

Sustaining Coral Reef Ecosystems and their Fisheries in the Kiunga Marine National Reserve, Lamu, Kenya

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Abstract Coral reefs in the Kiunga Marine National Reserve (KMNR) (40° 07' E, 2° 00' S) are located in a transition ecotone between the warmer East African coral reef bioregion to the south, and colder waters of the Somali Current to the north. The reefs have been monitored annually from 1998 to the present, documenting a range of ecosystem changes from large and small scale threats. Reefs in the area suffered ~60% loss of coral cover due to mass bleaching in the 1998 El Niño event, and 25-40% loss of coral species at individual site levels. Recovery of coral community structure has been variable, with some reefs showing strong recovery, while others have declined further. A harmful algal bloom and coral disease in early 2002 further impacted these reefs, causing mass mortalities of fish and coral, and failure of coral recruitment in that year. Fishing impacts to the reserve are high, with a strong south-north decline in fish density due to easier access to the migrant and large fishing communities to the south of the reserve. Responsibility for management of the KMNR falls under multiple institutions, including the Kenya Wildlife Service, Fisheries and Forestry Departments, and the local council. Overlapping mandates, unclear relationships, limited information and understanding, and lack of resources have hampered effective management. The monitoring programme reported here is one aspect of new collaborative approaches to coral reef and fisheries management, and has focused on improving the information and understanding of the biological and resource systems of the area. The ecosystem trends induced by larger scale threats and the south-north fish resource gradient caused by local use patterns will be analyzed in an attempt to develop sustainable management practices for the reserve.

Keywords: Status of reefs, benthic cover, fish densities, El Niño, harmful algal bloom, Kiunga Marine National Reserve

Introduction

The Kiunga Marine National Reserve (KMNR) lies in the Bajun archipelago at the northern extreme of the Kenya coast, at the border with Somalia (Fig. 1). It gained protection status in 1979, allowing traditional use of marine natural resources through direct management

efforts started only in the mid 1990's with the development of a co-management initiative (Gubelman and Weru 1996). The management mandate for the reserve lies with the Kenya Wildlife Services (KWS), and the mandate for managing the fisheries remains with the Kenya Fisheries Department. Reserve and fisheries regulations exist as a component of the country wide protected areas and fisheries regulations under the Wildlife and Fisheries Act.

The reserve extends over some 50 km in length by 3-5 km in width (from about 1°42.25'S 41°31.78E to 2°2.58'S 41°14.80E). The area is characterized by a linear series of barrier islands sheltering extensive mangrove stands in the protected lagoons, large areas of seagrass on rocky substrates on the shallow outer fringing reefs, and a submerged barrier rock reef some 3-4 km offshore (Weru 1991, 1996).

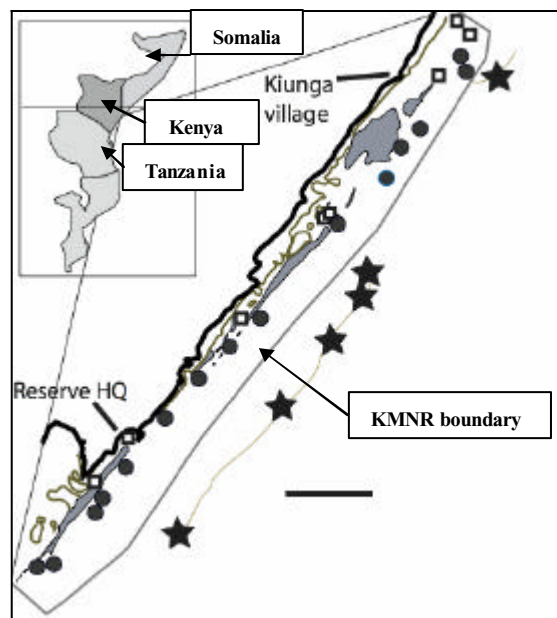


Fig. 1. Map of the KMNR located just south of the Kenya-Somalia border, showing the Reserve headquarters in the south, and Kiunga village in the north. The approximate seaward boundary of the MPA is shown by the straight lines. Study sites are shown for inner (open squares), outer (circles) and deep (stars) habitats, the latter lying outside the reserve boundary. North is upwards, and scale bar is approx. 5 km.

The KMNR has high biodiversity importance due to nesting activities of marine turtles and migratory birds, the presence of whales and some of the largest area of mangroves in Kenya, and a series of patch and fringing coral reefs around the barrier islands and on the offshore rocky rock reef (Obura 2001). Biogeographically, reefs in the area have a lower species diversity than those farther south (Yaninek 1976, 1978, McClanahan 1990) and are transitional with coral reefs on the Somali coast and the upwellings system to the north (Carbone and Accordi 2001).

The region is also of cultural significance, as the coastal zone is home to centuries-old Arab-Swahili cultures. The majority of those living in the KMNR and Lamu are Bajun. Just inland are the unique Boni people of Cushitic origin and one of the few remaining animistic tribes in Africa. Their lives are intertwined with the local natural resources and traditional resource use includes fishing, mangrove cutting, farming for subsistence and hunter-gathering. Lamu town, the principal town south of the KMNR is designated a United Nations Educational, Scientific and Cultural Organisation (UNESCO) World Heritage Site for its cultural significance.

The Reserve was gazetted in 1979 and supports globally outstanding marine and coastal habitats and is a priority site in the World Wide Fund for Nature (WWF) Eastern African Marine Ecoregion (WWF 2001). In 1980, the KMNR, together with the terrestrial Boni and Dodori National Reserves, were designated a UNESCO Biosphere Reserve, recognizing the biological sensitivity of the area and the intimate dependence of local communities on biological resources.

Currently, the resources within the reserve are being threatened. Traditional resource use is permitted and has included fishing for finfish and lobsters (Fielding 2002, Murage and Church 2004), mangrove cutting and farming (Gubelman and Kavuu, 1996). This is however, increasingly becoming a problem as the Reserve faces intensifying threats from destructive and unsustainable resource use (fishing, farming, and harvesting for timber), land acquisition, poorly planned tourism development, habitat loss, pollution, climate change and limited management.

To combat these rising threats, a number of important activities have been established through a joint project between the KWS and the WWF established in 1995 to build consensus-based management for the reserve area. Fundamental to improved management has been the development and continuation of a joint KWS, WWF and Coral Reef Degradation for the Indian Ocean (CORDIO) coral reef, fisheries and resources research and monitoring programme initiated in 1998, coincident with coral bleaching in the El Niño event of early 1998 (Obura *et al.* 1998). This paper summarizes some of the findings and management implications for the KMNR.

Methods

The annual coral reef monitoring programme in the KMNR was started in 1998 using rapid assessment visual

estimate methods to establish a baseline. Coincidentally, the mass coral bleaching event of 1998 first became apparent at this time, thus the data reflects pre-bleaching coral cover while also recording maximum bleaching levels. From 1999 transect-based survey methods for benthos, invertebrates and fish (English *et al.* 1998) were adapted for application by mixed monitoring teams of scientists, project and government staff and local fisher participants. The surveys were carried out every year for a three week period in March/April during the Inter-monsoon period, when the water is at its calmest. A detailed description of the methods used can be found in Obura and Church (2004). Patches of hard substrate with coral communities were selected as the focus for monitoring, excluding the surrounding seagrass and sand habitats that cover most of the area of the KMNR.

From 1999 benthic sampling was conducted using 10 m Line Intercept Transects (LIT, English *et al.* 1998), replaced by video then still-image analysis methods in 2001. Image analysis was done by recording substrate cover at the centre of five small circles taped onto a monitor, increasing the number to 25 circles for high quality digital photographs. One hundred points were aggregated to comprise one 'transect', with 10 transects being the target for permanent monitoring sites, and 5 for resource survey sites. Fish and invertebrates were recorded using both scientific and local names in common 50 m belt transects, with a minimum sample of 6 per site, using a width of 5m for fish and 2m for invertebrates. Attention was given to recording resource and indicator species of fish and invertebrates.

Recruitment and health of coral colonies were recorded in haphazardly placed 1m² circles. For juvenile and adult corals, colonies were identified to the species level, size estimated (in 5 size class intervals), and any signs of bleaching, disease or other stressful conditions noted (e.g. black band, purple deposits, predation scars etc.). Small corals <10cm that were very rounded in shape and appeared to be the result of recruitment were measured to the nearest mm and their condition also recorded.

A search census technique was used to record the incidence of new coral species in successive time intervals of 2.5 minutes, giving species accumulation curves over time. In general, samples were at least 40 minutes, and where possible up to 1 hour. Identifications were conducted using Veron (1986, 2000), Wallace (1999) and Sheppard and Sheppard (1991). Documentation of species with a digital still camera was undertaken from 2002 onwards, following methods outlined in Sheppard and Obura (2004). To enable standardized comparison of diversity among sites, logarithmic regression functions on site-specific accumulation curves were used to predict species richness for 90 minute samples.

Results

Coral reefs in the KMNR are separated into three zones:

a) *deep reefs*. The outer barrier is a rocky ridge running NNE to SSW, ranges between 8-25m depth, and is

exposed to heavy ocean swell and offshore oceanographic influences such as upwellings up the continental shelf and seasonal exposure to cold currents from the Somali current system. Offshore of the reef base a sand or rocky substrate slopes gently into deeper water, and is dominated by suspension feeding communities of sponges, ascidians, soft corals, tube worms and mixed invertebrate/algal clumps. Inshore of the reef is an extensive sand plain to maximum measured depths of 35m. In places the barrier reef is broken by channels, which correspond to the main barrier island channels. The channel bottoms are sand and rubble, with in some areas extensive coral patch reef communities leading in towards the islands. The deep reefs were not included in

of turbidity and sediments from adjacent mangrove and intertidal systems.

Fifty-eight survey sites were included in this analysis, comprising outer (n=35), deep (n=13) and inner (n=10) sites. Of the 58 sites, 41 sites within the KMNR provide the primary longitudinal dataset presented here. Seventeen sites outside the reserve, comprising inner and outer, but not deep reefs, were added in 2003 and are analyzed separately.

Coral reefs of the KMNR had coral cover varying between 5 and 14% (means) and zero and 21% (minimum/maximum) over the course of this monitoring period, from 1998 to 2003 (Fig. 2). The reefs were dominated by various forms of algae (40–80%),

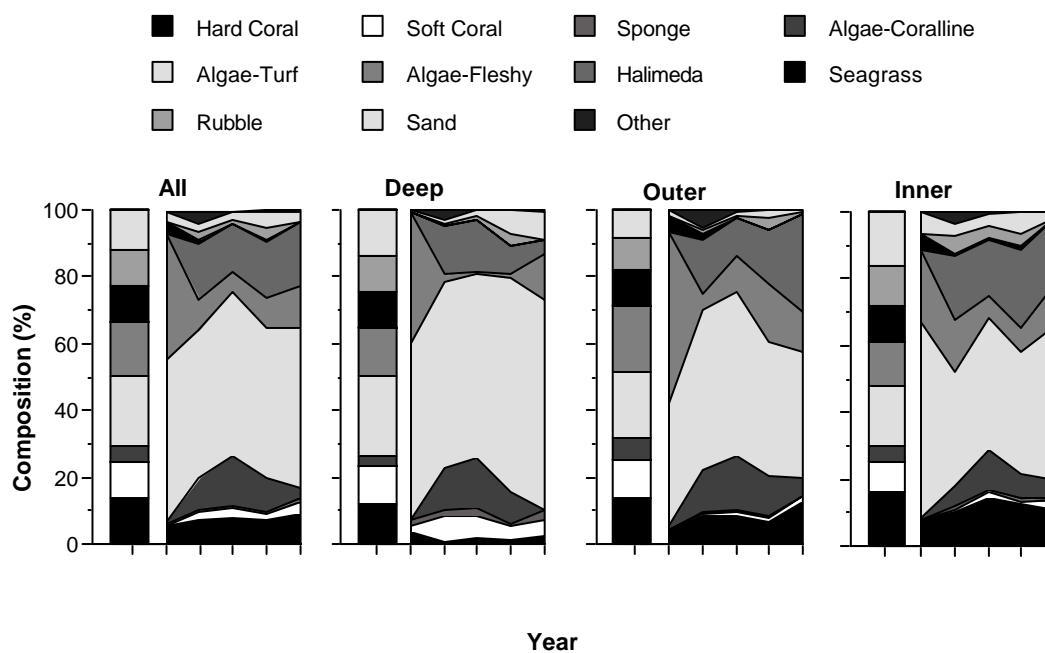


Fig. 2. Benthic cover of sites in the KMNR, 1998 - 2003. Data for 1998 are shown separately as they were collected using rapid assessment methods while those for 1999-2003 were collected using more standardized transect/quadrat methods. Out-of-reserve sites sampled in 2003 excluded from graphs.

the KMNR boundaries designated in 1979.

b) *outer reefs*. The eastern shores of the islands and mainland are fringed by hard substrates with patchy coral communities. Coral communities extend from the low tide level to 8-15m, and are aggregated on steeper slopes or on rock bommies. The coral communities are embedded in extensive seagrass beds (dominated by *Thalassodendron ciliatum*) on hard substrate, extending from the shallows to 10-12m. These give way to the sand in the south of the reserve and sand/rubble/suspension feeder communities similar to those of the outer deep reefs in the north. Fringing reefs are exposed to strong surf and breaking waves that pass over the outer barrier.

c) *inner reefs*. Patchy back-reef communities fringe the western edges of islands and fringing reefs and/or are located in sheltered bays and mangrove channels. They are never exposed to heavy wave energy, experience strong tidal currents, and some are exposed to high levels

predominantly turf algae, with varying levels of fleshy algae, coralline algae and *Halimeda*. Soft coral cover was consistently less than hard coral cover, and highest on deep reefs. Low levels of the minor cover categories (coralline algae, rubble, sand, seagrass, 'other') were recorded in all years.

The dominant trend in benthic cover was the decrease in hard coral cover of 62% from 1998 to 1999 (Fig. 3), mirrored by a similar decline in soft coral cover, as a result of mass coral bleaching and mortality coinciding with the El Niño Southern Oscillation (ENSO) event of 1997-98 (Obura 2002). From 1999 to 2003 there was a progressive recovery of hard corals to 66% of 1998 levels. Before the bleaching, inner reefs had slightly higher coral cover than outer and deep sites (16, 14 and 11%, respectively), though the differences were relatively small. Mortality of corals in the El Niño affected the reef zones differently, with the greatest drop

in coral cover in outer reefs (72%) compared to deep reefs (68%) and inner reefs (52%). Recovery also differed between the zones. Deep reefs showed continuing declines in coral cover to 2002 and then a small gain to 20% of 1998 levels in 2003. Outer reefs showed strong recovery to 88% of pre-bleaching levels in 2003; the rapid increase in 2003 being partially an artifact of sampling. Inner reefs showed the fastest recovery to 2002, reaching 77% pre-bleaching levels, then a decline in 2003 to 67 % of 1998 levels due to vulnerability to disease in 2002 (see later).

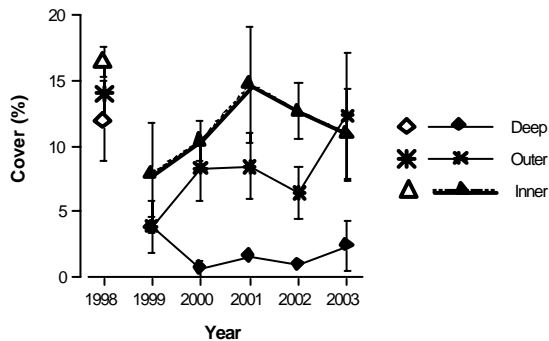


Fig. 3. Coral cover in the KMNR, 1998 – 2003 (mean and standard error) for each reef zone). Data for 1998 are shown separately as they were collected using rapid assessment methods while those for 1999-2003 were collected using more standardized transect/quadrat methods.

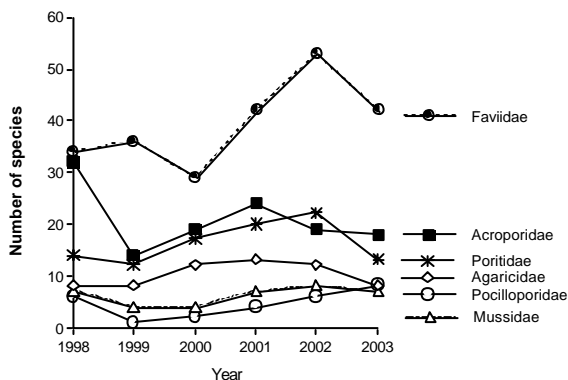


Fig. 4. Numbers of species by family of hard corals at all sites in the KMNR by year. Sites outside of the reserve, sampled in 1998 and 2003, are excluded, as species diversity of corals is higher at these locations due to biogeographic factors. Number of sites sampled each year: 1998- 9; 1999- 15; 2000- 22; 2001- 21; 2002- 22; 2003-15

Coral species diversity declined dramatically from 1998 to 1999, following which it increased at most sites to 2003. A total of 140 species were recorded in 1998 (at 2 sites), which declined to 104 in 1999, then climbed back to 140 in 2001, 150 in 2002 and 142 in 2003 (all with 8-12 sites).

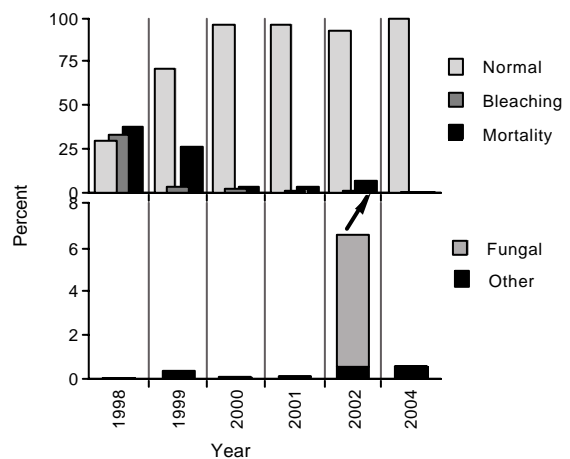


Fig. 5. Coral condition, 1998 – 2004, KMNR. Top – proportion of normal, bleached and dead colonies. Bottom – incidence of general and a fungal disease. Mortality related to the fungal disease in 2002 is shown by the arrow.

Shallow inner reefs had the highest species diversity, with predicted values of 89 species in 90 minutes of sampling for Chole (an inner channel patch reef) in 1998 and 101 species in 2003. Outer reefs with the most highly developed coral communities had predicted species richness of 60-80 species, while deep reefs had the lowest diversity, with 40-50 species. The approximately 40% reduction in coral species diversity at individual sites in 1999 was due to local mortality of species. The Acroporidae and Pocilloporidae, at the family level, and *Acropora* and *Pocillopora* at the genus level, were the taxonomic groups that suffered the greatest loss of species (Fig. 4).

A number of rare and/or regional endemic species have been recorded in the KMNR. A population of some 20+ colonies of *Siderastrea savignyana* is found at one site, often partially or totally buried by shifting sands, but persistent over the 6 years of surveys. The deeper sections of the fringing reefs, particularly towards the north, and the outer deep reefs (the whole length) hold the most extensive population of the regional endemic *Horastrea indica* that has been documented to date in East Africa (D. Obura, pers. obs.). This species is restricted to islands of the Western Indian Ocean and parts of the East African coast, but it appears most common in sediment-influenced and turbid sites, and at depths greater than 10 m. The marginal environment, under the influence of high sediments from mangrove creeks, and cool nutrient-rich upwelling waters appear to favour some of these minor and rare species over the species dominant farther south.

Mass coral bleaching was shown by the percentage of bleached corals in 1998 of 31% (and pale colonies 7%, total mortality >30%), during the El Niño bleaching event, compared to levels of <1% in other years (Fig. 5). In 1999, the proportion of normal colonies increased to 67%, and rose again to 85-90% in 2000–2002. In ‘normal’ years (excluding 2002, see below), partial and

full mortality varied around 5-10% and <1%, respectively. In February 2002 an unknown coral disease was seen for the first time (Pople and Church 2003), likely to be of fungal origin (McClanahan *et al.* 2004) that affected genera common in the KMNR, namely *Astreopora*, *Echinopora*, and *Montipora*, as well as a variety of other genera including the bleaching-vulnerable *Acropora* and *Pocillopora*. This resulted in a high incidence of disease records (6.0%) and a subsequent ten-fold increase in mortality over background levels to 3.3% and 4.5% (full and partial mortality, respectively, Fig. 5). Of the impacted genera, *Astreopora* suffered the highest mortality, with only 2-3 partially live colonies being seen in 2002, whereas it was previously common. We estimate mortality to have been >90% for *Astreopora*, and lower than 30-50% for the others mentioned above. At the time of the disease, *Acropora* and *Pocillopora* were mainly present as recruits in early recovery mode following the mortality of 1998, and suffered high mortality due to the disease. The disease resulted in reductions in the rate of recovery of coral cover at inner sites; the two prime coral community sites were strongly impacted by the disease and showed a decline in coral cover from 2001 to 2002, while at other sites coral recovery remained positive, but at lower levels.

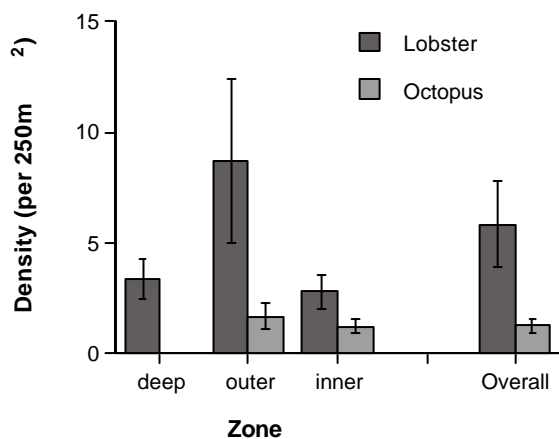


Fig. 6. Densities of octopus and lobster counted in 45-60 minute sample periods during sampling in 2002 (mean \pm standard error).

Lobster and octopus, the most valuable invertebrates in the local fishery, were found at low abundances, at 0.2 and 0.1 per 250m² respectively. Lobsters were most abundant at outer fringing reef sites (Fig. 6) due to the presence of suitable ledges and shelters, while octopuses were most common on inner and outer reefs. Crown of thorns seastars were only recorded within transects at one site in 2002 (at 0.09 per 250m²), and predation scars were not observed.

A total of 40 fish families were recorded in fish transects, comprising 208 species in 56 genera, though because surveys focused on resource species this does not provide an accurate species inventory. Sweetlips (Haemulidae) and snappers/emperors (Lutjanidae/Lethrinidae) were the most individuals

observed on transects, with sharp declines in abundance below the top five families (Fig. 7). *Lutjanus* (Lutjanidae), *Acanthurus* (Acanthuridae) and *Plectorhynchus* (Haemulidae) were the top three genera observed. Generalist carnivores, the snappers, emperors and sweetlips (Scavengers) were by far the most abundant trophic group of fish with a mean of over 500 per 250m², followed by herbivores, piscivores and invertivores (means of 115, 55 and 23 per 250m², respectively). Inner reef sites had the highest diversity of fish and outer reefs and some out of reserve sites had the lowest diversity as measured by number of families

Fish abundance varied over a wide range, from highs of >1,000 per 250m² to lows of 15 per 250m², and varied among years (Fig. 8). Highest densities were recorded on the deep reefs (>10m) where large aggregations of several thousands of schooling fish are found adjacent to ledges and small caves on the deep drop-offs.

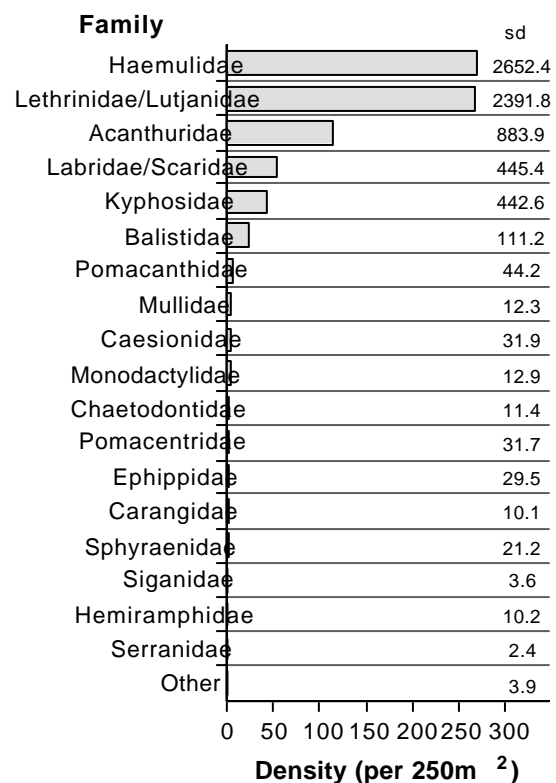


Fig. 7. Density of the top 18 fish families, KMNR (mean, standard deviation written to right of each bar).

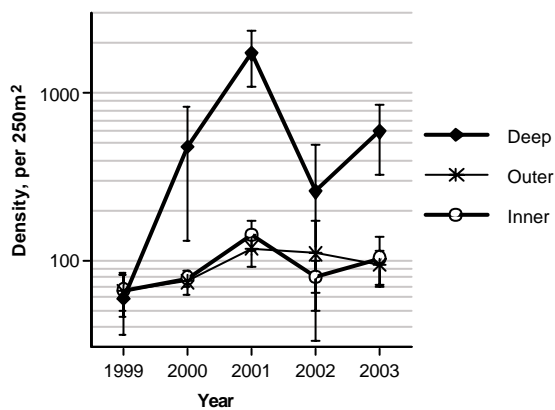


Fig. 8. Variation in fish density by reef zone in the KMNR for 1999-2003 (mean and standard error, number of fish per 250m²). Note logarithmic scale on y-axis.

These conditions are restricted to a small number of sites however, as for most of its length the deep reef is a smooth rounded ridge with low topographic complexity, where lower fish densities of several 100s per 250m² were recorded. The lowest densities of fish were recorded at shallow (approx. 5m), outer sites. These sites had low relief and were dominated by seagrass growing on rocky substrates with small rock ledges, providing little shelter for fish. These sites are also heavily fished. Overall densities were highest in 2001 with declines in all zones in 2002, most marked on the deep and inner reefs, and following the trends of the 4 most abundant families, the snappers, sweetlips, surgeonfish and triggerfish (Fig. 9). Standard deviations were also highest in 2001, suggesting high variability between transects and sites. In addition to potential causes of natural variability, the use of data collectors with no training in

sampling design and theory contributes additional sources of variation that make it difficult to attribute changes in mean and variation in population density to natural causes.

Across all sites, a clear increase in fish abundance is evident from south to north (Fig. 10). Fish densities at sites in the south of the reserve were constrained to a narrower range of values, while to the north higher densities were recorded. Deep sites in the southern part of the reserve had lower densities. This latitudinal density trend was dominated by abundance of the four dominant fish families (snappers, sweetlips, surgeonfish and triggerfish), 3 of which are primary resource species for fisheries in the reserve.

Large groupers (*Epinephelus lanceolatus*, *E. malabaricus*, *E. tukula*, *Plectropomus laevis*, *P. punctatus*, *Variola louti*), black jack (*Caranx ignobilis*), humphead wrasse (*Cheilinus undulatus*) and other species vulnerable to fishing were rare at inshore reefs, but commonly seen on the deep outer reefs, and abundant at sites with high topographic complexity. A range extension for the 'Red Sea' angelfish, *Apothemichthys xanthotis* was documented multiple times from 1998 to 2003. Reported as endemic to the Gulf of Aden, Arabian Gulf and Red Sea (Allen *et al.* 1998), aggregations of this fish were common on the deeper rubble slopes towards the north of the reserve.

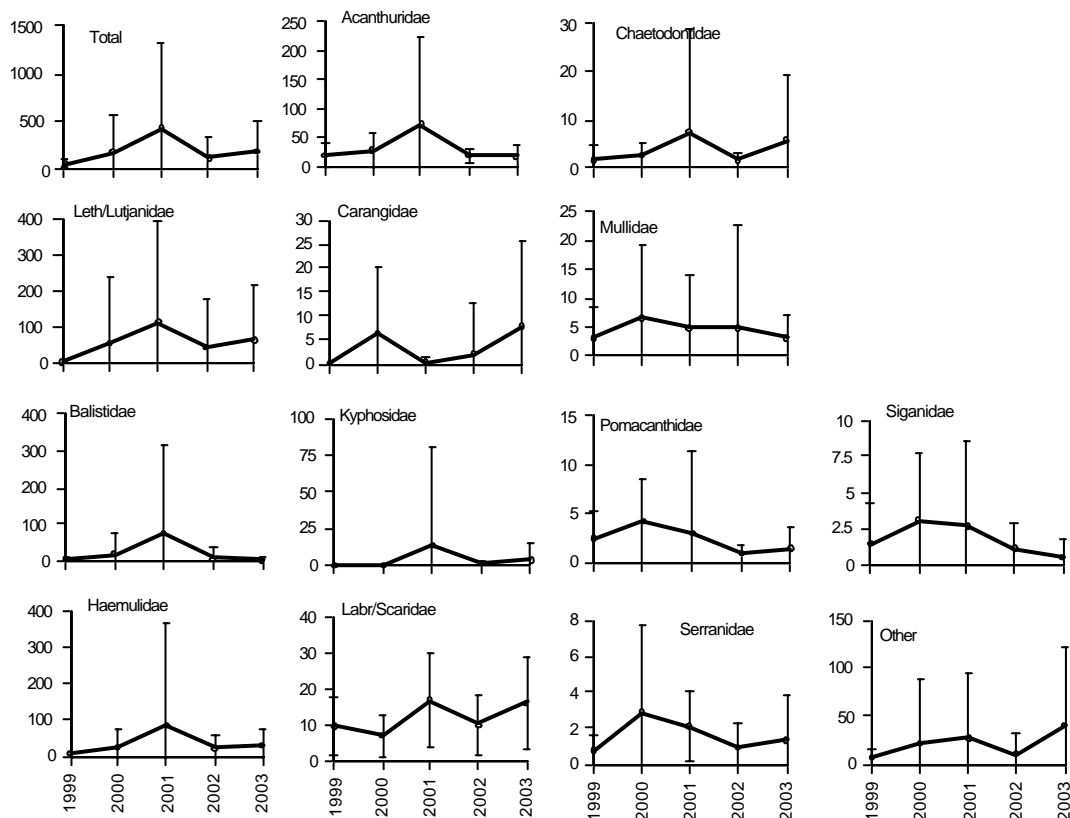


Fig. 9 Abundance of fish by family and year 1999-2003 Kiunga Marine National Reserve (mean and standard

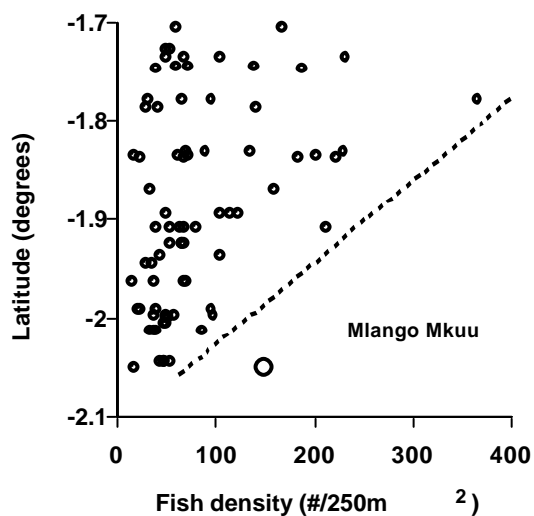


Fig. 10. Fish density (x axis) versus latitude (y axis) in the KMNR. The figure excludes high-density sites above 400 fish per 250m² as these make it difficult to visualize patterns in the other sites. The dotted line illustrates how maximum density increases northwards. The circle to the right of the diagonal is data from Ml. Mkuu, a large passage at the southern tip of Kiwayu Island and potentially with larger transient fish populations and/or difficult fishing access due to currents and tidal flows.

Here, either individual or paired adults, and a scattering of juveniles were observed on the rubble bottom, at times

with over 10-20 individuals. Numbers were higher than would be expected for a non-reproductive population of vagrants. *A. xanthotis* has not been seen at any sites at the southern end of the reserve, nor outside of the reserve in Kenya.

Seventeen out-of-reserve sites were sampled in 2003, focusing on the area offshore of southern Pate Island, Manda Island and the Islands of Kinyika and Tenevi south of Lamu, about 30-50 km south of the KMNR. Benthic cover in the out-of-reserve sites was almost identical to that measured inside the reserve, with average coral cover of just below 10% and dominance by algal turf at almost 60% cover. Lowest coral development was documented at inner and outer reefs in the Pate area, with evidence of high coral mortality during the bleaching in 1998, and these sites are heavily utilized for fishing. Fish populations at out-of-reserve sites were higher than those inside the reserve by a factor of 1.2 to 2, but not significantly, for both inner (108.3±115.5 vs. 88.1±85.1, respectively) and outer (150.0±140.1 vs. 70.4±34.2, respectively) reefs.

In early 2002, two previously unknown phenomena occurred in the KMNR (Table 1). Whether they were linked by a common cause is not established, but a combination of strong onshore winds and a strong Somali Current upwelling event resulted in the upwelling plume being closer onshore and further south than usual (D. Robins, pers. comm.). First, a harmful algal bloom (HAB) was reported from Bosaso on the Gulf of Aden coast of Somalia (latitude 10°N) and extended down the coast as far as Lamu (2.3°S), from early January to mid-

Table 1. Area and date for reporting of Harmful algal bloom and coral disease in East Africa and the Gulf of Aden, December–February, 2002. Communications reported through an email discussion thread (January–March 2002). Source Obura 2002.

Area	Date	Source
<i>Harmful Algal Bloom</i>		
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	United Nations Development Programme
Kiunga, northern Kenya	Jan 25, 2002.	J. Church, World Wide Fund for Nature (WWF)
Somalia-Kenya	29 Jan, 2002	Oceanographic conditions (1mg/m ³ chlorophyll) reported from Oceanography SeaWIFS (NASA) data by D. Robins, Marine Biological Laboratories, Plymouth, UK.
	31 Jan 2002 – 11 Feb 2002	Fisheries Department closed fishing in the KMNR
Northern Kenya	Feb. 6, 2002	Identification of <i>Gymnodinium mimikimotoi</i> or <i>G. brevisulcata</i> by G. Pitcher, South Africa plankton ID - possibly the latter as this is known to also produce aerosol toxins, which would explain eye irritations reported by fishermen in Somalia.
<i>Coral Disease</i>		
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

February 2002 (Pople and Church 2003). Just as this was dying out, a coral disease was reported from Kiunga (2°S) to Tanga in Tanzania (5°S), and persisted from mid-February to late-March 2002. The annual monitoring programme takes place in late March-early April each year, thus immediately followed these two phenomena in 2002. The impact of the coral disease was described above. Reports of fish mortality during the HAB included a wide range of fish, including pelagics, demersals and reef fish (Pople and Church 2003), but the impact of the HAB on fish populations is harder to identify in the monitoring data, due to high variability in the fish counts. Fish densities in 2002, recorded 2 months after the HAB were much less than in 2001, and showed some recovery in 2003. However, high variance prevents interpreting this due to the HAB.

Discussion

The coral community structure data confirms that coral reefs in the Kiunga Marine National Reserve are at the northern limit of true coral reef growth on the East African coast and marginal compared to reefs further south (Carbone and Accordi 2000, Samoily 1998, Kemp 2000, Obura 2001). Coral cover is lower than sites in southern Kenya and Tanzania (20-30%, e.g. McClanahan and Arthur 2001) and diversity is lower (140-150 species, compared to 200-270 species increasing southwards; Obura *et al.*, in press). Even within the reserve system, a transition from south to north was noted, in the decreasing coral species diversity, increasing abundance and shallowing distribution of the marginal coral species *Horastrea indica* and suspension feeding communities, and increasing abundance of the Red Sea angelfish *Apolemichthys xanthurus*, presumably at the southern end of its distribution range.

Recovery of reefs in the KMNR from the El-Niño bleaching event of 1998 has been limited and patchy, mainly constrained by recruitment of new corals. Recruitment has been overwhelmingly dominated by *Pocillopora* spp. and *Coscinaraea* spp. (Obura, in review) which are opportunistic and fast growing, but contribute little to reef framework development. Recruitment of massive species is typically low in coral reefs and the fast growing dominant corals, such as *Acropora*, *Galaxea* and *Montipora* that normally show high recruitment rates are not recruiting successfully to the KMNR.

The deep reefs of the KMNR are an important physical feature of the reef environments, and perhaps a dominant control on the ecology of the inshore reef system. While poor in terms of coral community structure and diversity, they harbour large fish populations, at some places in massive abundances (and see Samoily 1988) that may function as a reserve for inshore reefs. Additionally, they shelter inshore areas from direct upwelling of cool nutrient-rich waters, perhaps enabling the development of more southern-adapted coral communities in inshore, warmer waters.

Ecosystems at the margins of major distributions, or ecotones, have been variously hypothesized to have

higher ecological resilience to changes, due to their being accustomed to more marginal conditions, and conversely of having lower ecological resilience, being closer to a number of physiological limits. The KMNR coral reef communities appear to support the latter hypothesis, showing a high level of vulnerability to the threats documented here – coral bleaching in 1998, a HAB and coral disease epidemic in 2002, and increasingly severe fishing pressure. Recovery of coral communities and coral recruitment are lower in the KMNR than in other parts of the Western Indian Ocean (Obura 2005) and a progressive decline is apparent from 1998 to 2004 as a result of these threats, superimposed on the local intense threat of overfishing. We hypothesize that as a result of its marginal position the KMNR is showing vulnerability to large scale threats as follows, impacting directly on the conservation values and goals of the KMNR, and likely also on the management needs of the reserve:

- Due to the reduction of adult coral populations over scales of several 100s and even 1,000s of kilometres in 1998 and 2002, both reproduction and recruitment of corals is depressed, and the KMNR is further limited by its isolation from the nearest southern source reefs (over 200 km to Malindi, Kenya).
- Reduced cover and diversity of corals, particularly of fast growing and reef building corals may result in simplified reef structure over the long term, and reduction of holes, crevices and three-dimensional topography that other components of the reef community, such as algae, invertebrates and fish depend on. This may result in lower biomass and productivity, and ultimately reduced fish catch for human consumption.
- Many other taxonomic groups on coral reefs are also vulnerable to large scale threats (e.g. soft corals, anemones, sponges and their commensal organisms vulnerable to bleaching, fish vulnerable to HABs); local loss of species and a decrease in genetic diversity within species as vulnerable genotypes are lost may occur with consequent loss of biodiversity in the area.

The characteristic fish communities of the deep reefs suggest high productivity, but further work in this area may reveal the affinity to cold nutrient-rich upwellings such as the Somali Upwelling, suggested by the presence of the angelfish, *Apolemichthys xanthurus* previously thought limited to the Gulf of Aden (Allen *et al.* 1998). Shallow reefs have typical coral reef fish populations, though strongly impacted by fishing pressure from the south. The declining abundance of fish southwards reflects the accessibility of resources in the south of the reserve to the towns and larger fishing villages south of the reserve and as far as 150 km to Malindi and beyond. Villagers south of the reserve preferentially fish in the reserve, and appear to impact fish populations in the reserve to a greater extent than the out-of-reserve sites near Lamu Island. Inaccessibility of the deep reefs adjacent to the reserve enables the persistence of vulnerable large groupers and wrasses, and areas of high concentration of target reef fish.

The increase in fish populations from 1999 to 2001 is dramatic, but high variability makes it difficult to determine how reliable the mean values are. Interestingly, the strong decline in 2002 may be related to fish mortality during the harmful algal bloom in 2002 just prior to sampling, however again, high variability precludes making definitive judgements.

Like all marine reserves in Kenya, the management of the KMNRR is hampered by the lack of clearly defined reserve regulations, overlapping institutional mandates, and limited management. Despite being gazetted for management in 1979, and earlier studies highlighting the biographic importance and vulnerability of the area to fishing (Samoilys 1988, McClanahan 1990), it was not until the mid-1990s that the first steps in designing locally appropriate management strategies were taken (Gubelman and Kavuu 1996). The findings from this work have guided recommendations for improved coral reef and fisheries management in the reserve (Church and Obura 2004), which is becoming increasingly necessary to prevent further degradation of fish stocks as documented in the south of the reserve. It is the intention that the findings will assist in the formulation of reserve regulations specific to the needs of the reserve and the communities which live within and adjacent to the reserve; and to the review of the current management plan.

Acknowledgements

We are particularly indebted to the local resource users (fishers and women), the local administration, KWS KMNRR and Lamu, Fisheries Department KMNRR and Lamu and the WWF KMNRR personnel. Without the support of so many of these local stakeholders the research and monitoring would not have taken place. We are also grateful to the technical support provided by the CORDIO-EA office and to the Kenya Marine Fisheries Research Institute, and the operational support bestowed by tourism operators in the area. The majority of funding for the coral reef research and monitoring has been provided by the WWF and CORDIO.

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