cover of corallimorpharians was observed on the reef crest (9.8%) and reef flat (9.5%) than on the reef slope (3.6%). Four corallimorpharian species occurred on these reefs: *Rhodactis rhodostoma*, *Actinodiscus unguja*, *Actinodiscus nummiforme*, and *Ricordea yuma*. *Rhodactis rhodostoma* was the dominant corallimorpharian in terms of cover. Corallimorpharians distribution was associated with non-living substrate (rock and rubble) on degraded reefs suggesting that corallimorpharia, specifically *Rhodactis rhodostoma*, may become a threat on reefs that are degraded continuously, by taking up space that might otherwise be available for coral recruitment. It appears that when environmental factors stressful to corals decrease and coral recovery is started, proliferation of corallimorpharians ceases.

INTRODUCTION

Corallimorpharians are soft-bodied coral-like anthozoans. Superficially, corallimorpharians resemble actinianian sea anemones in that they lack a calcareous skeleton (Carlgren, 1949). However, they are more similar to scleractinian corals in most other aspects of their morphology. Corallimorpharians appear to establish themselves on degraded shallow reefs where they may effectively prevent coral settlement (Langmead & Chadwick-Furman, 1999; Chadwick-Furman & Spiegel, 2000) and competitively exclude reef-building corals, especially those in the families Acroporidae, Pocilloporidae, and Poritidae (den Hartog, 1977; Miles, 1991; Kuguru, 2002).

Since 1996, the abundance of corallimorpharians appears to have increased on frequently-fished and/or disturbed reefs (Johnstone *et al.*, 1998) along the east African coast of Tanzania, including sites in Zanzibar, Tanga, Dar es salaam, Mafia, and Mtwara (Kuguru, 2001; Muhando, 1998). The unprecedented bleaching and subsequent mortality of scleractinian corals on many reefs in the Indian Ocean in 1998 (Wilkinson *et al.*, 1999) resulted in further degradation (Muhando, 1999; Mohammed *et al.*, 2000) and may have increased space available for corallimorpharians and other organisms to become established (Rajasuriya & Kamnarathna, 2000; McClanahan *et al.*, 2001). Mortality of corals has been observed to induce a shift in the composition of other reef organisms and a significant reduction in biodiversity (Karlson 1980; Done 1992; McClanahan, 2001). Conversely, where a few species of hard coral dominated, their mortality may allow for localised increases in biodiversity.

Since corallimorpharians compete with stony corals and thus may inhibit their recovery after disturbance, it was considered necessary to monitor the relative abundance of corallimorpharians and their association with other benthic organisms. This study aimed at determining the relative contribution of corallimorpharians on the reef benthos on selected reefs as well as elucidating the relative importance of the contributing species, one year (1999) and three years (2001) after the 1998 coral-bleaching event. The study also examined the associations between corallimorpharians and other benthic organisms on the reef environment.

MATERIALS AND METHODS

The study was conducted in November 1999 and repeated in November 2001 at five coral reef sites in Tanzania: Mbudya and Bongoyo just north of Dar es Salaam, and Bawe, Changuu and Chumbe on the western side of Unguja Island, Zanzibar (Fig. 1). All sites were located on patch or fringing reefs, all located in relatively sheltered areas. With the exception of the Chumbe site, which is inside a marine protected area, the other coral reef sites are frequently fished and/or disturbed by human activities.

The abundance and distribution of corallimorpharians, scleractinian corals and other benthic organisms was assessed using a combination of line intercept transect (English *et al.*, 1994) and line-point intercept methods (Wiens & Rotenberry, 1981). At each study site, five 20-m-long transects were deployed parallel to the shoreline in each of the following five reef zones: the inner reef flat, middle reef flat, reef crest and at 2 m and 4 m depth on the outer reef slope. Along each 20-m line transect, five points were randomly selected, and benthic data was recorded on a 2-m-long line-intercept transect (1 m on each side) set perpendicularly at the points. Thus, a total of 10 m was examined within each 20-m-long transect. The types of benthic organisms and substrates in contact with corallimorpharians along the transects were recorded according to categories defined in English *et al.* (1994) and Veron (1986). Corallimorpharian species were distinguished based on comparison with specimens in the Natural History Museum of Leiden, Netherlands and upon consultation with Prof. J.C. den Hartog.

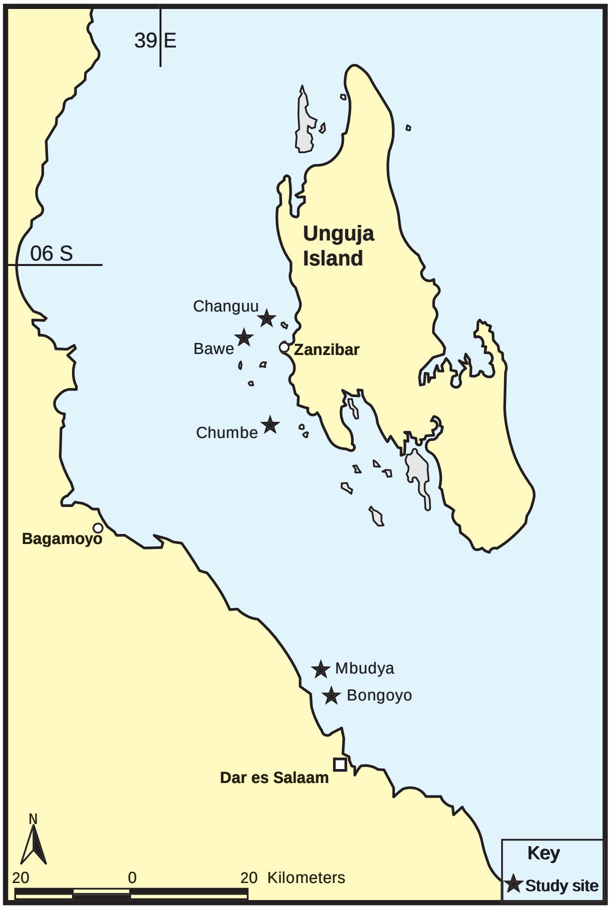

Figure 1. A map showing the location of the study areas.

Water visibility was estimated at each site using a Secchi disk (English *et al.*, 1994). The level of sediment resuspension was classified as high when the substrate at the base of the reef was composed of silt (very fine and

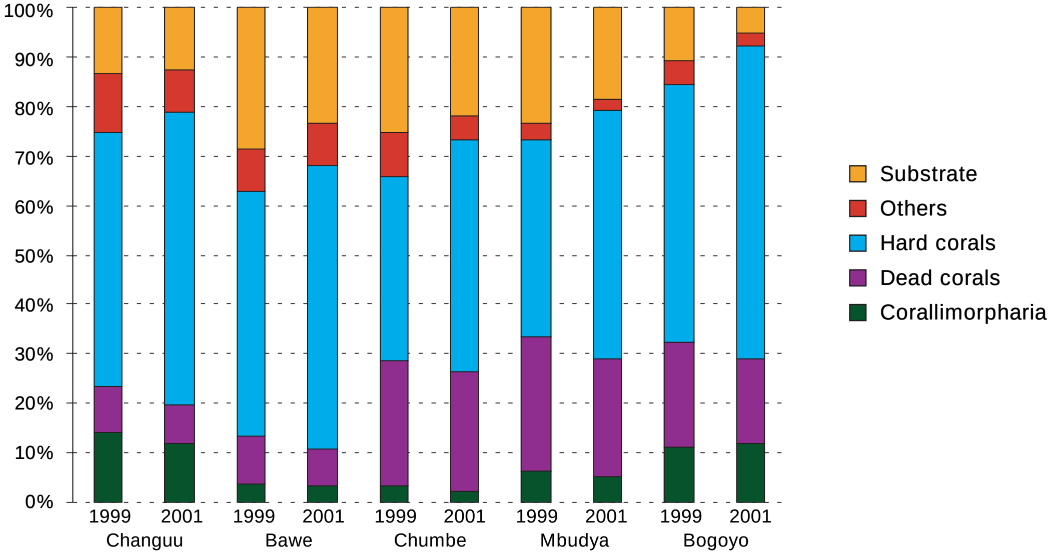

Figure 2. The relative contribution (%) of different benthic community categories in surveyed reefs in 1999 and 2001.

easily resuspended particles), low when the substrate was composed of non-resuspendable sand particles, and medium when it was a mixture of the two. The habitat selection or levels of association with other benthic categories by corallimorpharians was evaluated using Straus' (1979) linear electivity index (L).

RESULTS

The benthic category with the highest percent cover on all reefs was scleractinian corals (37–51% in 1999 and 46–63% in 2001), followed by non-living substrate i.e. rock, rubble and sand (10–29% in 1999 and 6–24% in 2001), then dead coral (9–27% in 1999 and 7–23% in 2001) (Fig. 2). The 'other' benthic category (soft corals, sponges, algae and others) had cover below 9% during both sampling periods. Corallimorpharians occurred at all of the coral reef study sites examined, with benthic cover ranging from 3–14% in 1999 and 2–14% in 2001 ranking second after hard corals in percent cover of live benthic forms in some reefs.

The cover of corallimorpharians varied significantly between sites (One Way ANOVA: F=2.92, df=4, $p<0.05$ in 1999 and F=2.8, df=4, $p<0.05$ in 2001) with Bongoyo and Changuu reefs having significantly higher cover than Bawe and Chumbe reefs (SNK test; Fig. 3). Due to high variance between transects, the Mbudya site did not appear to differ significantly from any of the other sites (SNK test; Fig. 3 on next page). The cover of corallimorpharians in 1999 at all sites was found to be not significantly different from that of 2001 (Paired sample t-test: t=0.82, df=24, p=0.40; Fig. 3). Similarly, there was no significant change in cover of corallimorpharians in the different reef zones (reef flat, reef crest and reef slope) between 1999 and 2001 (Paired Sample t-test: t=0.85, df=24, p=0.4053; Fig. 4 on next page).

There was no significant correlation between corallimorpharian abundance and scleractinian corals (Spearman Rank Order Correlation: r=–0.18, n=25, p=0.39) or dead corals (Spearman Rank Order Correlation: r=–0.17, n=25, p=0.13). Instead, the percent cover of corallimorpharians was negatively correlated with water visibility (Spearman Rank order correlation: r=–0.9, $p<0.05$; Table 1 on next page).

Four corallimorpharian species were identified: *Rhodactis (Discosoma) rhodostoma*, *Ricordea yuma*, *Actino-*

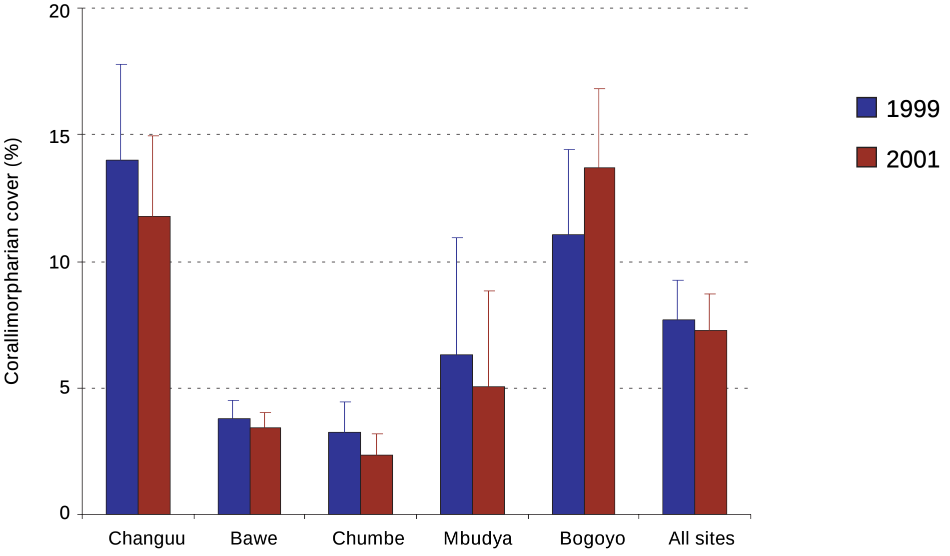

Figure 3. The mean cover of corallimorpharian (%) on the surveyed reefs in 1999 and 2001. NB: Error bars = SD.

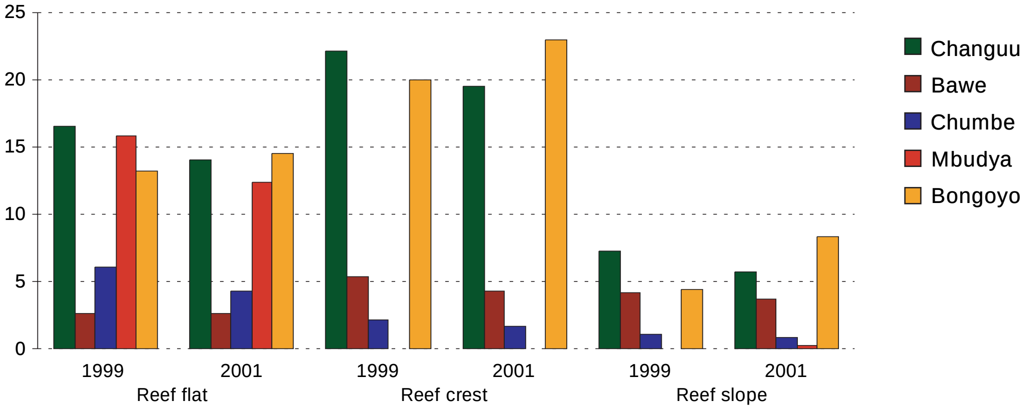

Figure 4. The average corallimorpharian cover (%) on the different reef zones at each study site in 1999 and 2001.

Table 1. Water visibility (Secchi disk), sediment resuspension levels, corallimorpharian cover (%) and fishery status on surveyed reefs in Tanzania

Site	Secchi disk (m)		Sediment resuspension	Corallimorpharia (%) cover		Fishery status
	1999	2001		1999	2001	
Changuu	3.3	3.2	High	14.0	11.8	Overfished
Bawe	10.5	8.8	Low	3.8	3.4	Overfished
Chumbe	10.5	6.2	Low	3.3	2.6	Marine Sanctuary
Bongoyo	3.0	2.8	High	12.9	13.7	Overfished
Mbudya	4.8	3.9	Medium	6.3	5.1	Overfished

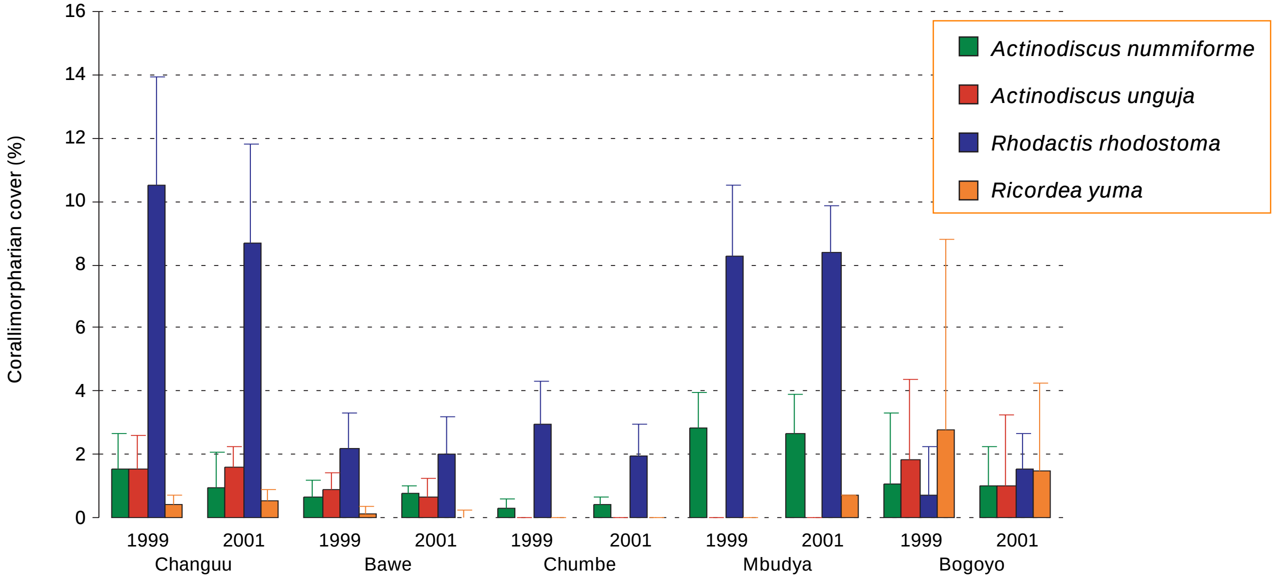

Figure 5. The average distribution and abundance (% cover) of corallimorpharian species on the surveyed reefs in 1999 and 2001. NB: Error bars = SD.

discus (Discosoma) unguja and *Actinodiscus (Discosoma nummiforme*. All four corallimorpharian species occurred on Changuu, Bawe and Bongoyo reefs (Fig. 5). Only two species, *A. nummiforme* and *Rhodactis rhodostoma*, occurred at Chumbe and Mbudya reefs. *Rhodactis rhodostoma* was the dominant corallimorpharian on four surveyed reefs (74% of all corallimorpharians in Changuu, 57% in Bawe, 85% in Chumbe, and 74% in Mbudya) (Fig. 4). *Ricordea yuma* was the dominant corallimorpharian on Bongoyo where it contributed about 44%.

The corallimorpharians were not uniformly distributed among the reef habitats surveyed (Friedman's test, $X^2 = 9.25$, DF = 4, $P < 0.05$; Fig. 6). *Rhodactis rhodostoma* and *Actinodiscus unguja* occurred in all habitats (reef zones) examined, *Actinodiscus nummiforme* was observed mostly on the reef crest and slope. *Ricordea yuma* was distributed mostly on the reef flat (Fig. 6).

Corallimorpharians were found growing on or in contact with dead corals, rubble, rock, living scleractinian corals, and soft corals. Scleractinian coral species that were in direct contact with corallimorpharians included *Acropora* spp, *Porites cylindrica*, *P. nigrescens*, *P. annae*,

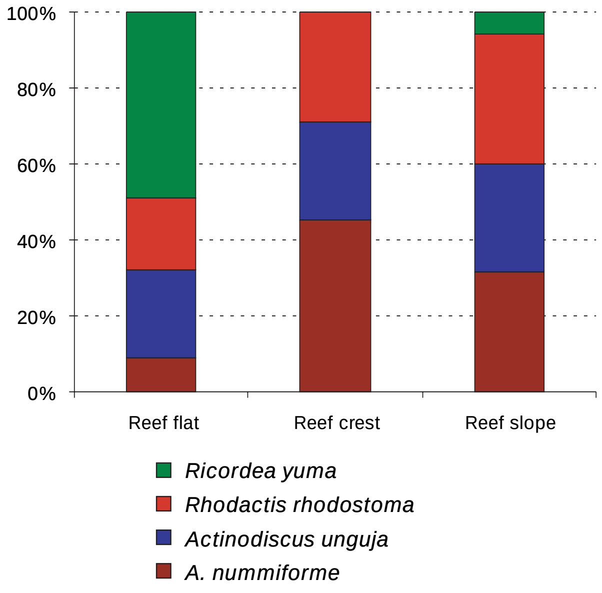

Figure 6. The combined (all sites) mean distribution and abundance of corallimorpharian (% cover) on the different reef zones (reef flat, reef crest and reef slope) studied.

Table 2. Proportions of corallimorpharians in contact with benthic categories (R), the relative contribution of the benthic category on the reef (P), and the Strauss (1979) Linear selectivity index (L) for each category (all sites pooled)

	R	P	L=R–P	L
Hard coral	0.184	0.455	–0.271	**
Dead coral	0.372	0.189	0.183	*
Rock and rubble	0.339	0.203	0.136	*
Algae	0.004	0.045	–0.041	ns
Soft coral	0.100	0.005	0.095	ns
Sponges	0.000	0.008	–0.008	ns
Others	0.000	0.014	–0.014	ns

NB: Selectivity index (L) values range from –1 to 1, with positive values indicating preference and negative values indicating avoidance or inaccessible; Dead coral=dead coral + dead coral with algae;
ns=L value not significant;
*=significant ($p<0.05$);
**=very significant ($p<0.01$)

P. lutea, *P. lobata*, and *Pavona* spp. The corallimorpharians were not in contact with all benthic categories with equal frequency (Friedman test: (X^2=9.25, df=4, $P<0.0001$). Corallimorpharian contacts also were not proportional to the relative abundance of the benthic category types (Strauss [1979] electivity index: Table 2). Corallimorpharians preferred to establish on dead coral (L=0.18) and non-living substrate, e.g. on rock and rubble (L=0.14), whilst live coral was either avoided or inaccessible (L=–0.27) (Table 2).

DISCUSSION

The corallimorpharians were not mentioned as an important coral reef benthic cover component on Tanzania reefs, in surveys conducted before 1996 (UNEP, 1989; Horrill *et al.*, 1994a, 1994b; Risk *et al.*, 1993). They were first noticed on severely fished and degraded reefs (Muhando, 1998; Kuguru, 2002). The unprecedented bleaching and subsequent mortality of scleractinian corals on many reefs in the Indian Ocean in 1998 (Wilkinson *et al.*, 1999) resulted in further degradation (Muhando 1999; Mohammed *et al.*, 2000) and may have increased the benthic space available for corallimorpharians and other organisms to become established (McClanahan *et al.*, 2001). Results in this study have confirmed that corallimorpharians have now become important, ranking second after hard corals in terms of benthic cover on Tanzania reefs (Fig. 2).

The highest cover of corallimorpharians in both sampling periods (Nov. 1999 and Nov. 2001) was observed on Bongoyo, Mbudya and Changuu reefs, sites with a history of frequent reef degradation caused by destructive fishing practices (dynamite, dragnets and anchor damage) and uncontrolled tourism (see UNEP, 1989; Kamukuru, 1998; Muhando & Francis, 2000). Corallimorpharian abundance was low on Chumbe reef, which is in a marine protected area (sanctuary) since 1994. Furthermore, corallimorpharian abundance was negatively correlated with water visibility, implying that these organisms proliferate on reefs with relatively murky water and low visibility. In Mbudya and Bongoyo, the poor visibility could be a result of various inputs (solid and liquid wastes) originating from Dar es Salaam city and carried through several small rivers, and then moved further through reefs by the predominantly north flowing East African coastal currents. Bottom stirring or resuspension of settled silt particles during flooding tides (especially during spring low tides and during the southeast monsoon winds) and the frequent passage of ferryboats and high-power sports boats about 100 m from the reefs increases the amount of suspended particles on the water column affecting the physiological activity of corals (Rogers, 1979), while allowing corallimorpharians to become superior competitors (Hamner & Dunn, 1980; Kuguru, 2002). At Changuu, the suspended solids could have originated from sources on land, especially the Zanzibar municipal sewage discharge system. According to Hamner & Dunn (1980), many corallimorpharia live in turbid water where plankton and drifting detritus are abundant.

Although an increase in corallimorpharian abundance was predicted in 2001 (Muhando *et al.*, submitted), the 2001 survey results have shown that there was no difference in cover from that of 1999 in all studied sites. The no change results could be accounted for by the fact there was no El-Nino related or other environmental stress that caused significant coral mortality in the study areas, and the dead coral structures (remnants of the 1998 coral bleaching event) though covered by algae, has remained largely intact in most areas. During the period between 1999 and 2001, coral cover increased on most reefs in Tanzania (Mohammed *et al.*, 2002). Proliferation of corallimorpharia probably stopped as corals and other reef organisms started to recover, occupying more space. Some species of corals, specifically *Montipora* appeared to competitively exclude some corallimorpharians (e.g. *Rhodactis rhodostoma*) on Mbudya reefs. Long-term trends from other studies suggest that the abundance of corallimorpharia may increase with time as degradation of the observed reefs continues (Speigel, 1998; Langmead & Chadwick-Furman, 1999; Kuguru, 2002). The baseline information obtained so far in this study indicates that corallimorpharian proliferation stops when the factors causing reef degradation cease.

In the present study, higher corallimorpharian cover was found on the shallow reef crest and reef flat than on the reef slope. Similar results were observed in other studies (Langmead & Chadwick-Furman, 1999; Chadwick-Furman & Spiegel, 2000). Reef flat areas are susceptible to disturbance from both anthropogenic and natural catastrophes such as exposure to air and direct sunlight (Chadwick-Furman & Spiegel, 2000). Corallimorpharians (specifically *Rhodactis rhodostoma*) have developed mechanisms to occupy these areas, such as the formation of large colonies of polyps, which reduces the chance of being swept away by waves or becoming desiccated, compared with solitary polyps (Kuguru, 2002). They reproduce asexually by fission and budding, which reduces polyp size and builds large aggregations of polyps (Chadwick-Furman & Spiegel, 2000).

Some corallimorpharians were observed to be in direct contact with living coral colonies. Closer observation showed that colonies of *Porites* spp and *Acropora* spp that were in contact with corallimorpharians had their contact edges bleached with no sign of skeletal extension, implying that they were negatively affected. Studies elsewhere have shown that *Rhodactis rhodostoma* tend to negatively affect scleractinian corals (den Hartog, 1977), sometimes by over-growing and killing them. Corallimorpharians use marginal tentacles, which are packed with powerful nematocysts (den Hartog, 1977; Miles, 1991; Langmead & Chadwick-Furman, 1999), and may also employ mesenterial filaments, which secrete digestive enzymes (Chadwick, 1987; Miles, 1991). Corallimorpharians grow on dead coral and rock by forming large aggregations that deny space for stony coral recruitment (Chadwick-Furman & Spiegel, 2000). In this study, corallimorpharians were found growing most often on flattened dead coral, rubble and rocky areas. There was no correlation between corallimorpharian cover and dead coral, probably because corallimorpharians were unable to immediately colonise the intact and branched dead corals. The association with rubble and rocky habitats suggests that the corallimorpharians are opportunistic colonisers of non-living substrate.

Only four corallimorpharian species occurred at the study sites, although reports from other sites indicated that there are more corallimorpharian species in Tanzania (Kuguru, 2002). Besides *Ricordea yuma* which was the abundant corallimorpharian at Bongoyo reef flat, *Rhodactis rhodostoma* dominated on the other surveyed reef sites. This species is known to follow a strategy for monopolisation of space for attachment (Jackson, 1977). Traits such as the capacity to reproduce asexually, move across the substrate (Spiegel, 1998), and damage or kill neighbouring organisms (den Hartog, 1977; Langmead & Chadwick-Furman, 1999) may be important during competition for space (Chadwick & Adams, 1991). The locomotory ability of *R. rhodostoma* (Chadwick & Adams, 1991) allows them to change position once attacked, which determines in part whether they can colonize adjacent open areas or escape from competitors and predators. The capacity for damaging or killing neighbours by using specialised marginal tentacles filled with penetrating nematocysts enables them to access limited resources (Chadwick & Adams, 1991).

This study has quantified the relative importance of corallimorpharians in terms of benthic cover and has identified contributing species and their relative distribution on the different reef zones. Destructive resource harvesting practices (fishing/tourism) and natural disasters (e.g. bleaching), which result in coral death, abundant rubble and denuded rocks as well as the amount of suspended solid particles (turbidity), which reduces light penetration and stresses the physiological process of corals, appeared to be important factors enhancing the proliferation of corallimorpharians before and after the 1998 bleaching event. This study has also demonstrated that when stressful environmental factors to corals are arrested and coral recovery is started, proliferation of corallimorpharians stops. However, continued observation is required to determine whether the coral recovery process will result in regaining the reef spaces now occupied by corallimorpharians.

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Seawater Temperature on Shallow Reefs Off Zanzibar Town, Tanzania

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Key words: climate change, coral bleaching, El Niño, sea surface temperature, Tanzania, western Indian Ocean

ABSTRACT

Knowledge on temporal and spatial patterns in seawater temperature is important not only because it influences ecological processes, but also because it is required in planning and execution of various integrated coastal management activities. In order to contribute knowledge, the seawater temperature was measured on Bawe and Chumbe reefs, off Zanzibar town, Tanzania. Results showed that there was a yearly seasonal pattern in seawater temperature with lowest temperatures (24.5–25.5˚C) occurring in July–August and higher annual peak occurring in March–April (29.2˚C in 1997; 30.7 ˚C in 1998; 29.7˚C in 1999, 29.3˚C in 2000 and 29.2˚C in 2001).

A small but consistent intrusion of a cold water mass was recorded in January–February each year. Coral bleaching and mortality occurred during the period of maximum seawater temperature in March–April 1998. The temporal patterns in seawater temperature correlated positively with macro-algal abundance and coral settlement. It is proposed that marine protected area management strategies could make use of seawater temperature records as proxy indicators for the when corals are most stressed and least resistant to threats, for example to intensify law enforcement activities during warmer periods and select cooler seasons for reef rehabilitation activities.

INTRODUCTION

Seawater temperature has been considered to be the primary factor limiting coral reef survival and development of reef biotopes throughout the world (Glynn *et al.*, 1988; Coles, 1988; Glynn & De Weerdt, 1991; Yap & Gomez, 1984; Goreau & Hayes, 1994; Winter *et al.*, 1998). The lower lethal temperature limit for coral development has been observed at 18˚C and the upper limit at 31–32˚C (Jokiel & Coles, 1977; Coles 1988, Jokiel & Coles, 1990). The optimum temperature for coral growth was suggested to occur at 26–28˚C (Jokiel & Coles, 1977), closer to the upper lethal limit. Prolonged exposure to temperature approximately 30˚C or higher causes loss of photosynthetic pigments (or bleaching) (Coles & Jokiel, 1977; Hoegh-Guldberg & Smith, 1989; Glynn & D'Croz, 1991), increased mortality (Cortes *et al.*, 1984), and reduced calcification (Jokiel & Coles, 1977; Jokiel & Guither 1978).

Changes in seawater temperature affect coral breeding and spawning as well as larval settlement and growth (Birkeland *et al.*, 1979; Szmant & Gassman, 1990; Fisk & Harriott, 1990). The breeding process involves gonad maturation and release of gametes, both of which have critical temperatures. Elevated temperature may stimulate release of immature planulae that are less capable of survival (Birkeland *et al.*, 1979). Coral recruitment depends on many factors including effects of temperature on production of planulae, on the planulae themselves, on settling and on the newly settled corals (Jokiel & Coles, 1990). Large fluctuations in temperature often lower the tolerance of organisms to other factors, e.g. salinity, sedimentation and ultraviolet radiation effects and the susceptibility to diseases, ability to compete, and the ability to shed sediments also decrease (Lesser *et al.*, 1990). Yap & Gomez (1984) observed reduced growth rates on coral transplants during warmer seasons and higher growth rates during cooler times of the year.

Given the influence of water temperature on coral reproduction and development, knowledge of seasonal changes in seawater temperature has applications in coral reef management. Attempts at coral transplantation, enhancing settlement by placement of artificial substrates and selection of appropriate coral harvesting schemes should consider the influence of seawater temperature. This report gives results of five-year seawater temperature records and also associates seawater temperature and macro-algal abundance on coral reefs situated off Zanzibar town, Tanzania.

MATERIALS AND METHODS

Seawater temperature was recorded using TidBit Stowaway Temperature Loggers (Onset Corporation, USA) tied to a branching coral at 3 m depth on Bawe and Chumbe coral reefs, Zanzibar, Tanzania (Fig. 1). The loggers were tied on coral branches to ensure that it records temperature of water mass that actually pass over the coral reef. Recording started in January 1997 and is continuing. The temperature loggers was set to record seawater temperature after every ten minutes for three months, after which the logger was replaced and data downloaded into a computer. Missing data from July to October 1999 was due to breakdown/expiry of the temperature logger that was subsequently replaced.

The mean daily temperature record was plotted and subsequently correlated with the coral bleaching event, and also with corresponding seasonal macroalgal abundance records and coral larvae settlement patterns in the study area (Muhando, in prep.).

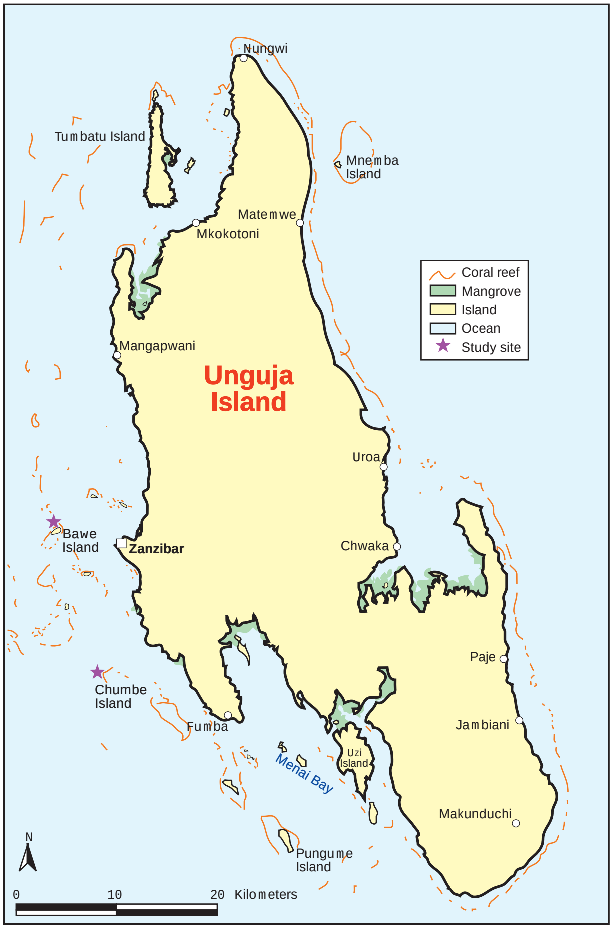

Figure 1. Map showing the location of the study area.

RESULTS

The high seawater temperature peaks were 29.2˚C on 22nd March 1997, 30.7˚C on 9th April 1998, 29.7˚C on 24th March 1999, 29.3˚C on 16th March 2000, 29.2˚C in 2001 and cold peaks were in July–August; 25.5˚C in 1997, 25.7˚C in 1998, 25.6˚C in 1999, 25.6˚C in 2000 and 24.5˚C in 2001. There was a small but consistent intrusion of cold seawater mass in January–February for all years, though most noticeable in 1997 and 2001. The seawater temperature was always higher during the Northeast (NE) monsoon (November–April: 28±0.85˚C) than during Southeast (SE) monsoon (May–October: 26±0.90˚C) (Fig. 2).

The 1998 maximum seawater temperature corresponded with coral bleaching at the study sites, between March and May 1998. The seawater temperature started to warm up in December 1997 but it was cooled by the intrusion of cold water mass in January–February before raising rapidly in March to stressful levels above 30˚C in mid March 1998 and then rapidly cooling from 30.1˚C to 28.5˚C in just 10 days (Fig. 2; Fig. 3). These rapid changes in seawater temperature in 1998 were associated with coral bleaching and subsequent mass mortalities of corals in the study area.

A positive correlation was observed between seawater temperature records and macroalgal abundance on reefs at the study sites (Fig. 4, Muhando, in prep.). Similarly, peak coral larvae settlement occurred during the NE monsoon when temperature was rising fast in October–November and during maximum seawater temperatures (January–March, Muhando, in prep.) both before and after the 1998 coral bleaching event (Fig. 5, see page 44).

DISCUSSION

For most of the year, seawater temperature was within the optimum (26–28˚C) coral growth and calcification range for scleractinian corals (Fig. 2; Jokiel & Coles, 1977; Jokiel & Coles, 1990), except between January and May when temperatures are rapidly changing and often above the optimum range, and between July and September when temperatures may be below 26˚C (Fig. 2; Jokiel & Coles, 1977; Yap & Gomez, 1984). Since the upper lethal limit is very near to the optimum range, the period between January and May can be isolated as the most stressful period for coral reefs. It was within this period, in March through May 1998 that coral bleaching and subsequent mortality of corals was observed at the study sites (Muhando, 1999) and other areas in East Africa (Wilkinson *et al.*, 1999; Obura *et al.*, 2000). The

Figure 2. The daily mean seawater temperature records on coral reefs off Zanzibar town, Tanzania, from January 1997 to February 2002. Higher temperatures in March – May 1998 were associated with coral bleaching in the study area. Note the yearly intrusion of cold water in January – February.

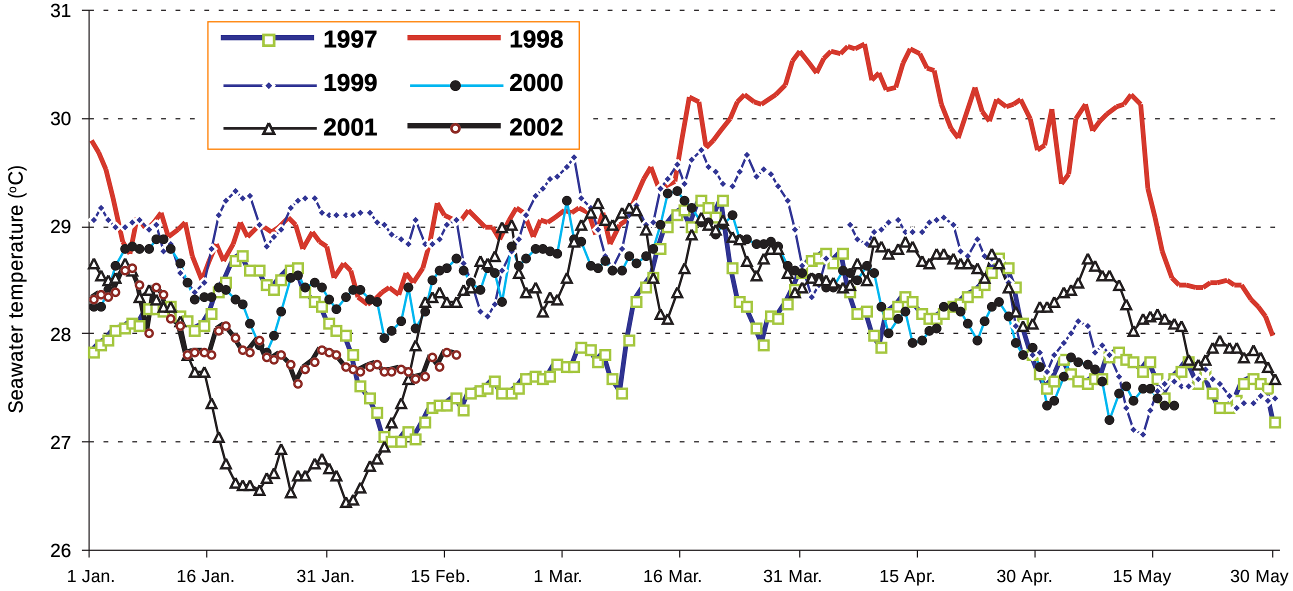

Figure 3. January to May daily mean seawater temperature records on coral reefs off Zanzibar town, Tanzania for the years 1997, 1998, 1999, 2000, 2001 and 2002. This period is characterized by higher year- to-year temperature variations.

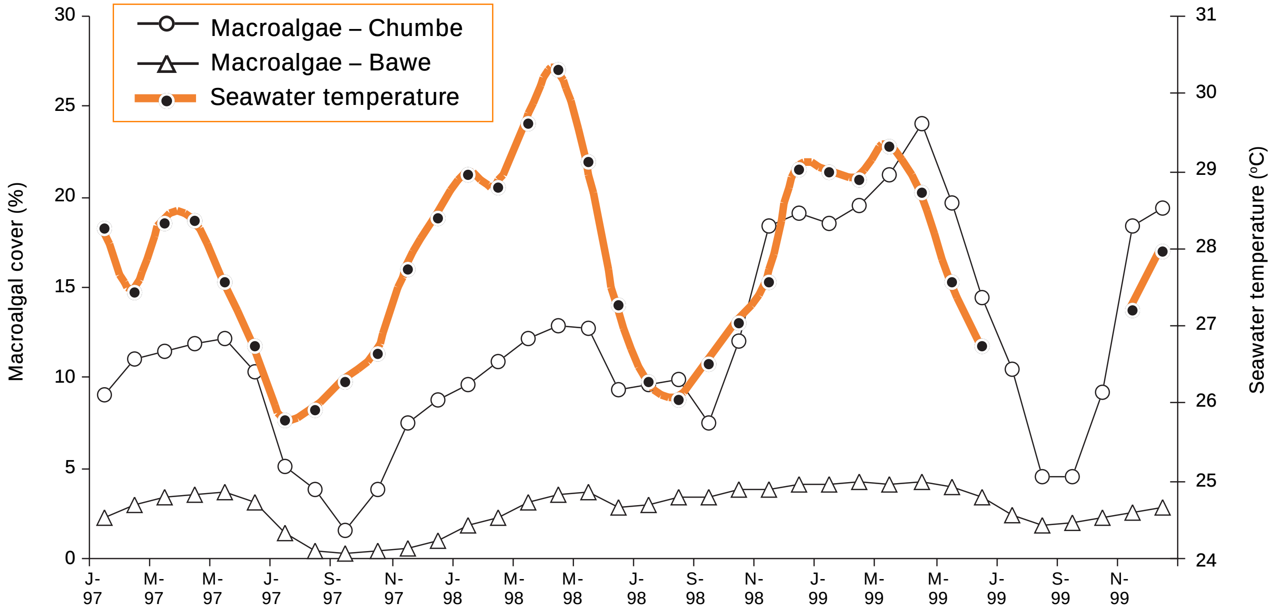

Figure 4. The monthly seawater temperature (ºC) and the macroalgal abundance (% cover) on Chumbe and Bawe reefs from January 1997 to December 1999. There was a significant correlation between seawater temperature and macro-algal abundance. (*Source:* Muhando, in prep.)

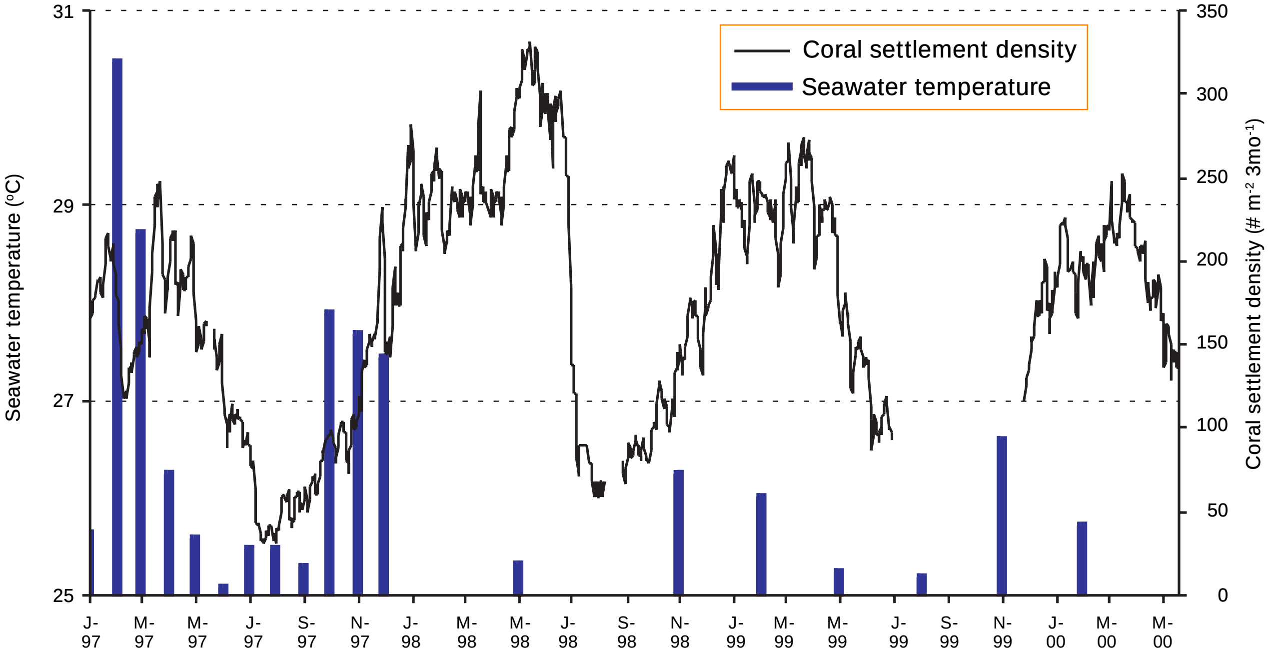

Figure 5. Seawater temperature and coral settlement density on artificial settlement plates incubated for three months (# m^{-2} $3mo^{-1}$) from January 1997 to March 2000. Note that settlement peaks occurred during rising or maximum seawater temperature. (*Source:* Muhando. in prep.)

warm NE monsoon months were previously considered a probable stressful period for growth of coral transplants in the Philippines (Yap & Gomez, 1984) and elsewhere (Meesters *et al.*, 1993).

The yearly intrusion of a cold water-mass in January–February (Fig. 2) played a role in reducing the bleaching and mortality impacts at the study sites (Zanzibar) (Muhando, 2001). Local knowledge of water temperature as well as factors influencing seawater circulation patterns are important for predicting disaster routes. Monitoring of seawater temperatures at various points within marine protected areas could identify areas with high risks or susceptible to high variations in seawater temperature. For example, coral mortality in 1998 was up to 90% on Tutia reef, while it was less than 20% on Chole Bay, about 16 km apart (Mohammed *et al.*, 2000).

The seasonal abundance of macroalgae (dominated by *Sargassum*) on coral reef environments off Zanzibar town correlated positively with the seawater temperature (Fig. 4). Macroalgae appear and start to grow in October (Shunula, 1988), attain their maximum size in April–May and are uprooted or broken between June and August. The high abundance of macroalgae when seawater temperatures are highest is an additional stress to corals, through competition and abrasion. The significant increase in abundance of macroalgae after the bleaching and mortality of corals could be attributed to the availability of additional substrate on dead corals (Done, 1992; Tanner, 1995; McClanahan *et al.*, 2001), and reduced competition from surviving corals (Tanner, 1995).

Although there was continuous settlement of coral larvae throughout the year, there were clear periods of increased settlement (Muhando, in prep.). These settlement peaks occurred during the Northeast (NE) monsoon period, in October–December when the seawater temperature was rising and in February–March when the seawater temperature was near or at summer maximum. Similar seasonal settlement patterns have been reported elsewhere in Tanzania (Franklin *et al.*, 1998; Nzali *et al.*, 1998) as well as in the Great Barrier Reef (Wallace, 1985), Maldives (Clark & Edwards, 1994), and in the Red Sea (Rinkevich & Loya, 1979).

Bleaching of natural coral colonies provides direct evidence that corals are stressed when seawater temperatures are approximately 30˚C or higher. I propose that coral restoration activities, such as by use of coral settlement plates or coral transplantations should probably not be done between February to May as seawater temperature during this period are not ideal for corals. Instead, such activities could start in September through December when seawater temperature are lower and more suitable for coral growth, and macroalgal cover is at its minimum. The management of marine protected area could make use of seawater temperature records and intensify law enforcement activities during warmer periods that are stressful to corals, because recovery and natural restoration of tissue colour (after stress) may be negatively influenced during these times (Yap & Gomez, 1984; Meesters *et al.*, 1994). Most of the broken or injured corals may not recover at higher seawater temperatures (Tunnicliffe, 1981; Hall, 1997).

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Status of Coral Reefs in Kiunga Marine Reserve, Kenya

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Key words: coral bleaching, coral degradation, coral recovery, Kenya, marine protected areas

INTRODUCTION

Monitoring of coral reefs in Kenya focuses on two main parts of the coast, representing the two regions of the coastline dominated by coral reefs. The southern coast of Kenya, from Malindi to the border with Tanzania is dominated by a near-continuous fringing reef, and monitoring focuses on lagoon patch reefs in protected and unprotected sites. This monitoring effort is conducted by the Wildlife Conservation Society (Coral Reef Conservation Project), together with the Kenya Wildlife Service and the Kenya Marine and Fisheries Research Institute (McClanahan, 1992; McClanahan & Obura, 1995), beginning in approximately 1985 and continuing until the present. The northern coast of Kenya contains more discontinuous reefs, in a system of barrier islands, mangrove and seagrass beds, the northern portion of which is contained in a marine reserve (the Kiunga Marine Reserve) that allows fisheries extraction. Coral reef monitoring in this region is coordinated by CORDIO and WWF, also in partnership with KWS, KMFRI, and recently, the Fisheries Department, and extends from 1998 to the present (Obura *et al.*, 1998; Obura & Church, in prep). Components of this montoring effort are also designed to include local fishermen.

This paper gives an overview of patterns observed in the Kiunga Marine Reserve monitoring dataset, compared to the latest reporting of coral cover data from southern Kenya.

METHODS

The two monitoring programmes differ in detail, but both include sampling of benthic cover, coral population information and fish diversity and abundance. The southern monitoring programme uses line intercept transects for benthic cover and coral population studies, and 100 x 5m belt transects for fish sampling, and works on 6 reef areas. The northern monitoring programme uses primarily video transects for benthic cover, $1m^2$ quadrats and timed searches for coral population data and 50 x 5m belt transects for fish sampling, and focusses on 10–12 primary sampling areas. Each programme also monitors a number of other variables not reported here.

Results

Coral cover fell on reefs throughout Kenya as a result of the 1998 El Niño, from varying levels between 15 and 45%, to 5–11% in 1999 (Fig. 1 on next page). This represented losses in cover of between 50–80% on average, with individual sites suffering higher losses approaching 100%. Since 1998, coral cover has generally increased,on the southern coast, and recovery of both protected and unprotected reefs is visible (McClanahan

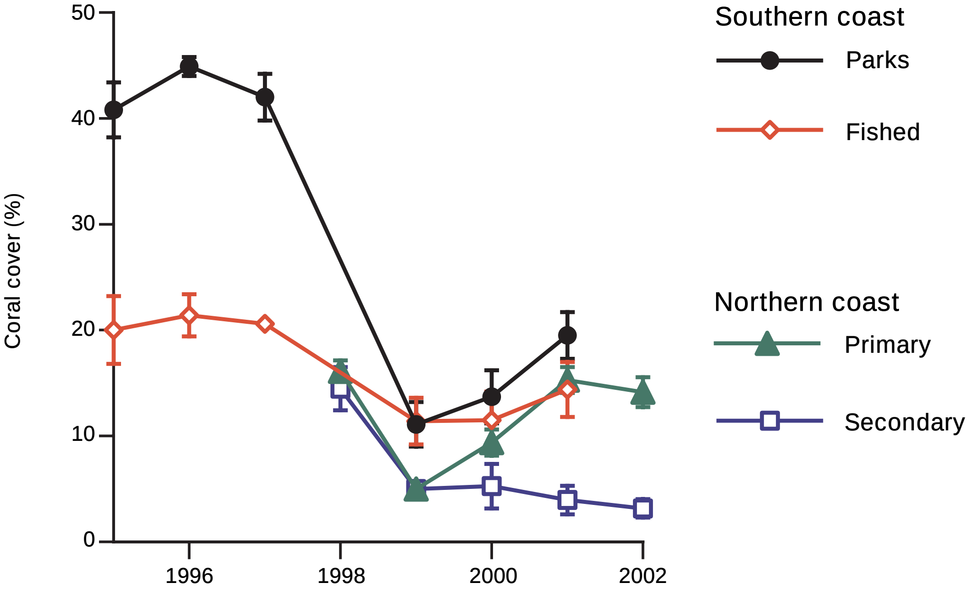

Figure 1. Coral cover from 1995 to 2002 for sites monitored in southern Kenya, and northern Kenya (mean and standard error). Measurements in 1998 made prior to El Niño impact. Southern sites split between protected (inside MPAs) and unprotected sites; data from McClanahan *et al.* 2002. Northern Kenya sites split between those showing positive trajectory after El Niño, and those showing negative trajectories; data from Obura and Church 2002 (in prep).

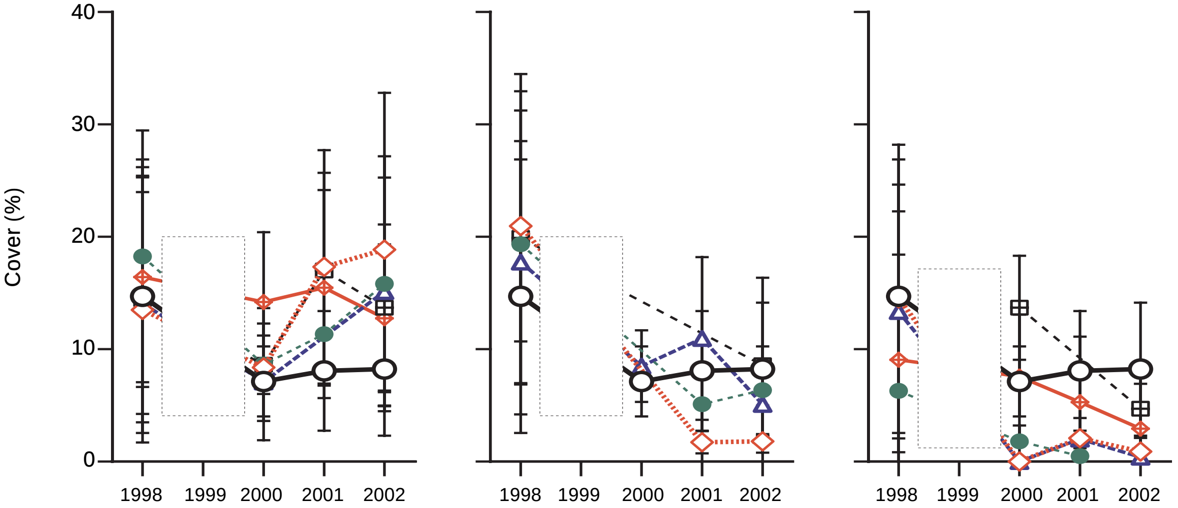

Figure 2. Coral cover before (1998) and after the El Niño at sampling sites in the Kiunga Marine Reserve, grouped into sites with initially high coral cover followed by high recovery (left) and low recovery (middle), and sites with initially low coral cover (right) with no recovery of corals. The heavy black line with open circles represents the overall average, and is the same in each figure and represents the average of all sites shown. The gap between 1998 and 2000 is due to differences in methodology, and poor data collection in 1999.

et al., 2002). In northern Kenya a subset of reefs has shown good recovery to 2001, followed by a fall in 2002 (Obura & Church, in prep), while other reefs have shown small but steady declines from 1999 to 2002, to an average coral cover of <3% in 2002 (Fig. 1). Coral reefs of the Kiunga Marine Reserve have significantly lower coral cover than those in southern Kenya, with average levels of 15% before the El Niño of 1998 (Fig. 1, Obura & Church, in prep), which is less than that of unprotected reefs in southern Kenya.

Following coral bleaching and mortality in 1998, coral reefs in Kiunga also showed a more variable recovery trajectory than did those in southern Kenya. While both unprotected and protected reefs in the south are increasing in coral cover to 2001, more than half of the monitoring sites in Kiunga show a decline (Fig. 1). These can be grouped into three (Fig. 2). The first group comprises previously high-coral cover sites showing good recovery (back to within 80% of pre-bleaching levels, Fig. 2 left). The second group comprises reefs with poor recovery (<50% of pre-bleaching levels, Fig. 2 middle). The third group comprises almost all of the previously low-coral cover sites showing low or even negative recovery of coral cover (Fig. 2, right).

Coral condition was recorded during the El Niño event in 1998 and in subsequent years, the results showing high levels of bleaching and mortality in 1998, and high mortality still visible in 1999 (Fig. 3 left). In 2002 mortality levels increased slightly, correlated with a high incidence of a new coral disease (Fig. 3 left and right) first reported in Kiunga (J. Church, pers. comm.) then in southern Kenya (T. McClanahan, pers.comm.; Wilkinson, 2002). Identified as a fungal disease (Mc-

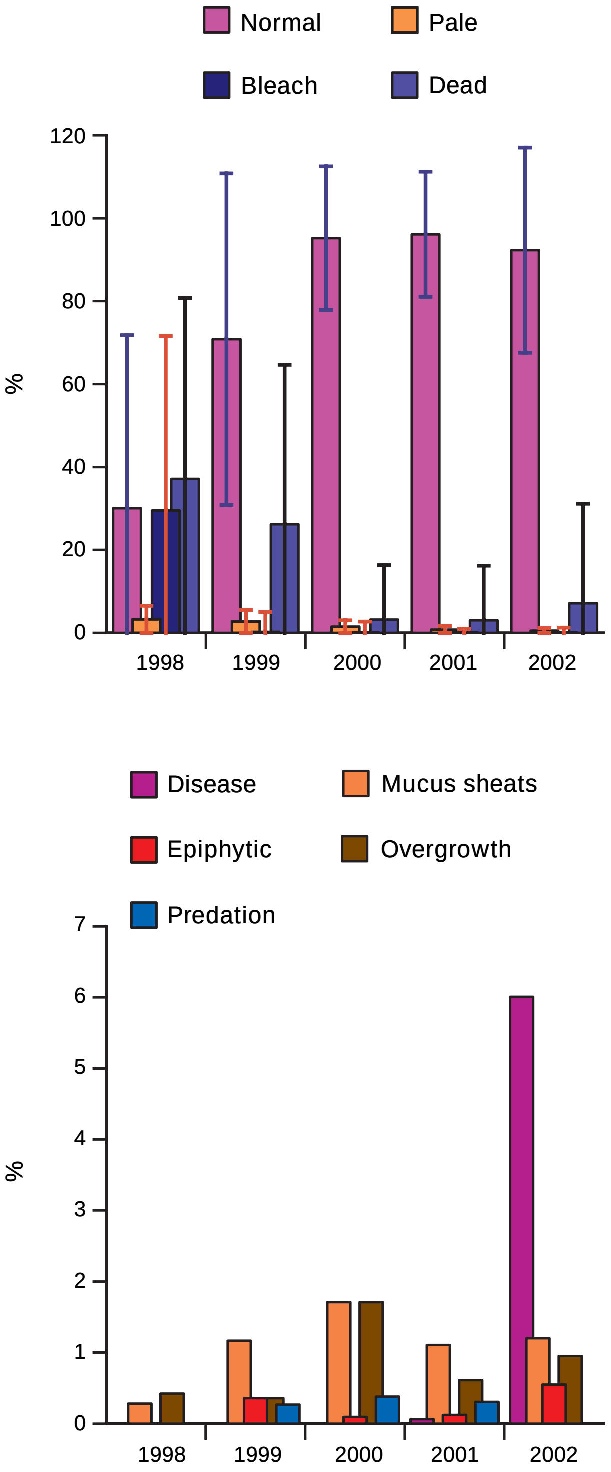

Figure 3. Coral condition recorded in 1m^2 quadrats in the Kiunga Marine Reserve, 1998–2002. Left: percentage of normal, pale, bleached and dead colonies. Right: incidence per 100 colonies of disease, mucus sheaths, various epiphytic conditions (eg. filamentous fungus, pink spots, red rusty deposit, etc.), overgrowth by algae and invertebrates, and predation.

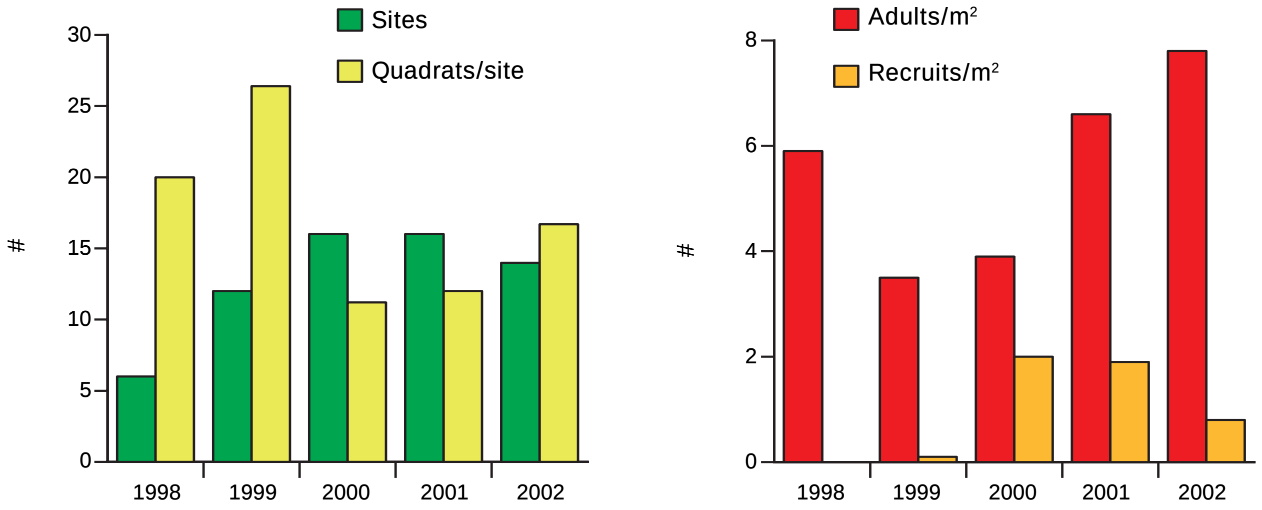

Figure 4. Sampling of coral adults and recruits in 1m^2 quadrats, 1998–2002, Kiunga Marine Reserve. Left: number of sites and average number of quadrats per site sampled. Right: density of adult and recruit colonies, per 1m^2.

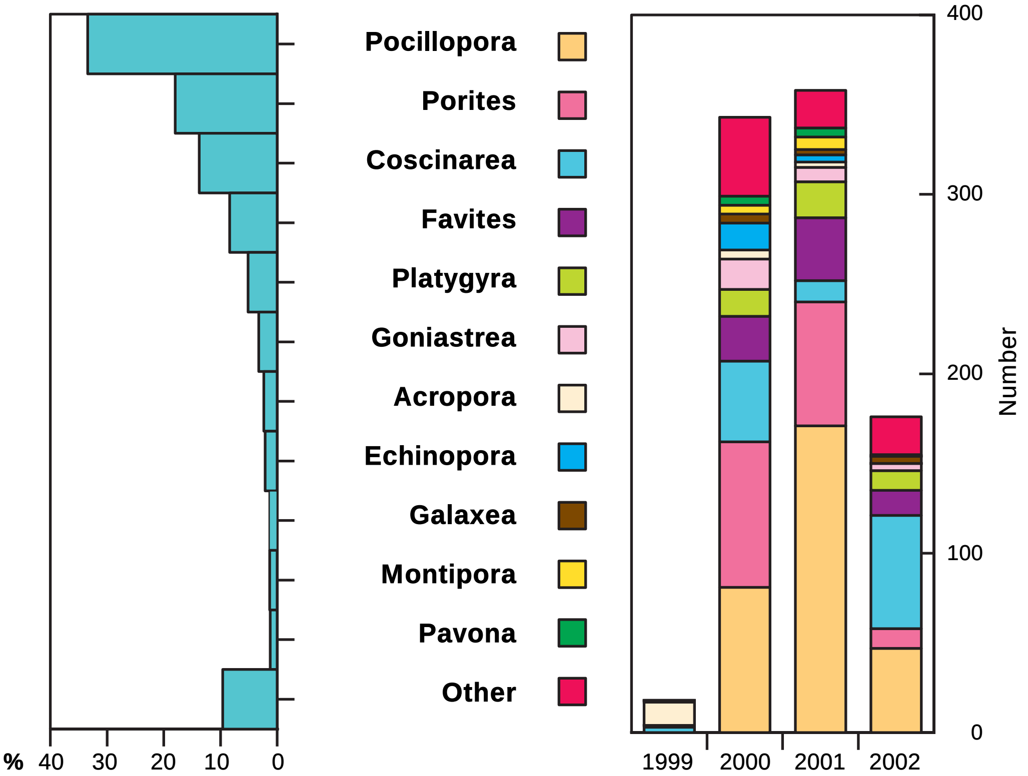

Figure 5. Genus composition of coral recruitment to Kiunga Marine Reserve, 1998–2001. Left: overall proportions by genus, for all genera >1%. Right: composition of genera by year, in numbers of recruits recorded.

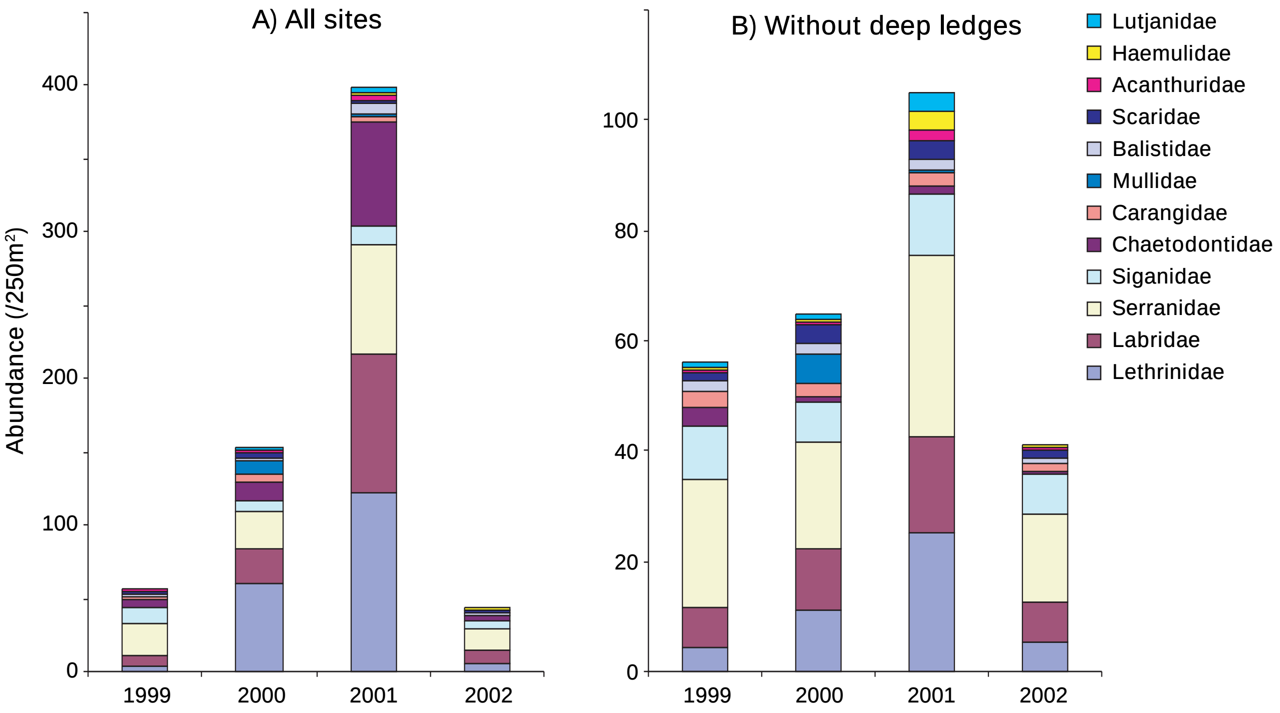

Figure 6. Overall composition of fish fauna for 12 families in the Kiunga Marine Reserve for 1999–2002, in numbers per 250m^2. A) All sites. B) Excluding deep ledges with extreme abundance of schooling fish associated with local topography.

Clanahan, pers.comm.), it affected some of the dominant coral genera at sites with good coral cover, including *Astreopora*, *Echinopora* and *Montipora*, *Pocillopora* and *Acropora*. The disease left a signature of reduced recovery and even a decline in coral cover from 2001 in some high-coral sites that contained populations of the vulnerable genera (Fig. 1, 2 left).

Coral recruitment was monitored from 1998 to 2002 in the Kiunga Marine Reserve (Fig. 4). Recruitment was zero in 1998 and negligible in 1999, with only 13 recruits counted over 300 1m^2 quadrats. Recruitment increased in 2000 and 2001 to average levels of 2 m^{-2}, followed by a strong decline in 2002 to <1 m^{-2}. Recruitment was overwhelmingly dominated by *Pocillopora* (33%), *Porites* (18%) then *Coscinarea* (13%), though the overall recruit population in 2000 and 2001 was highly diverse. Overall, 32 genera and 80 species of corals were recorded in the recruit population, the most common species being *Pocillopora verrucosa* (13.1%), *Pocillopora damicornis* (11.5%) and *Coscinarea mcneilli* (9%), though it was not always possible to identify small recruits reliably.

Fish densities in the Kiunga Marine Reserve vary on average between 50 and 110 individuals per 250m^2 (Fig. 6) at the majority of sites. A peak average density of 400 individuals per 250m^2 in 2001 was caused by massive schools of fish on deep ledges, approaching 3,000 per 250m^2 in a single transect, where local topography attracts schooling fish. Among years, fish density increased to 2001, but this is not significantly higher than in other years. A total of 41 fish families were recorded using local and scientific species names. Snappers (lutjanidae)

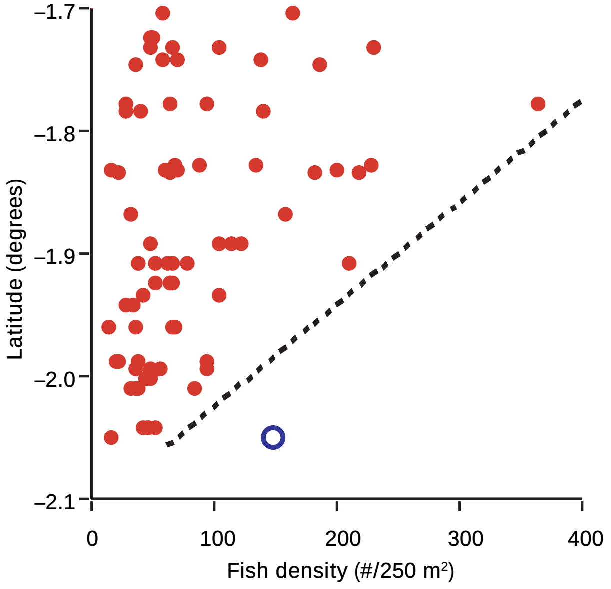

Figure 7. Fish density in the Kiunga Marine Reserve plotted against latitude for all sites (excluding the deep ledges) and years. The line shows the boundary for which the majority of points fall to the upper left. The outlier point (open circles) represents samples from 1999 and 2001 from a single site, Mlango Mkuu.

and sweetlips (haemulidae) were the most abundant fish families in the reserve with average densities >60 per 250m^2, and peak densities estimated at over 5,000 per 250m^2 at the sites with deep ledges. Surgeonfish, parrotfish and triggerfish were the most abundant of the non-schooling fish, with average densities of between 10 and 40 per 250m^2. No trends were observed in the fish populations related to the El Niño.

Fish density in the Kiunga Marine Reserve declines strongly with increasing latitude (Fig. 7). Fish densities at sites in the south of the reserve are constrained to a narrower range of densities, while to the north higher fish densities are found. Two outlier points that weaken, but don't negate, this tendency are from the same site, a large channel where currents are strong and wave energy relatively high. This latitudinal pattern is likely to be due to fishing pressure, which is stronger towards the south where large fishing communities outside the reserve can more easily access the resources in the reserve. Additionally, these communities tend to use beach seines which harvest larger proportions, and smaller size classes, of fish, and are detrimental to overall fish populations.

DISCUSSION

The high impact of the El Niño related bleaching event in 1998 on coral reefs in Kenya has been extensively reported in a number of sources (eg. Obura, 2000a, b; McClanahan *et al.*, 2001). Five years after the bleaching event, it is now possible to recognize the extent of recovery.

Recovery has been patchy, with strong variation within and between reef zones and along the coast. Declines in coral cover from mortality have been matched by increases in algal populations, with successional patterns in the algal communities clear over time (Uku *et al.*, this volume). The dominant pattern that emerges is one of strongest recovery in southern Kenya compared to the north (Fig. 1), and in Marine Protected Areas over unprotected areas (McClanahan *et al.*, 2002). The first difference, of latitudinal variation, is attributed to the more marginal conditions in northern Kenya for coral growth, due to influence of the Somali current system, i.e. cooler waters and higher nutrients (McClanahan, 1988; Samoilys, 1988a, b). Even before the El Niño, reefs in this region were known to be poorer in diversity, have less coral cover and with less accretion of carbonate substrate compared to the south (Samoilys, 1988a, b). Conditions less favourable to reef development appear to have placed a significant constraint on recovery from bleaching. With respect to protection effects on coral reef recovery, it appears that recovery is more robust on reefs under protective management, with higher levels of coral cover and diversity, however, the time to recovery may be longer for these reefs as unprotected reefs may achieve a return to pre-bleaching levels after a shorter time (Fig. 1).

The effect of other threats on the recovery of reefs is demonstrated by the coral disease that hit reefs in Kiunga and other parts of Kenya in early 2002. Data from

Kiunga shows that on reefs with high coral cover and diversity, that had large populations of the vulnerable speciese, recovery of coral cover was reversed. In southern Kenya, a similar effect may be found in reefs that had high abundance of *Astreopora*, *Echinopora* and *Montipora*. For example, a monospecific stand of *Montipora spongodes*, measuring some 30m in diameter that dominated one patch reef in a protected lagoon suffered >99% mortality, with only a few small fragments remaining alive, and depending on transect layouts, may cause over 30% reduction in coral cover at that site. As with recovery from bleaching losses, recovery from smaller impacts such as coral disease is likely to be slower in the north where conditions are less favourable for reefs, as well as in southern Kenya reefs suffering chronic stress from pollution, overfishing or other threats.

Coral recruitment has been at reasonable levels from 2000 to 2002, and on the whole at levels that suggest recovery in the long run may not be limited by larval supply. There do appear to be reproducing coral populations that survived the El Niño, though their output in 1998 and 1999 was significantly reduced during and immediately after the El Niño. Whether a reproduction impact of the coral diseas will be seen as depressed recruitment in 2003 of species affected by the disease will be a specific objective of monitoring in 2003.

The impact of the coral bleaching and mortality of 1998 on fish populations is not indicated in a consistent post-1998 trend in the data from the Kiunga Marine Reserve, though a lack of before data weakens the power of the dataset. Certainly gross changes in fish community structure is not reported for southern Kenya (McClanahan *et al.*, 2001). The Kiunga data does confirm prior findings from East Africa, that fishing and protection levels over-ride other factors that affect fish community structure (McClanahan & Kaunda-Arara, 1996; McClanahan, 1997), with the upper range for fish abundance declining with proximity to major fishing communities at the south of the marine reserve.

Continuation of coral reef monitoring in Kenya is a primary objective of the two programmes monitoring Kenya's reefs, with increasing attention being paid to the interaction of climate change and local threats. Additional effort is being made to understand recovery processes through recruitment and transplantation studies, and through feeding information from monitoring directly to management for more responsive interventions to ongoing and new threats. Extending monitoring to further sites and reefs zones, such as to fore reef sites, and to include non-scientists in monitoring, such as through dive operators and community groups is also being undertaken (eg. Obura *et al.*, this volume). Improving responsiveness of monitoring to threats, such as through rapid response and more frequent focussed sampling following the initial impact will become a greater component of monitoring. Experimental studies investigating different components of bleaching responses are continuing to be undertaken (Kirugara & Wawiye, this volume), to help explain patterns of response to environmental changes.

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Coral Reef Degradation in Mozambique, Results of the Monitoring 1999–2000*

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Key words: coral bleaching, coral recovery, flooding, Mozambique, sedimentation

INTRODUCTION

Mozambique initiated its Coral Reef Management Programme (MCRMP) in 1999 with support from various national and international entities, including CORDIO. Our activities included monitoring, research and training in 2000. This also marked the second year of CORDIO support for research on Mozambique's coral reefs. The activities were undertaken in the context of the recently drafted National Coastal Zone Management Programme (NCZMP), one of its foci being the critical ecosystems such as coral reefs that make up the coastal environment. A management plan for the coral reefs is being formulated within the framework of this Programme and includes, for example capacity building, the collection and synthesis of relevant information and scientific data to support sound management, the development of an appropriate and effective network for the co-ordination of sustainable coral reef use and ongoing monitoring.

This paper describes the activities and presents a summary of results for 2000. Technical support for those activities was provided by the Oceanographic Research Institute (ORI) in Durban, South Africa, with funding from CORDIO and DANIDA.

BACKGROUND

Mozambique has the third longest coastline amongst the countries adjoining the Western Indian Ocean, extending 2,700 km, much of it including coral reefs. The northernmost section of the coast extends for 770 km from the Rovuma River (10º 20' S) in the north to Pebane in the south (17º 20' S). In this section, coral reefs form an almost continuous fringe 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 mangroves. The coastal waters are shallow and this, combined with the sediment loading from the rivers, causes high turbidity. Coral reef formation in this area is consequently severely limited. The southern section stretches for 850 km from Bazaruto Island to Ponta do Ouro (26º 50' S). The coastline is characterized by high dunes, north facing bights and barrier lakes. Reef distribution along the coast and near-shore islands is patchy and the reefs are sparsely inhabited by corals.

The coral reef ecosystem constitutes an important biological resource in terms of its complex biodiversity. It also forms the basis for tropical fisheries and the marine ecotourism industry. Today, about 6.6 million

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*This paper was presented at the Regional ICRI Workshop for the Indian Ocean, held in Maputo, November 2001.

people live within Mozambique's 48 coastal administrative districts and the number is expected to grow at 3% p.a. (INE, 2001). Although this represents 42% of the current population of Mozambique (15.7 million), only 2–3% are fishermen or harvesters of marine resources. 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 focuses on activities such as fishing, SCUBA diving and snorkelling.

CORAL REEF RELATED ACTIVITIES IN MOZAMBIQUE DURING 2000

Monitoring

The sites selected during a preliminary survey in early 1999 (Schleyer *et al.*, 1999) and subjected to monitoring in November 1999 (Motta *et al.*, 1999) were again visited during November and December 2000 over a period of 27 days. These sites are widely distributed along the coast and represent different reef types (shallow fringing reefs, sheltered inshore reefs, reef outcrops and spur-and-groove formations).

METHODS

Surveys were carried out using the GCRMN-recommended methods of recording benthos, invertebrates and fish along the same transects (English *et al.*, 1994). A major modification was to use video transects to sample benthic cover, instead of line intercept transects (LITs). Details on the location of sites and the methods used are provided by Rodrigues *et al.* (2000).

Fish data was analysed according to 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 category) and size classes (total number of individuals in specific size classes for each family).

For the purposes of this report, only the percentage cover of the four main benthic categories encountered in the transects are presented (hard and soft corals, other invertebrates and algae). The fish data are presented as numbers of fish per 500 m^2 for each of the five trophic groups.

RESULTS AND DISCUSSION

The condition of reefs surveyed varied from healthy to heavily impacted by natural and anthropogenic factors. Among these, siltation, cyclone damage and destructive fishing techniques appeared to be the most important.

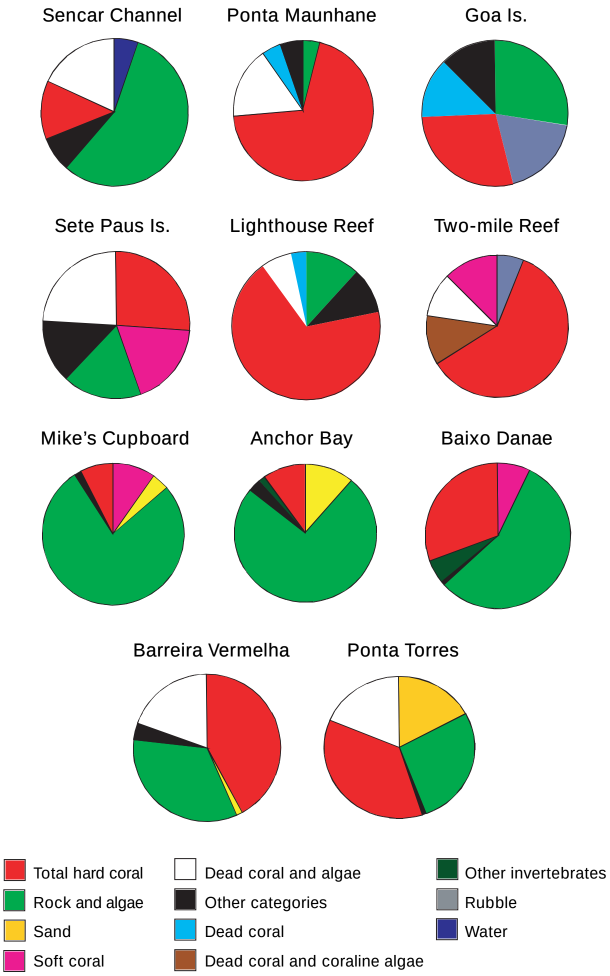


Figure 1. Percentagecover of the main benthic categories of the sitessurveyed.

Reefs that were already degraded by bleaching in 1998 and predation by crown-of-thorns starfish showed slow or no recovery. Coral cover was highest on the inaccessible reefs in northern Mozambique and in marine protected areas (Fig. 1). Fish populations were dominated by carnivores and positively correlated with the coral cover in these areas (Fig. 2). Increased algal cover (Fig. 3) reflected the mortality that was reported at these sites in earlier surveys (Schleyer *et al.*, 1999; Motta *et al.*, 2000). There was evidence of recovery on some reefs with primary colonisation by soft corals. High fishing pressure on some of the reefs was manifested by the small size classes of fish and a predominance of herbivores, some of which are not favoured by fishermen. A general decrease in abundance of fish was best illustrated in Sencar Channel where the fishing pressure is quite high (Fig. 4).

As in the last report, the reefs in the protected areas were in the best condition.

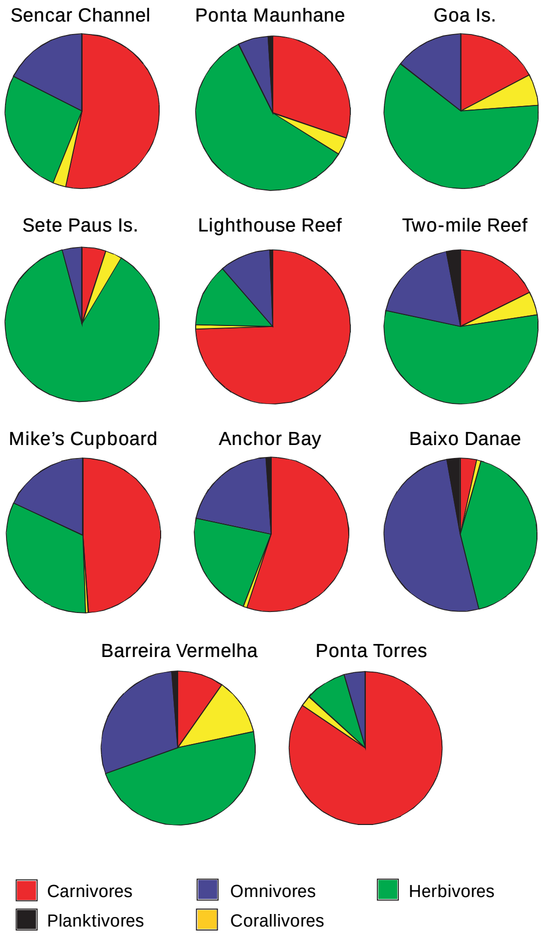
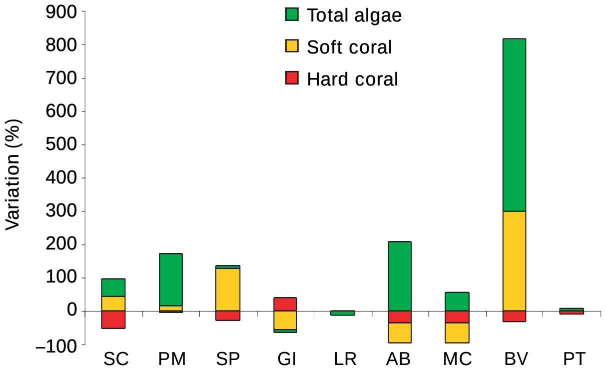


Figure 2. Abundance of the five fish trophic groups considered at each of the sites surveyed.



SC = Sencar Chanell, PM = Ponta Maunhane, SP = Sete Paus Is.,
GI = Goa Is., LR = Lighthouse Reef, AB = Anchor Bay,
MC = Mike's Cupboard, BV = Barreira Vermelha, PT = Ponta Torres

Figure 3. Variation (1999 to 2000) in percentage cover of the major benthic categories at the survey sites.

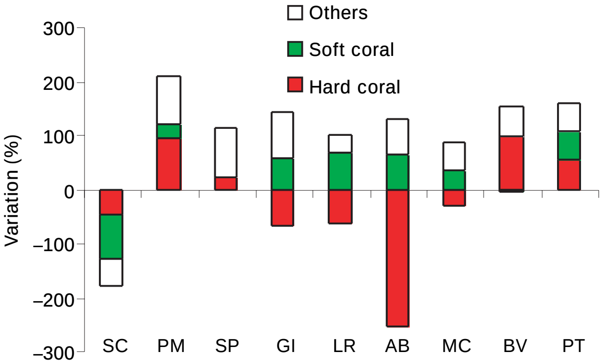


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Figure 4. Variation (1999 to 2000) in abundance of the major trophic categories of fish at the survey sites.

Effects of the Floods in Southern Mozambique in 2000

The southern part of Mozambique suffered the worst floods in 50 years from February to May in 2000 (INAM, 2000). The damage caused by this phenomenon was high with the loss of several hundreds of lives and millions of US dollars in damage.

A survey was carried out in Xai-Xai lagoon to assess the effects of the floods on reef communities, taking advantage of existing pre-flood data (Pereira & Gonçalves, 2000). Benthic composition was studied using the line intercept transect (LIT) technique (English *et al.*, 1994) and the organisms under the line were classified in the appropriate category with the length of each specimen measured to the nearest centimeter. Eight fixed 20 m LITs were completed at each sample time point in January 2000 (before the floods), September 2000 and again in December 2000 (after the floods). The percentage cover of the various categories was calculated from these measurements.

Results of this study are summarized in Fig. 5. The pre-flood hard coral cover decreased from 19.9% to 8.3% after the floods, representing a 58.4% decrease. Virtually all the branching corals (e.g. *Acropora*, *Pocillopora*) died and were covered by algae after the floods. The great majority of *Porites* colonies were bleached, only the small proportion at depths >1 m being unaffected. Massive corals (e.g. *Goniopora*, *Favia* and *Favites*) and encrusting forms (*Echinopora*) comprised most of the residual hard corals. The soft coral community suffered a dramatic decrease of 90.4%, dropping from 4.59% to 0.44% of the reef cover. The majority of *Sinularia* colonies were bleached, with small colonies located at depths >1 m appearing less affected. However, an increase in cover of both hard and soft corals was measured in December, in spite of the high sedimentation, turbidity and low salinity to which they are subjected at the site (Pereira & Gonçalves, 2000).

There was thus considerable post-flood mortality at Xai-Xai with a 60% reduction in coral cover. The factors responsible for this mortality appeared to be a reduction in salinity with an increase in turbidity and sedimentation. Rapid and drastic decreases in salinity cause physiological damage (Kato, 1987) and bleaching in corals (reviewed by Hoegh-Guldberg, 1999). Turbidity and sedimentation negatively affect coral communities through light attenuation, resulting in less light for photosynthesis by the symbiotic algae in the host tissues. Sediments also smother coral colonies, resulting in metabolic expenditure in sediment removal, reduced efficiency in gas exchange, and reduced food capture (McClanahan & Obura, 1996).

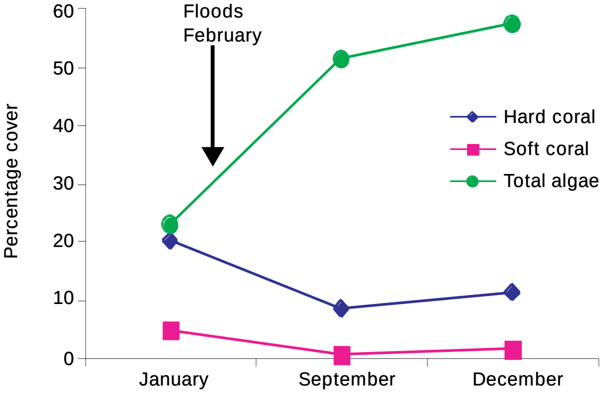

Figure 5. Variation in percentage cover of the main benthic categories at Xai–Xai lagoon before and after the February 2000 floods in southern Mozambique.

The corals' response to these stresses is mucus-sheet formation (Kato, 1987), which assists in sediment clearing and feeding (Barnes & Hughes, 1993). Mucus formation was widely observed in the present study, and the majority of massive (e.g. *Porites*, *Favia*, *Favites*, *Goniopora*) and encrusting (e.g. *Echinopora*) hard coral colonies secreted large amounts of mucus. As pointed out by Veron (1993), all these genera occur in high turbidity areas, suggesting that turbid conditions are not necessarily detrimental to corals and they may obtain nutrition from it (Anthony, 1999). At Inhaca Island, Kalk (1995) and Gonçalves (2000) also reported the predominance of these genera in reef environments subjected to high turbidity and sedimentation. These genera thus appear to have adapted to such stresses.

The deleterious effects of the floods on coral reefs, as illustrated by these results, were probably limited to Maputo Bay, the Greater Maputo Bight and the region between Xai-Xai and Inhaca Island, as there are no major rivers north and south of this region. Damage on the southern Mozambique reefs was probably minimal due to their location.

Training Activities

Coral reef-related training activities during the year 2000 in Mozambique, included:

- Forming and training a reef check team to work on the southern Mozambican reefs. This team consists mainly of undergraduate students from the Biological Sciences Department of Universidade Eduardo Mondlane;
- Graduation of two BSc Honours students in Marine Sciences with a particular interest in coral and reef fish studies;
- Enrolment of two new Honours students in coral reef research activities;
- Enrolment of an MSc student working on the effects of sport diving on reef fish communities.

The Way Ahead: Priority Areas for Action

In the years to come, priority areas for action within the scope of the MCRMP should include not only fieldwork and hands-on training of post-graduated students, but also lobbying for the following measures to increase coral reef and reef community conservation in Mozambique:

- An increase in the total area of MPAs (currently only 0.26% of territorial waters are protected) encompassing coral reefs and associated ecosystems;
- The promulgation of MPAs to protect representative species, environments and systems (including coral reefs, seagrass beds, mangroves, dunes, endangered marine species, etc.);

Support is also needed for the activities of the NPCRM and to build the capacity needed for execution of the work, viz.:

- Training of marine scientists in research on coral reefs and on associated ecosystems;
- Assessment of the biodiversity and other baseline parameters at priority sites;
- The continuation of the current monitoring programmes.

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Trends in the Distribution of Macroalgae in a Bleached Kenyan Reef Ecosystem

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Key words: algal succession, coral bleaching, Kenya, macro algae, marine protected areas

ABSTRACT

In May 1999, two sites were selected within the Mombasa Marine Park and Reserve for the study of macroalgal succession due to bleaching. One of the sites was Starfish, which lies within the Mombasa Marine Park and is protected from fishing and the other site was Ras Iwatine, which lies in the Reserve area and is subjected to fishing activities. The study revealed that the Starfish site had a significantly higher hard coral cover, which did not change over the study period. A comparison of the density of macroalgae revealed that a higher proportion of the substrate in Ras Iwatine was dominated by macroalgae. Settlement tiles, used to study algal recruitment, indicated that the succession process in the Starfish site passes through more stages compared to the Ras Iwatine site. This succession process suggested the influence of grazers in Starfish and their importance in maintaining a lower cover of macroalgae in this site.

INTRODUCTION

Studies along the Kenyan coast have indicated that the coral bleaching event of 1997/98 had a considerable impact on coral environments in terms of the loss of hard coral cover and an increase in the amount of fleshy algae (macroalgae). McClanahan and Mangi (2000) estimated the overall loss of hard coral to be 71% in protected reefs and 44% in unprotected reefs. This decline in hard coral cover coincided with an increase of 115% of fleshy algae in protected areas and a 220% increase of fleshy algae in unprotected areas. This study was undertaken in order to investigate the types of fleshy algal communities that emerged following the bleaching event as well as their succession patterns taking into consideration that degraded reefs are often colonized by different types of algae, which may undergo a succession sequence ending in a climax community that is different from the original community.

In May 1999, two sites within the Mombasa Marine Park and Reserve were selected for this study. One of the sites was Starfish, which lies within the Mombasa Marine Park and is protected from fishing and the other site was Ras Iwatine, which lies in the Reserve area and is subjected to fishing activities. This study formed part of the benthic project and the data collection ended in November 2000.

MATERIALS AND METHODS

At the beginning of the study, in May 1999, three line transects were established in the study areas. However the number of transects was increased to a maximum of 12 line transects in January 2000 in order to improve the quality of the data collected. A tape measure was used to measure the length of the different substrate types along the line transects. These measurements were later converted into percentage cover estimates of the line. Additionally, settlement tiles were used to study algal recruitment. Ceramic bathroom tiles were covered with a mixture of sand and cement in order to provide a rough substrate for the settlement of macroalgae. A total of 45 tiles were set out in each field site and three tiles were collected monthly. In the laboratory the tiles were studied for the percentage cover of macroalgae on them. Thereafter, small sections (2 cm x 2 cm sections) of the tile were scrapped off for weight estimates. The weight of the small area was extrapolated for the entire tile area. The estimation of weight proved to be imprecise due to the fact that cement fragments were often incorporated in the sample. Therefore, percentage cover estimates were used in the analysis of the tile data. The data presented in this report covers the period May 1999 to November 2000. However, sampling was not undertaken in some months, which affected the quality of the data collected. The data collected was transformed using the arcsine transforma-

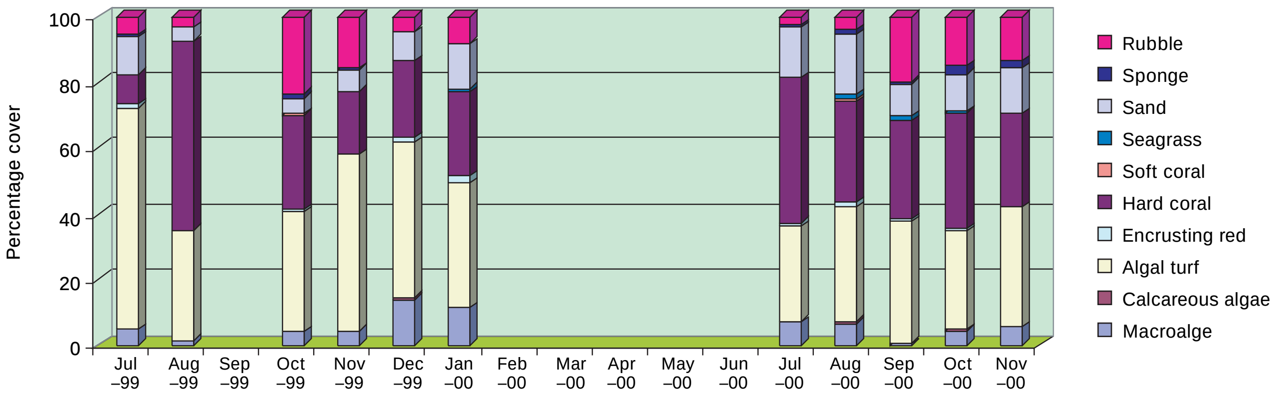

Figure 1. Substrate categories in Starfish.

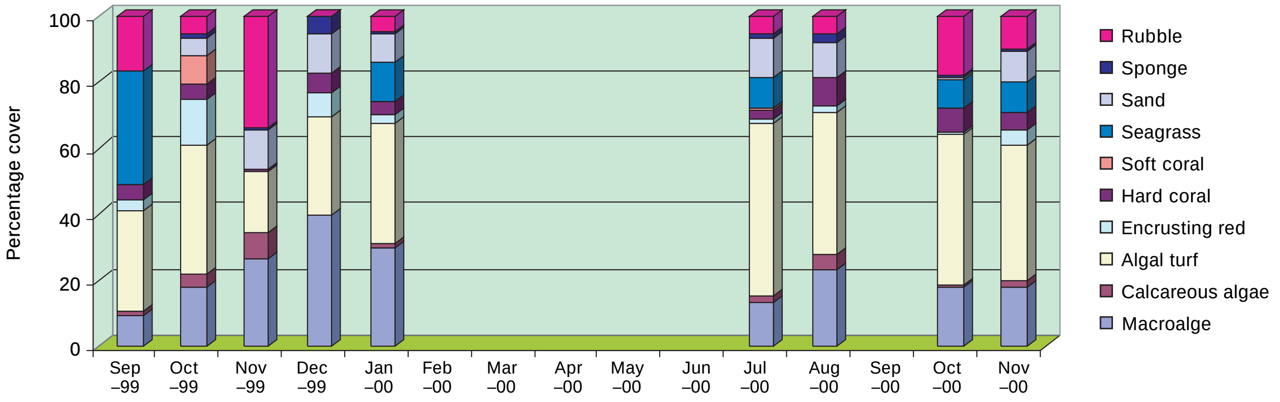

Figure 2. Substrate categories in Ras Iwatine.

tion and subjected to a two-way analysis of variance (ANOVA).

RESULTS

The Substrate Composition

The overall substrate types identified in the study areas were fleshy algae (such as *Sargassum* sp. and *Dictyota* sp.), calcareous algae (such as *Amphiroa* sp. and *Halimeda* sp.), algal turfs comprised of filamentous blue-green and red algae, encrusting red algae which formed crusts on corals, hard coral, soft coral, seagrass, sand, sponges and rubble. In the Starfish site algal turf and hard corals were dominant (Fig. 1). About 30% of the substrate in Starfish was comprised of hard coral while in Ras Iwatine hard coral covered about 5% of the substrate. This difference in the cover of hard coral was significant, however the hard coral cover did not vary significantly over the time period studied (Table 1).

In the sites studied in Ras Iwatine there was a dominance of algal turf on the substrate (Fig. 2). However there was no significant difference in the amount of algal turf when the two sites were compared (Table 1). Macroalgae were found to be more abundant in Ras Iwatine and this difference was found to be significant.

Approximately 20% of the substrate in Ras Iwatine was covered by macroalgae while the cover in Starfish was much lower. The macroalgal group showed variation with time (Table 1). The cover of calcareous algae, encrusting red algae and seagrasses was also found to be significant with a higher cover of these groups in Ras Iwatine.

The Composition of Macroalgae

The macroalgae found in the study sites were divided into upper canopy macroalgae, lower canopy macroalgae and calcareous algae. The upper canopy species were the large brown algae and these were *Sargassum* sp., *Dictyota* sp., *Turbinaria* sp. and *Padina* sp. These macroalgae dominated in the two study sites (Table 2) and their distribution patterns are shown in Fig. 3 (next page) and 4 (page 65). *Sargassum* sp. and *Dictyota* sp. were significantly different when the two sites were compared with higher amounts found in Ras Iwatine. Out of all the macroalgae only *Dictyota* sp. exhibited seasonality with a significant variation over time (Table 2).

The lower canopy macroalgae are shown in Fig. 5 and 6 (page 65). There were more species in this group in

Table 1. Substrate composition in Starfish and Ras Iwatine and summary of the statistical analysis

Substrate Categories	Starfish	Ras Iwatine	Analysis of Variance (ANOVA)			
			sites		time	
			F	P	F	P
Macroalage	5.78 ± 3.91	21.31 ± 8.52	125.85	<0.001	8.49	0.003
Calcareous algae	0.29 ± 0.37	2.53 ± 2.55	7.01	0.024	0.43	NS
Algal Turf	39.91 ± 11.66	36.67 ± 10.04	0.06	NS	0.08	NS
Encrusting Red algae	0.83 ± 0.68	3.86 ± 4.31	5.64	0.04	1.19	NS
Hard Coral	30.03 ± 12.8	4.81 ± 2.20	85.56	<0.001	1.35	NS
Soft Coral	0.11 ± 0.18	1.06 ± 2.66	1.13	NS	2.09	NS
Seagrass	0.34 ± 0.58	8.02 ± 10.94	5.86	0.04	0.88	NS
Sand	10.60 ± 4.68	7.62 ± 4.66	3.50	NS	2.35	NS
Sponge	0.97 ± 0.79	1.41 ± 1.63	0.29	NS	1.48	NS
Rubble	10.27 ± 7.01	11.09 ± 10.38	0.0004	NS	2.22	NS

NS: Not significant. Data presented as mean percentage cover ± sd.

Table 2. The substrate cover of the different macroalgae found in Starfish and Ras Iwatine and the summary of the statistical analysis

Macroalgae	**Starfish**	**Ras Iwatine**	**Analysis of Variance (ANOVA)**			
			sites		**time**	
			F	**P**	**F**	**P**
Upper canopy algae						
Sargassum sp.	1.71 ± 1.19	9.59 ± 3.05	87.07	<0.001	2.17	NS
Dictyota sp.	0.83 ± 0.76	3.57 ± 2.68	39.58	<0.001	4.78	0.02
Turbinaria sp.	0.40 ± 0.33	1.00 ± 1.88	1.55	NS	1.95	NS
Padina sp.	2.51 ± 2.72	6.12 ± 5.00	1.55	NS	1.95	NS
Lower canopy algae						
Hypnea sp.	0.22 ± 0.34	0.33 ± 0.65	0.90	NS	6.53	NS
Dictyospheria sp.	0.01 ± 0.03	0.10 ± 0.12	4.42	NS	0.97	NS
Blue-green algae	0.40 ± 0.33	0.22 ± 0.46	3.51	NS	1.87	NS
Dasya sp.	0	0.02 ± 0.06	1.00	NS	1.00	NS
Codium sp.	0	0.36 ± 0.65	4.21	NS	1.00	NS
Halymenia sp.	0	0.05 ± 0.11	2.25	NS	1.00	NS
Chaetomorpha crassa	0	0.01 ± 0.01	1.00	NS	1.00	NS
Calcareous algae						
Amphiroa sp.	0.05 ± 0.07	1.67 ± 2.37	12.21	0.008	0.96	NS
Jania sp.	0.10 ± 0.23	0.38 ± 0.84	0.94	NS	1.71	NS
Halimeda sp.	0.14 ± 0.31	0.47 ± 0.39	3.89	NS	0.23	NS

NS: Not significant. Data presented as mean percentagae ± sd.

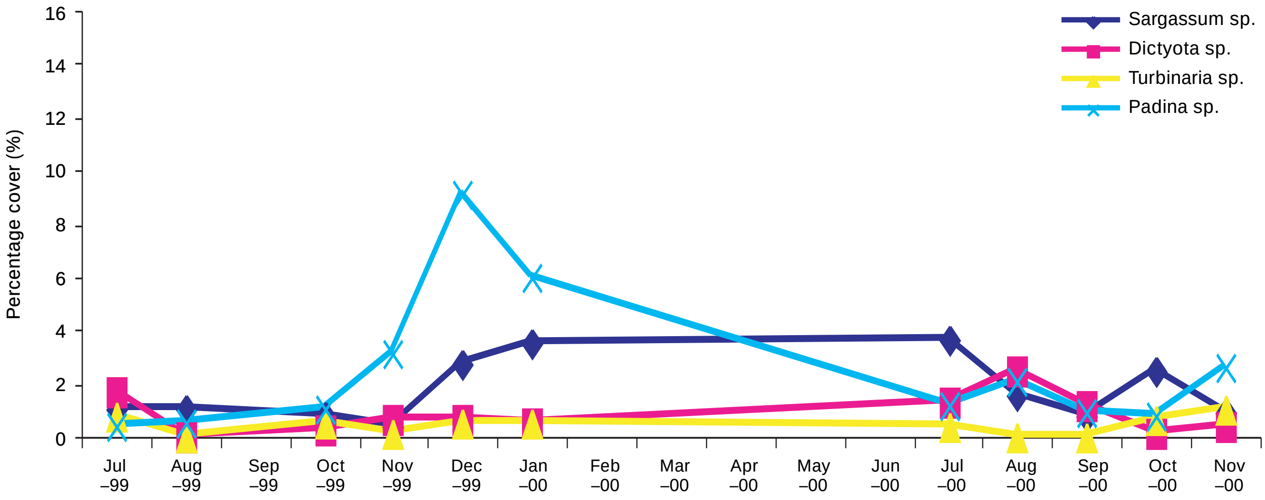

Figure 3. Upper canopy macroalgae in Starfish.

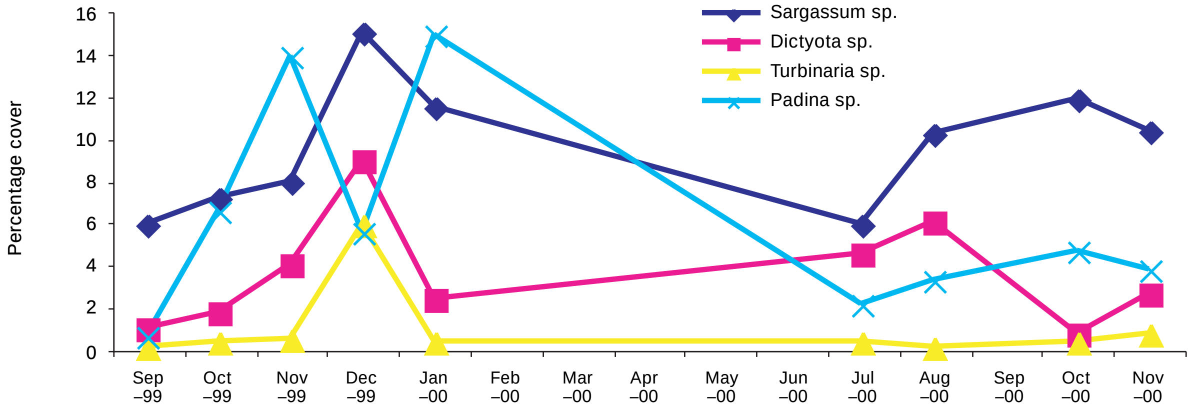

Figure 4. Upper canopy macroalgae in Ras Iwatine.

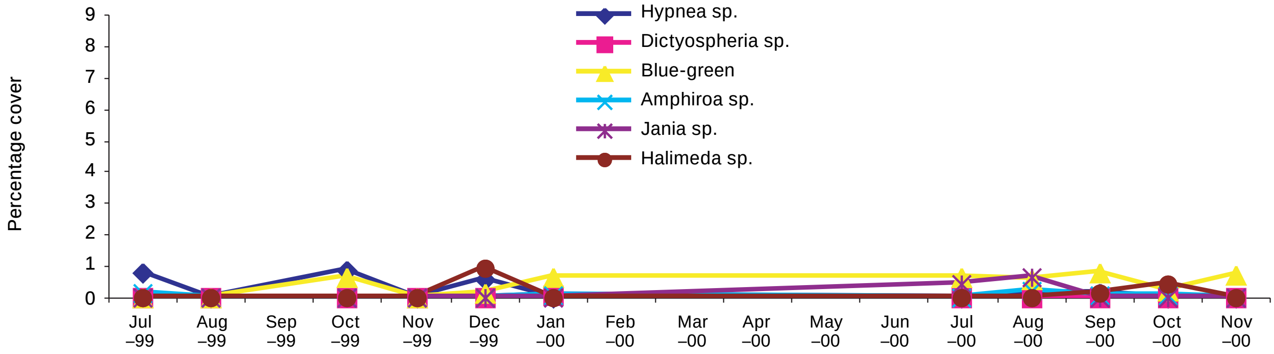

Figure 5. Lower canopy macroalgae and calcareous algae in Starfish.

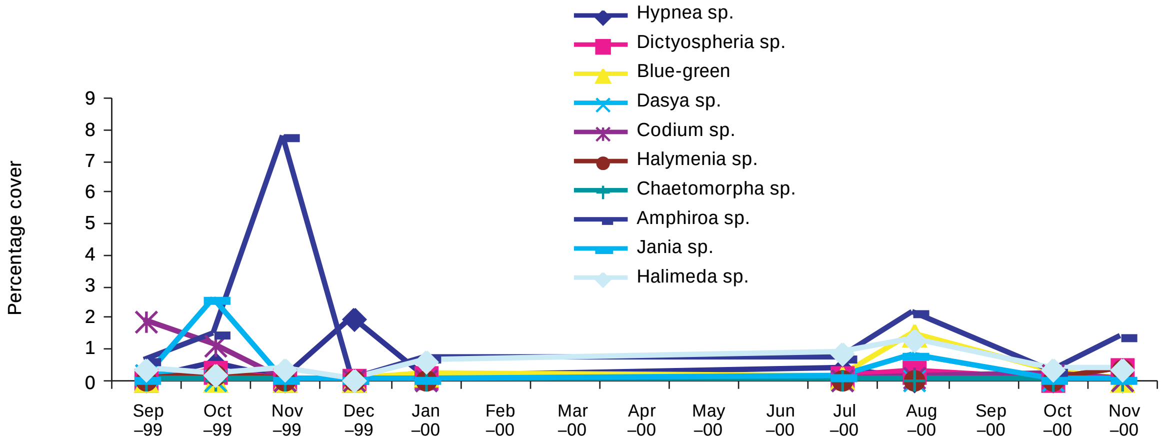

Figure 6. Lower canopy macroalgae and calcareous algae in Ras Iwatine.

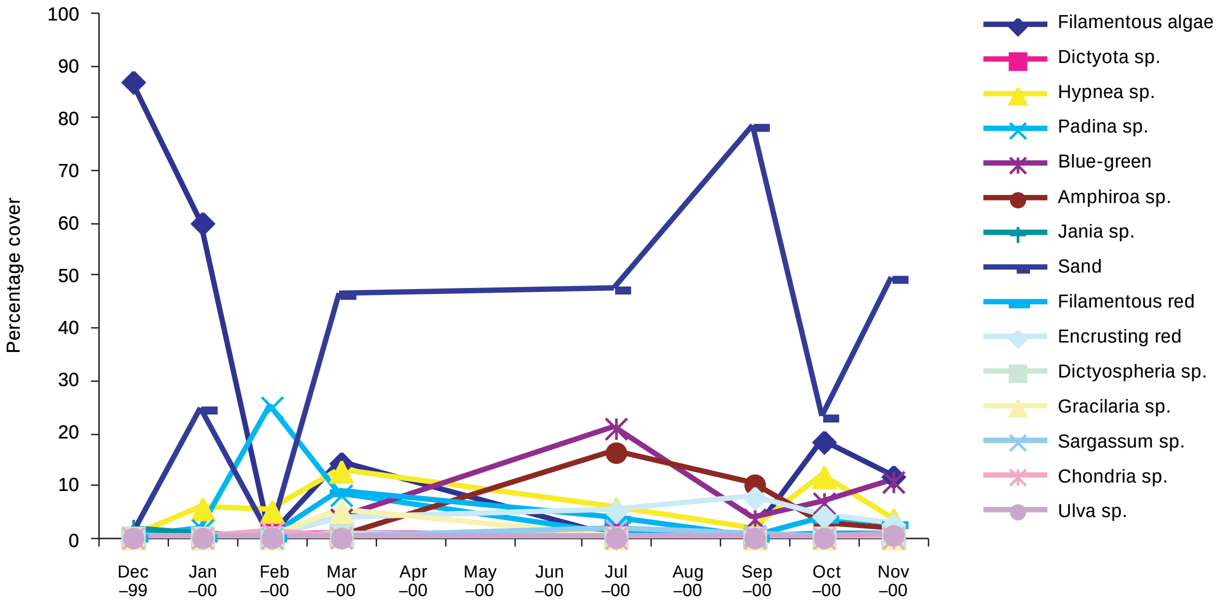

Figure 7. Succession of macroalgae on tiles in Starfish.

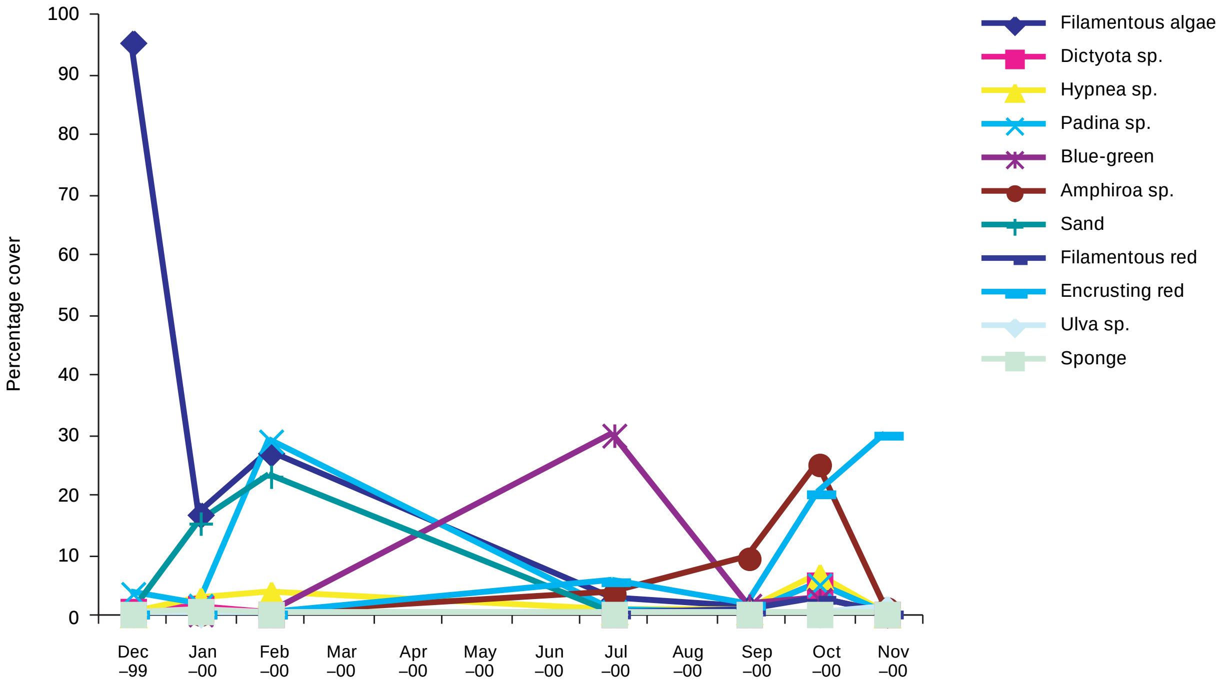

Figure 8. Succession of macroalgae on tiles in Ras Iwatine.

Table 3. The cover of the different macroalgae that colonized the tiles set in Starfish and Ras Iwatine and the summary of the statistical analysis

Macroalgae on tiles	Starfish	Ras Iwatine	Analysis of Variance (ANOVA)			
			sites		time	
			F	P	F	P
Filamentous algae	23.88 ± 32.08	20.63 ± 34.3	0.06	NS	5.82	0.03
Sand	33.38 ± 26.74	5.43 ± 9.51	3.56	NS	0.42	NS
Filamentous red	2.20 ± 3.02	0.40 ± 0.93	1.72	NS	2.20	NS
Blue-green algae	5.61 ± 7.16	4.89 ± 11.12	1.06	NS	7.09	0.02
Encrusting red algae	2.84 ± 2.75	8.17 ± 12.01	1.23	NS	2.95	NS
Hypnea sp.	5.73 ± 4.48	1.90 ± 2.38	11.52	0.015	7.74	0.014
Padina sp.	4.59 ± 8.69	5.59 ± 10.49	1.27	NS	15.62	0.002
Amphiroa sp.	3.70 ± 6.02	5.57 ± 9.23	0.08	NS	3.39	NS
Gracilaria sp.	0.63 ± 1.77	0	1.00	NS	1.00	NS
Jania sp.	0.36 ± 0.61	0	1.00	NS	1.00	NS
Sargassum sp.	0.23 ± 0.47	0	2.24	NS	1.00	NS
Chondria sp.	0.13 ± 0.35	0	1.00	NS	1.00	NS
Dictyota sp.	0.11 ± 0.16	0.96 ± 1.83	1.23	NS	0.58	NS
Ulva sp.	0.06 ± 0.18	0.14 ± 0.38	1.00	NS	31.76	<0.001
Dictyospheria sp.	0.04 ± 0.11	0	1.00	NS	1.00	NS
Sponge	0	0.04 ± 0.11	1.00	NS	1.00	NS

NS: Not significant. Data presented as mean percentagae ± sd.

Ras Iwatine while in Starfish there were fewer speices in this group and their abundance was very low. In the group of calcareous algae only *Amphiroa* sp. was significantly different when the two sites were compared with higher densities in Ras Iwatine (Table 2).

The Composition of Macroalgae on the Settlement Tiles

The tiles were set in the two sites to determine the colonization of newly opened space, which may be the outcome of coral death. The tiles were used to examine what would happen to the coral reefs once the corals die. The settlement patterns on the tiles are shown in Fig. 7 and 8. This part of the study focused on macroalgal colonization and succession.

In general, a more diverse assemblage of macroalgae colonized the tiles in Starfish (Fig. 7). In both sites, colonization was mostly by lower canopy macroalgae though some of the upper canopy species appeared occasionally. At the onset of the colonization process at the Starfish site, there was an abundance of filamentous algae (a mixed species assemblage of turf algae) but this was later covered with sand. Blue-green algae and *Amphiroa* sp. were also present in the later parts of the year and other algae like *Gracilaria* sp., *Dictyospheria carvenosa* and *Chondria* sp. were found periodically on the tiles. In Ras Iwatine, filamentous turf algae were also found at he onset of the tile colonization process (Fig. 7). *Padina* sp. was abundant in the early and later parts of the sampling period. Sand was present however the amount of sand on the tiles in Ras Iwatine was lower than that in Starfish though this difference was not significant (Table 3).

Several macroalgae varied significantly with time and these were the filamentous algae, *Hypnea* sp, *Padina* sp., Blue-green algae and *Ulva*. *Hypnea* sp. was the only algae

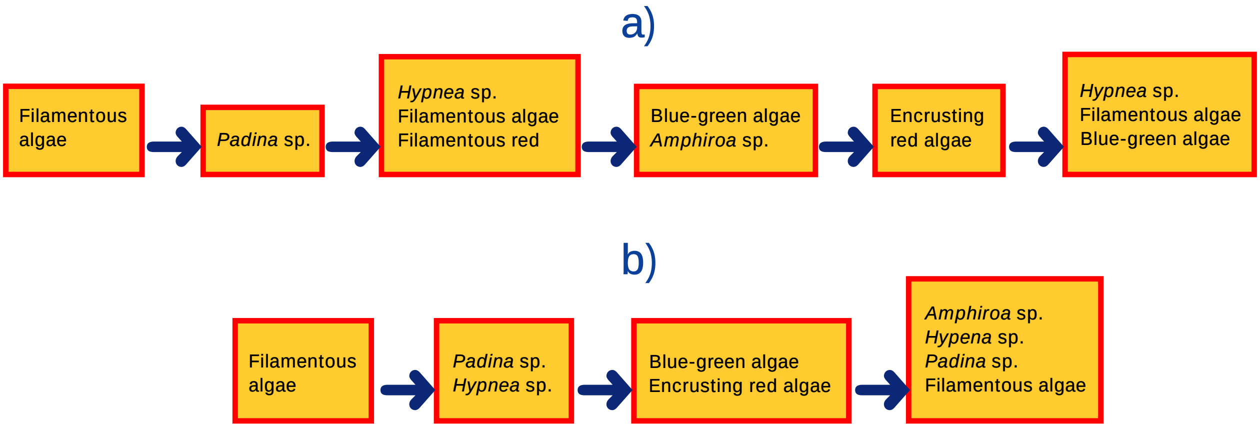

Figure 9. Succession of macroalgae on tiles at a) Starfish, and b) Ras Iwatine.

that showed a significant difference when the sites were compared. A more detailed analysis of the peaks in the succession process is shown (Fig. 9), indicating that the Starfish tiles had more succession stages compared to the tiles established in Ras Iwatine.

DISCUSSION

Previous studies in protected and unprotected reefs have indicated that protected reefs have a high abundance of hard coral, calcareous algae and coralline algae while unprotected reefs have a greater abundance of algal turf, seagrass and soft coral (McClanahan & Obura, 1995). The Starfish site falls within a protected area and the fact that the algal turf is not significantly different from that found in Ras Iwatine indicates that the area has may have experienced changes in substrate composition. The high abundance of algal turf in Ras Iwatine is typical of an unprotected site. Ras Iwatine lies in a marine reserve area where fishing is allowed and this has influenced the substrate composition in the area.

The presence of a significantly high coral cover in the Starfish sites indicates that hard corals were still prevalent during the duration of the study period. There were no significant changes in the hard coral cover during this time, which indicates that the expected decline in hard coral cover did not occur during the study period. Substrate data collected later in July and October 2001 indicated that the cover of hard corals was approximately 40% (Mwachyriea, unpublished data). This estimate was similar to the estimates made during the study period between May 1999 and Novemeber 2000.

The study sites in Ras Iwatine had a high cover of upper canopy macroalgae. This group was comprised of large brown algae and these species have been documented to be dominant in coral reef areas (McClanahan, 1997). *Turbinaria* sp. and *Sargassum* sp. have been found to be the most abundant canopy-forming genera in climax communities (McClanahan, op. cit). In Ras Iwatine, *Sargassum* sp. was significantly more abundant and its dominance in this site together with the other upper canopy species indicates that Ras Iwatine has a climax macroalgal community. The abundance of these upper canopy species is less in Starfish. Additionally, the number of smaller lower canopy species was higher in Ras Iwatine, which indicates the extent of algal dominance in this site. Seasonality patterns were hard to discern in the data set and this may be the result of infrequent sampling during the year.

The tiles were put in the study sites to try and understand the succession process that may occur in a newly opened space caused by coral death. The tiles in Starfish experienced more stages in the succession process. McClanahan (1997) noted that the type of herbivorous fish on a reef influence algal succession where large roaming fish like parrotfish and some surgeonfish allow the succession to pass through more stages. The fact that Starfish lies in a protected site means that there is a higher abundance of fish compared to Ras Iwatine, and this abundance of fish could have an influence on the colonization process.

The foregoing results of this study indicate that during the study period the Starfish site was not transformed into an algal reef and that the presence of herbivores in this site may have contributed to this. This shows that consideration must be given to the types of grazers found at a site, which are here controlled by the level of fishing disturbance and management, in order to understand the trends seen in the distribution of macroalgae.

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Participatory Monitoring of an Artisanal Fishery in Kenya

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Key words: artisanal fishery, Kenya, participatory monitoring, western Indian Ocean

ABSTRACT

Improved and increased monitoring of coral reef status and fishery production is a core need for improving coral reef fisheries management in Kenya. In the past, this has been done primarily by scientists and government officers, who are in limited supply. This paper presents a case study of the expansion of monitoring to artisanal fishermen, through participatory monitoring methods in the Diani region of Kenya, starting in 1998. Fishers are involved in monitoring of landed catch, underwater visual census of fish, and socio-economic variables. The paper presents an analysis of aspects of the data collected, focussing on linkages among the monitoring areas on the issue of gear choice by fishers and implications for management of the fishery. Spearguns are increasingly used by new and poor fishers due to their low cost (US$ 1.33) and ease of use, and probably also because of their marginally higher productivity than alternative gear of similar and even moderate cost (US$ 210). With the banning of spearguns by a Fisheries Department directive in 2001, considerable conflict may ensue as increasing poverty among fishers forces more of them to take up spearguns, despite the illegality of the gear. Additionally, the paper presents the argument that by involving fishers in monitoring, their acceptance of, and empowerment in management is increased to the benefit of all stakeholders, particularly themselves.

BACKGROUND

The Diani-Chale coral reef, 20 km south of Mombasa, supports one of the oldest artisanal fishing communities in Kenya (McClanahan *et al.*, 1997). Studies of the fishing community have included socio-economic analysis of local community livelihoods (Rubens, 1996), fishery trends and impacts on the reef (McClanahan *et al.*, 1997), ownership and power rights (King, 2000) and indigenous knowledge, site use and fisheries trends (Obura, 2001; Obura *et al.*, 2002). The area has suffered declining diversity and productivity due to overfishing (Khamala, 1971; Muthiga & McClanahan, 1987) for over 30 years, and is consistently the 'worst case' reef in a long term dataset, with coral cover of <15% and fish abundance <50 per 250 m^2 (compared to healthy reefs of coral cover >30% and fish abundance >100 per 250 m^2; McClanahan *et al.*, 1996; McClanahan & Arthur, 2001). The area also supports an active tourism industry that competes with fishing for use and access to the reefs, with tourism dominant in the northern, and fishing in the southern part of the area (Fig. 1 on next page). Due to increasing numbers of different users and declining resources, conflict among resource users has been severe, peaking in 1994 following attempts to implement a newly gazetted marine reserve by Kenya Wildlife Service. Opposition from fishermen was due to their fear of losing access to fishing resources due to increased inter-

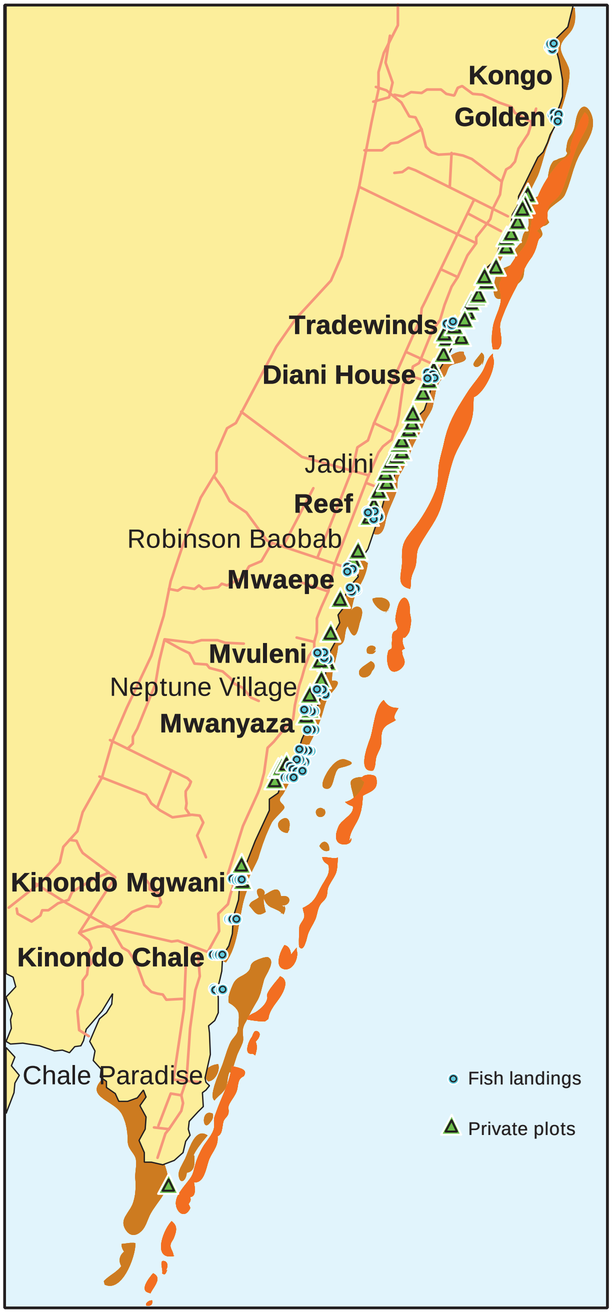

Figure 1. Diani-Chale reef, showing fisheries landing sites, major hotels and the approximate location of the reef crest. Note the dominance of hotels in the northern half, where development has excluded fishing access to the beach.

est and revenue to other stakeholders in tourism. A new coastal area management initiative is taking a holistic approach to management of the area, attempting to balance the interests of the multiple stake holders and achieve resource and environmental management.

As with the remainder of the East African coast, the Diani reef system was impacted by coral bleaching and mortality following the 1998 El Niño event, though recovery of coral cover and diversity to pre-bleaching levels has been rapid and no change in fish communities was recorded (McClanahan *et al.*, 2001). The sudden whitening of corals in 1998 and explanations offered by scientists that this may result in further reef degradation was alarming to fishers in Diani, and reinforced the message that management of the reefs and fishery may be necessary to avoid further declines.

To promote this awareness among fishers of the need for management, and to lay a foundation for their greater involvement in management in the long term, further support was directed towards an indigenous information and participatory monitoring project started in 1997 (Obura, 2001). The project design was altered to directly address issues of data availability and how to obtain it in a manner that is a) useful for management decisions in the future, and b) educational and development-oriented for fishers. Participatory monitoring became the focus of activity and research, extending from catch monitoring and reef mapping to underwater monitoring of fish and benthos and finally to socio-economic monitoring. At a broader level, the project is linked with a number of other participatory monitoring activities in East Africa to promote local level monitoring of coral reefs for conservation and management (Obura *et al.*, 2002).

This paper presents broad results in three areas of participatory monitoring: landed catch, underwater visual census of fish, and socio-economic monitoring. The paper will address the issue of gear productivity and selection, and how socio-economic considerations should be included in fisheries management alongside traditional resource information. The paper will also show that information collected by, and structured by the knowledge of, fishers, can be assessed quantitatively for management purposes and scientific analysis, and also contribute to the growing need for sound monitoring data on coral reefs and resources.

PARTICIPATORY CATCH MONITORING

Methods

Data collection at each landing site is conducted by young fishermen in a book kept by a local fishermen's group. The methodology was designed to be consistent with the fishermen's knowledge and practices, and with data needs for catch monitoring: name of the captain, number of additional crew, fishing gear used, fishing sites visited and catch weight (kg) and composition (number of fish per taxon). The dominant local language, Swahili, was used for data collection, and local names were used for gear (Table 1), sites and fish. The study has been conducted at four fish landing sites on the southern part of the study area – Mwaepe, Mvuleni, Mwanyaza and Kinondo Chale (Fig. 1).

Fish abundance, diversity, and size were recorded in 50 x 5 m belt transects by teams of four fishers. Two of the observers counted fish, 1 counted invertebrates and 1 layed the transect line. The transect-layer swam in the middle, with the two fish observers on either side separated by a spacer-line. This method was designed to ensure a maximum transect width of 5 m, with the observers only recording fish directly ahead and in towards the space between them. The method does, how-

Table 1. Fishing methods used in diani

Gear type	Type	Vessel	Description
Basket trap (malema)	Traditional, passive	Dugout canoe	Wood strips woven in hexagonal patterns. Left overnight with mixed bait and, checked on a daily basis. The trap usually reset in the same place or nearby.
Handline (mshipi)	Traditional, passive	Dugout canoe	Hook and line, now using steel hooks and nylon monofilament.
Hand spear (Mkuki)	Traditional, active	Walking, swimming	Wooden stick sometimes with a steel tip, targeting octopus, rays and some fish.
Gillnets (Nyavu, various sizes)	Traditional, passive	Dugout canoes, outrigger sail canoes	Now made of nylon, with legal minimum hole size of 5 cm. Set as in lines or rings, operated by 3 to 9 fishers depending on target species, net size and deployment method.
Bottom-set net (Jarife)	Traditional, passive	Dugout & outrigger sail canoes, sail boats	Deep gill nets with large mesh size, targeting large fish and sharks. Significant turtle bycatch.
Speargun (Bunduki or Mdeti)	modern, active	Swimming	Locally made using wooden or metal tube shafts, with steel harpoon powered by rubber/inner tube strips.
Seine net: pull- or beach-seine (Juya, nyavu ya kukokota)	Modern, active	Dugout & outrigger sail canoes, sail boats	Large robust nets often with fine mesh sizes of 2 cm and less, with fine mesh cod end. Dragged in a semicircle to trap fish, operated by 10–20 fishers.

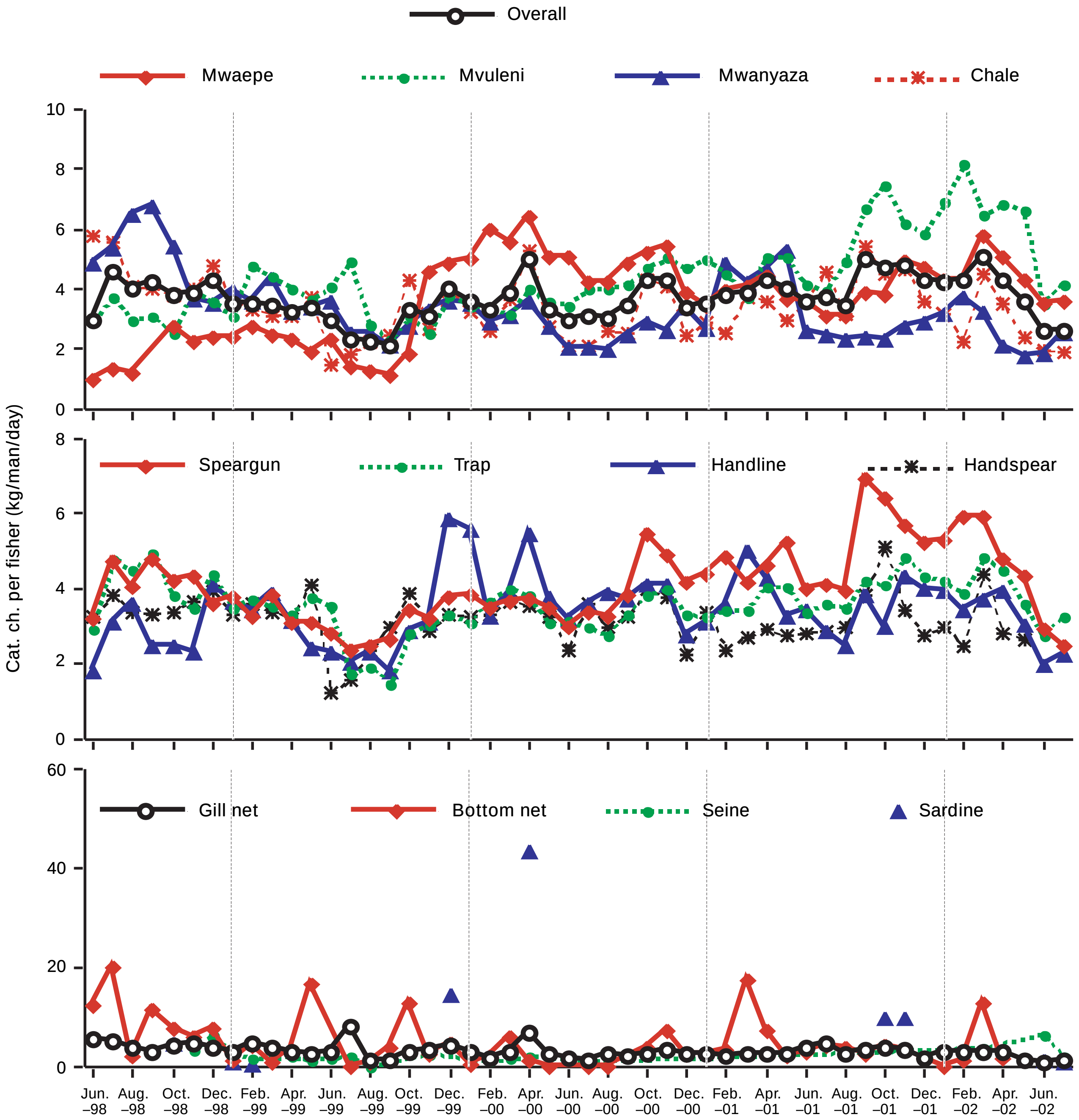

Figure 2. Catch per unit effort (kg/fisher/day) data from four landing sites in Diani, Kenya (Mwaepe, Mvuleni, Mwanyaza and Chale). Catch for each landing site and the overall average (top). Catch by gear type for all landing sites combined: spearguns, traps, handlines and handspears (middle); gill nets, bottom set nets, beach seines and sardine nets (bottom). Vertical lies show the start of each year.

ever, suffer from the presence of three people advancing in the water at one time, and the effect this may have on fish behaviour. Fishermen used local names and four size categories (1, 2, 3 or more 'hands' long, corresponding approximately to <20, <30, <40 or >40cm). Three transects were recorded at each site, with sampling conducted seasonally (twice a year) during 2001 and 2002.

Baseline socio-economic surveys were conducted in 2002 through informal discussions with random individual fishers and group discussions with a total of 248 fishers. Information was collected on reef use, fishermen's perception of reef condition. Building on this, a protocol for socio-economic monitoring, using members of the fishing communities as primary data collectors is under development (Malleret-King *et al.*, 2002). The methodology emphasizes participatory techniques and includes modules for documenting resource use patterns, occupational structure and attitudes related to conflict among users, with households in a fishing village as the primary unit of measurement. Pilot testing is being carried out in Diani and Kiunga (Kenya), and Misali Island (Pemba, Tanzania). The method is being developed to be sensitive to, and provide for, local information needs (e.g. for an MPA/community based project) while providing data that can be compared at the regional level.

RESULTS

Participatory Catch Monitoring

Between June 1998 and July 2002, 24,705 catch records were collected. Catch per unit effort in the Diani fishery varies between 2 and 6 kg/fisher/day, with high variation through the year (Fig. 2). In general, catch rates are highest during the northeast monsoon (November–April) than the rougher southeast monsoon (May–October), however this varies among gears and years. Analysis of catch trends over time and season show an overall increase in catch from 1998 to 2002 (Table 2). However, while the increase over time is significant (p<0.05), the

Table 2. Analysis of variance for CPF for each landing site and gear type, by year and season. The table gives the direction of trend (+ve or –ve) with successive years and for NEM to SEM, and the significance level of the F value.

	Model		**Year**		**Season**		**Year*Season**	
	direction	**sign**	**direction**	**sign**	**direction**	**sign**	**direction**	**sign**
Site								
Overall	+	+	+	*	–	ns	–	ns
Mwaepe	+	***	+	***	–	***	+	ns
Mvuleni	+	***	+	***	–	ns	–	ns
Mwanyaza	–	**	–	**	–	ns	–	*
Chale	–	ns	–	ns	–	ns	–	ns
Gears								
Individual								
Bunduki	+	***	+	***	+	ns	–	ns
Malema	+	ns	+	ns	–	ns	–	ns
Mshipi	+	**	+	ns	–	**	+	ns
Mkuki	–	ns	–	ns	+	ns	+	ns
Group								
Nyavu	–	ns	–	*	–	ns	+	ns
Jarife	+	ns	n/a	n/a	n/a	n/a	n/a	n/a
Juya	+	ns	n/a	n/a	n/a	n/a	n/a	n/a

ns: not significant; n/a: test not applicable due to insufficient degrees of freedom; *p<0.05; **p<0.01; ***p<0.001

overall ANOVA model is not significant. Between-season differences are not significant for all sites and gears combined. Catch at two landing sites (Mwaepe, Mvuleni) is increasing significantly over time (p<0.001) and decreasing at the other two, while seasonal differences are consistent though non-significant, with catch being higher in the northeast than southeast monsoons.

Bottom-set nets have the highest per capita catch rates, with levels of over 5.5 kg/fisher/day, significantly higher than all other gears, which have catch rates of 3–4 kg/fisher/day (Fig. 3). Lowest per capita catch rates occur for beach seine nets, at <3 kg/fisher/day. Over the four years, some gear show increasing catch trends, while others show decreasing trends (Fig. 2, Table 2). Gears that can be operated by a single fisher (speargun, trap, handline) show increasing overall catch, though this is only significant for spearguns. Handspears and multi-fisher gears (nets) show a declining trend. Spearguns show an increase in catch during the southeast monsoon, unlike the other primary gears, perhaps due to their lower vulnerability to rough conditions as the fisher is swimming, not using the relatively unstable dugout canoes. Handspears are used for catching octopus while walking or swimming on the reef, and are often used as a secondary gear when others are non-productive, hence may be used during the SEM when preferred gear cannot be used.

Recording of the primary fishing sites in the dataset allows mapping of all of the variables, and analysis of the degree of spatial overlap of gear use. Handlines, spearguns and basket traps show strong differences in site use (Fig. 4), shown by the separation of rays and points in the scatterplot, and dominance of the gear in the column graphs. By contrast, gill nets, bottom nets and handspears show less spatial separation amongst each other in cluster 4, and even where they are used, traps predominate. In general, basket traps and spearguns are used in similar reef zones (lagoons and back-reef coral communities), illustrated by their dominance of the left side of the scatterplot, though within these zones the

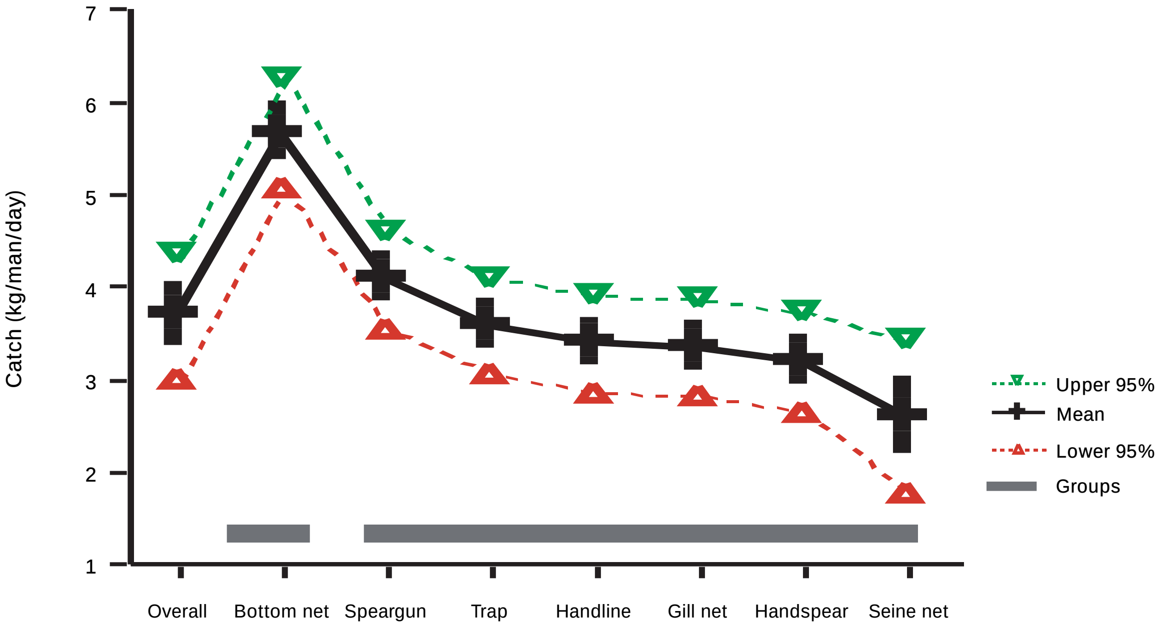

Figure 3. Comparison of catch per unit effort based on gear, Diani. Mean, standard error and upper and lower 95% confidence limits shown. Horizontal grey bars show significantly different groups (Tukey HSD test).

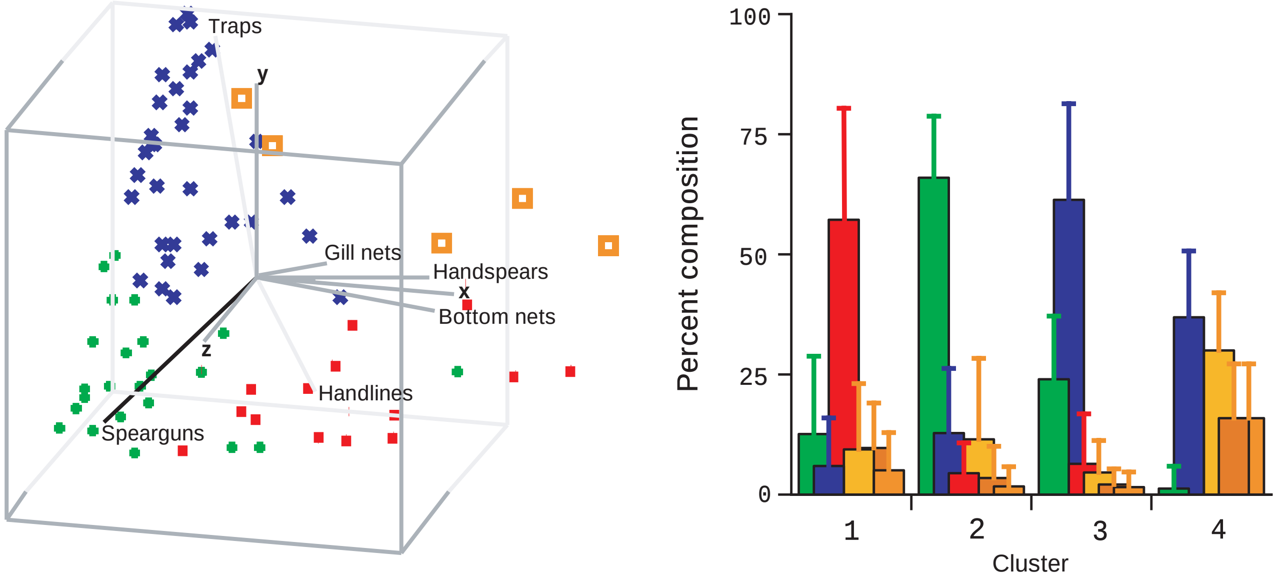

Figure 4. Gear use by fishing site, Diani, Kenya. Left – three dimensional scatterplot of first 3 Principal Components, of the percent composition of gear used at 91 sites in Diani. Rays show the location of gear types and points show the location of individual site records. Four clusters were identified by cluster analysis, with the percent composition of 6 gear types in each cluster shown on the right (mean and standard deviation). Numbers in the legend indicate the cluster to which each gear most strongly contributes.

data suggests there is some spatial segregation of use. Handlines are used in reef channels and on the fore reef, as are gill and bottom nets, hence their positioning to the right of the plot.

Fishing sites show distinctly different visitation rates and use by fishers. The maximum visitation rate recorded was just under 4,000 visits (in ≈ 1000 days of recording over 4 years), with a mean and median of 127 and 15 visits per site, respectively. Eight four out of 192 sites had less than 10 visits over the 4 years. Preferred sites are not associated with the highest per capita catch, indeed there is a slight negative correlation between the number of visits and the productivity of a site (Fig. 5 on next page, lower left). The most highly visited sites are those with average catch rates.

Underwater Fish Census

Fish abundance recorded in 2001 was higher than that recorded in 2002, though with no consistent difference among seasons (Fig. 6 on next page). While the decrease from 2001 to 2002 was significant (Table 3 on next page), the amount of variation explained by the ANOVA was very low ($r^2 < 0.05$). Indeed, the high point for 2001 was due to high abundance off glass-eye sweepers (Pempheridae), which are small-bodied and unimportant for the fishery (though occasionally fished), in a single transect at one site, Mtengo (Fig. 7 on next page). Notably, fish populations are extremely low in Diani (20–22 per 250 m^2 in 2002) compared to other parts of Kenya (Obura & Church, 2002), for example Kiunga in the north, where fish densities average 197±59.7 per 250 m^2 (m±se), and vary from a minimum of 15–30 per

Table 3. ANOVA of fish densities (#/250 m^2) by year (2001 and 2002) and season (northeast and southeast monsoon), Diani, Kenya. ANOVA r^2 = 0.04, n=136.

Source	DF	Sum of Squares	F Ratio	Prob >F
Year	1	10271.266	4.1879	0.0427
Season	1	5391.002	2.1981	0.1406
Year*Season	1	2678.797	1.0922	0.2979

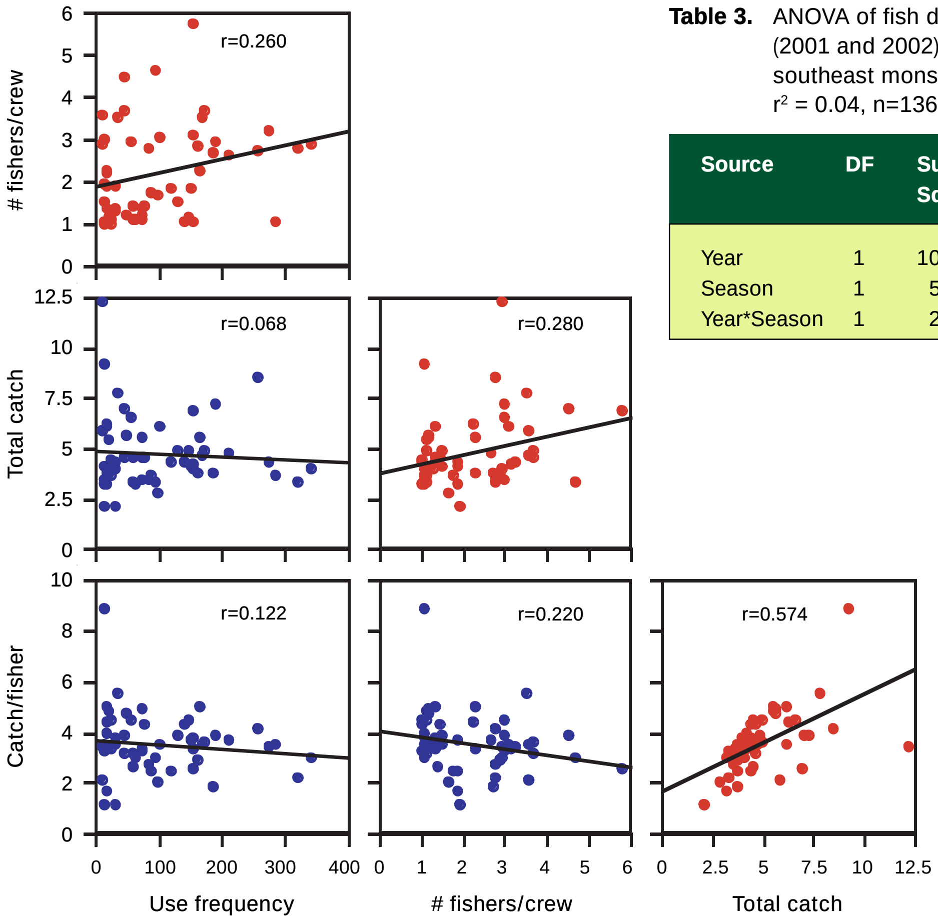

Figure 5. Multiple correlations between catch variables associated with individual sites – site use frequency, # fishers per crew, total catch and catch per fisher. Diagonal plots (red) show positive correlations, the others (blue) show negative correlations

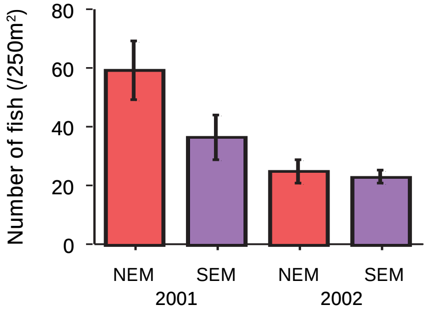

Figure 6. Fish density (per 250 m2) recorded by fishermen in the southeast and northeast monsoons (SEM, NEM respectively) of 2001 and 2002.

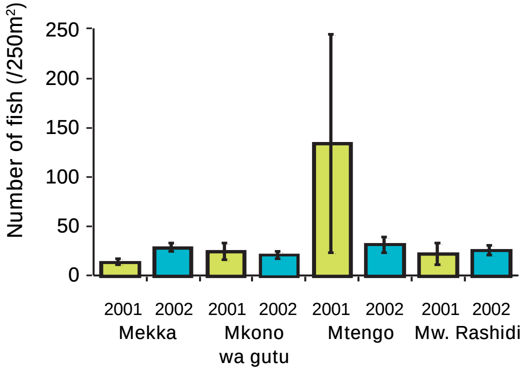

Figure 7. Fish density (per 250 m^2) recorded by fishermen at four selected sites in 2001 and 2002.

250 m^2 at highly fished sites (similar to Diani), to 3,250 per 250 m^2.

The inverse relationship between fishing effort and catch determined by site productivity (Fig. 5) is also shown between catch levels and censused fish populations (Fig. 8). Sites with higher fish densities yield lower catches, as shown by the linear regression line (though this has a low r^2), as well as by the general shape of the data, which is restricted to the space below the dotted curved line.

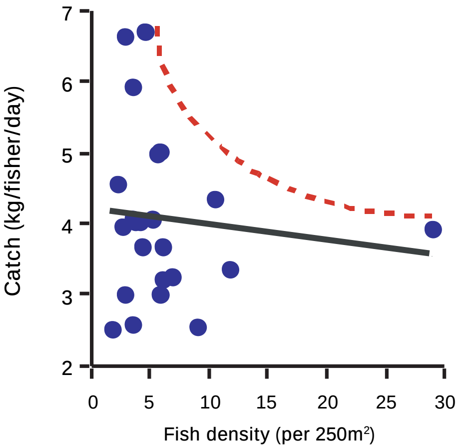

Figure 8. Relationship between catch and fish density, Diani. Linear regression line shown by solid black line ($y = -0.022x + 4.184$ $r^2 = 0.011$). Upper boundary of data points shown by dotted curved line.

Socio-Economic Surveys

At Chale landing site the majority of fishers were trap and speargun fishers (Fig. 9, left), working independently, or for some trap fishers, in pairs. However, 20 of the 47 fishers interviewed would prefer to use bottom-set nets (*jarife*, with large mesh sizes of >10cm for large fish and sharks) over their current gear, even though none of the respondents were actually using these nets. Thirty four out of 48 fishers gave limited financial resources as the primary reason for not using their preferred fishing gear. Gear use also varies with experience of fishers (Fig. 9, right), with spearguns predominant among younger fishers, being replaced by traps and nets for medium-experience fishers, and gillnets, handlines and hand-spears becoming more common among older fishers.

The absence of high cost fishing gear in the study area demonstrates the strong restriction placed by cost on gear use (Table 4 on next page). The most expensive gear used by fishers was KSh 36,000 (or U$ 480) for bottom-set nets, with most fishers investing about Ksh 16,000 (U$ 210) for use of traps, or KSh 100 (U$ 1.33) for spearguns. There are no fishers at the study sites using motorized boats and or larger 'dhows' for offshore fishing.

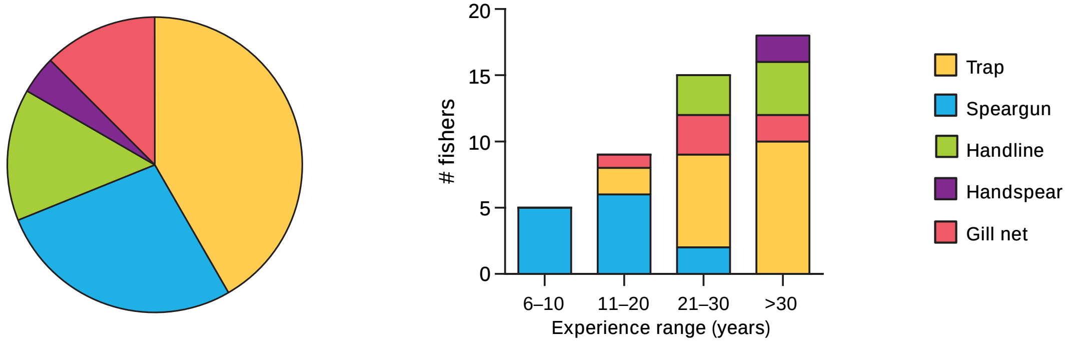

Figure 9. Gear use by fishers in Diani. Left – proportion of fishers using principal gear types. Right – distribution of gear types by experience of fishers (in years).

Table 4. Cost of different gear and vessel combinations. Exchange rate, approx. 75 KSh to U$ 1.00.

Gear	Vessel	Estimated cost (KShs)	Used in Diani
Bottom set net	Outboard powered	508,000	No
Bottom set net	Sailboat	220,000	No
Beach seine (large)	Canoe (2)	100,000	Yes
Beach seine (small)	Canoe (1)	50,000	Yes
Bottom set net	Outrigger sail	36,000	Yes
Gillnet	Canoe	23,000	Yes
Tramelnet	Canoe	23,000	No
Handline	Outrigger sail	21,000	Yes
Fixed line	Canoe	18,000	No
Trap	Canoe	16,000	Yes
Beach stick trap	–	10,000	No
Speargun	–	100	Yes
Handspear	–	100	yes

DISCUSSION

Fishery Trends

All of the data discussed above, save for the baseline socio-economic data, were collected by fishers. The data clearly show a variety of trends that vary by site, gear type and time, providing useful information for assessment of the fishery. The fishery shows a slight increasing trend over four years, which is corroborated by independent data, following a decline up to 1997 (McClanahan, unpublished data). This trend is contrary to expectations that coral reef fisheries may decline as a result of the El Niño-related coral bleaching in 1998, especially in a situation of an already degraded reef (Lindahl *et al.*, 2001). The lack of decline may relate to low vulnerability to temperature stress as a result of an already low abundance of corals, and/or a positive response of algal and invertebrate eating fish. Spearguns are the primary gear contributing to this increasing trend, with the other gears showing no significant change, or slight negative trends (Fig. 2, Table 2). The implications for fisheries management are complex, as spearguns are an entry-level gear for many young men and poor fishers, and in some landing sites account for >50% of the fishermen.

The reef fishery in Diani has been under high pressure for several decades (Khamala, 1971), having passed a typical coral reef Maximum Sustainable Yield (MSY) of 10–20 tons/km^2/yr, which is achieved at a fisher density of approximately 10 fishers/km^2 (McClanahan, 1992). The estimated total catch at Mwaepe, Mvuleni and Mwanyaza can be extrapolated from the participatory monitoring data, yielding figures of 22.0, 11.1 and 5.1 tons/km^2/year for Mwaepe, Mvuleni and Mwanyaza,

Table 5. Total fisheries output estimated for Mwaepe, Mvuleni and Mwanyaza, from December 1999 to March 2000

Landing site	Mwaepe	Mvuleni	Mwanyaza
#fishers sampled (/day)	12.13	16.16	11.58
#fishers total (/day)	33.91	33.28	43.5
% of fishers sampled	35.76	48.55	26.63
Est. total catch (/day)	182.88	195.27	133.71
Reef length (km)	2.9	4.2	6.2
Reef area (km^2)	2.5	5.3	7.9
# fishers (per km^2)	13.6	6.3	5.5
yield (tons/km^2/yr)	21.96	11.1	5.1
Exploitation level	over	even	under

Table 6. Interpretation of correlations among fishing variables – site use frequency, # fishers per crew, total catch and catch per fisher (Fig. 5)

Correlation	Interpretation	Behavioural interpretation
Negative		
Frequency of site use *vs* total and per fisher catch	Highly visited sites have lower catch rates than less visited sites	Fishing pressure to preferred sites is reduced when catch declines significantly below less preferred sites
Catch per fisher *vs*. number of fishers per crew	Individual share of catches decreases with increasing numbers in a crew	Fishers join larger crews for economic reasons and the perceived greater success of overall catch (see below). However, individual returns are less as the catch is shared among more individuals
Positive		
Number of fishers per crew *vs*. site use frequency	Larger crews tend to fish at more highly frequented sites	Larger crews may have more conservative fishing patterns controlled by gear or preference, while smaller crew teams may explore more and less productive sites
Total catch *vs*. number of fishers per crew	Larger crews catch more fish.	Larger crews catch more fish due to greater effort and/or more efficient gear
Catch per fisher *vs*. total catch	Larger total catches result in great catch per fisher.	–

respectively (Table 6). Mwaepe and Mvuleni are apparently at MSY, and Mwanyaza below. However, all sites have previously been reported to be overexploited (McClanahan *et al.*, 1997), and these numbers may reflect declining yields beyond the MSY. This difference in annual yield among the 3 landing sites is not clear in the seasonal and per capita catch levels from these sites (Fig. 2). This may be due to many factors, one of which may be the spatial overlap in fishing areas among the landing sites.

Local Geography of Fishing

Reconciling coral reef fisheries assessment with the geography of fishing is an important area that is generally ignored in management. While fisheries analyses for coral reefs tend to view the entire fishing area seawards of the landing site as homogeneous, fishers see and utilize a highly complex seascape of sites with different attributes, that are named and mentally catalogued. Fishing site use, such as by different gears, is an important feature of any fishery, and in particular artisanal ones (Johannes, 1981), for example in how it mediates interactions between users. Traps and spearguns are used in similar habitats (Fig. 4), and fishers that use these gears report high levels of conflict with each other (CORDIO, unpublished data). Beach seines require shallow habitats also used by most other gears (Obura *et al.*, 2001), and there has long been a high degree of conflict between them and other users (Rubens, 1996; McClanahan *et al.*, 1997; King, 2000). Handlines are more commonly used in reef channels and the fore reef where depth is too great for set-gear such as traps and nets, and their perceived levels of conflict with other users are low (CORDIO, unpublished data). Nevertheless, even with the high degree of overlap, data analysis does suggest consistent differences in spatial use among gear types (Fig. 4), and this information may be used to inform management decisions.

Site use by individual fishers may be affected by a number of factors, one of which can be expected to be site productivity. However, the most highly visited sites are those with average catch rates, not those with highest catch rates (Fig. 5). This suggests there may be a trade-off between the tendency to visit productive sites more often, which may depress catch rates at these sites to the extent that their production is fully exploited. Once catch rates at these sites are depressed, they no longer attract fishers, and other sites become preferred. Correlations among a number of spatially explicit variables (Fig. 5) suggest other patterns that potentially explain site use behaviour by fishermen (Table 3). These need to be tested through directed studies, following which they may be useful in developing management interventions. For example, the negative relationship between catch per fisher and number of fishers per fishing crew could form the basis of a campaign to discourage the use of illegal beach seines that require large number of fishers.

Gear Choice and Socio-Economic Factors

The results presented here show that only marginal differences in catch exist between the most common gear types of low to medium cost (handspears, spearguns, handlines, traps, gillnets and seine nets; Fig. 3, Table 5). This encourages fishers to use the gear of least cost, ie. handspears, spearguns or seine nets, as they are under intense pressures to minimize expenses. Spearguns have a slightly higher catch rate than the other gears which though not statistically significant is financially significant over the long term. Joining a seine net crew involves no cost to the fisher, and though the per capita catch is slightly lower than other gears, this may result in a higher net income. Handspears are also a cheap gear, but are often viewed by young fishers as a gear of the past, as they are popular among older men, and are seldom used by young fishers.

Increasing poverty of fishers hinders their ability to invest in gear, and new entrants to the fishery often select seine nets or spearguns to earn their living. Both of these gears are now illegal under Kenya's Fisheries Act (GOK, 1991). Beach seines have been illegal since the Act was first written (but with unclear references to coral reefs). Beach seines are the most common destructive fishing method in the Diani area (McClanahan *et al.*, 1997), with a long history of conflict here and in other parts of the Kenya coast (King, 2000). The destructive impact of beach seines has been well documented in Kenya and elsewhere, including the catch of juvenile fish and habitat destruction (Rubens, 1996; McClanahan *et al.*, 1997). Though illegal for many years, enforcement has been irregular, improving over the years as opposition to beach seines from other resource users, scientists and managers grows. Spearguns require an active mode of fishing, and it is perceived by other fishermen that this chases fish away from an area, that speargunners catch fish before they enter passive gear (such as traps and gillnets), and/or speargunners take fish out of passive gear. As a result of pressure from fishers groups, which are comprised mainly of older fishers, spearguns were declared illegal in a review of the Fisheries Act in late 2001, to stop the perceived disturbance by speargunners to set gear (S. Hemphill, personal communication). However, the impact of spearguns has not yet been quantified.

Management Implications

Fisheries management in Kenya focuses on prohibition of illegal gears, which currently covers the two main gear types accessible to poor fishers. For this reason, implementation of the laws will cause increasing conflict as the number of new and poor fishers grows with the expanding human population. Gear exchange is a traditional option, but the strong preferences shown by fishers needs to be overcome, by using for example the results on gear productivity above in a targetted education programme. Additionally, sustainability of gear exchange requires that maintenance and renewal costs remain within the financial range of poor fishers (ie. up to medium-priced gears, Table 5). Gear exchange to the most highly preferred gear, bottom nets, may only be possible for groups of fishers, also with appropriate financial assistance. An additional option is to alter the management regime from complete gear bans to temporal, seasonal, or site-based controls, for example in preventing spearfishing at night, during some months, or at certain sites designated for passive gear only.

The success of any management intervention, however, would be dependent on determining if catch rates will remain stable or even increase with changes in use patterns, as the exploitation level of fisheries is likely to be at or over MSY (Table 3; McClanahan, 1992). Increasing degradation of the local reef from overexploitation and climate change impacts may not yield further fisheries declines if the reef is already in a relatively stable degraded state and fisheries production depends more on algal and planktonic production than coral reef diversity and productivity. Continued monitoring of the system through participatory methods is a growing priority of the provincial Fisheries Department, through a collaboration with CORDIO.

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A Consideration of the Biodiversity and Future of Southern African Coral Reefs

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Key words: climate change, coral biodiversity, coral distribution, sea surface temperatures, Southern Africa

ABSTRACT

Coral communities in Southern Africa form a continuum from the more typical, accretive reefs in the tropics of Mozambique to the marginal, southernmost African distribution of this fauna in KwaZulu-Natal. While the latter are limited in size, they are gaining increasing attention as they provide a model for the study of many of the stresses to which these valuable systems are globally being subjected. Soft coral cover, comprising relatively few species, exceeds that of scleractinians over much of the southern reefs, and the coral communities attain a high biodiversity at this latitude on the East African coast. A long-term monitoring programme has revealed small yet significant changes in community structure on the reefs in recent years, concurrent with consistent increases in mean and maximum temperature. Insignificant bleaching was encountered during the 1998 ENSO event, unlike elsewhere in East Africa, but quantifiable bleaching occurred during an extended period of warming in 2000. Outbreaks of COTS have caused longer-term changes in isolated areas. A study of coral larval dispersal and recruitment has been initiated to establish the capacity of the reefs to recover from the latter form of disturbance. The marginal nature of the reefs is further manifested by corals that generate aseasonal and atypical natural products and have a reproductive pattern that conforms with the pattern found on marginal reefs in W. Australia. Calcium deposition on the reefs is also low due to physico-chemical factors that are related to latitude. Published projections on the long-term effects of climate change indicate that more reefs will become marginal as a result of global warming. Current monitoring on the South African reefs is being expanded to investigate the extent to which they will elucidate the future of more typical reefs.

INTRODUCTION

Coral communities in South Africa constitute the southern most distribution of this fauna on the African coast and fall almost entirely within MPAs in northern KwaZulu-Natal (Schleyer, 1999, 2000; Fig. 1). The reefs on which they are found are not true, accretive reefs since corals merely grow as a thin veneer on the limited Pleistocene sandstone substrata in the region (Ramsay & Mason, 1990; Ramsay, 1996). As coral reefs, they are thus marginal in nature. Soft coral cover, comprising relatively few species, exceeds that of the more diverse scleractinian cover over much of the reef area, yet the coral communities attain a high biodiversity (Table 1) at this latitude on the East African coast (Riegl *et al.*, 1995; Schleyer, 1999, 2000). The purpose of this brief review will be to assess how marginal the reefs are, the extent to which they may be affected by climate change and whether they will provide a model on the global future of corals by virtue of their unique attributes.

BACKGROUND

Kleypas *et al.* (1999) assessed the limitations on coral growth of a number of physico-chemical determinants at different latitudes using modelling techniques. They concluded that, in terms of the parameters they considered (salinity, the nutrients NO_3 and PO_4, light penetration, temperature and aragonite saturation state), the South African reefs fall within the marginal limits of the last three for coral growth. They anticipate that marginal reef development at high latitudes will become increasingly subject to the divergent forces of increasing temperature resulting in reef expansion and declining levels of aragonite saturation resulting in diminished reef accretion. They do not expect the incidence of light or light penetration to change or affect coral growth beyond the present level.

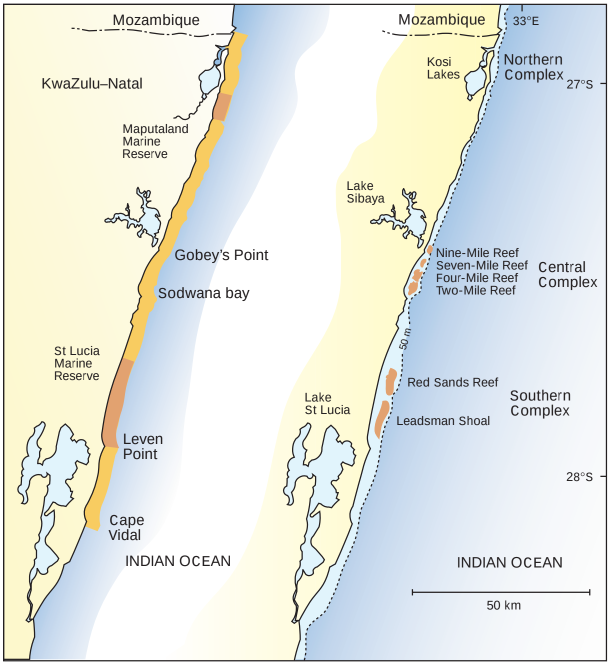

Figure 1. The major coral-inhabited reefs in KwaZulu-Natal fall within MPAs in reserve areas (light shading) and sanctuary areas (dark shading) in the Maputaland Marine Reserve (north of Gobey's Point) and the St. Lucia Marine Reserve (to the south).

Table 1. Biodiversity on the South African coral reefs in terms of Scleractinia (Riegl, 1995a, b, 1996a, b; Riegl *et al.*, 1995), Alcyonacea (Benayahu, 1993; Benayahu & Schleyer, 1995, 1996), Ascidiacea (Monniot *et al.*, 2001) and Porifera (Schleyer, unpublished data). Figures for the last-mentioned are for material identified thus far in current, incomplete work.

Taxon	Genera	Species
Scleractinia	43	93
Alcyonacea	11	39
Ascidiacea	17	30
Porifera	20	?

As we hope to demonstrate, we believe that the South African reefs are presently experiencing increased coral growth in the short-term due to the warming, greenhouse effect of increasing global atmospheric CO_2 (Table 2). However, in the long-term, Kleypas *et al.* (1999) expect that the CO_2 build-up will globally and radically retard coral accretion through a reduction in the marine aragonite saturation state. The present thin accretion on our reefs is already a manifestation of the marginal degree of aragonite saturation in South African waters and, with a further decrease, accretion will again diminish.

Table 2. The anticipated effects of climate change on coral growth in South Africa resulting from increasing atmospheric CO_2 (derived from Kleypas *et al.*, 1999)

	Effects on physico-chemical parameter	Effects on coral growth and reef formation
CO_2 Increase	Temperature ↑	Could increase in the short-term
	Aragonite saturation state ↓	Will decrease in the long-term
	Light (unchanged)	Unchanged

THE STATUS OF THE SOUTH AFRICAN REEFS

These changes were, to a degree, anticipated in the Coral Programme of the Oceanographic Research Institute (ORI) some years ago. A long-term monitoring site was established on one of the reefs at Sodwana Bay in 1993 and fixed quadrats have subsequently been photographed annually and stored for image analysis. An underwater temperature recorder (UTR) was also installed at the site in 1994 and this has yielded temperature records manifesting a steady temperature increase of 0.27°C p.a. (Fig. 2).

A preliminary assessment of changes in community structure by image analysis of the first five years of quadrat photographs yielded small changes in community structure (Schleyer *et al.*, 1998., and a ten-year data set is presently under investigation. The data is being analysed for changes in community structure and coral recruitment, growth and mortality. In general, the hard coral cover on the reef is increasing while that of the soft corals is decreasing (Fig. 3; Schleyer & Celliers, 2002, in prep.).

It would appear that the >2°C increase measured on the Sodwana reefs in the last decade has, up to now, encouraged scleractinian calcification and growth. If the projections modelled by Kleypas *et al.* (1999) prove correct, the scleractinian cover will later decrease as the aragonite saturation state diminishes and calcification becomes more inhibited. The monitoring is ongoing and is being expanded to include measurements of light and aragonite saturation state.

Interestingly, only slight bleaching was recorded on the South African reefs prior to and including the dramatic El Niño Southern Oscillation (ENSO) event in 1998 (Celliers & Schleyer, in press) that so devastated coral reefs in East Africa (Schleyer *et al.*, 1999; Obura *et al.*, 2000). Despite the temperature increase, the 'marginal' temperatures in the South African latitudes did not attain hazardous levels for the corals. However, a protracted period of elevated temperatures (see Fig. 2) with high levels of irradiance caused by exceptional water clarity elicited measurable bleaching in 2000 (Celliers & Schleyer, in press). The critical temperature at which bleaching appeared to be engendered was ~28.8°C.

Coles, Jokiel and Lewis (1976) suggested that the

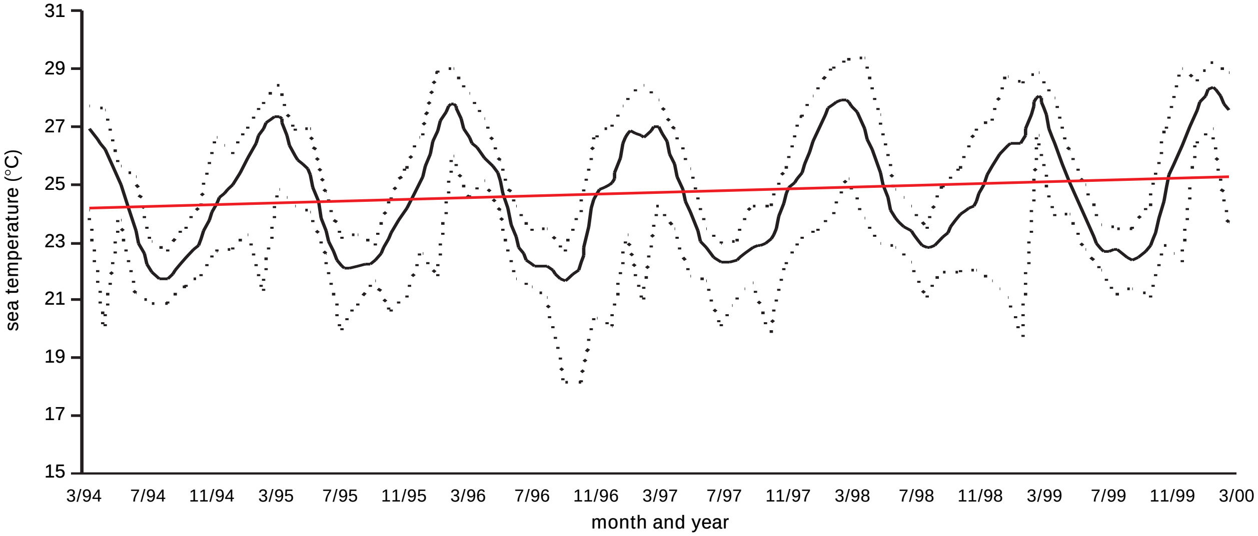

Figure 2. Monthly mean temperatures (solid line) with maxima and minima (broken lines) recorded since March 1994 at 18 m depth at the ORI fixed monitoring station on Nine-mile Reef at Sodwana Bay.

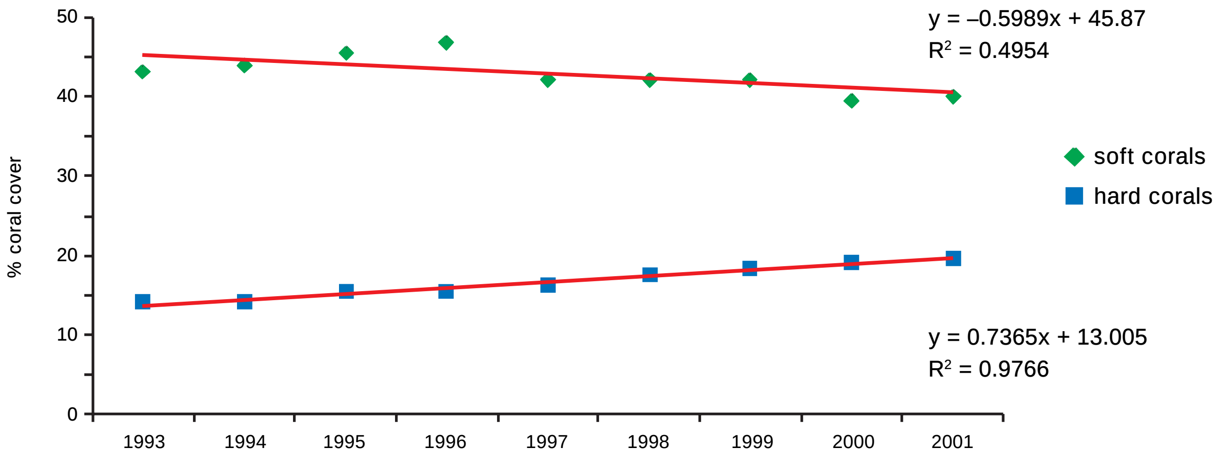

Figure 3. Mean percentage hard and soft coral cover recorded in fixed quadrats at a long-term monitoring site on Nine-mile Reef at Sodwana Bay between 1993–2001.

Table 3. Rates of sea temperature increase (°C.decade^{-1}) for a selection of tropical reef locations (adapted from Fitt et al., 2001)

Locality	Period examined	Rate of increase (°C.decade^{-1})	Source
Jamaica	1903–1999	0.125	Hoegh-Guldberg (1999)
Phuket, Thailand	1904–1999	0.154	Hoegh-Guldberg (1999)
Phuket, Thailand	1945–1995	0.126	Brown *et al.* (1996)
Tahiti	1926–1999	0.069	Hoegh-Guldberg (1999)
Tahiti	1956–1995	0.08	Brown (1997)
GBR, Australia	1902–1999	0.125–0.168*	Hoegh-Guldberg (1999)
Tropical oceans	1982–1996	0.001–0.053*	Strong, Kearns and Gjovig (2000)
Sodwana Bay, SA	1950–2000	0.1	This study

*latitude dependant

upper thermal limits for corals on subtropical reefs were approximately 2°C less than for congeners in tropical regions. The lower maximum temperature at which coral bleaching was observed at Sodwana Bay seems to support this. The lowest mean monthly and mean maximum monthly temperatures at which bleaching occurred were 27.5°C and 28.8°C respectively. The duration of the thermal exposure is also important (°C heating weeks or days) although it is not clear how the magnitude of SST anomalies and their duration combine to produce bleaching (Podestá & Glynn, 2001). The duration at higher temperatures in 2000 was 4 days at ≥28.8°C (67 days at ≥27.5°C) while in 1998 it was 5 days at ≥28.8°C (46 days at ≥27.5°C; Table 8.3). It would appear that the elevation in temperature contributed to the bleaching in 1998 while the duration of elevated temperatures was also a contributory factor in 2000.

A comparison of the Sodwana Bay region with other coral reef localities revealed a similar increase in sea temperature of 0.1°C.decade^{-1} (Table 3). The recent,

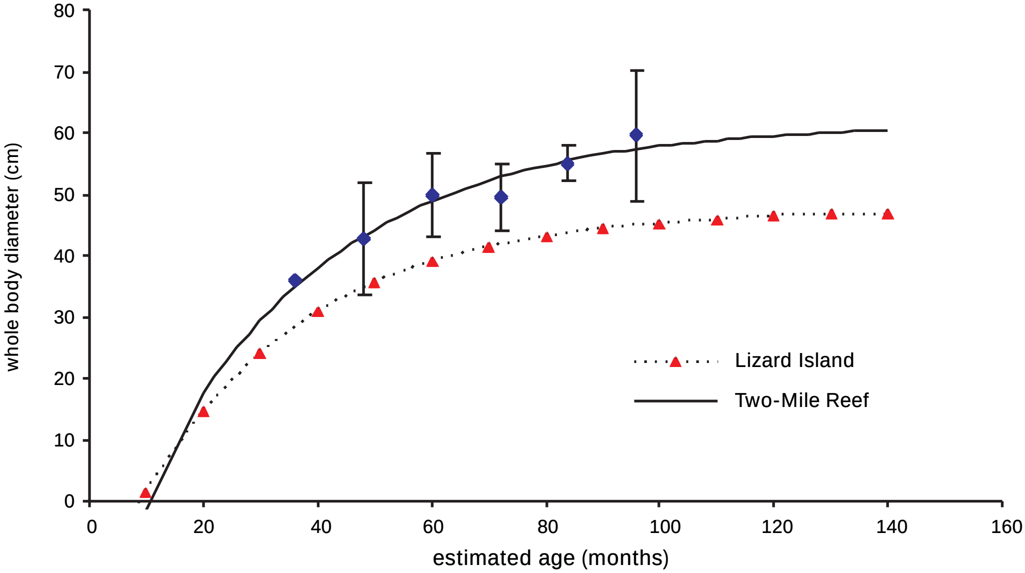

Figure 4. Growth of non-swarming *Acanthaster planci* on Two-mile Reef at Sodwana Bay compared with COTS swarming on Lizzard Island Reef on the Great Barrier Reef in Australia (Stump 1996).

more dramatic increase is thus probably part of a macrocyclical phenomenon and is unlikely to be maintained. The area is nevertheless undoubtedly undergoing the general temperature increase associated with global warming.

A crown-of-thorns starfish (COTS; *Acanthaster planci*) outbreak was also encountered on the reefs during the last decade (Schleyer, 1998) and became the subject of a Ph.D. study (Celliers, 2001). The behaviour of the COTS was atypical as they aggregated in summer, apparently to breed rather than feed in a swarm, and they dispersed in winter (Schleyer, 1998). They thus grew faster than COTS swarming and competing for food on the Great Barrier Reef (Celliers, 2001; Fig. 4), providing further evidence that the biota on the South African reefs live under marginal conditions. The outbreak caused a severe but patchy reduction in scleractinian cover. The affected reef communities entered alternative stable states subsequent to the outbreak, dominated either by soft corals of low diversity or hard corals of high diversity (Celliers, 2001).

In view of the marginal nature of the reefs, coral reproduction was studied to establish whether the local communities are self-perpetuating or dependant on recruitment from reefs to the north. Seven coral species, manifesting the full range of reproductive strategies, were thus examined histologically and normal reproduction was encountered (Table 4). It is of note that no evidence of the mass spawning in early summer typical of corals on tropical reefs was found. The alcyonacean broadcasting spawners studied did, however, appear to synchronise their spawning with the lunar cycle late in summer similar to scleractinia found on marginal reefs off Western Australia (Schleyer *et al.*, 1997).

Having established that the coral communities on the South African reefs are sexually active, it remains to be demonstrated that they are also self-seeding. A study of coral larval dispersal and recruitment has thus been initiated to establish the capacity of the reefs to recover from COTS and bleaching events. Monitoring of settlement is being accompanied by oceanographic measurements to assess dispersal using an acoustic Doppler current

Table 4. Reproductive strategies found in the corals studied on the Maputaland coral reefs (Schleyer *et al.*, 1997, in prep.; Kruger & Schleyer, 1998; Kruger *et al.*, 1998).

Species	Mode of reproduction
Anthelia glauca	Gonochoric brooder, repetitive seasonal planulation
Lobophytum crassum *Lobophytum depressum* *Sarcophyton glaucum* *Sinularia dura* *Sinularia gyrosa*	Gonochoric seasonal broadcast spawners
Pocillopora verrucosa	Hermaphroditic seasonal broadcast spawner

profiler (ADCP), wave height recorder, additional UTRs and CTD (conductivity, temperature and depth) determinations.

Finally, a correlation was anticipated between toxin production and gametogenesis during the reproductive cycle in soft corals. These natural products are known to have a number of functions in soft coral reproduction, e.g. sperm attraction and larval defence, and are usually produced in an annual cycle (Ketzinal, 1997). However, Ketzinal (1997) found no such correlation in South Africa. She studied two soft corals, *Lobophytum crassum* and *Eleutherobia aurea*, and they respectively produced five and eight different natural products randomly throughout the year, some of which were unique (Ketzinal *et al.*, 1996). This may be construed as further atypical behaviour of the corals in a marginal environment.

In conclusion, it is apparent that the South African reefs are marginal in terms of temperature, light, aragonite saturation state and many aspects of their biology. The effects of climate change are causing warming of our coastal waters, resulting in a shift in the coral community structure from soft to hard corals. This is probably related to an increase in coral accretion attributable to the rise in temperature. The trend will probably be short-lived as temperatures are now approaching the local coral bleaching threshold. A further build-up of the atmospheric CO_2 that is causing climate change is expected to reduce the marine aragonite saturation state globally, leading to a commensurate decline in reef formation and the scleractinian composition in coral communities. Changes on South African reefs are likely to precede those on more typical, tropical reefs. Studies on the former will thus elucidate the relationship between these complex mechanisms and may provide an insight to the global future of coral reefs.

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Part II

South Asia

Coral Reef Degradation in South Asia

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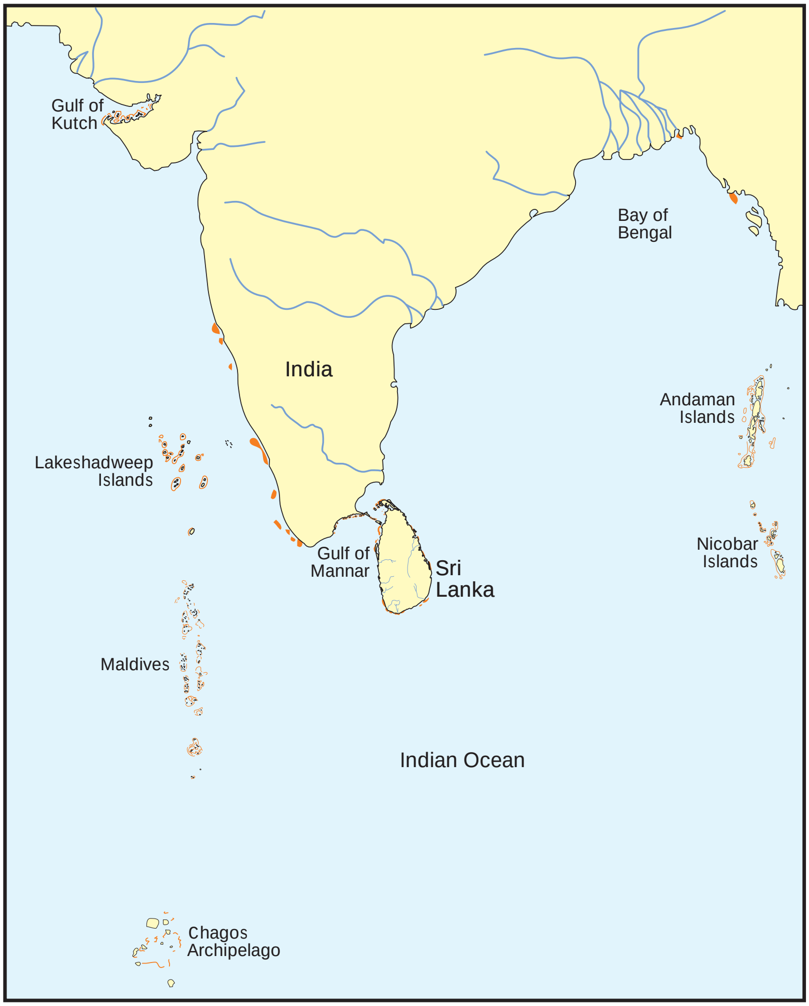
Key words: bleaching, coral reefs, CORDIO, South Asia

INTRODUCTION

During the severe El Niño event, with the resulting elevation of sea surface temperatures in 1998, most shallow coral reefs of South Asia (Fig. 1 on next page), here including India, Maldives and Sri Lanka, suffered extensive bleaching and subsequent mortality. While the deeper reefs, below app. 10 meters, to a large extent recovered from the bleaching, between 50% and 100% of the corals in many shallow areas were destroyed (Rajasurya *et al.*, 1999, Wafar, 1999, Rajasuriya & Karunarathna, 2000, Zahir, 2000). Other more chronic threats and causes of damage to the coral reefs in South Asia are the widespread destructive human impacts on the reefs, such as coral mining, destructive fishing methods, increased sedimentation, pollution, anchoring and tourism related activities (De Silva, 1985; Baldwin, 1991; Öhman *et al.*, 1993; Rajasuriya *et al.*, 1995; Rajasuriya *et al.*, 1998; Dharmaretnam & Kirupairajah, 2002; Patterson, this volume; Perera *et al.*, this volume). Several studies have pointed out positive correlations between fish abundance or diversity with structural habitat complexity or live coral cover (McClanahan, 1994; Jennings *et al.*, 1996; Munday *et al.*, 1996; Öhman & Rajasuriya, 1998; Öhman *et al.*, 1998; Sebens, 1991; Turner *et al.*, 1999; Williams, 1991), and alteration of habitats will affect the productivity of the ecosystem (Brown *et al.*, 1989; McManus, 1996; Öhman *et al.*, 1997). Consequently, the degradation coral reef habitats in South Asia also has secondary effects on the resource user groups, such as for example subsistent fishermen and ornamental fish collectors, directly dependent on the products of the reefs. Further, the reef related tourism in South Asia is affected by the loss of recreational value of the coral reefs, due to low abundances of corals and certain fish (Milon, 1989; Wilhelmsson *et al.*, 1998; Westmacott *et al.*, 2000). Since early 1999 the CORDIO program has supported bio-physical monitoring of coral reefs, including studies of changes in community structure of the reefs, recruitment patterns of coral and fish, rate of erosion of the reef framework, and prospects for recovery of coral reefs in the region. In addition, CORDIO has initiated studies on the trends in reef fisheries and the ornamental fish collection in Sri Lanka, and of the tourism industry in Maldives and Sri Lanka. Further, efforts of capacity building and raising of awareness on the issues have been made. This paper aims to give a brief update on the present status of the reefs in South Asia, and to report on progress and synthesised findings of CORDIO's activities in the region.

Figure 1.



STATUS OF THE REEFS

India

The reefs of Gulf of Mannar were severely affected by mortality of corals during 1998. Post bleaching surveys on the coral reefs of the 21 islands in the Gulf showed a mean cover of 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 (Pet-Soede *et al.*, 2000). Surveys at Tuticorin Coast, Gulf of Mannar in 2002, report of lustrous growth of *Acropora* spp. off the mainland in the protected areas with a live coral coverage of about 80%, compared to less than 5% in the unprotected areas. Around the islands off Tuticorin Coast, only about 20% live coral cover was noted (Patterson, this volume). The reefs of the Lakshadweep islands were reported to have lost between 43–87% of the live coral cover during the 1998 bleaching event, while in the gulf of Kutchch less than 30% of the corals were destroyed (Wafar, 1999; Pet-Soede *et al.*, 2000). Early reports from the Andaman and Nicobar Islands indicated that 80% of all corals bleached (Pet Soede *et* *al.*; 1999, Wilkinson *et al.*, 1999). Further, Ravindran *et al.* (1999) reported that 90% of the massive corals and 75% of the branching corals were bleached on 3 surveyed reefs in the Andaman's in July 1998. However, the mortality in this area seem to have been insignificant, or relatively local, since subsequent surveys have showed no signs of mass mortality at all in Andaman and Nicobar Islands (Turner *et al.*, 2001; Kulkarni & Saxena, this volume). Turner *et al.* (2001) further pointed out the significance of the Andaman Islands for global biodiversity, and their coral reefs potential as a source for natural seeding and re-colonisation of other more degraded reefs in the region.

Maldives

Results of monitoring of the reef tops conducted during 1999–2002 show that there has been no significant increase in live coral cover since the post bleaching surveys conducted in 1998, and the average remains at approximately 2–3% (Zahir, page 139, this volume). At present, although new recruitment has been noticed at all sites, the cover of live coral is almost 20 times lower on the shallow reef flats than was recorded before the bleaching event. However, the deeper reefs appear to be in better condition. Further, the relatively high coral cover recorded during surveys at Addu region in 2002, suggests that the most severe impacts of the bleaching of 1998 may not have been as geographically widespread as initially thought (Zahir, this volume). In addition to biophysical monitoring of the reefs, studies are conducted to determine the spatial and temporal patterns of coral recruitment in Maldives. Results suggest that there is potential for re-colonisation of corals on the degraded reefs of Maldives through the influx of coral planulae from surviving colonies elsewhere (Zahir & Clark, this volume). However, the studies further indicate a relatively poor supply of *Acropora* larvae. This make the recovery of this previously common genus slower, while corals of *Pavona* dominate among the new recruits. Also, the degree of erosion and changes in the topographic complexity of these reefs are being assessed following the extensive coral mortality in Maldives. Preliminary results illustrate a varying intensity of bio-erosion at different sites and depths (Zahir, page 135, this volume).

Sri Lanka

Surveys have revealed that most of the dominant forms of reef building corals in many of the shallow coral habitats (<8 m) were destroyed in Sri Lanka, during the bleaching event in 1998 (Rajasurya *et al.*, 1999). The dead coral reefs are largely dominated by algae, tunicates, and corallimorpharians (Rajasuriya & Karunaratna, 2000; Rajasurya, this volume). Corals growing in deeper waters (>10 m) survived the bleaching better, potentially providing a source for new recruits. Recovery of bleached corals in shallow reef habitats is variable between sites, but has in general been slow and in some places non-existent (Rajasuriya, this volume). Further, uncontrolled and destructive human activities continue to damage the reefs. Even the marine protected areas in Sri Lanka are largely unmanaged and increasing human activities within these protected areas continue to degrade their condition (Rajasuriya & Karunaratna, 2000; Rajasuriya, this volume). A significant decrease in number of butterfly fish (Chaetodontidae), many of which are usually associated with live coral, has been

Figure 2. The live coral cover in Bar Reef Marine Sanctuary, northwestern Sri Lanka, was high (around 78% (Rajasuriya and Karunarathna, 2000)) before the mass bleaching in 1998. *Photo:* ARJAN RAJASURIYA, NARA, Sri Lanka.

Figure 3. During the bleaching in 1998 nearly all corals of the dominating genus *Acropora* and also *Echinopora lamellosa* died, and the reef was covered with filamentous algae (Rajasuriya and Karunarathna, 2000) *Photo:* E. N. R. PERERA, Sri Lanka Sub Aqua Club.

observed on several reefs in Sri Lanka (Rajasuriya & Karunarathna, 2000; Rajasuriya, this volume; Wilhelmsson *et al.*, this volume).

CORAL REEF RESOURCES

Resources extraction from the coral reefs in South Asia include fisheries for finfish (e.g. trevallies (*Caranx spp.*), emperors (*Lethrinidae*), snappers (*Lutjanidae*), groupers (*Serranidae*) etc.), spiny lobsters (*Panulirus spp.*), sea cucumbers (*Holothuridae*), and also collection of fish, molluscs, crustaceans and corals for the aquarium trade. In addition, although illegal in most areas, mining of corals for road and building construction, and to produce lime for agriculture and industries, is still occurring in the region (Patel, 1985; Rashid, 1988; Wafar, 1986; Perera, 2002; Patterson, this volume). Further, coral reefs provide a number of non-extractive values, such as attraction of tourists and the function as break waves protecting coastlines from erosion. Moreover, coral reefs host a large portion of the known marine animals and plants, and thus support a high bio-diversity. The productivity of marine ecosystems is important as a source of food and livelihoods for many people in the coastal areas in South Asia. If used and managed in a sustainable manner, coral reefs can provide services of significant economic value. For example, Berg *et al.* (1998) estimated the minimum economic value of coral reefs in Sri Lanka at USD 140,000–7,500,000 per km^2 over a 20-year period. Further, the same study pointed out the great net national economic losses of for example coral mining in potential tourist areas in Sri Lanka. The annual economic net benefits of sustainable use of coral reefs have also been highlighted for other regions, such as for Indonesian reefs (Cesar *et al.*, 1997), and on average for South-East Asian reefs (Burke *et al.*, 2002). Despite all these values, pollution, sedimentation, and unregulated and destructive exploitation continue, resulting in rapid degradation of the coral reefs.

Reef Fisheries

Wafar (1986) estimated the landings of reef fisheries to contribute about 10% (0.2 million tonnes per year) to the total marine fish production in India. These figures for the reef fisheries in India notably exceeds the total near-shore and offshore landing for Sri Lanka in 1990 (NARA, 2001). A study of the socio-economic impor-

Figure 4. On many shallow reef tops in the Maldives, 95% of the corals died during the bleaching in 1998 (Zahir, 2000). *Photo:* HUSSEIN ZAHIR, MRC, Maldives.

tance of reef fisheries in India, suggested that the increasing demands for food caused by growing coastal populations, and overexploitation of coastal shelf areas, will increase the pressure on India's coral reefs to provide food and income to the people in the coastal areas and on the islands of India (Pet Soede *et al.*, 2000). In the Maldives, tuna is dominating the seafood, and reef resources are not extracted for food to the same extent, although bait and hook fishing are important in several atolls (Sheppard & Wells, 1988). In Sri Lanka, it is estimated that around 15% of the marine landings are from reef fisheries (Rajasuriya *et al.*, 1995). Destructive fishing methods, such as for example bottom-set net- and blast fishing, continue to damage the reefs in Sri Lanka and India (Öhman, 1993; Rajasuriya *et al.*, 1998; Patterson, this volume; Perera *et al.*, this volume). A study in three selected areas in Sri Lanka under the CORDIO program suggests that the mass mortality of corals in 1998 had adverse impacts on the fish catches. However, the fish stocks on the coral reefs where reported to be declined due to over fishing and the use of destructive fishing methods already before the bleaching event. In addition, although the fishermen where aware of the damage caused on the reefs, and the negative effects it could have on their future fishing, they continued to use the destructive nets due to the relatively good catches it generated in the short-term (Perera *et al.*, this volume).

Monitoring of Ornamental Fish Collection and Trade

In Sri Lanka, NARA (National Aquatic Resources and Research Agency), with support from CORDIO, has initiated a program for monitoring of the collection and trade of marine ornamental fish. The program will hopefully serve as a useful tool in management of the ornamental fishery industry in Sri Lanka. Since the collection of marine ornamental fish is practiced entirely for the export market a date base, building on packing lists from the customs, is being developed. This is supplemented by structured interviews with ornamental fish suppliers, and collection of logbooks. Analysis of the data suggests a decline in catches and abundance of ornamental fish on the reefs, due to increased collection efforts, destructive fishing methods and, especially for fish feeding on coral polyps, due to the coral bleaching in 1998 (Wilhelmsson *et al.*, this volume).

Coral Reef Degradation and Tourism

Coral reef related tourism is of particular importance in the Maldives, where travel and tourism contribute around 56% to the national economy and about half of the visitors are scuba divers (Westmacott *et al.*, 2000). Within the CORDIO program surveys were conducted to evaluate the potential economic impacts on tourism in the Maldives and Sri Lanka, due to the mass mortality of corals in 1998. Results suggest that the tourism industry was adversely affected by the coral mortality in the Maldives (Westmacott *et al.*, 2000; Cesar *et al.*, 2000). In Sri Lanka there seem to be no major impact on the scuba diving industry (Cesar *et al.*, 2000). On the other hand, according to continued surveys in 2000–2001, the snorkelling and glass bottom boat industries decline due to the degraded coral reefs along the south coast of Sri Lanka. For example in Hikkaduwa, south-western Sri Lanka, only half of the 80 registered glass bottom boats were in operation during 2000–2001. Moreover, the number of daily trips per boat dropped 5–10 times compared to before the bleaching event in 1998 (Amaralal *et al.*, 2002). However, the glass-bottom boats are also part of the problem, causing considerable damage on the reefs in Hikkaduwa Nature Reserve. Boats and anchors break the corals, and tourists walk on the reef flats (Rajasuriya, this volume).

Also the rapid increase of coastal tourism in the northern and eastern Sri Lanka after the initiation of the peace process in 2002, calls for extensive efforts in awareness building in the tourism sector in order to minimize the damage to the reefs in these newly accessible areas. Hence, during the next year, as part of the CORDIO program a small scale project working with the private sector (e.g. dive- and glass-bottom boat operators) will suggest options for more environment friendly coral reef tourism in Sri Lanka. The aim of the project is to reduce the destruction caused by different tourist activities, and also to enhance the qualitative experience

Figure 5. Each day about 9 tons of corals are illegally mined in Kalkudah, eastern Sri Lanka, partly due to lack of alternative income sources in the area (Dharmaretnam and Kirupairajah, 2001). *Photo:* DAN WILHELMSSON.

for the tourists. Amaral *et al.* 2002 points out the high economic potential of an expanded scuba diving industry in Sri Lanka. However, great care should be taken before any significant expansion of the coral reef related tourism is allowed to take place, primarily due to the almost inevitable damage from increased tourist activities on the reefs. For example, in Eilat, Israel, several hundred thousand coral colonies are broken each year from recreational scuba diving (Zakai & Chadwick-Furman, 2000). Worth pointing out is that this is despite relatively active and well-developed reef management, and high environmental awareness among wardens, rangers, and dive operators in the area.

Coral Mining

Coral have been the main construction material in the Maldives for hundreds of years. However, recognising the ecological and economic damage of coral mining, the government is now seeking to limit the extent of coral removal through regulations, and promotion of the use of concrete blocks as an alternative (Risk & Sluka, 2000). Mining of corals cause considerable damage to coral reefs in the South Asia region. A consequence of this activity is increased beach erosion particularly along the west coast south of Colombo and along the south coast in Sri Lanka. Also in India there are a number of examples of coastal areas suffering from erosion due to coral mining. For example in Tuticorin Coast, Gulf of Mannar, India, one island has been submerged due to erosion caused by extensive coral mining in the area (Devaraj, 1997; Patterson, this volume). Moreover, during three months in 1993 about 770 tons of coral were mined in three study sites (Hikkaduwa, Habaraduwa, Madiha) in southern Sri Lanka (Banda, 2002). The activities may have decreased in recent years, due to increased law enforcement efforts, but this has not been verified. Further, a case study of the status of coral mining in Kalkudah, Batticaloa, on the east coast of Sri Lanka was carried out in 2001 (Dharmaretnam & Kirupairajah, 2001). The study showed that approximately 9.7 tons per day of corals were collected from the coral reefs in the area. Although there is an awareness of the extensive harm to the coral reefs caused by coral mining, and the resulting increased erosion of the coastline the activity is likely to continue as long as alternative sources of income is not assured (Dharmaretnam & Kirupairajah, 2001).

Options for Alternative Livelihoods

In order to reduce the pressure on the coral reef resources, but also to mitigate any adverse social and economic effects due to declining coral resources or management interventions providing real protection to the reefs, projects investigating alternative livelihoods will be further developed. A CORDIO project lead by SACEP (South Asian Co-operative Environment Program) is reviewing past, current and future applications of alternative livelihood approaches in Sri Lanka, in order to identify examples of successful activities. The project also tries to incorporate experiences from other parts of the world. The project was started at the inauguration of the ‘Sri Lanka Coral Reef Forum’, April 24, 2002, and useful contributions were made already during the event. Valuable inputs and recommendations are made from various institutions, such as Asian Development Bank, universities and governmental departments, as well as from individuals with past experience of the subject. A working group is preparing specific programmatic recommendations for the development of alternative livelihoods for coral reef dependants in Sri Lanka, for presentation at an inter-ministerial meeting hosted by SACEP (Perera, 2002). Further, there is scope for regional co-operation on these issues through the intergovernmental mandate of SACEP.

RAISING OF AWARENESS

Studies have shown that the awareness regarding the destruction taking place on the coral reefs due to mining and fishing using destructive methods is significant, also among those directly involved in these activities (Weerakody, 2002; Perera *et al.*, 2002). However, the adverse effects of these destructive activities are affecting also other people, such as fellow fishermen, people involved in the tourism activities etc. Thus, raising the awareness of the broader public is necessary to support legal and enforcement measures. It appears particularly important to increase the awareness and will at the political level. In 2001 CORDIO, together with GCRMN (Global Coral Reef Monitoring Network), DFID and, Shell Lanka Pvt, supported an educational and awareness project titled 'A tomorrow for our reefs' implemented by IUCN (World Conservation Union) in Sri Lanka. The awareness campaign started with an eight days exhibition in Colombo, followed by a mobile exhibition in Hikkaduwa and Tangalle during April–May, 2001. In Tangalle the average number of visitors per day was four thousand. Recommendations will be made for implementation of similar projects in other areas of South Asia (IUCN, 2001). During the exhibitions, schoolteachers often asked for material to assist them in the teaching of subjects related to the marine environment. Thus, as learning about coral reef ecosystems are not included within the formal school curriculum in Sri Lanka, CORDIO is supporting IUCN in producing an educational package in Sinhala, Tamil and English, to be used by the schools in Sri Lanka. Further, CORDIO assisted in the organisation of a seminar on environmental issues held in July, 2000, in Batticaloa, Sri Lanka. During the first day, local school children and teachers were invited to participate in discussions. On the second day, governmental officers, NGO's, and different stakeholders contributed with their views. One of the major topics discussed was the extensive coral mining taking place off Batticaloa.

Figure 6. The loss of coral reef habitats affects the people depending on the productivity of the marine ecosystems, such as for example ornamental fish collectors. *Photo:* DAN WILHELMSSON.

CAPACITY BUILDING

Several of the CORDIO projects are implemented with components of local capacity building included. For example, in Maldives CORDIO is providing a researcher with an MSc degree through the coral recruitment studies carried out by Marine Research Centre (MRC). The CORDIO program has also trained several persons at MRC, responsible for conducting coral reef related monitoring activities in the Maldives. 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. Further, a pilot scale community based reef restoration and monitoring project in Tuticorin Coast, Gulf of Mannar (Patterson, this volume), will be the basis for the Ph.D.-research of an Indian student. In Sri Lanka, with assistance from National Aquatic Resources Research and Development Agency (NARA), CORDIO has provided training of students at Eastern University on the east coast of Sri Lanka. Due to the armed conflict during the past two decades, there is a lack of information on the status of reefs in that area. Thus, the basic techniques of coral reef monitoring were taught, with the aim of developing the local capacity to undertake such monitoring. In addition, lectures and indoor training have been given at Colombo University, in Sri Lanka. Also GCRMN (Global Coral Reef Monitoring Network) has trained a number of marine researchers in South Asia, and provides networking opportunities from which the CORDIO program has benefited. An illustrative example of the close co-operation between CORDIO and GCRMN, and the complementary functions of the two programs in the region, is the training course that GCRMN conducted in Puttalam, Sri Lanka, in the end of 2000. Several researchers carrying out CORDIO-projects attended the workshop, focusing on socio-economic monitoring, which enhanced their capacity to implement the socio-economic components of their projects. Further, CORDIO co-funded the planning and implementation of GCRMN´s training course on Socio-economic Monitoring for Coral Reefs, in the Andaman and Nicobar Islands, India, January 26–February 5, 2001.

LOOKING AHEAD

Although there is a significant variation in the status of the reefs of the region, the recovery from the mass mortality in 1998 has generally been very slow. Reefs of the Maldives show more promising signs of improvement than elsewhere, with quite large numbers of new coral recruits in many places. However, a shift in dominating species has taken place and other species of corals seem to take over the reefs. In other parts of South Asia the reefs are in poor shape with little or no signs of recovery. For most areas in South Asia, it is clear that the health of the coral reefs is under continuous pressure and decline. To hope for any general recovery after the 1998 bleaching event is not realistic as long as all the other causes for degradation of the reefs are still there; destructive fishing techniques, coral mining, pollution from land of sediment and sewage.

The loss of the coral reef habitats in South Asia inevitably and most exceedingly affects the people directly depending on the coral reefs for daily food and income. These groups are often poor, and usually lack access to alternative livelihoods. The lack of alternatives is also often the cause for the destruction, such as coral mining or bottom-set net fishing on the reefs. For example, in Kalkudah, Sri Lanka, many people lost access to their paddy fields during the armed conflict in the area. Around 100 of these families shifted into coral mining as an alternative source of income. Today over 400 families in Kalkudah (of totally 630), are in some way involved in the lime industry (Dharmaretnam & Kirupairajah, 2001). Also, in Rekawa, Sri Lanka, a prolonged drought during the 1980´s, and depletion of the fish stocks in the lagoon, forced many farmers and fishermen into coral mining and lime production (Perera, 2002). However, in other cases people take up for example dynamite fishing and other destructive practices simply because they offer relatively lucrative returns in the short-term (Weerakody, 2002; Perera *et al.*, this volume). In general, there is a need to improve the capacity and competence of the law enforcement. However, for the law enforcement to be effective, an enhanced political will to protect and manage the marine and coastal resources is required in South Asia. In addition, there is a lack of experiences and will when it comes to the application of findings from scientific research and monitoring into management actions.

In its next phase the CORDIO program will, in addition to supporting the research and monitoring of the status of the coastal reefs, seek to exemplify how scientific data and results can be used for management measures. Small-scale pilot projects are under development, which will use the knowledge gained from research and monitoring to improve the quality of life for coastal communities by assisting in developing alternative livelihoods, and at the same time protecting the coastal environment.

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The Conservation Status of Coral Reefs of Andaman and Nicobar Islands

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Key words: Andaman, Nicobar, conservation, coral reefs

INTRODUCTION

Coral reefs all over the world are under tremendous pressure of exploitation. Their very existence is now threatened by numerous problems such as industrial pollution, deforestation and, to some extent tourism, that are causing large-scale degradation and declines in biodiversity of reefs (e.g. Rogers, 1990). In addition to local factors, reefs are now also threatened by global climatic changes such as increases in sea surface temperature and changes in sea level. In order to determine the impacts of these threats, studies of the coral community structure are valuable tools (Downing, 1985; Sheppard, 1988; Riegel *et al.*, 1995).

In India, major coral reef formations occur in the Lakshadweep, Andaman & Nicobar Islands, Gulf of Mannar and the Gulf of Kutch (Pillai, 1983a, b; Wafar, 1986; Kulkarni, 2000). The Andaman and Nicobar Island chain constitute a string of the oceanic islands that are separated from the coasts of Myanmar and Thailand by the Andaman Sea to the east, from the Indian peninsula by the Bay of Bengal to west, and from Sumatra by the Malacca Strait in south. This Indo-Pacific reef province is more or less contiguous with that of Strait of Malacca, Arakan coast and the East Indies (Pillai, 1983a) but is separated from the Sri Lankan and southeast Indian coral formations by nearly one thousand kilometers, which represents a significant gap in the coral bearing areas of the Indo-Pacific.

The Andaman and Nicobar Islands are situated in the Bay of Bengal between 6° and 14° N latitude, 92° and 94° E longitudes and are separated by about 160 km (Fig. 1 on next page). There are 349 islands in the Andaman and Nicobar Island group. Together they constitute a Union Territory (U.T.) of the Union of India. Being close to the equator, the islands have a tropical climate that is characterized by heavy precipitation. Under the influence of both the northeast and southwest monsoons, it rains seven months in a year. Cyclones sometimes occur at the change of monsoons.

Prior to 1998, there was no proper assessment of the status of the coral reefs of the Andaman and Nicobar islands. However, training courses in monitoring techniques and coral taxonomy have been conducted by the Global Coral Reef Monitoring Network (GCRMN) and Indian Coral Reef Monitoring Network (ICRMN), which has enabled the local Department of Environment and Forests to monitor and effectively manage these reefs. In addition, two marine national parks, Mahatma Gandhi Marine National Park (MGMNP) at Wandoor on the western coast and Rani Zansi Marine National Park on eastern cost of South Andaman have been established to assist with the conservation of these ecosystems.

Although the coral reef science in Andaman and Nicobar Islands is in its infancy, assessment of the coral reefs remains a priority. Moreover, the results of moni-

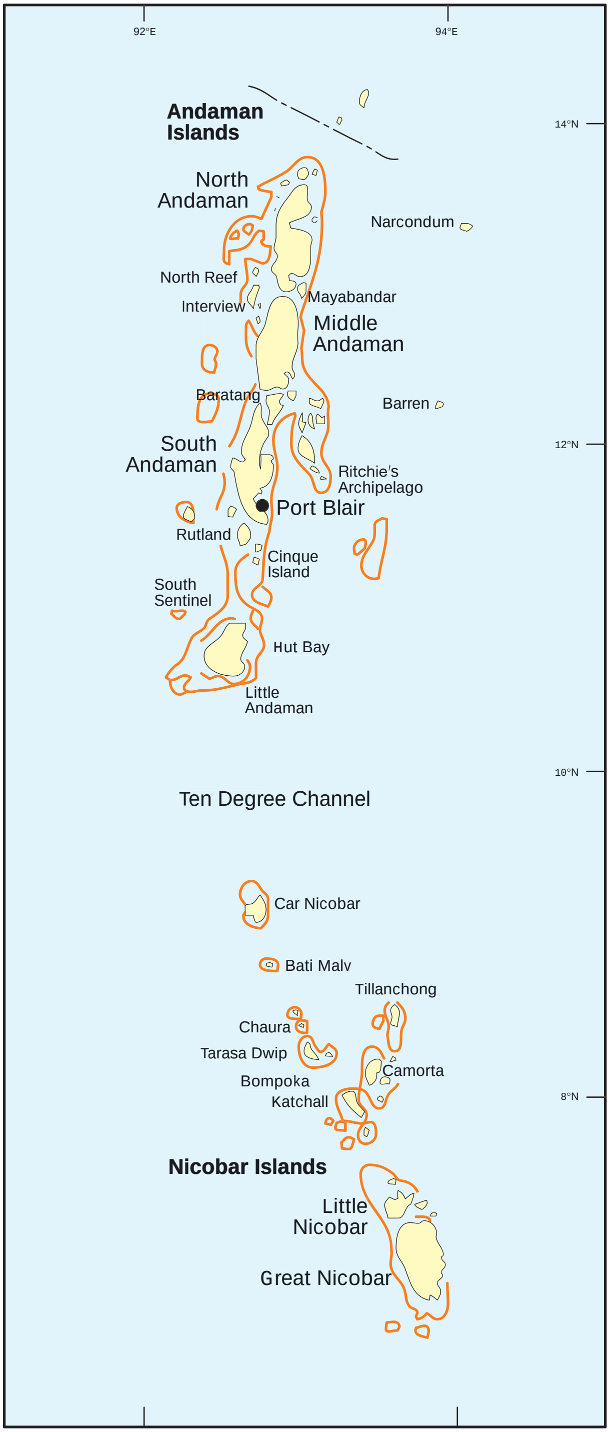

Figure 1. The Andaman and Nicobar Islands.

toring should be used to recommend actions that will promote the conservation of coral reef resources, rather than simply being an academic exercise.

Assessing the conservation value of natural habitats is important for the development of conservation policy, particularly for tropical biodiversity conservation (Edinger & Risk, 2000). Coral reef conservation value is related to biodiversity of corals, invertebrates and fish (Edgar *et al.*, 1997), fisheries potential (Jennings *et al.*, 1996) and habitat for rare or endangered species.

Estimates of coral reef conservation value can be used in choosing which areas are most deserving of protection (Done, 1995; 1996; Edgar *et al.*, 1997). Tools for making such decisions are badly needed, particularly in Andaman and Nicobar Islands (Kulkarni *et al.*, 2000).

In order to be useful in Andaman and Nicobar Islands, where most officials within the Department of Environment and Forests who are coral reef managers have little or no coral taxonomic expertise (Kulkarni *et al.*, 2000), measures of coral reef conservation value should not be taxonomically based at the level of coral species (Edinger & Risk, 2000). Here we demonstrate the use of coral reef classification based on conservation value, which was first used for coral reefs by Edinger & Risk (2000) for the reefs of Indonesia. We feel this method is very appropriate not only for these islands but also other reefs in India. This methodology helps to bridge the formidable gap that exists between reef scientists and reef managers (Risk & Risk, 1997).

Almost all studies in India on coral reefs are evaluated according to a linear scale of coral cover such that only those reefs with >75% live coral cover are considered to be in excellent condition. Reefs with 50–75% live coral cover are considered in ‘good’ condition; reefs with 25–50% live coral cover in ‘fair’ condition; and reefs with <25% live coral cover, in ‘poor’ condition. This use of coral cover to define the condition of reefs ignores the roles of disturbance, competition and stress in regulating coral species diversity (e.g. Huston, 1994). Sites with very high percentage of live coral cover are frequently composed of large mono-specific stands of corals such that their coral biodiversity is low (Connell, 1978; Aronson

Table 1. Study sites in Andaman and Nicobar Islands

Site	Code	Reef geomorphology	Depth	Visibility Hori./Vert.	Sources of Stresses
South Andaman					
Jolly Buoys	JB	Fringing reef/wall	4	16/19	Tourism, sedimentation, agricultural run off, storm damage
Twins	TW	Fringing reef	5	13/16	Storm damage, illegal fishing
Grub	GR	Fringing reef	4	12/14	Agricultural run off, sedimentation
Redskin	RS	Fringing reef/ Mangrove fringing reef	5	8/12	Tourism, agricultural run off, sedimentation
Chiriyatapu	CT	Fringing reef	3	11/15	Tourism, sedimentation
Pig head	PH	Fringing reef	6	15/18	Storm damage, sedimentation
Cinque	CQ	Fringing reef	7	17/22	Storm damage, sedimentation
North Bay	NB	Fringing reef	4	12/16	Oil Pollution, tourism, over fishing, sedimentation
John Lawrence	JL	Mangrove fringing reef	6	10/14	Agricultural run off, sedimentation, over fishing
Henry Lawrence	HL	Mangrove fringing reef	5	8/11	Agricultural run off, sedimentation, over fishing
Outram	OT	Fringing reef	6	7/10	Storm damage, agricultural run off, sedimentation, over fishing
Havelock	HK	Fringing reef	8	6/ 9	Tourism, agricultural run off, sedimentation, over fishing
Neil	NL	Fringing reef	6	6/ 8	Destructive fishing, agricultural run off, sedimentation, over fishing
Middle Andaman					
North Reef	NR	Fringing reef	6	17/23	Storm damage
Interview	IR	Mangrove fringing reef	4	8/13	Run off from mangroves
South Reef	SR	Fringing reef	6	9/11	Run off from mangroves
Sound	SN	Fringing reef	6	10/13	Over fishing, storms, sedimentation
Aves	AV	Fringing reef	5	8/11	Tourism, over fishing, storms, sedimentation
North Andaman					
Smith	SM	Fringing reef	7	9/13	Sedimentation, over fishing, tourism
Ross	RO	Patchy reef	5	8/11	Sedimentation
Lamia Bay	LB	Patchy reef	7	6/ 8	Sedimentation
Nicobar group					
Car Nicobar	CN	Fringing reef	9	19/24	Sedimentation
Nancowry	NC	Fringing reef	8	8/12	Sedimentation
Campbell Bay	CB	Fringing reef	6	10/14	Sedimentation

& Pretcht, 1995; Kulkarni *et al.*, 2000). It also ignores the role of habitat complexity in supporting reef fish diversity (Edinger & Risk, 2000). Because sites with >75% live coral cover are frequently composed of monospecific stands, they have relatively low spatial complexity, and provide relatively poor fish habitats (Roberts & Ormond, 1987).

Disturbance, competition and stress are three primary factors controlling diversity and abundance of plants and animals in natural communities (Huston, 1994). Grime (1979) proposed the response to these three factors as the primary adaptive strategies in land plant: ruderals (disturbance adapted), competitors, and stress-tolerators. Stress tolerators tend to be slow growing organisms that are able to survive in nearly all habitats, but only dominate in habitats where physiological stress precludes or slows the growth of ruderals and competitors, and stress tolerators. Grime further showed that different plant communities could be characterized by the relative importance of ruderals, competitors, and stress tolerators within them, displayed graphically using r-K-S ternary diagrams.

Ternary diagrams are triangles reflecting composition, in which each corner represents 100% make-up by one subset, and position within the relative importance of those three subsets. The relative importance of these three groups changed through plant succession, and ternery diagrams of plant adaptive strategies can be used in vegetation management (Grime, 1979).

Edinger & Risk (2000) demonstrated successfully that using coral morphology triangles, coral reef's conservational values could be estimated for the better management of Indonesian coral reefs. The assignment of morphologies and conservation classes in this report is appropriate to the coral morphologies and threats to coral reefs in Indonesia (Edinger & Risk, 2000) as well as in Andaman and Nicobar Islands, where land based pollution is a primary threat to coral reef biodiversity (Chou, 1997; Edinger *et al.*, 1998; Kulkarni *et al.*, 2000).

Coral Morphology Triangles

We have classified reef communities using r-K-S ternary diagrams (Fig. 2) according to the relative abundance of

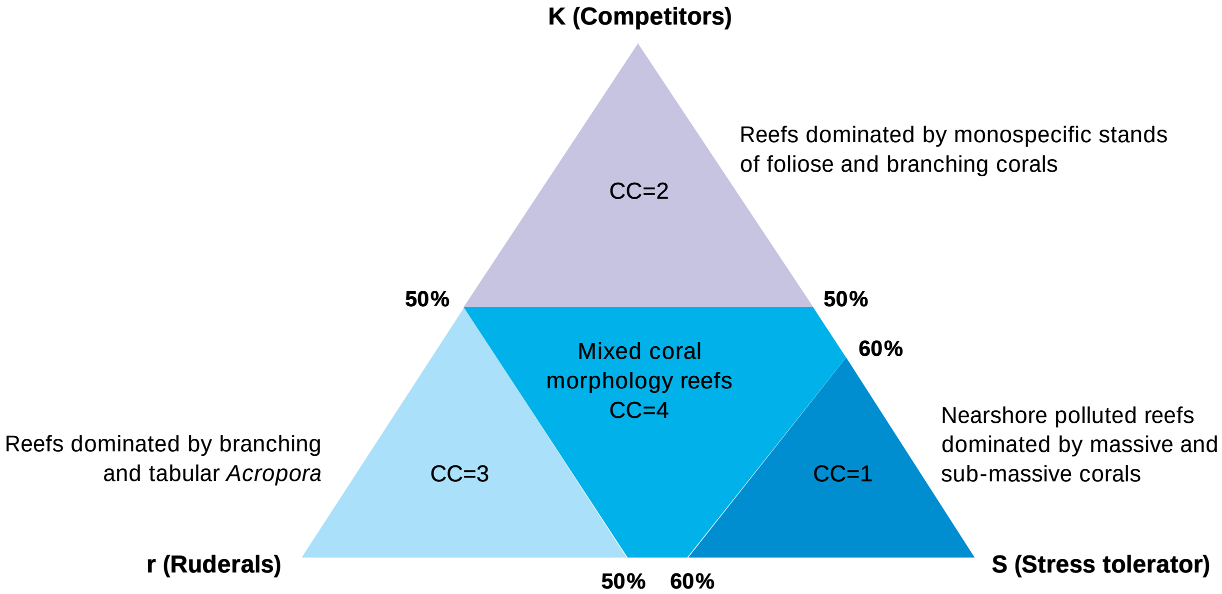

Figure 2. Idealised r-K-S ternary diagram for coral reef conservation classes (after Edinger & Risk, 2000)

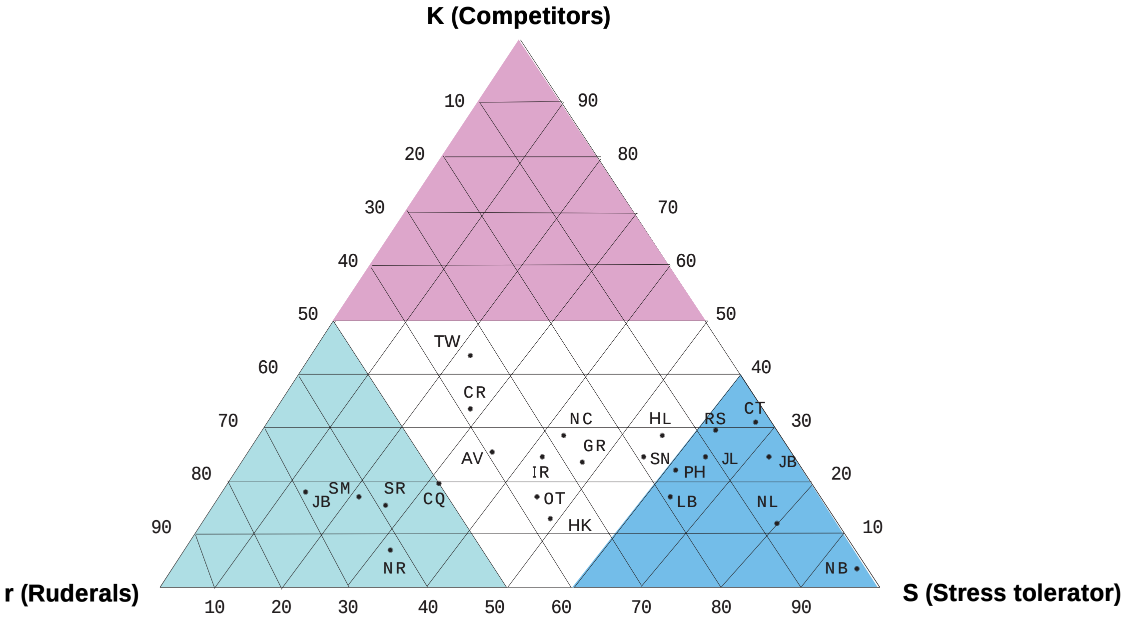

Figure 3. All study sites plotted on on r-K-S ternary diagram (n=22). Some sites are strongly affected of disturbance (NB, SM, NL, LB, PH, JL, RS, CT, JB) are dominated by stress tolerates and receive conservation class CC=1. Sites such as CB, SR, NR and CQ have higher proportion of *Acropora* and are classed as CC=3. Sites (TW, CN, AV, IR, OT, GR, NC, HL, SN) have an equal proportion of *Acropora*, non-*Acropora* competitors ; and classed as CC=4. No sites felled in CC=3, where proportion of competitors is high.

standardized coral morphology categories (Table 2 on next page). Here we have defined *Acropora* corals as disturbance-adapted ruderals (r), due to their rapid growth and mechanical fragility (Done, 1982; Karlson & Hurd, 1993; Kulkarni *et al.*, 2000; Edinger & Risk, 2000). Branching non-*Acropora* corals and foliose corals which grow and recruit more slowly than *Acropora* are the competitive dominants on many Andaman and Nicobar Islands coral reefs protected from heavy wave exposure (Kulkarni *et al.*, 2000). We define these morphologies as competition adapted (K). Massive and submassive corals more tolerate of high sedimentation and/or eutrophication are defined as stress-tolerators (Veron, 1986; Rogers, 1990; Kulkarni *et al.*, 2000).

Reefs dominated (>60%) by stress-tolerators are assigned to conservation class I (Fig. 2). Reefs dominated (>50%) by competitors (foliose corals and branching non-*Acropora* corals) or ruderals (*Acropora* corals) are assigned to conservation classes 2 and 3 respectively. Reefs falling in the centre of the diagram where all three morphological groups are represented in approximately equal proportions are assigned to conservation class 4. S-dominated sites were defined as >60% massive and submassive corals, rather than 50% as for the other categories because stress tolerators are usually abundant in all communities but only dominate those subjected to environmental stress (Grime, 1979; Edinger & Risk, 2000), and in which *Acropora* corals, branching non-acropora and foliose corals are mostly excluded as a result of pollution or sedimentation (Pastroak & Bilyard, 1985; Rogers, 1990; Edinger & Risk, 2000).

Table 2. Coral morphology categories used for Coral Reef Classification

Acropora branching	ACB	Staghorn corals, long thin branches	r
Acropora bottlebrush	ACBB	Manly *A. echinata*	r
Acropora corymbose	ACC	Stout branches, low bushy shape	r
Acropora digitate	ACD	Digitate, Stubby, mainly *A. humilis*	r
Acropora tabular	ACT	Tabular, *A. hyacinthus*	r
Acropora submassive	ACS	Columnar, mainly *A. palifera*	r
Branching coral	CB	Branching non Acopora such as *P. cylindrical*, *P. nigrescens*, *Seriatopora hystrix*, *Stylophora pistallata*, *Pocillopora damicornis*, *P. eudoyxi, Hydnophora rigida, Echinopora gemmacea* and some other species	K
Encrusting coral	CE	Mainly *Porites rus* and *Montipora efflorescens*	K
Massive coral	CM	Massive corals such as *Porites* sp. *Favia* sp. and *Favites* sp.	S
Foliose coral	CF	Foliose mainly Montipora and Echinopora	K
Submassive coral	CSM	Multilobate or lumpy corals, sometimes columnar or mixed massive columnar especially *Gonipora*, *Galaxea*	S
Mushroom coral	CMR	Free living fungiid corals	K
Millepora	CME	Various species of *Millepora*	r
Heliopora	CHL	Blue coral	r

METHODOLOGY

Study Areas

Twenty-three sites were sampled on coral reefs in four regions of Andaman and Nicobar Islands: South Andaman (13 reefs), Middle Andaman (5 reefs), North Andaman (3 reefs) and Nicobar groups (3 reefs). Study sites and stresses are listed in Table 1. All these sites were sampled during November 2000 to April 2001. These sites were sampled between 3 and 9 meter depths.

Coral Community Structure

Sampling was conducted on the outer reef crest using line intercept transects. Five transects of 20 m were placed parallel to depth contour at each site. To account for intra-site variation, transects were separated by between 10 and 30 m, such that surveys were distributed over approximately 150 m to 250 m of each reef. The determination of coral species coverage and abundance was carried out using permanent line intercept transect (LIT). The intercepts of all underlying coral species, typology of dead and live coral (massive/submassive, branching/digitate, mushroom and foliose), soft coral, sponge, macro-algae and sand were recorded to the nearest 1 cm along each transect.

Coral Morphology Triangles

Morphological composition of the corals at each site was calculated from the percentage cover of each morphological category at each site. R-K-S ternary diagrams were constructed by summing the total cover of the various morphologies in each of the three adaptive strategy groups: ruderals (r), competitors (K), and stress-tolerators (S). Percentage cover values for each adaptive strategy group were calculated as a proportion of the total live coral cover recorded at each site, such that the sum of all three categories always equals 100%. The proximity of a reef site to any corner of the ternary diagram indicates the relative contribution of that adaptive strategy group.

Conservation Class Scores

Twenty-two sites were plotted on r-K-S ternary diagrams and conservation class was assigned to each site according to its location on the ternary diagram (Fig. 3). Sites dominated (>60%) by stress tolerators were assigned to conservation class 1; sites dominated (>50%) by K-strategists were assigned to conservation class 2, and sites dominated (>50%) by r-strategists were assigned to conservation class 3. Sites plotting on the centre of the diagram were assigned to conservation class 4. The conservation class of each reef was determined by calculating the average conservation class (1–4) of the various sites on that reef.

RESULTS AND DISCUSSION

Coral Community Structure of Each Group of Islands

Data obtained from each group of islands (Table 1) were pooled. The islands in South Andaman and North Andaman groups were dominated by *Porites* while the Middle Andaman and Nicobar groups of islands were dominated by *Acropora*.

Fifteen of the 23 sites exhibited good live coral cover while the live coral cover at the remaining eight sites was fair. The highest live coral cover was recorded at North Reef (73.49%), Car Nicobar (72.68%) and Outram (67.33%). In contrast to reports by Wilkinson *et al.* (1999) who reported 60 to 90% mortality of corals, these observations clearly indicate that the ENSO event of 1998 did not affect the coral reefs of Andaman and Nicobar Islands. The mortality index was high (>0.6) at Jolly Buoys, Neil, Pig Head and Red Skin. Almost all reefs were dominated either by *Acropora* or *Porites* except Twins Island where *Montipora* dominated. At almost all reefs, colonies of the encrusting *M. turgescens* were growing on the dead corals. In addition, in some areas, such as Grub Island, colonies of *Echinopora lamellosa* occupied large areas and were competing with slow growing species such as *Porites*. *Acropora* dominated the reefs of Aves, Cambell Bay, Car Nicobar, South reef, North Reef, Smith and Cinque Island suggesting that, because *Acropora* spp. are sensitive to any environmental changes (Kulkarni *et al.*, 2000), the reef environment in these areas seems to be in better condition.

Eight sites from two regions were of conservation class (CC) 1, seven from these were from South Andaman Islands. Sites such as Jolly Bouys, Redskin, and John Lawrence were located in marine national park areas. These sites were composed of stress tolerant species and were mainly dominated by *Porites*. None of the sites were in CC 2. Cambell Bay, North Reef, Smith, South Reef and Cinque were of CC 3. All these sites were dominated by ruderals (*Acropora* spp.) except Twins Island where *Millepora* dominated the reef. Twins Island, Car Nicobar, Aves, Interview, Outram, Havelock, Grub, Nancowry, Sound and Henry Lawrence were of CC 4.

The coral reefs that were designated as CC 1 reefs were severely affected by sedimentation and the impacts of tourism. Reefs in this class will probably not recover until anthropogenic impacts are removed. According to the concept of Triage (Risk, 1994), remediation and protection efforts should focus on reefs in better health (CC 3 and 4), where these efforts are more likely to succeed. Mechanically damaged reefs, such as reef in Jolly Buoys which is severely affected by tourism, that are not affected by sedimentation may be able to recover from damage by protection.

There was no direct correlation between diversity and coral reef health. Earlier studies by Bak & Luckurst (1978) and Porter *et al.* (1981) demonstrated that changes in reef structure could occur even in the absence of stresses. This study was unable to determine changes in community structure because of methodological inconsistencies with previous studies. However, the present study provides the basic information with which the results of further long term monitoring can be compared. Comparison with the earlier studies of Dorairaj *et al.* (1997) and Arthur (1996), which estimated live coral cover, clearly indicates a decline in the condition of these reefs. Future monitoring will help researchers and managers to predict the changes in this fragile ecosystem, particularly if monitoring is expanded to include more sites and different zones at each site. The results of this study indicates that the reefs of Andaman and Nicobar are suffering from anthropogenic stresses but that it would be premature to implicate *El Niño*.

Studies and observation of other reefs are beginning to suggest that the species assemblages are not stable equilibriums as was originally thought (Connell, 1978; Woodley, *et al.*, 1981). Coral reefs can change rapidly as a result of storms (Woodely *et al.*, 1981), predation (Endean, 1976; Wood,1989), sedimentation (McCook, 1999; Umar *et al.*, 1998; McClanhahn & Obura, 1998), anchor damage and blast fishing. The shallow reef is changing as the dominance of *Acropora* is drastically reduced by the increased abundance of *Porites*.

Generally, coral reefs have the ability to grow rapidly and re-colonise large areas of destroyed reef in a relatively short period, but this does not seem to be happening on the reefs within Mahatma Gandhi Marine National Park (MGMNP). Moreover, whereas intermediate levels of disturbance may promote species diversity in some cases,

Table 3. The relative abundance of each category in transects (Data of five transects were summed and averaged for each site)

Site code	Live coral	Dead coral	Total coral cover	Rubble	Sand	Algae	Soft corals	Others
AV	62.32	23.86	86.18	4.72	7.11	1.14	0.59	0.26
CB	67.23	22.13	89.36	8.92	1.72	0	0	0
CN	72.68	20.41	93.09	2.74	3.81	0	0	0.36
CQ	61.2	10.11	71.31	7.03	18.23	2.11	1.32	0
CT	51.31	38.94	90.25	4.17	5.01	0.19	0.12	0.26
GR	57.66	36.29	93.95	1.73	3.78	0.28	0.16	0.1
HK	59.68	37.22	96.9	0	2.96	0.14	0	0
HL	45.13	51.23	96.36	0.56	0	0.61	2.04	0.43
IR	67.73	24.59	92.32	1.06	3.74	2.77	0	0.11
JB	37.36	55.37	92.73	4.13	2.56	0	0	0.58
JL	65.46	14.21	79.67	4.56	12.11	1.89	1.77	0
LB	41.08	40.86	81.94	11.76	5.81	0.49	0	0
NB	44.47	43.87	88.34	2.86	6.92	1.11	0.77	0
NC	60.74	35.44	96.18	1.03	2.08	0	0.71	0
NL	27.82	55.87	83.69	6.39	8.22	1.01	0.69	0
NR	73.49	14.21	87.7	2.06	5.95	1.79	2.5	0
OT	67.33	28.12	95.45	2.38	1.56	0.61	0	0
PH	26.94	60.21	87.15	8.24	3.56	0	1.05	0
RS	32.6	48.51	81.11	9.16	8.41	1.03	0.29	0
SM	53.98	29.87	83.85	9.32	4.11	2.53	0	0.19
SN	46.64	46.17	92.81	2.79	4.4	0	0	0
SR	63.2	23.81	87.01	7.11	5.11	0	0.77	0
TW	35.6	49.26	84.86	8.13	6.34	0.29	0	0.38

Table 4. Mortality index, relative composition of stress tolerators = s; ruderals = r; competitors = k; species diversity index (H'); evenness (E') and conservation class value of each site

Region	Site	Live coral (%)	Condition index	*MI* __	r ___	K ___	S ___	H' __	E' __	CC
SA	AV	62.32 ± 10.29	Good	0.28	40.21	25.29	34.5	2.44	0.74	4
N	CB	67.23 ± 16.04	Good	0.25	67.06	18.25	14.69	2.3	0.54	4
N	CN	72.68 ± 13.82	Good	0.22	39.22	33.35	27.43	2.24	0.72	4
SA	CQ	61.2 ± 14.63	Good	0.14	50.34	18.54	31.12	2.79	0.73	3
SA	CT	51.31 ± 6.15	Good	0.43	3.35	30.65	66	1.83	0.61	1
SA	GR	57.66 ± 16.48	Good	0.39	27.49	23.98	48.52	2.37	0.84	4
SA	HK	59.68 ± 6.73	Good	0.38	38.4	10.89	50.71	2.09	0.65	4
SA	HL	45.13 ± 18.23	Fair	0.53	15.37	25.79	58.84	2.37	0.64	2
MA	IR	67.73 ± 11.36	Good	0.27	33.91	24.4	41.69	2.63	0.82	4
SA	JB	37.36 ± 11.50	Fair	0.6	5.69	23.21	71.11	2.04	0.64	1
SA	JL	65.46 ± 15.05	Good	0.18	12.01	25.11	62.88	1.81	0.61	1
NA	LB	41.08 ± 8.35	Good	0.5	18.76	18.24	63.1	1.89	0.49	1
SA	NB	44.47 ± 14.24	Fair	0.5	1.16	3.67	95.17	1.13	0.33	1
N	NC	60.74 ± 11.05	Good	0.37	27.5	28.72	43.78	2.81	0.82	4
SA	NL	27.82 ± 13.17	Fair	0.67	7.81	12.9	79.29	1.34	0.49	1
MA	NR	73.49 ± 17.88	Good	0.16	60.91	6.9	33.19	2.12	0.79	3
SA	OT	67.33 ± 16.73	Good	0.29	35.19	19.86	44.96	2.88	0.75	4
SA	PH	26.94 ± 6.77	Fair	0.69	12.21	22.24	65.55	1.92	0.4	1
SA	RS	32.6 ± 12.65	Fair	0.6	8.21	28.33	63.46	2.6	0.48	1
NA	SM	53.98 ± 9.87	Good	0.36	60.58	19.21	20.21	2.38	0.41	3
MA	SN	46.64 ± 10.58	Fair	0.5	18.24	23.41	58.35	2.13	0.58	4
MA	SR	63.20 ± 9.15	Good	0.27	58.32	16.93	24.75	2.03	0.68	3
SA	TW	35.6 ± 10.11	Fair	0.58	35.53	41.12	24.14	2.01	0.52	4

the magnitude of disturbance on MGMNP seems to be such that it is inhibiting re-colonisation of corals and diversity is, in fact, declining. This observation must be taken into account when trying to assess the vitality of coral reefs and constructing management plans for their preservation.

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The Coral Reefs of the Tuticorin Coast of the Gulf of Mannar, South-East India

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Key words: coral reefs, Tuticorin

INTRODUCTION

The Gulf of Mannar (GOM) is one of the four major coral reef areas of India, covering an area of approximately 10,500 sq. km from Tuticorin to Rameswaram. The area includes a chain of 20 coral islands surrounded by fringing reefs, and patch reefs rising from the shallow sea floor.

The area is predominantly a high-biodiversity coral reef ecosystem with 117 species of corals from 33 genera recorded from GOM and Palk Bay during 1971 and further the faunal list was consolidated in 1986 to only 94 species of 37 genera (Pillai, 1998). In addition, 13 species of mangrove (Perichiappan et.al., 1995), 147 species of seaweed (Kaliyaperumal, 1998), 13 species of seagrass (Rajeswari & Anand, 1998), 17 species of holothurian (James, 2001), 510 species of fish (Dorairaj, 1998) 106 species of crustacean (Jeyabaskaran & Ajmal Khan, 1998), four species of shrimp (Ramaiyan *et al.*, 1996) and four species of lobster (Susheelan, 1993) have been recorded from this area. At Krusadai Island alone, 450 species of mollusc were reported in 1952 (Satyamoorthi, 1952) and 306 species in 2000 (Unpublished TMMP report, 2000).

Large numbers of traditional fishermen from Rameswaram Island and the mainland use the reefs as fishing grounds. In 1982, the fishery production, including pelagic fish, in the area was 2375 tonnes and in 1983, it was 2150 tonnes (Venkataramanujam & Santhanam, 1985). Further, molluscs, holothurians and algae are harvested in large quantities. Also, dynamite fishing and coral mining still occurs, although enforcement efforts by the conservation authority of Gulf of Mannar Marine Biosphere Reserve (GOMMBR) have curtailed these activities considerably. The southern-most part of GOMMBR, the Tuticorin Coast, currently consists of four islands. Originally, five islands were present in this area but as a result of soil erosion caused by excessive coral mining, Villanguchalli now lies 30 cm below mean low water level. There are five fishing villages along the Tuticorin Coast namely, Thirespuram, Siluvaipatti, Vellapatti, Tharuvaikulam and Puthu Kadarkarai. About 35,000 registered fishermen from these villages depend solely on fishing for their livelihood.

CORAL REEF MANAGEMENT ISSUES IN TUTICORIN COAST

Tuticorin Coast is the most environmentally stressed of the coastal areas in GOM due to human activities such as destructive fishing, coral mining and pollution from the industries along the coastal belt. Although illegal, coral mining for the lime industry, for road constructions and for ornamental purposes has been practiced for several decades and involves many poor fishermen who depend on it for their daily livelihood. The number

Figure 1. Fixing of coral fragments on stones and cement slabs.

of boats involved in mining varies seasonally with the highest number involved during the lean fishing season (mid-May to mid-August). Usually, fishermen will mine corals four times per week during the low tide and each boat extracts an average of 1.5 tons, which is comprised of about 98% dead corals and 2% live corals. However, since coral mining is an illegal practice, the fishermen are reluctant to reveal the actual information about the quantity of coral they are quarrying.

Cyanide fishing is used to catch reef fishes, particularly groupers that fetch high market prices, and ornamental fishes like clownfishes (*Amphiprion* spp.), dottybacks (Pseudochromidae), damsels (Pomacentridae), and surgeons (Acanthuridae). Further, various types of destructive fishing nets such as beach seine nets and trawl nets, are damaging the benthic environment at Tuticorin. Further, a small section of fishermen are also involved in dynamite fishing using sticks of gelignite to blow up the reef and kill shoaling fishes. However, strict regulations are implemented to prevent this type of fishing. The Ramanathapuram district administration has banned dynamite fishing in the Palk Strait and the Gulf of Mannar and those using explosives to catch fish would be booked under the stringent Goondas Act.

Pollution is another major problem affecting the Tuticorin Coast due the discharge of untreated and partially treated effluents from industries like Tuticorin Alkali Chemicals and Fertilizers Ltd. (TAC), Southern Petrochemical Industries Corporation Limited (SPIC), Tuticorin Thermal Power Station (TTPS), Heavy Water Plant (HWP), Dharangadhara Chemical Works Limited (DCW), Sterlite Copper etc. which are located in the coastal belt. In addition, the domestic sewage and pollution from port exacerbate the problems caused by industrial effluents to the coastal ecosystem.

More than 69% of the people living along the Tuticorin Coast live below the poverty line and, as a consequence, many rely on illegal coral mining and exploitation of natural resources for their daily livelihood. The literacy rate is very low and local people have a poor knowledge of the effects of unsustainable resource extraction on the coral reefs.

Although there are several research initiatives underway, and the conservation authorities have been very active with their limited facilities, no considerable larger scale contribution to coral reef research has been made at Tuticorin Coast after the 1980´s. As a consequence, up to date information describing the status of the coral reefs along the Tuticorin Coast and their resources is scarce. However, the effects of coral mining and destructive fishing have clearly degraded the reefs in many parts of the area, and the reef ecosystem is under immediate threat.

PILOT SCALE COMMUNITY BASED REEF RESTORATION AND CAPACITY BUILDING IN TUTICORIN COAST

To date, coral reef restoration has not been tried in India. Although restoration programs are successful only at a smaller scale it also provides an opportunity to involve local people in conservation and management, and for raising of awareness. One of the new initiatives that has recently started in Tuticorin is a pilot project focusing on community based coral reef restoration and capacity building. The project is conducted by Suganthi Devadason Marine Research Institute (SDMRI) with support from the CORDIO Programme and will include:

- Transplantation of branching coral fragments for restoration of the reefs on a small scale, involving the local fisher community;
- Monitoring, growth and survival, and recruitment studies, and maintenance of the transplanted corals;
- Creation of awareness among fisher folk along the Tuticorin Coast of the value and services provided by the coral reefs;
- Capacity building among fisher folk, with focus on women, through training in alternative livelihoods;
- Capacity building through Ph.D programmes.

The work within this CORDIO project was started in March 2002 and, as a first step, a preliminary survey was made in order to determine the current condition of coral reefs along the Tuticorin Coast. Experimental restoration studies using various coral species have also been initiated.

STATUS OF THE REEFS IN TUTICORIN COAST (PRELIMINARY SURVEY)

In the present preliminary survey, the diversity and distribution of corals along the Tuticorin Coast were assessed mainly around the islands (up to 300 m from the low water mark) and along the coast of the mainland. Surveys were carried out for four months by visual observation through snorkeling in the shallow area and SCUBA diving in deeper areas. Live corals were distributed around the Van, Koswari, Kariyachalli and submerged Villanguchalli islands. Live corals were also recorded at several areas along the coast of the mainland, particularly in the protected areas coming under the control of Tuticorin Port Trust. Both solitary and branching corals occurred in these areas and many new recruits were observed at all sites. Pillai (1972) reported of 21 species of hard coral from Tuticorin. However, there is little coincidence in the species composition reported by Pillai (1972) and that which was recorded in this study.

For example, in the genus *Acropora*, Pillai (1972) reported only two species, *Acropora formosa* and *A. corymbosa*, while in the present survey, *Acropora* was the domi-

Figure 2. *Acropora nobilis* cultured (one month) in clay cups.

Figure 3. Twelve months cultured coral fragments of *Acropora humilis* and *A. nobilis*.

nant genus with nine different species being recorded (Table 1).

In total, 22 species of coral were recorded belonging to the genera *Acropora*, *Montipora*, *Porites*, *Favia*, *Turbinaria*, *Hydnophora*, *Goniopora*, *Goniastrea*, and *Cycloseris* (Table 1). Around the islands, live corals were sparsely distributed, while on the fringing reefs of the mainland lush coral growth was observed. Branching *Acropora* and *Montipora* were dominant at both the island and mainland sites. Of the nine species of *Acropora* that were recorded, three species (*A. stoddarti*, *A. hyacinthus*, *A. valida*) were confined to island area. The only common acroporid species observed in both areas was *Acropora formosa*. The

Table 1. List of species of hard corals recorded along the Tuticorin Coast

Family: Acroporidae
Genus: *Acropora*
1. *Acropora formosa*
2. *A. nobilis*
3. *A. humilis*
4. *A. valenciennesi*
5. *A. stoddarti*
6. *A. cytherea*
7. *A. hyacinthus*
8. *A. valida*
9. *Acropora* spp.(unidentified)

Genus: *Montipora*
10. *Montipora digitata*
11. *M. foliosa*

Family: Poritidae
Genus: *Porites*
12. *Porites* spp.
13. *Porites lutea*

Genus: *Goniopora*
14. *Goniopora* spp.(unidentified)

Family: Faviidae
Genus: *Favia*
15. *Favia favus*
16. *F. pallida*

Family: Dendrophylliidae
Genus: *Turbinaria*
17. *Turbinaria crater*
18. *T. peltata*

Family: Faviidae
Genus: *Hydnophora*
19. *Hydnophora microconos*

Genus: *Goniastrea*
20. *Goniastrea pectinata*
21. *G. rectiformis*

Family: Fungiidae
Genus: *Cycloseries*
22. *Cycloseris cyclolites*

Figure 4. Lush growth of corals inside the protected area.

presence of suitable substrate is critical to successful recruitment (Westmacott *et al.*, 2000) and many new recruits were detected on the dead corals in both areas. The continuing illegal quarrying of the dead corals in the Tuticorin region definitely reduces the survival of new recruits.

Luxurious growth of *Acropora* spp. was observed along the mainland, especially in the protected areas inside the port where there is less human disturbance (Fig. 4). The live coral cover in this area is about 80%. Also, recruitment of colonies of *Acropora*, *Porites*, *Turbinaria* and *Goniastrea* was observed in this area. However, in the unprotected areas, live corals were sparsely distributed and the coral cover was less than 5%. The species recorded in these areas were *Porites* spp., *Turbinaria crater, Goniastrea* spp. and *Cycloseris cyclolites.* The cover of live coral around the islands was approximately 20% and dominated colonies of *Montipora digitata* and *Acropora* spp.

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Status of the Coral Reefs of Maldives

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Key words: bleaching, coral reefs, Maldives

INTRODUCTION

Conservation, sustainable use and management of coral reef resources are among the national priorities in the Maldives, the country's economy being dependent on the health and function of the coral reefs. Degradation of coral reef resources because of natural and/or human impacts has been felt more frequently and intensively in recent years. This paper reports on the national coral reef monitoring programme that began after the severe bleaching in 1998 that lead to high mortality of the reefs in the Maldives. Monitoring of selected sites was conducted in 1998, 1999, 2000, and 2002 to assess the status of the coral reefs and the recovery processes after the mass bleaching and mortality of corals in 1998. The aim of this monitoring exercise is 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.

METHODS AND SURVEY LOCATIONS

Site Selection

The reef area of Maldives is enormous and the resources available for monitoring it are small. The sampling sites were chosen in the following regions (Fig. 1 on next page):

- *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).

In each of these regions three reefs were selected, including reefs that had been surveyed in the recent past and thus providing baseline data against which monitoring data could be compared.

All of the quantitative surveys were conducted on the reef tops, since this has been the case for virtually all of the previous studies, and for logistical efficiency. 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. Oc-

casionally, when it was judged effic[redacted], a 50 m
long line point insect transect was ot study
was done in 1998, and surveys were the same
sites in 1999 and 2000 as part c ing pro-
gramme to provide an insight to s of reef
recovery especially after the bleach After a
data gap of one year (2001) reef ese sites
resumed during 2002.

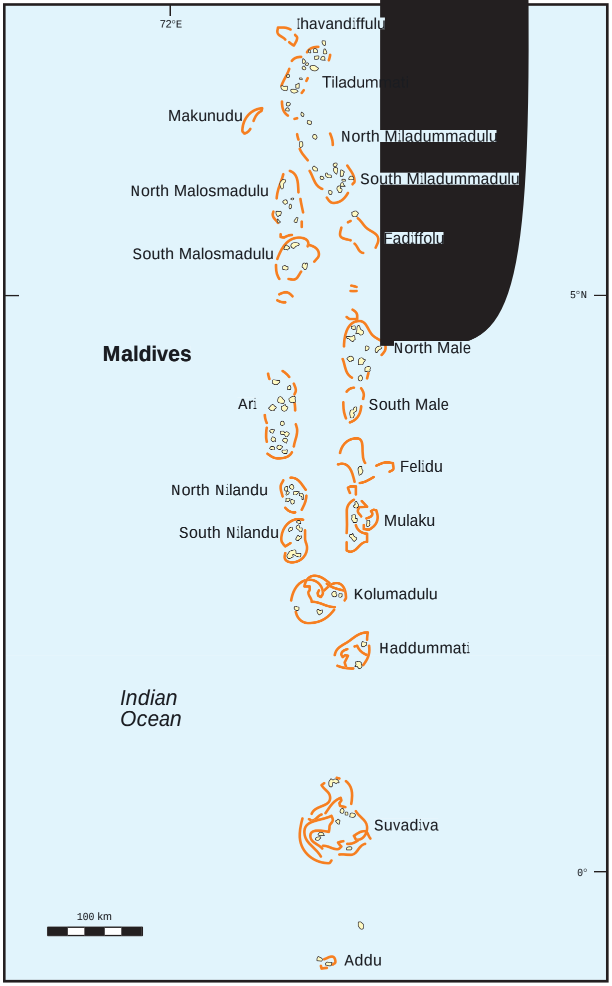

Figure 1. The reef area of Maldives with each survey site illustrated.

RESULTS

A summary of post-bleaching data in 1998 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) (Fig. 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 poritiids and agariciids, despite suffering

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 1	2	3	Aggregate
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				

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									

high mortality, survived the mass bleaching best. Results from the 1999 surveys showed a mean cover of live coral of 1.9% ranging between 0.33% and 3.04% among the atolls surveyed (Table 4 on next page). The results from the 1998, 1999, 2000, and 2002 surveys are shown in Fig. 3 (next page). Recovery is slow but evident at all regions, although variable between regions. Reef recovery (live coral cover) has been recorded highest in Addu followed by Ari region but highly variable within regions (e.g. large standard deviations).

Additional information on the reef status was also collected as part of the 2002 surveys to compare reef recovery at deeper reef areas. Surveys at several deeper locations in Addu region indicated a relatively higher live coral cover compared to shallow reef areas surveyed in that atoll (Fig. 4 on next page). Coral cover ranged from 41–54% on the reef slope (7–10 m) and 52% at 3 m (atoll outer reef). Villingili (1 m) and Villingili (10 m) are directly

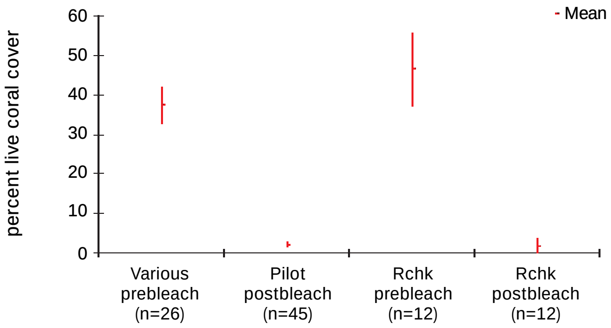

Figure 2. Estimates of live hard coral cover prior to and after the bleaching event plotted as mean bounded by upper and lower 95% confidence intervals on the mean. Pilot post-bleaching data from pilot project field work. Various pre-bleaching data from Coral Reef Research Unit, Riyaz et al. (1998).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

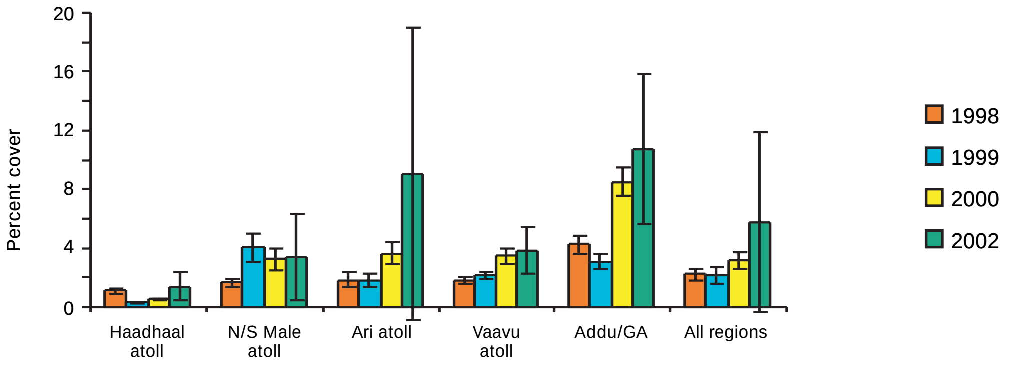

Figure 3. Comparison of estimates of the cover of live coral obtained immediately following the bleaching event (1998) and yearly surveys since then to present (no data for 2001). Values are mean coral cover and the error bars are standard deviation of the means.

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

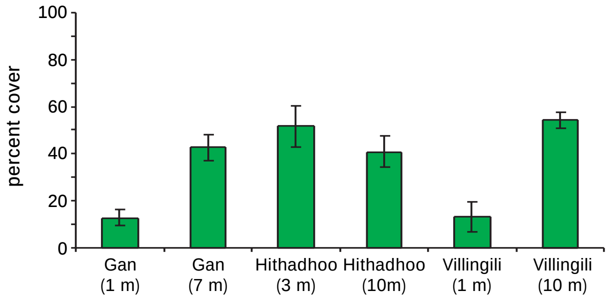

Figure 4. Reef surveys at Addu atoll in 2002, comparing the coral cover between shallow and deeper sites. Values plotted are means and error bars are standard deviation of the means.

comparable sites in Addu atoll clearly demonstrating the different levels of coral cover at this region (Fig. 4).

DISCUSSION

The post bleaching study data from 1998 show that only a small amount of live coral cover remained on the reef tops surveyed (Table 1). 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 was 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

Figure 5. Large colonies of *Acropora* and *Porites* at Villingili (10 meters depth).

(Tables 2 and 3). 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 in 1999 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.

However, the survey team has observed new coral recruits at all sites. Re-colonisation of fast growing branching growth forms where 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 belong to the genus *Acropora* which was the genus most seriously affected by the bleaching in 1998. In addition, on some reefs encrusting coralline algae are abundant providing potential areas for coral settlement and recruitment and in some regions (e.g. Haa Dhaal) large *Acropora* tables that were believed to be dead are regenerating live tissue indicating prolonged recovery of some species of coral. Reef recovery has been reported to be highest in Addu atoll compared to other regions.

Figure 6. Reef survey location at Hithadhoo (10 meter depth). Large acroporids are common indicating less impact compared to shallow areas.

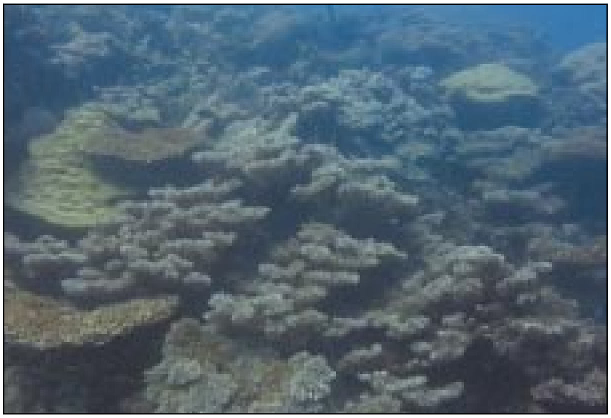

Figure 7. Tabulate acroporids in Hithadhoo (shallow, 3 meter depth) survey location, the most severely affected coral genus from the 1998 bleaching.

Comparison of the reef recovery at Addu region clearly demonstrated highly variable conditions at the surveyed sites. This high coral cover compared to other regions indicated that the severity of the impact of bleaching in 1998 was not as widespread as it has been previously thought (Fig. 5–7). There was considerable evidence that the reefs in Addu were less affected compared to other regions surveyed in the country. Large colonies of acroporids were common in these reefs that have clearly survived and outlived the bleaching impact. More interestingly, corals of the genus *Seriotopora* and *Stylopora* were common in both atoll outer reef (Hithadhoo 3 m) and deeper reefs (Hithadhoo and Villingili) which were not recorded elsewhere during the past 4 years of reef surveys as part of this long term reef monitoring program since the bleaching in 1998.

On average, coral cover increased approximately 3% from 1998 to 2002 (see Fig. 3). However, it is evident that the rate of recovery is variable among and between sites, demonstrating the complexity of the reef recovery processes. Therefore, 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 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 ($CaCO_3$) fragments. Where reef disturbance is severe, boring and grazing organisms may remove $CaCO_3$ faster than primary frame-builders can add to it. Such biogenic processes will determine whether the integrity of the reef structure will be compromised.

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Spatial and Temporal Patterns of Coral Recruitment Following a Severe Bleaching Event in the Maldives

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Key words: coral reefs, Maldives, recruitment

ABSTRACT

Reef recovery processes were investigated at shallow depths (between 5 and 10 m) within two geographically distinct atolls in the Maldives archipelago from February 2000 to July 2002. Artificial settlement plates were used to determine potential availability of coral spat on settlement plates and repeat censuses of small (0.25 m^2) permanently marked quadrats were used to investigate spatial and temporal recruitment patterns over a period of two and half years. The number of spat settling on the settlement plates after approximately 8 and 12 months submergence was low, and densities ranged from 2.5 to 4.27 spat/m^2. There was no difference in spat density between cryptic and exposed surfaces however, there were significant differences between sites (Kruskal-Wallis, $p=0.02$ and $p<0.001$ for 8 months and 12 months respectively). The dominant coral settling on the plates was *Pavona* spp., which accounted for over 34% of the total. Recruit densities and size frequency distributions for one site in Malé atoll were consistent between years. Density of recruits was high ranging from 30 to 49/m^2 and did not differ between years. The survivorship of recruits between each census was high (>80%) and this combined with a consistent supply of new recruits meant that recruit turnover was low. The most common genus in the recruit and juvenile populations was *Pavona* spp., a finding consistent with the adult population. Whilst density and size frequency data confirm that reef recovery processes are well underway the population at Feydhoo fonolhu is still dominated by one genus, *Pavona* spp. indicating that the coral community structure may take several decades to recover to the pre-bleaching level. Recovery of branching corals belonging to Acroporidae and Pocilloporidae appears to be limited by the impoverished adult population. Based on our observations to date, we conclude that longer time-scales are required to determine whether the temporal patterns demonstrated in this study are typical or whether there are occasionally years with a high or low influx of new recruits.

INTRODUCTION

Coral reefs in the Maldives were severely impacted by the 1998 coral bleaching event, with subsequent bleaching-induced mortality reducing coral cover from about 42% to 2% (Zahir, 2000). In cases where coral mortality is severe reef recovery is largely dependent upon factors that influence coral settlement and recruitment. However, the survival of fragments and asexual reproduction can be important in some systems (Pearson, 1981). Various studies have demonstrated that coral recruitment in open systems is highly variable in space and time (Smith, 1992; Connell *et al.*, 1997; Hughes *et al.*, 1999) indicating the need for studies on recovery processes at several scales. There have been several studies on patterns of coral recruitment in the Atlantic and Pacific reefs but there are few published studies on coral recruitment in the Maldives or Indian Ocean in general. A study on coral recruitment patterns on artificial reef structures at a shallow reef-flat site (<2 m) in North Malé atoll found an average of 12.6 recruits/m^2 between 1991 and 1994 (Clark & Edwards, 1999). The community was dominated by acroporids and pocilliporids with few massive corals (<2%) present. However, a survey on the same structures, conducted 11 months after the bleaching event, established that recruit densities varied from 0.6 to 2.8 recruits/m^2 and the overall ratio of branching to massive corals was 69:33, which represents a shift in the taxonomic patterns compared to the pre-bleaching coral community. It was concluded that despite the severity of the bleaching-induced mortality there were viable supplies of coral larvae from upstream or deeper reefs to recolonise the degraded reefs.

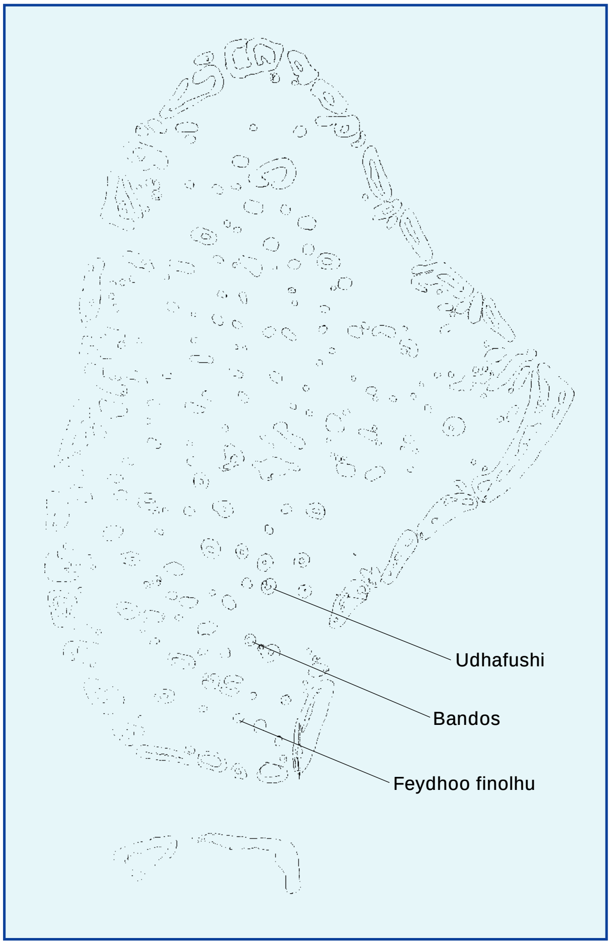

Figure 1. The three sites in North Male Atoll: Bandos island reef, Udhafushi reef and Feydhoo finolhu reef.

In 1999, an investigation funded by the CORDIO programme was initiated to determine the spatial and temporal patterns of coral settlement and recruitment at six reefs within two central atolls, North Malé atoll and Vaavu atoll. Integrated studies combined the use of ceramic tiles to investigate the potential availability of coral spat on settlement plates with in-situ studies of recruitment to the natural reef. In February 2000 a preliminary assessment on coral recruitment (approximately 21 months after the bleaching event) at Feydhoo finolhu in Malé atoll found that recruit density ranged between 19–26 individuals/m^2 at 10 and 5 m respectively (Clark, 2000) suggesting that recovery is not recruitment limited. This report presents the results of the research and monitoring that has been continued from the preliminary studies. The aims of this study were to:

- determine the spatial and temporal patterns of coral spat settlement and recruitment;
- investigate the relationship between availability of larvae and in-situ recruitment;
- determine sucessional processes of sessile communities to natural and artificial substrates.

MATERIALS AND METHODS

Sites

A hierarchical sampling design was adopted with 3 random sites (reefs) within each location (atoll) nested within two geographically distant atolls (70–80 Kms). Three sites were selected in North Male Atoll (Fig. 1): Bandos island reef, Udhafushi reef and Feydhoo finolhu reef. An additional three sites were selected in Vaavu atoll (Fig. 2): Wattaru reef, Kuda Ambaraa reef and Foththeyo reef. These sites are also part of a long-term reef monitoring study using line intercept transects to document spatial and temporal changes in benthic communities since the bleaching in 1998.

Figure 2. The three sites in Vaavu atoll: Wattaru reef, Kuda Ambaraa reef and Foththeyo reef.

Coral Spat Settlement

To investigate potential larval availability artificial settlement tiles were deployed directly onto the reef and exposed for periods between 160 and 395 days. An overlapping design was adopted so that tiles with suitably conditioned surfaces were always available. Deployment and retrieval dates for both atolls are shown in Table 1 (next page). Ceramic bathroom tiles were selected because they are locally available and cheap and have been shown to be a suitable surface for coral spat settlement (Harriot & Fisk, 1987). The tiles were 20 cm x 20 cm and 5 mm thick and had a central hole drilled to facilitate attachment to the reef. For each pair of tiles a cryptic push-mount plug was fixed into a solid section of the reef framework, and each tile pair was attached directly to the mount using two plastic cable ties. This method of attachment allowed tiles to be fixed in various orientations, facilitated the removal of tiles and has been shown to be very robust (Mundy, 2000). Twenty pairs of tiles (with the unglazed surfaces facing out) were deployed at each site, within the upper reef-slope between 5–10 m depth. After retrieval from the field, a point count method was used to determine the benthic sessile community on the tiles. Tiles were then bleached and examined microscopically for the presence of coral spat.

Coral Recruitment

Coral recruitment was assessed using small permanently marked quadrats 50 x 50 cm (0.25 m^2) on hard substrates at 2 depths (5 m and 10 m) on the upper reef slope. The quadrats were divided into twenty-five grid squares, each 5 x 5 cm, to enable observers to closely scrutinise the reef surface for recruits and small juveniles. Care was taken to distinguish between recently settled recruits (with intact

Table 1. The sampling protocol for the investigation of coral spat settlement using an overlapping design. Settlement tiles were deployed and retrieved between December 1999 and April 2001 at three sites within two atolls (Male and Vaavu atoll)

	Deployment date	Retrieval date	No. days	No. months
Male atoll				
Feydhoo finolhu				
Sample 1	07-Dec-99	Apr-00	137	5
Sample 2	07-Dec-99	Aug-00	245	8
Sample 3	Apr-00	Mar-01	331	11
Bandos				
Sample 1	13-Dec-99	12-Apr-00	121	4
Sample 2	13-Dec-99	11-Sep-00	273	9
Sample 3	21-Mar-01	Mar-01	343	11
Udhafushi				
Sample 1	12-Dec-99	10-Apr-00	120	4
Sample 2	12-Dec-99	13-Sep-00	276	9
Sample 3	Apr-00	Mar-01	349	12
Vaavu atoll				
Ambaraa				
Sample 1	09-Mar-00	16-Aug-00	160	5
Sample 2	09-Mar-00	Apr-01	392	13
Wattaru				
Sample 1	10-Mar-00	Aug-00	160	5
Sample 2	10-Mar-00	04-Apr-01	390	13

margins) and small colonies as a result of partial mortality or fragmentation. Here we define a recruit as an individual coral colony that can be detected by visual inspection and which has a maximum colony diameter of 50 mm. Each individual colony was given a unique identification code and its' position within the quadrat grid noted to determine recruit turnover in terms of survivorship, mortality, losses and new stock. Quadrats were resurveyed as part of a repeated measures design after 6, 13, 19, 24, 27 and 30 months. Detailed information on the status of individual colonies (i.e. partial mortality, fragmentation, overgrowth by algae) was recorded. Individual colony sizes (maximum and minimum colony diameter) were measured to nearest millimetre using vernier callipers.

RESULTS

Coral spat settlement on artificial settlement plates

Overall densities of coral spat on settlement plates were very low (ranging from 0.1 to 1.71 spat/0.04 m^2) at all sites after approximately 8 and 12 months deployment (Fig. 3). There was no significant difference in spat density between the two sampling periods (Kruskall-Wallis test H=0.66, p=0.417) but there was however, a significant difference in spat densities between sites after eight months (Kruskall-Wallis test H=11.34, p=0.023) and 12 months (Kruskall-Wallis test H=21.6, p=<0.001). This suggests that larval settlement is patchy between reefs. Overall there was no significant difference in spat settlement on cryptic or exposed surfaces after eight months

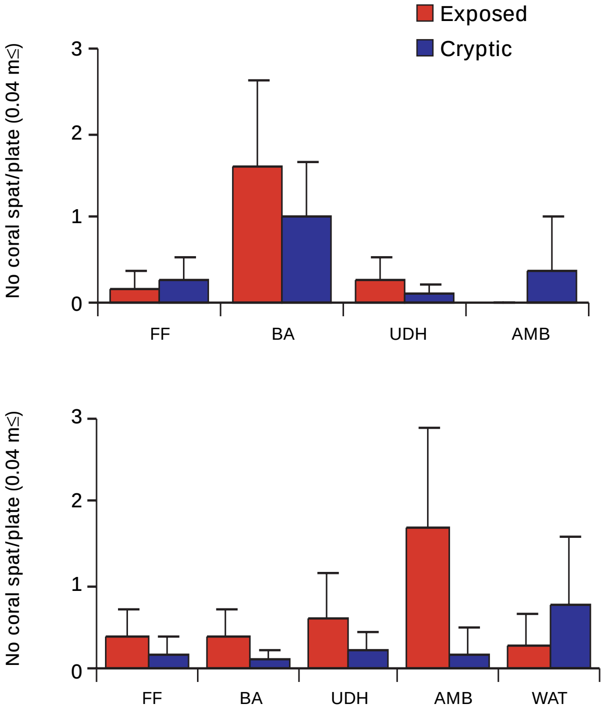

Figure 3. Mean density of coral spat on the settlement plates for plates collected after a) approx. 8 months and b) approx.12 months. Site codes: FF = Feydhoo finolhu; BA = Bandos; UDH = Udhafushi; AMB = Ambaraa and WAT = Wattaru. (Error bars are 95% CI).

(Kruskal-Wallis test H=1.63, p=0.202) or twelve months (Kruskal-Wallis test H=0.34, p=561). Taxonomic patterns were consistent on all plates. Dominant corals identified to generic level belonged to Agariciidae which accounted for over 34% of the total, however approximately 60% of the spat were too small to group taxonomically with certainty, thus a mixture of species may be represented.

Patterns of Succession and Community Structure on Artificial Surfaces

An initial pilot study demonstrated that a 60 random point sampling method was sufficient to represent all major benthic categories present on the artificial settlement tiles. The benthic communities on the tiles were classified according to nine major benthic categories and the percent cover of each group was determined. The data presented below is for one site – Feydhoo fonolhu. However, similar patterns were observed at all sites in Malé atoll.

The benthic communities that colonised the settlement plate surfaces after 126, 245 and 341 days of submergence are shown in Fig. 4 (next page) for two depths. After 126 days submergence the exposed surfaces were dominated by filamentous and turf algal groups (72% and 86% at 5 m and 10 m respectively). Algae was also the dominant group on the cryptic surfaces at both depths however, the percent cover was lower (46% at 5 m and 52% at 10 m) than that observed on the exposed surfaces. The second major benthic group on the cryptic surfaces was bryozoans but this group was virtually absent from exposed surfaces. Competition for space was so intense that there was virtually no bare space on either cryptic or exposed surfaces after 4 months of exposure for both depths.

After 265 days the percent cover of algae declined on all surfaces and at both depths, although it was still the dominant group on exposed surfaces (47% at 5 m and 49% at 10 m). Conversely, the cryptic surfaces were dominated by bryozoans (52% at 5 m and 35% at 10 m). Bryozoan diversity appeared to be high but taxonomic identification was not carried out for the purpose of the study. There was high variability in the percent cover of crustose coralline algae on both surfaces and at both depths. However, there was an overall increase in cover with increased submergence. Other invertebrates such as molluscs and polychaetes also increased in abundance on both surfaces with increased submergence. However, colonisation by colonial ascidians was more variable over all exposure periods and *Diademnum molle* was the dominant species present.

Data was pooled for sites and orientation to investigate temporal trends in community structure at the atoll level (Fig. 5). Overall similar trends in algal colonisation were observed in both atolls; the percent cover of algae was high initially but declined with increased exposure and at the same time the diversity of assemblages on the tiles increased. The major difference in the benthic com-

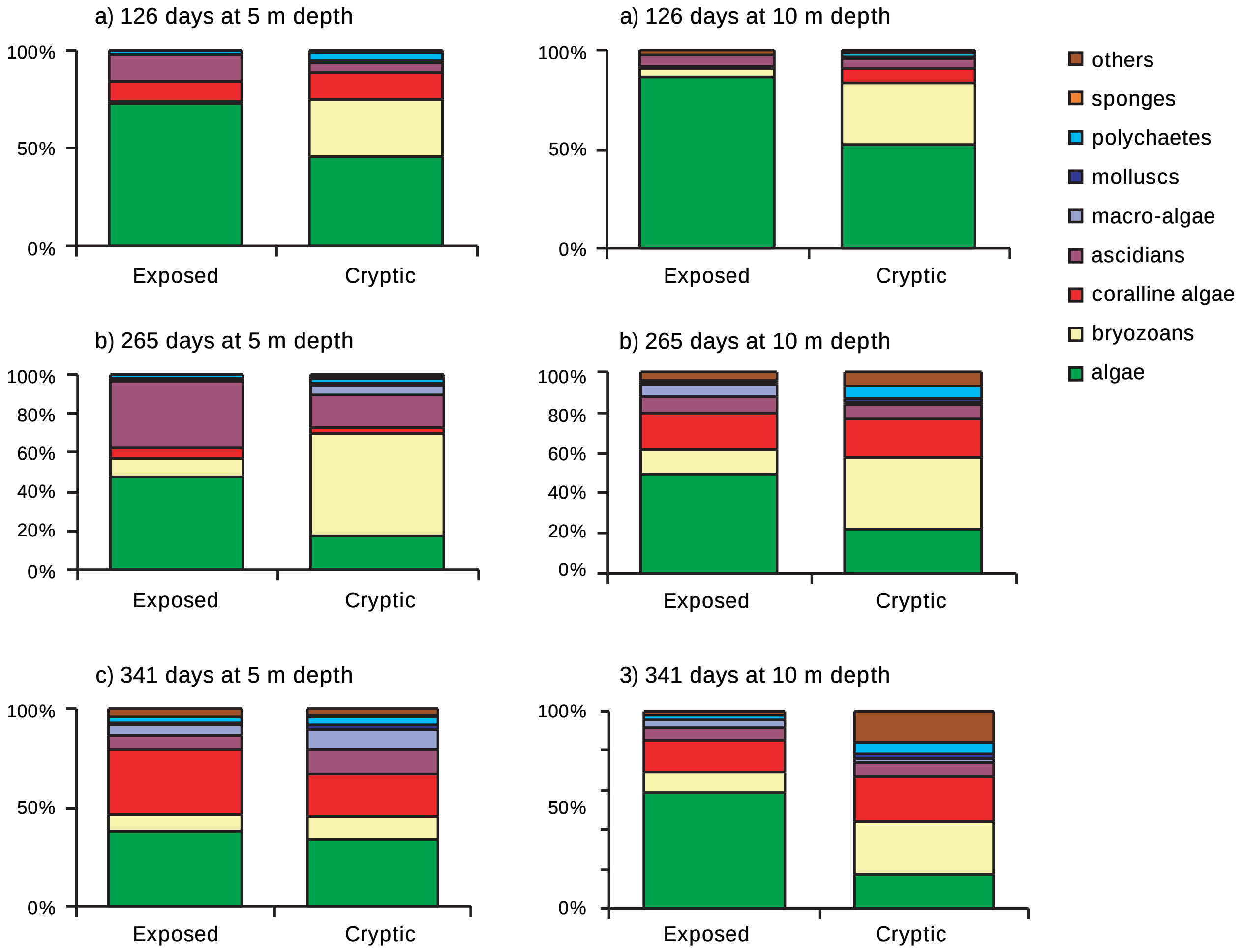

Figure 4. Benthic sessile communities developing on cryptic and exposed surfaces of settlement plates deployed at Feydhoo finolhu after 3 periods of submergence at 5 m and 10 m depth.

munities between the two atolls was the low colonisation of bryozoans in Vaavu atoll compared to Malé atoll. Further analysis will investigate the relationship between the benthic community structure on tile and spat settlement preferences.

Coral Recruitment

Data on coral recruitment has been collected for 6 sites over 7 sampling periods between February 2000 and July 2002. A database has been designed to facilitate the storage, manipulation and analysis of the data-set on coral recruitment. At this time a detailed analysis of temporal patterns in coral recruitment processes is only available for one site – Feydhoo finolhu. The remaining sites will be analysed in the next 3 months.

Overall densities of coral recruits (<50 mm) on the natural substrate within permanently marked quadrats were high for all sampling periods (Fig. 6a). Generally the highest densities of recruits within both reef zones were observed during the initial census in February

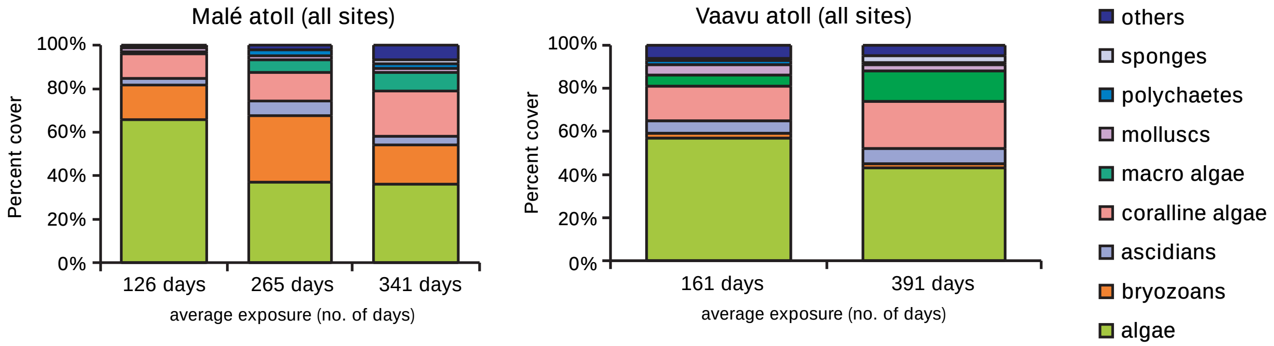

Figure 5. Temporal patterns in benthic communities developing on settlement plates in Malé and Vaavu atolls.

2000 (49.2 per m^2 at 10 m) with the lowest densities being recorded in September 2001 (30 per m^2 at 10 m). There was however, no significant difference in recruit densities with sampling periods within both reef zones (Kruskal-Wallis test; H=2.35, p=0.885 for 5 m and H=8.76, p=0.188 for 10 m). Similarly, pairwise comparisons indicated no differences in recruit density with depth at each census. Densities of juveniles colonies (>50 and <100 mm mean colony diameter) within the same quadrats were generally lower than recruit densities (Fig. 6b). There were no significant differences in juvenile densities with sampling periods for both reef zones however, pairwise comparisons (Mann Whitney – p<0.001) revealed that there were significantly more juveniles at 5 m than 10 m for February 2000 and August 2000.

Coral recruitment at all sites was dominated by colonies belonging to Agariciidae followed by Poritidae and Siderastreidae. The branching corals Acroporidae and Pocilloporidae were found in low densities at all sites suggesting that recovery will be slow, presumably due to the limited source of larvae.

Recruit Turnover

Survivorship of recruits and small juvenile corals within the permanent quadrats was high (>80%) and did not differ significantly over the six survey periods, between February 2000 and July 2002. The number of colonies missing (e.g. no evidence of skeletal remains) from the

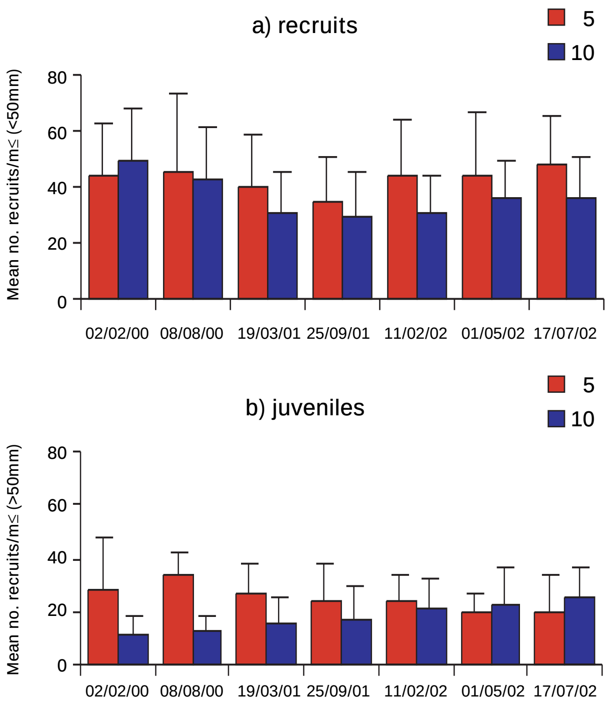

Figure 6. a) Densities of the mean number of coral recruits/m^2 (colonies < 50 mm mean diameter) at Feydhoo Finolhu. b) Densities of the mean number of coral juveniles/m^2 (colonies > 50 mm mean diameter) at Feydhoo Finolhu. (Error bars are 1 S.D.)

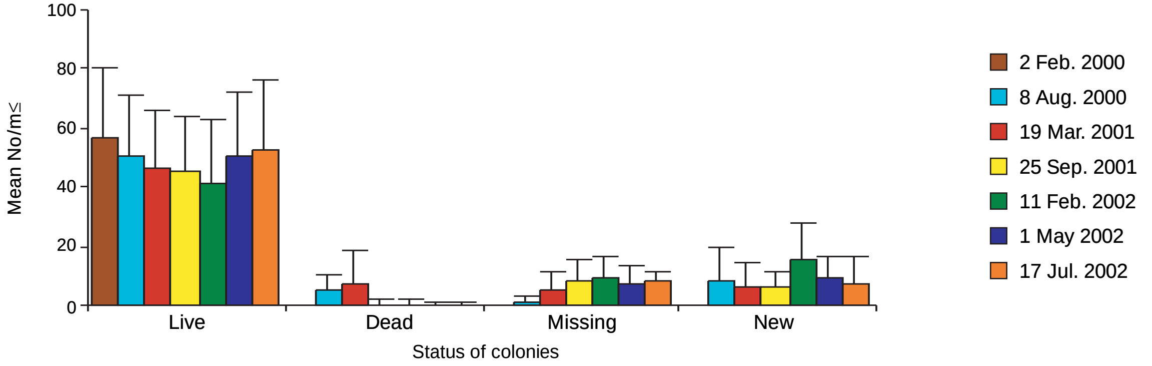

Figure 7. The status of coral recruits measured within small permanent quadrats between six census periods from February 2000 to July 2002 at Feydhoo finolhu. (Error bars are 1 S.D.)

Table 2. The abundance of coral colonies within 3 size classes measured in permanent quadrats between February 2000 and September 2001.

	Colony size (mm)		
Date	**<40**	**40–80**	**>80**
Feb-00	40.4	8.4	0.9
Aug-00	46.2	8.7	1.6
Mar-01	48.4	10.4	1.1
Sep-01	52.0	13.3	1.8

sampling area was less that the numbers found dead (e.g. skeletal remains present) in the first two surveys, but this trend was reversed in the subsequent surveys (see Fig. 7). Overall losses (dead and missing) were low in all survey periods, and this low turnover combined with a high influx of new recruits meant that the final stock densities were equal or greater than the initial standing stock, indicating that the current potential for reef recovery in the Maldives is favourable.

Colony Size Frequency Distributions

The results of the colony size frequency distributions presented in Fig. 8 demonstrate that the coral population at Feydhoo finolhu had a truncated size distribution at the initial census (approx. 2 years after the 1998 bleaching event) with very few large colonies present. Although a similar pattern in size frequency distribution was observed in subsequent census periods there was a slight increase in the frequency of larger colonies over time (see Table 2).

Conclusions

The preliminary results have shown that there was high spatial variation in coral spat settlement on the artificial surfaces at the level of regional atolls. Such variation may have profound effects on the long-term recovery from the 1998 bleaching event, with some reefs displaying slower rates of recovery than others. There was however little taxonomic variability. The dominant taxonomic group of coral spat settlement belonged to the genus *Pavona*, which is consistent with the existing adult population on the reefs (see Zahir, this volume). The survey design (i.e. frequency of tile deployment and retrieval) was not sufficiently sensitive to detect any seasonal patterns in coral settlement thus a more intensive sampling programme has been initiated to collect data at bi-monthly intervals for three sites in Male atoll between 2001 and 2002.

Rapid colonisation of settlement plates by various algal groups (mainly turf and filamentous algae) and bryozoans may pre-empt space and limit the settlement of coral spat. The lack of bare space on tile surfaces after 4 months suggests that competition for space may have been a limiting factor in this study. As coral spat were observed on both cryptic and exposed surfaces there was

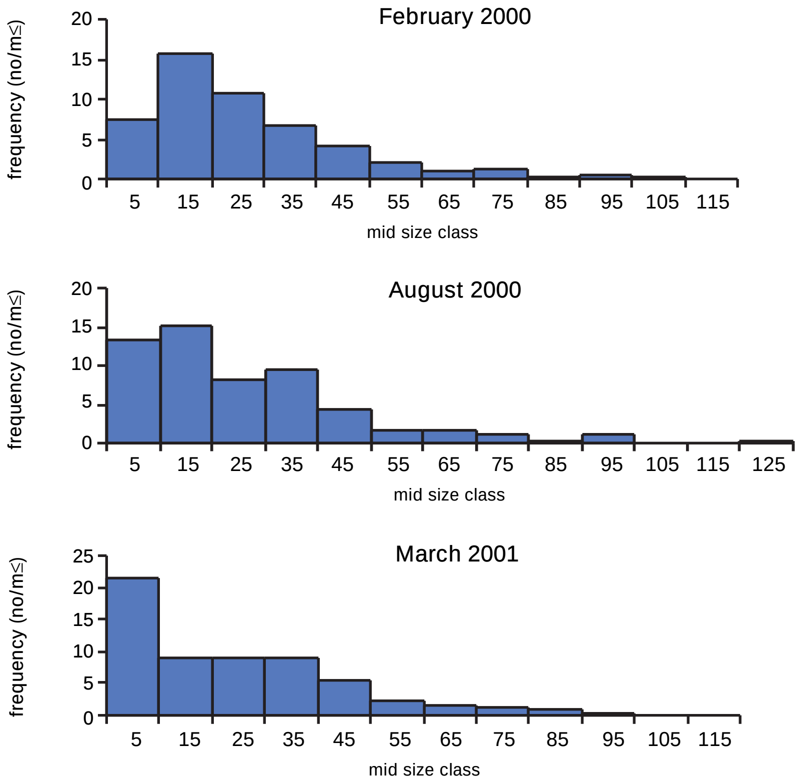

Figure 8. Size frequency distributions for all coral colonies recorded in the permanent quadrats at Feydhoo finolhu between February 2000 and March 2001.

no apparent relationship between the sessile benthic community developing on the settlement plates and spat settlement preferences.

The study of coral recruitment patterns at Feydhoo finolhu between April 2000 and July 2002 indicated that recruit densities were high overall (up to 49 individuals/m^2) and that there was very low variation in the densities of new recruits and small juveniles over the two and half year period. Further analysis of data collected for five additional sites will investigate spatial patterns in coral recruitment processes. Small individuals representing the new influx of recruits dominated the size frequency distribution of colony sizes. Such results are expected in a population that has suffered a major mortality event. Whilst density and size frequency data confirm that reef recovery processes are well underway the population at Feydhoo fonolhu is still dominated by one genus *Pavona* spp. indicating that diversity is low and that the coral community structure may take several decades to recover to the pre-bleaching level or may never recovery to its' former state.

Turnover of recruits in terms of mortality and loss versus new influx was consistently low at all sites and sampling periods The low level of turnover together with the high influx of new recruits meant that the recorded densities of recruits were equal or greater than initial standing stock indicating that the potential for reef recovery is high. However, longer time scales (e.g. a minimum of five years) would be required to determine whether the temporal patterns demonstrated in this study are typical or whether there are occasionally years with a high or low influx of new recruits. Provided post-recruitment mortalities are low (as indicated in this study) it is likely that coral populations will develop with a multitude of cohorts over the next 5–10 years and this in turn may result in higher recruitment.

Taxonomic patterns of spat settlement on artificial surfaces and natural recruitment are consistent with *Pavona* being the dominant genus in both studies. This also correlates with results of the long-term coral monitoring study (see Zahir, this volume), which show the most abundant adult colonies in reefs across several atolls are *Pavona spp*. Branching corals belonging to Acroporidae and Pocilloporidae, which were severely affected by the bleaching-induced mortality, were relatively low in abundance at all survey periods suggesting that the source and supply of larvae for these species are limited.

In the Chagos archipelago where bleaching-induced mortality was less severe than that observed in the Maldives (Sheppard, 1999) a recent study of new coral recruitment found average densities of juveniles was approximately 78 individuals/m^2 with twenty five coral genera observed and *Acropora* being the most dominant (Sheppard *et al.*, 2002). This included larger colony sizes than those typically measured in recruit studies however, after adjustment to represent smaller colonies (i.e. 20 mm) the levels of recruitment were still greater than those reported here for the Maldives. In the Seychelles a broad-scale assessment of 15 reefs found that recruitment was patchy and low, particularly for *Acropora* and *Pocillopora* (Turner *et al.*, 2000). Densities of small corals (2–15 cm) varied between 0.25 and 8 individuals/m^2 (converted from densities per 24 m^2 area for comparative purposes). Although the densities of recruits recorded in the Maldives were higher than those reported for the Seychelles the taxonomic patterns were very similar. These results indicate that recruitment processes throughout the Indian Ocean are very patchy and that a range of studies at several spatial and temporal scales are required to understand the mechanisms underlying the observed patterns.

ACKNOWLEDGEMENTS

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Assessing Bioerosion and Its Effect on Reef Structure Following a Bleaching Event in the Maldives

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Key words: bioerosion, bleaching, coral reefs, Maldives

INTRODUCTION

The unprecedented coral bleaching in the Maldives in 1998 resulted in extensive mortality of reef invertebrates with symbiotic algae, especially reef building corals. The percentage of living coral cover declined markedly immediately after the bleaching event, from around 30–60% pre-bleaching to 0–5% post-bleaching for shallow reef-flat areas (Allison, 1999). Fast-growing branching corals, particularly *Acropora* spp., were more susceptible to bleaching, and, as a consequence, suffered greater mortality than slow-growing massive corals.

The ultimate consequences of the 1998 bleaching event will not be fully understood for some time, possibly for decades. However, it is clear that reefs will be modified as a result of this bleaching event. In the short term (<5 years), coral reefs that were formerly dominated by branching species will now be dominated by non-living substrate that support a few surviving colonies of massive species. The consequences of bleaching for the reef framework will largely depend on the transport and fate of $CaCO_3$ fragments. Where reef disturbance is severe, boring and grazing organisms may remove $CaCO_3$ faster than primary frame-builders can accrete it. Such biogenic processes will determine whether the structural integrity of the reef will remain intact.

This study was conducted to investigate the processes of reef recovery after the bleaching event in 1998. The primary objectives of this study were to identify the main contributing groups of reef boring organisms and their relative rates of bioerosion.

METHODS

Coral blocks (10 cm x 5 cm x 1 cm) were cut from live colonies of *Porites* collected from a nearby reef. After these blocks were cut, they were soaked in fresh water and oven dried to a consistent weight. At Gulhifalhu reef (atoll outer reef, facing open ocean) and Feydhoofinolhu reef (atoll inner reef, atoll lagoon) in North Male atoll (Fig. 1 on next page), seven blocks were placed at depths of 5 m and 10 m. Each block was attached to a plastic push mount plug drilled to the reef substrate using cable ties via a central hole drilled in each block.

Each coral block was collected after a period of 14 months between February 2001 and April 2002 and was examined visually to record the types of fouling community and their relative densities. Each block was dried and sectioned in both the longitudinal and transverse planes to yield eight cut surfaces per block. The relative area removed by various bioeroding organisms was estimated by placing a plastic sheet printed with small dots evenly spaced over each cut surface of the block and counting the number of points covering the area removed by borers. The intensity of bioerosion was calculated as the percentage of the cross sectional area removed by each type of boring organisms divided by the cross sectional area of each block. Boring organisms were identified as worms, sponges or molluscs by the characteristic cavities they left in the coral blocks. Bioeroding worms were further classified into two size classes, micro worms (<1 mm in diameter) and macro worms (>1 mm in diameter).

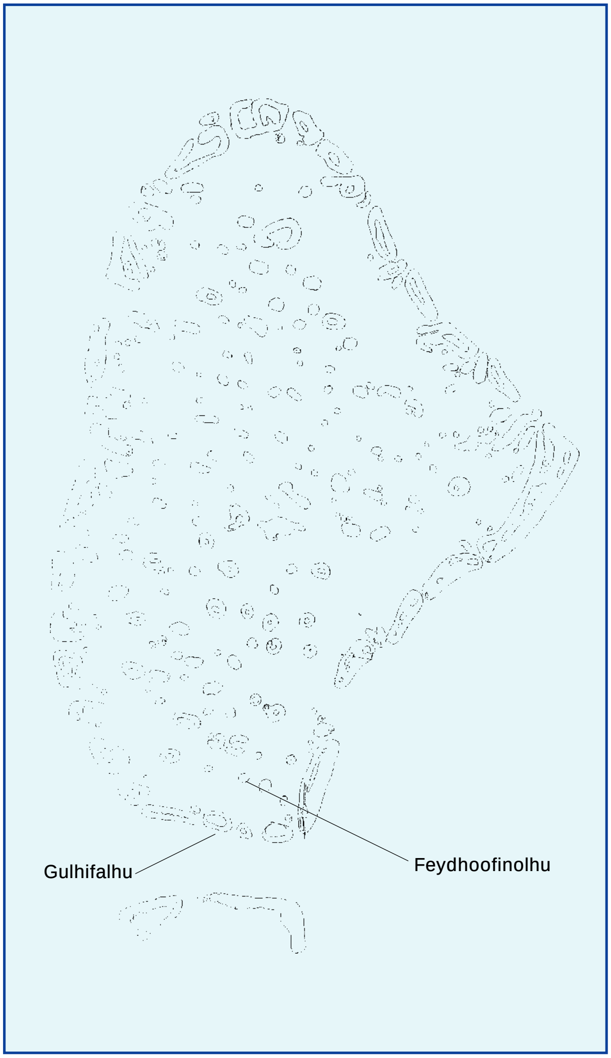

Figure 1. Male atoll.

RESULTS

The total area removed by bioeroders was greater at Feydhoofinolhu than at Gulhifalhu (17.3% and 14.3 % for Feydhoofinolhu and Gulhifalhu respectively). Of the four groups of bioeroders idenitified, sponges removed the most of the calcium from the blocks (Feydhoofinolhu, 10.1%) and (Gulhifalhu. 4.8%) followed by clams 2.3% and 4.2% for Feydhoofinolhu and Gulhifalhu respec-

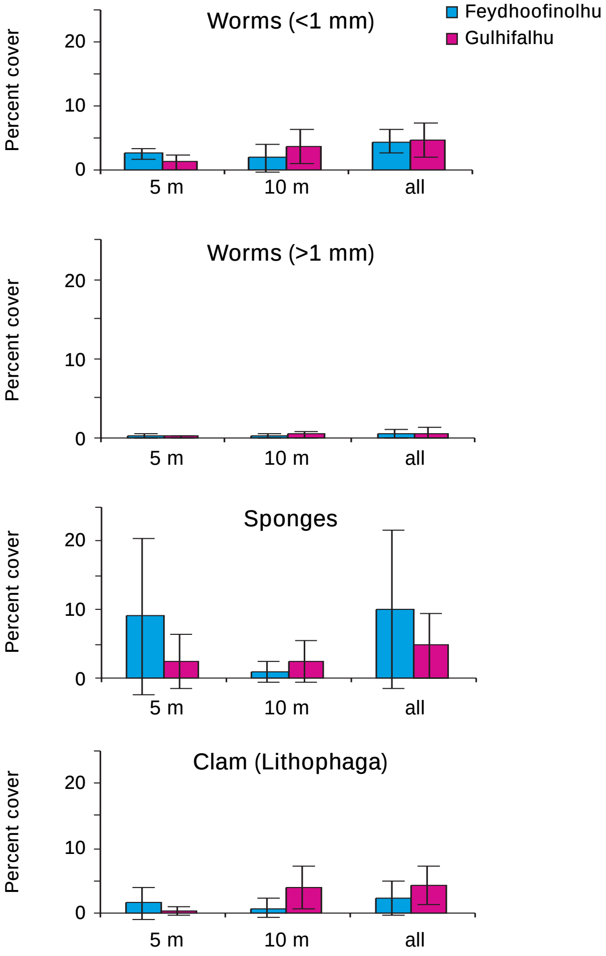

Figure 2. Intensity of boring at the two study locations. Boring intensity is given as the percentage of the area removed by each group from the cross sectional area of the block. (4 separate plots one for each group)

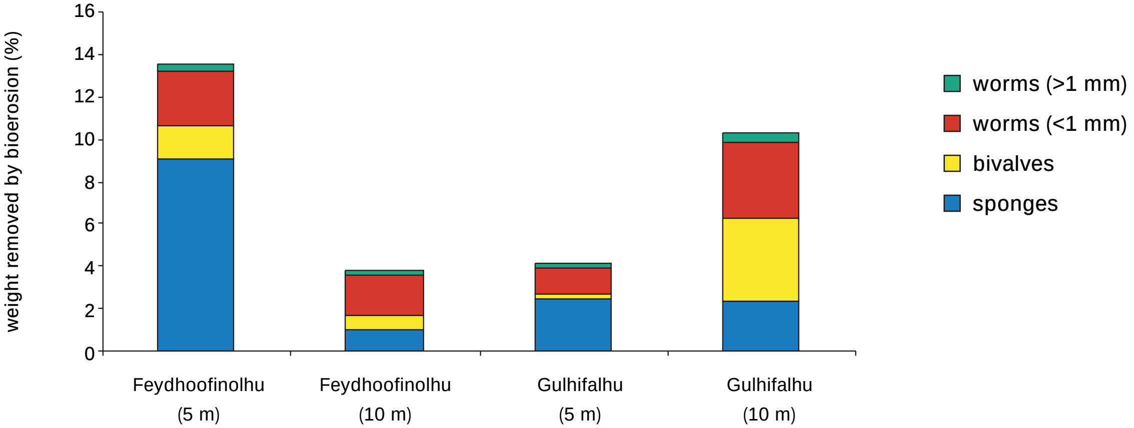

Figure 3. Comparison of the borers at the two study sites Feydhoofinolhu and Gulhifalhu.

tively (Fig. 2). Boring by worms was less at all sites compared to the previous two groups.

There is a significant difference in the intensity of bioerosion between the two depths at each study site. The relative intensity of bioerosion at Feydhoofinolhu at 5 m (13.5%) was higher than at 10 m (3.8%). However, this patterns was reversed at Gulhifalhu where the intensity of bioerosion at 5 m (4.1%) was less than that recorded at 10 m (10.3%) (Fig. 3). Overall, boring by sponges (e.g. *Cliona* sp.) was highest followed by bivalves (mostly *Lithpphaga* sp.) and worms <1 mm group (mostly spirobids).

DISCUSSION

The level of bioerosion at Feydhoofinolhu was significantly higher compared with Gulhifalhu. This difference in the intensity of erosion cannot be attributed to any environmental factor related to these sites because no environmental parameters were recorded for the purpose of this study. Eutrophication has been implicated in causing greater levels of bioerosion (Holmes *et al.*, 2000), but neither study sites have suffered excess nutrient levels.

After the coral blocks were deployed, their entire surface was rapidly colonised by fouling organisms, including filamentous and macro algal groups, calcareous algae, sponges and bryozoans (Fig. 4). Identification of the endolithic borers was carried out using the shape of the boreholes, which were grouped into three major categories (Fig. 5 on next page). Detailed grouping and identification of the organisms will be carried out at the later part of the study.

The intensity of bioerosion at the study sites over the 14-month period ranged between 10–20% and was attributed to three major taxonomic groups. The preliminary finding from this study gives some indication of the rate of biological erosion contributed by the endolithic borers at the sites investigated. Bioerosion has been reported as a key process limiting the rates and patterns of coral reef growth (Hutchings & Bamber, 1985) and, as a

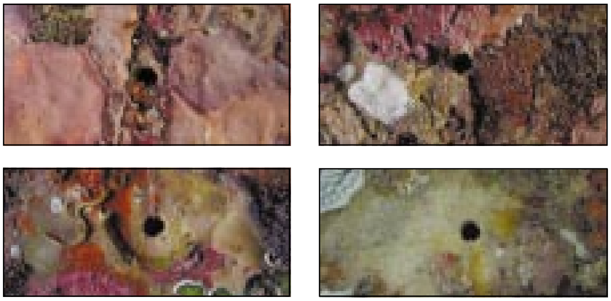

Figure 4. Fouling community succession on settlement tiles. Surfaces were completely covered by coralline algae, sponges, ascidians and bryozoans within four months of deployment.

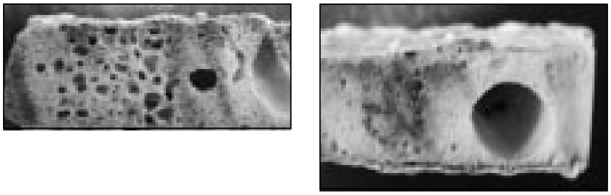

Figure 5. Examples of endolithic borers identified. The large hole is from *Lithophaga* (clam), the very small bores are mainly from worms and the spongy appearance boring with cavities are from boring sponges (e.g. *Cliona*).

consequence, can be viewed as a significant factor affecting reef recovery processes in the Maldives following the mass coral bleaching in 1998. These preliminary findings provide some insight to the intensity of bioerosion by the specific groups of borers identified. It also gave some indication of the important role these organisms play in breaking and restructuring of the reef framework.

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Status Report on the Condition of Reef Habitats in Sri Lanka, 2002

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Key words: coral reefs, management, monitoring, Sri Lanka

ABSTRACT

Reefs habitats were surveyed at selected locations during 2001 and 2002. Surveys indicated that recovery of damaged reefs is slow and variable at different locations. Bar Reef Marine Sanctuary and Weligama had better natural recolonisation by *Acropora* species. Overgrowth by *Halimeda*, and various filamentous algae and sedimentation appear to inhibit recruitment and growth of corals in other areas. Butterfly fish have become scarce on shallow water coral habitats. Human activities continue to damage reef habitats. Management and law enforcement is weak.

INTRODUCTION

Sri Lanka has a coastline of about 1585 km without coastal indentations, bays, offshore islands and lagoons (CCD, 1990). The continental shelf area of about 28,000 km^2 is narrow, and its average width is approximately 20 km. The climate is driven by the northeast and the southwest monsoons. The annual rainfall in the central highlands is about 2500 mm mostly during the southwest monsoon. The coastal plains and the rest of the island receive about 1500 mm of rain (Baldwin, 1991). Major rivers flow throughout the year and discharge large amounts of particulate matter into coastal waters that results in high turbidity in the coastal areas during the monsoons. During periods between monsoons the visibility varies between 5 to 30 m depending on the location and the sea conditions (Rajasuriya & Premaratne, 2000). Surface currents are primarily driven by the monsoons. Tidal range is low (0–1 m) and the salinity of the coastal waters is 35 ppt. Seawater temperature varies between 26°C and 30°C (Swan, 1983).

Sri Lanka has a population of about 19 million, the majority living in the western and southern parts of the country. More than 30% of the population lives in coastal areas (Baldwin, 1991). Coral reefs are important for the fisheries industry and for tourism and protection of the coastline (Rajasuriya & Premaratne, 2000). Nearshore fish production amounts to 70–80% of the total marine fish catch. Coastal tourism including dive tourism brings in US$ 200 million annually. The marine ornamental fish trade is valued around US $ 3 million per year.

Many government organisations are responsible for the management of coral reefs and the coastal environment in Sri Lanka (Rajasuriya & White, 1995; Wilkinson, 2000). In the recent past several laws and regulations have been introduced to improve management of resource use and prohibit activities that cause damage to coastal habitats. In spite of such initiatives there has not been a noticeable improvement in management.

Surveys have shown the presence of three types of reef habitats along the coast of the country: coral, sandstone and rock (Rajasuriya & De Silva, 1988). Live coral is found at varying levels on sandstone and rock habitats (Rajasuriya *et al.*, 1995). Coral reef habitats have been seriously degraded during the last decades primarily due to influences from various human activities on land and fishing using destructive techniques (see for example Öhman *et al.*, 1993). Furthermore, the coral reefs of Sri Lanka were seriously damaged during the Indian Ocean coral-bleaching event in 1998 (Wilkinson *et al.*, 1999; Rajasuriya & Karunarathna, 2000).

METHODS

The reef status was determined at selected locations in the southern and western coasts during the period from 1999 to 2002. Shallow coral reef habitats were surveyed using the line intercept transect method and observations were carried out on deep offshore reefs to record the status of degradation and recovery.

Reef habitats were surveyed using snorkel and scuba diving. Fifty meter line intercept transect (LIT) method was used to determine the percentage cover of benthic organisms (English *et al.*, 1997). The presence of selected coral dependent fish species was recorded in an area of 250 m^2 (50 m x 2.5 m on either side of the transect) at each transect. In addition, sandstone reef habitats and offshore reefs were examined at selected locations to determine the impact of fishing activities.

Further marine algae, molluscs and echinoderms were recorded along 50 m transects at four locations. These records were carried out in parallel with the coral and fish counts. Specimens of algae were collected only when their identity was not known, or as a confirmation if a known alga was found in a new locality. Specimen collection was restricted to macro algae, mollusc shells and sea urchins (Fernando, 2002).

REEF STATUS

Bar Reef Marine Sanctuary

Prior to bleaching in 1998 Bar Reef was one of the richest reef areas in Sri Lanka with extensive areas living corals and abundant fish resources (Öhman *et al.*, 1997). Coral areas were dominated by branching *Acropora* spp., and large patches of *Echinopora lamellosa* also existed on one of the shallow reef patches. Crown-of-thorns starfish had damaged some parts of the reef. Fishing was the main anthropogenic impact (Öhman *et al.*, 1993; Rajasuriya *et al.*, 1995) although it was relatively limited.

Figure 1. New coral growth at the Bar Reef Marine Sanctuary at 1.5 m depth (January 2002). *Photo:* Dr. MALIK FERNANDO

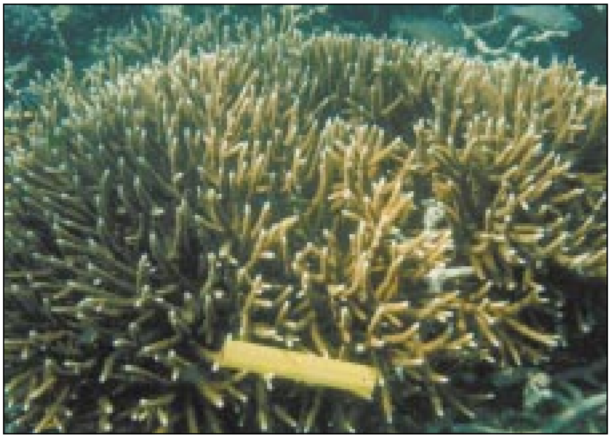

Figure 2. New coral growth at the Bar Reef Marine Sanctuary in January 2002. Size of ruler (6 inches). *Photo:* ARJAN RAJASURIYA

During the immediate post-bleaching period there was an almost total mortality of *Acropora* and *Echinopora* corals. Large areas were covered with dead corals. Most species that survived severe bleaching were those that was below 10 m. In the latter part of 1998 the shallowest (1–3 m) areas of the reef was covered by a dense carpet of alga (*Bryopsis* sp.), which lasted for about two months. Butterfly fishes which were previously abundant became rare. Overall the abundance of fish in the shallow areas of the reef was reduced (Rajasuriya & Karunarathna, 2000).

Many branching and tabulate *Acropora* colonies and *Pocillopora damicornis* were observed during the surveys in 2002 in areas that were devoid of live coral during the previous surveys. In the shallowest coral habitat (1–3 m depth) the number of tabulate *Acropora* were relatively high compared to branching *Acropora* (Fig.1). The largest colonies of branching *Acropora* were about 100 cm in diameter and 35 cm high (Fig. 2). Although live coral still is sparse, much of the reef structure is intact and crustose coralline algae cover much of the dead coral.

Better coral growth was recorded at a depth of 7–8 meters at a site about 300 m south of the shallow coral area. Many species of hermatypic corals (*Acropora, Montipora, Favites, Favia, Galaxea fascicularis, Pavona, Cyphastrea, Hydnophora microconos, Podabacea crustacea*) were present (2002) and the maximum diameter of colonies was about 50 cm. Live coral cover was 14% of which 3% was branching *Acropora* species whilst 0.3% was tabulate *Acropora* species. Coral rubble, dead coral with algae and small sandy patches comprised about 80% of 4 transects.

In the survey in 2002 butterfly fish were rare amongst all the shallow coral reefs due to the scarcity of live corals. Herbivores such as *Siganus lineatus, Scarus dimidiatus* and *Acanthurus* species were common. Newly settled juvenile *Dascyllus aruanus*, and *Chromis viridis* were present on several branching *Acropora* colonies.

The main threat to the Bar Reef is fishing (Dayaratne *et al.*, 1997). Fishing is carried out within the Bar Reef Marine Sanctuary with the use of two to three types of gillnets and bottom-set nets. The main target species belong to Hemiramphidae, Haemulidae, Siganidae, Lutjanidae and Lethrinidae. Bottom-set nets are also used to catch spiny lobsters. Moreover, a new form of barrier net is being used in the area. In this fishing divers spot the fish among relatively shallow coral patch reefs up to a depth of about 10 m. The divers then drive the fish into the net, which is placed in an arc at one side of a selected patch reef.

Colombo

Pitagala and Gigiripita reefs were surveyed. Both reefs are large coral mounds, 2–3 hectares in extent at a depth range of 15 to 22 m. They are surrounded by deeper (35 m) water. Corals belonging to the families Poritidae and Faviidae were common. Live and dead coral was 26% and 66% respectively. Sponges, soft corals, algae and sand amounted to 8%. The underwater visibility often exceeds 20 m.

There are strong currents, and occasionally gill nets drift over the reef and cause damage to corals. A greater threat exists from shipping because both reefs are situated within the outer harbour anchorage in Colombo. The author has observed serious anchor damage to corals in 1999 on the Pitagala reef.

Hikkaduwa

In the Hikkaduwa Nature Reserve live coral cover was reduced from 34% to 7% during the 1998 coral-bleaching event (Rajasuriya & Karunarathna, 2000). Eight line intercept transects in 2001 indicated that the live coral cover was 12%. Three genera contributed more than 1% each to the percentage of total live coral cover: *Porites* (7%), *Montipora aequituberculata* (3%), *Acropora muricata* (1%). During the 2002 surveys *Montipora aequituberculata* had begun to recolonise the dead *Acropora* areas (Fig. 3 on next page). All *Acropora muricata* recorded as live coral in 2002 were colonies that had been introduced by man after the bleaching event. These colonies are alive but are stunted (Fig. 4 on next page) with little indication of healthy growth after more than three years. Natural recruitment of *Acropora* spp. was not detected within any part of the reef lagoon. Colonies of *Goniastrea*

Figure 3. Recolonisation of dead *Acropora* areas by *Montipora Aequituberculata* in January 2002. *Photo:* ARJAN RAJASURIYA

Figure 4. Condition of corals transplanted in 1999 in the Hikkaduwa reef lagoon (Photo January 2002). The transplanted colonies were the same size indicating that conditions are unsuitable for the growth of branching *Acropora* species. *Photo:* ARJAN RAJASURIYA

retiformis, *Echinopora lamellosa*, *Favia speciosa*, *Faivites chinensis*, *Platygyra lamellina*, *Platygyra sinensis*, *Leptoria phrygia*, *Acanthastrea echinata*, *Galaxea fascicularis* and *Pavona varians* appear to be healthy and tolerate the high levels of sand and sediment accumulation within the reef lagoon.

About 59% of the reef consisted of dead corals and coral rubble, while sand amongst corals and algae made up 29%. The percentage of *Halimeda* on coral rubble, dead coral and sand were 13%, 0.3% and 2% respectively. There was an almost total absence of butterflyfish and damselfish such as *Plectroglyphidodon lacrymatus, P. dickii* and *Gobiodon* spp. which used to be common among branching corals in the area.

Rajasuriya and Karunarathna (2000) reported that the sandstone/limestone reef outside the southern boundary of the Hikkaduwa Nature Reserve contained corals in good condition. This section of the reef contained 17% live coral, 40% dead coral and 43% of (limestone substrate, rock, sand, *halimeda*, soft corals and coral rubble). Genera which had more than 1% cover were *Porites* (4%), *Acropora* (4%), *Favites* (2%), *Montipora* (1%) and *Platygyra* (1%). *Acropora rudis* comprised half of the total amount of *Acropora* spp.

Accumulation of sand and sediment within the Hikkaduwa reef lagoon appears to be the major factor that prevents recruitment and the healthy growth of corals within the nature reserve. Direct human impacts are from glass bottom boats, trampling of corals, anchoring of fishing boats and pollution add to the stress. The Coastal Resources Management Project (CRMP) funded by the Asian Development Bank (ADB) has planned to zone the reef lagoon according to the zoning plan under the ‘Special Area Management Plan for Hikkaduwa Marine Sanctuary and Environs’ (HSAMMSCA). Attempts have been made earlier to zone the reserve but without success. Thus marker buoys installed in 1997 during the previous zoning were lost due to lack of maintenance (De Silva, 1997).

Kandakuliya

Reefs at four different depths (5 m, 10 m, 18 m and 20 m) were investigated at Kandakuliya. In the shallow (1–5 m) coral reef small colonies of *Acropora* and *Montipora* species were recorded in 2000 (Rajasuriya & Karunarathna, 2000). By 2002 nearly all of the young coral colonies had been lost due to an overgrowth of *Halimeda* (Fig. 5). Further, fish such as Soldierfish (*Holocentrus* spp.), Squirrelfish (*Myripristis* spp.) and Bullseyes (*Priacanthus* spp.), caught on the coral reefs in Kandakuliya, were previously considered undesirable. However, today these fish are used for consumption, which suggests a decline in fish stocks in the area (Fig 6.).

The sandstone and limestone platforms at a depth of 18 m were examined from Kandakuliya to Bar Reef Marine Sanctuary. There is limited coral growth, mainly species belonging to the families of Faviidae, Mussidae, Poritidae and Dendrophylliidae (Rajasuriya *et al.*, 1998a). No obvious damage to the reef from human activities or natural causes were detected. Small groups of 5 to 6 individuals of large (up to 1.5 m) hump head wrasses (*Cheilinus undulatus*) and 9 large (>1 m) groupers (*Epinepheplus malabaricus*) were observed as well as schools of snappers (Lutjanidae) and emperors (Lethrinidae) which were common as well as *Naso brevirostris*, *Acanthurus* spp., *Plectorhinchus ceylonensis* and *Plectorhinchus schotaf*.

Figure 5. Shallow coral area covered by Halimeda at Kandakuliya (January 2002). *Photo:* ARJAN RAJASURIYA

Figure 6. Soldierfish (*Holocentrus* sp), Squirrelfish (*Myripristis* sp) and Bullseyes (*Priacanthus* sp) caught at Kandakuliya using bottom-set nets. All three species were not used for consumption until recently (January 2002).

The reef habitat at around 20 m depth consisted of mushroom corals (*Zoopilus echinatus*, *Diaseris fragilis*, *Cycloseris cyclolites*, *Fungia fungites* and *Leptoseris papyracea*). This habitat was restricted to an area of about 50000 m^2 and is located at the edge of a relatively steep slope (~60°). Coral mortality or diseases was not observed in any of the reef habitats at 15 to 20 meters depth.

Moratuwa

Itipandama reef is a sandstone and limestone platform at a depth range of 8–12 m situated about 1 km offshore. Common coral species in this area are *Favia* spp., *Favites* spp., *Goniastrea pectinata*, *Goniastra retiformis*, *Gardineroseris planulata*, *Symphyllia radians*, *Montastrea valenciennesi*, *Pleasiastrea versipora*, *Pavona varians* and *Leptastrea* spp. Live coral cover amounted to 22%, soft corals 5%, sponges 3% and the substrate was 71%.

Major threats to reefs in coastal waters around Colombo are caused by intensive fishing using bottom set nets and gill nets and from pollution from land-based sources.

Negombo

The main coral reef habitat at Negombo is located approximately 20 km offshore. It is dominated by *Porites* spp. and is similar to the offshore reef habitats in Colombo. Live coral was 14% (Rajasuriya & Karunarathna, 2000). In 2001 the corals were found to be healthy although many large *Porites* domes contained dead coral patches. A short filamentous alga was abundant on the substrate.

Drift gill nets and bottom-set nets continue to damage the reef. A marked reduction was noticed among the schools of surgeonfish and unicornfish usually present at this site.

Polhena

The fringing reef at Polhena was seriously damaged due to coconut husk seasoning along the shoreline. The coconut husks are used for the production of coir. It is also a popular recreational area because of the shallow small back reef lagoon. Visitors walk on the reef flat during low tide and trample corals (Rajasuriya *et al.*, 1995; Öhman *et al.*, 1998). Rajasuriya and De Silva (1988) reported that the live coral cover (up to 10%) varied at different sections of the reef. A survey in 2000 revealed that the live coral cover, dead coral and algae were 12%, 41% and 16% respectively (Fairoze *et al.*, 2000). They also reported on the presence of several sea grasses within the lagoon area: *Cymodocea* spp. covering 21% of the seabed, *Halodule* sp. covering 24% and *Halophila* sp. covering 5%.

Major anthropogenic threats come from seasoning of coconut husks along the shore causing pollution and excessive growth of algae, trampling of corals and coral collection for souvenirs by visitors, and uncontrolled collection of marine ornamental species.

Rumassala

Live coral cover at Rumassala reef varied between 26% to 50% depending on the different sections of the reef (Rajasuriya & De Silva, 1988). Karunarathna and Weerakkody (1994) reported a live coral cover of 71% for the shallow coral patches and the scattered coral area near the headland at the entrance to the Galle Bay. Rajasuriya *et al.* (1998b) reported live coral of 45% prior to bleaching, which included the shallow coral patches and the spur and groove formations in deeper areas. The coral bleaching event in 1998 destroyed much of the live corals. However several species, such as *Porites rus*, *Psammacora digitata*, *Montipora aequituberculata* were only marginally affected. Live coral cover in 2000 was 20% (Rajasuriya, unpublished NARA report).

At present there is better coral growth on the deeper sections at 4–5 m. Many species including *Acropora aculeus*, *Hydnophora microconos*, *Galaxea fascicularis*, *Favia* spp., *Favites* spp. have colonised the deeper areas. However the shallow areas near the surface contains much dead coral.

The major threats are blast fishing, sedimentation, pollution and ornamental fish collection.

Trincomalee

The Pigeon Islands fringing reefs were in good condition during a survey carried out in 1999 (Christofellsz *et al.*, 2000). They also reported that a population explosion of corallimorpharians was a threat to the corals. Moreover there was blast fishing carried out in areas close to the reef by local people. The reef sustained minor damage during a cyclone that affected the north-eastern coastal area in 2000 (personal communication, marine ornamental fish collectors, 2001).

At present the islands are visited by large numbers of local tourists and the coral reef has come under threat from souvenir collectors and people trampling corals. Moreover visitors pollute the islands by dumping picnic refuse such as polythene bags, bottles and cans. In an attempt to combat pollution at Pigeon Islands, the Nilaveli Beach Hotel has initiated a programme to remove waste matter from the islands twice a week. However, unless the visitors are stopped from carrying non-biodegradable waste into the island the problem of pollution would become difficult to contain.

Unawatuna

The Unawatuna nearshore coral reef had 47% live corals prior to bleaching in 1998 (Rajasuriya *et al.*, 1998b) but was reduced to less than 1% due to bleaching. In January 2002 there were several small patches (*Acropora* spp., *Pocillopora eydouxi*, *Favia* spp., *Favties* spp., *Leptoria phrygia*, *Platygyra lamellina* and *Pavona varians*) of live corals on the nearshore reef (Fig. 7). An area of about 20 m^2 near the offshore rocky islets contained large stands of *Porites* and *Pocillopora eydouxi* that had recovered from bleaching. Overall fish abundance including butterfly fish was low.

Figure 7. Recolonisation by *Acropora* in the shallow coral reef at Unwatuna in January 2002. *Photo:* ARJAN RAJASURIYA

Major anthropogenic threats are ornamental fish collection and pollution due to unplanned tourism within the Unawatuna Bay. Due to lack of enforcement of regulations beach huts and temporary structures have increased in recent times. These are causing damage to the vegetation on land and pollution of the tidal and sub-tidal areas.

Weligama

Rajasuriya and Karunarathna (2000) reported that *Acropora muricata* recovered relatively well within the reef lagoon and that *Montipora aequituberculata* was not bleached and continue to exhibit healthy growth. At present these two species make up much of the live coral within the reef lagoon.

A study was carried out in 2000 to determine natural coral recruitment within the reef lagoon. A coral recruit was defined as a recently settled coral large enough to be visible to the naked eye. Recruits were counted within 27 belt transects (0.5 x 10 m). Main genera recorded were *Acropora*, *Montipora* and *Alveopora*. A 10 m transect was placed in the center of the belt transect and the percentage cover of live corals, dead corals and algae was measured. Live coral covered 21% of the seabed, dead coral and rubble 26%, algae including *halimeda* 28%, and other organisms including sponges, and substrate 25%. Filamentous alga was common on dead corals. The density of recruits was 2.4 per 0.5 m^2 (Rajasuriya & Wilhelmsson, 2000).

Rajasuriya and Karunarathna (2000) reported that the live coral cover on the *Acropora* banks was 28%, and similar figures were reported by Fairoze and others (2000). In December 2000, Fairoze and colleagues reported that the live coral cover had increased to 31%. The latest surveys in 2002 indicate that the live coral cover on the *Acropora* banks had reached 54%. The sea grasses present were *Cymodocea* sp. (26%), *Halodule* sp. (7%) and *Halophila* sp. (1%) (Fairoze *et al.*, 2000).

Very high pressure from ornamental fish collecting continues to damage the reef due to the use of 'moxy nets' which is an illegal fishing technique in Sri Lanka. In addition, anchor damage by fishing boats causes a great deal of physical damage to the reef. Local visitors trample the reef and remove corals for souvenirs. Furthermore, pollution due to oil discharges from fishing boats and wastewater discharged from a large hotel nearby may contribute to overall degradation of the reef.

DISCUSSION

The recent studies indicate that reef recovery is variable particularly in the shallow coral reef habitats. Good coral recruitment and reef recovery was observed at several sites such as the Bar Reef and Weligama (Table 1 on next page). Sudden transformation of a recovering reef as in Kandakuliya to a bed of *Halimeda* indicate that there could be reversals in the process of recovery and the presence of young coral colonies do not guarantee the complete recovery of a reef in the short term.

McCook (2001) had reported that filamentous alga was not successful in inhibiting growth of *Porites*. He also refers to the work of Potts (1977), who found that filamentous algae inhibit coral growth within the territories of damselfish whilst Gleason (1996) reported that filamentous algae could enhance coral growth. These varying results indicate that the recruitment and growth of corals in the presence of filamentous algae could be

Table 1. Percentage of live corals at selected reef sites.

Location	Depth (m)	Pre-bleaching (Percentage of live coral)	1999–2000 (Percentage of live coral)	2001–2002 (Percentage of live coral)
Bar reef	0– 3	78.5% overall in 1993–1994[2]	Nearly 100% mortality[2]	Several new colonies recorded but too sparse to estimate percent cover
Bar reef	7– 8		Nearly 100% mortality	14
Kandakuliya	0– 5	22[1]	Small colonies were present[2]	Corals smothered by *Halimeda*
Hikkaduwa Nature Reserve	0– 4	47.2[2]	7[2]	12
Hikkaduwa, South of Nature Reserve	7–13	Not estimated	Percent cover of live corals was not measured	17
Rumassala	1– 5	45[5]	19.6[4]	Better recovery at 4–5 m depth. Live coral cover not estimated
Weligama	0– 2	92[2]	28[2] 31.04[3]	54

Sources: [1]Öhman and Rajasuriya 1993; [2]Rajasuriya and Karunarathna 2000; [3]Fairoze, Cumaranatunge and Amarasinghe 2000; [4]Rajasuriya (unpublished report); [5]Rajasuriya et al. 1998b

highly variable depending on many factors. McCook (2001) cites previous studies (Lapointe, 1997; McCook, 1999) where it has been shown that algae can overgrow corals and lead to a phase-shift from a coral dominated habitat to an algal community as a result of reef degradation due to natural causes or eutrophication combined with overfishing. Factors such as over fishing and pollution are problems on Sri Lankan reefs although it is not known whether such factors contribute to increased growth of algae that has been observed on many reef habitats.

In addition to filamentous algae there appear to be an increase in calcareous alga such as *Halimeda* and most of the dead corals are covered by crustose algae (encrustrating coralline algae). Crustose algae help to bind the reef and prevent erosion of the reef structure (Fabricius & De'ath, 2001). They cite research carried out by Morse *et al.* (1988), Heyward and Negri (1999) that indicate that crustose algae induce settlement of reef organisms including hard corals.

Management of human activities is the single most important aspect in the recovery of coral reefs.

Threats from human activities such as coral mining in the sea, destructive fishing methods and uncontrolled resource extraction continue to be chronic problems and difficult to control (Rajasuriya *et al.*, 1995; Rajasuriya *et al.*, 2000; Spalding, 2001). Recently the government of Sri Lanka banned the use of coral lime for buildings constructed by the government (Rajasuriya & Premaratne, 2000). However this has not prevented or reduced illegal coral mining in the southern coastal belt. Coral miners are arrested occasionally in the southern coast but are released with minimal fines. There are no recent known instances where the law against coral mining has been implemented in the east coast where this practice is rampant, for example in Passikudah and Kalkudah. In Sri Lanka it is well known that there is partiality in applying the laws and that politicians interfere in the legal processes whenever people are arrested for illegal activities. Such problems are the main reasons for continued reef destruction in Sri Lanka.

The increase in population involved in fishing can lead to a much greater problem in the future. A large number of young males enter the fishing industry, mainly as ornamental fish collectors. Although a management mechanism is in place in the form of licensing under the Fisheries Act of 1996, the vast majority of collectors remain unlicensed. Similarly most fishermen are unlicensed and even today it is possible to enter the fishing industry without a license although the regulations regarding fishing licenses had been introduced many years ago.

Management is further complicated by the sectoral approach practiced by different parts of the government and government agencies. For example the Ministry of Fisheries and Ocean Resources and the Ministry of Industries are expected to promote fish production whilst the Ministry responsible for the Environment and the Department of Wildlife Conservation are expected to conserve biodiversity. However, there is very little or no collaboration between these agencies in implementing their policies. Due to such reasons there is an absence of a holistic approach to management and the environmental degradation and the overexploitation of the coastal areas continue.

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Reef Fisheries and Coral Reef Degradation in Sri Lanka

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Key words: coral reefs, fisheries, Sri Lanka

INTRODUCTION

Sri Lanka is situated in the Bay of Bengal, southeast of India. The island, with a population of about 18 million people, has an area of approximately 65,000 km^2 and a coastline of more than 1700 km. For generations fishing, including offshore fisheries (pelagic fisheries) and nearshore fisheries (semi-pelagic and demersal), has been a major source of food and income for people in Sri Lanka. Annual landings has increased from 183,990 MT in 1990 to a provisional estimate of 304,380 MT in 2000 (NARA, 2000), which means a 65% increase in production over the last decade. In 1996 marine fisheries provided full time employment to nearly 83776 people (FAO, 1999), and contributed 1.95% to GDP in 1998. Nearly 65% of the animal protein and about 13% of the total protein consumed in Sri Lanka is provided by the marine fisheries (Baldwin, 1991). The portion of marine fish harvested from coastal reefs has never been calculated as a separate category in Sri Lanka. However, considering the overall marine fish catch data for 1989 from categories such as large demersals, Carangids and Scombrids, it is estimated that reef fish comprise about 15% of the total fish landings in Sri Lanka (Rajasuriya *et al.*, 1995). Food fish from the reefs (e.g. emperors (*Lethrinidae*), snappers (*Lutianidae*), groupers (*Serranidae*), and goatfishes (*Mullidae*) etc.) is to a large extent consumed locally, but ornamental fish and lobsters are mainly caught for export. Reef fishery resources are heavily exploited in Sri Lanka. The fish stocks are under pressure from uncontrolled harvesting, and destructive fishing methods are causing major harm to the coral reef habitats.

Other human impacts threatening the reefs of Sri Lanka include coral mining, increased sedimentation on the reefs due to poor land use practices, pollution, boat anchoring and tourism related activities (De Silva, 1985; Baldwin, 1991; Öhman *et al.*, 1993; White, 1995; Rajasuriya *et al.*, 1998). Also, periodic infestations of the crown-of-thorn starfish threaten the reefs in the northwest and the east coast of Sri Lanka (De Bruin, 1972; Rajasuryia & Rathnapriya, 1994). In addition, the coral reefs around the Indian Ocean suffered severe mass mortality, due to elevated sea surface temperatures, during the strong El Nino in 1998 (Lindén & Sporrong, 1999; Wilkinson *et al.*, 1999). Most shallow coral reefs along the coast of Sri Lanka experienced extensive mortality, ranging between 50% and 100% (Rajasuriya & Karunatne, 2000). Recovery has in general been poor, and in some places non-existent. There is a close relationship between habitat structure and fish community composition (Öhman & Rajasuriya, 1998), and alterations of habitats will affect the productivity of the ecosystem (McManus, 1996; Öhman *et al.*, 1997). For example at some reef sites in Chagos the abundance and diversity of fish dropped to less than 25% of their former levels, as a result of the mass mortality of corals in 1998 (Sheppard, 1999). Further, on several shallow reefs in Sri Lanka the number of butterfly fish (*Chaetodontidae*), of which many are directly feeding on the corals, has decreased dramatically (Rajasuriya & Karunaratna, 2000). The degradation of the coral reefs should also have an impact on the pelagic fish foraging over the reefs, or living on coral reefs in their juvenile stage, and therefore affect the pelagic fisheries. However, the loss of the coral reef habitats is likely to have clearer adverse effects on the near shore small-scale fisheries, which is to a large extent carried out directly on the reefs.

To relate the coral reef degradation to declines in the fish catches is difficult due to the problems in estimating the individual importance of the various factors influencing the fish stocks and the catches. A general problem is that fishermen for obvious reasons immediately try to adapt to decreasing catches by shifting to new fishing areas. In addition fish catch data rarely admit the identification of catch sites, i.e. if a fish was caught on the coral reefs or in surrounding habitats such as rocky and sandstone reefs, seagrass beds etc. Hence, to illustrate the importance of the coral reefs for the communities dependent on fisheries for their livelihood, there is a need to give examples where fishing is carried out directly on the coral reefs. Further, it is also useful to, on a local case study basis, show the interdependency between the fishing activities and the status of the reefs. Thus, this study focuses on fishermen operating special gear types solely on coral reefs. An effort is made to map the magnitude and nature of the different fishing practices carried out on coral reefs in three selected sites in Sri Lanka, what damage they may cause the reefs, and their relative effectiveness. Also, fishermen's perception of various threats to the sustainability of reef fisheries, and the impacts of the degradation of the coral reefs on the fisheries in Sri Lanka, with special reference to the mass mortality of corals in 1998, is investigated. Further it seeks to establish a mechanism for continuous monitoring of future trends in reef fisheries, aligned with studies of changes in the status of the coral reefs, carried out by the Coral Reef Unit of NARA.

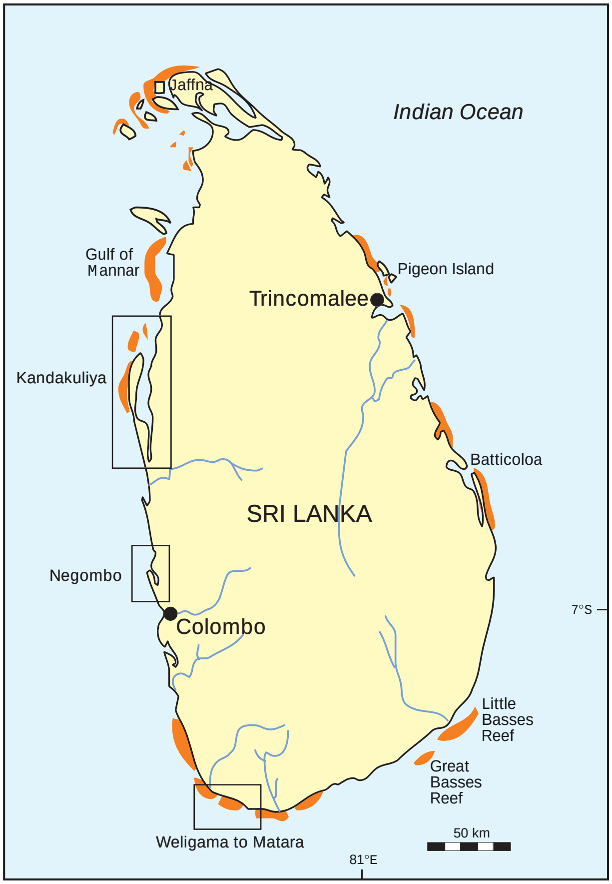

Figure 1. Study sites.

METHODS

The study was carried in Negombo, Weligama to Matara, and Kandakuliya (Fig. 1), from May 2000 to June 2001.

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 is dominated by large *Porites* domes, of which many have dead patches. After the bleaching in 1998, the amount of filamentous algae increased on coral rubble and dead coral. No reduction on fish abundance was recorded in 2000 (Rajasuriya & Karunaratna, 2000), but subsequent underwater surveys showed a decrease in surgeonfish and unicornfish (Rajasuriya, this volume).

Weligama fringing reef is located within a bay in southern Sri Lanka. Before 1998 the percentage live coral was as high as 92%, but dropped to 28% during the mass-bleaching event (Rajasuriya & Karunarathna, 2000). However, *Montipora aequituberculata* escaped the bleaching, and continued healthy growth for this species, and some recovery of *Acropora* spp., has been reported (Rajasuriya, this volume; Fairoze *et al.*, 2000). The dead parts of the reef are now covered with *Halimeda* spp. and filamentous algae. A reduction in numbers of fish, particularly butterfly fish (*Chaetodontidae*) was recorded during post-bleaching surveys (Rajasuriya & Karunaratna, 2000).

Kandakuliya reef is situated about 1 km offshore of Kandakuliya and Kudawa fishing villages, 10 km south of the shallow reef areas of the Bar Reef Marine Sanctuary. There are also near shore reef patches in the area that fringes the southern beach of Kandakuliya (Rajasuriya *et al.*, 1997), which is largely buried by sand from beach erosion along the northern shore of Kandakuliya village (Rajasuriya & Karunaratna, 2000). Also, the shallow reefs are degraded by destructive fishing methods such as for example bottom-set nets (Öhman *et al.*, 1993), and fish abundance has been reported to be low compared to Bar Reef Marine Sanctuary. The deeper offshore reefs are in better condition, and no reduction of fish population was recorded after the bleaching (Rajasuriya & Karunaratna, 2000).

Information on the general patterns of fishing activities in the study areas were gathered from primary sources, through structured interviews, participatory observations, and casual conversation. Supplementary literature studies were also conducted. Based on the initial surveys, past catch records (e.g. log books) from fishermen catching lobsters and finfish on the reefs, using special gears such as bottom-set trammel nets, encircling seine nets for trevallies, and hook and line, were collected. This was supplemented by distribution of forms for data collection. Further, to investigate the fishermen's perception of the degradation of the coral reefs, including the bleaching induced mass mortality of corals in 1998, and the changes in catch and income over time, ten special gear operators from each of three study areas were selected for a questionnaire survey.

RESULTS

Fishing practices on coral reefs

About 30 fish species caught for food consumption in Sri Lanka was classified as directly associated with coral habitats (through review of Bruin *et al.*, 1994). Groupers (*Serranidae*) represent one of the largest families of coral reef dwellers (Gerald *et al.*, 1996). The initial survey identified the main fisheries carried out, and gears used, on, or close to, coral reefs in Negombo, Weligama, and Kandakuliya (Table 1, and Table 2 on next page).

Presently there are several fishing methods used on the coral reefs in the study areas, including bottom-set trammelnets, bottom-set gillnets for lobsters, encircling seine nets for trevallies (*Caranx* spp.), drift gill nets for barracudas (*Sphyraenidae*), moxy nets (small cast nets) for ornamental fish, hook and line, and spear fishing. The bottom set trammel nets are operated at depth between 10 and 30m, and are set in the evening and hauled in the following morning. Catches mainly consist of emperors (*Lethrinidae*), snappers (*Lutjanidae*), groupers (*Serranidae*), and goatfishes (*Mullidae*). Hook and line fishery is one of the most traditional fishing methods in Sri Lanka, but the fish catches are low, and most of the fishermen are engaged in this fishery mainly for subsistence. However, when they can catch more than enough for themselves, they sell it to other people in the community. Hook and line fishery requires large quantities of small live bait fish, which are generally caught close to reefs early in the morning. Several methods are currently

Table 1. Methods for reef fisheries in three study sites in Sri Lanka

Category	Fishing Gears Used
Spiny lobsters	Bottom-set nets and diving.
Finfish	Bottom-set trammelnets, encircling seine nets for trevallies, hook and line, and spearing
Sea cucumber	Diving

Table 2. Different fishing methods used in the study areas, and their relative destructiveness to the coral reefs

No.	Fishing Gears	Negombo	Weligama	Kandakuliya
Very destructive				
1.	Bottom-set gillnets for lobsters	+	–	+
2.	Bottom–set trammelnets	+	–	+
Destructive				
3.	Encircling seine nets for trevallies	–	–	+
4.	Drift gill nets for barracudas	–	+	+
5.	Moxy nets	–	–	+
Relatively harmless				
6.	Hook and line	+	+	+
7.	Diving	+	+	+

(+) Existing; (–) Not existing

used to catch bait fish, and they vary between regions, villages and fishes. The most common methods are cyanide, hook and line and trap fishing (Bently, 1999).

Bottom-set gillnets and the bottom-set trammelnets, used in Negombo and Kandakuliya but not in Weligama, are considered the most destructive fishing gears used in coral environment. These types of nets are placed directly on the coral beds to catch lobsters and finfish, and are damaging the reefs especially when retrieved for collection of the catch. This is particularly evident from the amount of coral parts brought to the landing sites. In comparison, encircling nets for trevallies (*Caranx* spp.), and drift gillnets for barracudas (*Sphyraenidae*), are less destructive not being laid directly on top of the coral beds. However, these nets are often used close to the reefs and can cause considerable damage when entangled in the corals. The small cast nets (moxy nets) used to catch ornamental fish are also destructive to the corals. The fishing methods considered least destructive are diving, and the traditional hook and line fishing. Diving can however cause damage depending on the equipment used to catch fish. For example divers who collect sea cucumbers *(Holothuroidae*) do not usually destroy the reef structure, since it is mostly carried out using a hand net, and spear fishing is very selective, targeting big fish, and is therefore of no or minor harm to the corals.

Reef Fisheries Catch Data

SPINY LOBSTERS

The gross value of the spiny lobster catch in Sri Lanka during the financial year 1997/1998 was around US$ 20 Million. The most profitable lobster fisheries today in Sri Lanka are concentrated along the South Coast (Jayakody, 1999). According to Joseph (1993) the newly introduced techniques such as bottom-set nets and scuba diving primarily targeting spiny lobsters cause a steady decline in catches. Mostly bottom-set gillnets, but sometimes also

Figure 2. Most of the bottom-set trammelnet catches consist of coral reef associated fish such as, *Parupeneus indicus* (Indian goatfish), *Scolopsis bimaculatus* (Thumbprint spinecheek), *Lethrinus* spp. (Emperor), *Macolor niger (*Black snapper) *Sufflamen fraenatus* (Bridled triggerfish), *Acanthurus* spp. (Surgeonfish), *Apolemichthys xanthurus (Indian yellowtail angelfish) and Amphiprion sebae* (Sebae anemonefish). *Photo:* CHINTHA PERERA, NARA, Sri Lanka.

Table 3. Spiny lobster catches in Weligama, Negombo and Kandakuliya, Sri Lanka

Area	Year	Species wise catch per day/fisherman (kg) *Panulirus*					Total catch per day/fisherman (kg)	Income per day/fisherman (US$)	Source
		longipes	*ornatus*	*homarus*	*penicillatus*	*versicolor*			
Weligama	1997	1.97	0.013	0.45	0.11		2.5	35	8 months data from 22 fishermen
	1998	2.20	0.02	0.16	0.12		2.5	35	5 month data form 31 fishermen
	1999	3.06	0.014	0.1	0.01		3.2	44	6 months data from 23 fishermen
	2000	1.46	0.006	2.16	0.14		3.8	52	7 months data from 15 fishermen
Negombo	1998						1.3	19.7	12 months data from 15 fishermen
	1999						1.04	15.3	As above
	2000						1.10	16.3	9 months data from 15 fishermen
Kandakuliya	2000–01	0.30	0.11	0.12		1.40	1.93	23	2 months data from 5 fishermen

Note: For the calculation of daily catches and incomes, the estimation of 20 fishing days per month has been used consequently, and comparison in catch per unit effort (CpUE) between years is not possible here.

moxy-nets, are used to catch spiny lobsters, which are found at depths of between 8 to 30 meters. The spiny lobster species usually caught on coral reefs in Sri Lanka include *Panulirus longipes*, *P. ornatus*, *P. homarus*, *P. penicillatus* and *P. versicolor*. From 1997 to 1999, *Panulirus longipes* made up the main catch, while in year 2000, *Panulirus homarus*, became the most commonly caught lobster species in Weligama. In Negombo bottom-set gillnet is the main fishing gear to catch lobsters, while in Kandakuliya also diving and occasionally moxy-nets are used.

FINFISH

Results give an indication of the differences in daily fish landings between the fishing practices (Table 4 on next page). Worth noting is the high catch figures of the encircling seine nets for trevallies in Kandakuliya.

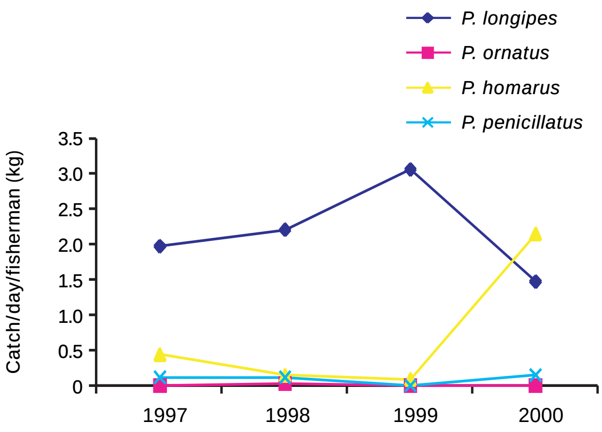

Figure 3. Daily catches of spiny lobsters, species wise, in Weligama, Sri Lanka, 1997–2000

Table 4. Reef fisheries catches in Negombo Kandakuliya and Matara/Polhema, Sri Lanka

Area	Type of fishing	Average catch per day of fishing effort/ fisherman	Income per day of fishing effort/ fisherman	Figures calculated from
Negombo	Bottom-set trammelnet	48 kg	Rs 4,842 (≈ US$ 54)	21 fishing days of 3 different fishermen in 2000
Kandakuliya	Bottom-set trammelnet	32 kg	Rs 3,200 (≈ US$ 36)	134 fishing days of 9 different fishermen in Dec. 1997 and Jan. 1998
Kandakuliya	Encircling seine net for trevallies	250 kg	Rs 25,000 (≈ US$ 277)	109 fishing days of 1 fisherman, 1997–2001
Negombo	Hook and line fishery	6 kg	Mainly for subsistence	9 fishing days of 1 fisherman, in January 2001
Matara/ Polhema	Hook and line fishery	6 kg	Mainly for subsistence	27 fishing days of 1 fisherman, in Dec. 2000.

Fishermens Perception of Factors Influencing Fish Catches

Coral reefs play a crucial role to the well being of coastal communities in Sri Lanka and unsustainable exploitation, and destruction of reef habitats, will affect the segments of the population involved in fishing more than any other group. In order to assess how the fishermen perceive the factors influencing the fish catches structured interviews were carried out in three areas. These interviews show the following:

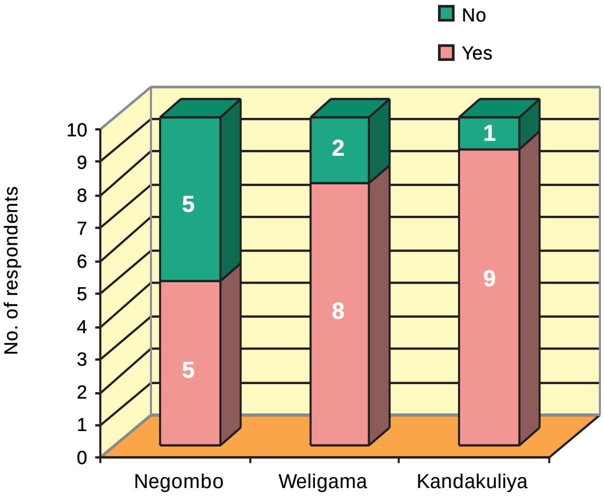

Figure 4. Fishermen's opinions on whether the fish catches declined after the mass coral bleaching in 1998.

- most of the fishermen commented that shoals of parrotfish commonly observed before 1998 were only rarely noticed after that year;
- a reduction in lobster abundance on the coral reefs had been noticed in recent years;
- most of the fishermen were not sure whether the changes had occurred before or after the bleaching event;
- overall catches on the reefs had declined since 1998 and most of the respondents indicated that coral bleaching was one of the reasons for declined catches in recent years (Fig. 4);
- most fishermen indicated that, in their opinion, the use of destructive fishing techniques as well as increasing number of fishermen were major threats to fish stocks (Fig. 5);
- the fishermen were generally of the opinion that the introduction of new fishing gear and efficient methods exacerbates the problem with decreasing catches;
- most of the respondents in three sites agreed that,

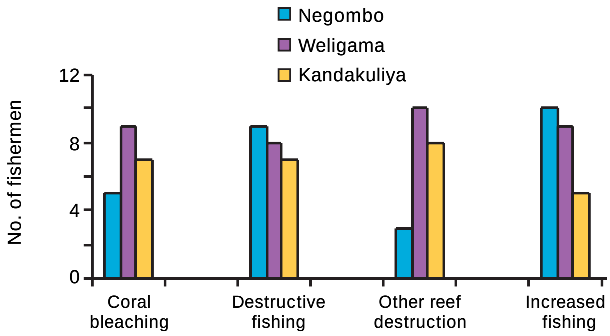

Figure 5. Fishermen's views on potential causes for decline in reef fish catches.

except for natural disasters, destructive fishing gears are the main cause for coral degradation (Fig. 6);

- all respondents in Negombo and Kandakuliya, where bottom-set gillnets are used, identified this gear type as the most harmful for coral habitats;
- many of the interviewed fishermen in all three sites pointed out bottom-set trammelnets as being among of the most destructive fishing gears;
- the interviewed fishermen also considered discarded nets and litter such as polythene shopping bags as a major problem causing coral degradation;
- collection of live corals and dynamite fishing were mentioned as problems in Weligama and Kandakuliya;
- predation on corals by the crown-of-thorns starfish was mentioned at all three sites.

DISCUSSION

The coral reefs of Sri Lanka are under serious stress. At the same time the reef related fishery is facing decreasing catches. The present study has shown that, in the opinion of the fishermen, the mass mortality of corals in 1998 had adverse impacts on reef fisheries catches. However, many of the fishermen were uncertain about the actual timing of the decline in catches. Most of them claimed the fish catches were declining also before the bleaching event, and that the decline was a result of over-fishing and destructive fishing practices. Where pre- and post bleaching catch data is available in the log books, it does not give any indication of major changes

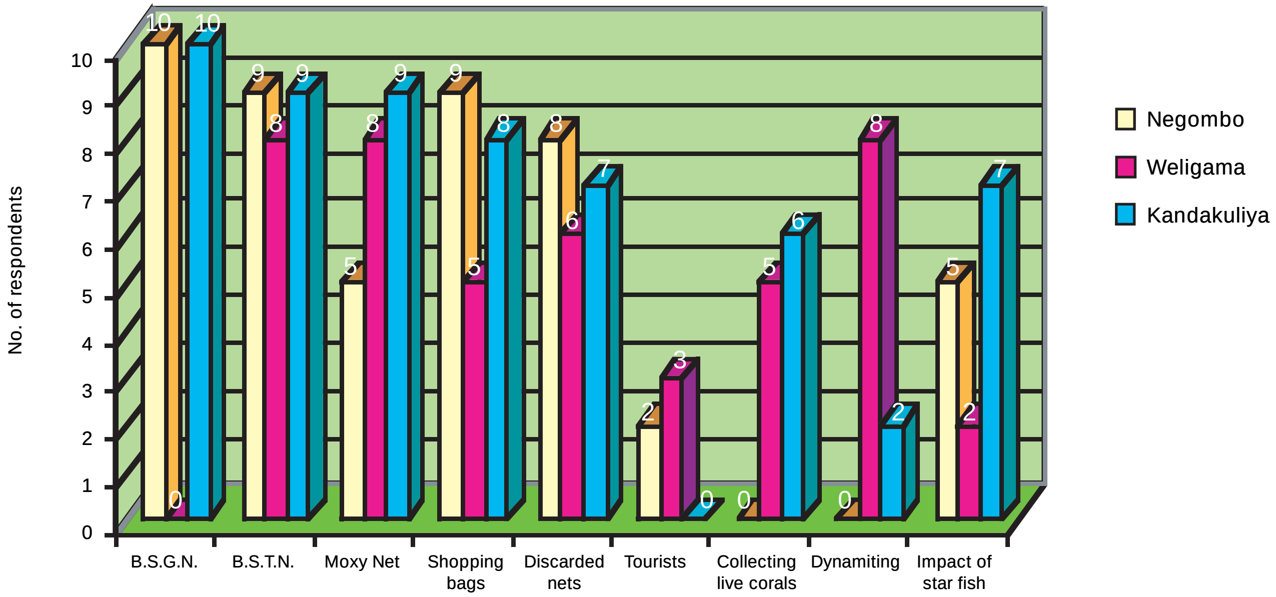

Figure 6. Fishermen's ranking of causes for degradation of coral reefs in their fishing areas.

over time. On the other hand the analysis of the finfish catch data is based on fishing efforts in terms of number of days. Thus, the daily effort may have increased to maintain a constant income, even if the data do not admit any conclusions in this regard. Changes in the catch composition over time may provide further information. The shift in the catches from *Panulirus longipes longipes* to the less coral associated *Panulirus homarus* during year 2000 (Fig. 3) is of interest. Further monitoring needs to verify whether this is a long-term trend or not. The figures from the encircling seine net targeting trevally during 1997–2001 (Table 4) suggest this reef fishing practice to be the most effective in terms of fish landings and earnings for the fishermen. However, when considering the annual income for these fishermen, it should noted that according to the same log books, the number of fishing days per month average only around 6.6. The catches of this particular gear mainly consist of trevallies (*Caranx* spp.), trumpet emperors (*Lethrinus miniatus*), spangled emperors (*Lethrinus nebulosus*) and barracudas. This is an illegal fishing method and only a few fishermen are engaged, opposing competition from any new fishermen taking up this practice.

Interestingly, the seine net fishery in Kandakuliya is both less destructive and more effective than for example the bottom-set nets in the same area and in Negombo. However, also bottom-set trammelnet as well as the seine net fishing can apparently harvest significant fish catches and earn considerable amount of money. It should be pointed out that these methods are banned in the *Fisheries Act*. However, fishermen are still operating these kinds of fishing gears due to the good income they generate. The results also indicate that the fishermen are aware of the negative impacts of the fishing gear on the reef habitats, and the likely long-term consequences for the fish catches.

The system of free entry to reef resources, leading to an uncontrolled increase in fishing pressure, largely from destructive fishing methods, is a serious threat to the coral reef habitats of Sri Lanka. Sustainable use and management of reef fishery resources is vital in the country, and improvement of management is urgently needed. Therefore, it is recommended that laws and regulations aimed at preventing destruction of reefs and over-fishing should be implemented. In the study sites priority should be given to the Kandakuliya area, where the destructive methods at present seem to be more widely used than in the other two sites. The contribution to total fishery landings made from offshore fishery is increasing (NARA, 1998). However, the present harvest is lower than the estimated production from offshore sea areas (UNEP, 2001). This may indicate that the offshore fishery resources can provide an alternative to fishing in near shore areas. However, further studies are required to verify this. Moreover, actions to increase the fishing effort in offshore areas need to be carefully monitored and managed in order to avoid simply adding to the problem of overfishing in the country. Nevertheless, it is recommended that the illegal use of bottom-set gillnet and bottom-set trammelnet is stopped, and that the operators are provided opportunity to shift towards alternative livelihoods.

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Monitoring the Trends of Marine Ornamental Fish Collection in Sri Lanka

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INTRODUCTION

Marine organism, including fish, corals, clams, sea anemones, echinoderms, and soft corals are being collected and exported from most of the world's coral reef areas (Philipsson, 1989) to supply the aquarium hobbyists and public aquaria, mainly in United States, Europe and the Far East (Holthus, 2001). In 1986 the global ornamental trade was calculated to a value of around US$ 7.2 billion (Andrews, 1990; Holthus, 2001). Around 10% of the fish involved is thought to be marine species (Holthus, 2001), and it is estimated that between 14 and 30 million coral reef fish are annually collected for the global aquarium fish market (Wood, 2001). The trade includes around 1000 species of fish, and the main supplying countries are the Philippines and Indonesia covering about one third each of the global market (Wood, 2001). Other important exporting countries are Brazil, Maldives, Vietnam, USA (Hawai), and Sri Lanka (Wood, 2001).

Sri Lanka was one of the pioneer countries in the collection and export trade of coral reef fish, with small-scale collectors operating off Colombo in the 1930's (Jonklaas, 1985). The trade then expanded during the 1950's as the air fright facilities developed (Wood, 2001). The high prices for marine aquarium fish do not attract the domestic market in Sri Lanka, and thus the collection is practiced entirely for export (Wijesekara & Yakupitiyage, 2001). There are no official statistical records on the export value specifically for the marine fish. However, today the aquarium fish sector as a whole constitutes around 1.4 % of Sri Lanka's total export trade (EDB, 2000), and ranks the third highest in volume and value among the fishery export products after prawns and lobsters (Baldwin, 1991). Marine, freshwater, and brackish water species of fish and invertebrates, at a total value of approximately 8 million US$ (EDB, 2000), are exported from Sri Lanka to around 52 different countries throughout the world, the major markets being Europe, USA, and Japan. (EDB, 1999). It is estimated that marine fish constitute about 70% (Wijesekara & Yakupitiyage, 2001) of Sri Lanka's total export value of ornamental organisms. Further, nearly all the marine ornamental fish is collected on or around coral reefs (Wood, 2001), and the industry is not only directly dependent on the productivity of the reefs, but also, as it is largely practiced today, cause damage to the same. This paper presents the nature of marine ornamental fish collection and trade in Sri Lanka. Also, it reports on the progress and findings up to date of the ongoing CORDIO project, assessing and monitoring the trends of the country's ornamental fisheries industry.

THE ORNAMENTAL FISH COLLECTION AND TRADE IN SRI LANKA

Collection and Supplying to the Sri Lankan Export Market

There are around 60 ornamental fish export companies operating in Sri Lanka (EDB, 2000), but figures from 1998 show that more than 80% of the total export was handled by only five regular export companies (EDB, 1998). Some exporters deal with either marine or fresh water species, but many exporters handle both categories (Haputhantri *et al.*, 2002). Further, marine fish are mostly obtained from fish collectors in Sri Lanka, but about 7% (by value) of the specimens are brought in from mainly the Maldives for re-export to supply the export market (Wood & Rajasuriya, 1996a). Since no technology for breeding of marine ornamental fish has been successfully introduced in Sri Lanka, the export totally depends on wild-caught specimens. Cleaner Wrasse (*Labroides dimidiatus*), Powder-blue Surgeonfish (*Acanthurus leucosternon*) and Three-spot Damsel (*Dascyllus trimaculatus*) are among the most commonly collected fish species (Ekaratne, 2000), but most genera are represented in varying numbers. The number of ornamental fish collectors currently involved in Sri Lanka is difficult to estimate. There is no system for issuing of permits for collection, and no records are maintained. However, according to some of the ornamental fish exporters, the total number, including both full-time and part-time involved collectors, is more than 1000 (Haputhantri *et al.*, 2002). The same figure was presented in Wood, 2001. Most of the full time collectors are scuba divers operating in groups, while part time collectors usually limit their activities to snorkelling in shallow waters around their home villages. Some of the largest exporters maintain their own teams of divers, but most of the exporters, or their agents, purchase the fish needed to fulfil the export orders from independent collectors. Further, there are so called suppliers acting as middlemen between the collectors and exporters. A significant portion of these suppliers are also ornamental fish collectors themselves (Haputhantri *et al.*, 2002).

Figure 1. Over 200 species of marine fish are collected in Sri Lanka to supply the aquarium hobbyists and public aquaria in mainly USA, Europe and Japan. *Photo:* OLOF LINDÉN

Many fish collectors use relatively non-destructive hand-nets to catch the fish. However, the use of small cast nets, also called moxy-nets, are causing significant destruction on the reefs. The moxy nets, which have been banned but are still in extensive use, are placed around the coral colonies or other structures where the desired fish is hiding. By rocking or hitting the hiding place with an iron bar, the fish are frightened into the surrounding net. Marine ornamental fish collection is allowed throughout the coast. There is a formal ban on ornamental fish collection in Hikkaduwa Nature Reserve and Bar reef Sanctuary, which have been declared as protected areas under the *Sri Lanka Fauna and Flora ordinance*. Collectors operate mainly along the west, southwest and south coasts with the activities concentrated in Chilaw, Negombo, Colombo, Galle, Unawatuna, Weligama, Polhena, Matara and Tangalle. On the East Coast the main centers are Batticaloa and Trincomale. On the northwest coast, the key centers are Kuliyapitiya and Kandakuliya. Fishing activities in the north and northeast parts of Sri Lanka has been limited by the prevailing political situation. Further, the organized scuba diving fish collectors normally move around according to the seasonality of the sea conditions. In general, the monsoons restrict the fish collection on the east coast from April to October, and on the west coast from May to November (Haputhantri *et al.*, 2002).

Management Issues

Estimations of the annual number of ornamental fish exported from Sri Lanka indicate an increase from around 200,000 in 1985 (Wood, 1985) to around 1 million at present (UNEP, 2001). Not only the number of collectors involved in ornamental fish collection has increased, but there has also been a gradual diversification to a wider range of species from 139 in 1985 (Wood, 1985) to over 200 today (Ekaratne, 2001; Monagurusamy & Dhanasiri, 2001; Wood, 2001). Seven species of marine ornamental fish are protected under the *Fauna and Flora Protection (Amendment) Act 1993*, and since 1996 also under the *Fisheries Ordinance*. In addition, the *Fisheries Ordinance* lists sixteen species of marine ornamental fish that are exportable only on permit (Wood & Rajasuriya, 1996b). Although systematically collected evidence is scarce, overfishing of ornamental fish in general, and extensive harvesting of certain rare or popular species may cause changes in the fish community composition (Tissot, 1999; Wood, 2001), and threaten the biodiversity of the reefs. In addition, the illegal destructive fishing methods still in use for ornamental fish collection (e.g. Moxy nets) continue to damage the coral reefs.

Other human impacts threatening the reefs of Sri Lanka include destructive fishing methods for food fish, coral mining increased sedimentation on the reefs due to poor land use practices, pollution, boat anchoring and tourism related activities (De Silva, 1985; Baldwin, 1991; Öhman *et al.*, 1993; Rajasuriya & White, 1995; Rajasuriya *et al.*, 1998). In addition, periodic infestations of the crown-of-thorn star fish threaten the reefs in the north-west and the east coast of Sri Lanka (Rajasuryia & Rathnapriya, 1994; De Bruin, 1972). Furthermore, during the severe El Nino event with the resulting elevation of sea surface temperatures in 1998, the shallow coral reefs of Sri Lanka suffered extensive bleaching and subsequent mortality. While the deeper reefs, below approximately 10 meters, to a large extent recovered from the bleaching, between 50% and 90% of the corals in most shallow areas died (Rajasuriya & Karunarathna, 2000). Surveys conducted in 1998 revealed that many of the dominant forms of the reef building corals, such as *Acropora* spp., *Pocillopora* spp., and *Echinopora lamellosa* in the shallow coral habitats had been destroyed (Rajasuriya *et al.*,1999). Further, recovery of the coral reef habitats has in general been poor (Rajasuriya & Karunarathna, 2000; Rajasuriya, this volume). This dramatic decrease in live coral cover on the reefs, and a subsequent erosion of the coral skeletons by boring organisms and wave action, has altered the community structure on the reefs, and resulted in decreased complexity of the reef framework. Further, resent studies on the post bleaching status of the coral reefs of Sri Lanka (Rajasuriya & Karunarathna, 2000) revealed a significant reduction in the number of butterfly fish after the bleaching event in 1998.

Although generalizations of the effects of habitat changes on the reef fish may be difficult due to the influence of a variety of different factors (Öhman, 1999), the destruction of the coral reef in Sri Lanka is likely to negatively affect the abundance and diversity of the fish inhabiting the reefs. Habitat effects on reef-fish communities have been described in a number of studies, as assemblages may associate with distinct habitat units within reefs at various scales (Galzin, 1987; Russ, 1989; Letourneur, 1996; Öhman *et al.*, 1997) Also, positive correlations between fish abundance or diversity with general habitat descriptors such as structural complexity or live coral cover have been pointed out in several studies (Sebens, 1991; Williams, 1991; McClanahan, 1994; Jennings *et al.*, 1996; Munday *et al.*, 1996; Öhman & Rajasuriya, 1998; Öhman *et al.*, 1998; Turner *et al.*, 1999). Consequently, since the vast majority of ornamental fish are collected on coral reefs, and many of the collected species only occur within healthy coral reef systems, the sustainability of this industry and livelihood is threatened by the rapid degradation of the coral reef habitats in Sri Lanka.

ASSESSING THE TRENDS IN MARINE ORNAMENTAL FISH COLLECTION AND EXPORT IN SRI LANKA

At present, insufficient information is a major problem in studying the trends in the collection and trade of marine ornamental fish in Sri Lanka. As mentioned before, no licenses are issued for fish collection, and only a few ornamental fish collectors and suppliers keep records for longer periods. Since the entire marine ornamental fish collection is practiced for the export market any significant changes in the total catch of ornamental fish in Sri Lanka should be reflected in the amount of fish being exported. It should be noted here that according to the exporters there is a constant high demand for aquarium fish for the export market (Haputhantri *et al.*, 2002). However, as with most other countries, Maldives and Singapore being among the exceptions (Wood, 2001), there is a substantial gap in the available information on fish species and numbers of specimens exported from Sri Lanka. The information provided in government statistics only outlines the value and the weight of the products, including the water it is transported in. In addition, the statistics do not treat marine fish, freshwater fish and plants as separate items.

The Marine Aquarium Council (MAC) and the World Conservation Monitoring Centre (WCMC) has established a data base (Global Marine Aquarium Database) on the global aquarium trade, providing country specific import and export data, building on information from export and import companies. However, it is under development and at present only a selection of the exporters in Sri Lanka contributes the data. Moreover, the actual abundance and availability of fish for the collectors needs to be estimated through catch per unit effort (CpUE), or if done in retrospect, by structured interviews with the fish collectors. This information can further be combined with data from the CORDIO supported underwater surveys in Sri Lanka.

In this CORDIO project, carried out by NARA (National Aquatic Resources and Research Institute, Ministry of Fisheries and Aquatic Resources), a mechanism for enhanced monitoring and management of the marine ornamental fishery resources in Sri Lanka is being established. The monitoring program, focusing on the trends in marine ornamental fish collection and trade in Sri Lanka, comprises the following components:

- Analysis of secondary trade statistics, to identify any major trends in the ornamental fisheries export from Sri Lanka.
- Assessment of the impacts of coral reef degradation on marine ornamental fish collection through structured interviews with collectors and suppliers, and analysis of log books.
- Development of a database built on export packing lists from the customs department, to fill the existing information gap.

The hope is that these actions will create a tool for monitoring of the marine ornamental fisheries in Sri Lanka, as well as a foundation for sustainable management of the resource.

Results up to Date

EXPORT STATISTICS

During the 1980's the aquarium fish sector as a whole increased with approximately 150% (Beets, 1994), and in most of the 1990's a rapid increase was registered, followed by a drop in the end of the decennium (Fig. 2 on next page). Poor performance of three big export

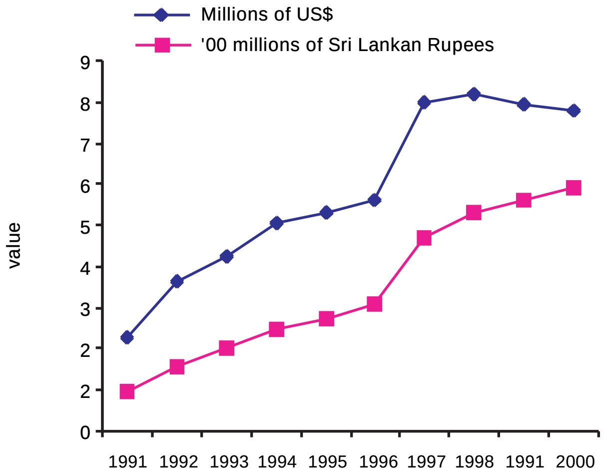

Figure 2. Export trade of marine and freshwater fish, water plants, and invertebrates, for ornamental purposes, during the period 1991–2000 (*Source:* EDB, 2000, and *Fisheries Year Book*, 2000).

companies contributed to the setback. Export of aquarium fish to Japan, Germany and U.K. declined in 2000, while export to USA showed a substantial increase during the same year (EDB, 2000). The Sri Lankan Rupie (SLR) has been devaluated compared to the U.S. dollar, explaining the slightly different patterns of the curves.

COLLECTORS AND SUPPLIERS PERCEPTIONS OF THE TRENDS IN ORNAMENTAL FISH COLLECTION

12 ornamental fish suppliers were interviewed to obtain information on trends of collecting and supplying of marine ornamental fish during the last 10 years (Haputhantri *et al.*, 2002). At the time of the interviews these suppliers were based in Unawatuna, Weligama, Polhena (south coast), Negombo (west coast) and Kandakuliya (southwest coast). All the suppliers had more than ten years experience of ornamental fisheries and all except one were also directly engaged in the fish collection. According to the responses, there has been a declining trend in marine ornamental fish abundance over the last five years (Haputhantri *et al.*, 2002). A detailed analysis of the responses shows that:

- periods of maximum fish collection took place before 1995;
- periods of minimum collection period took place 1996-2000;
- coral bleaching was considered to be one of the main causes for the declining catches;
- some recovery had been noted since the end of 2000;
- the total effort to catch ornamental fish has increased significantly over the last few years.

Results suggest that overexploitation of the resource is likely to have contributed to the declining trends in ornamental fish abundance. Reasons mentioned for their individually poor collection of marine fish over the past years include:

- coral bleaching in 1998;
- increased number of ornamental fish collectors;
- blast fishing;
- increase in other destructive fishing practices;
- coral mining;
- water pollution;
- non-biodegradable waste dumping;
- use of destructive methods for ornamental fish collecting.

DEVELOPMENT OF A DATA BASE

A data base on marine ornamental fish export data, based on packing lists submitted to the customs by the exporters, has been established by Haputhantri *et al.* (2002), as a tool for monitoring of the trends in the trade, and for more sufficient management of the resource. Export statistics, however only available from December 1998 and onwards, was extracted from the packing lists stored at Sri Lanka Customs Departments, and entered into the database. A packing list prepared by the exporter should be presented to the department with every consignment of ornamental specimens for export. The packing lists provide the details of the contents of each box by species, number of specimens and price. However, many exporters are reluctant to provide the export values of the species. Data on 52 species from all the packing lists of 1999, except for the missing month of September, and packing lists from December 1998, January, February, June and August 2000 have so far been included in the database (Haputhantri *et al.*, 2002). Available data for the remaining months, and additional species, are being processed at the moment.

Of particular interest are if any effects of the mass mortality among corals in 1998 can be traced in the data. It should be noted that little information is available from the period before 1999. However, Wood and Rajasuriya (1996a) examined a large sample of packing lists for the period August 1995 to July 1996, and estimated annual export numbers at species level. This allows for a comparison of export data for 31 species before and after the 1998 coral mortality. The total number of specimens annually exported from the 31 species showed an increase of 4% between Wood and Rajasuriya's study in 1995/96 and Haputhantri's *et al.* data for 1999. However, this was attributed to a 60% increase in export of the herbivorous Powder-blue surgeonfish (*Acanthurus leucosternon*), from 12,061 to 19,299 specimens. Excluding the Powder-blue surgeonfish, the total for the remaining species had decreased with 9%. The export data of the four butterflyfishes (chaetodontids) feeding exclusively on coral polyps (*Chaetodon meyeri, C. octofasciatus, C. trifasciatus, C. trifascialis*), that were included in the comparison, indicates a 59% decrease between 1995/96 and 1999.

Figure 3. Around 1 million coral reef fish are exported from Sri Lanka each year. *Photo:* OLOF LINDÉN

Discussion

There has been a rapid increase in the total export of ornamental fisheries export during the 1980's and most of the 1990's (Fig. 2). In addition, the data presented suggest a reduction in the catches for the ornamental fish collectors in the last few years. This reduction appears to have been caused by over-harvesting of fish probably in combination with a decrease in absolute numbers of fish due to the coral mortality in 1998. Further, a decline in total collection and export is shown for many species. Öhman (1998) demonstrated a positive correlation of coralivores and live coral cover, and the fish species expected to be directly affected by the major decrease in live coral cover in 1998 are the obligate coralivores. Correspondingly, the obligate coralivores in this study showed the most significant decrease in export numbers. Regarding the high export numbers for the herbivorous Powder-blue surgeonfish it is worth to mention the increased abundances of herbivores, benefiting from enhanced algal growth after the bleaching event, that has been reported for some reefs in the Indian Ocean (Lindén & Sporrong, 1999; Lindahl *et al.*, 2000). Some other species also show an increase between 1996 and 1999. However, there is no export data available from mid 1996 to the end of 1998. Thus, in theory, a continued increase between 1996 and 1998 could conceal any stagnation or drop for these species after the mass-bleaching event. Also, if the export data for the 31 species in this study is representative for the whole ornamental fish sector, the 4% increase (including Powder-blue surgeonfish) over three years is remarkably low compared to the rapid expansion during previous years.

Interesting to note is the recent growth of the fresh water ornamental fishery sector (Haputhantri *et al.*, 2001), which could, in the unspecific secondary export statistics (Fig. 2), over-shadow any major changes of the marine sector.

As mentioned above, according to fish collectors and suppliers, there is a constant high demand for the ornamental fish. If this is the case the lower export numbers of many species in 1999 compared to 1995/96 could be explained by decreased abundance of these fish on the reefs. It should be pointed out the export statistics alone is not enough to explain the rather complex picture probably consisting of fluctuations in demand, variations in prices etc. This limits the ability to directly link the export figures to the actual availability of the fish species on the reefs. In addition, the overall export numbers of the obligate coralivorous butterflyfish referred to above are relatively low, making the data more sensitive to random fluctuations. The prices of the exported fish could reflect changes in availability, but not having access to information on pre-bleaching prices at the moment, no such analysis has been done. Nevertheless, some of the exporters have stated that they are having difficulties to fulfil their export orders. However, natural fluctuations in abundance of certain species in major collection areas, influenced by variability in recruitment patterns, food availability or predation, may also affect the export numbers, making it difficult to directly relate any changes in fish abundance to the health of the coral reefs.

The biophysical monitoring of the reefs of Sri Lanka, also carried out within the CORDIO program, provides information that can be compared with the export data. The underwater surveys have shown a dramatic decrease in number of butterflyfish on several reefs after the mass coral bleaching and mortality in 1998 (Rajasuriya & Karunaratna, 2000). For example, only a total of 10 butterflyfish fish where recorded in seven transects during underwater surveys in 1999 in the Bar Reef Sanctuary in north-western Sri Lanka (Rajasuriya & Karunarathna, 2000), where previous surveys in 1993–94 showed an average abundance of approximately 10 chaetodontids per transect (50m Line Intercept Transect (LIT) (Öhman *et al.*, 1998). Furthermore, similar densities of 10 chaetodontids per transects where found on Weligama and Hikkaduwa lagoon reefs in 1997 (Öhman *et al.*, 1998), to compare with post bleaching surveys during 1998–2002 showing average densities of 2.4 and 6 cheatondontids per transect in Weligama and Hikkaduwa respectively (unpublished data submitted to CORDIO by Rajasuriya *et al.*). There was no *Chaetodon trifacialis* exported at all during 1999, compared to approximately 408 specimens in 1995/96, and the export number of *C. trifasciatus* was 38% lower in 1999 (1152 in 1996 to 710 in 1999). Correspondingly, the post-bleaching underwater surveys indicate a dramatic decrease of both these species compared to Öhman's studies in 1993/94 and 1997. Further, the export of the filefish *Oxymonacanthus longirostris* that feeds exclusively on *Acropora* polyps, was 94% lower in 1999 compared to 1995/96. Interesting to note is that in the Maldives this species, previously being among the most commonly exported ornamental fishes, is nearly absent on the coral reefs today (Saleem & Naeem, 2000). In conclusion, the available information presented above suggests a declining availability of the marine ornamental fish, and that this decline is correlated with the extensive degradation of the coral reefs in Sri Lanka. The database of marine ornamental fish exported from Sri Lanka is under development, and the monitoring program will hopefully provide deepened knowledge on the long-term trends and nature of this industry.

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Future Effects of Climate Change on Coral Reefs and Mangroves in South Asia

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Key words: climate change, coral reefs, Mangroves, South Asia

INTRODUCTION

Coral reefs are highly vulnerable to the effects of increased water temperatures, changes in precipitation, cloudiness, wave activity, ocean circulation and chemistry, and sea-level rise. Current climate models predict globally-averaged surface air temperature increases of between 1–5°C by 2100. Mean global sea-surface temperature is expected to rise by around 2°C, with possible greater regional and local increases. Sea-level projections predict a globally-averaged rise of up to 90 cm by 2100. There are also predictions of possible changes in storm intensity and frequency, and disruption of monsoon systems. In the South Asia Region, where so much of the coastline is subject to strongly seasonal (associated with two monsoons) and inter-annual (El Niño) weather patterns, storm events, and sea-surface temperature fluctuations, the task of understanding the effects of climate change poses many challenges. South Asia covers a large geographical area, with numerous islands, and densely populated low-lying coastal areas. It has large areas of geographically scattered and diverse coral reef and mangrove systems. Many coastal communities, often reliant on subsistence-level practices, will be particularly hard-hit by climate change. The Region offers opportunity for comparative studies of the interplay between climate change and 'direct' human impacts along heavily populated coastlines with areas where the local population is minimal and climate change is the primary factor affecting the coastal ecosystems. Up to now, very little information has been collected in terms of monitoring climate change and the associated effects on coral reefs and mangroves in the South Asia Region. This report gives a climatological background for the Region and provides a summary of some of the climate-related studies on coral reefs and mangroves that have taken place to date. It discusses projections for future climate change and the possible effects on coastal ecosystems in the South Asia Region. It is hoped that this report will serve to motivate research efforts and monitoring programmes to enhance our understanding of the relationship between climate change and coastal ecosystems in this often-neglected region. These studies are vital if we are to make attempts to mitigate the negative impacts at the political, socio-economic and conservation levels. This report draws heavily on information in the most recent Intergovernmental Panel on Climate Change (IPCC) Reports and the Global Coral Reef Monitoring Network's (GCRMN) 'Status of the Coral Reefs of the World 2000'. Much of the discussion below applies not only to the South Asia Region, but also on a global scale.

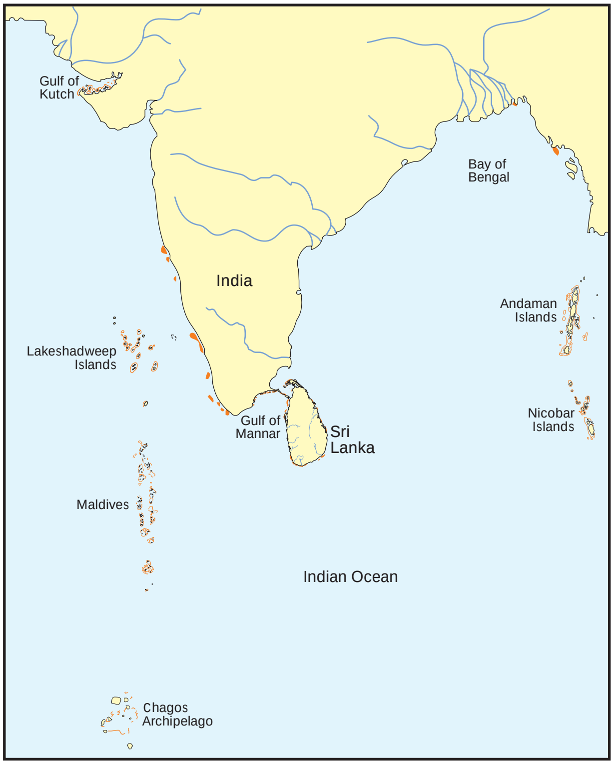

Figure 1. The distribution of coral reefs in South Asia.

CURRENT KNOWLEDGE

Climate Change

South Asia, with the interplay of the Gulfs of Arabia and Bengal, the complexities of two annual monsoon systems, and the Region's susceptibility to major sea-surface temperature (SST) excursions during El Niño events, is climatically complex. Seasonally, the climate is dominated by two monsoons: the summer southwest monsoon influences the climate of the Region from May to September, and the winter northeast monsoon occurs from November to February. These monsoons bring most of the Region's precipitation. Indian Ocean SST's affect the dynamics of the monsoon system (Zhu & Houghton, 1996). A large part of the Region experiences annual droughts and floods. The northern Indian Ocean is affected by tropical cyclones (on average 5.5 cyclones per year (1958–1984) (IPCC, 1997)), with Bangladesh being particularly affected. No long-term trend in mean rainfall has been discernible over the period 1958–1984 for the South Asia Region (SAR) or individual countries, although many countries have shown a decreasing trend in the past three decades. Similarly, no identifiable change in the number, frequency, or intensity of tropical cyclones has been observed in the Bay of Bengal and Arabian Sea over the past 100 years; however, substantial decadal-scale oscillations have occurred, with a rising trend during 1950–75 and a declining trend since that time (Gadgill, 1995). The southwest monsoon in India has shown definite changes in the period 1961–1990, in comparison with either 1901–1930 or 1931–1960, though no consistent longer-term trend is detectable (IPCC, 1997).

The globally-averaged surface temperature has increased by 0.3–0.6°C during the 20th Century (IPCC, 2001). Over the past 100 years, mean surface temperatures across the SAR have followed a similar increase, in the range of 0.3–0.8°C (IPCC, 1997). Wilkinson (2000) reports that SST's in the Indian Ocean have risen by 0.12°C per decade over the last 50 years.[1] The SAR displays marked seasonal fluctuations in SST associated with the monsoons, and interannual variability associated with El Niño. El Niño plays an important role in interannual temperature, rainfall and sea level variability. El Niño is generally linked with a drier southwest and wetter northeast monsoon. Warm episodes associated with El Niño appear to have become more frequent, longer, and more intense since the mid-1970's compared to the last 100 years (IPCC, 2001). Studies suggest that the very large 1982/83 and 1997/98 warm events, only 14 years apart, might be outside the range of variability of the past few centuries. Despite the recent strong El Niño episodes, the inverse relationship between El Niño and the Indian summer (southwest) monsoon (weak summer monsoon arising from an El Niño event),

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[1] Detailed maps of SST's can be viewed, for example, at the Tropical Ocean Global Atmosphere programme (TOGA) and the US National Atmosphere and Ocean Administration (NOAA) websites).

which operated on multi-decadal time-scales, has broken down in the last two decades (Kumar *et al.*, 1999a,b).

Global mean sea level has risen 10–20 cm over the last 100 years, with no detectable acceleration of sea-level rise during this century (IPCC, 2001). There are currently major difficulties in discerning regional sea-level trends for the SAR due to lack of tide-gauge data (only Mumbai has a record that is longer than 75 years), in addition to high decadal and interannual variability (such as changes associated with El Niño). Statistics from the Khulna region, Bangladesh, with extensive mangrove systems, show a sea-level rise of 5.18 mm/year. In some cases, shoreline erosion results in observed local sea-level rise, unassociated with the effects of climate change.

Effects of Climate Change on Coral Reefs and Mangroves

To date, the focus of work that has been carried out in the South Asia Region on climate effects on coral reefs has concentrated on the degree, areal extent, and consequences of the 1998 bleaching event associated with SST increases caused by El Niño. In past decades, there have been cycles of changes in coral cover and species diversity, which probably correlated with SST fluctuations. In some areas, such as Maldives, these fluctuations have been documented, (W. Allison, pers. comm.). However, the 1998 bleaching event was unprecedented in both extent and severity and seriously damaged almost all reefs in South Asia, with losses of around 60–80% of live coral cover. In many areas during this event, SST's rose by as much as 5°C for periods of up to several months. The warm pool of water was first observed in satellite images from NOAA in January 1998. Warm surface waters migrated from south to north during the first six months of the year. A short summary follows, highlighting the main findings of the bleaching event caused by the elevated SST's in the SAR in the first half of 1998. Details are outlined in the GCRMN Status of Coral Reefs of the World reports (Wilkinson, 1998, 2000) and the CORDIO Status Report 2000 (Souter *et al.*, 2000).

The warming event destroyed many of the shallow-water corals, in the previously remote healthy communities of Maldives (losses of up to 80% on many reefs), Chagos and Ladshakweep, as well as around Sri Lanka and the Gulf of Mannar of India, which were all particularly badly hit. There are some anecdotal reports of bleaching in the Gulf of Kachchh and the Andaman and Nicobar Islands, but bleaching in the latter appears to be less severe than elsewhere. In the Andaman and Nicobar Islands, there has been conflicting information with initial anecdotal reports of up to 80% bleaching mortality of coral (Pet-Soede *et al.*, 2000). However, subsequent reports concluded that there was actually very little bleaching in these areas (K. Venkataraman, pers. comm.). The greatest impacts were at depths of less than 10 m, among branching and tabulate corals. There is still a lack of data on the impacts and recovery of reefs from this event and much of the information gathered has been based solely on anecdotal accounts, especially for remote reefs. Species composition shifts in colonizing corals have been documented in some areas, such as Maldives (Clark, 2000; Zahir & Clark, CORDIO Newsletter, Oct 2001, with a shift from branching to massive corals.

In many places, the bleaching is similar, but more severe, than during the 1987 El Niño, which caused some mortality, but reefs in Maldives were observed to recover quite quickly from this event. In general however, recovery from the 1998 event has been slow, and in many areas close to human populations, there has been little or no recovery, often accompanied by significant disintegration of reef structures (e.g the southwest coast of Sri Lanka (Rajasuriya & Karunarathna, 2000)).

Mangrove areas, which play a key role in maintaining the healthy balance of the coastal ecosystem, are expected to be prone to the effects of sedimentation from debris of damaged bleached reefs, but currently studies are lacking.

Presently, there exists a need for the expansion of existing reef monitoring programmes to include additional sites and mangrove habitats There is a lack of current studies on the impacts of sea-level change on both coral reefs and mangrove systems in the SAR. This is particularly crucial in those areas most likely to be exposed to the highest risks.

POSSIBLE FUTURE SCENARIOS

Climate Change

Up until now, the majority of the predictions for future climate change scenarios by climate scientists have been on the global level. Closing in on the regional level is, of course, extremely difficult, and the level of uncertainty rises. The task of modelling regional climate is complicated by the interplay of global, regional and local forces, in addition to teleconnections from distant regions. The climate dynamics of the SAR are not thoroughly understood, and there are few existing studies of climate projections. The coral research community will need to concentrate on the collection and consolidation of climatic data, and work closely with climate modellers to produce climate scenarios for the coming decades. The following paragraphs on climate summarize work highlighted in IPCC (1997).

Modelling studies by Whetton (1994) and the Climate Impact Group (1992) suggest that surface land-sea temperatures will increase throughout most of the SAR, although they do indicate that the amount of warming will be less than the global average. Whetton (1994) predicts a regionally-averaged warming of up to 3°C by 2070. There are expected to be large regional differences, with coastal areas and islands experiencing lesser warming. Model results described in Supiah (1994) indicate strengthening of the monsoon circulation and an increase in wet-season rainfall under increased warming, i.e. an increase in summer rainfall in the southwest monsoon region and an increase in winter rainfall in the northeast monsoon region. Increases could be greater at the local level. Average rainfall intensity and frequency of heavy rainfall events are expected to increase (Whetton *et al.*, 1994). Current modelling capabilities do not allow for reliable prediction of future changes in storm and cyclone activity. It is unclear whether the area affected, frequency, timing, intensity, or duration will change. Recent modelling studies indicate that increases in intensity, if they do occur, will be small. The uncertainties in predicting future climate impacts of aerosols, of particular significance over the Indian landmass, are great and they are likely to be extremely variable temporally and at the regional level. It is likely that they could have a significant effect on the summer monsoon since aerosols may produce a smaller warming over the land of the Indian subcontinent than in the adjoining ocean (Lal *et al.*, 1995). This will reduce the land-sea thermal contrast, which is the primary factor responsible for the onset of summer monsoon circulation. Therefore, expected increased concentrations of aerosols may actually produce a decrease in summer monsoon rainfall, in contrast to the results from the studies mentioned above. Data from an ongoing international research programme called INDOEX (the Indian Ocean Experiment) into the climatic effects of aerosols over the Indian subcontinent and tropical Indian Ocean have recently shown an expansive dense cloud of particulates that forms a ‘brown haze’ which extends beyond the Indian landmass across large areas of ocean. The climatic effects of this unexpectedly vast area of haze are likely to be considerable.

Currently, predictions about future changes in El Niño are still limited by the ability of models to simulate the El Niño system accurately. Palaeoclimate studies utilizing stable isotope thermometry on coral cores to determine frequency, intensity and areal extent of El Niño conditions under globally warmer conditions in the past, provide valuable information for predicting future behaviour under global warming.

The mean predicted rise in global average SST is around 2°C (IPCC, 2001), with most modelling studies restricted to projections of combined land-sea surface temperatures. The author is unaware of existing modelling studies which are capable of reliably predicting increases in SST at the regional level in the SAR over the next century. There are likely to be considerable regional and local variations.

Globally averaged sea level is predicted by models to rise 0.09–0.88 m by 2100 (IPCC, 2001). Maximum elevation in Maldives is only 5 m, similarly in Lakshadweep and Chagos.

Effects of Climate Change on Coral Reefs and Mangroves

The effects of climate change on coral reefs have been reviewed by Wilkinson & Buddemeier (1994) and Brown *et al.* (1994). Due to the many uncertainties and lack of existing studies of climate projections for the SAR, assessment of possible effects of climate change over the coming century is generalized, qualitative, and contains some speculation. Some of the possible effects in South Asia follow, based on the climate scenarios discussed above.

Increases in mean SST's and extremes and duration of maximum SST's are highly likely to result in more frequent and more severe bleaching episodes. With increased frequency of such events, there will be less opportunity for recovery (typically, full recovery takes at least 15–20 years). The recent apparent increase in frequency and intensity of bleaching is quite possibly a sign of things to come. However, some coral reefs will be more resistant than others to increases in water temperature. In the Chagos, corals in the lagoons have survived better than those on outer reefs, possibly because they are subjected to higher temperatures and have built up a tolerance to increased SST's. It is unclear why corals in the Andaman and Nicobar Islands apparently suffered insignificant bleaching, despite observations of significant SST increases in the area. Perhaps their exposure to a wide seasonal range in SST due to the current systems associated with both summer and winter monsoons will make them more resistant to SST warming events in the future. The corals that have so far survived largely intact may prove to be tolerant to further temperature increases in the future. However, they may also be at the limit of their tolerance and would suffer with any greater SST increase in the future.

Increased rainfall, and in particular high-precipitation events, will result in increased runoff in some areas. Where the coastline is already degraded by deforestation, agricultural activity and human constructions, increased amounts of sediment will be transported into coastal waters, resulting in increased sedimentation of coral reefs and mangroves. Flow of industrial pollutants into the ocean will increase. The sediment loading of waters, particularly in areas that are already prone to influxes of sediment such as in the Gulfs of Kachchh and Mannar, and surrounding St. Martin's Island, and settled islands in the Andaman and Nicobar Islands, will be expected to increase. In reef areas close to pristine shorelines, there will be more frequent influxes of freshwater.

Increased storm severity and/or frequency will bring increased wave damage to reefs, which themselves act as wave barriers during these storms. Coral skeletons that have suffered a mass bleaching event will be exposed to the erosive forces of wave action and will be more susceptible to such damage. Bleached coral skeletons will also be exposed to bioeroders. With damaged reefs, the ability to protect the coastline from increased storm activity will be reduced. The protection of shorelines from rising sea level will also be affected. Healthy mangroves also play an important role in protecting the coastline from storm and flood events. Mangrove habitats are expected to be lost with the inundation of ocean water and erosion of the coastline. If an increase in cyclone activity occurs, it will affect the coral aggregations around St. Martin's Island, which are presently heavily influenced by monsoons and frequent cyclones.

Coral reefs and associated photosynthesizing communities will also be affected by the fluctuations in insolation, caused by changes in cloudiness associated with modulations of the monsoons and storm and cyclone activity and increases in atmospheric aerosol concentrations. The effects of insolation changes are currently not well understood and more research is required. A reduction in cloudiness will exacerbate damage to these communities from harmful UV radiation, which will increase with the projected advancing ozone depletion in the stratosphere as we progress further into the 21st Century, but presently more research is needed to be able to make any reliable predictions about the negative effects.

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Predictions for the rate of sea-level rise are well within reported rates of vertical reef growth for present reef communities, but locally the rates of sea-level rise could be higher. The most vulnerable areas are the atolls and low-lying coastal areas, especially deltas with mangroves. Likely to be a more important factor is a combination of sea-level rise and increased erosion, which would result from increased wave action. In Sri Lanka, large areas of the coast, some of them important mangrove habitats, are currently naturally eroding because of monsoon waves from the Indian Ocean (exacerbated in many places by destruction of coral reef and mangrove systems and sand harvesting from rivers). The effects of increased storm frequency and intensity will also be exacerbated by sea-level rise.

The temperature effect on growth and species diversity of mangroves is not known and research in this area is very much needed. Sea-level rise may pose a serious threat to mangrove ecosystems. In Bangladesh, for instance, sea-level rise, in combination with possible increased cyclone and storm activity, will threaten species in the three distinct ecological zones that make up the Sundarbans (including presently protected areas), the largest continuous mangrove area in the world. In populated areas, landward migration of mangroves and tidal wetlands in populated areas will be constrained by human infrastructure and activities.

Changes in ocean circulation brought about by global and regional climate change (scenarios are highly uncertain for the Indian Ocean), along with changes in wave conditions, will cause changes in the salinity and nutrient content of the water which will probably affect the rates of growth, calcification and reproductive potential of corals. Conversely, these changes may bring more favourable conditions to some areas. Changes in ocean circulation will affect recruitment, dispersal and recovery, but such effects are currently very difficult to predict and will require more study with more sophisticated models in the future.

Reduced calcification rates are expected at higher CO_2 levels.

The SST warming resulting from climate change will have beneficial effects in some areas. While coral species will die in waters that become too warm, new communities will also establish themselves in waters that were previously too cool (provided that shorter wintertime day length at higher latitudes does not restrict coral growth), although this latter scenario is not likely to be relevant to the SAR. Another possible beneficial effect is the expansion of coral communities to colonise areas that have been inundated by sea level rise. Shifts in community composition after the 1998 bleaching event, such as those that have been documented in Maldives (Clark, 2000; Zahir & Clark, CORDIO Newsletter, Oct 2001), are likely to result in massive corals replacing branching corals. Corals growing at greater depth are likely to be less severely affected by climate change and will be healthier than shallower ones.

It is not clear how the incidence of starfish predation will change on reefs that have been degraded by climate change. In recent years in Sri Lanka, northwest and east coasts have been under persistent attack from the crown-of-thorns starfish *Acanthaster planci* (Rajasuriya & Karunarathna, 2000). Already-damaged reefs will be less likely to recover with such predation episodes. Warmer temperatures may bring an increase in frequency and severity of disease, as well as susceptibility.

Some coral and mangrove communities are already surviving in conditions which are thought to be at their tolerance limits, e.g. the fringing reefs in the Gulf of Kachchh and the extensive mangroves in the Indus River Delta (these reefs are subjected to high temperatures, high salinity, high sediment discharge and tidal ranges which can be as great as 12 m). The extremes of these conditions would be expected to increase with regional warming and only small increases will result in widespread mortality.

The predicted warming will produce net economic loss in many parts of the South Asia Region, which will result in increased poverty and, in turn, increased pressure on coral reef and mangrove resources. This would particularly affect India, Bangladesh and Sri Lanka. More damaging impacts are predicted for these nations because of their lesser adaptive capacities relative to wealthier countries. The effects will be felt most by the poor coastal communities living at or just above subsistence level.

SUMMARY AND RECOMMENDATIONS

At present, the relationship between climate change and coastal and marine ecosystems, specifically coral reefs and mangroves, in the South Asia Region is still poorly understood. Surface temperatures have shown a clear increase over the last century and will continue to increase, with possible greater increases on the regional and local scale. Changes in the monsoons over this period have also been recorded, but it is unclear how monsoon or cyclone activity will change in the future. The behaviour of El Niño appears to have shifted since the mid-1970's, and the very large events of 1982/3 and 1997/8 may be a sign of things to come. Sea level is predicted to rise by 10–90 cm over the next century. While the most pressing immediate danger to these ecosystems comes from direct human impacts, for those areas that remain isolated from the effects of population pressure, climate change is the only significant threat. Furthermore, the combination of the effects of human impacts and climate change poses significant danger for many of the reef and mangrove habitats in heavily-populated South Asia. Climate change is likely to exacerbate the impacts of human activities in and around these habitats. The importance of the effects of climate change will increase as we progress through the 21st Century.

Climate change represents an important additional external stress on ecological and socio-economic systems in South Asia which are already adversely affected by air, water, or land pollution, as well as increasing resource demands, environmental degradation, and unsustainable management practices. Awareness of the effects of climate change needs to be improved at all levels of society, from government to local and community levels, and alternative sources of income need to be found for those communities reliant on resources from coral reefs and mangroves. Studies have also shown that awareness among tourists is low (Westmacott *et al.*, 2000) and education of tourists visiting coastal areas needs to be improved. With reefs and mangroves along both densely populated and very remote coastlines, the SAR is an ideal place for a comparative study of the impacts of climate change alone, and the combined effects of climate change and local and regional human activities.

The 1998 bleaching event across the Region was dramatic but, more than three years on, the need still exists for a pooling of these individual studies and a thorough summary of their findings. There is a need for continued observations to determine whether this is a rare, severe event, or part of a pattern of increasing disturbance associated with global climate change. In addition, more studies are required to investigate if and why some areas are likely to be less affected than others: e.g. a study into why there was no bleaching in some areas, e.g. around Trincomalee (Pigeon Island) on the northeast coast of Sri Lanka, while other areas of the Sri Lanka coastline were badly affected (this may be due to localized upwelling of cooler waters in the northeast). In Maldives, some reefs and portions of reefs escaped significant damage, as did the reefs in the Andaman and Nicobar Islands. In some corals, differences in zooxanthellae composition may be an important factor in affecting the tolerance level to elevated SST's. These unaffected sites should be studied and their usefulness as larval sources investigated (W. Allison, pers. comm.). It is also unclear why some species are more susceptible than others: e.g. an investigation of the corals in the lagoons and those on the outside reefs in the Chagos to understand why the lagoon corals have shown more resistance to bleaching would provide valuable information. In some places, the fact that there have been no reports of bleaching may be due to insufficient observations, e.g. in remote regions of the Andaman and Nicobar Islands, the Gulf of Kachchh, and the coral aggregations around St. Martin's Island. More detailed studies of the variations in the tolerance limits of corals in different locations are needed to address this very important aspect of their response to the effects of climate change. Such studies will also aid understanding of how the habitat zones of coral reefs might shift in future climate scenarios. These studies can only take place if there is a thorough inventory of sites. This is still lacking for the SAR.[2] For those sites where studies have taken place or are ongoing and for which data already exist, these data are not being fully utilized or consistently recorded. A review of management strategies with a focus on identifying and managing the reef and mangrove sites with a regional approach is necessary.

In past decades there have been cycles of changes in coral cover and species diversity, which probably correlate with variations in SST. Historical trends need to be studied in order to put recent trends into an historical context and aid predictions about the future, when such SST fluctuation episodes may likely become more extreme and more prolonged. Palaeoclimate studies in coral cores, utilizing stable isotope thermometry, will also provide a more detailed picture of past temperature fluctuations, but so far there have been very few documented studies in South Asia (whereas other regions have received considerable attention), and this Region needs to be targeted. Currently, the scientific expertise in the Region for such research is limited and could be improved with increased training and collaboration with scientists in other countries.

There is a need for closer collaboration between climate scientists and coral reef specialists. It is vital that there is the implementation of a systematic, regionally cohesive programme of long-term climate and sea-level data monitoring, particularly the measurement of SST's. The establishment of permanent monitoring sites is required. The author is unaware of the existence of a database of instrumental data which would allow, for example, the 1998 SST event to be placed into a regional historical context for South Asia . There are only a few programmes that are currently measuring SST's, but these are not widely known about in the coral research community and are isolated data sets. These data need to be consolidated. There should be deployment of temperature loggers in coral reef areas for long-term monitoring of water temperature. These time series would also be extremely useful for calibration purposes in future studies that use coral stable isotope thermometry to study historical trends in sea water temperature. The ability of climate models to reliably predict future climate is increasing, and advances in their performance demand the collection and consolidation of more climate data time series. Remotely-sensed data should also be pooled into a South Asia database that is made readily available to both climate and coral reef specialists. These studies and monitoring efforts are urgently needed if we are to gain an understanding of the role of climate change in the degradation and adaptation of coral reefs and mangroves.

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[2] The Zoological Survey of India has proposed a 10-year inventory and monitoring programme in the four main coral reef areas of India. The Gulf of Mannar study is already underway(2002-2012) and will be followed by Lakshadweep (2005-2015).

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Part III

Indian Ocean Islands

2001 Bleaching on *Acropora* Dominated Reefs in Réunion

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Key words: Acropora muricata, bleaching, coral reef, LIT, Réunion Island.

ABSTRACT

As coral cover and diversity are affected on many reefs of the Western Indian Ocean by natural and human threats, especially coral bleaching, a clearer understanding of reef dynamics after bleaching is necessary. This paper presents a study designed to quantify the impact of a localized bleaching event which occurred in the lagoon of La Saline in February–March 2001. Phase 1 focused on assessing the spread of the phenomenon to all coral reefs of Réunion. Phase 2 was a rapid response to provide the initial temporal baseline for long term monitoring. Data were collected along 20-m transects located in the reef back boat channel, where *A. muricata* was the dominant coral and was suffering from bleaching. From March to July, LIT were used to estimate the percentage cover of bleached coral and algae, according to five previously-defined scales. Algal turfs covering dead corals and associated microalgae were collected at different steps of the process in order to estimate the colonisation patterns of dead corals. The results described in this paper confirm that *A. muricata* may remain white for more than one month, that 5 stages can be identified for algae cover and that assemblages of potentially toxic dinoflagellates were immediately present. Recovery processes will be studied during phase 2 to assess the topographic and community structures, including recruitment and growth from remaining live tips of *A. muricata.*

INTRODUCTION

As coral cover and diversity are affected on many reefs of Western Indian Ocean by natural and human threats, especially coral bleaching, a clearer understanding of reef dynamics after bleaching is necessary. Réunion island was not severely affected by the 1998 event, compared to Mayotte-Comoros and Seychelles. As part of the GCRMN-CORDIO regional components and IFRECOR Action Plan, Réunion is conducting activities that are relevant to monitoring and specific research on coral reefs.

This paper presents the results of a study designed to quantify the impact of a localized bleaching event which occured in the lagoon of La Saline in February–March 2001.

MATERIALS AND METHODS

Phase 1

Phase 1 (April–May) was focused on assessing the spread of the phenomenon to all coral reefs of Reunion. Rapid surveys of lagoons were conducted by scientists and Marine Park eco-guards. A total of 80 sites were surveyed (coral cover, species, coral bleached). Temperature records were obtained using *in situ* data loggers from La Saline lagoon (–1 m) and a coastal reference site, Le Port (–6 m).

Table1. Successive steps in the algal colonisation of post-bleached corals

Steps	Description
0	Corals lose their colours, become white. In this step, corals survive or die.
1	First step of colonisation. A light coloration appears at the surface of corals.
2	A short algal turf partially cover the coral surface. Skeleton is still visible.
3	The short algal turf covers the entirety of the coral surface. Corallites are visible.
4	The algal turf beco,es dense and some corallites are not visible.
5	Final step of the colonisation. All corallites are invisible. Algal turfs are similar to those habitually observed on the site.

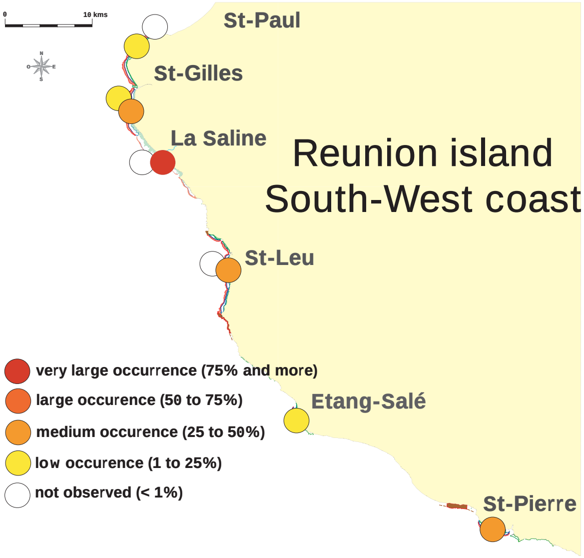

Figure 1. Coral bleaching in April 2001.

Phase 2

Phase 2 (June–October), was a rapid response to provide the initial temporal baseline for long term (phase 3) monitoring.

- Ground truthing the biophysical status of reef communities, performed according to the GCRMN-adapted methodology (Conand *et al.*, 1998), including 5 scales of algal cover (Table 1). Data was collected along 20 m LIT transects.
- Temperature recording using *in situ* data loggers from both La Saline lagoon (–1m) and a coastal reference site, Le Port (–6 m).
- Obtaining photographs for an image database.
- Collection of dead *A. muricata* tips for algal turfs and epiphytic microalgae analysis (regional methodology in Turquet *et al.*, 2000), in order to estimate the colonisation patterns of dead corals.

RESULTS & DISCUSSION

Eighty sites were surveyed by stakeholders involved in the GCRMN local network and showed that all reefs had been affected by a local SST anomaly (small scale). Variable intensities of bleaching were recorded. The most affected places were those from La Saline lagoon where *A. muricata* dominates the benthic community (Fig. 1), with levels of 50% to 90% recorded from the Livingstone site.

Fig. 2 shows that, from the end of February to mid-May, coastal waters were 0.5 to 1°C warmer than average (1993–2001 reference period). Lagoonal waters are subjected to large tidal and diurnal variations (6°C difference) and 32°C was reached for a whole week in February (Fig. 3).

Benthic LIT monthly monitoring of the 2 stations (South-Livingstone and Livingstone) are presented in Fig. 5 and 6 (page 182), respectively. The onset of the bleaching event was estimated as the end of February. Observations made in June indicated that the same colonies may stay white for 3 months.

At the medium scale of La Saline, 80% of the bleached corals recovered. The same pattern was reported

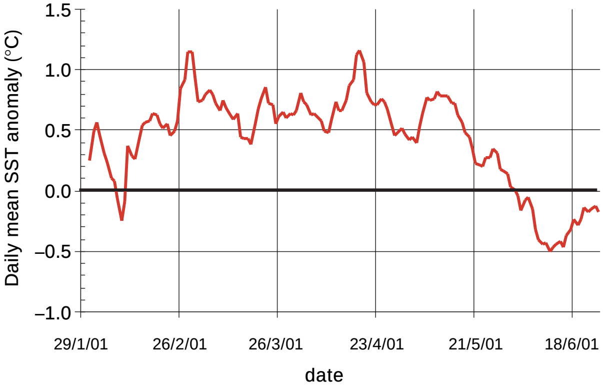

Figure 2. SST in Réunion: daily mean anomaly of coastal water.

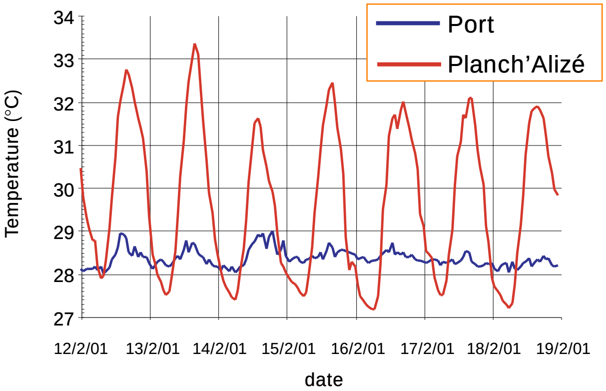

Figure 3. SST in Réunion in coastal water (black line) and in lagoonal water (dotted line).

Figure 4. Coral bleaching in LivingStone (April 2001). *Photo:* J. P. QUOD, ARVAM

for South-Livingstone (Fig. 5), while Livingstone, located 200m north, suffered higher mortality of 90% (Fig. 6). Pockets of resistant colonies were rare (5% to 10%) and patchy. Mortality was followed by a succession of algal-turf assemblages on the dead corals. Five steps were characterised, including the classic 'algae-step 5' (dense algal turfs with *Stegastes* damselfish) and 'eroded-step'. The latter is supposed to be a key factor for recruitment, settlement and subsequent recovery of the coral community.

This event was an opportunity to understand better (Quod *et al.*, 2000) the colonisation patterns of dead corals by potentially toxic microalgae. A classic assemblage of dinoflagellates was observed very early (white colonies, step 0) and increased in abundance with time (Fig. 7, page 183).

Spatial and temporal analysis of collected data (field observations, LIT, temperature variations, algal cover) are synthesised in Fig. 8 (page 183). As resistance to bleaching of *A. muricata* colonies varied from place to place, we supposed that environmental factors (topography, hydrology and currents, nutrient load) may be important explanatory factors for bleaching and subsequent consequences: mortality, recovery, recruitment.

CONCLUSION

A localized bleaching event occured in the lagoon of La Saline in February–March 2001. Variable bleaching intensities were recorded and the most affected places were those from La Saline lagoon where *A. muricata* dominates the benthic community. Bleaching affected 50 to 90% of corals from Livingstone site, in La Saline. During this period, coastal waters were 0.5 to 1°C warmer than average (1993–2001 reference period). Lagoonal waters were subjected to large tidal and diurnal variations (6°C difference) and 32°C was reached for a whole week in February. Monitoring showed that same colonies may remain white for 3 months.

Spatial and temporal analyses of collected data (field observations, LIT, temperature variations, algal cover) are synthesised in Fig. 8. As resistance of *A. muricata* colonies vary from place to place, we supposed that

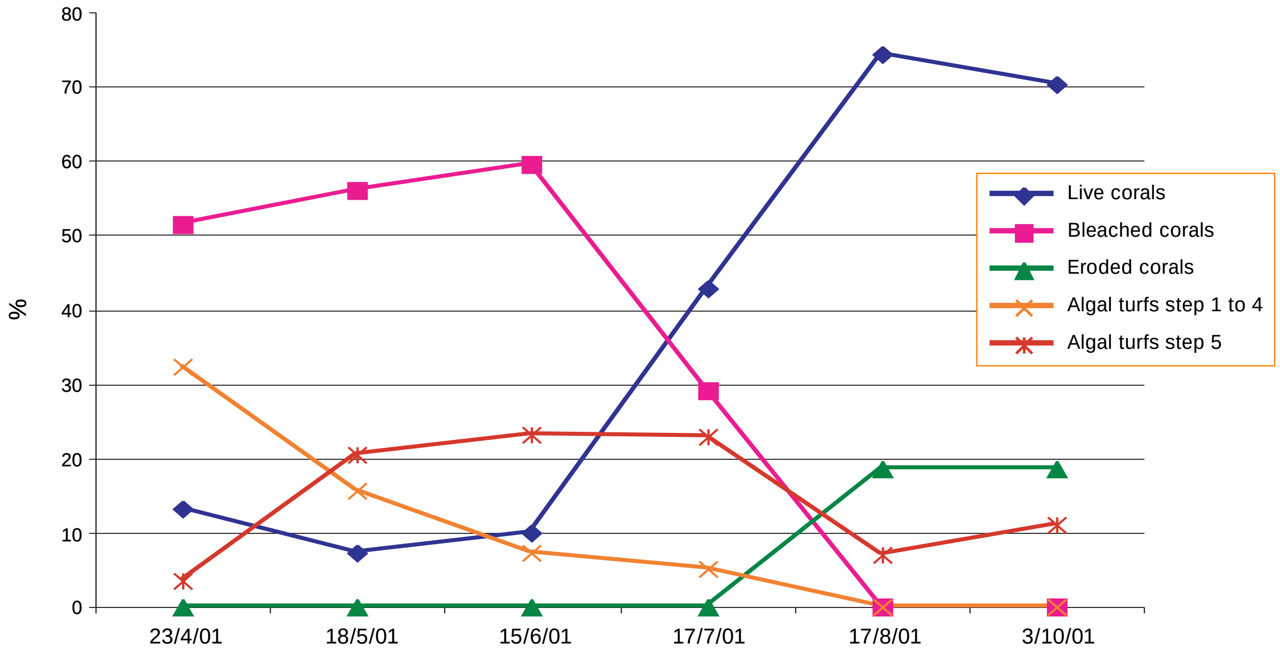

Figure 5. LIT monitoring at South-Livingstone.

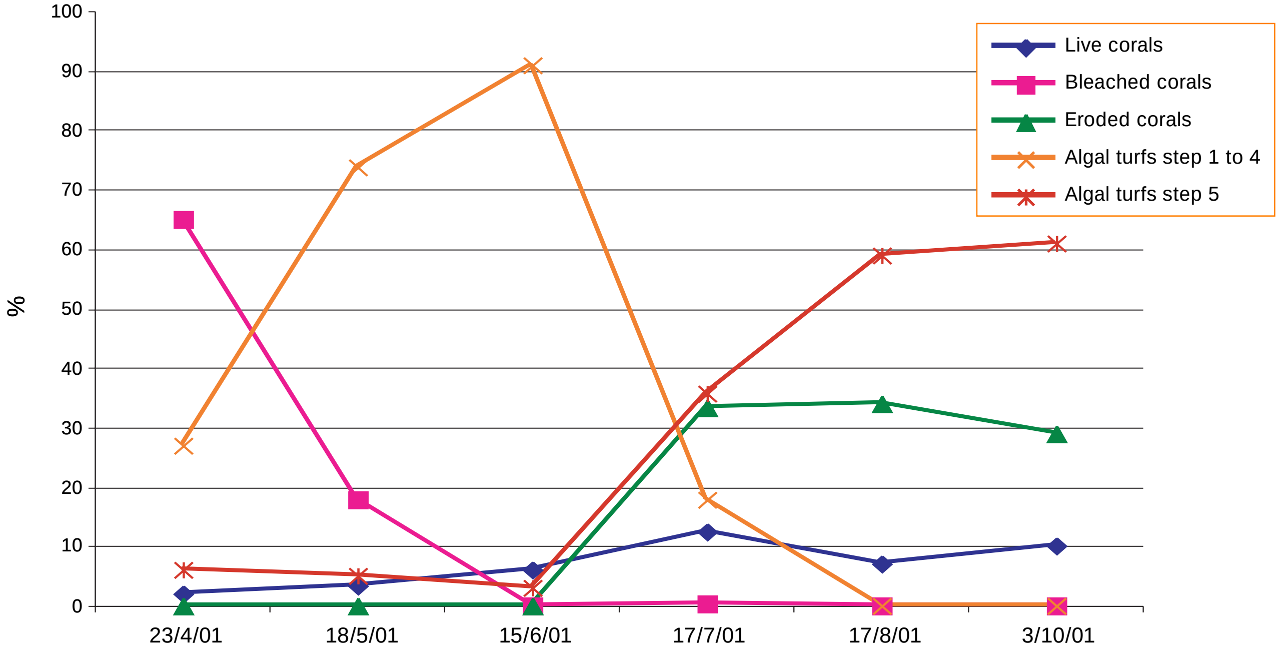

Figure 6. LIT monitoring at Livingstone.

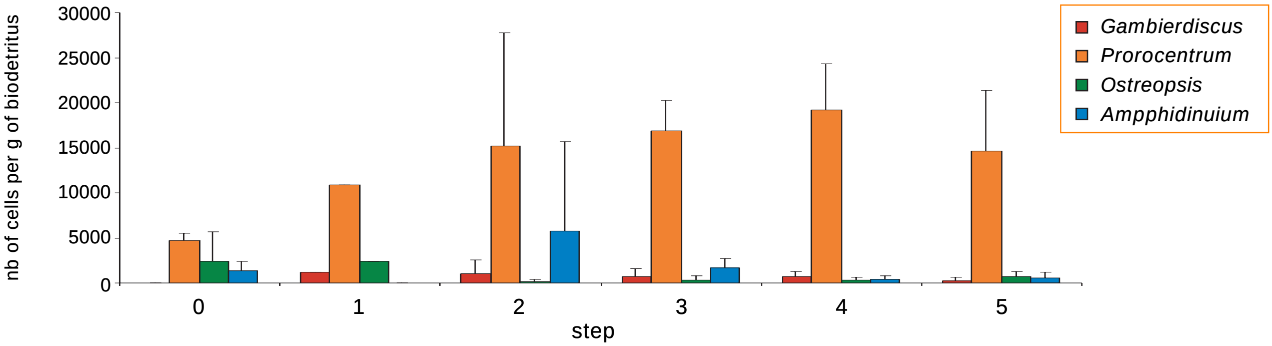

Figure 7. Dinoflagellates colonisation on the 5 step algal turfs.

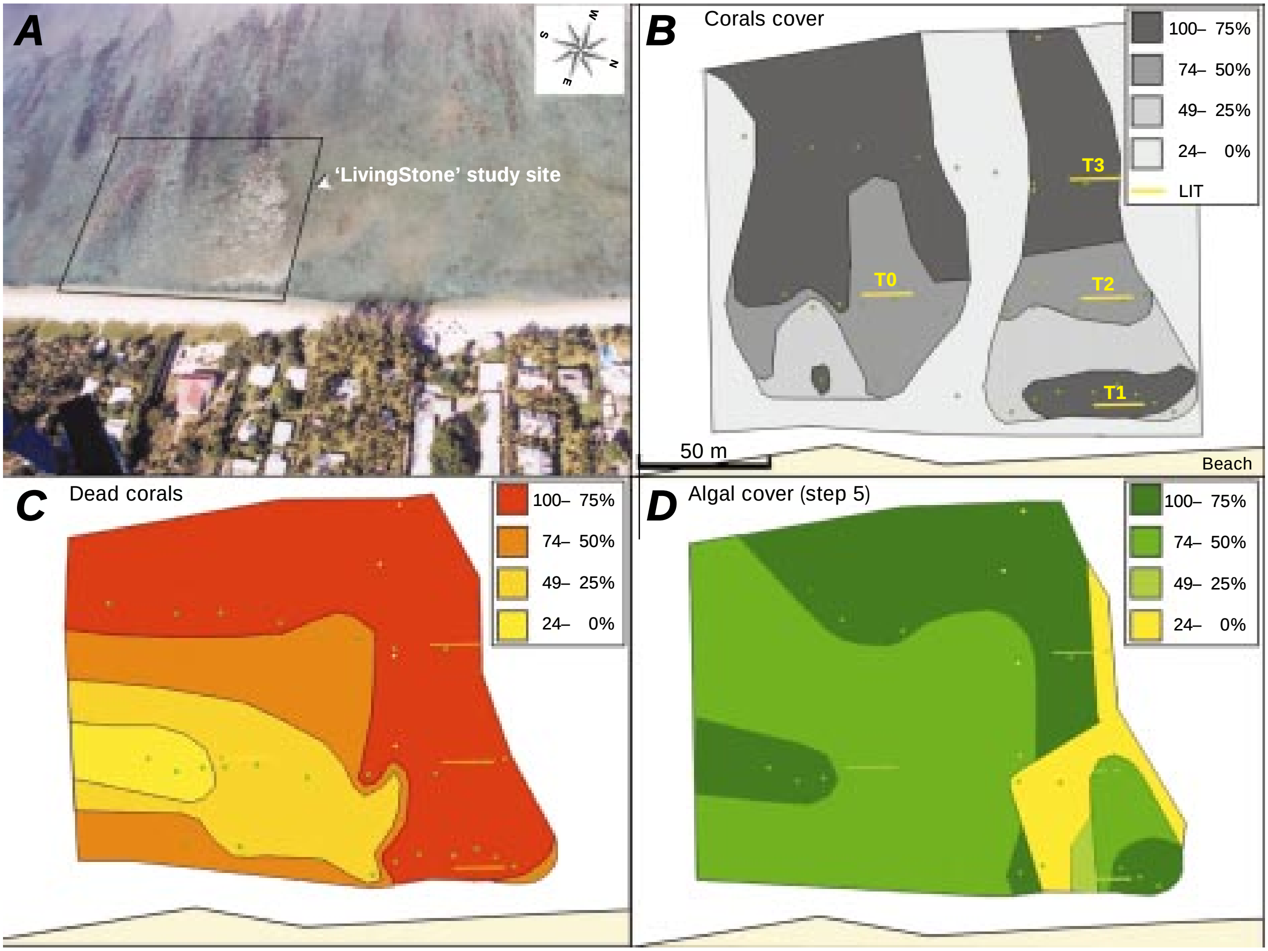

Figure 8. Spatial analysis (October 2001).

environmental factors (topography, hydrology and currents, nutrient load) may be important explanatory factors for bleaching and subsequent consequences:

Following phase 2, decision-makers will propose a 3–5-year monitoring programme, focused on 'Livingstone' site, which will include bi-annual monitoring of benthic and fish communities, recruitment patterns and recovery from remaining live colonies.

This programme will contribute to determine resilience and resistance factors for lagoonal communities of Réunion reefs. These will be important for effective local issues regarding both Integrated Coastal Zone Management and natural threats such as Cyclones and/or Climate Change.

ACKNOWLEDGMENTS

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Status of Coral Reefs at Réunion, Mayotte and Madagascar

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Key words: coral bleaching, Réunion, Mayotte, western Indian Ocean island

INTRODUCTION

The massive coral bleaching that affected the Indian Ocean in 1998 caused a drastic decline in the coral cover in large parts of the central and western Indian Ocean. However, certain areas, particularly in the south of the western Indian Ocean region, were not affected to the same extent. The present study report, the results from three such areas: Réunion, Mayotte-Comoros and Tuléar-Madagascar. Réunion was not seriously affected, while Mayotte-Comoros and Tuléar-Madagascar showed intermediate impacts.

The paper presents the status of the sites 4 years after the mortality caused by coral bleaching due to the El Niño of 1997/8. with special emphasis to Mayotte and the recovery process.

MATERIALS AND METHODS

Three islands were considered due to (i) their representative position regarding coral bleaching and (ii) the occurrence of existing reef monitoring programs:

Mayotte Island in Comoros, Tuléar in southern Madagascar, and La Saline in Réunion. Investigations were conducted using the LIT techniques adapted from GCRMN (see Conand *et al.*, 1998) and data was stored in the COREMO database developed for COI (Indian Ocean Commission). Major aspects of the monitoring were benthic fauna and flora including Live Coral Cover – LCC, the occurrence of bleached coral and coral recruits (less than 2 years old) and the diversity and abundance of fish. Sampling was performed on permanent 100-metre transects located either on the outer slopes or reef flats of fringing, inner, barrier reefs.

Dataloggers at Mayotteare have recorded sea temperature every 10 minutes since 1998 and downloaded every 2 months.

REGIONAL STATUS OF CORAL REEFS

Coral reefs are presently monitored at 73 stations in the COI countries (Bigot *et al.*, 2000; Ahamada *et al.*, 2002), 22 in Mayotte-Geyser (Dahalani *et al.*, 2002) and 1 in Europa. Both coral fauna, certain invertebrates and fish are assessed. Table 1 (next page) considers only the CORDIO stations, as defined previously (Quod & Bigot, 2000). Europa was surveyed in 2002 and added to the network as it will be regularly monitored in future.

The maximum temperatures of the surface waters in the area occur normaly during the period February–March. However, since 1999, coral reefs from the western Indian ocean islands have not suffered significantly from increased sea surface temperatures (SSTs).

At Réunion, reefs where affected to a limited extent by the 1998 bleaching (less than 5 percent of the reef flat

Table 1. Live Coral Cover from the 17 GCRMN-CORDIO LIT stations. Data obtained either from the COI monitoring network,or from the Coral Reef Observatory of Mayotte. (rf: reef flat; os : outer slope; D1: difference between 2001 and 1999; D2: difference between 2002 and 1999).

country	sector	sites	stations	depth	Live Coral Cover						
					1998	1999	2000	2001	2002	D1	D2
COMOROS	Gde Comore	Mitsamiouli	rf	1	46.0	44.2	nd	nd	36.5		–7.70
	Mohéli	Itsamia	rf	1	35.0	30.0	nd	nd	5.0		–25.00
MADAGASCAR	Tuléar	Ifaty	os	8	30.8	40.7	41.9	nd	42.0		1.30
		Nosy Ve	os	10	nd	42.3	nd	nd	nd		
MAYOTTE	S Nord Est	Surprise	rf	–0.5	6.3	4.6	16.1	15.0		10.40	
			rf	–6	15.0	28.0	44.1	53.7		25.70	
	S Ajangoua	Longogori	os	–0.5	11.9	6.2	15.3	15.9		9.70	
			os	–6		21.6	16.2	22.3		0.70	
	Geyser/Zélée	Geyser	rf	–6		34.3	46.7		nd		
			os	–6		63.1	65.3		nd		
RÉUNION		La Saline/PlanchAlizés	rf	–0.5	18.0	40.0	39.2	39.1	37.1	–0.90	–2.92
			os	–12	38.0	40.0	48.6	45.2	37.0	5.20	–3.00
		La Saline/3Chameaux	rf	–0.5	31.0	30.0	23.4	36.60	35.7	6.60	5.70
			os	–12	57.0	50.0	39.4	42.9	38.7	–7.10	–11.30
		Saint Leu/Corne Nord	rf	–0.5	–	50.0	53.4	62.6	44.2	12.60	–5.80
			os	–8	–	60.0	73.5	77.1	74.4	17.09	14.40
FIOT		Europa	os	–8	–	–	–	–	36.0		

communities were affected). Here a slow increase in live coral cover (LCC) have been observed between 1999 and 2001. The cyclone Dina, which hit the west coasts of Réunion in February 2002, decreased the live coral cover especially on the reef flats (18.7 percent at Saint Leu Corne Nord station).

In February–March 2001, a localised bleaching event was observed close to the 'lagoon' station of Trois Chameaux. Bleaching affected 50 to 90% of corals at the Livingstone site, in La Saline. (Turquet *et al.*, this issue). This event was associated with a warm water 'hot spot' observed only at Réunion and not at Mauritius (A. Strong, pers. comm.).

In 2002, the cyclone Dina led to high level outputs of freshwater and siltation in the lagoons with subsequent coral bleaching and mortality varying from place to place.

In Mayotte, coral cover increased significantly since 1998, especially at Surprise inner reef (LCC at 6 m increased from 15 percent in 1998 to 54 percent in 2001).

The impact of the 1998 bleaching and coral mortality on fish communities was studied by Chabanet (2002). Results of studies of the trophic struture and bioindicators show that coral bleaching has influenced particularly the Chaetodontidae and Scaridae. The reason for the observed effect are habitat alteration and massive algal cover of dead substrates.

In Madagascar, no significant change was observed in the Ifaty outer slope. Here, the cover of hard coral has been approximately 40 percent since 1999.

Cover of live corals in Comoros showed a decrease over the 4 year period. The decrease was particularly pronounced at Itsamia where an extensive mud run-off caused a 25 percent breakdown in reef flat live coral cover in year 2001.

Geyser bank, located between Mayotte and Madagascar was surveyed only in 1999 and 2000. No change was observed at stations on the slope. However, there was a significant incerase in live coral cover at the inner station (12% per year).

The French 'Ile Eparse' of Europa, located in the south of the Mozambique channel was added in 2002 as a CORDIO site as it is located at a strategic place in terms of coral distribution in the region. Cover of hard corals was approximately 36 percent at stations on the outer slopes. It has been confirmed that both the 'lagoon' and the outer slopes were affected by the 1998 coral bleaching (Quod and Garnier, 2002). Large strands of dead Acroporidae were observed and recruitment was poor and limited to a few locations.

RECOVERY PATTERNS IN MAYOTTE

As show in Fig. 1, after the widespread coral bleaching, which occured in the summer of 1998, the process of recovery has been assessed and differs from Mayotte to Réunion.

In Mayotte Surprise inner reef (Mayotte), live coral cover at the sites on the slope (6 m depth) showed a signifi-cant recovery compared to stations located on the reef flat.

Depending on the sites, increased live coral cover may result either from growth of remaining live colonies or recruitment or both. The majority of the hard coral recruits that settled and survived on the sites in Mayotte consist of species from the families Acroporidae and Pocilloporidae. By family, the recruitment is Pocilloporidae: 28%; Acroporidae: 60%; Faviidae: 6%; Poritidae: 1%; and other groups: 1%. These data are in agreement with those from the other stations in Mayotte (Dahalani *et al.*, 2002). Poritidae is the group showing the greatest success in recruitment at the fringing reefs of the main land (Grande Terre).

TEMPERATURE RECORDING

Both Réunion and Mayotte now have sets of temperature recorders located in the lagoon and on the outer slopes. An additional temperature recorder was deployed at Europa in 2002.

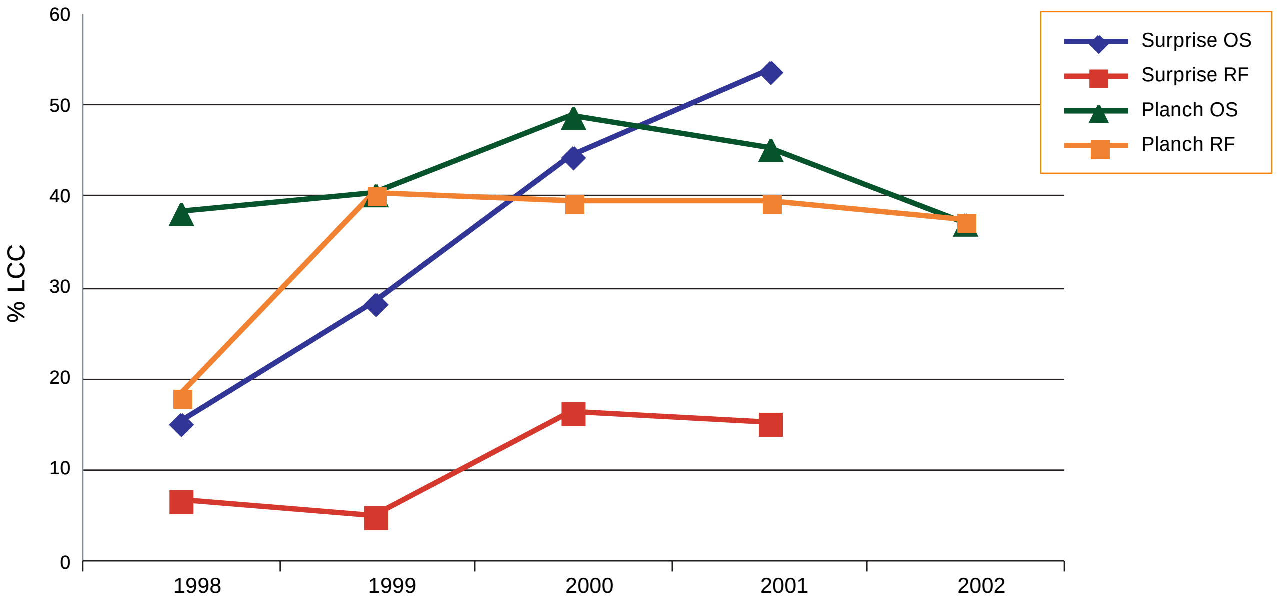

Figure 1. Live Coral Cover (LCC) trends in Mayotte and Réunion CORDIO sites from 1998 to 2002.

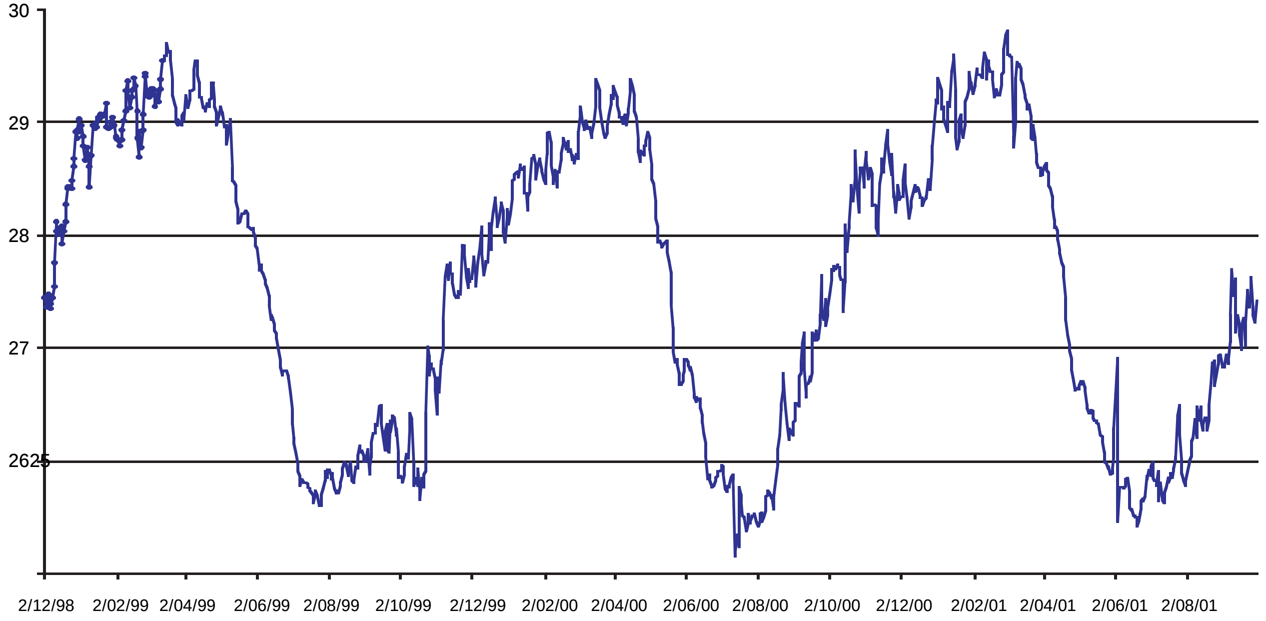

Figure 2. SST in Mayotte, from december 1998 to august 2001. Average of 4 stations, including Surprise and Longogori.

In Réunion, data were collected during the 2001 bleaching event when the coral reefs were affected by a increased sea surface temperatures (SST) (Turquet *et al.*, this issue).

In Mayotte, seawater temperatures have been recorded from 4 sites since December 1998 (Fig. 2). Average temperatures of the lagoon for the period 1999–2001 were very similar between the years: 27.6°C ± 1.4 in 1999, 27.3°C ± 1.3 in 2000 and 27.6°C ± 1.3 in 2001. Mean annual variation was 3.5°C and maxima were

^{2}=0,81).

These results may be useful for an alert system to be able to predict future events of coral bleaching, and also for other management issues of coral reefs in relation to human threats.

In Réunion, the occurrence of coral bleaching in relation to sea surface temperatures have been described by Conand *et al.* (2002). The authors demonstrated that in March 1998, sea surface temperature was 1.5°C higher than during the 1994–2000 reference period and that the ‘hot period’ exceeded 15 days.

DISCUSSION AND TRENDS

For 2001 and 2002, collection of data at the CORDIO sites was carried out in collaboration with two other regional initiatives: (i) the GEF-COI coral reef network and (ii) the IFRECOR action plans at Mayotte and Réunion. Results from the field differs whether the location was badly affected by the 1998 event or not.

In Mayotte, it has been demonstrated that live coral cover has decreased significantly and that benthic communities are now dominated by algal communities. The observed shift in the fish community two years after the severe mortality (80 to 90 percent of the outer slopes) could be related to the replacement of hard live corals by algal substrates. A positive shift to increasing coral cover was now observed (25%) for stations at the outer slope of Longogori). Thus, it appears that recovery is proceeding both from growth of remaining colonies and recruitment. Future studies should define whether the positive trend will continue and if a recovery will take place also in the fish populations.

The results from various places in the region are highly variable and no general trend is obvious. The highest live coral cover was found in Saint Leu (Réunion) outer slope (77% in 2001). Average figures from other stations in outer slopes of Réunion show about 30–40 percent of hard coral cover. The variations are related to the exposed nature of several stations that make them vulnerable to cyclones.

The reefs in the shallow lagoons were affected to some extent by bleaching induced mortality. However, land-based sources of pollution and pollution via the groundwater is more important to these areas. Furthermore, in 2002 freshwater and siltation from land caused significant reductions in living coral cover from which only some of the reefs have recovered. Studies will be performed at Livingstone to monitor the condition of this site. The total area of coral reef in Réunion is only 12 km^2. However, the reefs may have a greater importance as source reefs for coral and coral fish larval recruitment in a much larger area of the western Indian Ocean. Therefore, it is of great importance to follow the status of the corals and fish communities in Réunion.

Temperature recorders should in the future be deployed in each of the major reef areas of the western Indian Ocean. It has been confirmed both in Mayotte and Réunion that good correlation may exist with air temperature recording and/or satellite information. More research has to be conducted to identify the temperature thresholds at which corals bleach and the duration and extent of temperature stress that causes mortality. Preliminary data confirm that the upper temperature limit that causes bleaching and mortality in corals depends on where they are found (lagoon, outer slopes), depth and geographic location. As both Mayotte and Réunion are now recording the sea surface temperatures using data loggers, increase in temperature may, in the future, be studied to define critical levels for coral bleaching.

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The authors acknowledge the national teams involved in coral reef monitoring for assistance in field data collection.

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An Assessment of Coral Reefs in Northwest Madagascar

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Key words: coral bleaching, coral recovery, Madagascar

ABSTRACT

Madagascar has an extensive coastline and a large area of coral reef. These reefs are under threat from sedimentation, pollution, global warming, over fishing and mining for building materials. Limited information has been published on the status of coral reefs in Madagascar. This study focused on six fringing reefs in the Nosy Be region, north west Madagascar. Nosy Be is an principle tourist destination and healthy reefs are an important part of the local economy.

Six reefs were examined between July and August 1999. The reefs examined appeared to be in relatively good condition compared with other areas in Madagascar and in the western Indian Ocean region. The percentage of live coral cover was high and dead coral cover low. Coral bleaching was minor, and it appeared the area was little affected by the 1998 bleaching event. The reefs were diverse with a high number of coral genera recorded along each transect. Species such as the crown of thorns starfish *Acanthaster planci*, and the spiny sea urchin *Diadema setosum*, were absent or at low densities. Soft coral cover was less than 10% and the presence and abundance of certain indicator fish species also suggests that the reefs were in good condition. Immediate threats to the coral reefs at each site were identified and management recommendations were made.

INTRODUCTION

Madagascar is the world's fourth largest island, approximately 1,600 km long and 580 km wide situated in the Western Indian Ocean spanning tropical and subtropical latitudes. The extensive coastline (4,500 km) including approximately 270 small continental islets and islands, supports about 3,450 km of coral reefs. Emergent fringing reefs, barrier reefs, patch reefs and submerged coral banks and shoals are all present (Cooke *et al.*, 2000). Considering the extent of coral reefs little work has been done in the region.

Coral diversity in Madagascar is high with 304 scleractinian coral species present (Conservation International, 2002). Earlier reviews report 62 genera in Toliara and 63 in Nosy Be (IUCN, 1988). At the Grand Reef of Toliara 130 species (Pichon, 1978) and in the Masoala Peninsular 164 species of scleractinian corals have been identified (McClanahan & Obura, 1998).

There are a number of threats to coral reefs in Madagascar. Extensive clearing of the land for timber and agriculture has resulted in massive soil erosion and subsequent sedimentation in coastal areas, especially western Madagascar (Cockroft & Young, 1997). Pesticide runoff from intensive agricultural areas is also a potential threat (Cooke *et al.*, 2000). With the majority of Madagascar without sewage treatment networks, eutrophication is often significant (Hinrichsen, 1998). Bleaching has impacted large areas of coral reef in Madagascar, especially in 1998 when coastal water temperatures were 2°C above the seasonal average. Mortality was 80 to 90% in some areas (McClanahan & Obura, 1998).

Apart from direct effects on coral, there are secondary effects to the health of coral reef ecosystem from fishing. Fishing is an important source of food and income to many Madagascans living on the coast. Commercial fishing in the northwest of Madagascar mainly for shrimp, *Panaeus monodon* and *P. indicus*, has a significant by-catch (Jain, 1995). Disruptions to reef ecosystems from activities such as fishing, can lead outbreaks of certain species such as the spiny sea urchin, *Diadema setosum* (Andrianarivo *et al.*, 1999).

Healthy coral reefs are critical to the livelihoods and cultures of millions of people in tropical coastal environments like Madagascar and also form part of the crucial life support system of the biosphere (Wilkinson, 2000). With the paucity of information on corals in Madagascar (IUCN, 1998) and the potential threats to coral reef health, the aim of this study was to assess the health of six coral reef systems in the northwest of Madagascar, around Nosy Be Island. Potential threats to the coral reefs at each site were identified. Coral reef health was inferred from; live and dead coral cover, coral diversity, the density of outbreak species and the abundance of indicator fish species.

METHODS

Study Region

Nosy Be is an island of 290km^2 located in the northwest of Madagascar. Coastal marine ecosystems of this region are vulnerable to impacts from sedimentation and the primary land uses on the island and nearby mainland are coffee and sugar plantations. A refinery is located in Djanrndjara, Nosy Be for processing sugar cane where, up until 1999 the effluent was pumped directly into a creek running into the sea.

Commercial shrimp trawling centered on Nosy Be and processing of the fish also occurs on the island. Other commercially fished species industries include pelagics, sea cucumbers, octopus, lobster, crabs and reef fish. Artisinal fishing is important and has increased in intensity in the Nosy Be region in the last decade (Andrews, pers comm., 1999). Artisinal fishing practices include the use of fish traps, line and net fishing and collecting invertebrates on reefs at low tide. Trampling of coral is a threat in these areas.

Nosy Be is Madagascar's principle tourist center with coral reefs a significant attraction for diving and snorkeling. Threats to the coral from the tourist industry include direct damage to coral reef systems from anchoring of boats and collection of live coral and shells for sale. Associated with this is increased fishing pressure for restaurants, increased rubbish and effluent.

In the Nosy Be region, there is only one marine protected area with some form of conservation protection, Nosy Tanikely. This island is a conservation site declared by ministerial order in 1968 that forbids fishing within 300 m of the shore. However, this Preserve is not actively policed and some fishing is known to occur within the boundary.

Madagascar has two principle wind systems, the south east trade winds and the monsoon. The south east trade winds blow strongly for eight to nine months of the year with a median windspeed of 6 ms^{-1}, however the north western areas of Madagascar including Nosy Be are much less influenced, being protected by the Tsaratanana masif. These areas are affected by the monsoon season of which there are two, the northeast (November to March) and south east (April to October) winds average 3.5 ms^{-1}. The associated wave and swell action is usually low. Madagascar has significant daily onshore and offshore breezes. The country is strongly affected by cyclones in the warm season (December to March) with an average of 1.3 per year. The tidal range in Madagascar is greatest on the west coast with a spring range of 3.8 m recorded approximately 300km south of Nosy Be, but up to 5 m in rare instances (Cooke *et al.*, 2000).

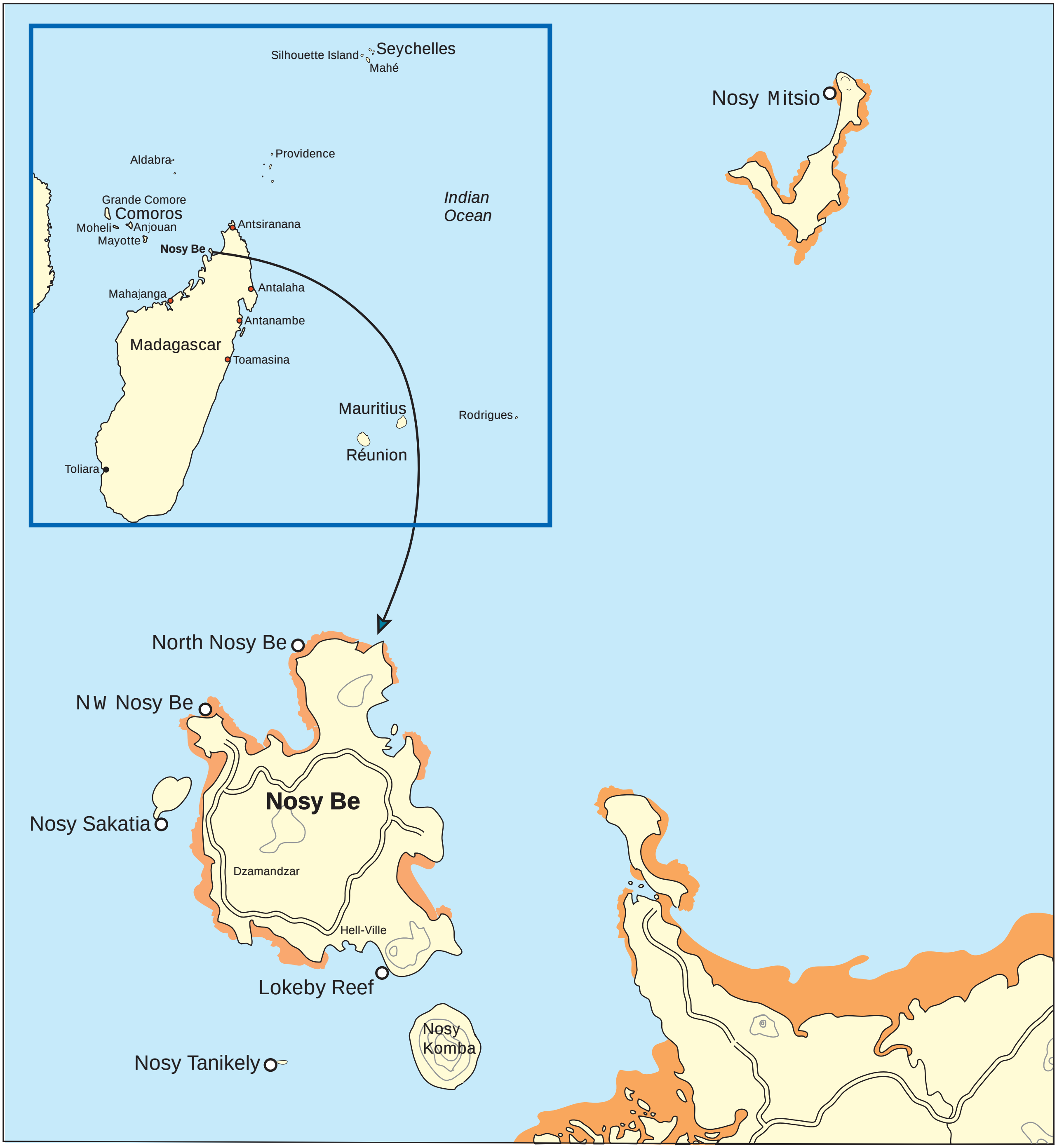

Figure 1. Location of Nosy Be region and study sites (indicated by stars). Map of Madagascar adapted from Wilkinson 2000.

Table 1. Location of the six study sites in the Nosy Be region, northwest Madagascar. Coordinates are given based on the (WGS 84). Bearing is the direction the sampling transects were laid

Site Name	Location	Bearing
Nosy Mitsio	S 12º 50' 20" E 48º 36' 49"	310º
Nosy Sakatia	S 13º 19' 13" E 48º 09' 13"	250º
Nosy Tanikely	S 13º 28' 40" E 48º 14' 30"	40º
Nosy Be – North	S 13º 12' 20" E 48º 15' 50"	225º
Nosy Be – Northwest	S 13º 14' 20" E 48º 11' 40"	40º
Nosy Be – Lokebe Reef	S 13º 25' 27" E 48º 18' 30"	150º

Study Sites

Six fringing coral reefs were studied in detail during the months of July and August 1999, in the Nosy Be region, northwest Madagascar. Three of the reefs studied were adjacent to Nosy Be Island, and three were adjacent to other smaller islands in the region (Table 1 and Fig. 1).

Potential Threats to Coral Reefs

A list of potential impacts to the coral reef was compiled for each site from observations of land and marine use practices and conversations with local residents. The following potential impact sources were examined:

- Nearby infrastructure;
- Fishing activities;
- Presence of obvious sedimentation;
- Presence of coral bleaching;
- Presence of *Acanthaster planci* (Crown of thorns starfish);
- Rivers, streams, creeks etc.

This methodology was adopted from Regional environment Programme of the Indian Ocean Commission (PRE/COI).

Physical Characteristics

At each site the following physical parameters were recorded:

- Visibility (m);
- Current Direction;
- Water depth (m).

Coral Reef Health

Coral reef health was assessed at each site by a number of parameters and techniques:

- coral cover (alive and dead);
- coral genera composition and diversity;
- abundance of *Diadema setosum* (a potential outbreak species);
- abundance of indicator fish species.

CORAL COVER ESTIMATES

At each site three 40 m transects, parallel to the shore and along the reef slope were established. At 0 m, 20 m, and 40 m, along each transect, 1 m^2 quadrats were set. Depths ranged between 3 and 6 m. Transects were accurately measured with a tape measure on a constant compass bearing and depth, and temporarily fixed to the reef. Coral cover was assessed visually by recording the percentage of live scleractinian coral and dead coral in nine 1m^2 quadrats. Dead coral was either recently killed and white with no polyps or covered in algae but the coral skeleton intact. Coral rubble was not included as dead coral.

CORAL GENERA COMPOSITION AND DIVERSITY

The total number of scleractinian coral genera was recorded in a survey zone extending 1 m either side of each transect. All coral genera present in the belt transect were identified *in situ* on SCUBA or if possible specimens collected for later identification using Veron (1986).

PERCENTAGE COVER OF CORAL GENERA

The percentage cover of different coral genera was measured by stratifying and intensifying the sampling along the three 40 m transects. At 0 m, 20 m and 40 m along

Table 2. Indicator fish species and their diet identified in the surveys. Adapted from Regional environment Programme of the Commission de L'Ocean Indien (PRE/COI)

Fish Family and Species	Diet
Chaetodontidae	
Chaetodon trifasciatus (Park 1797)	
Redfin butterfly fish	Exclusively coral polyps
Chaetodon lunula (Lacepede 1802)	
Racoon fish	Nudibranchs, worms, coral polyps, algae
Lethrinidae	
Lethrinus harak (Forsskal 1775)	
Thumbprint emperor	Polychaetes, crustaceans, mollusks,echinoderms, small fish
Lutjanidae	
Lutjanus monostigma (Cuvier 1828)	
One spot snapper	Fish, benthic crustaceans
Macolor macularis (Fowler 1931)	
Black snapper (juvenile)	Zooplankton
Haemulidae	
Plectorhinchus flavomaculatus (Cuvier 1830)	
Gold spotted sweetlips	Small fish and crustaceans
Plectorhinchus gaterinus (Forsskal 1775)	
Blackspotted sweetlips	Small fish, crustaceans
Serranidae	
Epinephelus fasciatus (Forsskal 1775)	
Black tipped cod	Fish, crustaceans, octopus

each transect a 1m^2 quadrat was established, which was divided into 25 points, 10 cm apart as shown in Fig. 2. At each of the 25 points the substrate was categorized as:

- live scleractinian coral genera;
- soft coral;
- no coral.

For each transect a total of 225 observations were recorded (25 observation points/quadrat x 3 quadrats/transect x 3 transects/site). Frequency of coral genera was calculated for each transect.

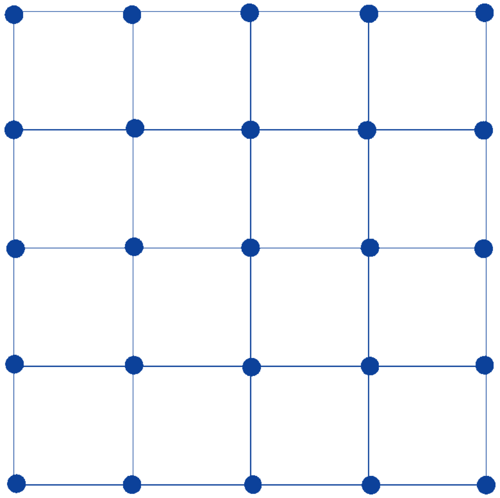

Figure 2. Sampling points in 1m^2 quadrat, located 10 cm apart.

ABUNDANCE OF *DIADEMA SETOSUM*

The density of *Diadema setosum* was recorded by counting the number of individuals in each of the nine, 1m^2 quadrats, established along the transects.

ABUNDANCE OF INDICATOR FISH SPECIES

At each site, the abundance of indicator fish (Table 2) were recorded along three replicate 50 m x 5 m transects. Fish surveys were carried out prior to any coral surveys to minimize disturbance to the fish. Additional species common in the area were also added (Table 2). These species contained a number of corallivores that are reliant soley on coral polyps for their diet and higher tropic level fish that require a varied diet found in a coral reef system such as fish, crustaceans and mollusks. *Chaetodon trifasciatus*, *Chaetodon lunula* and *Epinephelus fasciatus* are caught for the aquarium industry and the remaining fish are important in commercial fishery for human consumption.

Data Analysis

The coral cover data was not normally distributed and log transformation did not increase normality. Therefore, where possible the percentages of cover were statistically compared using the non-parametric Kruskal-Wallace ANOVA.

At each site, coral diversity was calculated by adding the total number of genera observed along the 3 replicate transects. For each site the Shannon Weiner Diversity Index was calculated using the coral genera frequency data and sites were compared using a series of t-tests (Zar, 1996).

RESULTS

Potential threats to coral reefs

At all sites except for Nosy Tanikely there was a village within a kilometer of the reef examined (Table 3). Major infrastructure was present at Nosy Sakatia (sugar refin-

Table 3. Information relating to the reef health and potential threats at each site

	Nosy Mitsio	Nosy Sakitia	Nosy Tanikely	Nosy Be North	Nosy Be Northwest	Nosy Be Lokebe Reef
Nearby infrastructure						
Small village	•				•	
Medium village		•		•		•
Hotel		•				
Other		Sugar refinery	Light-house			Abandoned soap factory Hellvile port
Fishing activities						
Nets		•		•	**?**	•
Lines		•		•	**?**	•
Fish traps	•	•		•	**?**	•
Tourist activities						
Sailing	•	•	•	**?**		
Diving	•	•	•	**?**	•	•
Boat tours	•		•			
Sedimentation	•	•		•	•	•
Coral bleaching				•	•	•

- Denotes infrastructure or activity present

? Not directly observed, but the threat probably exists ery on the mainland opposite) and nearby to Lokebe (Hellvile port). Fishing using nets, lines and/or traps (or remnants of fishing gear) was observed at all sites except Tanikely and Nosy Be NW. Fishing was carried out from pirogues (small, wooden canoes). Nosy Mitsio, Nosy Sakatia and Nosy Tanikely were observed as popular anchorages for yachts and also for snorkeling and SCUBA diving. Fine silt was present at most sites and abundant at Nosy Sakatia. There was evidence of coral bleaching at the Nosy Be North, Nosy Be NW and Lokebe reef reefs. *Acanthaster planci* and *Drupella cornus* were not observed at any of the reefs.

Physical Characters

At all sites the reef slope was between 2.8 and 5.8 m water depth (Table 4). On the change of tide the current was significant. Apart from Nosy Sakatia, the visibility was good, ranging between 5–10 m, however decreasing with tidal currents.

CORAL REEF HEALTH

Live coral cover ranged from 28% ± 5% to 51% ± 10% ($\bar{x}$ ± *SE, standard error*) although there were no significant differences between the sites (Fig. 3). Dead coral cover ranged from 0% at Nosy Sakatia to 14% ± 9% at

Table 4. The water depth, visibility and movement at each of the sites

	Nosy Mitsio	Nosy Sakitia	Nosy Tanikely	Nosy Be North	Nosy Be Northwest	Nosy Be Lokebe Reef
Water Depth (m)	2.8	5.8	4	2.9	4.3	4.0
Current Direction	N parallel to shore	SSW	S	W	NW parallel to shore	NW
Current Strength	strong on lowering tide	average with tide lowering	strong mid tide	weak	strong on lowering tide	strong on lowering tide
Visibility Range (m)	5–10	2– 5	5–10	5–10	5–10	5–10

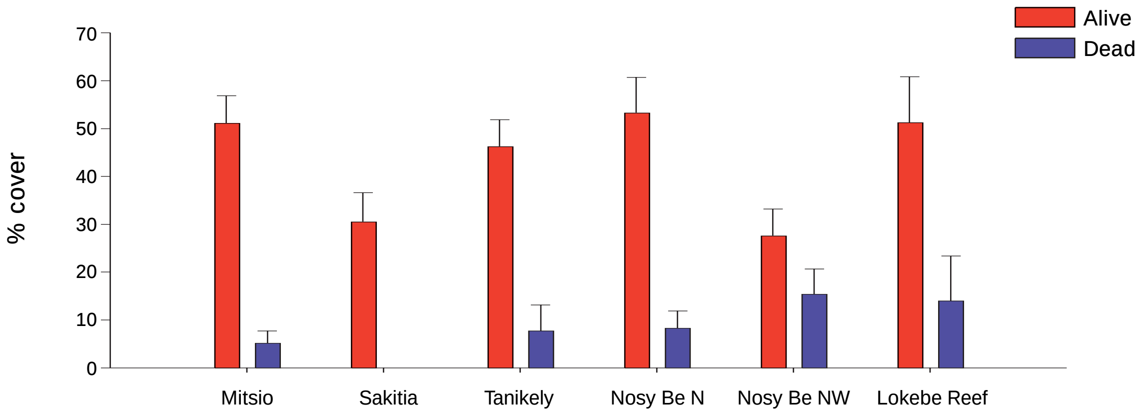

Figure 3. Average percentage of live and dead coral cover measured in 1m^2 quadrats at each site. Bars indicate standard error of the mean.

Table 5. Total number of genera observed and measure of diversity (Shannon Wiener Index) at each site.

Site	Total number of genera	Diversity Index	Site comparison Statistical difference p<0.01
Nosy Mitsio	26	0.53	c
Nosy Sakatia	30	0.75	a
Tanikely	35	0.74	a
North Nosy Be	18	0.40	b
North West Nosy Be	24	0.46	bc
Lokebe Reef	26	0.37	b

Table 6. Percentage cover of the five most common coral genera and soft coral at each site ($\bar{x}$ ± SE). Other coral genera were recorded but at a percentage of <2% and not presented here.

Nosy Mitsio	%	Nosy Sakatia	%	Nosy Tanikely	%
Acropora	28.2 ± 5.9	*Acropora*	11.5 ± 1.2	*Acropora*	26.2 ± 10.5
Montipora	21.5 ± 6.2	*Diploastrea*	10.7 ± 2.0	*Leptastrea*	3.5 ± 2.2
Leptastrea	4.0 ± 4.0	*Porites*	5.3 ± 3.1	*Favites*	1.8 ± 1.2
Unidentified	1.8 ± 1.8	*Favia*	1.3 ± 0.8	*Galaxea*	1.8 ± 0. 4
Goniastrea	0.9 ± 0.9	*Favites*	1.3 ± 0.8	*Goniastrea*	1.8 ± 1.2
Soft coral	3.6 ± 2.2	Soft coral	5.8 ± 1.2	Soft coral	0
N Nosy Be	**%**	**NW Nosy Be**	**%**	**Lokebe Reef**	**%**
Montipora	26.2 ± 8.	*Acropora*	20.4 ± 6.7	*Montipora*	29.3 ± 25.4
Acropora	25.8 ± 11.3	*Montipora*	4.4 ± 1.8	*Acropora*	19.1 ± 16.5
Favites	1.3 ± 1.33	*Astreopora*	1.3 ± 1.3	*Porites*	1.8 ± 0.4
Astreopora	0.4 ± 0.4	*Porites*	1.3 ± 0.8	Unidentified	0.9 ± 0.9
Pscammocora	0.4 ± 0.4			*Pocillopora*	0.4 ± 0.4
Soft coral	8.0 ± 3.1	Soft coral	0	Soft coral	0

Lokebe reef. Statistical comparisons of dead coral cover were not possible due to the highly skewed nature of the data. The remaining substrate at all sites was bare sand or limestone reef.

CORAL GENERA DIVERSITY

The most coral genera were present at Nosy Tanikely (35) and the lowest at Nosy Be North (18). Nosy Mitsio (26), Nosy Be NW (24) and Lokebe reef (26) had similar numbers of coral genera (Table 5). The total number of coral genera observed for all sites was 42.

The coral reefs at Nosy Sakatia and Nosy Tanikely are similar and significantly more diverse than all other sites as determined by the Shannon Weiner diversity index ($p<0.01$). The diversity indices measured at all of the Nosy Be sites (North, North west and Lokebe Reef) were not statistically different (Table 5).

PERCENTAGE OF CORAL GENERA

At most sites the most commonly recorded coral genera was either *Acropora* or *Montipora* at a frequency between 20–30% (except Nosy Sakatia) (Table 6). At Nosy Mitsio, North Nosy Be, Nosy Be NW and Lokebe reef, if *Acropora* was dominant, *Montipora* was the second most dominant, or vica versa. However, at Nosy Sakatia, after *Acropora* the next most coral genus was *Diploastrea* (10 ± 2%) and at Nosy Tanikely, *Leptastrea* (3 ± 2%). The frequency of all other coral genera accounted for less than 6% of all observations, and the majority were not recorded. The frequency of soft coral accounted for less than 10% cover at each site.

Table 7. Number of *Diadema setosum* per m^2 for each site ($\bar{x}$ ± SE)

Site	Average number
Nosy Mitsio	0 ± 0
Nosy Sakatia	0.2 ± 0.2
Nosy Tanikely	0
North Nosy Be	0
North west Nosy Be	0.2 ± 0.2
Lokebe Reef, Nosy Be	2.4 ± 1.3

DIADEMA DENSITY

Diadema was absent at three sites with the highest density at Lokebe reef (Table 7). The sites could not be statistically compared due to the distribution of the data. There was no correlation between the density of *Diadema* and the percentage of live coral (r^2= 0.018).

ABUNDANCE OF INDICATOR FISH SPECIES

The highest total number of indicator fish species was observed at North Nosy Be. Nosy Tanikely, Nosy Mitsio and Lokebe reef also had relatively high numbers of indicator fish species with a high proportion of predator species.

At least 50% of the indicator fish species were observed at all sites, with at least one coralivore and a few predators (Tables 8 and 9). The two most common fish,

Table 8. Total number of indicator fish species observed along three 50 m x 5 m transects at six sites in Nosy Be region

Fish Species	Diet	Site					
		Nosy Mitsio	Nosy Sakitia	Nosy Tanikely	Nosy Be North	Nosy Be Northwest	Nosy Be Lokebe Reef
Chaetodon trifasciatus	Corallivore	5	15	8	18	10	10
Chaetodon lunula	Corallivore	7	0	0	5	0	3
Lethrinus harak	Predator	16	1	5	1	1	21
Lutjanus monostigma	Predator	5	0	0	39	7	0
Macolor macularis	Zooplankton	0	0	9	7	1	0
Plectorhinchus flavomaculatus	Predator	2	1	0	0	0	1
Plectorhinchus gaterinus	Predator	0	2	20	0	1	0
Epinephelus fasciatus	Predator	0	0	0	0	1	0
Total number observed		35	19	42	70	21	35

Table 9. Summary of indicator fish data from six sites in Nosy Be region

	Nosy Mitsio	Nosy Sakitia	Nosy Tanikely	Nosy Be North	Nosy Be Northwest	Nosy Be Lokebe Reef
Number Species Observed	5	4	5	6	4	4
Large schools	N	N	Y	N	Y	N

observed at all sites, were the red fin butterfly fish, *Chaetodon trifasciatus* and the Black spotted emperor, *Lethrinus harak*. *Chaetodon trifasciatus* was found, usually in pairs associated with coral. Nosy Sakitia and North Nosy Be supported the highest abundance of this species. *Lethrinus harak* was found in high numbers associated with sandy patches at Nosy Mitsio and Lokobe Reef (Table 8).

Most other fish indicator species were found rarely or in low numbers, however, two species were observed in schools in high numbers, *Lutjanus monostigma* at North Nosy Be and *Plectorhinchus gaterinus* at Nosy Tanikely (Table 9).

DISCUSSION

Health of Fringing Reefs

The coral reefs examined in this study in 1999 appeared to be in good condition. The percentage of live coral cover (27–53%) was slightly lower than reported for other studies in the Nosy Be region (Wilkinson, 2000). However, compared to other major islands in the region (Wilkinson, 2000); areas along the African coast (McClanahan & Obura, 1997; McClanahan & Shafir, 1990); and the global average of 33% (Hodgson, 1999), live coral cover was high.

The percentage of dead coral cover was generally less than 10%, which suggests the areas examined were not strongly affected by the 1998 global coral bleaching event. There was some evidence of low levels of bleaching at all of the North Nosy Be, North West Nosy Be and Lokebe Reef. Other studies in different parts of Madagascar report much higher levels of bleaching (80–90% in Mclanahan & Obura, 1998; 20–50% in Wilkinson, 1998). All of the reefs examined were; in >3m of water, along a reef slope, and exposed to currents and, therefore, maybe have been less vulnerable to increased water temperatures. Alternatively, coral bleaching may have been more extensive earlier and the reefs have recovered. The low percentage of dead coral is consistent with the known absence of highly destructive activities such as dynamite blasting and coral mining.

The number of outbreak species was generally low. No specimens of *Acanthaster planci* or *Drupella cornis* were observed at any of the sites and, in general, the densities of *Diadema setosum* were much lower than reported for similar unprotected areas in Kenya (McClanahan & Shafir, 1990). Whilst not examined directly it was noted that the density of all sea urchin species was much lower than reported on heavily fished areas in Kenya (140 ± 81 per 10 m^2 in McClanahan & Shafir, 1990). These authors have reported a negative association between numbers of fish and fish species and sea urchin density. This may indicate reefs examined in the Nosy Be region were not under severe stress or similar intense fishing pressure.

Coral diversity in terms of total number of genera observed at each site (ranging from 18–35) and for all sites combined (42) was lower than reported for studies around Nosy Be (63) (IUCN 1988). However, considering the survey areas were restricted to transects, the number of coral genera recorded were high. A similar transect based study in Kenya identified only up to 25 coral genera (McClanahan & Obura, 1997).

The sites with the greatest diversity were Nosy Sakatia and Nosy Tanikely. *Acropora* and *Montipora* did not dominate the reefs at these sites and genera from other coral families were well represented. The three reefs examined at Nosy Be, were relatively similar in terms of coral diversity.

Whilst a comprehensive survey of fish species and densities was not conducted, certain indicator fish species were examined to provide some indication of reef health and fishing pressure at each study site. The abundance of *Chaetodon* and *Lutjinas* species was similar to those observed in relatively unaffected coral reefs in the Seychelles (Spalding & Jarvis, 2002). The presence of corallivores and predator species and large schools of predatory fish suggests that the coral reefs are in good health, especially at Nosy Mitsio, North Nosy Be, Nosy Tanikely and Nosy Sakatia. However, fish abundance is not always a sensitive measure of reef health (Heyward *et al.*, 1999; Holbrook *et al.*, 2002).

A recent study by Conservation International in January 2002, also assessed reef health in the Nosy Be region. The results published in the preliminary report are consistent with this study, that being the reefs in the region are generally in good condition with high coral cover, coral and fish diversity (Conservation International 2002).

Potential Threats

The coral reefs examined in the Nosy Be region are exposed to similar threats experienced elsewhere in Madagascar, which include; sedimentation, pollution overfishing, coral bleaching, and damage associated with tourism (Gabrie *et al.*, 2000). However, the reefs which were studied do not appear to be as severely impacted as other areas in the country. All sites were exposed to strong currents and well flushed which may ameliorate some environmental pressures. Gabrie *et al.* (2000) reported that the reefs in North West Madagascar were strongly affected by land erosion and sedimentation. However, in this study elevated sediments were only observed at Nosy Sakatia and North Nosy Be, but the true risk is difficult to predict in the dry season. At all sites, soft coral cover was low, which may indicate that sediment was not causing stress (McClanahan & Obura, 1997).

Most reefs examined were exposed to pollution from nearby villages. However, at all sites the abundance of algae was low suggesting that the concentration of nutrients was not excessive. The area probably most at risk from pollution was Lokebe reef with the port and associated town located around 1km away. Fishing was observed at all sites except Nosy Tanikely. Whilst not directly measured, it is known that artisinal and commercial fishing pressures are significant. The pressure from tourism is also significant in the Nosy Be region. Nosy Tanikely receives a large number of tourists daily and many undertake snorkeling or diving activities which can damage the reef. However, tourism can also have a beneficial effect by providing an income which is not detrimental to coral reefs. Tourism operators support the development of marine parks.

Marine Protected Areas

The Nosy Be region was given a high priority for the National Coastal Zone Management Program in Madagascar, in particular of Nosy Tanikely and Lokebe Reef were included as part of the marine protected areas initiative (Gabrie *et al.*, 2000). The reefs at Nosy Tanikely were in excellent condition and with many species that were not observed at other reefs including: large anemones, live ornamental shells, crayfish, turtles and holothurians.

The results from this study suggest that Nosy Sakatia may be an area more worthy of protection rather than Lokebe Reef. The coral reef diversity at Nosy Sakatia was significantly higher and the percentage of dead coral lower than at Lokebe Reef. Nosy Sakatia is known to be important for sea turtle nesting and for the existence of a well preserved forest on the island (Sagar, 2001; Rabarison *et al.*, 2000). Furthermore, the Lokebe Reef is more vulnerable to pollution and overfishing from the nearby Hellvile port and associated town.

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Island Elevations, Reef Condition and Sea Level Rise in Atolls of Chagos, British Indian Ocean Territory

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Key words: bioerosion, Chagos, coastal erosion, coral bleaching

ABSTRACT

Three years after most corals died on the central Indian Ocean reefs of Chagos, mortality remains very high to 15 m deep in northern atolls, and to >35 m in central and southern atolls. Many shallow reef surfaces have ‘dropped’ 1.5 m due to loss of dense coral thickets of *Acropora palifera*, coral bioerosion is substantial, and there is much unconsolidated rubble. Juvenile corals are abundant, though mostly are found on eroding or unstable substrates. There is a ‘race’ between erosion and new growth, whose outcome is unknown at present.

Sea surface temperature (SST) has risen 0.65°C since 1950. In 1998, the critical SST causing mortality in these atolls was 29.9°C. Accompanying sea level rise in this region is predicted to be over 0.5 cm y^{-1}.

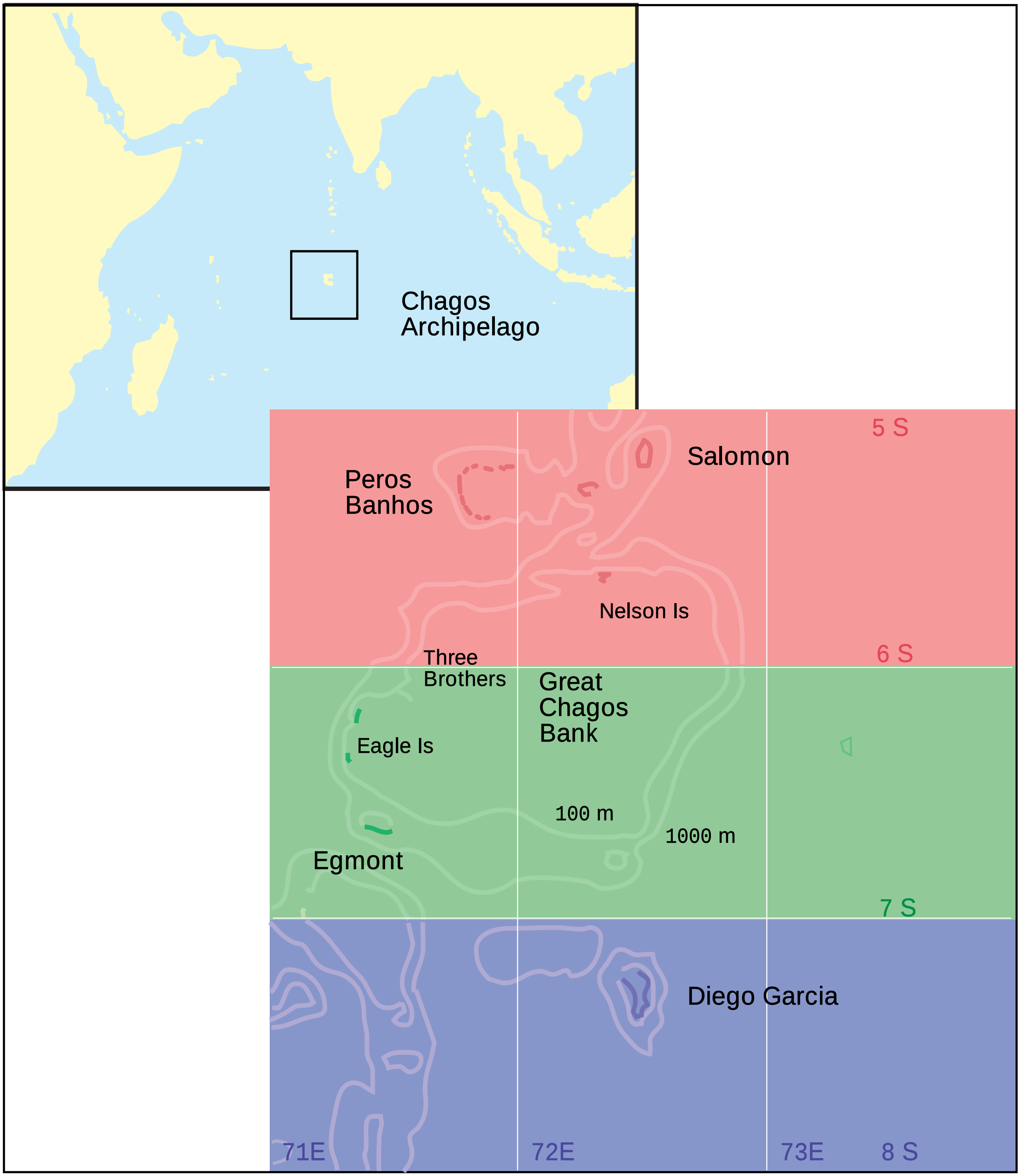
Profiles of several islands were surveyed over 25 years ago. Most islands have a raised perimeter surrounding a central depression located near or even below sea level. Protecting the islands from erosion are (or were) three ‘lines of defence’: firstly the now absent seaward coral thickets of *A. palifera*; secondly the *Porolithon* algal ridges at the seaward edge of the reef flats and, thirdly, wide expanses of reef flat located near present sea level, across which waves decay. Reduction in effectiveness of any of these will transfer wave energy inward to the shores and elevated rims of the islands. Consequences could include erosion or even breaching of island rims.

INTRODUCTION AND METHODS

Following the massive coral mortality in many Indian Ocean archipelagos in 1998 (McClanahan, 2000; Spencer *et al.*, 2000; Sheppard *et al.*, 2002), effort is being made to determine what consequences might arise which will affect local communities and shorelines. All three major functions of coral reefs, namely maintenance of diversity, productivity and coastal protection, are at risk from the mass mortality which has left many reefs markedly depleted. Much new coral recruitment has been observed (Turner *et al.*, 2000; Sheppard *et al.*, 2002) but it is uncertain what proportion of juveniles will survive the unstable substrates to reach maturity.

In the Chagos Archipelago (Fig. 1), air temperature patterns have risen about 1°C in 25 years (Sheppard, 1999a), with a corresponding reduction in cloud and an increase in wind variance. No sea temperature series were measured during that period, though recently, interpolated sea surface temperature (SST) data has become available (Sheppard & Rayner, 2002) whose patterns match closely those of actual air temperatures.

Figure 1. Location and map of Chagos Archipelago, and the grid of 9 one-degree cells for which sea surface temperature is available. Bathymetry lines are 200 and 1000 m depth. Colour coded blocks relate to figure 2.


Chagos
Archipelago
5 S
Salomon
Peros
Banhos
Nelson Is
6 S
Three
Brothers
Great
Chagos
Bank
Eagle Is
100 m
1000 m
Egmont
7 S
Diego Garcia
71E
72E
73E
8 S

It has been predicted that significant changes would take place to reefs in all affected areas of this ocean (Wilkinson, 2000). Many of the corals killed were hundreds of years old, and it was clear that, whether or not the cause is cyclical, natural, or even an isolated occurrence, an event of this magnitude has not occurred for several centuries or even millennia (Aronson *et al.*, 2000).

As well as having great intrinsic interest, the Chagos archipelago is an important stepping stone in East-West migrations of marine species across the Indian Ocean (Sheppard, 1999b). Belying the small area of its islands (4000 ha), reef habitat to 60 m depth exceeds ~13 000 km^2 (Dumbraveanu & Sheppard, 1999), giving it enormous biogeographical importance.

The responses of the coral reefs to the 1998 mortality is discussed elsewhere (Sheppard *et al.*, 2002). To date much less work has been done on the consequences of the coral mortality on the islands (in this or any other Indian Ocean archipelago), although the latter aspect is of major importance to island communities.

There are three main physical barriers to coral island erosion: the shallow part of the seaward reef slope, the reef crest, and the expansive reef flat. Abundant data on pre-1998 conditions exist for all three broad zones (e.g. Sheppard, 1980,1981), but assessment of their condition post-1998 is so far limited to that on the first, namely reef slopes (Sheppard *et al.*, 2002). Regarding the second barrier to erosion, there has been, regrettably, no good quantification of calcareous algal cover on the reef crest, though it was observed (Sheppard, 1999c) that this was noticeably reduced compared with 20 years earlier.

Regarding the third barrier (the expansive reef flats and the shorelines of the islands themselves) two relevant sets of information are now available or have been rediscovered: first is predicted sea level rise based on climate models, and second is surveyor transect levels measured in the 1970s, across several islands, from seaward to lagoon shore. These were performed on Joint Services Expeditions to Egmont atoll (southern Chagos) in 1972/3 and to islands of the Great Chagos Bank in 1975. Traditional methods of levelling using tapes and theodolites were performed along transects cut through vegetation to allow adequate lines of sight (Anon, 1973; Baldwin, 1975; and unpublished charts). These data are re-examined here in detail. The accuracy of the levels is not questioned, though the absolute elevations above a sea level datum was not fixed as accurately as would now be desirable. However, discussions with some of those who prepared the island profiles has meant that the levels redrawn here are considered most useful in the context of assessment of possible island erosion and inundation.

RESULTS

Surface Sea Water Temperature Changes

Fig. 2 shows mean SST in 9 cells of 1° latitude and longitude, centred on the Chagos archipelago. Mean SST rose over 50 years by nearly 0.7°C, or about one quarter of the magnitude of an annual cycle which is ~3°C. Air and SST track each other closely (Sheppard & Rayner, 2002); air temperature is cooler by 1–2°C, but is rising faster. The higher specific heat capacity of water explains this difference, but acceleration of SST rise seems possible.

Critical SST values for coral mortality vary according to region (NESDIS, 2001). In Chagos, 29.9°C is a key indicator of coral mortality, providing a useful index for prediction of future similar coral mortalities, though factors such as UV penetration are also important. Simple extrapolation of these data (not shown) suggests that the straight-line rise in April SST will reach this critical value in about 2020, and even mean annual SST will do so in 2030. Annual fluctuations above this 'lethal index' would be expected repeatedly well before then, and indeed nearly did so in April 2001 (NESDIS, 2001).

Sea Level Rise

Sea level (Fig. 3, page 206) is predicted to rise between about 5–20 cm by the year 2040 (IPCC, 2001). This rate equates to up to 0.5 cm per year, a value which is actually less than measured sea level rise in the nearby Maldives (Singh *et al.*, 2001) where values of between

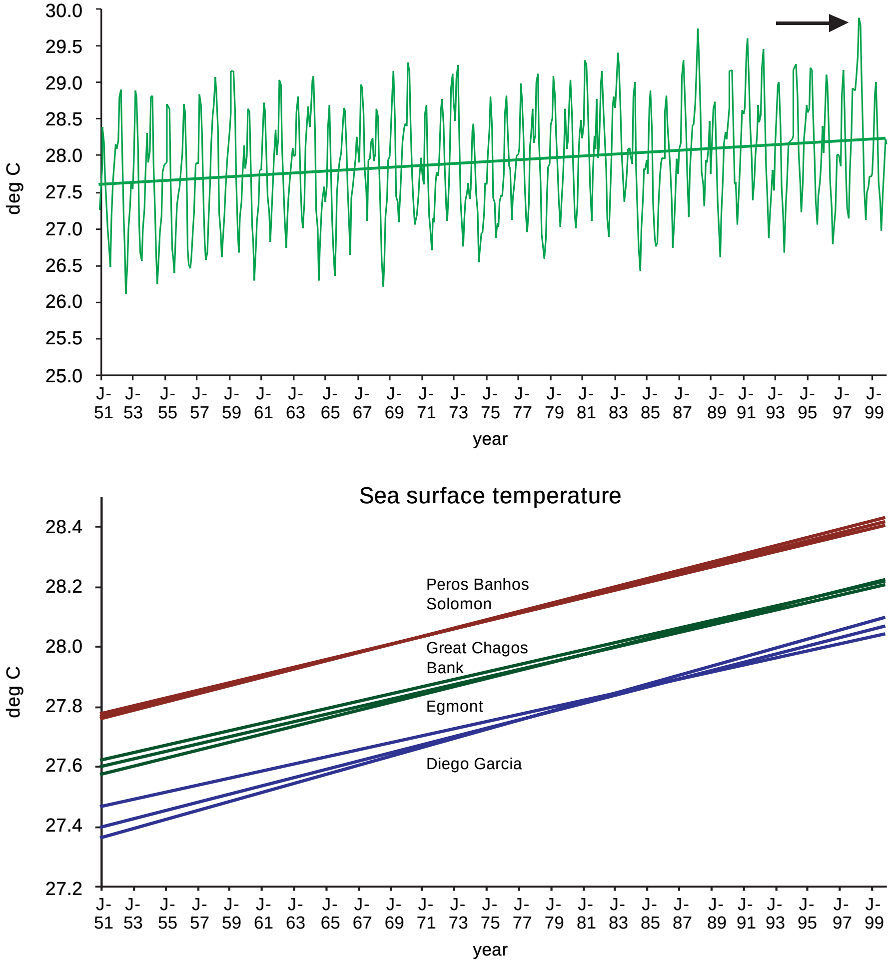
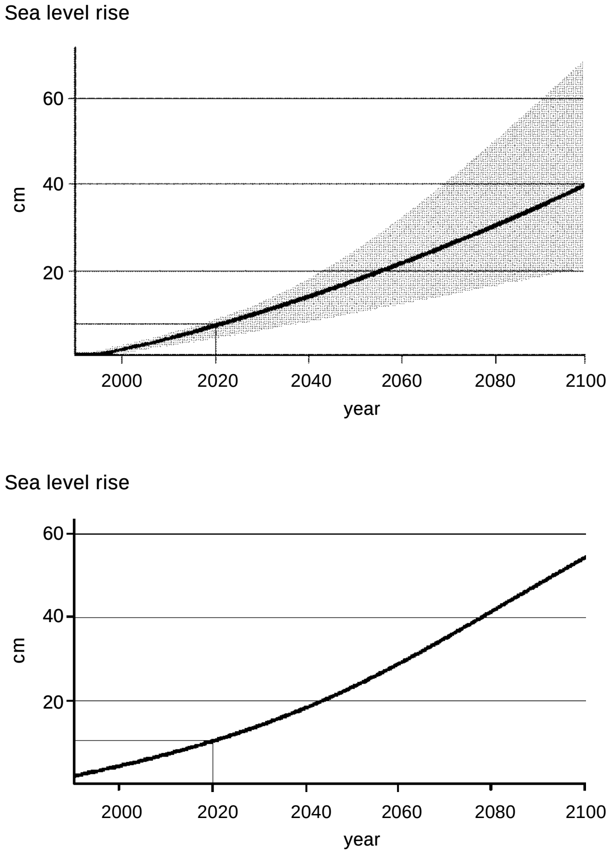


Figure 2. *Top:* Monthly mean sea temperatures of the middle row of latitude (including much of Great Chagos Bank and Egmont atoll whose islands are focussed upon here). Straight line is the simple regression line. Arrow marks the point which caused the 1998 mortality.
Bottom: Lines of best fit for all 9 cells.

5.8 and 8.5 mm y^{-1} (depending on season) have been measured over the past few years. A short series of sea level data from the Chagos indicates a rise currently of 5.5 mm per year. IPCC (2001) predicts a likely acceleration in sea level rise as time passes.

Coral Erosion and Recruitment, Sand and Rubble

In shallow water, erosion of dead corals is marked. Where the large and robust *Acropora palifera* used to dominate on seaward facing reefs, the killed colonies have been almost totally removed. Since these used to form a 1–2 m tall, impenetrable thicket to about 5 m deep, their removal has effectively lowered parts of these reef surfaces by about 1 to 2 m (Fig. 4 on next page). This is a region of great consequence to attenuation of wave energy, though erosion, rubble formation and mortality was recorded to depths considerably greater as well (Sheppard *et al.*, 2002).


Sea level rise

Sea level rise

Figure 3. Sea level rise prediction (taken from IPCC, 2001).

Figure 4. A remnant of *Acropora palifera* on a seaward reef slope. The standing skeleton shows the depth to which substrate has been removed.

Rubble is extremely abundant and shows geographical patterns. The western edge of the Chagos Bank has smaller rubble, suggesting a more advanced state of erosion. Smaller rubble (more advanced erosion) also occurs in larger atolls. Of probable importance, but so far unmeasured here or elsewhere, is the fact that further erosion of rubble will create large quantities of sand. The importance of this here is the question whether this increased sand will assist maintenance of islands *via*

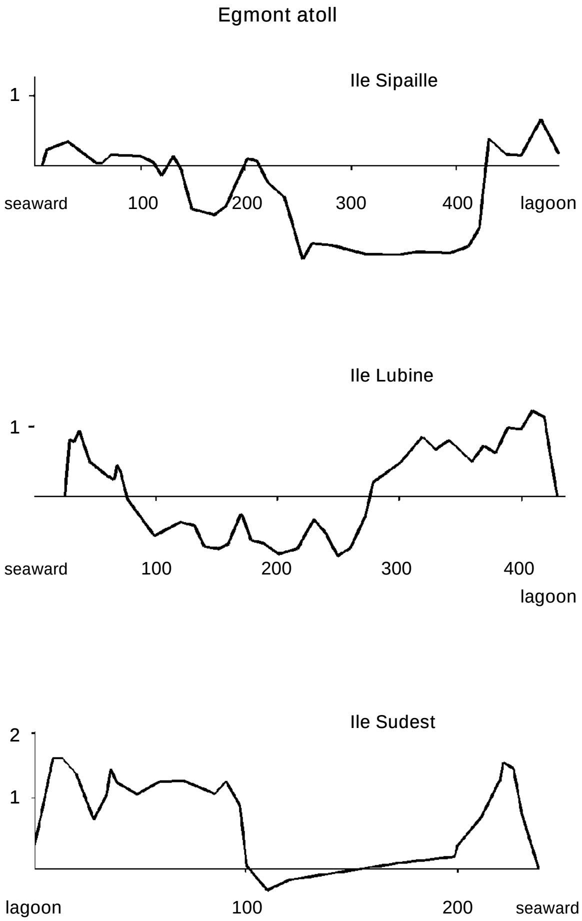

Egmont atoll

Figure 5. Profiles of islands of Egmont atoll drawn for survey transects carried out in 1972/3. Numbers are all metres. Redrawn from unpublished charts loaned from M. Hirons.

build-up of beaches. However, at some locations, chutes of scoured substrate indicate that much of this newly created rubble is being carried into deeper water.

Shores and Island Profiles

Fig. 5–7 show profiles of several islands, measured in the 1970s. Remarkably, most of them show elevated rims surrounding a depression which reaches to near sea level or even below it. It was noted that the absolute elevation above a sea level datum was not accurately fixed, but discussion with the chief surveyors has allowed the x-axes of these profiles to be adjusted approximately to present high tide level. Spring tidal range here is approximately 1 m.

The central depressions of these islands are sometimes marked and are clearly obvious even on thickly vegetated islands. Formed possibly by solution by mildly acidic rain acting on the basic limestone rock, the de-

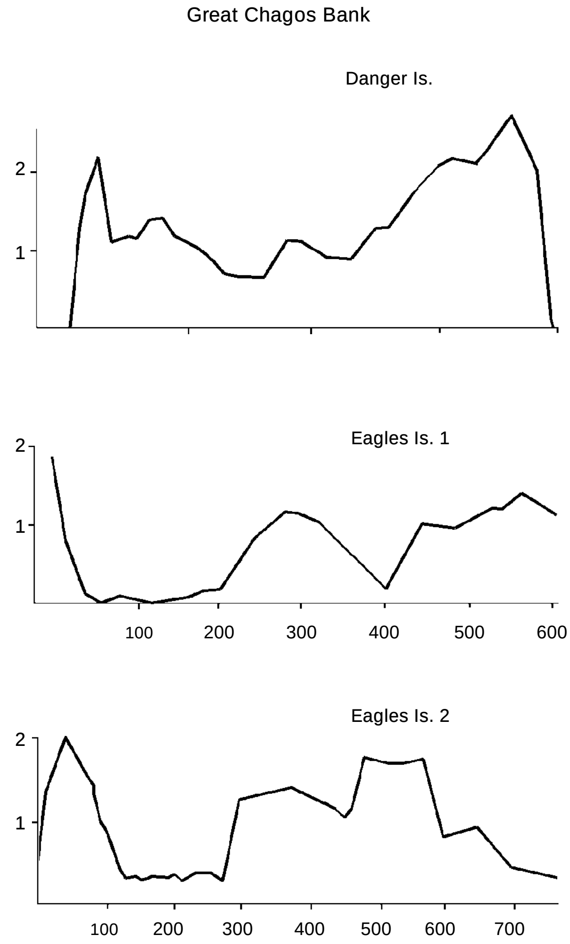


Figure 6. Profiles of islands of the Great Chagos Bank drawn for survey transects carried out in 1975. Numbers are all metres. Redrawn from Baldwin (1975).

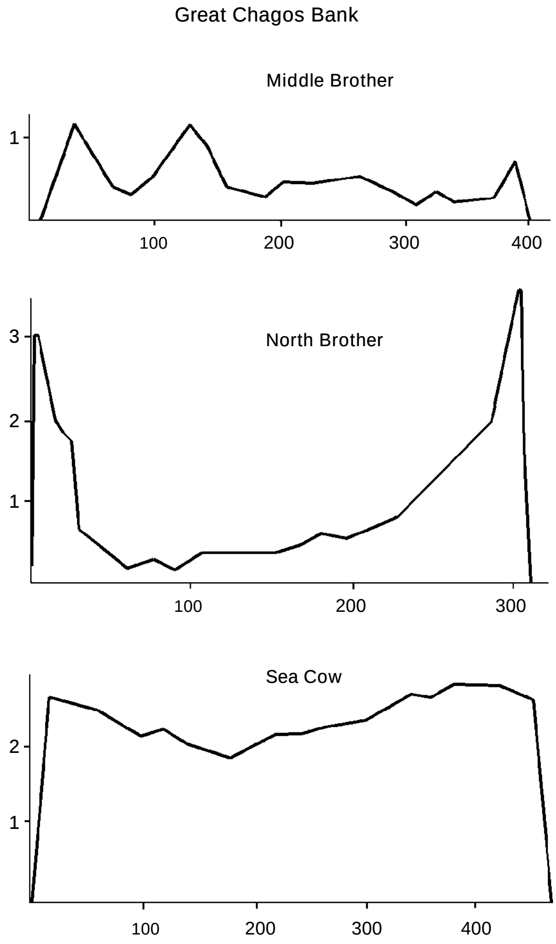


Figure 7. Profiles of islands of the Great Chagos Bank drawn for survey transects carried out in 1975. Numbers are all metres. Redrawn from Baldwin (1975).

Figure 8. North Brother island whose small but impressive cliffs are raised reefs. Note the profile of this island in Figure 7.

pressions extend variously over a small portion to most of an island. On some (e.g. North Brother, Fig. 8) the effect is almost of a thin shell surrounding a fertile and vegetated centre, while on others (e.g. the similarly sized island Sea Cow) the depression is small and the entire island is well above sea level. The largest islands measured in this way are those of Egmont atoll (Fig. 5) and Eagle Island (Fig. 6), both previously inhabited when copra was a valuable product. During and after heavy rain, freshwater 'lakes' may form in the interior of these islands, which seep gradually away over a few hours. Such is the flow of water that after episodes of rain, elevated nutrient levels may be detected offshore (Rayner & Drew, 1984). Given these island profiles it is likely that these nutrients accompany seepage through the very porous limestone rock rather than surface run-off. This high porosity is confirmed by the fact that disused wells located well inland in these Chagos islands show tidally linked rises and falls in fresh water level (Baldwin, 1975; Griffiths, 1979; Hunt, 1997).

Similar depressions are clearly visible in many islands in the two northern atolls. Commonly the effect may be visually exaggerated due to piling up of sand around the rims of islands, but in most cases coral rock visibly outcrops around the rims, and it is predicted that profiles are similar. Two sites in western Peros Banhos atoll appear to see this process taken much further: In Ile Monpatre and Ile Diamant, tidal channels split each island into two along their long axes, i.e. parallel with the atoll rim, leaving separate 'seaward-side' and 'lagoon-side' islands.

DISCUSSION

Most work following the coral mortality of 1998 has focussed on sublittoral reef conditions. Little so far has been done on researching consequences of reef mortality to biodiversity or to fisheries productivity, and even less has been done on the possible consequences of this mortality and sea level rise to shorelines and to islands hitherto protected by the reefs. Indeed, shoreline protection is traditionally seen as an engineering problem rather than one for biologists to worry about!

The effects of mortality, its possible recurrence, and sea level rise will clearly compound each other to a considerable degree. SST is trending upwards, so the strong possibility exists of repeat occurrences of coral mortality. Trends of sea level rise are likewise upwards (IPCC, 2001), further increasing clearance between wave crests and reef substrate. Rising sea level means increasing depth of water above the reef flat, which thus becomes decreasingly able to attenuate wave energy. The loss of shallow corals to seaward of the reef flat decreases the capacity for wave attenuation before the reef flat is reached. There is no good data on the condition of the reef crest, but incidental observations (Sheppard, 1999c) suggested that cover by red algae in such areas was reduced by about half following 1998. At present, therefore, it appears likely that all three natural 'lines of defence' are being reduced.

Two consequences are likely. First is erosion of the elevated rims of islands. This was observed to be taking place in one or two areas which are well known to this author, though no measurements have yet been taken. If erosion progresses, clearly breaching of the rim could occur, which would lead to sea water flooding of the central portion of the island. Fig. 9 shows erosion of the

Figure 9. Eroded rim of the island of Ile de Coin, Peros Banhos atoll. Vertical height of the eroded ‘cliff’, from vegetation at top to sand at foot, is nearly 1 metre.

rim of one such Chagos island on Peros Banhos atoll, inhabited until the mid 1970s. This erosion certainly took place since that time (personal observations). Fig. 10 illustrates the results of flooding of depressions in two Chagos islands; in one case a breach is also visible. It is not necessarily suggested that this is a consequence of recent sea level rise, but it illustrates the effect.

Secondly, rainfall on these islands is between 2,500 and 4,000 mm y^{-1} (Topp & Sheppard, 1999), which is the highest of any Indian Ocean archipelago. It may be presumed that the concave surfaces of these islands retain most of the rain water, which then percolates into the rock, with very little being lost by lateral surface run-off. Because of this and the high fresh water input, all larger islands maintain a fresh water lens, which in turn supports vigorous vegetation and, in past times, a human population. Small water lenses can be maintained only when sufficient fresh water falling on the island can maintain a pressure, or flow, outwards through the porous rock to a degree which exceeds inward pressure from sea water. A rising sea level will increase the inward pressure of sea water, potentially compromising the fresh water interior. This appears likely even if there is no breach of the raised island rim. These islands include both remarkable remnants of native Indian Ocean island hardwood (Topp & Sheppard, 1999) and very important populations of sea birds (Symens, 1999) as well as recovering populations of species such as turtles (Mortimer & Day, 1999), all of which are likely to be threatened by a sea level rise much smaller than that needed to completely cover the islands.

Figure 10. *Top.* Flooded depression in Ile Anglais, Salomon atoll. *Bottom.* Small islet in Diego Garcia atoll, illustrating a breach of island rim and ingress of sea.

It might be suggested that reefs can simply grow upwards to match a rising sea level. After all, colony extension rates of most coral species exceed the rate of sea level rise. However, reef growth and coral colony growth are not the same thing, though they have been commonly confused, and reef growth in atolls has long been known to be only 0.2 to 3 mm y^{-1}, much slower than rates of extension of coral colonies (e.g. Hopley, 1982). Formation of durable reef substrate such as that of which a reef flat is made, as contrasted with simple growth of coral colonies, is a complex and poorly understood issue, and indeed, reef growth and coral growth even become 'uncoupled' in many situations where conditions are sub-optimal (Sheppard *et al.*, 1992). In the Chagos archipelago there are as many reefs and 'drowned atolls' which failed to reach present sea level as there are islanded atolls, even though the last several thousand years has showed no significant sea level change. We can, unfortunately, have little hope that reefs will keep up with a sea level rise of the magnitude currently being predicted over the next 20 years and longer. This means there will be an increasing height of water above the reef flats, with corresponding reduction in the ability of reef flats to attenuate wave energy.

Although new coral recruitment is high, over the next few years and decades it is likely that coral mortality events will recur. The projected temperature trends illustrate the point (IPCC, 2001). SST in April 2001 reached to about 0.5°C of the very damaging 1998 value. This being the case, the fine balance between reef growth and erosion could worsen, and there may be more widespread problems of erosion throughout the archipelago. Even if artificial means such as breakwaters could be developed to resist breaching of island rims without creating damage to other aspects of these ecosystems, the worry remains that increased inwards pressure from rising sea levels will increase. This could threaten the island interiors, their freshwater lenses and the biota supported by them.

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I gratefully acknowledge the efforts of many expeditionary companions who (with this reluctant author) cut numerous transects through very dense vegetation across these islands, in the 1970s. At the time we had no idea why we were doing such arduous work when we could have been diving, but in good Services tradition relied solely on the wisdom of our expedition leaders. Now we clearly see their prescience and foresight!

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Coral Communities around the Seychelles Islands 1998–2002

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INTRODUCTION

In recent years, coral reefs around the inner granitic islands of the Seychelles have suffered significant reductions in the live cover of scleractinian hard corals as a result of localised crown-of-thorns starfish (*Acanthaster planci*) outbreaks as well as the most severe mass coral bleaching event on record. Active outbreaks of *A. planci* were first reported on reefs in the northern parts of Mahé Island in late 1996. Localised starfish outbreaks remained active until the middle of 1998. The most severe coral bleaching event recorded in recent history (Wilkinson, 1998) affected Seychelles' reefs in early 1998 with the peak in bleaching-induced coral mortality occurring between February and May. Locally, hard coral mortality peaked at around 80–95% of the initial population (Engelhardt, 1998). Excessively high sea surface temperatures combined with excessively high levels of UV radiation during prolonged calm periods provided the trigger for the bleaching event. Temperatures in access of 33° C were recorded not only in surface waters but also at greater depths down to some 20 metres (D. Rowat, pers. com.). The 1998 circum-global coral bleaching event has variously been described as the worst such event in recorded history with large-scale coral death observed in many reef areas throughout the Indo-Pacific region (Wilkinson, 1998).

Dedicated benthic surveys aimed at providing an assessment of selected biological and environmental characteristics of bleaching affected coral communities were initiated in late 1999. Since then, repeat surveys conducted in 2000 and, most recently, in early 2002 have provided a detailed assessment of the early stages of local and regional-scale patterns of hard coral recovery following the mass bleaching event. This status report provides an overview of recent trends in coral community characteristics such as percent cover of both hard and soft corals as well as hard coral diversity.

MATERIALS AND METHODS

Benthic surveys were conducted in November 1999, November 2000 and most recently in January and February 2002 with some 15, 22 and 40 individual reef sites surveyed respectively. The recent geographic expansion of the study area reflects the increased availability of dedicated research funds through UNEP's Global Environmental Facility (GEF). A complete listing of the 40 reef sites surveyed in early 2002 is provided in Table 1. Study site locations are shown graphically in Fig. 1 (page 214).

Table 1. Overview of the 40 coral reef sites surveyed in January and February 2002

Region	Location	Site Name	Site No.	Habitat
West	Ile Thérèse	S Point	1	G
West	Ile Thérèse	N Reef	2	C
West	Mahé Island	Port Launay SW Point	3	G
West	Conception Island	NE Reef	4	C
West	Mahé Island	Baie Ternay W Reef	5	C
West	Mahé Island	Baie Ternay E Reef	6	C
West	Mahé Island	Baie Ternay E Point	7	G
West	Mahé Island	Ray’s Point	8	G
West	Mahé Island	Willie’s Bay Point	9	G
West	Mahé Island	Willie’s Bay Reef	10	C
West	Mahé Island	Anse Major Reef	11	C
West	Mahé Island	Anse Major Point	12	G
West	Mahé Island	Black Rock	13	G
West	Mahé Island	White House Rocks	14	G
West	Mahé Island	Auberge Reef	15	C
West	Mahé Island	Corsaire Reef	16	C
West	Moyenne Island	E Reef	17	C
West	St Anne Island	SE Point	18	G
West	Beacon Island	N Reef	19	C
West	Beacon Island	S Rocks	20	G
East	Cousine Island	S Point	21	G
East	Cousin Island	W Reef	22	C
East	Aride Island	W Reef	23	C
East	Aride Island	E Reef	24	C
East	Curieuse Island	W Rocks	25	G
East	Praslin Island	Anse Petit Cour	26	C
East	Curieuse Island	Coral Gardens Reef	27	C
East	Curieuse Island	Point Rouge Rocks	28	G
East	St Pierre Island	N Point	29	G
East	St Pierre Island	SE Point	30	G
East	Roches Boquet	S Rocks	31	G
East	Round Island	SW Rocks	32	G
East	Round Island	SC Point	33	G
East	Ave Maria Rocks	S Point	34	G
East	Petit Soeur	SE Point	35	G
East	Grand Soeur	W Reef	36	C
East	Felicité Island	NW Reef	37	C
East	Felicité Island	SW Reef	38	C
East	Marianne Island	SW Reef	39	C
East	Marianne Island	S Reef	40	C

C = carbonate reef habitat; G = granitic reef habitat

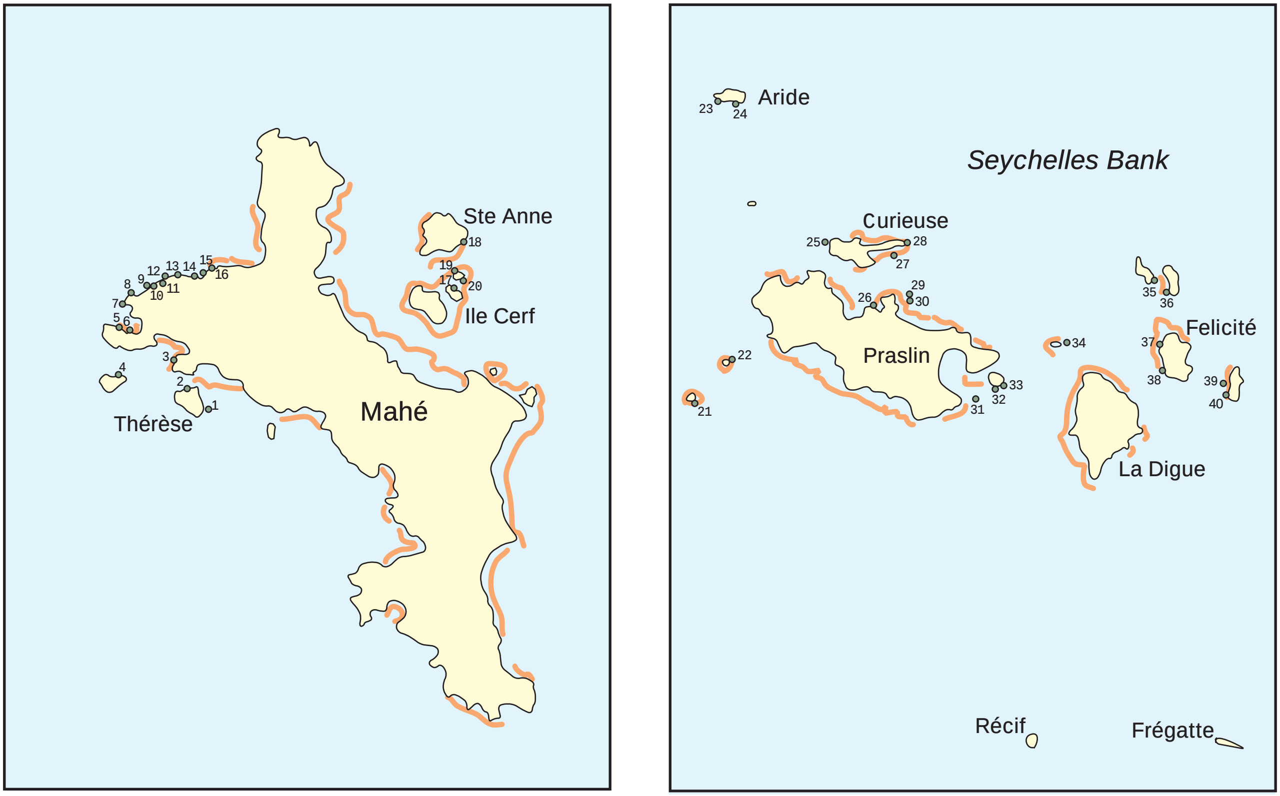

Figure 1. Map of the inner granitic islands of the Seychelles showing the approximate location of the 40 individual reef sites surveyed in January and February 2002.

Sampling Protocol

The benthic sampling protocol used here aims to provide reliable scientific data on (i) hard and soft coral abundance (quantitative % cover estimates) as well as (ii) hard coral diversity (qualitative estimates of coral diversity at the Family and Genus level).

Estimation of Hard Coral Abundance (Quantitative % Cover Estimates)

A comprehensive methodological study by Mapstone and Ayling (1998) showed that, for visually assessing the abundance of discrete benthic organisms, such as corals and some mobile invertebrates, benthic belt transects measuring 50 x 5 metres often provide the least biased density estimates, a key concern for field surveys, particularly when considering the operational constraints of most survey programs. In the central Great Barrier Reef region, 50 x 5 m benthic belt transects have been used in a detailed ecological field study undertaken to investigate the complex predator-prey relationship that exists between scleractinian hard coral communities and the crown-of-thorns starfish (*A. planci*) (Engelhardt *et al.*, 1997, 1999 & 2001). Similarly, 50 x 5 benthic transects have also been employed as part of the extensive coral reef monitoring program now operating in the Republic of Seychelles (Engelhardt, 1998, 2000 & 2001).

Throughout this study, both the selection of survey sites and the placement of individual transects within sites were haphazard at all times. Two replicate 50 x 5 metre (2 x 250 m^2) transects were sampled at each site.

Transects were placed at an oblique angle down the available site profile from as shallow as possible (typically 1–2 m depth) to a maximum depth of 15 metres. The placement of transects across the available reef profile and depth range ensures the collection of a representative sample that truly reflects the range of environmental conditions that exist at the site. In contrast to rigidly depth-stratified sampling (i.e. where two fixed depths have been pre-selected for transect placement), the abovementioned method avoids possible problems of missing data and, consequently, insufficient replication due to local-scale differences in individual sites' profiles where exact matching pairs of depth-specific transects can not be obtained due to differences in reef topography.

Since the inception of this study in 1999, reliable estimates of hard coral abundance have been obtained using either one of two widely used and accepted estimation methods – the Line-Intercept Transect (LIT) methodology and the Visual Estimation Transect (VET) methodology. Surveys conducted in 1999 and 2000 used visual estimation transects to estimate % coral cover. Visual estimates of live hard coral cover (% LHCC) and soft coral cover (% SOCC) for individual transects were recorded as '10%-range estimates' – that is coral cover was visually estimated as a % cover category with a 10%-range (i.e. 5–15%, 25–35%, 40–50% etc.). The only exception to this rule was the use of a smaller 5%-range estimate where LHCC or SOCC was found to be extremely low and was estimated to be less than 5% cover. Individual range estimates were recorded for each of the five 10-m segments that comprise the full 50 x 5 m benthic transect – namely the 0–10, 10–20, 20–30, 30–40 and 40–50 m segments.

The mid-points of the 10 individual range estimates (5 segments x 2 transects) recorded at each site were used to calculate the mean percent cover for individual reef sites, (i.e. a midpoint of 10% is used where the range estimate is recorded as 5–15%). Throughout the analyses, the estimation error was set either at ± 5% (for all 10%-range estimates) or ± 2.5% (for 0–5% cover estimates only). This conservative error margin (which is typically higher than the calculated standard error would be!) was chosen as it more accurately reflects the inherent spatial variability and patchy distribution of most coral reef organisms including corals (Mapstone & Ayling 1998).

In contrast, the 2002 surveys used line-intercept transects throughout to obtain coral cover estimates for the 40 sites surveyed. Line-intercept data were recorded within three separate 10-m long segments along each of the full 50 m long belt transects. Specifically, LIT data was recorded in the 0–10, 20–30 and 40–50 m segments along each transect. As a direct result of the transect placement down the reef slope at an oblique angle, the 0–10 m segments always corresponded to the shallowest depths sampled while the 40–50 m segments corresponded to the deepest section with the 20–30 m segment sampling benthic cover at intermediate depths.

This status report presents LIT summary data on percent live hard coral cover (% LHCC; combined total estimate for all scleractinian hard corals recorded) as well as percent soft coral cover (% SOCC; *Sinularia* and *Sarcophyton* spp. only!). For subsequent statistical analyses, individual LIT segments were considered as independent measurements as their respective starting and/or end points did not affect each others physical location. A site-specific estimate of the % cover of respective benthic life forms was calculated using the data obtained from the 6 individual LIT segments sampled at each site (3 segments x 2 transects). The sampling design used to test for the possible effects of selected environmental parameters is shown in Table 2 (next page).

Estimation of Scleractinian Hard Coral Diversity (Qualitative Description)

Site-specific measures of scleractinian coral diversity were obtained by locating hard corals occurring within the full 50 x 5 m transect and identifying them to both Family and Genus level. An overview of the observed local and regional-scale patterns of hard coral diversity is provided in the resulting absence/presence data tables (see Results section).

A site-specific categorical abundance ranking for in-

Table 2. Overview of sampling (ANOVA) design used to test for effects of Region, Habitat and Depth on the abundance (% cover) of scleractinian hard corals (LHCC) and soft corals (SOCC).

Factor	Levels	Type & Selection	F – RATIOS
(A) Region	West (1 East (2)	Orthogonal / fixed	MS(A) / MS(r)
(B) Habitat	Carbonate (1) Granitic (2)	Orthogonal / fixed	MS(B) / MS(r)
(C) Depth	Shallow / Intermed. / Deep	Orthogonal / fixed	MS(C) / MS(r)
(AB) Region x Habitat	Interaction term	–	MS(AB) / MS(r)
(AC) Region x Depth	Interaction term	–	MS(AC) / MS(r)
(BC) Habitat x Depth	Interaction term	–	MS(BC) / MS(r)
(ABC) Region x Habitat x Depth	Interaction term	–	MS(ABC) / MS(r)
Replicate sampling units residual error term (r)	10-m LIT segments (total of 240 segments)	–	

dividual coral groupings (i.e. families or genera) was calculated using the density criteria listed in Table 3. Furthermore, a regional abundance index for individual coral families and/or genera was calculated by dividing the sum of all numerical abundance codes determined for each coral type across each region by the total number of reef sites surveyed in that region (n = 20).

Table 3. Overview of categorical abundance classes and density criteria used to classify the relative abundance of scleractinian hard coral families and/or genera as recorded across reef sites in the inner granitic islands of the Seychelles in January and February 2002

Abundance category	Abundance code	Mean no. of coral colonies per 12 m^2	Standardised mean no. of colonies per 1 m^2
Absent	0	0	0
Rare	1	0.1– 3.0	0.1 –0.25
Low	2	3.1–12.0	0.26–1.0
Moderate	3	12.1–36.0	1.1 –3.0
High	4	36.1–60.0	3.1 –4.0
Very high	5	>60.0	>4.0

Statistical Analyses

Survey data were analysed using both simple descriptive statistics and, where appropriate, more advanced parametric or non-parametric statistical procedures. However, commonly used parametric tests could not be applied in some instances where actual count data were highly non-normally distributed. As a result of the substantial numbers of individual 'zero' counts, even commonly advocated data transformations did not achieve acceptable levels of 'normality' to allow for the use of standard parametric tests such as ANOVA techniques. Consequently, such data sets were analysed using non-parametric Kruskal-Wallis tests (Kruskal & Wallis, 1952). Kruskal-Wallis analyses of variance by rank procedures test for significant differences between sample medians. These procedures test the null hypothesis that the medians of the selected variable within each of the selected levels are the same. Data from all the levels are first combined and ranked from smallest to largest. The average rank is then computed for the data at each individual level. Where the P-value is less than 0.05, there is a statistically significant difference amongst the medians at the 95.0% confidence level. Actual Kruskal-Wallis test statistics and related *p* values are shown throughout relevant sections of the survey results.

RESULTS

Scleractinian Live Hard Coral Cover

Across the survey area, scleractinian live hard coral cover (LHCC) remains very low with a grand mean of just under 5% live coral cover recorded across the 40 sites surveyed in early 2002 (Fig. 2). However, whilst the overall percent cover is still very low the latest results do suggest that there has been a slight increase in LHCC since the previous survey conducted in November 2000 when the mean percent cover of live corals was recorded at only 3% across the 22 sites surveyed that year. It should, however, be noted that this is an indicative trend only as more powerful statistical comparisons of the data are precluded due to differences in both sampling effort and methodology.

A comparison of the LHCC estimates obtained for the 20 individual sites surveyed in each of the two habitat types – carbonate and granitic reefs – identified some significant differences. On average, carbonate reef sites recorded significantly lower percent coral cover values than those of granitic reef sites. A total of 15 (75%) of the 20 carbonate reef sites surveyed recorded mean LHCC values of less than 5% with 3 (15%) sites recording values of between 5 and 10% live cover (Fig. 3). Only 2 (10%) sites (No. 26 – Anse Petit Cour & No. 27 – Coral Gardens Reef) recorded slightly higher mean values of between 10 and 15% LHCC.

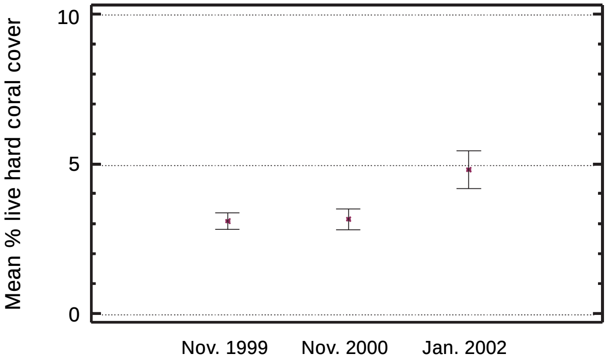

Figure 2. Graph showing recent trends in the *percent cover of live hard corals (LHCC)* across reef sites located in the inner granitic islands of Seychelles. Trends shown are indicative only as both sampling methods and sampling intensity differed between periods. Both 1999 (n=87) and 2000 (n=132) estimates are based on visual estimation (VET) data whilst the 2002 (n=240) estimate is based on line-intercept (LIT) data. (Error bars show 95% confidence intervals around the mean.)

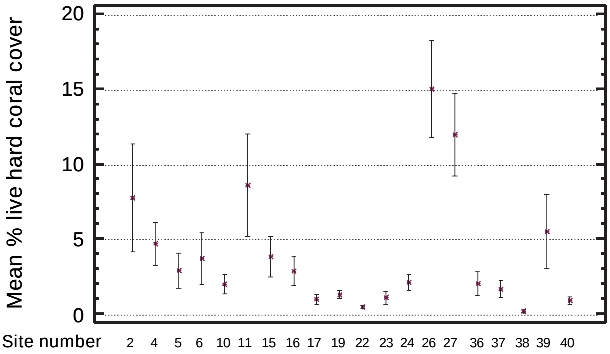

Figure 3. Graph showing *live hard coral cover (LHCC)* estimates recorded across the 20 carbonate reef sites surveyed in January and February 2002. (Error bars show SE.)

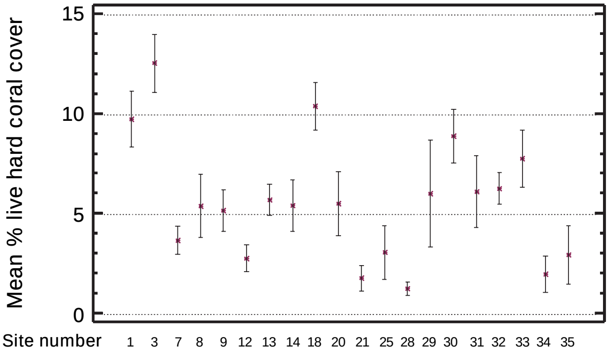

Figure 4. Graph showing *live hard coral cover (LHCC)* estimates recorded across the 20 granitic reef sites surveyed in January and February 2002. (Error bars show SE.)

In contrast, mean LHCC estimates for granitic reef sites were slightly higher with only 7 (35%) sites recording percent cover estimates below 5% live coral cover (Fig. 4). A majority of granitic reef sites (55%) recorded

Table 4. ANOVA table showing the results of statistical significance tests to assess the effects of Region, Habitat and Depth on the abundance (% cover) of *scleractinian live hard corals (LHCC)* (*p* values of less than 0.05 indicate the existence of a significant effect on LHCC).

```
Analysis of Variance for LHCC - Type III Sums of Squares
--------------------------------------------------------------------------------
Source             Sum of Squares     Df      Mean Square    F-RatiO    P-Value
--------------------------------------------------------------------------------
MAIN EFFECTS
  A:Region                47.8827      1          47.8827       2.01     0.1571
  B:Habitat                157.14      1           157.14       6.61     0.0108
  C:Depth                 43.7016      2          21.8508       0.92     0.4002

INTERACTIONS
  AB                      77.2935      1          77.2935       3.25     0.0726
  AC                      18.4456      2          9.22279       0.39     0.6788
  BC                      121.995      2          60.9975       2.57     0.0790
  ABC                     17.2217      2          8.61087       0.36     0.6964

RESIDUAL                  5418.13    228          23.7637
--------------------------------------------------------------------------------
TOTAL (CORRECTED)         5901.81    239
--------------------------------------------------------------------------------
All F-ratios are based on the residual mean square error.
```

LHCC values of between 5 and 10% with a further 2 (10%) sites estimated to support between 10 and 15% live coral cover.

Statistical significance (Analysis-of-Variance, ANOVA) tests confirmed the significant effect of the factor Habitat on the mean abundance of scleractinian hard corals (*p* = 0.01; Table 4; Fig. 5). However, the other two factors tested – Region and Depth – did not have any significant effect on the mean % cover of live hard corals (Table 4).

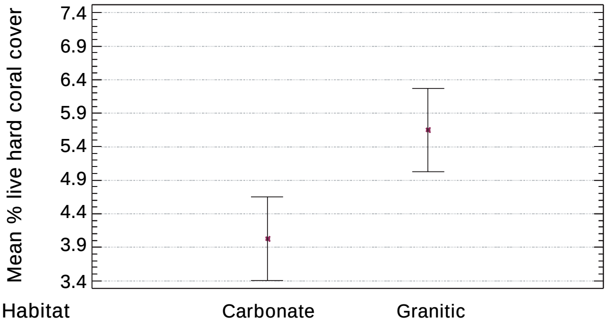

Figure 5. Graph showing significant *Habitat-effect* on the abundance (mean % cover + LSD) of *scleractinian live hard corals (LHCC)*.

Live Soft Coral Cover

Live Soft Coral Cover (SOCC; *Sarcophyton* and *Sinularia* spp. only) too remains at very low levels with grand mean percent cover across the 40 sites estimated at around 4.5% (Fig. 6). As with live hard coral cover (LHCC) these latest estimates are indicative of a small positive trend in the overall abundance of soft corals. Whilst factors such as differences in sampling methodology and sampling effort again preclude the application of more powerful statistical tools, soft coral abundance data collected since 1999 are suggesting a recent slight increase in the percent cover of *Sarcophyton* and *Sinularia* spp. in at least part of the study area.

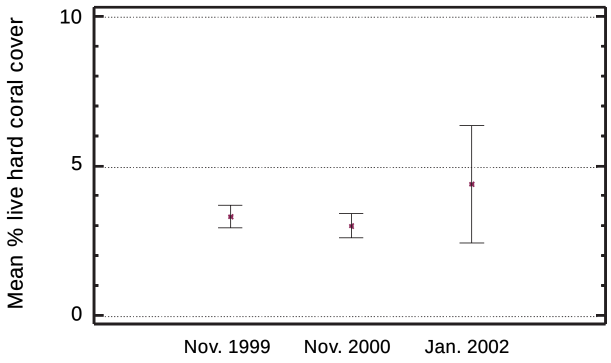

Figure 6. Graph showing recent trends in the *percent cover of soft corals (SOCC)* across reef sites located in the inner granitic islands of Seychelles. Trends shown are indicative only as both sampling methodology and effort differed between periods. Both 1999 (n=87) and 2000 (n=132) estimates are based on visual estimation (VET) data while the 2002 (n=240) estimate is based on line-intercept (LIT) data. (Error bars show 95% confidence intervals around the mean.)

Statistical significance tests (ANOVA) reveal the importance of a number of key environmental factors on the overall abundance of soft corals, with two of the three factors tested identified as having a significant effect on % cover estimates of soft corals (Table 5). The analyses identified both Region as well as Habitat effects on the mean % cover of soft corals. However, the emerging pattern is complicated by the fact that the analysis also identified a significant interaction between these two factors.

The mean percent cover of soft corals was found to be significantly higher in the western region of the study area ($p<0.01$; Table 5; Fig. 7 on next page) where SOCC was estimated to average about 9% across the 20 sites surveyed. In contrast, soft corals were virtually absent from the eastern region with a mean estimate of less than 1% SOCC being recorded.

A similarly significant difference in soft coral abundance was also identified to exist between the two habitat types surveyed in this study. The mean % cover of

Table 5. ANOVA table showing the results of statistical significance tests to assess the effects of Region, Habitat and Depth on the abundance (% cover) of *soft corals (SOCC)*.

```
Analysis of Variance for SOCC - Type III Sums of Squares
-----------------------------------------------------------------------
Source             Sum of Squares     Df      Mean Square   F-Ratio   P-Value
-----------------------------------------------------------------------
MAIN EFFECTS
  A:Region                3969.07      1          3969.07     19.07    0.0000
  B:Habitat               2747.27      1          2747.27     13.20    0.0003
  C:Depth                 279.075      2          139.537      0.67    0.5124

INTERACTIONS
  AB                      2196.15      1          2196.15     10.55    0.0013
  AC                      189.758      2          94.8792      0.46    0.6344
  BC                      97.7083      2          48.8542      0.23    0.7909
  ABC                      34.225      2          17.1125      0.08    0.9211
-----------------------------------------------------------------------
RESIDUAL                  47443.4    228          208.085
-----------------------------------------------------------------------
TOTAL (CORRECTED)         56956.7    239
-----------------------------------------------------------------------
All F-ratios are based on the residual mean square error.
```

soft corals was significantly higher in carbonate reef areas as opposed to granitic reef sites ($p<0.01$; Table 5; Fig. 8). On average, carbonate reef sites supported some 8% SOCC with granitic reef sites returning lower estimates in the 0.5 to 1% cover range.

However, the analyses also identified the existence of a significant interaction between the two factors Region and Habitat. The interaction plot (Fig. 9) clearly shows that the identified regional differences in soft coral abundance are most pronounced at carbonate reef sites whilst differences between the two habitat types are at their strongest in the western region.

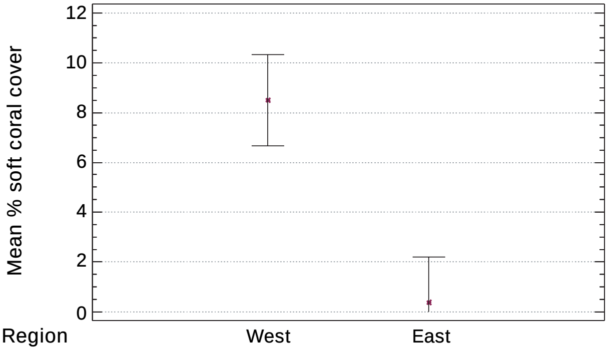

Figure 7. Graph showing significant *Region-effect* on the abundance (mean % cover + LSD) of *soft corals (SOCC)*.

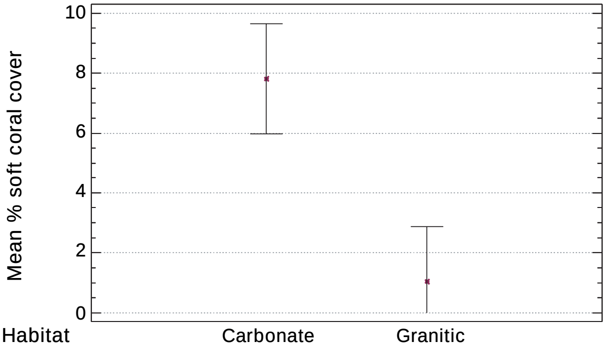

Figure 8. Graph showing significant *Habitat-effect* on the abundance (mean % cover + LSD) of *soft corals (SOCC)*.

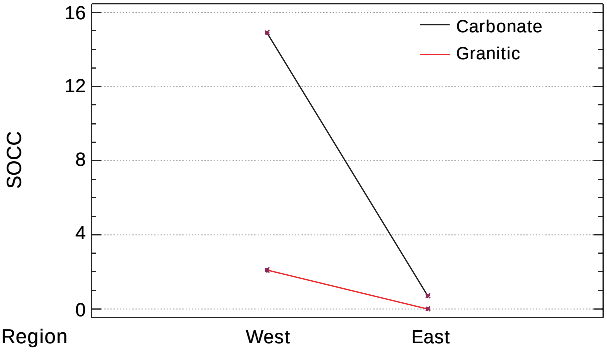

Figure 9. Graph showing significant *Region x Habitat interaction* affecting the abundance (mean % cover + LSD) of *soft corals (SOCC)*.

Estimates of Scleractinian Hard Coral Diversity

Surveys of scleractinian hard coral diversity conducted on Seychelles' reefs over past decades have identified a total of 67 individual coral genera belonging to some 16 different coral families (Veron, 2000). The 2002 survey of reef areas in the inner granitic islands of the Seychelles recorded a total of 44 individual genera of corals belonging to some 14 different families. These latest results suggest that some 67% of the generic-level and 88% of the family-level biodiversity of hard corals is currently represented in the region. Compared to the November 2000 survey the latest surveys recorded an additional 10 coral genera as well as 1 additional coral family. Whilst it is possible that these results do represent a real increase in regional hard coral biodiversity, the results may be

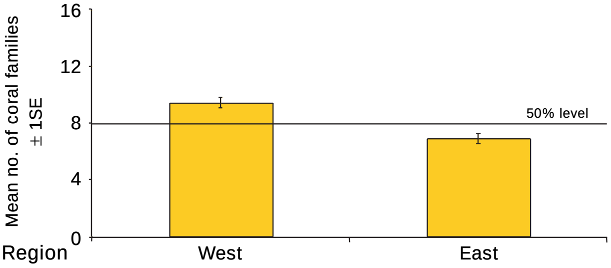

Figure 10. Bar graph showing statistically significant differences in the mean number of scleractinian hard coral *FAMILIES* recorded across the two regions surveyed in January and February 2002.

partly confounded due to the significant increase in sampling effort resulting from the expansion of the study to cover some 40 individual reef sites as opposed to the 22 sites surveyed previously. A number of the additional records obtained this year may relate to coral types that could have been present in 2000 but remained undetected due their potentially limited abundance and typically patchy distribution. A complete overview of the scleractinian hard coral families and genera recorded in early 2002 is provided in Tables 6a & 6b on the following pages.

Family-Level Coral Biodiversity

Hard coral biodiversity at the Family-level was found to differ significantly between the two regions with the average number of hard coral families per site being higher in the western region of the study area. The mean number of coral families present across the two regions sampled was calculated at 9.4 ± 0.4 (mean ± 1SE) for the western region and 6.9 ± 0.4 for the eastern region (Fig. 10). Only 3 (15%) of the 20 sites surveyed in this region recorded Family-level coral biodiversity at below the 50%-level. In contrast, 15 (75%) of 20 sites surveyed in the eastern region registered less than 50% of the total number of coral families known to occur in the area. The highest diversity of scleractinian coral families at any one site was recorded at White House Rocks (Site 14) in Baie Beau Vallon and Anse Petit Cour (Site 26) at Praslin. Both these sites recorded some 12 (75%) individual families of hard corals (Tables 6a & 6b). Members of the Family Pocilloporidae were the most abundant and widely distributed corals across both regions in early 2002 (Table 6a & b) with members of the Families Poritidae, Faviidae and Agaricidae also relatively widespread.

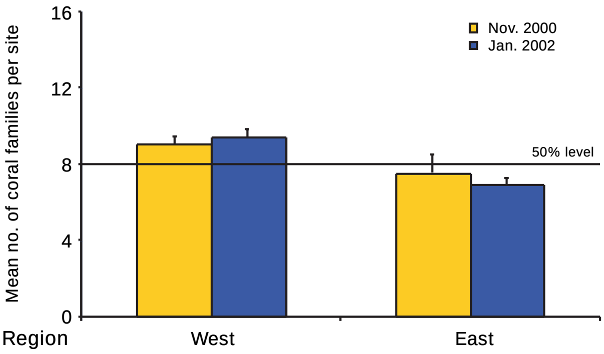

Figure 11. Bar graph showing a combined regional and inter-annual comparison of the mean number of scleractinian hard coral *families* recorded in both November 2000 and January–February 2002. Numbers above individual bars show the total number of sites surveyed for each combination of region and sampling period. Error bars show SE.

A preliminary comparison of Family-level coral biodiversity as recorded in November 2000 and January–February 2002 suggests that the mean number of coral families across the two regions has remained virtually unchanged over this time (Fig. 11).

Genus-Level Coral Diversity

Hard coral biodiversity at the Genus-level was also found to differ markedly between the two regions with the average number of hard coral genera per site again being higher in the western region of the study area. The mean number of coral genera present per site was calculated at 20.0 ± 0.4 (mean ± 1SE) for the western region and 13.9 ± 0.4 for the eastern region (Fig. 12 on page 223). Every one of the 40 sites surveyed recorded Genus-level coral diversity below the 50%-level (Tables 6a & 6b). The highest diversity of scleractinian coral genera in the western region was recorded at Auberge Reef (Site No. 15), whilst in the eastern region Anse Petit Cour at Praslin (Site No. 26) again recorded the highest local coral diversity. Respectively, these two sites recorded some 27 (40%) and 25 (37%) individual genera of hard corals.

A between-region comparison of the relative abundance indeces for the 11 most genera of scleractinian hard corals shows an appreciable degree of similarity in the indeces for the two regions. The two most common coral genera, the Genus *Pocillopora* and Genus *Porites*, both record an abundance ranking of ‘moderate’ which

Table 6a. Overview of scleractinian hard coral diversity recorded at 20 reef sites surveyed in the western region in January/February 2002

Type	Site No. 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
ACROPORIDAE																				
Montipora	R	R	R	R	–	R	R	–	–	–	R	R	–	R	R	R	–	–	–	L
Acropora	R	R	L	R	R	R	R	R	R	R	–	R	R	R	R	R	R	R	R	L
Astreopora	R	R	R	R	–	–	–	–	R	–	R	–	R	–	–	–	–	–	–	–
ASTROCOENIIDAE																				
Stylocoeniella	–	R	–	–	–	R	–	–	–	–	–	–	–	R	–	R	R	–	–	–
POCILLOPORIDAE																				
Pocillopora	L	R	M	M	L	M	M	M	V	L	R	M	M	V	L	L	L	M	L	L
Seriatopora	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–	–
Stylophora	R	R	–	–	R	–	–	–	–	R	–	–	–	–	R	R	R	R	–	–
EUPHYLLIDAE																				
Physogyra	–	–	–	R	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–	–
OCULINIDAE																				
Galaxea	R	R	R	R	R	R	–	R	–	R	–	–	–	R	R	R	–	–	–	–
SIDERASTREIDAE																				
Pseudosiderastrea	–	R	R	–	–	–	R	R	–	–	R	R	–	–	R	R	–	R	R	R
Siderastrea	R	–	R	R	R	R	R	L	R	R	R	R	R	R	R	R	R	R	–	–
Psammocora	–	–	–	–	R	L	–	R	–	L	–	–	–	–	R	R	–	–	–	–
AGARICIIDAE																				
Pavona	L	R	L	L	L	M	L	L	L	L	L	L	L	L	R	–	R	R	–	R
Leptoseris	R	L	L	L	L	M	R	L	L	L	R	R	R	L	L	L	R	R	L	R
Coeloseris	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	R	–	–	–	R
Gardineroseris	R	–	R	R	–	R	–	R	R	R	–	R	R	R	R	–	R	–	R	R
Pachyseris	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
FUNGIIDAE																				
Cycloseris	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–
Diaseris	–	–	–	–	R	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–
Fungia	–	L	–	R	L	L	–	R	–	L	–	L	–	R	L	M	L	–	L	–
Herpolitha	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R
Podabacia	–	–	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–	–	–
PECTINIIDAE																				
Echinophyllia	–	–	–	–	–	R	–	–	–	–	–	R	–	R	–	R	–	–	–	–
MERULINIDAE																				
Hydnophora	R	–	R	–	–	–	R	R	R	–	–	R	–	–	–	–	–	–	–	–
DENDROPHYLLIIDAE																				
Turbinaria	R	–	R	R	–	–	R	R	R	R	R	R	R	R	R	R	–	–	–	R
MUSSIDAE																				
Blastomussa	–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–
Acanthastrea	R	–	R	R	–	–	R	R	R	–	R	R	R	R	–	–	–	L	–	R
Lobophyllia	–	R	L	–	–	R	–	R	R	R	–	R	–	R	R	–	–	–	–	–

Table 6a. (continued)

Type	Site No. 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
FAVIIDAE																				
Favia	M	R	L	L	R	R	L	R	M	–	R	L	R	R	R	R	–	L	–	R
Favites	M	R	L	R	–	R	M	L	M	R	L	L	L	L	R	R	R	M	R	L
Goniastrea	M	R	L	L	R	R	R	R	R	R	R	L	R	R	L	R	R	R	–	–
Platygyra	–	–	R	R	–	R	R	–	R	R	–	R	R	–	R	–	–	R	–	R
Oulophyllia	–	–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–
Leptoria	–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–
Montastrea	L	–	L	R	–	–	R	R	L	–	R	R	R	R	–	–	–	L	–	L
Plesiastrea	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–	–
Diploastrea	–	–	–	–	–	R	–	–	–	–	–	–	–	R	R	–	–	–	–	–
Leptastrea	L	R	L	L	R	R	M	L	M	R	R	L	L	L	R	R	–	M	–	R
Cyphastrea	R	–	–	R	–	–	–	R	–	R	R	R	–	R	L	R	–	R	R	–
Echinopora	R	R	–	R	–	R	R	R	R	–	R	–	R	R	R	–	–	R	–	–
PORITIDAE																				
Porites	M	M	L	L	R	L	H	M	V	L	L	M	H	V	M	L	R	L	R	R
Poritipora	–	–	–	–	–	R	–	R	–	–	–	–	–	–	–	–	–	–	–	–
Goniopora	L	L	L	R	R	L	R	R	R	L	R	L	R	R	L	R	–	–	–	–
Alveopora	–	–	–	–	R	–	–	–	–	R	R	–	–	–	–	–	–	–	–	–
No. of families	10	10	10	11	7	11	9	11	9	10	8	11	8	12	11	11	8	6	7	8
% of families (16)	*63%*	*63%*	*63%*	*69%*	*44%*	*69%*	*56%*	*69%*	*56%*	*63%*	*50%*	*69%*	*50%*	*75%*	*69%*	*69%*	*50%*	*38%*	*44%*	*50%*
No. of genera	22	18	21	23	15	23	21	25	20	21	19	24	18	25	27	22	12	17	9	17
% of genera (67)	*33%*	*27%*	*31%*	*34%*	*22%*	*34%*	*31%*	*37%*	*30%*	*31%*	*28%*	*36%*	*27%*	*37%*	*40%*	*33%*	*18%*	*25%*	*13%*	*25%*

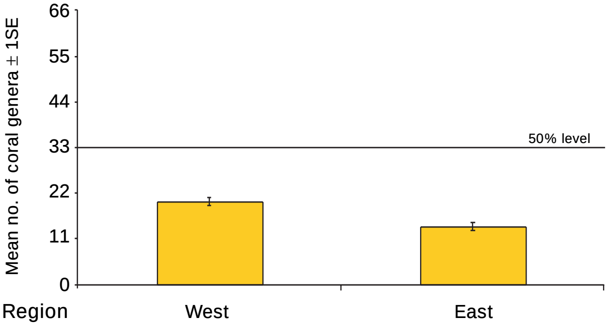

Figure 12. Bar graph showing statistically significant difference in the mean number of scleractinian coral *genera* recorded across the two regions surveyed in January and February 2002.

applies across the two regions sampled (Fig. 13 on next page). Similarly, the four genera identified as being the next most abundant and widespread coral types (Genera *Favites*, *Leptoseris*, *Pavona* and *Leptoria*) also show similar within-region patterns. These four genera all record relative abundance indeces in the 'low' category. However, 7 of the 11 most common hard coral genera recorded in early 2002 had higher mean abundance indeces in the western region of the survey area (Fig. 13).

A preliminary comparison of interannual trends in regional coral biodiversity at the Genus-level suggests that the western region may have experienced a recent increase in diversity, whilst reefs in the eastern region may have seen little or no change in coral diversity since the earlier surveys in November 2000 (Fig. 14 on page 226).

Table 6b. Overview of scleractinian hard coral diversity recorded at 20 reef sites surveyed in the eastern region in January/February 2002

Type	Site No. 21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
ACROPORIDAE																				
Montipora	R	–	L	R	R	M	L	R	–	R	–	L	R	–	–	R	–	–	–	–
Acropora	R	–	R	R	R	R	R	R	R	R	R	L	R	R	R	–	–	–	V	R
Astreopora	–	–	R	–	R	–	R	R	R	–	–	–	–	–	R	–	–	–	–	–
POCILLOPORIDAE																				
Pocillopora	L	R	L	M	L	L	L	L	M	L	M	L	L	M	M	L	L	L	L	L
Stylophora	–	–	–	R	–	–	–	–	–	–	R	–	–	R	–	–	–	–	–	–
EUPHYLLIDAE																				
Euphyllia	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Physogyra	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	R	–	–	–	–
OCULINIDAE																				
Galaxea	–	–	–	–	–	R	R	–	–	–	–	–	–	–	–	–	–	–	–	–
SIDERASTREIDAE																				
Siderastrea	–	–	–	–	–	R	–	R	R	–	–	–	–	–	–	R	–	–	–	–
Psammocora	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	R	–	–	–
Coscinaraea	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–
AGARICIIDAE																				
Pavona	R	R	R	R	R	L	R	R	R	M	R	R	R	R	R	H	R	R	R	R
Leptoseris	R	R	L	L	R	M	L	R	R	R	R	R	R	L	R	M	M	R	M	L
Coeloseris	–	–	–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–
Gardineroseris	R	R	–	R	R	R	R	–	–	–	–	–	–	–	–	R	R	–	R	R
FUNGIIDAE																				
Fungia	–	L	R	M	–	M	R	–	L	–	–	–	–	–	R	M	M	–	M	M
Herpolitha	–	–	–	R	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–
PECTINIIDAE																				
Echinophyllia	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
MERULINIDAE																				
Hydnophora	–	–	–	–	–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–
Merulina	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–	–
DENDROPHYLLIIDAE																				
Turbinaria	–	–	–	–	R	R	–	–	R	–	R	–	R	–	–	–	–	–	–	–
MUSSIDAE																				
Blastomussa	–	–	–	–	–	–	–	–	–	R	R	R	R	–	–	–	–	–	–	–
Acanthastrea	R	–	–	–	R	R	–	R	–	R	L	L	L	R	R	–	–	R	–	–
Lobophyllia	–	–	–	–	–	R	R	–	–	–	–	–	–	–	–	L	–	–	–	–
FAVIIDAE																				
Favia	R	–	–	R	L	R	–	R	R	L	L	R	R	R	R	R	–	–	R	–
Favites	M	R	R	R	L	L	R	L	L	L	M	H	M	L	L	–	R	R	R	R
Goniastrea	–	–	R	R	R	L	R	R	R	L	L	L	R	–	R	R	R	R	–	–
Montastrea	R	–	R	R	R	L	–	–	L	L	M	L	M	R	R	–	–	–	–	–

Table 6a. (continued)

Type	Site No.	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
Diploastrea		–	–	–	–	–	–	R	–	–	–	–	–	–	–	–	–	–	–	–	–
Leptastrea		R	R	R	R	R	R	R	R	M	M	L	R	R	L	R	R	R	R	–	–
Cyphastrea		R	–	R	–	R	R	R	–	R	R	R	–	R	–	–	–	–	–	–	R
Echinopora		–	R	–	–	R	R	M	–	R	R	R	–	–	–	–	–	–	–	–	–
PORITIDAE																					
Porites		R	R	L	M	L	M	M	M	M	M	L	M	L	M	L	R	M	L	L	L
Poritipora		–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	R
Goniopora		–	–	–	–	–	–	–	R	–	–	–	–	–	R	–	–	–	–	–	–
No. of families		7	5	6	6	7	12	8	7	8	6	8	6	7	6	7	9	6	5	6	6
% of families (16)		*44%*	*31%*	*38%*	*38%*	*44%*	*75%*	*50%*	*44%*	*50%*	*38%*	*50%*	*38%*	*44%*	*38%*	*44%*	*56%*	*38%*	*31%*	*38%*	*38%*
No. of genera		14	9	13	15	17	25	17	14	17	15	17	13	15	12	13	14	10	8	9	10
% of genera (67)		*21%*	*13%*	*19%*	*22%*	*25%*	*37%*	*25%*	*21%*	*25%*	*22%*	*25%*	*19%*	*22%*	*18%*	*19%*	*21%*	*15%*	*12%*	*13%*	*15%*

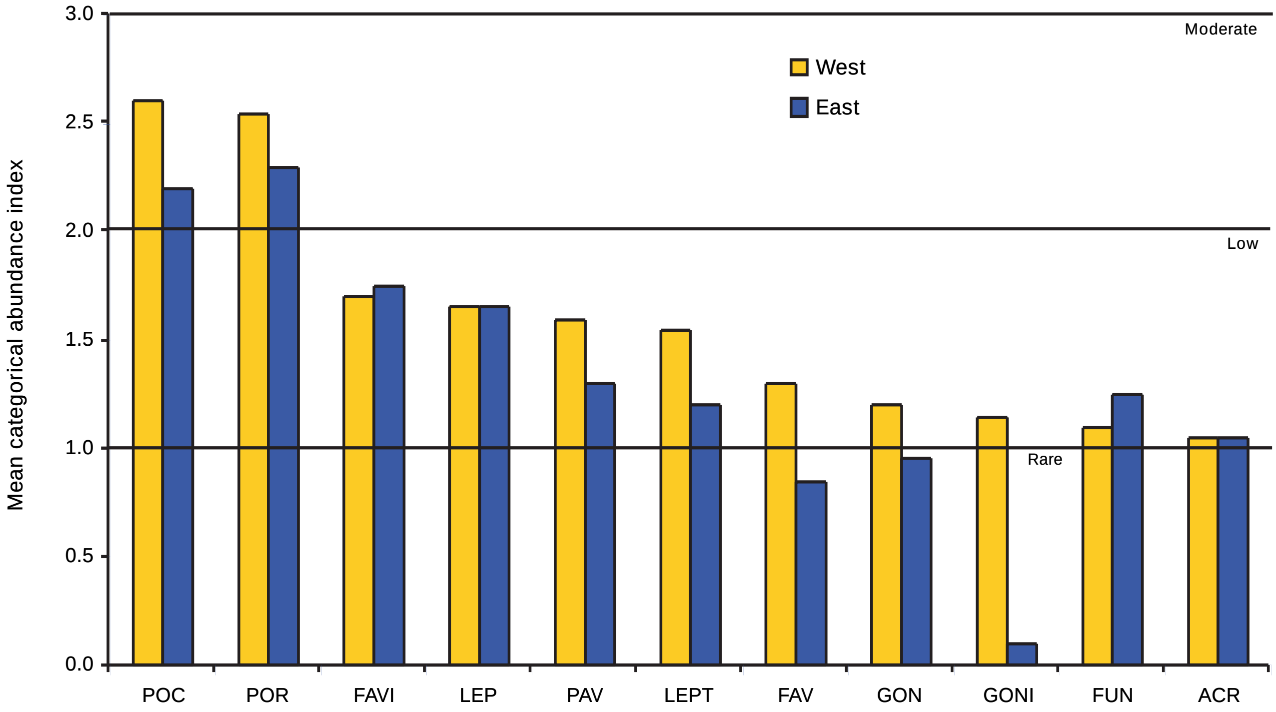

Figure 13. Bar graph showing a regional comparison of the calculated mean categorical abundance indeces for the 11 most abundant scleractinian coral GENERA recorded in early 2002 (where: POC = *Pocillopora* spp., POR = *Porites* spp., FAVI = *Favites* spp., LEP = *Leptoseris* spp., PAV = *Pavona* spp., LEPT = *Leptoria* spp., FAV = *Favia* spp., GON = *Goniastrea* spp., GONI = *Goniopora* spp., FUN = *Fungia* spp., and ACR = *Acropora* spp.)

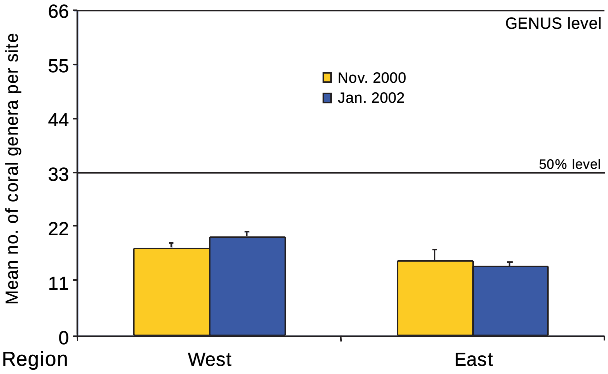

Figure 14. Bar graph showing a combined regional and inter-annual comparison of the mean number of scleractinian hard coral GENERA recorded in both November 2000 and January–February 2002. Numbers above individual bars show the actual number of sites surveyed for each combination of region and sampling period. Error bars show SE.

DISCUSSION

Scleractinian Hard Corals

Some 4 years after the 1998 mass coral bleaching event, the percent cover of reef-building or scleractinian hard corals on reefs around the inner granitic islands of the Seychelles remains at very low levels. Benthic surveys in January and February 2002 recorded a grand mean of just under 5% live hard coral cover across the 40 reef sites surveyed. However, the latest results do suggest a slow upward trend. In November 2000, mean percent cover of hard corals was recorded at only 3% across the 22 sites surveyed that year. Provided that no further large scale ecological disturbances will impact local reefs in the near future, there is a clear potential for a significant acceleration of the annual rate of increase in live hard coral cover.

A possible increase in the percent cover of scleractinian corals will occur largely as a result of the growth of recently settled hard coral recruits rather than through growth of the few remnant corals that survived the 1998 mass bleaching event. This emerging characteristic of the recovering coral communities is a likely reflection of the differential rates of bleaching-induced mortality experienced by the different functional components that comprised pre-bleaching coral communities. With few exceptions, remnant coral populations throughout the study area are dominated by members of relatively slow growing coral genera such the massive corals of the genera *Porites* and *Goniopora*. In contrast, virtually none of the previously dominant *Pocillopora* and *Acropora* corals survived. However, both of these fast-growing genera are key components of Indo-Pacific coral reef communities and are often responsible for the, at times, exponential rates of increase in percent live cover recorded some years following a major ecological disturbance (Wallace, 1999). If the recovery of coral communities around the inner granitic islands of Seychelles was to proceed as predicted, future surveys should describe a gradual shift in the relatively abundance of certain coral types away from slow growing massive and encrusting corals and back to a dominance of the previously prominent fast growing branching corals.

Overall, carbonate reef sites recorded significantly lower percent live hard coral cover (LHCC) values compared to granitic reef sites with 75% of carbonate reef sites recording mean LHCC values of less than 5%. In contrast, only 35% of granitic reef sites recorded percent cover estimates below the 5% mark. A majority of granitic reef sites did record percent coral cover values of between 5 and 10% with a further 10% of sites estimated to support between 10 and 15% live coral cover. As the great majority of corals recorded in both habitat types were recently recruited colonies, it appears highly likely that granitic reef sites will also show significantly higher rates of overall reef recovery in the immediate future.

It is noteworthy that the observed pattern of greater abundance of hard corals in granitic reef habitats is matched by a similar trend recently demonstrated for local populations of coral reef fishes (Engelhardt, 2001). A preliminary survey of selected species belonging to 14 individual families of coral reef fishes found that all 14 families of fishes were again significantly more abundant in granitic reef habitats (Engelhardt, 2001). The observed close correlation between habitat type and overall

Figure 15. Recent *Acropora* sp. recruit growing on granitic boulder.

Figure 16. Recent *Pocillopora* sp. recruits growing on granitic boulder.

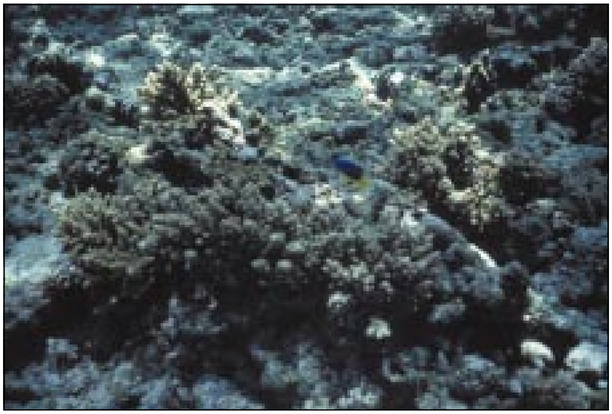

Figure 17. Regrowth of *Acropora* sp. at carbonate reef site in the eastern part of the study area.

Figure 18. Dead standing corals and coral rubble at carbonate reef site in November 2000 – some 2 1/2 years after the 1998 mass coral bleaching event.

abundance of two very important components of coral reef ecosystems – corals and fishes – warrants further investigation. Data obtained to date suggest that, over the past few years at least, granitic reef habitats in the Seychelles have apparently provided significantly more favourable environmental conditions that have facilitated reef recovery. The observed difference in the apparent recovery potential of the two habitat types has significant implications for the management of local coral reefs. It would appear that carbonate reefs throughout much of the survey area are in greater need of protective as well as mitigative management measures aimed at improving ecosystem functionality.

Soft Corals (*Sarcophyton* & *Sinularia* spp.)

Soft coral cover too has remained at very low levels over recent years with grand mean percent cover across the 40 reef sites surveyed in early 2002 estimated at around 4.5%. Similar to the trends observed for hard corals, soft coral abundance data collected since 1999 are also suggesting the possibility of a recent slight increase in the percent cover of *Sarcophyton* and *Sinularia* spp. throughout the study area.

The mean percent cover of soft corals (SOCC) was found to be significantly higher in the western region of the study area where SOCC was estimated to average around 9% across the 20 sites surveyed. In contrast, soft corals were virtually absent from the eastern region with a mean cover estimate of less than 1% being recorded. The observed pattern of higher soft coral abundance in the western region warrants further attention. Most members of the target genera *Sarcophyton* and *Sinularia* are known to have considerable tolerances to elevated levels of both sedimentation and nutrients (Rogers, 1990). These tolerances often result in a numeric dominance of these genera in areas affected by a high nutrient status and/or elevated levels of sedimentation.

The notion that the observed patterns of soft coral abundance may be closely linked to nutrient and sediment dynamics is further supported by the fact that the mean % cover of soft corals was found to be significantly higher in carbonate reef areas as opposed to granitic reef sites. On average, carbonate reef sites supported some 8% SOCC with granitic reef sites returning considerably lower estimates in the 0.5 to 1% cover range. A majority of the carbonate reef sites surveyed as part of this study are located in relatively protected bays and close to human habitation. Water circulation patterns within, as well as rates of water exchange from the open ocean into these bays, may differ markedly from those typically encountered at the more exposed granitic reef sites. The reduced rate of flushing and general water exchange found at most carbonate reef sites may result in a more frequent entrapment of dissolved particles as well as polluted waters. The greater abundance of soft corals in carbonate reef areas may be a reflection of the existence of these types of environmental gradients. It follows that hard coral recovery in these areas may remain significantly depressed for as long as nutrient pollution and sediment stress persist. Consequently, our results suggest the possibility of significant differences in water quality and sediment characteristics at carbonate reef sites in the two regions surveyed with parts of the western region possibly subject to persistent elevated levels of some of these parameters.

The greater abundance of soft corals in the western region may also explain a previously documented pattern of distribution for one of the major families of coral reef fishes. The reef fish pilot survey conducted as part of the GEF Coral Reef Study in July 2001 showed that the abundance of large angelfishes (Family Pomacanthidae) was significantly higher in the western region of the inner granitic islands (Engelhardt, 2001). Clearly, the abundance of pomacanthid fishes appears to be strongly correlated with the abundance of soft corals in the genera *Sarcophyton* and *Sinularia* who comprise a significant part of the diet of many tropical angelfishes, including members of the genus *Pomacanthus*.

Scleractinian Hard Coral Diversity

Surveys of scleractinian hard coral diversity conducted on Seychelles' reefs in recent times have identified a combined total of 66 individual coral genera belonging to some 16 different families of corals (Veron, 2000). The first comprehensive scientific survey of Seychelles' corals conducted in the early 1980's recorded some 47 individual scleractinian genera (Stoddart, 1984). More recently, a geographically more localised survey of fringing reefs around the northern parts of Mahé Island recorded some 31 different hard coral genera (TRMU, 1996). The 2002 survey of reefs throughout the inner granitic islands of the Seychelles recorded a total of 44 individual genera of corals belonging to some 14 different families. These latest results suggest that some 67% of the generic-level and 88% of the family-level biodiversity of hard corals is currently represented in the region. Compared to the November 2000 survey the latest surveys recorded an additional 10 scleractinian coral genera as well as 1 additional coral family.

Hard coral diversity at the Family-level was found to differ significantly between the two regions with the average number of hard coral families per site being higher in the western region of the study area. The mean number of coral families present across the two regions sampled was calculated at 9.4 ± 0.4 (mean ± 1SE) for the western region compared to 6.9 ± 0.4 for the eastern region. Members of the Family Pocilloporidae were the most abundant and widely distributed corals across both regions with members of the Families Poritidae, Faviidae and Agaricidae also relatively widespread. However, the present rarity of many of the coral families recorded highlights the fact that coral communities throughout the region are only in the early stages of recovery with many coral families still highly vulnerable to ecological disturbances that could, at least in the short term, lead to local-scale extinctions. Clearly, it remains very important to attempt to minimise any additional, unnatural disturbances to reef sites throughout the study area.

Scleractinian coral diversity at the Genus-level was also found to differ significantly between the two regions with the average number of hard coral genera per site again being higher in the western region of the study area. The mean number of coral genera present across the two regions sampled was calculated at 20.0 ± 0.4 (mean ± 1SE) for the western region and 13.9 ± 0.4 for the eastern region. All of the 40 sites surveyed across the two regions recorded Genus-level coral biodiversity at below the 50%-level. A between-region comparison of the relative abundance indeces for the 10 most genera of scleractinian hard corals shows an appreciable degree of similarity in the indeces for the two regions. The two most common coral genera, the Genus *Pocillopora* and Genus *Porites*, both record an abundance ranking of ‘moderate’ which applies across the two regions sampled. Similarly, the four genera identified as being the next most abundant and widespread coral types (Genera *Favites*, *Leptoseris*, *Pavona* and *Leptoria*) also show similar within-region patterns. These four genera all record relative abundance indeces in the ‘low’ category. However, 7 of the 11 most common hard coral genera recorded in early 2002 had higher mean abundance indeces in the western region of the survey area. A preliminary comparison of interannual trends in regional coral biodiversity at the Genus-level suggests that the western region may have experienced a recent increase in diversity, whilst reefs in the eastern region may have seen little or no change in coral diversity since the earlier surveys in November 2000.

Given the lack of high resolution coral survey data predating the 1998 mass coral bleaching event it remains unclear whether or not current regional differences in coral biodiversity already existed prior to the event or, alternatively, are the result of major differences in the supply of coral larvae to these regions. It would, however, appear unlikely that differential rates of bleaching-related mortality could have given rise to the observed pattern. Rather, the data are suggesting that the recent supply of coral larvae from a diversity of coral families has been greater to the western region of the study area. Should the observed pattern of supply and recruitment of scleractinian hard corals persist over time, coral communities in the western region will almost certainly develop into more complex and diverse communities compared to their counterparts in the east. Developing between-region differences in scleractinian coral diversity would also be likely to affect the composition of other reef-associated communities including coral reef fishes. Future surveys as part of the GEF Coral Reef Study will focus on providing additional high resolution data on the ecological characteristics of these developing communities, providing a critically important baseline for the development of a comprehensive coral reef management plan that aims to fully protected coral reef biodiversity throughout the study area. The current distribution of scleractinian coral families and genera shows a high degree of spatial patchiness giving rise to highly variable between as well as within-region patterns of coral distribution. This degree of spatial variability has important implications for future reef management measures aimed at protecting and promoting scleractinian coral diversity around the inner granitic islands of the Seychelles. Such aims may only feasibly be achieved with an extensive network of suitably protected marine areas that are actively managed to reduce the potential impacts of factors known to reduce coral settlement, recruitment and subsequent growth and reproduction. Such a network of protected areas needs to be developed on the basis of the emerging patterns of hard coral distribution and abundance with the aim of incorporating as much of the local coral diversity as possible.

CONCLUSIONS

Whilst the reefs of the inner granitic islands of the Seychelles have suffered extensive and system-wide degradation in recent years there is also a clear potential for significant reef recovery to occur in the near- to mid-term. Coral recovery around the inner granitic islands of the Seychelles remains in the early stages of succession with hard coral abundance and diversity well below pre-1998 levels. However, there are some positive trends emerging with mean % live coral cover and scleractinian coral diversity showing small but significant increases over the past 15 or months. However, the rate at which local coral communities will recover will be largely dependent on the environmental conditions encountered by newly settled coral recruits (Wallace *et al.*, 1986; Harriott & Fisk, 1989). Targeted, pro-active reef management measures aimed at maximising recruitment success will play a critical role in determining the direction and rate at which this recovery is likely to proceed.

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Aldabra: Monitoring the Path to Recovery

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Key words: coral, fish, monitoring, recruitment, Seychelles

ABSTRACT

In February 2001 the Aldabra Marine Programme completed the second annual survey in the long term monitoring of the corals and reef fishes of Aldabra Atoll, southern Seychelles. Eight outer reef sites and three lagoonal sites were surveyed, and a coral recruitment study was initiated at all sites.

The coral community did not change significantly between November 1999 and February 2001, but there were some signs of recovery. Coral recruitment was high with most recruits settling at approximately 10m depth. Fish survey results showed a sharp decline in the fish population during this period, but this difference is possibly due to surveys taking place at different times of year, or to schools of small fish drastically increasing the count numbers.

More recent surveys conducted in February 2002 suggest that the fish population at Aldabra is healthy and that coral cover is increasing. During these surveys the AMP established a 12th survey site at Aldabra and increased the scope of the study to three islands east of Aldabra at Assomption, Astove and St. Pierre (for reports see www.aldabra.org).

INTRODUCTION

The devastating effects of the 1997–98 ENSO (El Niño Southern Oscillation) have now been well documented (eg. Spencer *et al.*, 2000; Wilkinson, 1998, 2000). In the western Indian Ocean, ENSO-associated sea temperature rises led to extensive and pervasive coral bleaching in this region, including the southern Seychelles (Teleki *et al.*, 1998). In April 1998, at the peak of the coral bleaching event, a Cambridge Coastal Research Unit (CCRU) research team, the Southern Seychelles Atoll Research Programme (SSARP), found that widespread bleaching and mortality was common on the outer reef slopes (3–25 m) surveyed from the western to northeastern sides of Aldabra Atoll. Coral coverage was 37%, with 41% of those bleached or recently dead. The intensity of the coral bleaching in Aldabra was not as high as in other areas of the southern Seychelles because peak warming was 0.5° C lower (Spencer *et al.*, 2000). The CCRU expedition to Aldabra also conducted surveys of fish communities on the reefs at four locations along the western and northeastern shorelines and recorded 287 species from 35 families (Spalding, 1998; University of Cambridge, unpublished).

The Aldabra Marine Programme (AMP) was established following the discovery of the extensive damage to the coral reef ecosystem at Aldabra Atoll, and to address the growing concern throughout the coral reef scientific community that reefs worldwide were being altered at increasing and unprecedented levels. The primary goal of AMP was to establish the first permanent underwater survey sites for long-term, quantitative studies at Aldabra. These would determine the ability of coral reef systems to replenish lost coral populations, reinstate framework growth, and to recover reef habitat structural complexity and its fish communities. The degradation of coral reef communities and their subsequent rates of recovery are important early-warning indicators of local and global marine ecosystem health. The need for long-term monitoring of coral reefs in a relatively natural state, to establish benchmarks for measuring changes and recovery in impacted reef systems, has been recognised (Wilkinson & Buddemeier, 1994), but there are few anthropogenically unaltered sites to study (Roberts *et al.*, 1999; Steneck, 1998). Determining the recovery dynamics of coral reefs altered by natural disturbances is a long-term process. It can be complicated, prolonged, or rendered impossible, in systems already stressed by anthropogenic sources. Long term studies at Aldabra will provide benchmarks in the recovery process free from these disturbances, against which anthropogenic impacts in other coral reef systems may be assessed, and rates of recovery evaluated.

The AMP was also formed to provide information for the development of a marine conservation programme for Aldabra to enhance the protection and preservation of the atoll's marine resources (Teleki *et al.*, 2000b). AMP studies are conducted to fulfil the marine management priorities of the Seychelles Islands (as outlined in Beaver & Gerlach, 1997). The research findings at Aldabra are disseminated to national, regional, and global initiatives. The AMP studies also contribute to a growing and valuable information database on global climate change and the environment.

Aldabra Atoll: A Natural Science Laboratory

Aldabra Atoll (9°24' S, 46°20' E), Southern Seychelles Islands Group, is one of the world's largest raised coral atolls (34 km long, maximum 14.5 km wide, area 155 km^2). It is located 1150 km southwest of the capital of the Republic of Seychelles, Mahé, and 420 km north of Madagascar (Fig. 1 and 2). Late Quaternary raised reef limestones, averaging two km in width and up to 8 m

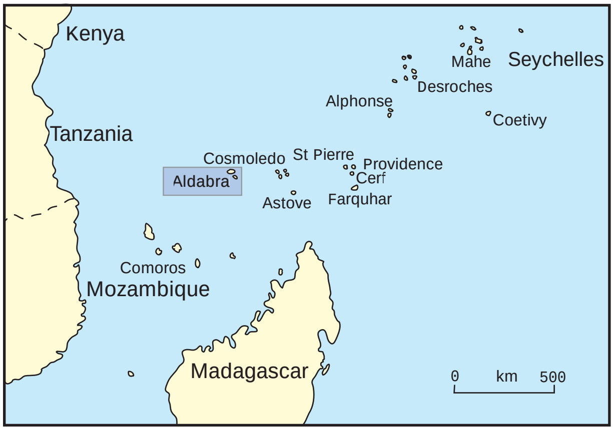

Figure 1. Location of Aldabra Atoll in the Western Indian Ocean.

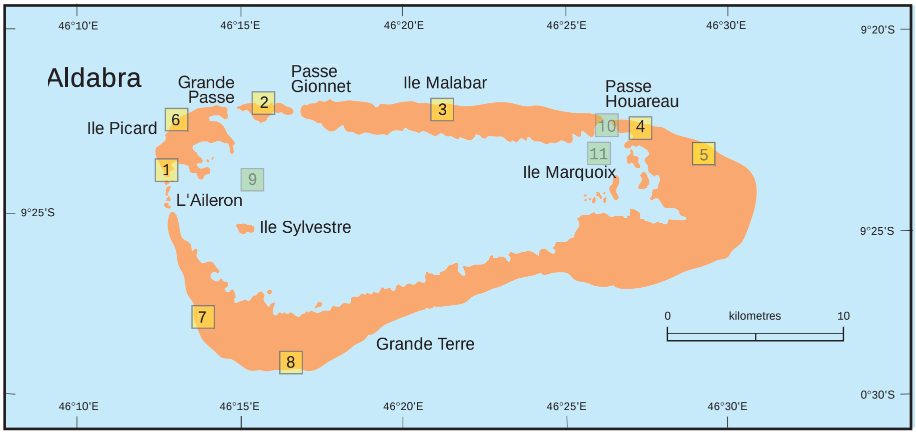

Figure 2. Aldabra Atoll with the 11 permanent Aldabra Marine Programme monitoring sites for coral and reef fish studies on the outer reef and within the lagoon.

above sea level, surround a shallow, central lagoon (190 km^2). The lagoon is on average 2–3 m deep at low tide. The coastline consists mainly of deeply undercut limestone cliffs and a broad intertidal reef flat (200–500 m). The lagoon is linked to the ocean by two major and one smaller channel and by several smaller reef passages (Stoddart, 1984). The tidal range is 2–3 m and results in large-scale hydrodynamic exchanges between the lagoon and the ocean through the channels. The main channel alone drains approximately 60% of the lagoon (Stoddart, 1971).

Aldabra Atoll has been characterised as 'one of the wonders of the world' and 'one of the world's greatest surviving natural treasures' (Attenborough, 1995). The isolation of Aldabra in a remote area of the Indian Ocean has helped preserve this large atoll in a relatively natural state. Aldabra Atoll was designated a UNESCO World Heritage Site in 1982 in recognition of the atoll's environmental importance. A recent global study by Conservation International has included Aldabra in one of the areas identified as a *marine biodiversity hotspot* (Roberts *et al.*, in press). High endemism and species richness are used to identify priority sites for marine conservation worldwide. The Conservation International study demonstrates the present critical importance of the marine environments of Aldabra for ecological studies that have global significance. Although increasing levels of stress from human activities are contributing to the decline of the world's coral reefs (Bryant *et al.*, 1998; Hodgson, 1999; Wilkinson, 2000) the marine environment of Aldabra with its coastal reefs and expansive lagoon has remained untainted. Aldabra is surrounded by a region which has a number of coral reef systems at high risk from activities ranging from coastal development and destructive fishing practices, to the overexploitation of resources, marine pollution, and runoff from inland deforestation and farming (Bryant *et al.*, 1998). Aldabra is an ideal natural laboratory for studying tropical marine ecosystems and related environments (i.e. seagrasses and mangroves).

The 1999 Aldabra Marine Programme Phase I

Phase I of the Aldabra Marine Programme took place in November 1999 (Teleki *et al.*, 1999, 2000a). Seven permanent survey sites were established on the outer reefs along the northern and western coasts of Aldabra. The survey sites were located to give an even geographic distribution around the atoll, and to coincide with previous transects by Barnes *et al.* (1971) and Drew (1977). At each survey site, permanent 50 m transects were located along the 10 m and 20 m depth contours, with the exception of one site where the transects were located at the 5 m and 15 m contours due to limited coral cover at depth. Surveys on each transect assessed the diversity and abundance of corals and fishes, and quantified the impacts of the 1998 bleaching event. During this first phase the corals on the Drew (1977) transect were resurveyed and it was concluded that the corals were beginning to recover.

In 1999, it was concluded that there had been high coral mortality in water shallower than 5 m, and that coral vitality increased with depth. Live coral coverage ranged from 3–21% in shallow water (0–10 m) and 0.5–34% in deep water (>10 m). Most of the live branching corals were dominated by *Pocillopora* spp. and *Porites* spp., which represented 20% of the coral cover. Massive species formed 63% of total live coral coverage on deep transects, compared to 45% on shallow transects. *Physogyra* spp. formed 65% of the live, massive coral coverage on the deep transects, but only 3% on the shallow transects. The foliose *Echinopora* spp., *Pachyseris* spp., and *Turbinaria* spp., and the encrusting *Montipora* spp., were common live corals at all sites, but the foliose genera were almost exclusively found on the deep transects. Plate and fine branching corals were not found at any of the locations surveyed, probably due to the relatively high hydrodynamic conditions at Aldabra. From west to east along the coastline of Aldabra there was a prominent gradient of decreased coral growth, related to levels of increasing hydrodynamic energy. Overall, comparisons with coral coverage data collected in 1998 revealed that in shallow depths bleaching primarily led to mortality, while in deeper water bleached corals were more likely to recover.

The quantified fish surveys on the outer reefs at Aldabra found 165 species representing 27 families, and the qualitative (off-transect) surveys identified an additional 46 species and 6 families. The total of 211 species from 32 families found in November 1999 were generally in agreement with the 287 species from 35 families (Spalding, 1998; University of Cambridge, unpublished) reported by the CCRU expedition to Aldabra in April 1998.

The densities of fishes on the transects were often dominated by large schools of a few species from the families Serranidae (groupers and basslets), Apogonidae (cardinalfishes), Pomacentridae (damselfishes), and Caesionidae (fusiliers). The abundance of fishes in the smallest size category was caused primarily by large numbers of small sized species, and secondarily by juvenile life stages. Although there was a prominent gradient of decreased coral growth from west to east along the coast line of Aldabra, there was no correlation between either the number of species, or the density of fishes, and the corresponding west to east locations of the transects. However, there were positive correlations between the amount of live coral habitat, and the number of species, and the density of Chaetodontidae (butterflyfishes), Labridae (wrasses), and Serranidae. These families each have several species that are commonly associated with live corals and habitat structure formed by erect dead corals (Crosby & Reese, 1996).

METHODOLOGY

Phase II of the Aldabra Marine Programme took place between 8th and 22nd February 2001. During this phase 4, additional sites were established at Aldabra, and the 7 permanent monitoring sites established in 1999 were resurveyed. New sites consisted of one on the outer reef (Site 8), and three in the lagoon (Sites 9–11; Fig. 2).

Coral and Fish Transects

The procedures used in the 2001 AMP survey to locate and mark permanent survey sites, and the methods used to quantify corals and fishes on the transects, replicated those used in AMP Phase I (Teleki *et al.*, 1999, 2000a).

The new outer reef site was situated on the exposed southern coast of Aldabra, and completed the spread of sites around the Atoll. At this site, two 50 m transects were laid along the 10 m and 20 m depth contours. Permanent survey sites in the lagoon were established in areas where there was abundant live coral habitat. At the lagoon sites a single 50 m transect extended along a depth contour appropriate to the habitat. All 50 m transects were permanently marked with steel stakes at the beginning and end, and each stake location was fixed using a Global Positioning System (GPS). Prior to each coral and fish survey, a tape or line was placed along the transects between the two stakes, following the appropriate depth contour. Digital underwater videography was used to record the benthic habitat on both sides of the 50 m transect lines. The imagery from the secondary transects was later analysed using the AIMS 5-dot method (Osborne & Oxley, 1997). Below 20 m, reference footage was filmed to characterise the general habitat.

Fish transects were conducted using protocols developed for rapid visual assessments (Ginsberg *et al.*, 1998; English *et al.*, 1997). Two divers recorded the species, number, and sizes of fish in a 2 m corridor extending out from either side of the 50 m transect, and vertically to the surface. The sizes of the fish were recorded in six total length categories: 0–5 cm, 6–10 cm, 11–20 cm, 21–30 cm, 31–40 cm, and >40 cm. Transects were surveyed in 25 m sections, first for larger/conspicuous fishes, and then immediately re-surveyed for small/cryptic fishes. Only the first 25 m section of the transects at 20 m depth were surveyed. Thus, over 300 m^2 of benthic habitat was surveyed for fishes at each site. The fish transect data was analysed using Analysis of Variance (ANOVA) and regression statistical procedures (Zar, 1984).

Qualitative surveys were conducted for species of fish off the transects on the outer reef, and at the lagoon survey sites, to develop a more complete inventory of fishes at Aldabra Atoll. On the outer reef, these surveys were done visually after a transect survey was completed, and concentrated on locating species not seen on the transect. In the lagoon, digital videography was used to record fishes near the permanent survey sites, and fo-

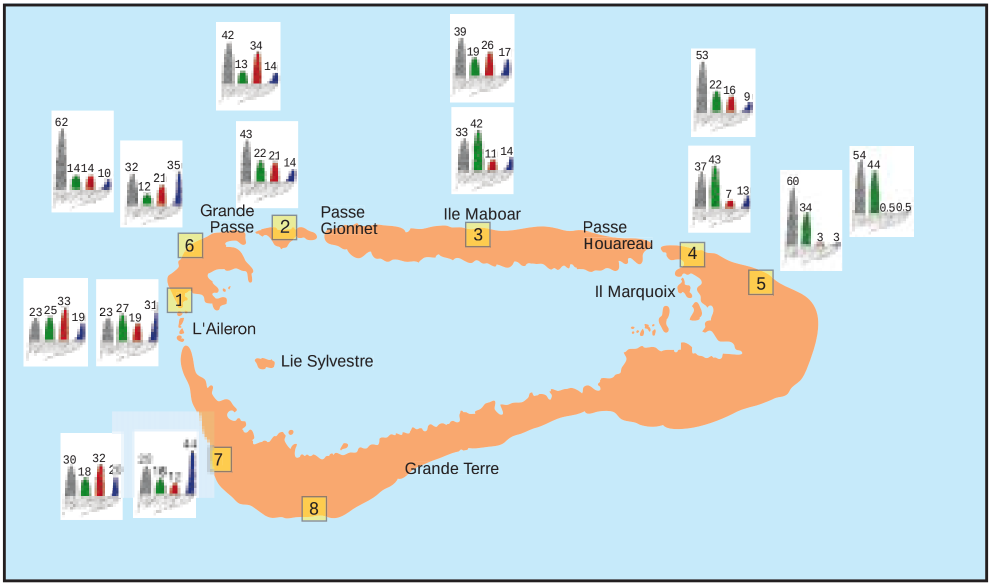

Figure 3. Calculated general substrate cover for outer reef shallow and deep transect sites (numbers in boxes) around Aldabra in November 1999. Shoreward coral graph = 10m depth, offshore = 20m. Coral branch length and numbers indicate Percent cover of category.
Colours represent: Grey: sand, rock, rubble; Green: algae; Red: live coral; Blue: dead coral.

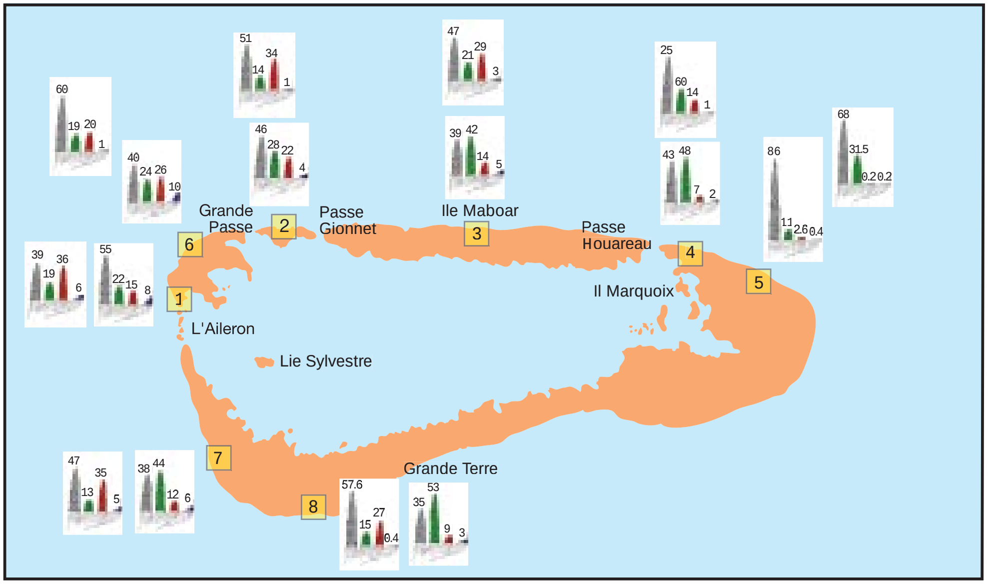

Figure 4. General substrate cover for outer reef shallow and deep transect sites (numbers in boxes) around Aldabra in February 2001. Shoreward coral graph = 10m depth, offshore = 20m. Coral branch length and numbers indicate percent cover of category.
Colours represent: Grey: sand, rock, rubble; Green: algae; Red: live coral; Blue: dead coral.

cussed on obtaining a video record of the many juvenile life-stages found in the shallow water.

Coral Recruitment and Tagging

In February 2001 coral recruitment was assessed at all permanent survey sites on the outer reef and in the lagoon for the first time, using counts of recruits in 1 m^2 quadrats, with up to nine randomly positioned 4 m x 1 m quadrats. At each site recruits were counted at 6 m, 10 m, and 20 m survey depths. A maximum width of 5 cm was used as the upper limit for defining recruits (Engelhardt, 2001). These criteria are based on *in-situ* coral growth rate observations of Wallace (1985), Wallace *et al.* (1986) and van Moorsel (1988). Fast growing *Acropora* corals have been shown to reach a maximum size, two years after settlement, of between 2 cm and 5 cm (Wallace, 1985). At lagoon sites, counts were only made at the depth of the single transect. The recruits were measured across their greatest width and the width at 90 degrees to this. Recruit size was calculated by averaging the two widths. Where possible, recruits were identified to genus level.

RESULTS

Coral Transects and Benthic Habitats

The video transect analysis for Sites 1–7 shows that the percentage of live coral cover around Aldabra has changed little between November 1999 and February 2001 (Fig. 3 and 4). There is a small increase (maximum 6%) in percent live coral cover except at Site 1 (10 m depth) and Site 4 (20 m) where cover decreased by 4% and 2%, respectively. Live coral cover ranged from 3–28% in shallow water (10 m) and 0.2–36% in deep water (20 m). Algal cover also remained generally unchanged, though there was also a tendency for small rises in percent cover. Sites 6 and 7 showed considerable increases in algal growth over the 15 month period (12 and 28%, respectively), possibly due to the large amount of dead coral substrate available for colonisation at these sites in 1999. Most algal cover at Site 7 is coralline algae, which are quick to colonise dead coral colonies.

The percentage of dead coral cover decreased considerably at all sites between November 1999 and February 2001. There was a corresponding increase in the amount of sand, rock and rubble due to the break-down of the dead coral, with the exception of Site 6 where the amount of sand, rock and rubble decreased by 2% in both shallow and deep water. Coral and algal growth over the 15 month period was greatest at Site 6, which accounts for the decrease in dead coral not leading to an increase in sand, rock and rubble.

The new Site 8 showed a unique benthic composition compared with the other sites. At 10 m there was no loose coral rubble, but a predominance of coralline algae and live coral cover was low. Other sites around Aldabra have an abundance of loose rubble in both shallow and deep water, Site 5 being completely dominated by rubble. Site 8 is highly exposed and therefore rubble would soon be washed into deeper water or along the coast. It is the high exposure of this location that made it impossible to set up a permanent survey site here in 1999, and the potential for adverse sea conditions may make re-surveying Site 8 difficult in the future. At 20 m depth the benthic habitat composition at Site 8 resembled other sites around the atoll.

The amount live coral cover in shallow water (5–10 m) around Aldabra in both 1999 and 2001 closely follows the pattern of exposure. Live coral is most abundant at Site 6 which is on the sheltered northwest coast, and progressively decreases moving around the atoll in both directions (Fig. 3 and 4). Algal cover tends to increase in more exposed areas. This trend is particularly evident from the February 2001 data, and accounts for the increase in coralline algae that favour the more exposed sites. There is also an increase in *Halimeda* spp. cover moving east along the north coast. However, *Halimeda* spp. was not common at Sites 7 and 8.

Of the three new sites established in the lagoon, two had a very high percentage of live coral (Fig. 5 on next page), Site 9 at Grand Passe (45%) and 10 at Passe Houareau (40%). These sites have strong currents, are

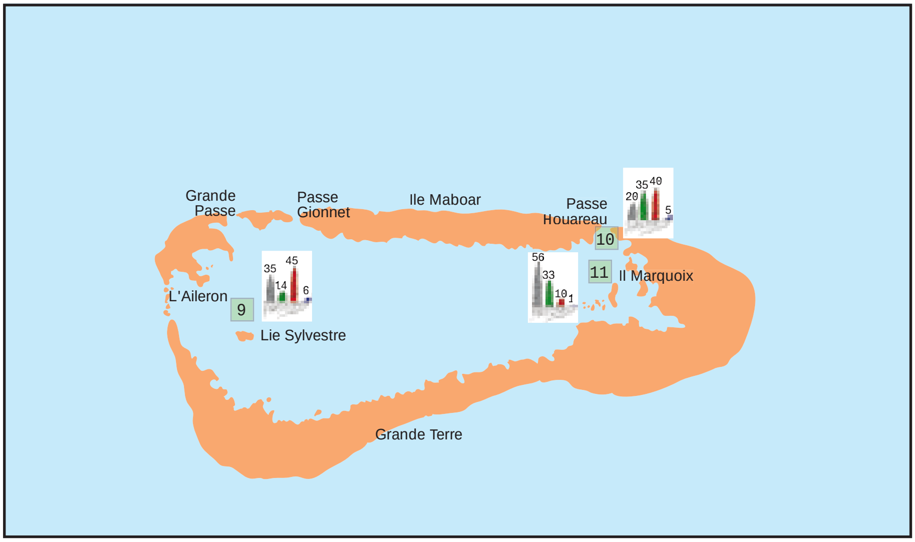

Figure 5. General substrate cover for lagoon transect sites (numbers in boxes) at Aldabra in February 2001. Coral branch length and numbers indicate percent cover of category.
Colours represent: Grey: sand, rock, rubble; Green: algae; Red: live coral; Blue: dead coral.

Table 1. Average numbers of recruits per metre squared for three depths at sites around Aldabra Atoll

Depth	Statistic	Site number										
		1	2	3	4	5	6	7	8	9	10	11
6m	Mean	3.5	3.8	7.9	8.1	8.7	3.6	7		7.8	3.9	10.6
	SE	0.6	0.4	0.7	0.8	0.8	05	0.8		1.0	0.5	1.2
	SD	3.5	2.4	4.1	4.4	5.1	2.3	4.3		6.6	2.9	7.6
	Max nº	15	11	17	22	20	9	23		25	12	46
	nr	119	137	278	259	313	91	216		314	160	425
	n	34	36	35	32	36	25	31		40	41	40
10m	Mean	4.6	5.6	8.7	8.6	5.6	2.2	8.6	12.6			
	SE	0.5	0.4	0.8	0.9	0.6	0.2	0.7	1.0			
	SD	2.7	2.5	5.1	5.0	3.4	1.3	4.3	5.6			
	Max nº	14	12	21	24	13	5	19	30			
	nr	156	169	329	293	184	71	277	377			
	n	34	30	38	34	33	32	32	30			
20m	Mean	5.8	3.1	6.3	4.1	0.7	1.7		5.7			
	SE	0.8	0.3	0.8	0.5	0.2	0.3		0.6			
	SD	4.4	1.7	4.5	2.7	1.1	1.3		3.6			
	Max n	19	6	22	9	4	4		13			
	nr	192	75	201	99	24	34		194			
	n	33	24	32	24	32	20		34			

Note that at Site 7 the recruit estimate below labelled 10m was made at 15m.
SE = standard error, SD = standard deviation, Max nº = maximum number of recruits per quadrat, *nr* = total number of recruits, *n* = number of quadrats.
*Note depth recording differences for Site 7.

rarely exposed at low tide, and are well flushed with water giving rise to good conditions for coral growth. Site 11 at Passe Houareau had poor coral live coral cover (10%), but this site is at the end of the channel and the corals are probably often exposed at low tide, experience temperature extremes and are regularly inundated by lagoonal sediment. Algal cover at these lagoon sites was similar to outer reef sites, with a predominance of coralline algae at Sites 10 and 11, and turf algae at Site 9. Levels of sand, rubble and rocks were low at Site 10 due to currents and the steep walls of the channel (Fig. 5).

During Phases I and II, 41 scleractinian coral genera and 57 species were recorded. A further 10 non-scleractinian genera were noted, and the true figure is expected to be considerably higher.

Coral Recruitment and Tagging

NUMBER OF RECRUITS

The average number of coral recruits per m^2 at each of the sampling sites is given in Table 1. The mean number of recruits per m^2 for all sites varied with depth from 6 recruits at 6 m depth (S.E. ± 0.32, standard deviation = 4.4, n° recruits = 1197, n = 198 quadrats), to 7 recruits at 10 m depth (S.E. ± 0.33, standard deviation = 5, n° recruits = 1582, n = 231), and 4 recruits at 20 m depth (S.E. ± 0.27, standard deviation = 3.8, n° recruits = 823, n = 199). Coral recruitment was patchy, and the average figures for individual sites varied from 1 recruit per m^2 at Site 5 (20 m depth), to 13 per m^2 at Site 8 (10 m depth; Table 1). The maximum number of recruits recorded in a single metre squared was 46 at Site 11 in the lagoon.

Coral recruit size-frequencies varied between genera. There was some indication that *Acropora* species may have had two recruitment cohorts in the last few years, as suggested by the two peaks in the size distribution histogram (Fig. 6). *Pocillopora* and *Favia* had bell shaped distributions suggesting a single cohort, though the *Pocillopora* distribution has a positive skew, which may indicate a second older cohort. Other genera, such as *Pavona* (n = 235) and *Psammocora* (n = 367), showed normal distributions, while *Acanthastrea* (n = 99) showed a strong negative skew, again suggesting two cohorts.

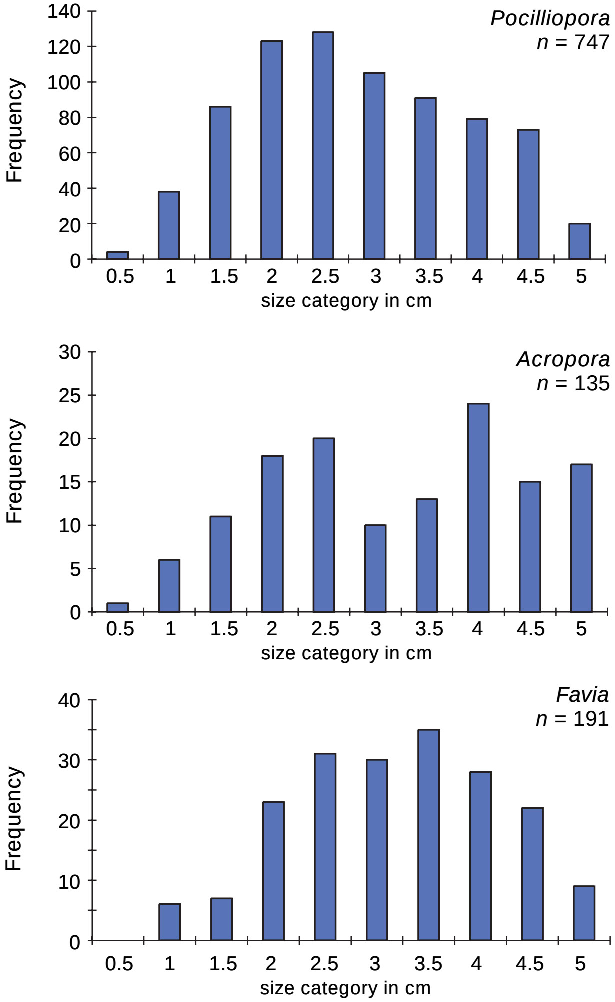

Figure 6. Size frequencies of three genera of coral recruits at Aldabra Atoll in February 2001.

FAMILIES AND GENERA OF RECRUITS

Coral recruits from 27 genera representing 13 families were recorded in quadrats around Aldabra. On the outer reefs, there was little difference in the percentage of the nine main families of recruits found at the 6 m and 10 m

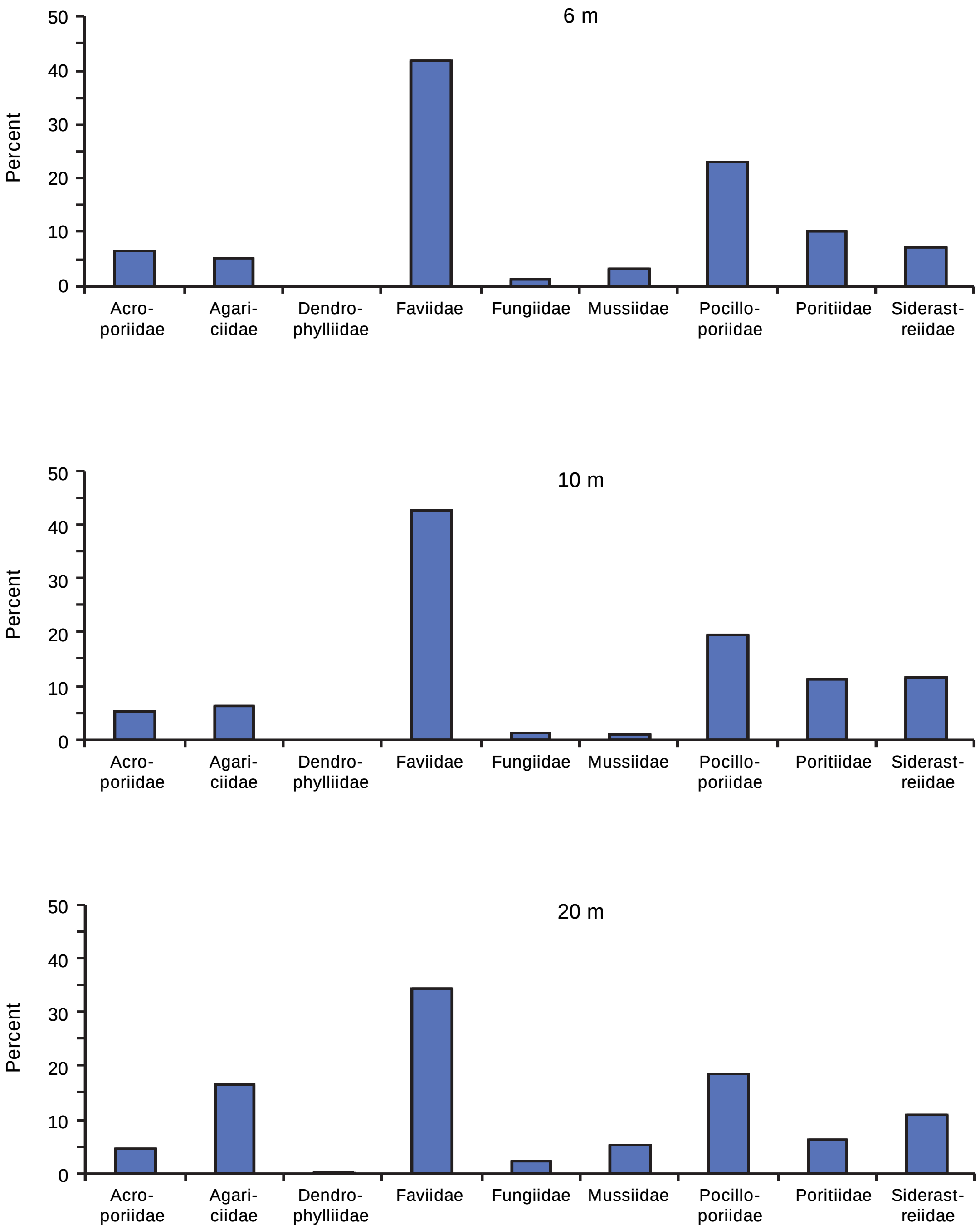

Figure 7. Percentage of juvenile coral families in shallow (6m, *n* = 1409), intermediate (10m, *n* = 1579) and deep (20m, *n* = 819) water at Aldabra Atoll.

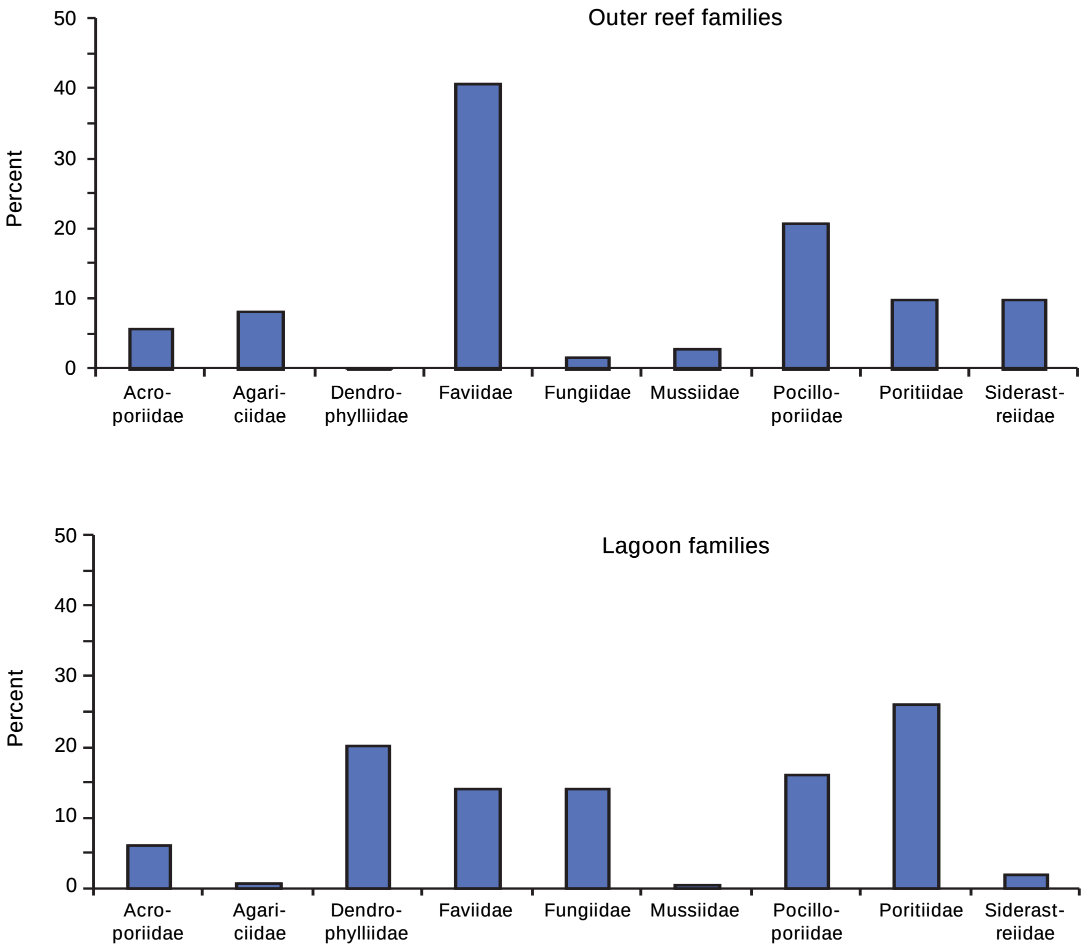

Figure 8. Percentage of juvenile coral families at Aldabra Atoll outer reef sites (pooled depths, *n* = 3715) and lagoon sites (*n* = 881).

depths (Fig. 7). However, there was a difference at 20 m depth where there were fewer faviids and poritids, and more mussids and agariciids. Lagoon sites were again different, with fewer faviids and siderastreiids, and many more fungids and poritids (Fig. 8).

Corals tagged at Sites 1,3,6 and 7 were primarily *Pocillopora* spp. and where insufficient recruits of this species were found, *Acropora* spp. recruits were also tagged. At Site 6, neither of these two genera were abundant enough so other genera such as *Pavona* and *Leptastrea* were also tagged and measured. Tagged colonies will be re-measured during the next survey.

Fish Transects

NUMBER OF SPECIES RECORDED

In February 2001, fish transects were completed at all the survey sites. However, sampling errors resulted in only 150 m^2 of benthic habitat surveyed at Site 1, and 275 m^2 surveyed at Site 6, compared to the usual of 300

m^2. At Sites 1–8, 191 species, representing 32 families were identified on the transects. The checklist surveys added 14 species and 8 families, giving 205 species representing 40 families. In November 1999, surveys of Sites 1–7 yielded 165 species representing 29 families on the transects, and 46 additional species and 5 additional families on the checklist surveys, giving a total of 211 species representing 35 families.

NUMBERS OF FISH RECORDED

The total fish count in 2001 at all 8 sites was 34,901, giving an average of 1,501 fish per 100 m^2. At the shallower sites the density was 1,431 fish per 100 $m^{2,}$ and at the 20 m depth sites 1,628 per 100 m^2).

Although fish were recorded according to six total length categories (see Methodology), the data is summarised here into three categories: 1 to 10 cm, 11 to 20 cm, and >20 cm and by transect depth. At the shallow and deep transects, the largest percentage of fish was in the 1 to 10 cm category (82% and 70%, respectively), and the smallest percentage in the 11–20 cm category (4% and 5%, respectively). The abundance of fishes in the 1 to 10 cm category at both transect depths was primarily due to large numbers of fish from families with numerous small-sized species (Serranidae (fairybasslets), Apogonidae, and Pomacentridae), and secondarily because of the number of juvenile life-stages (e.g. Serranidae).

FISH SPECIES DISTRIBUTION

The pattern of fish species distribution around the outer reef of the atoll in February 2001, followed the same general pattern as in November 1999 (Fig. 9). Sites 5, 3, 1, and 4 on the eastern portion of the atoll ranked 1st, 2nd, 3rd and 4th, respectively, for the least number of species in both years. Sites 2, 6, 7, and 8 (2001 only) on the western portion of the atoll accounted for the most number of species of fish on the transects in both years.

The difference in the species numbers from one end of the atoll to the other is striking. Thirty species were counted at Site 5 compared to a range of 62 to 115 species at the other sites. Furthermore, there were significant correlations between the number of species of fish surveyed on both the 10 m depth transects ($R^2 = 0.41$, $F_{.05},1 = 4.16$, $P = 0.087$) and the 20 m depth transects ($R^2 = 0.47$, $F_{.05},1 = 5.23$, $P = 0.062$) at Sites 1–8, and their relative positions from east to west along the shoreline of Aldabra Atoll.

FISH NUMBERS DISTRIBUTION

In February 2001, the number of fish per 100 m^2, combining shallow and deep transects ranged from 299 fish (62 species) at Site 3, to 3,867 fish (89 species) at Site 6 (Fig. 9). The 550 fish per 100 m^2 at Site 5 is unusually high for the low number of species at this site and was due to a school of 800 large fusilier, *Pterocaesio tile* (Caesionidae) crossing the transect line. The numbers of the fish counted in both shallow and deep transects combined, at Sites 1, 2, 3, 4, 6, and 7 were greatest in the 1 to 10 cm category. This ranged from 70% at Site 1 to 97% at Site 7. At site 8, 66% of the fish were in the >20 cm category. This was due to an extremely large school of >30 cm *Aethaloperca rogaa* (Serranidae) aggregated on a coral outcrop at 20 m depth.

There were no significant correlations between the densities of fishes at Sites 1–8, and the east to west positions along the outer reef.

RELATIONSHIP WITH CORAL HABITAT

There was no significant correlation between the number of species of fish, nor the density of fishes, surveyed at both depths at Sites 1–8, and the amounts of live coral habitat, and live coral and dead coral habitat combined, at each site.

The families Pomacentridae, Chaetodontidae, Labridae, and Serranidae each have several species that are commonly associated with live coral habitat, and habitat structure formed by erect dead corals. These selected fishes were examined for any associations with these measures of habitat structural complexity.

A) FISH DENSITY

Only the density of labrids at 10 m depth ($R^2 = 0.78$, $F_{.05},1 = 21.33$, $P = 0.004$) and at 20 m depth ($R^2 = 0.54$, $F_{.05},1 = 7.02$, $P = 0.038$), and the number of species of

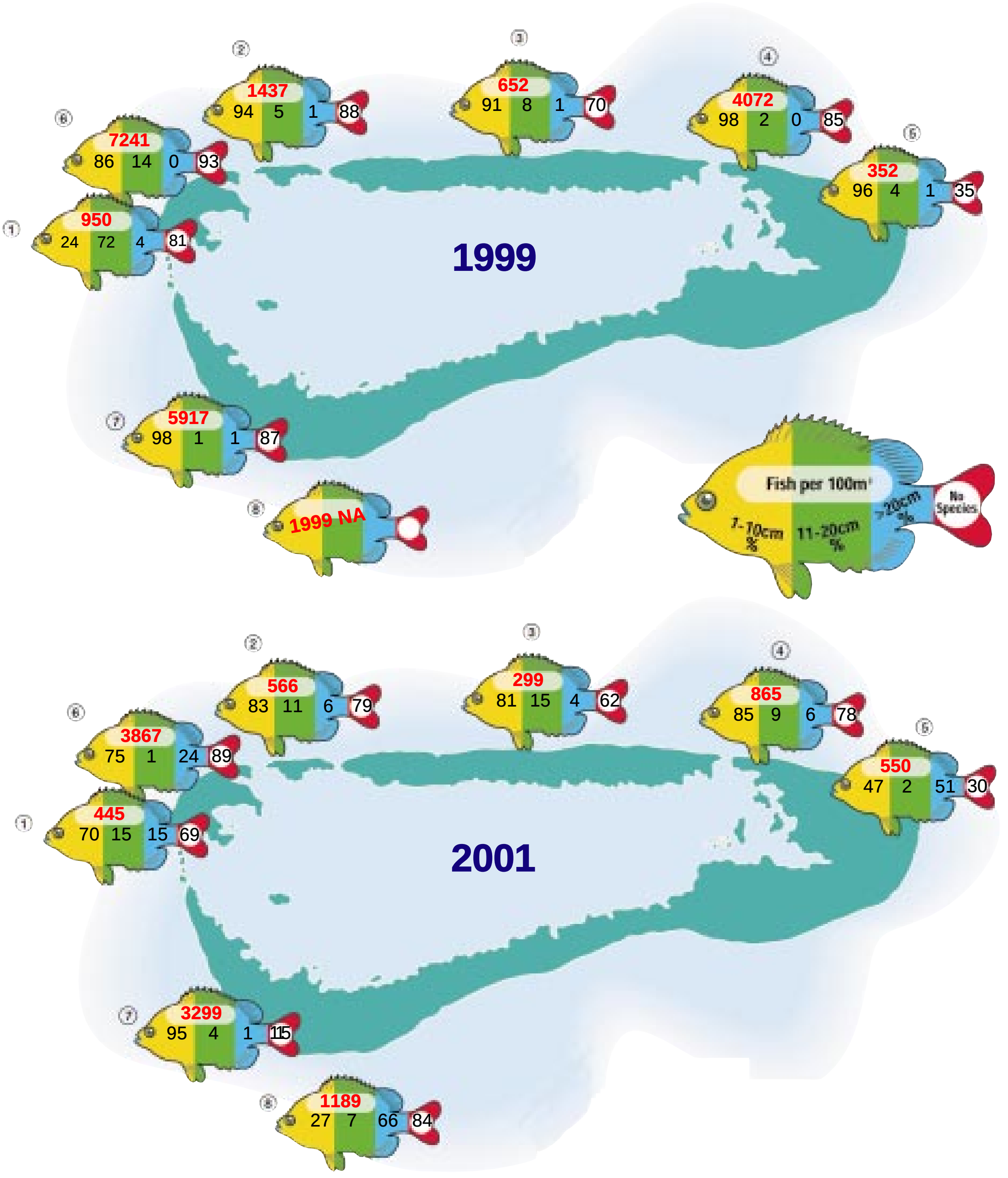

Figure 9. The pattern of fish species distribution around Aldabra Atoll in November 1999 and February 2001.

pomacentrids at 20 m depth ($R^2 = 0.67$, $F_{.05},1 = 12.18$, $P = 0.013$), were significantly correlated with the amount of live coral habitat at each site. In comparison, both the density of chaetodontids at 10 m depth ($R^2 = 0.50$, $F_{.05}$, $1 = 6.01$, $P = 0.050$) and 20 m depth ($R^2 = 0.60$, $F_{.05}$, $1 = 9.08$, $P = 0.024$), and the density of labrids at 10 m depth ($R^2 = 0.70$, $F_{.05},1 = 14.21$, $P = 0.009$) and 20 m depth ($R^2 = 0.52$, $F_{.05},1 = 6.61$, $P = 0.042$), were significantly correlated with the amount of combined live and dead coral habitat at each site.

B) FISH SPECIES

For the number of species of fish, only the chaetodontids ($R^2 = 0.52$, $F_{.05},1 = 6.53$, $P = 0.043$) and pomacentrids ($R^2 = 0.64$, $F_{.05},1 = 10.73$, $P = 0.017$) were significantly correlated with the amount of combined live and dead coral habitat at each site.

DISCUSSION

Coral Community

The 1999 survey results showed that coral bleaching in 1998 had had a pronounced effect on the reefs around Aldabra Atoll, particularly in shallow water (Teleki *et al.*, 1999; 2000a). Between 1999 and 2001 there has been little change in the outer reefs, with some suggestion of coral recovery at both the depths studied. There is no evidence of high macro algal growth since 1999, with the exception of site 7 and, to a lesser extent, Site 6. These sites had high coral mortality in 1999 providing new space for algae to colonise, particularly at Site 7 which appears to have provided suitable conditions for abundant coralline algae growth.

By 2001, the dead corals still visible in 1999 had been mostly broken down to rubble and sand, or were covered by other organisms. The shapes of dead massive colonies were still visible in 2001, and some showed evidence of tissue regeneration from surviving areas around the side and base of the colonies. Many are now covered with coralline algae that are known to provide a good substrate for settlement of coral larvae.

The detected natural variation in live coral cover around Aldabra Atoll can be attributed to the prevailing weather and current patterns. The area encompassing Sites 1, 2 and 6 is the most sheltered and has the highest live coral cover. Live coral cover at depth was good around most of the atoll, with the exception of Site 5 that appears to be a rubble zone.

The new sites established in the lagoon have shown that coral cover is high in the vicinity of the drainage channels. Live coral cover at Sites 9 and 10 was estimated to be double that of most outer reef sites, though species diversity appears lower. There is also less evidence of old dead coral colonies in the lagoon, suggesting that the effect of the 1998 bleaching may not have been as great here, a hypothesis supported by observations made by Teleki *et al.* (1998) just after the 1998 bleaching. Lagoon species may be acclimatised to extreme temperature changes, and new tides may bring in deep cooler water through the channels on every tide change reducing the time lagoon corals are exposed to extreme temperatures.

The temperature data recorded by the three loggers during April through June/July 2001 supports the suspected normally wider range in temperatures in the lagoon. During this period the temperature variation at the two lagoon sites was 9.1 and 9.4 ^{o}C respectively and at the outer reef site 6.1^{o}C.

Coral recruitment at Aldabra appears to be good, with most occurring at around 10 m. This would be expected as corals in shallower than 10 m depth have to cope with higher exposure to swell and a mobile substrate making settlement difficult, while corals in deeper water have less available light and reduced available settlement space due to the higher proportion of live coral. Levels of recruitment at Aldabra are consistent with those found at other locations, though direct comparison is not always possible due to differing sampling methods (Miller *et al.*, 2000). The diversity of coral families recruiting to Aldabra is high and to be expected in view of the high coral diversity on reefs around the atoll.

Size frequencies of recruits measured in 2001 suggests that there may have been several cohorts recruited since the coral bleaching of 1998. It appears that the fast colonisers such as *Acropora* and *Pocillopora* have had the best chance to produce several cohorts due to their high rate of growth and reproduction. The presence of so many recruits and evidence of several cohorts suggests that the atoll may be primarily self-seeding, though it is conceivable that the predominant southeasterly current may be carrying recruits from Assumption, Astove, Cosmoledo and potentially from reefs much further afield.

Fish Community

Unlike sessile coral colonies, fish populations are notoriously variable, even when surveyed at short time intervals. The following conclusions are therefore made within the limitations of two surveys having been undertaken some 16 months apart.

Comparing the 1999 and 2001 fish species lists there is generally good agreement between the two surveys even though the total numbers identified vary somewhat: in 1999, 35 families and 211 species and in 2001, 40 families and 205 species. Using a different survey method, Spalding identified 35 families and 287 species in 1998, prior to the bleaching episode.

Given the broadly similar diversity, it is therefore of note that a comparison of the actual counts of fishes and species at each site in 1999 and 2001 appear to be very different (Fig. 9). At this stage it is not known whether this difference is attributable to the different seasons in which the surveys were conducted, some structural change in the fish community, or a survey error. Nevertheless, it was concluded that surveys should, if possible, be conducted at similar times of the year, ideally in November and February.

If one compares the findings of the 1999 and 2001 surveys, the following conclusions emerge:

- In 1999, the lowest fish density was at Site 5, and the highest at Site 6. In 2001, the highest count was still at Site 6, but the lowest at Site 3. This may be due to an anomaly at Site 5 where a large school of fusiliers crossed the survey area. If this number is excluded, Site 5 yields the lowest count in 2001.
- In both years most of the fish counted were in the 0–10 cm length category. This was due to large numbers of fish from families with numerous small-sized species (Serranidae (fairybasslets), Apogonidae, and Pomacentridae), and because of juvenile life stages (e.g. Serranidae). Exceptionally at Site 8, most fish were in the >20 cm category.
- In both years there was no correlation between the densities of fishes counted and the relative positions of the survey sites east to west along the atoll.
- In 1999, there was also no correlation between the number of species counted and the east/west position of the survey sites, but in 2001 a significant correlation did appear, both at the 10 m ($R^2 = 0.41$) and 20 m depths ($R^2 = 0.47$).
- On coral reefs, fish diversity and habitat complexity are often linked (Ebeling & Hixon, 1991; Sebens, 1991; Williams, 1991; Turner *et al.*, 1999). In both 1999 and 2001, a distinction was made between benthic habitat whose structural integrity was still evident (live or dead coral), and habitat without such structural integrity (sand, rock and rubble). In 1999, no correlation was found between the density of fish, nor the number of species, and either the live or dead coral cover. In 2001 the analysis was slightly different, but a similar conclusion was drawn.
- However, in 1999, an analysis of fish families that have species commonly associated with structurally complex habitats concluded that there were positive correlations amongst the Chaetodontidae, Labridae and Serranidae and live or dead coral. Although this association was not found amongst the Serranidae in 2001, it remained within the Chaetodontidae and Labridae, and was evident amongst the Pomacentridae at 20 m depth.

Although it is too early to establish any significant trends in Aldabra's fish populations, the nature of any long term study means that these will emerge in time. In particular, close attention should continue to be paid to the relationship between coral reef related species and reef structure. For example, the numbers of specific corallivorous chaetodontids may prove to be an effective indicator of reef health (Crosby and Reese, 1996).

The Aldabra Marine Programme: Short Term and Long Term

The Aldabra Marine Programme has now established 11 permanent monitoring sites around Aldabra, seven of which have now been surveyed twice. These sites provide baseline data and will generate increasingly more important data as they are resurveyed on a yearly basis. Furthermore, as the information is gathered and analysed more questions about the atoll are generated, and therefore the scope to amplify the study.

After completing the 1999 survey, the AMP set out a list of five goals for the future. All of these goals have been achieved with the exception of the initiation of a study of mangrove ecosystem dynamics. Therefore, AMP has increased its research base in 2001 by:

- Continuing the fish and coral monitoring of permanent sites, and establishing one new long term site on the highly exposed southern shoreline;
- Establishing permanent monitoring sites in the lagoon and Passe Houareau;
- Initiating a coral recruitment study in the lagoon and outer reef slopes, and tagging corals to follow their survival and growth;
- Deploying temperature data loggers.

There was also the opportunity to train Aldabra staff in survey techniques for coral recruitment and echinoderm population assessment, and to discuss suitable monitoring programmes for the station staff. AMP remains committed to training Seychellois rangers in marine survey techniques and hopes to continue doing so in the future.

A long term marine science programme for Aldabra has now been established by AMP. Future surveys will continue to monitor the permanent sites and it is hoped that another site will be set up on the southern coastline along with further lagoon sites. The team also aims to improve the scope of the physical marine monitoring and expand the lagoon work, hopefully by appointing a PhD student.

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Part IV

Thematic Reports

Economic Analysis of Coral Bleaching in the Indian Ocean – Phase II

HERMAN CESAR[1], LIDA PET-SOEDE, SUSIE WESTMACOTT, STEPHEN MANGI, & ANNABELLE AISH

INTRODUCTION

The compilation of studies presented in this chapter forms the end-product of the socio-economic component of Phase II of the Coral Reef Degradation in the Indian Ocean (CORDIO) Project. The research was commissioned by the World Bank and carried out by the Institute for Environmental Studies (IVM) at the Vrije Universiteit in Amsterdam. It was funded through the World Bank's Netherlands Consultants Trust Fund.

Phase I involved a rapid assessment of the socio-economic impacts of coral bleaching and was carried out in all countries of the Western Indian Ocean (Westmacott *et al.*, 2000). This research took place directly after the 1997–1998 bleaching event. It used existing data to provide a general qualitative overview of the importance of reef-dependant economic activities, and presented country case studies on two important coastal industries: fisheries and tourism.

In terms of the effects of coral bleaching on fisheries, it was concluded that while direct impacts on reef habitats were clearly observable, the indirect impacts on fish community structures (and thus fisheries themselves) were less visible. For tourism, no clear changes in arrivals were noted, as divers had already made their holiday bookings or were not informed about the coral bleaching event. At that stage, it was suggested that possible effects on fisheries and tourism would become visible with time (Cesar *et al.*, 2000). Phase II of the socio-economic research focused on selected countries (Kenya, Tanzania and the Seychelles) where detailed data collection and analyses were carried out. The work evaluated the socio-economic importance of these countries' reefs to society and estimated the economic value of the reef resources for both fisheries and tourism.

THE EFFECTS OF CORAL BLEACHING ON FISHERIES IN THE INDIAN OCEAN

The aim of this section was to summarise two CORDIO fisheries papers by regional expert scientists under Phase II of the project. The first is by T.C. McClanahan which described the current status of the reef habitat and associated reef fish at several sites in Kenya, as well as the current status of the reef related fisheries at Kenyatta Beach. The second is by Edwin Grandcourt, who de-

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scribed the current status of reef-related fisheries in comparison to that of years before the bleaching event in the Seychelles. They are presented here as one integrated text.

Impact of Bleaching on Reef Fisheries in Kenya and the Seychelles

A true comparison of the results from the two case studies can only be made for the fishery-dependent data as these were collected and analysed in both cases. The Kenyan fishery that was monitored at landing sites included beach seine, spear gun, net, traps and hand line gear. The Seychelles fishery monitored at the landing sites included harpoon, trap and hand line gear. Time series of the Kenyan landings included 3 years before and after the bleaching event and time series of the Seychelles landings included 28 months before and after the bleaching event as well as a longer data set with information collected between 1991 and 2001. The Seychelles data were mostly analysed per fish category, while the Kenyan case also looked at total catch biomass. To account for the impact of changes in fishing pressure, fishing effort was monitored in both cases by recording the number of fishers per gear that went fishing on sampling days.

Total Catch, Individual Catch and Catch per Unit Effort

In Kenya, the total catch per landing site and catch per man decreased by 7% and 21% over the entire period before and after bleaching. The trend observed after the event was not significantly different from that before the event. Beach seines experienced the greatest decrease in total catch of 43% over this period while traps experienced a more modest decrease of 10%. On a catch per man basis all gear experienced significant declines of between 13% and 37%. Trends were not significantly related to the bleaching event.

In Seychelles, catches of Serranidae, Lethrinidae, Siganidae and Octopodidae were decreasing before the bleaching event. Comparing catch per unit effort (CpUE) and monthly yields before and after 1998 indicates that there were no significant differences for the families exploited by the hand-line fisheries: Lutjanidae, Serranidae, Lethrinidae and Carangidae. There was a significant increase in the abundance index and yield per km^2 for Octopodidae. Also, the CpUE and yield per km^2 of the family Siganidae decreased significantly following the bleaching event. Whilst there was no significant differ-

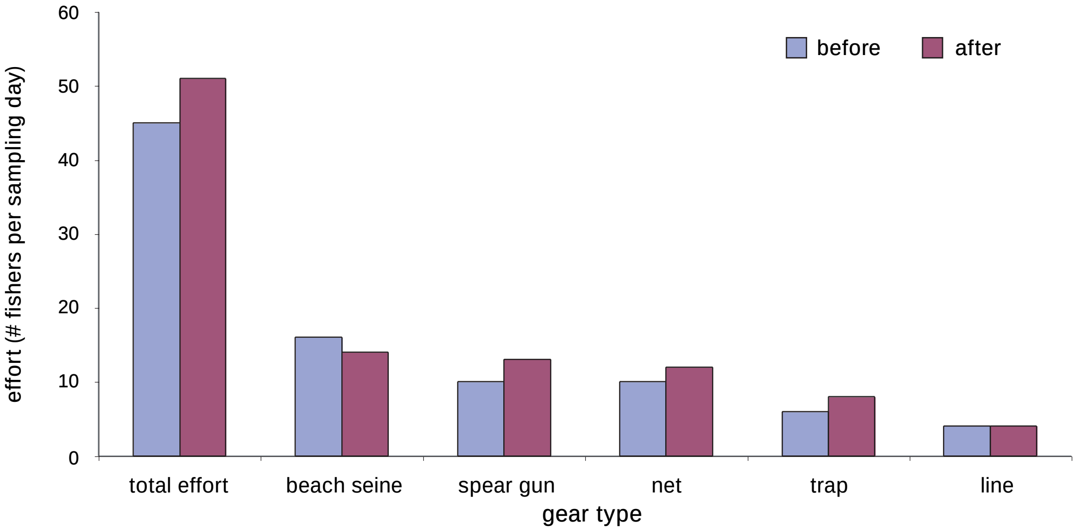

Figure 1. Comparison of fishing effort per gear per sampling day between the period before bleaching and after bleaching in the Kenya case.

Table 1. Comparison of cost and benefits per fishing type based on average catch per trip before and after the 1998 bleaching for the Kenya case

Method	Effort # Fishers		Gross income Ksh. per man/trip		Ksh. all fishers/trip		Ksh. per man/month*		Costs Ksh. per month^	Net income Ksh. per man/month
	before	after	before	after	before	after	before	after		
Beach seine	16	14	245	198	3790	2682	5868	4741	1042	3699
Spear	10	13	302	305	2956	3878	7240	7329	83	7246
Net	10	12	251	279	2510	3466	6025	6708	514	6194
Trap	6	8	249	230	1595	1820	5981	5530	150	5380
Line	4	4	266	288	956	1152	6376	6911	83	6827
Total	45	51			11808	12998				
average	9	10	262	260	2362	2600	6298	6244	374	5869
increase		12		–0.86		10.08		–0.86		

Source: Data from McClalahan (presented in Cesar et al., 2002)
^=Cost includes average initial gear price, the replacement rate and repair costs per month
=assuming 24 trips per month on average
Ksh.= Kenya shillings
#= Number

ence in the abundance index for other trap fish, a significant increase in yield was noted. The results of partial correlations showed that the 1998 bleaching event was only significantly associated with the changes for Octopodidae and not with the changes for Siganidae.

Catch Composition

In Kenya, only the total catch of rabbit fish decreased by 8%; on an individual fisher basis, there was a 20 to 30% decrease in average daily catch of rabbit fish, parrotfish and octopus over the monitoring period. For the Seychelles, there were no significant changes ($P>.05$) in the relative proportions of principal families in the trap and demersal hand-line fisheries following the bleaching event.

Fishing Effort

In Kenya, fishing effort (as measured by numbers of fishermen and boats) increased significantly by 17% and 10% respectively over this period (Fig. 1). In the Seychelles, a significant positive correlation was found in the number of trips targeting semi-pelagic species and a significant negative correlation was found in the number of trips targeting demersal species between 1991 and 2001. This implies a decreasing fishing pressure on demersal resources. This is supported by the shift in importance between demersal and pelagic fish production.

Fishery Economics

For Kenya, there was a modest increase in the price of fish over the study period and, assuming a constant cost of fishing, the total demersal fishery became 18% more profitable (Table 1). However, as the number of fishermen and the catch per fish category showed significant changes over the period, the actual increase in gross individual incomes resulting from the price increase was negligible (0.8%). Averaging the gross benefit per type of fishery, an increase of 10% was observed; however the per fisher gross income decreased slightly.

For the Seychelles, revenues per trip for the semi-pelagic hand-line fishery increased following the 1998-bleaching event, but were not significantly explained by the bleaching event. Net income per fisherman per trip in the demersal hand-line fishery decreased, but again this was not significantly related to the bleaching event. Differences between net income for trap and spear fishers were not significant. Mean prices for demersal species at landing sites dropped from US$ 2.9/kg in the pre-bleaching period to US$ 2.4/kg in the post-bleaching period. Conversely, inflation increased from a mean of –0.17% to 6.3% in these periods respectively.

HABITAT

The statistical analysis indicates that benthic cover varied both according to management and also between the years before and after 1998. This implies a bleaching impact, which is not surprising as coral bleaching most directly affects benthic reef organisms. The management factor was important for nearly all benthic cover categories apart from fleshy algae and soft coral. Hard coral cover was most drastically reduced in the marine park. Soft coral was reduced drastically at fished reefs but not in the marine parks. Open space was colonized by turf and coralline algae at all sites and by fleshy frondose algae (i.e. *Sargassum* and *Turbinaria*) on marine park reefs. Green calcareous algae did not change in either management category. There were more sponges on the fished reefs three years after the bleaching compared with three years before the event, but no change in the marine parks.

FINFISH

Analysis plots suggest large differences between the fish community according to management categories; however there are few consistent differences for the years before and after 1998. There was a greater density of finfish in marine parks than at fished reefs for all groups except the pufferfish (Diodontidae), wrasses (Labridae), and damselfish (Pomacentridae) families. In marine parks, the density of parrotfish (Scaridae), goatfish (Mullidae), wrasses and butterflyfish (Chaetodontidae) was greater during the three years before than three years after bleaching. On the fished reefs, the density of wrasses and damselfishes decreased over the bleaching period, while the angelfishes (Pomacanthidae) and the mixed group numbers increased. Surgeonfish (Acanthuridae) density increased by 267% in the fished reefs in 1999, but was unchanged in marine parks. The biomass was, however, 244% higher (F=6.2, $p<0.02$) indicating larger individuals after 1998.

These results, while significant, cannot be explained by the bleaching event only as the management factor obscures clear trends. However, these shifts in fish abundance and composition may change catch opportunities for fishers that target the reefs at some point.

Discussion

Neither study provides straight forward evidence of an impact of coral bleaching on particular fisheries. The only remarkable observation is the significant associated increase in average Octopodidae yield. For the principal coastal fisheries of Seychelles, there were however no significant trends related to bleaching. The Kenyan case study shows significant changes in the benthic community and some lesser changes in the fish community that may result in changes in the fishery over time; however the fisheries data suggests that there were no large changes in fish catch only 2 years after the event.

The changes in the Kenyan fisheries are more likely to be related to the reported increase in effort. The increase in fish prices contributed to making the fishery slightly more profitable but this increase is more likely to be an adjustment to the country's economic inflation. The increase in fish prices may have been an incentive for newcomers to join the fishery, but the net benefits remained marginal and there was no increase in profits for those individuals that joined. The increase in fisher numbers is similar to the 2.2% rate of population growth for Kenya and may simply reflect this demographic trend. Similarly in the Seychelles case study, there are some critical socio-economic and resource base management issues relating to the demersal hand-line fishery that will have to be looked at for a different purpose, considering the strong decline in relative importance of demersal fisheries. Quantitative analyses of changes in fishing pressure for the various gears would provide a possible explanation for some of the observed changes. In the Seychelles case study discussion, it is suggested that the decrease in siganids may have resulted from high levels of fishing effort.

It would have been interesting to have fishery-independent data available for the Seychelles case. Also, not all gears monitored were operated at depths where the reported bleaching occured in the Seychelles (Turner *et al.*, 2000). There are claims that deeper strata were unaffected by the mass bleaching event (Bigot *et al.*, 2000), however, these have yet to be authenticated with survey data. While this must be considered when studying the results from the Seychelles, the results from Kenya do not implicate drastic impacts after such a short time span.

IMPACTS OF CORAL BLEACHING ON TOURISM IN THE WESTERN INDIAN OCEAN

Coral mortality from the 1998 bleaching event exceeded 50% on many Indian Ocean reefs (Wilkinson *et al.*, 1999). This reduced coral cover to between 10 and 90% of previous levels in most areas along the western Indian Ocean (Goreau *et al.*, 2000; McClanahan *et al.*, 2001). Given that coral reefs are an environmental resource on which much coastal tourism activity is based, major economic impacts from the 1998 bleaching episode were to be expected. Not only does tourism provide direct revenue, it also stimulates other economic sectors to grow and support the tourism industry.

The three studies underlying this section evaluate the economic costs to tourism in Kenya, Zanzibar and the Seychelles. These studies (by Stephen Mangi – Mombasa/Malindi; by Annabelle Aish – Seychelles; and by Zeinab Ngazi – Zanzibar) are reported in Cesar *et al.* (2002). The studies are based largely on an appraisal of the awareness of tourists visiting these countries about coral bleaching and their willingness to pay for improvements in reef quality. A Contingent Valuation Method (CVM) was used to estimate the economic value of the reef by asking tourists to state how much extra they would pay to enjoy healthy coral reef environments.

Methods

In order to gauge tourists' reactions to coral bleaching and reef degradation, questionnaire surveys were used. These were administered to departing tourists in the major airports of the three Indian Ocean countries, thereby targeting those who had just completed their holidays. Surveys were conducted during the months of September and October in 2001 in both Zanzibar and Kenya, and in June 2001 in the Seychelles. A sample of between 150–200 tourists was required; 167 respondents were questioned in Kenya, 157 in Zanzibar and 199 in the Seychelles.

In this study, the economic values of the reefs were measured through respondents' Willingness To Pay (WTP) for the use/enjoyment of this resource. They then had to state how much extra they would hypothetically pay per holiday to enjoy a better reef quality, irrespective of other aspects of their holiday. Clearly, some people find it hard to place a price on natural assets, and react by giving 'protest bids'. Nevertheless, this method remains one of the only viable ways of measuring environmental resources in an economic way. The collected data were analysed and a mean WTP for reef improvements was generated. The WTP data were then regressed against variables that were thought to be determinants of these bids, including income, level of education, age and length of holiday.

Results

SELECTED STATISTICS FOR SAMPLE POPULATIONS

The majority of tourists surveyed in Kenya and the Seychelles were from the UK (Fig. 2, next page). For Kenya, British tourists accounted for 41% of the total sample, while for Seychelles they comprised 26% of all respondents. Most tourists in Zanzibar were from other European countries, although tourists from the UK still comprised a high proportion (21%) of the sample.

Income distribution of the respondents (Fig. 3, next page) showed that in both Kenya and Seychelles, the majority of respondents fell into the income category of US $30,000–49,999. This was in contrast to tourists

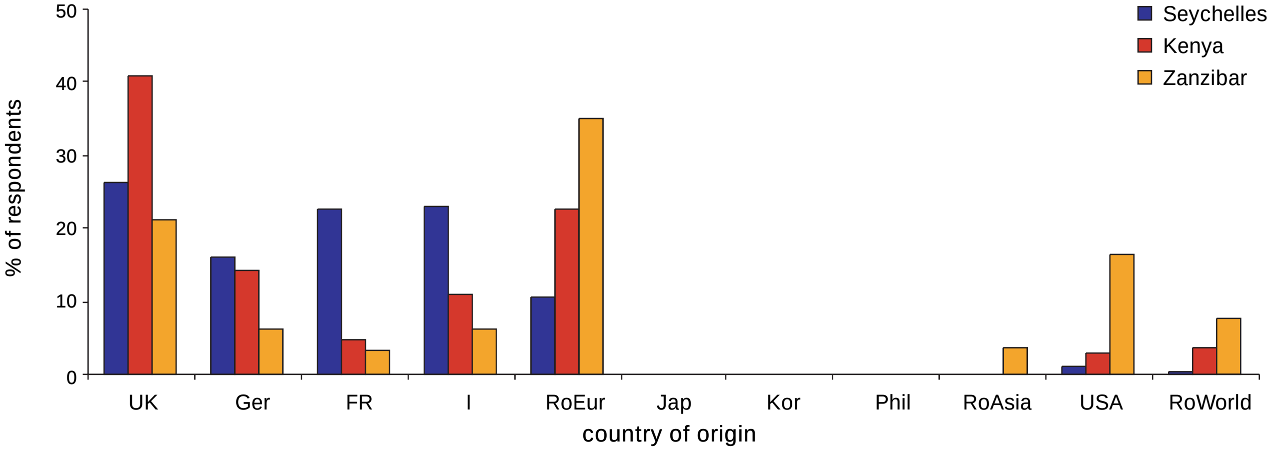

Figure 2. Nationality of survey respondents.

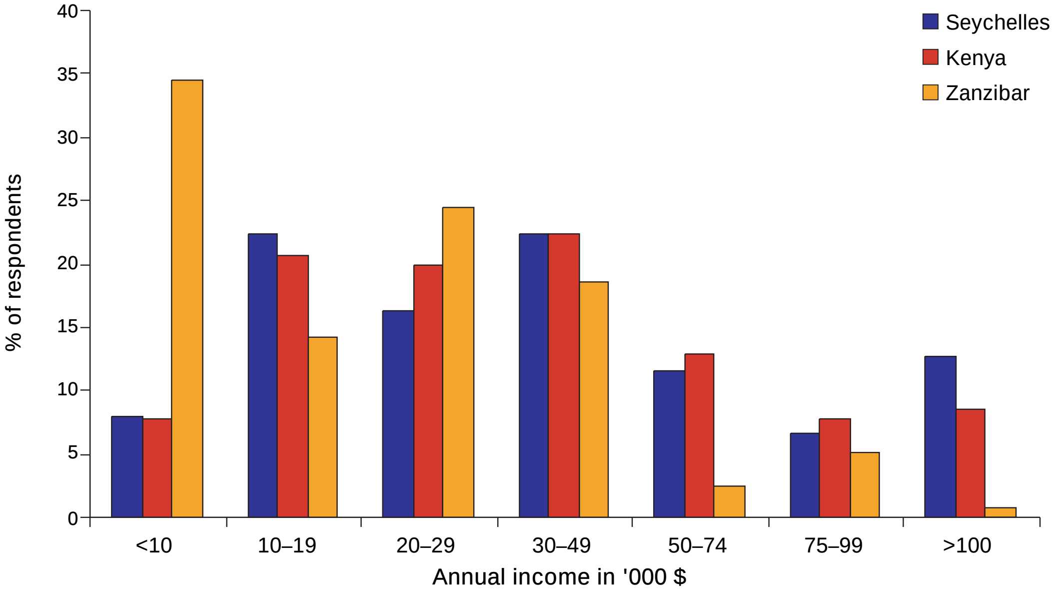

Figure 3. Annual income of survey respondents.

questioned in Zanzibar, the majority of which earned below US$10,000 annually. The most probable explanation for this variation in salaries is the influence of younger age-groups in the Zanzibar survey sample. Being students or potentially in their first job, they would automatically have lower incomes than their fellow tourists in Kenya and the Seychelles. Furthermore, the Seychelles is a relatively expensive holiday location that, as such, could be out of reach for tourists with lower salaries.

SAMPLE STATISTICS ON HOW MARINE LIFE WAS ENJOYED

With regard to how tourists enjoyed the marine life, most tourists in Kenya (53%) and the Seychelles (74%) chose to snorkel (Fig. 4). In contrast, 65% of respondents questioned in Zanzibar came into contact with the marine life through SCUBA diving. This result could be explained by Zanzibar's high number of young 'backpacking' tourists. This group may be slightly more adventurous than the average tourists in the Seychelles and Kenya, and more likely to engage in activities such as SCUBA diving. An additional determining factor might be the high price of diving in the Seychelles. Tourists who are certified divers may have decided against spending money on diving when the marine life could be viewed by much cheaper snorkelling excursions.

Irrespective of the method in which the marine life was viewed, the most important marine characteristic for the tourist was the variety of fish. This was cited as 'most important' by around 54% of all respondents in the Seychelles, around 37% in Kenya and finally 28% in Zanzibar. In general, fish life seems to have been sustained despite the widespread coral bleaching. Whether these fish populations will remain on reefs that disintegrate further is unknown. Interestingly, in the Seychelles, the characteristic coastal 'granitic structures' are usually surrounded by a considerable variety of fish; this provides entertainment to snorkelling/diving tourists regardless of the state of nearby corals.

PERCEPTION OF CORAL BLEACHING

Respondents were asked, who they felt was responsible for the corals dying around the world. The results were mixed. Around 27% and 23% of respondents in Kenya and Zanzibar respectively decided that industrialized countries were responsible for global coral mortality (Fig. 5, next page). Tourists in the Seychelles were more certain: 45% maintained that it was the fault of industrialized nations, thereby potentially making a link between level of greenhouse gas emissions, global warming and coral bleaching. However, respondents were never directly questioned about why these nations might be responsible. Governments and the whole (global) population came second and third respectively in terms of their level of responsibility for coral degradation. Divers and fishermen were identified as the least guilty parties.

VALUATION OF THE REEF RESOURCES

Respondents were asked to indicate how much extra money they would be prepared to pay so as to enjoy better reefs (assuming that the fish abundance in those reefs would be same). The results from the three surveys are presented in Table 2 (next page). According to this CVM analysis, respondents were willing to pay US$ 98.7 extra per holiday in the Seychelles, US$ 87.7 extra in Zanzibar and US$ 59.0 extra in Kenya to experience healthy reefs.

However, WTP responses varied a great deal, particularly among tourists in Seychelles and Kenya. They were less varied in Zanzibar. Approximately 90% of the respondents in Kenya, 73% in Seychelles and 71% in

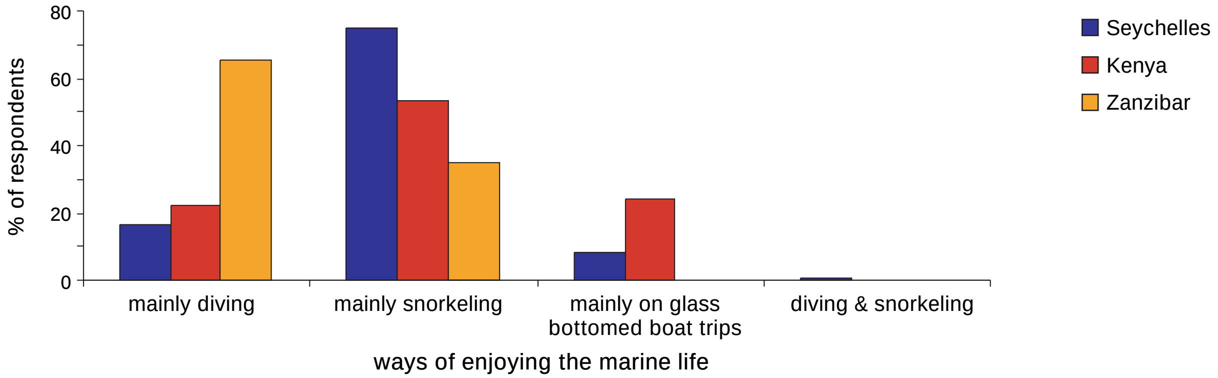

Figure 4. Ways in which respondents enjoyed marine life.

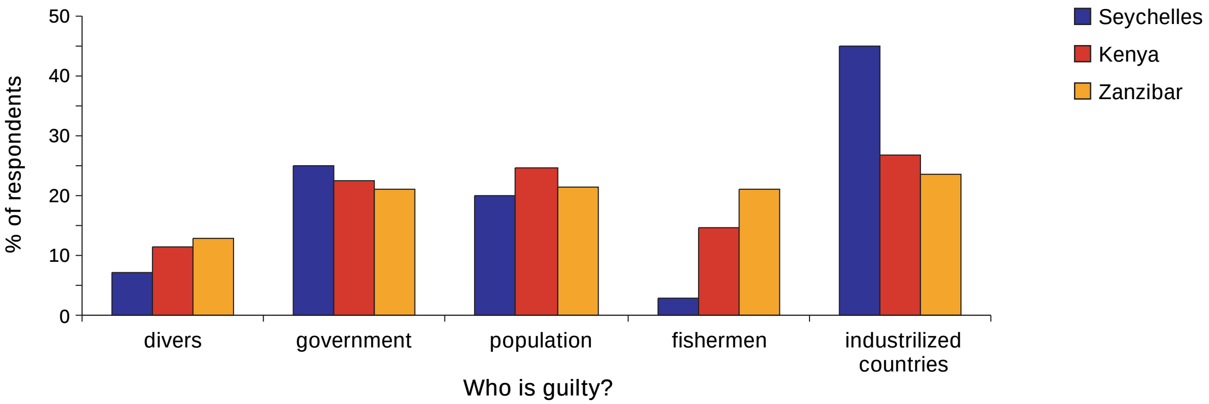

Figure 5. Respondents' views on who was to blame for global coral mortality.

Zanzibar were prepared to pay between US$ 0–100 extra for unbleached reefs. These losses of 'welfare' incurred by tourists are relatively significant; they give weight to the notion that healthy reefs are an important factor for successful Indian Ocean tourism. The WTP amounts were regressed against income, level of education, age, level of dive certification and the presence of marine life as a reason to visit the destination. This allowed for further insights into which socio-economic variables might drive respondents' willingness to pay. The data showed a statistically significant correlation with some of these variables (F=3.3, p<0.01) and the results are displayed in Table 3.

WTP amounts were statistically significant when regressed against income level of the respondents, and the presence of marine life as a reason to visit the area. Wealthier tourists and those with higher expectations of marine life were therefore prepared to spend more to enjoy good quality reefs. On the other hand, age, level of dive certification and level of education did not produce statistically significant relationships.

Table 2. Tourists' willingness to pay to experience better reefs

Willingness to pay extra money to experience better reefs (US$)				
country	**average**	**deviation**	**high**	**low**
Seychelles	98.7	267.9	2000	0
Kenya	59.0	201.3	1500	0
Zanzibar	87.7	100.0	500	0

In concluding, the tourist populations chose their holiday location according to a variety of factors, yet placed considerable importance on the presence of marine life. Many stated that they enjoyed the reefs. The maintenance of this satisfaction may lie with the continued abundance of a varied fish life around the reefs. As a result, visitors' awareness of bleaching did not seem to translate into a rejection the destination as a whole. Nevertheless, the 'welfare losses' from coral bleaching are significant. The majority of respondents in all three destinations claimed that they would be willing to pay US$ 98.7 extra per holiday in the Seychelles, US$ 87.7 extra in Zanzibar and US$ 59.0 extra in Kenya in order to experience healthy reefs. The economic consequences of these estimates are discussed in the next section.

ESTIMATING ECONOMIC LOSSES OF CORAL BLEACHING

The aim of this section is to transform the impacts on fisheries and tourism summarized in the two previous sections into economic costs for the countries of Kenya,

Table 3. Regression results for respondents' WTP to experience better reefs

WTP Regression statistics **variable**	**F-value**	**p-value**
Whole model test	3.25	0.01
Income	6.71	0.01
Age	3.15	NS
Level of dive certification	1.47	NS
Marine life presence as a reason to visit	4.07	0.05
Level of education	0.96	NS

R^2 = 0.16
Adj. R = 0.11

Tanzania and the Seychelles. Unlike the estimates in Phase I, no attempt will be made to generalize the data in order to obtain an overall estimate for the Indian Ocean. As discussed in Phase I and in Hoegh-Guldberg *et al.* (2000), there is still large uncertainty surrounding the economic costs of coral bleaching and subsequent mortality. In Phase I, scenarios were discussed and estimates were presented providing a plausible range of economic costs (see Table 4). In Phase II, instead, the two main impacted industries (tourism and fisheries) have been studied in much more detail and local estimates of the ecosystem goods and services on which they depend will be discussed in the next two sections.

Fisheries

In Phase I, fisheries losses were discussed according to both hypothetical scenarios and field observations. The field observations were, however, inconclusive and more research was called for at that stage. There are basically two competing ideas of the effects of coral mortality and fisheries which could explain the inconclusive results in the short run. One the one hand, coral mortality opens up space for algae, leading to an increase in primary production. Secondly, coral mortality will lead to reduced rugosity and hence less possibilities for fish to hide and feed and this will lead eventually to lower fish yields.

The hypothetical scenarios discussed in Phase I were necessarily arbitrary but they enabled the cost estimates presented in Table 4. In the recent fisheries studies for Kenya and for the Seychelles, fisheries data-sets from the three years since the bleaching event were used, together with pre-bleaching data, in order to analyse the socio-economic impacts. Quite surprisingly, the results were still indecisive. Species composition did change, in some cases considerably. However, overall yield and income did not change significantly (see Fig. 6 on next page for Kenya). The Kenya data showed more or less constant individual incomes for fishermen (an increase of 0.8%), resulting from increased effort and prices. For the Seychelles, catch per unit effort and monthly yield increased for octopi but not for rabbit fish (two of the main reef fisheries in the Seychelles). With the exception of a decrease in net income per fisherman per trip in the demersal hand-line fisheries, the economic indices in all other fisheries were not significantly affected.

These results suggest that the costs to fisheries due to bleaching are still following what is described as the 'optimistic scenario'. Continuation of underwater fish counts and fishery economics data are needed in the coming years to see whether this optimistic scenario holds in the long run or whether there is only a lag in the negative effects of bleaching. Ideally, the two conflicting hypotheses about the relationship between coral mortal-

Table 4. Estimates of the overall economic valuation of the socio-economic impacts of the 1998 coral bleaching event in the Indian Ocean (Net present value in million US$ over a 20 year time horizon with a 10% discount rate).

Coral Reef Ecosystem Services	**Optimistic Scenario**	**Pessimistic Scenario**
Food Production (e.g. Fisheries)	0	1361
Tourism and Recreation	494	3313
Disturbance Regulation (coastal protection)	0	2152
Other Services	114	1200
Sum Total	608	8026

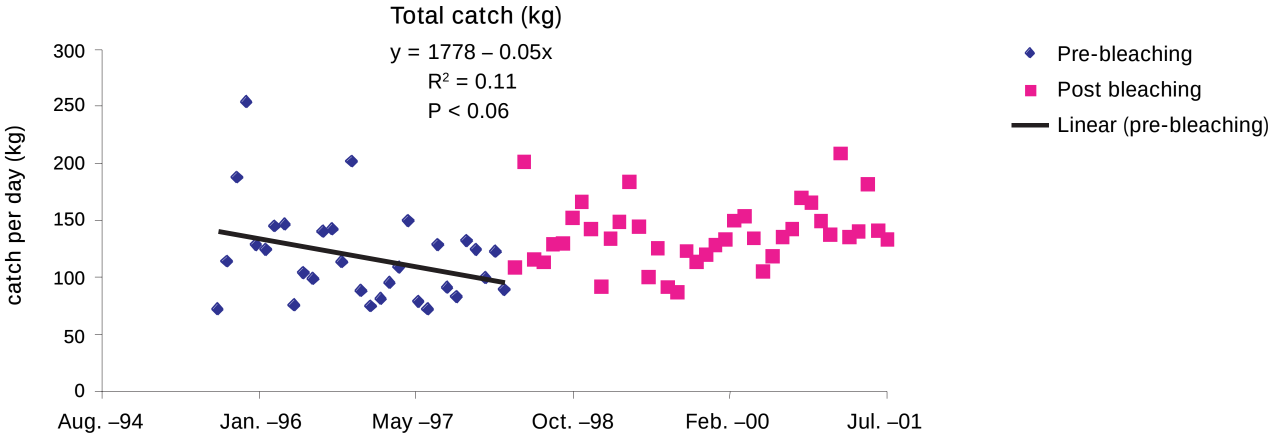

Figure 6. Plots of monthly catch as total catch per landing site (McClalahan et al., 2002).

ity and fishing yields can be tested. This would allow us to state whether negative fishing impacts from reduced rugosity is stronger or weaker than the positive fishing impacts from the increase in primary production.

Tourism

The WTP-data presented in Table 2 enabled the rough estimation of total welfare losses for Kenya, Zanzibar and the Seychelles as a result of coral bleaching. For this estimation, the following information was required:

NUMBER OF TOURISTS

In 2001, 76,329 tourists visited Zanzibar. For the Seychelles, 2000 saw 130,046 visitor arrivals. The number of tourists visiting Mombasa in 1998 was 137,402 (1998 data from Kenya Hotel Association). As there are no data available for Malindi, we have only used estimates for Mombasa. Tourism is expected to grow by 5.2% per year in Eastern Africa from 2000 to 2020 according to the World Tourism Organisation.

INTEREST MARINE LIFE

The WTP questions were only asked to visitors who expressed interest in the marine environment. This sample percentage was 72% in the Seychelles, 68% in Kenya and 81% in Zanzibar.

RECOVERY OF CORALS

In order to evaluate the total welfare loss over a 20 year time period, we need to estimate how quickly the coral reefs recover from bleaching. Applying conservative estimates, the corals should recover at a linear rate over a 20 year period with full recovery after 20 years. It is also assumed that WTP relates linearly to recovery.

Combined, this gives the WTP estimates presented in Table 5. Losses in 2001 would be US$ 9.7 million for the Seychelles, US$ 6.4 million for Mombasa and US$ 5.4 million for Zanzibar. Net present values of these annual welfare losses over a 20 year time period with a 10% discount rate were generated. These values take into account the effects of tourism growth combined with coral recovery and are presented in the last column of Table 5. This table shows considerable losses: a total of US$ 71.5 million for the Seychelles, US$ 47.2 million for Mombasa and US$ 39.9 million for Zanzibar.

In the tourism case, it is more difficult to state which scenario, optimistic or pessimistic better describes the actual trends in the researched countries, as we focused on willingness-to-pay estimates only and not on general economic value (including producer surplus). However, it appears so far that the current trends are closer to the optimistic than to the pessimistic scenario.

Other recent studies looked at the estimated financial

Table 5. WTP Estimates for Kenya, Zanzibar and the Seychelles (in million US$)

country	welfare loss in 2001	welfare loss 2001–2020
Seychelles	9.7	71.5
Mombasa	6.4	47.2
Zanzibar	5.4	39.9

losses in the Indian Ocean (Cesar *et al.*, 2000) from coral bleaching. That study showed that live coral cover in the Maldives was reduced from over 50% to less than 5%, though surprisingly, this has not lead to a major decrease in tourist arrivals. In fact, it was estimated that tourism growth was only 1% lower due to coral mortality. The main reason for this small decline is the successful shift the Maldives has made towards other types of tourism, especially ‘honeymooners’. In addition, with double digit annual international growth in the number of certified divers, and the relative proximity of the Maldives to the European market, this archipelago remains in a good position. It is guaranteed a fresh supply of relatively inexperienced divers mainly interested in charismatic marine megafauna which are readily available in the Maldives due to low reef fishing pressure.

This percentage decline for the Maldives is, however, rather low compared to declines found in East Africa. A study by Westmacott *et al.* (2000), based on tourist questionnaires, found a drop of 19% in dive tourists to Zanzibar, due to severe coral bleaching. This corresponds to an estimated decrease of around 10% of total tourist arrivals (Westmacott *et al.*, 2000). The difference between the Maldives and Zanzibar appears to be that the marketing possibilities of the Maldives allow for substitution among different sub-groups of the tourism market.

Besides these changes due to coral bleaching – small and large – global events such as the attacks on the US on September 11, 2001 have had a very large impact on the tourism sector in the Western Indian Ocean with losses corresponding to the pessimistic scenario albeit not bleaching related.

CONCLUSIONS AND FUTURE RESEARCH NEEDS

Summarising, the economic costs of the 1998 bleaching event in the Indian Ocean are still inconclusive. It may be that the time-lapse between the actual bleaching event and the surveys was too short or that other processes (such as changes in fishing pressure) obscure the analysis of bleaching impacts. This is especially true for fisheries. But, both for tourism and fisheries, the main difficulty lies with the fact that there are a number of different trends taking place at the same time. For example, within tourism, financial costs are currently severely affected by the post-9/11 economic downturn and drop in tourism arrivals. In fisheries, overall increases in effort are obscuring the bleaching impacts.

For tourism, both the analysis of welfare losses and the estimation of financial costs need to continue for more decisive conclusions to be reached. Key informant interviews in the tourism sector may be of more use than an examination of national statistics in this case. Furthermore, a study of the possible alternatives in coastal tourism is important to determine which areas are more prone to tourism losses than others.

For fisheries, more research is urgently needed. This is especially true given that corals killed after the 1998 bleaching event are beginning to disintegrate and a decrease in rugosity could lead to significant declines in fish catches. Also, more bio-physical research on coral recovery in bleached areas will be required. Notwithstanding the studies already carried out, few concrete results are available as yet.

ACKNOWLEDGMENTS

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Coral Reef Rehabilitation: Feasibility, Benefits and Need

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ABSTRACT

This study explores the benefits and feasibility of coral reef recovery through natural and artificial means, in Kenya, East Africa. Recruitment of new colonies to the reefs and recruit growth indicate the potential for natural recovery, while the survival and growth of transplanted fragments was used as an indicator for the potential for artificial restoration. Natural colonies fared much better than transplants in terms of both growth and survivorship, indicating the ecological benefit of transplanting corals is very small, especially in areas where they are most needed, i.e. in stressed areas where no or very low recovery has been recorded. We conclude that rehabilitation might be more efficient in the form of stress relief rather than through transplantation, that is, as protection targeted at priority areas. However, re-colonisation alternatives for areas showing low or no recovery needs to be explored further.

INTRODUCTION

The decline in the health of coral reefs caused by chronic stress such as overfishing and destructive fishing, increased sedimentation and turbidity due to high run-off from land caused by deforestation, and increased disturbance by pollution and litter has been continuing for decades in many areas, including Eastern Africa. This, in combination with perturbations such as the El Niño induced mass bleaching and mortality in 1997–1998, which was particularly destructive in the Western Indian Ocean, has further increased the threat to coral reefs in the region. At the same time, the dependence on coral reef resources is not declining. On the contrary, the coastal population that relies on reef based resources for their livelihood, directly or indirectly, is increasing.

In addition to illustrating the need for sustainable utilisation of coral reef resources and reduction in the effects of land-based activities on coral reefs, this also emphasises the need for activities that can halt and reverse the deteriorating trends. Among the proposed remedial actions is the active rehabilitation or restoration of damaged reef areas.

This ongoing study explores the benefits and feasibility of, and the need for, coral reef rehabilitation through comparing growth and survivorship in coral transplants with natural recruits and surviving pre-bleaching colonies in three areas along the Kenyan coast. Low-cost, low-tech methods were used to test and question the benefits of transplantation by weighing the costs and ecological impact of transplantation against that of natural recovery in the form of recruitment and re-growth of surviving colonies.

Recruitment of new colonies to the reefs and recruit growth indicate natural recovery is underway. Natural colonies fared much better than transplants in terms of both growth and survivorship, indicating the ecological benefit of transplanting corals is very small, especially in areas where they are needed the most, i.e. in stressed areas where no or very low recovery has been recorded.

Thus rehabilitation might be more efficient in the form of stress relief rather than through transplantation, i.e. as protection targeted at priority areas. However, recolonisation alternatives for areas showing low or no recovery need to be further explored.

MATERIALS AND METHODS

Sites

Four sites in the Bamburi-Shanzu lagoon, circa 10 km North of Mombasa Island, Kenya, were included in the study: Coral Gardens North, Coral Gardens South, Starfish and Ras Iwatine.

Ras Iwatine is located in Mombasa Marine National Reserve, where fishing using traditional methods is allowed and the fishing pressure is high, while the other sites are located in Mombasa Marine National Park, where all extractive use is prohibited. The sites are hydrographically similar and the depth is approximately 1 m at mean low water, with stronger tidal currents and indications of higher larval supply at Coral Gardens North and South (Tamelander *et al.*, in press).

Three sites in different parts of the Kiunga Marine National Reserve in the far north of the Kenyan coast were chosen for participatory coral transplantation: Shimo la Tewa, Mkokoni Bay and Boso. This area generally has higher nutrient concentrations and lower temperatures than Mombasa due to the interaction of two geotropic currents. The sites range from sheltered to exposed, with varying degrees of sediment input and turbidity. Fishing using artisanal methods is practised at all the sites, although fishing pressure is lower than at Ras Iwatine mostly due to the remoteness of the area. The tidal range is 4 m at all sites

All sites were heavily affected by the mass mortality in 1998, with an 80–95% reduction in coral cover (Obura, 1999; Obura *et al.*, 2000). Some signs of recovery have been seen since then as an increase in live coral cover, but there is also some formation of rubble caused by bio-erosion and breakdown of dead coral structures.

Transplantation

In Mombasa, transplantation was carried out at North Coral Gardens. Fragments of seven scleractinian corals species were transplanted using two methods, and growth and survivorship was measured over a period of one year.

The species used were *Echinopora gemmacea*, *Goniopora* sp., *Hydnophora microconos*, *Pavona decussata* as well as massive *Porites lutea* and branching *P. nigrescens and P. harrisoni*. Fragments of circa 15–30 cm^2 were cut from large source colonies and, using epoxy putty, cemented to either a cleared natural substrate (coral skeleton) or slightly elevated racks. Five colonies of each species were transplanted on natural substrate, distributed over three racks and five plots. Some replacements were made after losses or mortality. The initial size of the transplants was measured a few days after transplantation, after which growth was measured monthly for 2–4 months and later approximately once every three months.

In Kiunga, a participatory approach was used, and a transplantation team consisting of one scientist, four fishers, three Kenya Wildlife Service rangers and two WWF staff was formed from a group of people involved in bio-physical monitoring in the area. General information about corals, their reproduction, growth and life span was given before the sessions and during the work, contributing to general knowledge of coral reefs among the participants. Species were chosen based on availability, hence the selection of the three massive species *Porites lutea*, *Goniastrea retiformis*, *Platygyra* sp. and the encrusting/sub-massive *Hydnophora microconos*. Fragments of corals were cut from source colonies using hammer and chisel at a maximum of two transplants from each source colony. Based on experiences from the transplantation experiments in Mombasa Marine National Park the size of the transplant was kept large, >15 cm^2 and at times >30 cm^2, as smaller transplants have exhibited lower growth rate and survivorship (Tamelander *et al.*, in press).

Since Kiunga is remote, with low or no availability of transplantation materials, ans also to keep the cost of transplantation at a minimum, building cement was chosen for fixing transplant to the reef substrate. This allowed for use of mostly local materials with no need for expensive equipment or machinery. Dead corals were cleaned using wire brushes and transplants were fixed using a cement-sand mixture (approximately 1:1) as some cement gets washed away by water movements while fixing the transplants and before the cement hardens. Approximately 100 transplantations were made at each of the three sites, placed in plots of 5–10 transplants over approximately 1 m^2.

One person was identified as 'warden' for each site, with a responsibility to visit sites regularly to look for signs of mortality, predation etc. Another major task was to communicate the initiation and aim of the project to local fisher communities as well as report on progress.

Recruitment

At the four sites in Mombasa, 35 permanent quadrats were established, and maps prepared of all coral colonies and other features in the quadrats. Individual recruits and surviving pre-bleaching colonies were measured quarterly during one year

Growth and Survivorship

In natural recruits, surviving colonies and transplant maximum diameter and maximum perpendicular diameter were measured to the nearest millimetre from edge to edge of the living part of the colony using a flexible tape measure (transplants in Mombasa were measured using callipers). Surface area was estimated to be that of a circle with a diameter equal to the average of the maximum diameter and its maximum perpendicular (Obura, 1995). Cumulative growth over time was calculated, from which growth rate was estimated as rate of increase in area per unit area per unit time, or $mm^2*mm^{-2}*d^{-1}$, using the median of the areas and number of days between subsequent measuring events. By measuring growth per unit area, growth rates are comparable between colonies regardless of colony size, and provide better indications of their relative importance in the recovery process of coral reefs.

Survivorship was measured as the percentage of individuals surviving from one measuring event to the next. Growth and survivorship in recruits, surviving colonies and transplants were analysed separately.

RESULTS

Growth Rate

At an overall average growth rate of 0.0039 $mm^2mm^{-2}day^{-1}$ (±0.00021 SE) growth rates in coral recruits were approximately twice as high as in surviving colonies with an average growth rate of 0.0019 $mm^2mm^{-2}day^{-1}$ (±0.00035), and more consistent. There was considerable variation in growth rates between sites in surviving colonies (Fig. 1). Recruits were also less prone to negative growth, with an occurrence of negative growth in 6% of the recruits and 11% of the surviving colonies. The highest growth rates (>0.005 $mm^2mm^{-2}day^{-1}$) were recorded in *Echinopora*, *Galaxea*, *Acropora*, *Pocillopora* and *Porites* recruits and surviving *Montipora*, the lowest in surviving *Galaxea* and *Pavona* colonies, exhibiting average growth rates below zero.

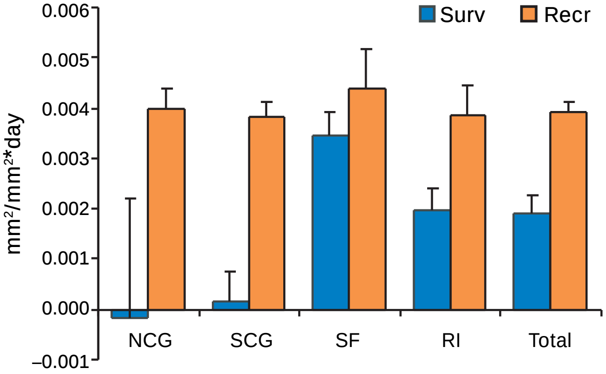

Figure 1. Average growth rate in coral recruits (Recr) and surviving colonies (Surv) at North Coral Gardens (NCG), South Coral Gardens (SCG), Starfish (SF) and Ras Iwatine (RI), as well as the overall total. Growth rate is presented in $mm^{2}*mm^{-2}*day^{-1}$, error bars indicate standard error of the mean.

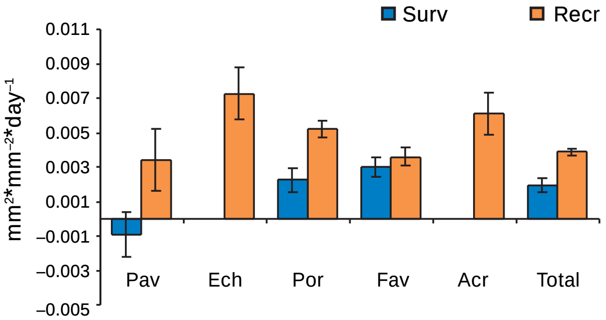

Figure 2a. Average growth rates of coral recruits (Recr) and surviving colonies (Surv). Genera from left to right: *Pavona, Echinopora, Porites, Favites* and *Acropora* as well as overall Total (the total includes growth rates in 31 genera from 13 families). Growth rate is presented in mm²*mm⁻²*day⁻¹, error bars indicate standard error of the mean.

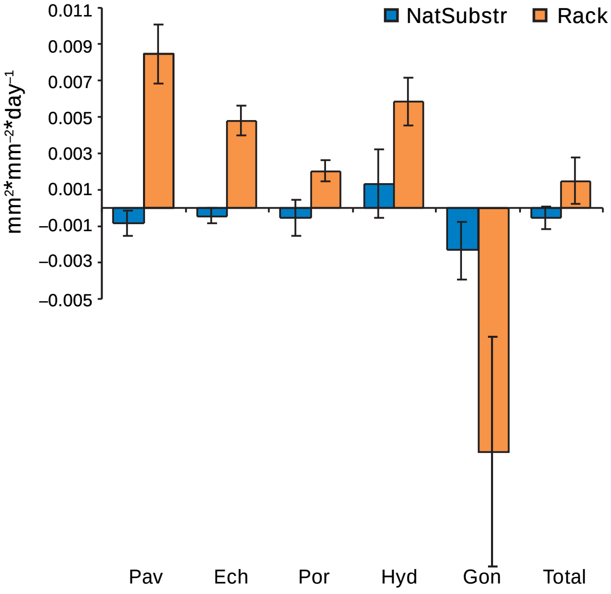

Figure 2b. Average growth rates of coral transplants on natural substrate (NatSubstr/dark) and racks (Rack/grey). Genera from left to right: *Pavona, Echinopora, Porites, Hydnophora*, and *Goniopora* as well as overall Total. Growth rate is presented in mm²*mm⁻²*day⁻¹, error bars indicate standard error of the mean.

Among transplants the highest growth rates were recorded in *Pavona* and *Hydnophora* on rack (0.0085 $mm^2mm^{-2}day^{-1}$ ± 0.0020 and 0.0058 $mm^2mm^{-2}day^{-1}$ ± 0.0013 respectively), the lowest in *Goniopora* on racks (–0.0134 $mm^2mm^{-2}day^{-1}$ ± 0.0063), while the average growth rate for all genera was 0.0015 $mm^2mm^{-2}day^{-1}$ (± 0.0012) on racks and –0.0005 $mm^2mm^{-2}day^{-1}$ (± 0.0006) on natural substrate. Negative growth was recorded in 32% of the transplants on natural substrate and 24% on racks. Growth rate in natural colonies and transplants is presented in Fig. 2a and b.

Cumulative Growth

Over a period of 13 months the total surface area of natural coral colonies increased by 231% in recruits and 31% in surviving colonies (Fig. 3). There was considerable variation between different genera and individual colonies, with a higher increase in surface area and stronger variability over time in *Pocillopora* and *Acropora*, while growth was generally slower and more stable in the Faviidae and Poritidae.

There was a decrease to 59% of initial area in transplants on natural substrate and 73% on racks 300 days from transplantation (Fig. 3). After 600 days the remaining area was 37% and 9% respectively. Higher fluc-

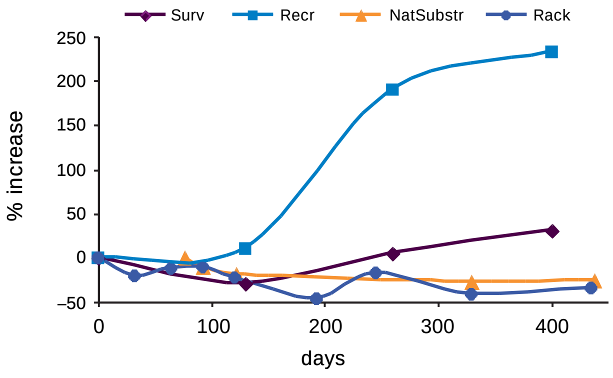

Figure 3. Cumulative growth in surviving colonies (Surv), recruits (Recr), transplants on natural substrate (NatSubstr) and transplants on racks (Rack). Growth is expressed as percent increase in surface area.

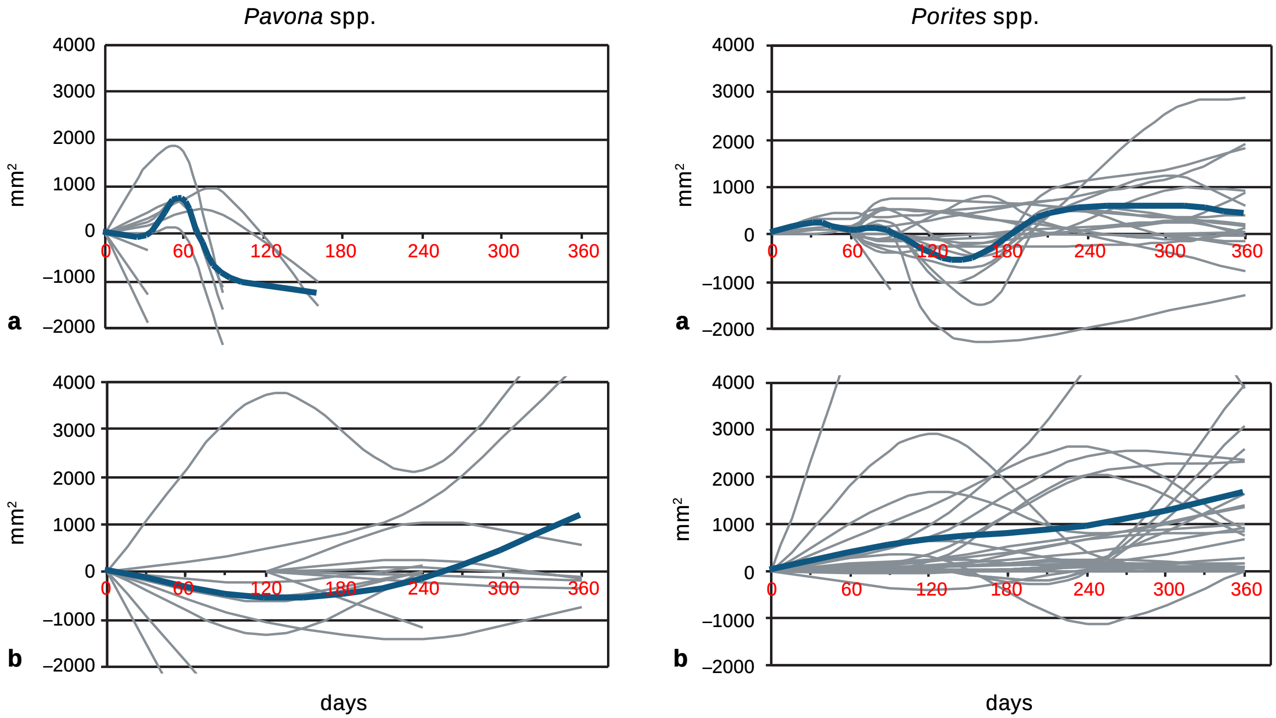

Figure 4. Cumulative growth in *Pavona* spp. and *Porites* spp. in a) transplants and b) recruits. The thin grey lines illustrate individual colonies, the thick line average cumulative growth. The graphs are based on four measurements of 45 recruits and 9 measurements of 32 transplants. Transplants on racks and on natural substrate are included, as well as both the massive and branching growth forms of *Porites*.

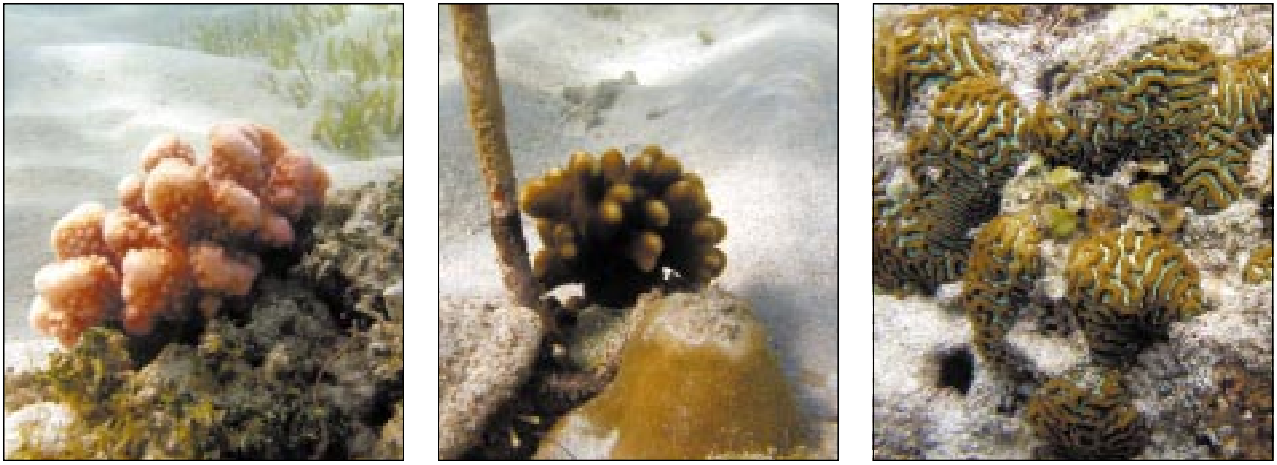

Figure 5. Photos of a) a *Pocillopora* recruit, b) transplants of branching and massive *Porites* and c) fragments of a surviving *Platygyra* colony. Note the large amounts of sand in a) and c), as well as the sand and sediment accumulated in the algal turfs in b) and c). Degradation of dead coral through bioerosion and wave action has led to increased sand and rubble formation, increasing stress through abrasion and suffocation. *Photos:* J. TAMELANDER 2002.

tuations in living tissue area over time were observed in transplants on racks.

Variation in growth between individuals was high in both natural colonies and transplants, and most genera exhibited higher cumulative growth in recruits than in transplants (Fig. 4).

Survivorship

Survivorship in natural colonies was high and virtually the same in recruits and surviving colonies, with an overall survivorship of 81% and 83% respectively. However, there were stronger variations between genera and over time, with higher mortality in branching recruit genera such as *Pocillopora* and *Acropora*, especially between October and February when survivorship dropped from 65–80% to 40–50%, and more stable and consistently over 80% in some predominantly massive genera such as *Favites* and *Porites*.

Transplant survivorship varied considerably between genera and methods. Three hundred days after transplantation 56% of the transplants on natural substrate and 42% of the transplants on racks remained. Six hundred days after transplantation only two transplants on the racks were alive, or 6% of the initial number, and 10 transplants or 31% of the initial number on natural substrate. During the first year after transplantation survivorship was higher in *Porites* and *Hydnophora* than in *Echinopora*, *Goniopora* and *Pavona*. Although growth rate was initially high in *Pavona* transplants on racks all but one had died within 6 months of transplantation.

Recruitment

There was a net influx of 3 recruits per m^2 every three months, ranging from 0 to 8.2 between different time intervals and sites. Recruitment was consistently higher at Coral Gardens North and South than at Starfish and Ras Iwatine.

Transplantation in Kiunga

Quantitative data on transplant growth and survivorship in Kiunga has not been obtained due to logistical problems. However, the survivorship was estimated at 75% or above after the first six months, with visible growth at least in *Porites*. Early 'mortality' appeared to be primarily due to loss of transplants rather than death. *Hydnophora* and *Platygyra* showed some susceptibility to tissue death at the edges where the transplants had been cut.

Costs of Transplantation

Based on the transplantation activities in Mombasa and Kiunga the cost was calculated for 50 small and 50 large transplants using epoxy and 300 large transplants using cement, taking into account the differences in crew, transplant numbers, transportation and materials used. The price per transplants was approximately US$ 1.95 using epoxy and US$ 0.71 using cement (Table 1).

Table 1. The cost of a) 50 small and 50 large transplants in Mombasa and b) 300 large transplants in Kiunga.

a			**b**		
Material	**Units**	**US$**	**Material**	**Units**	**US$**
Epoxy Putty	10 packs	34.62	Cement	1 bag	12.82
Racks	1 set	25.64	Sand	1 bag	2.56
Tools	1 set	19.23	Tools	1 set	19.23
Fuel	4 trips	64.10	Fuel	3 trips	115.38
Allowances	8 man days	51.28	Allowances	20 man days	64.10
Total		194.87	Total		214.10
Per Transplant		1.95	Per Transplant		0.71

DISCUSSION AND CONCLUSIONS

Recruits exhibited higher growth than surviving colonies, both of which fared considerably better than the transplants in terms of growth rates, survivorship as well as cumulative growth. The reason for this difference may lie in the manipulation of the transplants, i.e. the stress of cutting the transplants, making them vulnerable to predation and diseases especially at the damaged edges of the fragments. The higher growth rate in transplants on racks was probably caused by lower algal competition and lower accumulation of sediment in algal turfs adjacent to the coral tissue as well as less abrasion from moving sediments, while the lower survivorship could be an effect of their clustered distribution, making them more sensitive to predation or localised disturbances. All *Hydnophora* transplants on the racks, which exhibited among the highest growth rates and survivorship recorded in transplants, died circa two years after transplantation as a result of sand accumulation in a piece of cloth stuck on the rack covering the transplants.

In addition to the stress of cutting and cementing, transplants are affected by the same stresses as natural recruits and surviving colonies, implying that where background coral growth is poor transplant growth is also likely to be poor. Where re-colonisation is inhibited only by lack of larvae or by low settlement/recruitment success, viable populations could possibly be brought in through transplantation, provided a sufficient source population is accessible. In the studied area, there is an abundant supply of larvae (Tamelander *et al.*, in press) and high settlement and recruitment is underway with densities of over four recruits per m^2 recorded at all sites and over 20 recruits per 1 m^2 at Coral Gardens South. The species diversity of transplantation can also not be as high as in a natural population – while nine species were used for transplantation 31 genera from 13 families were recorded among the recruits.

It is possible that in some areas coral growth is not inhibited *per se*, but settlement and recruitment is inhibited due to low substrate availability or high mortality in newly settled larvae, particularly in high-stress environments where moving sand or rubble cause abrasion. Since rubble formation is increasing in many areas due to mortality following mass bleaching in 1998 (Turner, 2000) longer survivorship and growth studies are needed to determine whether the present level of recruitment is sufficient for reef recovery.

It is uncertain how well growth rates in different genera of natural coral colonies and transplants reflect the actual significance of their contribution to the overall recovery of the reef, as growth rates and recovery are dependent on a number of factors. For example, the strength of the skeleton is related to the rate of aragonite deposition – faster deposition creates a weaker skeleton. Further, faster growing species tend to be more vulnerable to perturbations because of a trade-off between growth and stress resistance, which was seen in the high mortality among generally faster growing, branching genera such as *Pocillopora* and *Acropora* during the mass-mortality (Obura, 2001; McClanahan *et al.*, 2001). Thus, fast recovery from the mass mortality can leave the reef more vulnerable to stress until the species composition and the reef ecosystem in general has recovered sufficiently to reach a balanced community structure.

The relatively short time since transplantation and the lack of quantitative data from Kiunga prevents wide-ranging conclusions, but the method used there appears better for its simplicity and community involvement. The strength of the cement over time is questionable, but it appears to be strong enough to allow coral fragments to grow onto the underlying substratum before it is weakened and eroded away.

The methods used were relatively cheap, although the initial cost per transplant increases significantly if survivorship of the transplants is taken into consideration. From an initial cost of US$ 1.95 per transplant using epoxy, the cost had increased to US$ 3 per surviving transplant after one year. The cost per surviving transplant in Mombasa after 600 days was US$ 8.10. The method used in Kiunga would be cheaper, circa US$ 5.30 per square meter using the method and transplant densities described, rising to US$ 8.20 after a year based on an estimated survivorship rate of 65%.

Some studies in the region have, however, reported higher success rates using different species and methods (Muhando, unpublished data; Lindahl, 1998, 2000), although the transplants still were as vulnerable to major perturbations as the surrounding reef. One potential use of transplantation is in small-scale 'cosmetic' improvements, particularly in the rehabilitation of craters caused by dynamite fishing (Wagner, in press).

The results from this study indicate that restoration is not feasible using the methods described, nor is it needed, as growth and survivorship in natural colonies far exceeds those observed in transplants. Additionally, the disease that caused coral mortality on Kenyan reefs in February–April 2002 is likely to have affected transplants as much as natural colonies. It is also obvious that the geographic extent of the coral mortality of 1998 and its severe impacts in some areas renders large-scale restorative interventions economically impossible, and transplantation would be ecologically questionable as the coral community is severely depleted. Further, many of the reefs most affected by the mass mortality in 1998 are located in developing countries, where funds available for protection and rehabilitation of coral reefs are very limited but the dependence of the population on reef resources high. Because of this it is vital that appropriate methods and technologies are used that will maximise the effect of the input. With these limitations of restoration it appears protection and management of reefs and their associated ecosystems, as well as the areas that influence them, is a better investment in the long-term.

There was, however, some benefit in the transplantation activities of this study, although not so much in the rehabilitation of a reef area, but rather in the use of transplantation as an awareness-enhancing tool through the participatory approach of the study. There was a noticeable increase in the knowledge of coral reefs and their importance among the team members, and through communicating the aims of the activities to local communities some of the transplantation sites were actually closed from fishing, reducing a major stress in the area. This aspect of rehabilitation and management needs to be further developed as, besides poverty, lack of awareness is one of the primary threats to the health of coral reefs. However, when initiating rehabilitative efforts on coral reefs it is imperative that the objectives are relevant, clearly defined and achievable, that outputs match the objectives and that the cost per output is kept within acceptable limits.

ACKNOWLEDGEMENTS

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UV-Radiation and Recruitment of Microalgal Assemblages in a Coastal Lagoon

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Key words: Kenya, micro algae, micro algae recruitment, uv- radiation

ABSTRACT

A field experiment was carried out in November 2001 to investigate the effect of UV radiation on the micro algal recruitment patterns in the Mombasa Marine Park, also known as Bamburi lagoon. It involved both manipulations of grazing (consumption) and light regimes on settlement tiles as proxies for newly dead corals offering new substratum. It was hypothesized that the relatively clear and shallow waters of the lagoon ensured that at least 30% of the surface (0.36 Wm^{-2}) UV-B irradiation penetrated down to the bottom of the lagoon (c. 3 m) where the experimental block was placed. There were no statistical differences in the abundance of microalgal recruitment in the different light treatments. However the total abundance decreased in the following order: uncaged > caged > half caged with total means of 3500±1900>2000±800>1500±700 cells/cm^2 respectively. The generic richness was statistically different between the light treatments ($p=0.032$) with highest levels in the PAR+UVA+UVB and control (no filter) treatments as opposed to the treatments where UV-B and UV-A + UV-B were excluded. Colonization of the control tile surface by microalgae and sand deposits seemed to have increased competition among the different genera for light and resources hence favouring colonization of the more resilient genera at the expense of other competing genera, thus lowering generic richness. The combined effect of caging and manipulation of light regimes did not have any significant influence on the diversity ($p=0.320$), evenness ($p=0.323$) and richness (0.505). The experiment needs to be repeated on a large scale including seasonality as a factor, and with greater replication.

INTRODUCTION

The amount of UV-B (280–315 nm) radiation reaching the earth's surface is high near the equator throughout the year due to a naturally low ozone concentration, coupled with a low solar zenith angle (Madronich, 1992; Cutchis, 1982). The incoming UV dose per day depends on latitude, season, ozone layer condition, cloud cover, humidity, and other meteorological parameters. The penetration at the water surface is influenced by angle of sun and wave conditions. When penetrating into the aquatic medium the amount of UV radiation is attenuated by surface reflection, scattering by particles and absorption by dissolved organic carbon, phytoplankton and macroalgae and inorganic matter. Depending on the composition of seawater, substantial amounts of UV-B (>10% of surface irradiation) reach water depths where biological processes take place (Häder, 1995). UV-C (100–280 nm) and UV-A (315–400 nm) irradiation, unlike UV-B, are less affected by ozone but are also less hazardous to marine life.

The most generally accepted primary factor in geographically widespread bleaching is the elevated sea temperatures sometimes associated with the El Niño Southern Oscillation and global greenhouse warming (Glynn, 1993, 1996). However, the correlation is not exact, and elevated temperatures and conditions promoting them (e.g. low cloud cover) may interact with other factors such as increased penetration of solar radiation, including UV (280–400 nm) and PAR (400–700 nm) into clear seawater (Glynn, 1993). UV related bleaching was postulated to account for massive bleaching reported in the warmest months of the year in 1987 and 1990 in the GBR and the Caribbean because it occurred only on the upper and unshaded colony surfaces extending to deeper colonies in very clear waters (Harriott, 1985; Goenaga *et al.*, 1989). However, corals that contain zooxanthallae produce UV-B screening substances such as mycosporine-like compounds.

During the recent severe El Niño warming event of 1997–98, Mdodo (1999) observed that bleaching of corals in Bamburi lagoon, Mombasa was mainly due to expulsion of zooxanthallae rather than to reduction of chlorophyll-a per zooxanthallae. This is reported to be typical of UV related bleaching that occurs at depths even >10m (Gleason & Wellington, 1993) in sharp contrast to the co-occurence of zooxanthallae expulsion and decrease of chlorophyll per zooxanthellae recorded in temperature related bleaching (Glynn & D’Croz, 1990).

Holm-Hansen *et al.* (1993) and Helbling and Villafane (1992) reported a decrease of production, biomass and/or a change in species composition for phytoplankton communities, diatoms and macroalgae in marine benthic systems under natural UV-B irradiation levels compared to conditions where UV radiation is completely blocked. They studied the influence of UV-R on phytoplankton growth, reproduction and photosynthetic rates. Surprisingly few studies have investigated UV-R effects on the structure of communities or on multi-level trophic interactions.

It was shown from a recent study at Mombasa Marine Park that dead corals provided a substrate for the proliferation of microalgae assemblages both on their bare surfaces as well as on the macroalgal turf which colonized them after a bleaching event (Obura *et al.*, 2000). Settlement tiles were used in this research to act as new substratum (equivalent to dead corals) and to follow the micro algal spp proliferation dynamics in a controlled field experiment. A UV radiation component was incorporated within the study since microalgae recruitment strategies may be influenced by the magnitude of UV wavelengths, and differentially by UV-A and UV-B. The objectives of this project were to investigate the effects of PAR, UV-R and grazing levels by macroherbivores on the recruitment of microalgal assemblages in the Bamburi lagoon, Mombasa.

METHODS

Bamburi reef lagoon is situated north of Mombasa Island, along the Nyali-, Bamburi- Shanzu coastline, at 4° 01' S 39° 44' E 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. It is described in Kirugara (2000) and Mwangi *et al.* (2001).

Three main factors influenced the choice of this site. There is substantial background research carried out at the site. It is also expected that there is a high penetration of UV-B radiation due to the relatively shallow (<5 metres), clear and calm waters throughout the year. The experiment was performed in November 2001, during the North East monsoon when the lagoon waters are calm due to reduced wind and wave action. This is also the season of cloudless skies and hence it is expected that UV radiation can penetrate substantially into the warm lagoon waters. Finally, it is also expected that the biological diversity is high and so is the expected number of sessile organisms.

The experimental unit was a platform (132 x 66 cm) on which eight tiles and containers could be fixed, each under different treatment levels (Fig. 1). Colonization of microalgae was sampled on tiles (70 x 70 mm), held in containers adapted for each treatment factor. The tiles were fixed to the bottom of each container to allow for removal and replacement. Replication was achieved by repeating the experiment 4 times. The experimental unit was placed 30 cm from the bottom of the lagoon (c 2 m at spring low water).

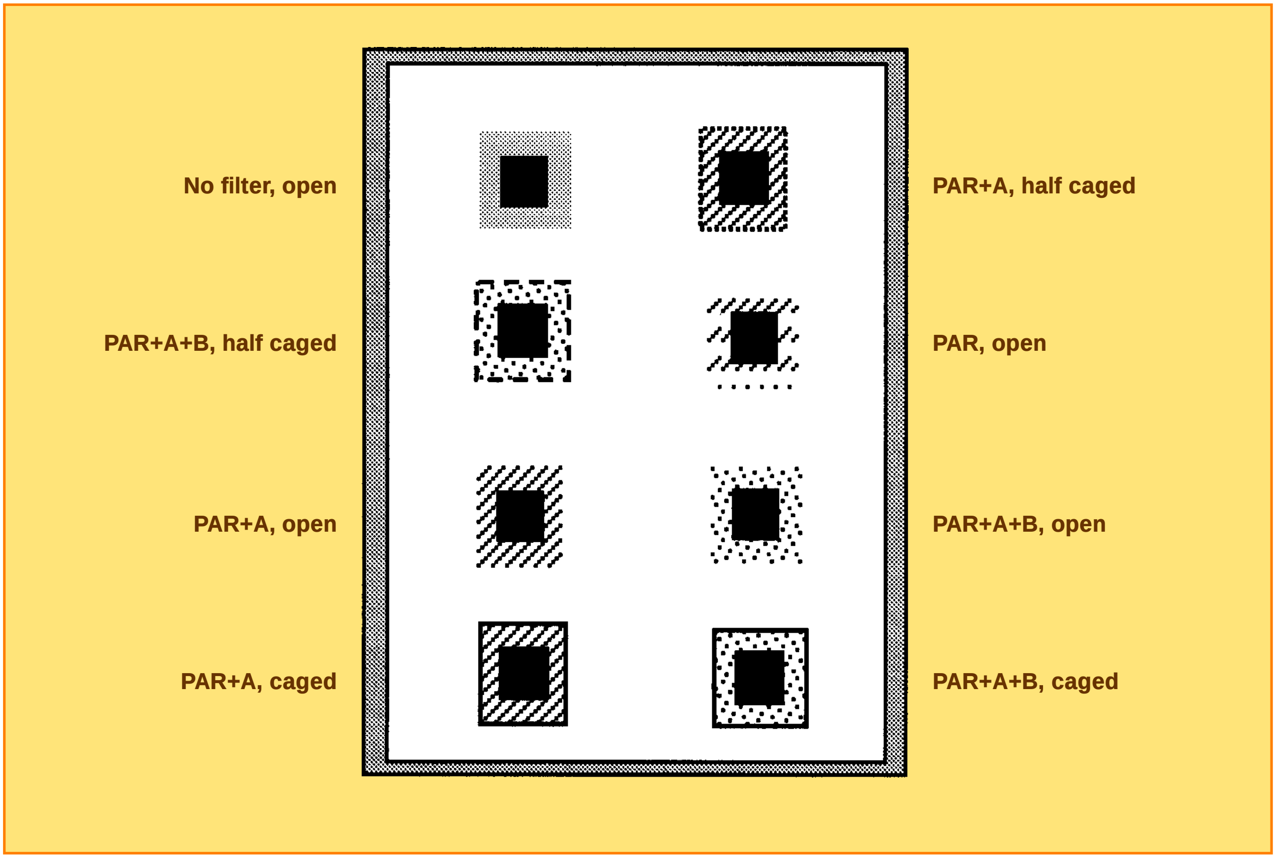

Figure 1. Schematic representation of the random experimental block design, measuring 132 x 66 cm.

The experiment involved manipulation of grazing (consumption) and radiation in a two-way factorial experiment. Three grazing treatments were set up. Uncaged containers had all sidewalls removed, partially caged containers (cage-controls) had one sidewall removed to allow access by consumers, and fully caged containers were completely enclosed to prevent access to the tiles by consumers. Fifty percent of the wall surface of the containers was perforated by 6 mm diameter holes to allow sufficient water flux for arrival of larvae. Radiation was manipulated by placing filters over the top of the containers (Table 1 and Fig. 1). There were four levels of light manipulation allowing: 1. PAR+UV-A+UV-B, 2. PAR+UV-A, 3. PAR only and 4. ‘no filter’. Panels under unfiltered radiation were used to control for fouling of filters.

On a sampling trip, all the tiles were removed from the experimental block and placed in labelled trays and brought to the laboratory as soon as possible. A visual observation was made on the substrate settled on the tile and recorded, excluding a 1 cm margin all around the periphery of the tile to cater for edge effects. Three sample guadrats per tile, of 1cm x 1cm cm were chosen randomly for quantitative inspection. All the organisms in each quadrant were scrapped off gently and placed in a labelled bottle containing a small volume of filtered seawater and preserved with formalin. A 5 ml sub-sample was placed in a Petri-dish with known number of circles

Table 1. Filter specifications for the experiment

Treatment	Filter material	Optical properties
PAR+UV-A+UV-B	3 mm thick Perspex (GS 2648 Röhm, Germany)	80–90% transparent to solar UV-radiation (290–400nm) and 90% transparent in the PAR region (400–700nm)
PAR+UV-A	3 mm thick Perspex (GS 2648 Röhm, Germany) covered by a 0.1 mm thick clear polyester film (LTF NashuaCopy)	Virtually opaque to solar UVB radiation between 290nm–313nm (i.e. <1% transmission) and its transmittance increased sharply to 70% at 320 nm. From 320–360 nm transmittance gradually increased to 85% and levelled off at 90% beyond 360 nm throughout the PAR region of the spectrum
PAR only	4 mm thick Makrolon (longlife plus 293, Röhm, Germany)	Blocked all (i.e. <1% transmission) of the UVB and UVA radiation (290–400nm). At 400 nm transmittance increased sharply to 90% and stayed at that value throughout the PAR region (400–700 nm).

(154) at a specific magnification for counting using an inverted microscope. The phytoplankton present were identified using appropriate identification keys.

The tiles and floats were completely cleaned during each weekly field visit. The data collected were pooled together into four replicates for each of the eight different treatments.

RESULTS AND DISCUSSION

During a two-month survey undertaken in February–April 2001, instantaneous UV-B, UV-A and PAR measurements were recorded at the Kenya Marine and Fisheries Research Institute, 3 km away from the study area using an UV data logger of the type LI-1400 (Walz GmbH, Germany) from 10:00–16:00 hrs. Fig. 2a (next page) shows the half hourly instantaneous measurements for solar (short wave) radiation (300–2500nm) and net (reflected long wave back) radiation (300–6000nm) obtained on 1st February 2001 from an Aanderaa Automatic weather station and Fig. 2b (next page) shows corresponding UV-B and UV-A measurements from a UV data logger. The UV-A sensor was faulty and hence did not record any data. The data shows that the intensity of UV radiation is a function of the total solar radiation with a peak occurring at 14:00 hrs after a one hour lag of the maximum solar radiation.

Analysis of the data collected during the two month period, though not coinciding with this sampling period, indicated that the mean values for the instantaneous fluxes at midday were 0.36 Wm^{-2} and 60 klx for UV-B and PAR respectively. Attenuation coefficients (kd) for UVB and PAR were calculated as 1.66 m^{-1} and 0.142 m^{-1} respectively for the location, which is in a mangrove creek (Tudor creek, Mombasa). This means that 81% of the surface UV-B irradiation was absorbed or scattered within the first 1 metre, compared to 14% of surface PAR. It is expected that the attenuation coefficient for the Bamburi lagoon waters are lower than the Tudor creek values hence 50% of the surface UV irradiation can penetrate down to the bottom of the lagoon (c. 3 m) due to the clarity of the water.

Total cover of microalgae on the tiles varied between 81% and 95% (Table 2 on page 277). All of the tiles with filters had 93–95% cover of microalgae, with the uncovered Full Sunlight treatment having 81%, due to higher cover of sand. Total abundance of microalgae decreased in the sequence uncaged > caged > half caged, with 3500±1900 >2000±800>1500±700 cell/cm^2 respectively (mean±std deviation). However, a one-way ANOVA indicated that there were no significant differences between treatments (p=0.724). Ten genera occurred in proportions >1% of the total abundance of

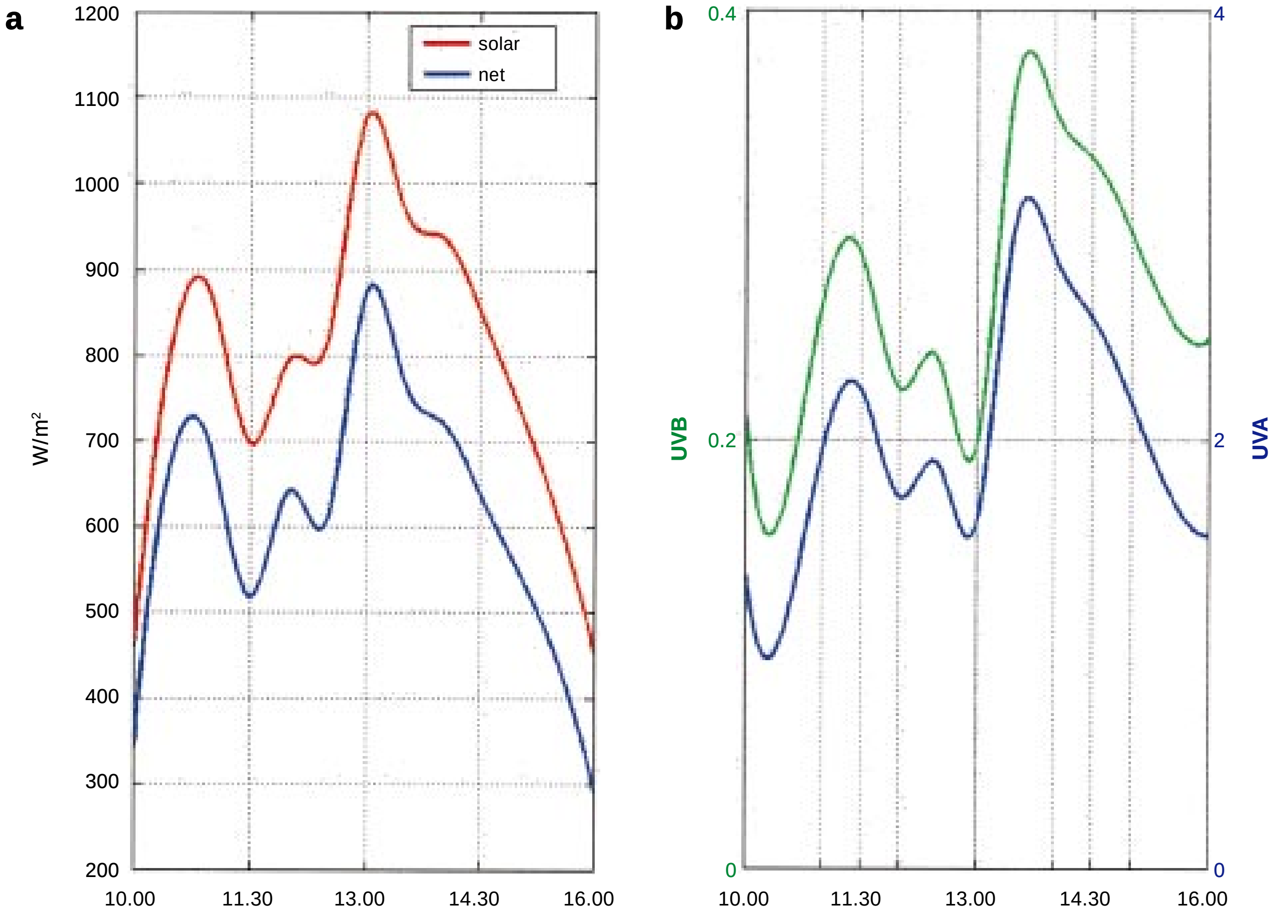

Figure 2. A six-hour record of solar radiation and net radiation (a) and simultaneous records of both UV-B and UV-A (b) obtained on 1st February 2001 between 10:00 and 16:00 hrs at the Kenya Marine and Fisheries Research Institute.

microalgae on the tiles. *Synedra*, was present in all the different light treatments, though *Navicula* and *Nitzchia* were the two most dominant genera, together accounting for >50% of the total cell counts (Table 2). One of the dominant genera, *Navicula* also constituted a major genus in the previous benthic survey at the same site (Obura *et.al* 2000). However, *Chaetoceros. was* noticeably absent, unlike in the previous study where its presence was significant. *Oscillatoria* and *Coscinodiscus* that were abundant during the previous survey occurred in less than 1% of the population.

There were no significant differences for diversity (H') and evenness (J) between the different light treatments within the uncaged (open) containers (One Way ANOVA, p=0.282 and 0.631, respectively). Diversity (H) was highest at PAR+A+B (1.718±0.231) and lowest at PAR (1.327±0.384) treatment effects (Fig. 3a). Evenness (J) was highest at PAR+A (0.765±0.070) and lowest at PAR (0.661±0.178) treatment effects (Fig. 3b). Generic richness was statistically different between the different light treatments (p=0.032), (Fig. 3c). Generic richness was highest in the PAR+UVA+UVB and the no filter (control) treatments as opposed to the treatments where UV-B as well as UV-A + UV-B were excluded.

Table 2. The percentage cover of the most abundant species recorded during the entire study period. FS: Full sunlight allowing PAR+UV-A and UV-B

Species	Uncaged				Half caged		Caged	
	FS	PAR	PAR+A	No filter	FS	PAR	FS	PAR
Amphora		3.5	1.6	13.3		2.1		4.2
Bacillaria				2.3	3.8	5.8	4.6	
Fragilaria			7.8		2.3			
Licmophora	14.3	6.6	6.2		3.0			
Navicula	26.1	28.8	28.7		28.8	37.9	31.2	25.4
Nitzchia	27.3	48.4	32.0		26.4	15.7	26.7	30.2
Oscillatoria	4.5	2.2	11.8					
Pleurosigma	2.3					2.8	4.4	
Prorocentrum					2.6		1.8	2.3
Sp x		2.7						
Synedra	6.9	3.3	6.9	.2	25.	29.3	25.4	31.1
%	81	95	95	94	93	94	94	93
Species richness	6	7	7	8	7	6	6	5

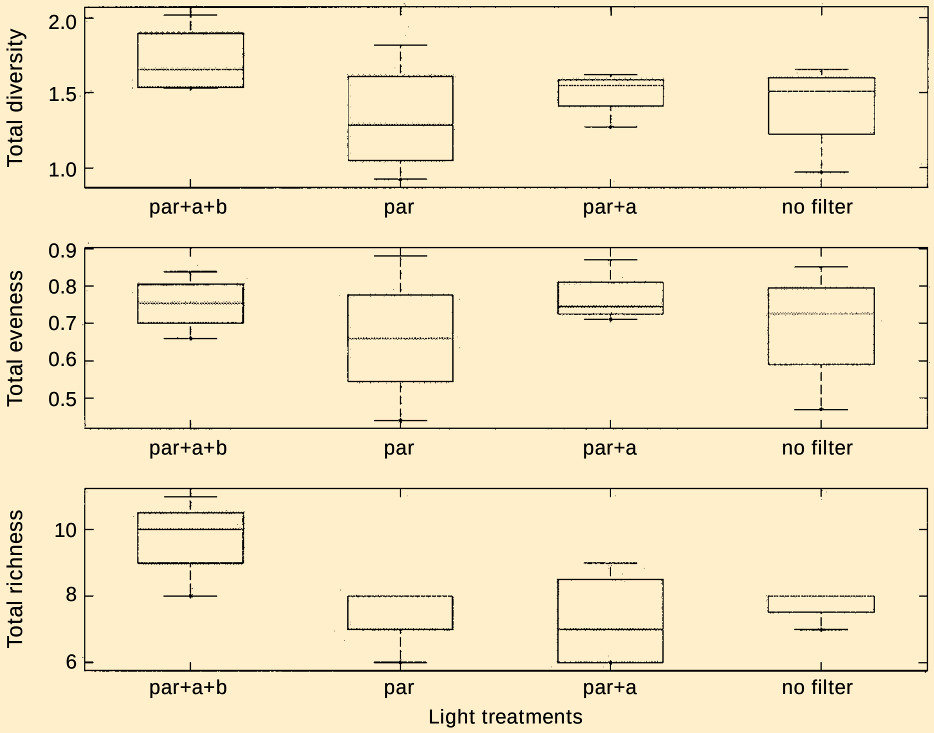

Figure 3. Box plots for a). Total diversity (H'), b). Eveness (J) and c). Richness (R) at different light treatments.

The lower generic richness on the No filter treatment compared to the PAR+UVA+UVB treatment could be due to the higher sand deposits, possibly excluding some genera by smothering and/or increasing competition for light and resources, favouring the more resilient genera at the expense of others.

A 2-factorial ANOVA was used to test the interaction of the two extreme light treatments (PAR+UVA+UVB and PAR) and three consumption levels (fully caged, half caged and uncaged) on diversity (H'), evenness (J) and genera richness (R). Diversity and genera richness were highest in treatments allowing consumers (1.522±0.360 and 8.625±1.598, respectively) than caged treatments (1.417±0.195 and 7.125±1.727). Evenness followed an opposite pattern, being highest in caged treatments (0.735±0.050) and lowest in the presence of consumers (0.708±0.014). Interaction effects of caging and light were not significant for any of the variables.

The results of the experiment are consistent with the literature on herbivore-algae impacts, where consumption increases diversity by preventing competitive exclusion of minor species. Nevertheless, we have some reservations on the design of the holes allowing grazers through the container side walls. It is possible that the holes were too small, resulting in the water carrying the grazers flowing around the cages as opposed to through the holes. Replication of the experiment was low and the time relatively short. The light filters should also be cleaned more frequently. This experiment needs to be repeated on a larger scale taking seasonality as a time factor, and with more experimental blocks for increased replication for better statistical analysis.

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