

CORAL AND ALGAL RESPONSE TO THE 1998 EL NIÑO CORAL BLEACHING AND MORTALITY ON KENYA'S SOUTHERN REEF LAGOONS

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INTRODUCTION

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1987 and 1994 with the 1987 event causing significant mortality in corals and other benthic invertebrates (McClanahan, unpublished data and observations).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

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

METHODS

Study sites included nine sites in the four MPAs, Malindi, Watamu, Mombasa and Kisite Marine National Parks (MNP) and seven sites in four unprotected reefs, Vipingo, Kanamai, Ras Iwatine, and Diani. Sites within a reef are often separated by 20 m to 100 m and the reefs are

distributed along 190 km of coastline with distance of 3 km to 50 km between reefs. Sites were in back reef lagoons with shallow water (< 3 m) at low tides (Kenya has a 4 m tidal range) dominated by hard substrate colonised by corals and other benthic invertebrates and algae (McClanahan & Shafir, 1990). Benthic cover, sea urchins, and major fish groups have been monitored in these sites since 1987. Artisanal fishing, which dominates the fishery on the Kenyan coast, is mostly restricted to these shallow reef lagoons in Kenya.

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

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

Sea urchins, surgeonfish and parrotfish were censused in each of the study sites both before and after the bleaching event to estimate their wet weights and rates of consumption. Sea urchins were identified to species and counted in nine to 12 10 m² circular quadrats in each study site. The biomass of each species was calculated by multiplying the average wet weight (McClanahan, unpublished data) with the average density of each species, and the total weights of each species were summed to estimate the total sea urchin wet weight. Consumption rate studies of the four most common species of sea urchin undertaken on Kenyan reefs indicate that the average daily consumption for these four species is ~1.6% of their wet weights (McClanahan & Kurtis, 1991; Silva, 1999). Consequently, we multiplied the total sea urchin wet weight (kg/ha) by 0.016 to estimate sea urchin daily consumption in kg/ha/day.

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

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

RESULTS

Sea water temperatures

Sea water temperature data indicate that average midday water temperature during March 1998 was between 30° C and 31° C (Figure 1). The NOAA satellite data suggest that this was the warmest month on record for the Malindi site with an elevation of 1° C to 1.5° C for the month of March and April. 1998 was also unique in that there was only a very small drop in water temperature during the onset of the north-east monsoon winds, usually between January and February, prior to the maximum temperatures in March. The satellite and logger data were well correlated, although the intercept was not zero, nor was the slope 1, which suggests that the logger was less responsive than the satellite data to temperature changes.

Post-bleaching benthic surveys

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

Analysis of the benthic cover approximately one year after the bleaching event comparing 12 sites for the relationships between the loss of coral cover, the estimated daily consumption rates of algae by herbivores and the increase in fleshy algae indicate that there was no clear relationship between the change in fleshy algae with the change in coral, but a weaker relationship with the estimate of herbivory (Figure 2). Some reefs appeared to fit a clear negative relationship with the change in algae and consumption estimates (Ras Iwatine, Diani 2, Mombasa 2, Malindi 1 and 2, and Diani 1) while another group of sites that experienced only a small or no increase in fleshy algae had low to moderate levels of herbivory (Vipingo, Kanamai, Mombasa 1 and Watamu). Consequently, six of the 12 sites experienced an increase in fleshy algae equal to or more than 10 cm/m over the year, but there was no clear distinction between protected and unprotected reefs.

DISCUSSION

The warm north-east monsoon of 1998 was unique in being one of the strongest El Niño events since 1877/78 (McPhaden, 1999). Both NOAA satellite data and our *in situ* logger suggest that

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

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

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

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