

KENYA, REEF STATUS AND ECOLOGY

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INTRODUCTION

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

BENTHIC COMMUNITIES

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

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

Monitoring of reefs in northern parts of Kenya (in the Kiunga Marine Reserve) and in a few fore reef sites in both northern and southern parts of Kenya has been established more recently. Monitoring in the remote northern sites has involved collaborations among many organisations, including, Kenya Wildlife Service, WWF, UNEP, FAO, CRCP, and recently, CORDIO (reported in Obura *et al.*, 1998). Transect surveys of deeper reef sites in the Malindi and Watamu areas are based on sites established in Obura (1995). Coral mortality shows the same patterns at these additional sites (Figure 1b, c, d), with coral cover falling from levels of 30% (southern, deep reefs) and approximately 10% (northern reefs) to about 5% at all sites. Lower pre-bleaching coral cover on northern Kenya reefs is thought to be due to a gradual transition from the coral-reef dominated East African fauna in the south, to the cold-water influence of

the Somali upwelling system further north (Samoilys, 1988; McClanahan, 1988; 1990). While coral bleaching and mortality in shallow waters in this northern region were as dramatic as southern areas, deeper corals suffered less mortality (though > 50% bleaching was observed, pers. obs., J. Church, pers. comm.), and the small losses in coral cover were not distinguishable due to the differences in methods used in the before/after surveys.

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

CORALS

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

Almost all of the coral species that suffered high bleaching and mortality rates are fast growing with branching morphologies, which enable them to dominate reef communities through sexual and asexual reproduction and competitive overgrowth. Many of the surviving species are massive or sub-massive in morphology and slower growing, or only attain small sizes and don't tend to dominate reef communities. These patterns in the bleaching response of coral species are reflected in cluster analysis groupings from a subset of 16 coral species and one soft coral genus (*Sinularia*) from the same data set. The primary division separates the fast growing, low resistance species, group A, from the others (Figure 2), with the main distinction between groups B and C being in higher mortality but less tendency to bleach, and also higher levels of variation in response, in group C. The divisions among the groups are somewhat consistent with differences in coral species resistance to other stresses, such as sedimentation (Obura, 1995) and in studies of coral life history strategies (Hughes & Jackson, 1985; Kojis & Quinn, 1991). These patterns offer hypotheses for attempts to understand the ecological and evolutionary effects of El Niño-related and other stresses on the long-term prospects for coral reef survival. It also offers the possibility of using a suite of species with known responses as indicators for reef status either through observation of natural corals, as analysed here, or observation of transplants as bioassays (Obura, 2000; this volume).

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

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

ZOOXANTHELLAE AND CHLOROPHYLL IN CORALS

Research on zooxanthellae and chlorophyll concentrations in Kenyan corals started in late 1997 just prior to the El Niño (Mdodo, 1999; Mdodo & Obura, 1999), and sampling has continued almost continuously since then. Five coral species were selected for monitoring (Figure 4), with 'normal' zooxanthellae and chlorophyll concentrations of 1-5 million/cm² and 1.5-8.0 mg/cm² respectively, with consistent species specific differences for zooxanthellae numbers. Sampling of bleached colonies of two of the species showed significant decreases in both the number of zooxanthellae and the concentration of chlorophyll in both species. Ongoing monitoring of these variables is being conducted to establish a good baseline of data and to maintain capacity to sample in a subsequent bleaching event.

MACROALGAE

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

There were more taxa and higher cover of macroalgae at Ras Iwatine compared to Starfish, though the cover of most species remained below 10% (Figure 5). *Padina*, *Sargassum*, and *Dictyota* were the most prominent algal genera, in common with other Kenyan reefs (McClanahan, 1997). Also, macroalgal populations were higher on the unprotected reef, Ras Iwatine, where herbivory by sea urchins would be higher (see Bioeroders section), also indicated by the high cover of herbivore-resistant encrusting coralline algae (Figure 5, McClanahan, 1997). At the Starfish site, all macroalgae remained low till November 1999 where *Padina bega* to increase at the onset of the north-east monsoon (Figure 5).

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

MICROALGAE

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

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

BIOERODERS

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

The study sites show typical sea urchins communities for Kenyan reef lagoons dominated by diadematids (especially *D. savignyi*), *E. mathaei* (Echinometridae) and *Tripnuestes gratilla* (Toxopneustidae) (Table 3). Sea urchin population densities in the marine park show considerable reduction compared to past years (McClanahan & Shafir, 1990), possibly due to increases in predation as fish populations within the marine park mature following 10 years of protection. These preliminary results show that *D. savignyi* makes the greatest contribution to overall bioerosion, with the highest gut sediment content (37%) followed by *D. setosum* (28.1%) and *E. mathaei* (25.6%) (Table 4). However, the method used in this study did not differentiate between sources of carbonate sediment, which included both corals and coralline algae. The internal bioeroder community was dominated by sponges (32.3%), polychaetes (14.5%), bivalves (22.8%) and sipunculid worms (16.8%) (Figure 8). Risk *et al.* (1995) contend that morphology of corals may be an important determining factor in total bioerosion levels. More holes were recorded in *Porites* skeletons, however the bioeroder community was similar for both *Acropora* and *Porites*.

ACKNOWLEDGEMENTS

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

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