Artisanal fishery analysis within the Mpunguti Marine Reserve (Southern Kenya): Gear-based management towards sustainable strategies.

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Artisanal fishery analysis within the Mpunguti Marine Reserve (Southern Kenya): Gear-based management towards sustainable strategies.

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No data can be taken out of this work without prior approval of the thesis-promoter.

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The sustainable management of coral reef fisheries subjected to overfishing is challenged by the complex multi-species, multi-gear and poverty context of its artisanal fisheries. Worldwide coral degradation and overfishing are setting an increasing pressure to resource users and managers to reconsider current management practices and explore innovative strategies. This study was carried out in the Mpunguti Marine National Reserve, contiguous to the Kisite Marine National Park, a no-take coral reef fisheries closure, located off the south coast of Kenya in the Western Indian Ocean. We explored a gear based management approach by incorporating escape gaps (3 cm x 30cm) in traditional basket traps (malema) and comparing the catches with the traditional traps (controls). This gear based option exploits differences in selectivity among gear types to control catch composition, reduce the catch of juveniles and bycatch species, without compromising the fisherman's income. Of the 2060 fish sampled, we identified 93 species belonging to 26 families, during 213 sampling occasions. There was no significant difference in the total catch per unit effort (CPUE, kg/trap) between gated and traditional traps, but gated traps significantly ($p < 0.001$) reduced the catch of non commercial fish (low-value, juveniles and narrow-bodied species). Moreover, for the most important local commercial species, the African white-spotted rabbitfish (*Siganus sutor*), the gated traps significantly increased the mean length of capture (by 13 %) and weight (by 32%) and decreased the proportion of catch under length at first maturity (Lmat) from 19.9% (traditional traps) to 3% (gated traps). Therefore, escape gaps did not reduce the catch of high value fish and decreased the catch of juveniles and narrow bodied coral reef herbivore species, increasing biodiversity, promoting sustainable practices and ecosystem health, without compromising fishermen's revenues.

Keywords: marine fisheries reserves, coral reef fisheries, catch composition, fish traps, bycatch, escape gaps, gear based management

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Introduction

Although coral reefs cover less than 1% of the oceanic area (Hughes *et al.* 2003) they are known to support almost one third of the world's marine fish species (Newton *et al.* 2007) and to contribute to around 10% of the total fish consumed by humans (Pauly *et al.* 2002). When compared to the world's global fishery, approximately 9 to 12% are considered artisanal fisheries based directly on coral reefs (Mumby *et al.* 2007). The definitions of traditional, artisanal and small-scale fisheries have however been subjected to different interpretations. These designations will be used in this study to describe typically near-shore fisheries, performed by fishers using small-sized vessels and labour intensive methods with little or no modern technology (FAO 2009). Still, coral reef fish fisheries have been described as being more efficient, less wasteful and supporting far more livelihoods per ton produced than industrial scale fisheries (Sadovy 2005).

The central target of coral reef management is to preserve the capacity of tropical reefs to sustainably support the ecosystem goods and services, with regard to fisheries, tourism and cultural values (Moberg & Folke 1999). Yet, coral reefs worldwide are declining due to anthropogenic impacts such as over-harvesting, pollution, disease outbreaks and climate change (Hughes *et al.* 2003; Bellwood *et al.* 2004; Cinner *et al.* 2012). As a result, coral degradation is expected to have longer-term effects on coral reef fisheries (Graham *et al.* 2007) and to affect the lives of millions of people who directly or indirectly depend on this ecosystem for income and protein source (Sadovy 2005).

The annual coral reef cover decline has increased to 2% between 1997 and 2003 (Bruno & Selig 2007) and this is setting an increased pressure on scientists, fisheries managers and stakeholders to reconsider current management practices and discover innovative and adaptive management tactics (Mcclanahan & Mangi 2004; Bellwood *et al.* 2004; Rinkevich 2008). An improved knowledge of the processes engaged in coral reef ecology and resilience (Hughes *et al.* 2003; Bellwood *et al.* 2004), as well as a greater understanding of the socio-economic and cultural dimension is needed in order to accomplish effective conservation outcomes (Mcclanahan *et al.* 2005; Cinner *et al.* 2009c). Furthermore, a recent evaluation of literature on applied coral reefs science has revealed an extensive documentation on the impacts of the coral reef climate and fisheries crisis, while highlighting the need to explore practical and realistic management solutions (Mcclanahan 2011). An up to date analysis of 464 coral reef fisheries peer-reviewed articles concluded that only 22% of those actually presented management recommendations based on the conclusions of its research (Johnson *et al.* 2012). So, there is an urgent need for solutions that will make coral reef fisheries ecologically, socially and economically sustainable.

Kenyan Fisheries

The Western Indian Ocean (WIO) belongs to one of the most dynamic and variable Large Marine Ecosystems in the world and seasonality is the major cause affecting annual patterns of physical, chemical and biological processes along the East African coast (McClanahan, 1988). Cyclic meteorological and oceanographic patterns are dictated by the Inter-Tropical Convergence Zone (ITCZ) which creates 2 distinct seasons - the northeast (NE) and southeast monsoons (SE). The SE monsoon (April to October - *Kuzi*) is characterized by high cloud cover, rainfall, river discharge, wind energy and lower temperatures. The ocean experiences high water-column mixing, fast currents and wave energy. These parameters are inverted during the NE monsoon (November – March – *Kaskazi*). Kenyan fish catches are low during the SE monsoon and high during the NE monsoon with a peak in March (McClanahan 1988; Kaunda-Arara 1997; Weru *et al.* 2001). The reason for this seems to be related to multi-factors involving reduced fishing effort during the SE monsoon due to rough sea conditions, fish migration, dispersal and recruitment patterns and a deeper thermocline and cooler waters (McClanahan 1988).

The Kenyan coastline is 640km long and lies on the western border of the Indian Ocean, between the latitudes 1º41'S and 5º40'S, from its border with Somalia to its border with Tanzania. It consists of 12 nautical miles of territorial waters and an EEZ extending to 200 nautical miles, with a total area of 142 400 km^2 (FAO 2009).

Most of the fishing sector in Kenya involves inland fisheries (particularly from Lake Victoria) and the marine fishery production in Kenya only contributes 2 - 6% of the total national fish production (FAO 2009). The great majority is carried out by artisanal fisherman (only 5 % of the catch comes from commercial trawlers) and over 60 000 coastal people depend directly or indirectly on these fisheries (UNEP 2006). The artisanal fishery involves multiple gear types, such as basket traps, hand lines, spear guns, beach seines, fence nets, gill nets and more recently, ring nets. The deep sea (EEZ) fishery resources are currently exploited by Distant Waters Fishing Nations (DWFNs) through a licensing system, but only a small quantity of that catch (migratory tunas including skipjack, yellowfin and bigeye tuna) is landed in Kenya (FAO 2009).

The coastal area is characterized by a variety of tropical marine and wetland ecosystems including coral reefs, sea grass beds, mangroves and salt marshes. The coral reefs and associated sea grass beds are the basis for a multispecies, multigear small-scale fishery, and recent studies report that the national artisanal fishery employs almost 23,000 fishers catching over 16,000 tonnes of fish annually (Tuda *et al.* 2008) and provides monetary income and animal protein to about 70% of the coastal communities in Kenya (Glaesel 1997).

Population growth, poverty and unemployment in the Kenyan coastal region, has contributed to a rise in the number of small-scale fishers, with a 34% increase documented between 2004 and 2008 (Ochiewo 2004). Excessive and destructive fishing has been described as one of the major problems facing the local reefs (McClanahan & Shafir, 1990; McClanahan & Obura, 1995). In order to maximize their catch, fishermen are driven by Malthusian overfishing, where fishing effort and use of destructive gear increase due to the increase of local population and declining resources (Pauly 1994; McClanahan *et al.* 2008b). Signs of overexploitation include an increase in sea urchin population (McClanahan & Muthiga 1988; McClanahan & Obura 1995; Glaesel 1997) and a decrease in the catch (reduced sizes and wet weight of landed fish) (McClanahan & Mangi 2001, Obura 2001).

Therefore, artisanal fishing represents an activity that urgently needs to bring back together the matters of human development and environmental sustainability if it is to continue to support the lives of millions who largely rely on coastal resources.

Fisheries Legislation and Coastal Governance

Kenya was the first developing country to have legislation for the protection of its marine resources (Malleret-King 2000). Marine protected areas (including marine reserves and national parks) coverage increased from nothing in 1967 to 8.7% of the total continental shelf in 2004. Furthermore, 8.6% of the Kenyan reefs are fully protected from fishing (Wells *et al.* 2007).

Marine reserves and National Parks were established in Kenya over 30 years ago (Wells *et al.* 2007). Kenyan **marine national parks** provide coral reefs with total protection from extractive exploitation (no-take zones, IUCN Category I) and their use is restricted to visitation, education and research (McClanahan 2005). Kenyan **marine reserves** allow fishing with traditional or non-destructive gear, mainly fishing lines and traps. However, these conditions inside the reserves are open to various interpretations and few restrictions have consistently been imposed in the reserves (Kaunda-Arara & Rose 2004a; McClanahan *et al.* 2005, 2007).

All protected areas fall under the jurisdiction of the Wildlife Act Cap 376 of the Laws of Kenya. Currently, Kenya has created nine MPAs which include five no-take marine parks and four marine reserves, that cover approximately 9% of the coastal shelf (Wells *et al.* 2007) (Table 1). This percentage constitutes a good result regarding the 1992 Convention on Biological Diversity MPA target (10% of all marine ecological regions effectively conserved by 2012) (Wells *et al.* 2007).

Table 1 - Marine protected areas in Kenya (adapted from Wells *et al.* 2007).

Date established	**Kenya Marine Protected Area**	**Total Area (Km^2)**
1968	Malindi Marine National Park	6.3
1968	Watamu Marine National Park	10
1968	Malindi-Watamu Marine National Reserve	245
1973	Kisite Marine National Park	28
1978	Mpunguti Marine National Reserve	11
1979	Kiunga Marine National Reserve	250
1986	Mombasa Marine National Park	10
1986	Mombasa Marine National Reserve	200
1995	Diani Marine National Park	75

The fisheries sector in Kenya has been regulated by the two main legislations: the Fisheries Act (Cap 378) of 1989 and the Maritime Zones Act (Cap 371) of 1989 which provide the legal framework for the management, exploitation, utilization and conservation of fisheries in Kenya. Fish stocks that are found in the marine parks are managed by the Kenya Wildlife Service while stocks in the marine reserves fall under the jurisdiction of both Kenya Wildlife Service and Fisheries Department. In 2001 a number of restrictions were added which included the exclusion on the use of mono-filament nets, seine nets, spear guns, and seasonal restrictions on trawling (McClanahan 2005). The small-scale fisheries are characterized by open access and fishermen require an annual license of KSh.100 (about US$1.17) to access the inshore fishery grounds.

In order to decentralize fisheries management, the Fisheries Department started in 2006 a community-based management strategy, forming Beach Management Units (BMUs). It delegates responsibility to stakeholders to administer their natural resources at the local level. As a result, BMUs have now the jurisdiction to co-manage with the Fisheries Department the activities and legislation within their landing site; namely the control of gears, assistance in data collection, implementing management practices and solving disputes on fishing grounds (Obura *et al.* 2008). Although it constitutes a positive step in decentralizing management authority and enables communities to locally develop appropriate regulations it is argued that there is still a need for appropriate training, capacity building and technical proficiency by governmental agencies and local NGOs (Oluoch *et al.* 2006).

Gear based Management

A great challenge in coral reef fisheries management is finding competent means to achieve ecological outcomes that are accepted and implemented by fishing communities. Management efforts are further complicated in the context of the multi-species and multi-gear reef fisheries with poor resource users highly dependent on fishing (McClanahan *et al.* 2008b; Cinner *et al.* 2009c).

In developing countries, fishing gear limitations have been proposed as the more acceptable and ethically preferred form of management, as opposed to restriction of access, effort or catch (Hicks & McClanahan 2012; McClanahan & Cinner 2008). However, access restrictions have become the default fisheries management tool in low-income countries (Mumby & Steneck 2008). In Kenya, no-take zones have been reported to have greater hard coral cover and higher biomass and diversity of coral reef fish compared to unprotected reefs (McClanahan & Shafir 1990; McClanahan 1994; McClanahan & Arthur 2001). Still, this management option might undermine local livelihoods and become difficult to justify and enforce in areas where poor people are faced with few alternative source of revenues (McClanahan & Mangi 2001; McClanahan *et al.* 2008). Small-scale artisanal fishermen tend to be more supportive of limitations on particular types of fishing gear when compared to fishing-ground closures, as they perceive closed areas as a direct gain to central government through high tourism revenues, with little return to the local community (McClanahan *et al.* 2005; Cinner *et al.* 2009b).

Gear selectivity can produce impacts at different levels, ultimately affecting the population size structure and the composition of its food webs and fishery. A well-managed fishery is expected to employ gears that catch most of the available species at sizes that do not undermine sustainability (McClanahan & Mangi 2004). Gear-based management is an approach that exploits differences in selectivity among different gear types to ultimately control catch composition (McClanahan & Cinner 2008). As an example, the catch per unit effort in the Kenyan south coast was able to increase by

20% after the elimination of the beach seines, since this gear was responsible for catching very small fish (McClanahan 2010). This approach has the potential to address multiple objectives and to be flexible to different socio-economic and ecological settings (McClanahan & Cinner 2008; Hicks & McClanahan 2012).

A good understanding of fishing gear species and size selectivity is fundamental to manage fisheries, especially in multi gear contexts, as gear will eventually influence the size frequency and catch composition of fish resources (McClanahan & Mangi 2004). Furthermore, gears can be actively managed to encourage ecosystem health; to recover selected functional groups, reduce high erect algae cover and sea urchin dominance, increase coral cover, and reduce detrimental effects of coral bleaching and climate change (McClanahan *et al.* 2008a; Cinner *et al.* 2009a). In addition, a broad understanding of the fisheries' social setting is crucial to sustain viable management options based on the active support by the fishing community. Kenyan artisanal fishermen are still influenced by traditional values associated with gear use that is either consented to or not by elders (McClanahan *et al.* 1997). Community leaders, with the support of government and local NGOs, can approve specific gear use and generate easily enforceable ecosystem-based co-management initiatives.

Kenyan Trap Fishery

The trap fishery accounts for almost 40% of Kenyan reef fish landings by weight (FiD, 2008), and is currently unregulated (Mbaru & McClanahan 2012, submitted). Artisanal coral reef trap fisheries can result in serious over-fishing, reduce biodiversity, and alter ecosystem structure (Hawkins & Roberts 2004).

Studies in southern Kenyan coral reefs revealed that traps are considered by elders the most traditional fishing gear (McClanahan *et al.* 1997), are relatively unselective with selectivity largely dependent on mesh size (Mcclanahan & Mangi 2004) and cause low physical damage to corals (Mangi & Robers 2006; Cinner *et al.* 2009a). On the other hand, the use of traps results in the capture of large numbers of small, low, or no-value fish due to the low size selectivity of its diamond-shaped mesh (Mbaru & McClanahan 2012, submitted). Local trap fisheries target high value fish such as rabbitfish (Siganidae), emperors and snappers (Lethrinidae and Lutjanidae), goatfish (Mullidae) and groupers (Serranidae) but also capture other non commercial but ecologically important herbivores especially parrotfish (Scaridae), surgeonfish (Acanthuridae), moorish idol (Zanclidae) and butterflyfish (Chaetodontidae).

Compared to other fishing gear frequently used on coral reefs, traps also catch the greatest proportion of herbivores (McClanahan *et al.* 2008a) and target a high proportion of species likely to be affected by climate change induced bleaching and that are key for the recovery of corals (reef scrapers/excavators and grazers) (Cinner *et al.* 2009a). However, traps are efficient and cost-effective (Miller, 1990) representing a a strong candidate for management restrictions to help reduce the catch of juveniles, herbivore mortality and hence increase ecosystem health and resilience.

Escape Gaps as a Management Option in trap fisheries

Studies on **trap mesh selectivity** have demonstrated that mesh size is a determinant of catch rates and the size composition in fish traps (Mahon & Hunte 2001). The management of coral reef trap fisheries has traditionally focused on the use of larger mesh sizes to reduce the catch of juveniles (Sary *et al.* 1997; Robichaud & Hunte 1997) but this approach has a major limitation of finding an optimal mesh size to maximize the yield and respect the maturity schedules for the full range of exploited species (Mahon & Hunte 2001). Also, increasing trap mesh size resulted in short-term loss in revenue for fishers (Mahon & Hunte 2001; Baldwin *et al.* 2002) and therefore becoming a difficult measure to implement and monitor (Baldwin *et al.* 2002).

Escape gaps started to be considered as a management option to increase size selectivity in crustacean trap fisheries back in the 1950s (Templeman 1958). Several studies examined its effectiveness in lobster pots (Brown 1982; Lanteigne *et al.* 1995; Treble *et al.* 1998; Clark & Sussex 2007), and in crab collapsible pots (Jirapunpipat *et al.* 2008; Boutson 2009). Nowadays, most commercial lobster fisheries worldwide have regulations requiring crustacean traps to have escape-gaps with a size that can conciliate reduced catch of undersize lobsters and maintenance of CPUE (catch per unit effort) of legal-size lobsters (Treble *et al.* 1998).

The inclusion of escape gaps in coral reef traditional fish traps as a method of reducing the catch of juveniles and narrow-bodied species without compromising the fisherman's income is a promising method that has recently won the attention of various conservation efforts. Munro (2003), conducted experiments in the Caribbean trap fisheries and concluded that rectangular escape gaps were effective in releasing undersized fish but do not significantly decrease the catchability of target species. A study in an overfished Caribbean coral reef (Johnson 2010) revealed that gated traps caught significantly fewer bycatch fish and increased the mean length of captured fish, maintaining the total market value of the catch. The fisheries department of the

Caribbean island of Curaçao has recently proposed the required use of short escape gaps (20 × 2.5 cm) in all fish traps (Johnson 2010).

A recent study (Mbaru & McClanahan 2012, submitted) carried out in a Kenyan mixed-use fishing ground evaluated the income of the two trap designs (traditional and gated traps). The authors recorded a 40% bycatch reduction and an increase in overall catch value with use of escape gaps. Gate traps can potentially yield higher valued catch due to the strong positive relationship between fish size and price per unit weight (McClanahan 2010) and density-dependent behavior of fish in traps (Mbaru & McClanahan 2012, submitted). Contrasting with the results from increasing mesh size experiments, gated traps showed no immediate economic loss for the fishermen and, by decreasing the catch of juveniles and key herbivores represents a plausible solution to create long term achievements in terms of fisheries productivity and improved ecosystem health.

The present study has two overall objectives; the first one is to assess gear selectivity between traditional and gated traps in a multi-species and multi-gear artisanal fishery within a marine reserve in Southern Kenya. The selectivity is evaluated in terms of species composition and size, total catch, functional groups and trophic level of the catches. The second objective sought to evaluate the practical implementation of a gear based management approach at a community level, considering social and ecological implications of escape gap use in order to support appropriate low-cost fisheries restrictions.

Methods

Study Area

The work was undertaken on the Mpunguti Marine National Reserve, located at the southern tip of the Kenya coastline in the District of Kwale, which borders Tanzania. In 1973 the whole area was considered a strict non take Marine National Park. However, the pressure from local communities due to the loss of fishing grounds changed the directive and the park boundaries were revised and reallocated. Mpunguti islands were opened back to fishing and it became a Marine National Reserve in 1978, only allowing traditional fishing methods (hand line - *mshipi* and basket traps - *malema)* to take place.

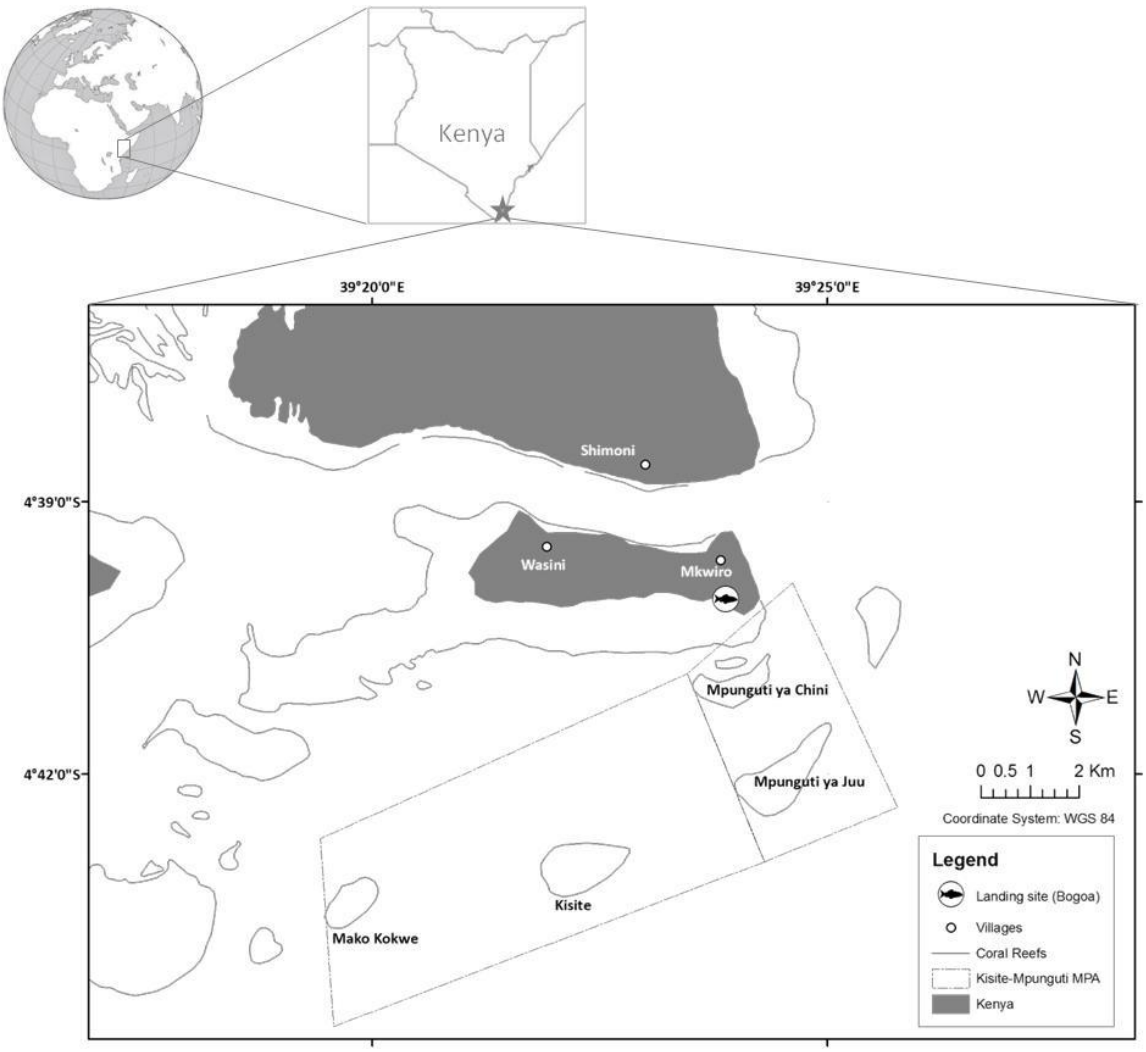

Fig.1. Map of the study area showing the location of the landing site (Bogoa), the Mpunguti Marine Reserve (Mpunguti ya chini and Mpunguti ya juu) and Kisite Marine National Park (Mako Kokwe and Kisite).

Kisite Marine Park is the biggest Marine Park (28 km²) while the Mpunguti Reserve is the smallest Marine Reserve in Kenya (11 km²). The reefs themselves represent a series of 4 small islands (Kisite and Mako Kokwe in the park and Mpunguti ya juu and Mpunguti ya chini in the reserve) together with adjacent small patch reefs on a shallow shelf 10 to 20 m deep (Watson & Ormond 1994). The area is rich in valuable natural and tourism resources: coastal forests, patch and fringing coral reefs, sea grass beds, reef flats, sand bars, important bird areas and mangrove forests which support a highly diverse ecosystem. The area is also home for a resident population of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*), Indo-Pacific humpback dolphins (*Sousa chinensis*) and seasonal migrations of humpback whales (*Megaptera novaeangliae)* (Perez *et al.* 2011). In addition, it serves as an important habitat to the critically endangered *Hawksbill Turtle* (*Eretmochelys imbricata*) and the endangered Green Turtle (*Chelonia mydas) (IUCN 2012)*.

The entry fee in the Kisite Mpunguti Marine Protected Area (KMMPA) is currently USD20 for non resident adults. Its administrative capacity has been improving over time and currently comprises a regional headquarters office, housing, a visitor center and patrol boats (Weru *et al.* 2001). More recently, an observation tower was constructed at Lower Mpunguti to observe the seascape, monitor wildlife and illegal activities such as fishing (KMMCA Management Plan 2011-2021). Since the economy of the park is strongly dependent on tourism, it is also significantly affected by its instability. Civil clashes on the coast, driven by cross-border conflicts and insecurity during election campaigns, as well as torrential rains during El Niño phenomenon, have decreased the number of visitors to the area. This decline was notorious after the 2007 post election violence in Kenya, where visitors to KMMPA decreased from about 60 000 visitors in 2007 to 28 000 annual visitors in 2008. Numbers in 2010 were approximately of 50 000 (KWS data, 2010) but the present-day political situation and the imminence of elections are expected to reduce the national tourism once more.

Mkwiro Fishing Community

Mkwiro is an Islamic fishing community located in the east side of Wasini Island (Fig.1 & 2). Its population grew from approximately 400 inhabitants and 60 households in 1986 (Wynne-Jones & Walsh 2010) to 1260 residents and 135 households in 2012 (Mkwiro Dispensary data, 2012). The people of Mkwiro still identify themselves as Shirazi (from Shiraz in Persia) and speak their own Swahili dialect, Chifundi.

Traditional fishing (basket traps and hook and line) using dugout canoes is the main source of livelihood in village. After the set up of the marine park, Mkwiro fishermen were denied access to traditional fishing grounds and their perception of the MPA was strongly negative: *“the sea is God's sea; it is the people's sea; not the government's* “ (Malleret-King 2000). This village was very proactive in the fight for the reallocating the marine park’s boundaries; in the 1970s fishers made roadblocks and met with park authorities, until they finally succeed to shift the limits of the no-take area. The Mpunguti Islands are currently their major fishing ground, although the neighboring villages of Kibuyuni, Kichangani and Shimoni also utilize the reserve to fish (Emerton 1999; Malleret-King 2000).

Mkwiro fishers are highly grounded on tradition and strongly disapprove destructive fishing methods (such as seine, ring nets and spear guns). In the 1990s, they collectively set a ban on Wapemba (Tanzanian migrant fishers from the island of Pemba) seine netters; an action that was later sustained by the fisheries administration (Malleret-King 2000).

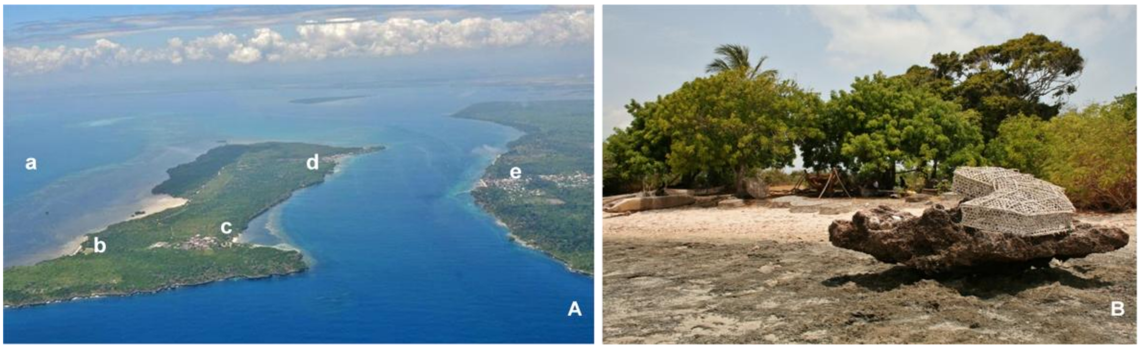

Fig.2. **A**- Aerial view of Wasini Island (photo credit *ShimoniReef*): a- Kisite and Mpunguti Marine Protected Area (partial view), b- Bogoa landing site, c- Mkwiro, d- Wasini, e- Shimoni (mainland). **B**- Bogoa, view from the shore.

Fishing is considered a male activity in the village, but women also harvest octopus, cowry shells and are currently engaged in a seaweed farming project, encouraged by local NGOs with the support of governmental agencies. (Fig.3). Although younger men are now finding jobs in local tourism, the dependence on this activity is still low in this community, compared to the tourism-orientated neighbor village of Wasini (where 36% of the families depended on tourism contrasting with only 10% in Mkwiro) (Malleret-King 2000).

Fig.3. Marine resource utilization by women: **A** – Octopus catch; **B**- Seaweed farming; **C**- Cowry shells harvesting.

A study on fishing communities adjacent to the Kisite Mpunguti MPA reported that although the choice of gear was driven by a combination of economic, physical (distance from landing site) and seasonal factors, traps were the most attractive gear, particularly for older fishers (Malleret-King 2000). The hexagonal shaped fish traps are deployed from canoes and constructed of wooden frames meshed with reed strips and weighted with stones on the side (Kaunda Arara & Rose 2004a). A trap has one funnel-shaped door and an underside opening to remove the catch. Fish enter the trap through the funnel entrance but cannot escape due to the constriction at the internal end of the funnel. Traps can be classified as either big (2 x 1.3 x 0.3 m) or small (1.2 x 1 x 0.2 m) with a mesh size of 5 and 3 cm and volumes of approximately 0.8 m^3 and 0.2 m^3 respectively (McClanahan & Mangi 2004). Smaller traps are deployed in shallow areas while big ones are laid at depths up to 30m (Glaesel 1997).

In Mkwiro, the use of basket traps is dominated by the older fishermen who construct their own traps and use a variety of baits, consisting of mixtures of seaweed and green and red algae. The catch is checked and removed on a daily basis, during low tide, with the bait replaced and the trap reset in the same place or nearby area. The trap is only brought to shore for maintenance if it is damaged or shows signs of algae overgrowth. Usually, traps can stay in the sea for about 30-40 straight days and have a fishable life of about 3-6 months (fishermen *pers. comm*.). Fishermen use paddles or sails to travel in small dugout canoes (carrying 1-3 fishermen) and set the traps in the Mpunguti Marine Reserve (Mpunguti ya chini and Mpunguti ya juu) on coral reef habitats, seagrass meadows or sand flats (McClanahan & Mangi, 2001). Fishing occurs all days except Friday (Muslim's holy day) and through severe weather conditions. The fish is gutted onboard and sold to fish dealers in Mkwiro. Because there is no electricity on the island, local fish dealers do not have access to ice and need to cross to the mainland (Shimoni) with no delay. Fish is then resold in Shimoni for local use or continues its journey to more densely populated areas, like Mombasa.

Fig.4. **A**-Mkwiro fishermen setting up a fish trap in the Mpunguti Marine Reserve. **B**- Trap deployed in a sea grass habitat.

Migrant foreign fishermen, coming from Pemba and mainland Tanzania, also exploit the local marine resources as they establish themselves in the village during seasonal fishing periods. These migrations are closely associated to the monsoon seasons and have taken place since historical times

This area is representative of the multi-gear, multispecies artisanal coral reef fishery in Kenya. The social characteristics of this fishing village, namely its high dependence on fishing, its profound roots in tradition and disapproval of destructive fishing methods made it the ideal place to try the implementation of the gated traps at a community level, In order to increase the sustainability of the local fisheries.

Trap use and design

From September 2011, several meetings were held in Mkwiro between researchers, the local Beach Managements Unit (BMU), the park authorities (Kenya Wildlife Service –KWS) and trap fishermen to introduce the new gated trap concept and explain the aims of this project. Results from previous experiments involving gated traps in Mombasa Marine National Reserve were presented and the trap fishing community showed interest in participating in the project. Some fishermen agreed to modify their own traditional traps with 3cm wide x 30cm escape gaps to evaluate the results of the modified traps on their catch. Fishermen also agreed to allow researchers to regularly sample their catch during the following months.

In January 2012, a group of three fishermen modified one of their traps, using local material, with the guidance of a fishery researcher and deployed it in the sea. Modified traps consisted of two escape gaps inserted at either side of the V-shaped corners of the traditional fishing traps (Fig.4.).However, due to rough weather conditions, the gate was damaged and it was decided to start employing rebar metal gates. These gates are more robust and can be inserted into existing traps with an installation time of about of 30 minutes per trap. Several rebar gates were purchased and distributed amongst the trap fishing community. By March 2012, there were 7 modified traps being used in the Mpunguti Marine Reserve involving 7 fishermen. Fishers used their regular fishing grounds, which varied to a certain extent in terms of reef slope, depth (around 5-15m) and benthic substrate. No changes of the fishing method were required, as gated traps were used in the exact same manner as the traditional traps.

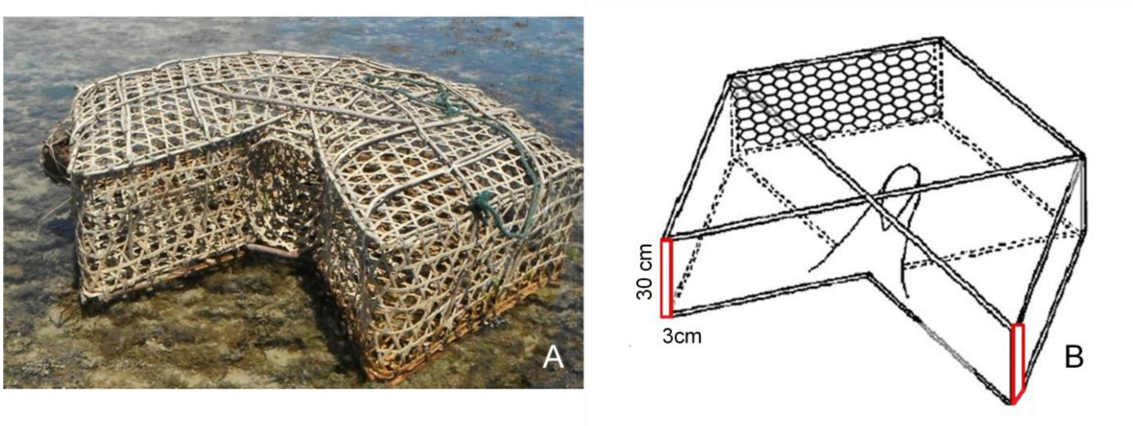

Fig.5. Two types of fish traps used in this experiment: **A -** control (traditional African basket trap) and **B** – modified basket trap with escape gap (Adapted from image in Johnson 2010).

Data collection

This study was carried out during the period from January 23rd to April 23rd 2012, throughout the northeast monsoon season and includes a total of 65 visits and 213 sampling occasions at Bogoa landing site (Fig. 1). Standard sampling methods were used and the analysis comprises a total of 2060 sampled fish, from 93 species belonging to 26 families.

We examined fish landings by approaching and asking consent from fishers as they arrived from fishing activities. Fishermen that were using both trap designs were asked to sort the catch into separate bags, separating traditional from gated traps catch.

Landed fish were identified to the species level (Allen *et al.* 2003) and individual standard lengths (tip of snout to end of last vertebra) and maximum body depth was measured to the nearest 0.1 centimeter. Individual fish were also weighed to the nearest 0.1g and the total catch was weighed to the nearest 0.1kg using spring scales. The entire catch was sampled whether the fish was for market (sold to local fish dealer) or for direct home use, and for large catches a randomly collected sub-sample (around 10% of the catch) was measured, to avoid delaying the fishermen (McClanahan & Mangi 2004). The total catch, species composition, type and number of traps (traditional or modified), fishing ground and use of catch (market – commercial species or home use – non commercial species) were recorded for each sampling day. Fish were photographed using a Canon EOS 400D digital camera and a species catalogue was compiled to cross check species identification and to assist in further data collection in the field.

In order to map the locations used by trap fishers within the reserve, a full day mapping trip was performed within the Mpunguti Marine Reserve on board a local vessel. This trip had the guidance of an experienced fisherman from Mkwiro who distinguished the various fishing grounds used by the local fishermen. The coordinates were recorded using a handheld GPS Garmin Etrex.

Data handling and Analysis

Using SPSS statistics v.17, we tested for differences in catch biomass, and catch composition between trap types (traditional and gated traps), fishing ground locations (Mpunguti ya juu and Mpunguti ya chini) and trap distance to the park boundary (Close 0-500m, Medium 500-2000m, Far ≥2000m) using a univariate general linear model, treating trap type, location and distance as fixed factors. Data on biomass was normalized by $\log_{10}$ transformation. Distances to the marine park were calculated using *ArcGIS* 9.3 and categorized in 3 levels based on the distance to the park boundary and to the park nearest reef.

Multivariate statistical analysis (correspondence analysis) was performed (using BiodiversityPro v.2) to determine the degree of similarity of catch composition amongst trap type and location. This analysis was only performed with species which accounted for more than 1% of relative abundance (19 species). The technique ordinates gear based on the number of individuals caught for each species, for each location, and plots them in two dimensional space (McClanahan & Mangi 2004). The distance between gears in the graph is a measure of the similarity in their species selectivity.

The mean trophic level was considered at species level, based on diet composition data compiled in FishBase families (for the complete set of values consult appendix II). The mean trophic level of the catch for each gear type (K) was calculated as:

Eq (1) $$TL_K = \sum_{i=1}^{m} Y_{ik} TL / \sum Y_{ik}$$

where Y_{ik} is the catch of species *i* in gear *k*, TL is the trophic level of species *i* for *m* fish species (McClanahan & Mangi 2004).

Standard length distributions of the dominant species in the catch (*Siganus sutor*) for both trap types were compared by the Kolmogorov-Smirnov test implemented in SPSS. Additionally, we compared the proportion of immature individuals (≤Lmat*)* in the both catches. Theoretical maturity length (Lmat) was taken from Hicks & McClanahan 2012 and recalculated using length-length relationship (SL=0 + 0.846 * TL) (http://www.fishbase.org/) and considered as the size that will allow the fish to gain significant biomass to spawn at least once in its life. Furthermore, and in order to investigate the effects on the retention properties of the traps with escape gaps on Siganus sutor, the logistic selectivity model was fitted to width morphometric data

Eq (2) $$S_i = 1 / (1+e^{(-b(Li-L50))})$$

S_i is the proportion retained for width class I, b is the slope and L50 is the width at which 50% of the fish are retained. It was assumed that the traditional trap without an escape gate was not size selective, and therefore the observed S_i was calculated as the number of fish caught in the traps with an escape gate divided by the sum of the fish caught by both types of trap for each width class:

Eq (3) $$S_{iobs} = N_{Igated} / (N_{Igated} + N_{Itraditional})$$

Solver (Microsoft Office Excel 2007) was used to estimate the parameters of the logisitic model my minimizing the sum of squares of the difference between the observed and expected size selectivities:

Eq (4) $$\text{Min } \Sigma(S_{iobs} - 1/(1+e^{(-b(Li-L50))}))^2$$

Results

Over the course of the study period, we had 65 visits to the landing site and 213 sampling occasions to examine the catch of 21 trap fishers from Mkwiro (Table 2). The number of fishers in each canoe varies from 1 to 3, depending on the distance to the fishing ground, size of the canoe and number of traps to check. Fishermen owned from 1 to 15 traps and the seasonal migrant fishermen from the island of Pemba (Tanzania) owned the greatest number of traps (4 fishermen, 2 canoes and 30 traps).

Table 2 – Fishermen who participated in the study, number of sampling occasions and fish sampled according to fishing ground location (Mpunguti ya chini and Mpunguti ya juu) and trap type (Traditional and Gated trap).

Fishing community				Fish sampled				Total # of fish sampled
				Location Mpunguti ya Chini		Location Mpunguti ya Juu		
Fisherman	# people in the canoe	# of traps owned	# sampling occasion	Traditional Trap	Gated Trap	Traditional Trap	Gated Trap	
Hamadi Salim	2	15	4	-	-	61	-	61
Hatibu	1	3	3	25	-	-	-	25
Juma Hassan	1	1	1	-	-	6	-	6
Keia Vuya	1	2 to 4	7	63	-	-	-	63
Kiboga	2	6 to 10	34	-	-	188	99	287
Makame	2	5	1	9	-	-	-	9
Mchasa	1	4 to 7	4	42	-	-	-	42
Mohamed	2	15	2	-	-	40	-	40
Mohamed Ali	2	2 to 6	21	239	-	-	-	239
Mohamed Mtwana	2	2 to 5	51	-	-	435	80	515
Mwalimu	3	5 to 10	79	7	46	405	227	685
Saidi Mbaruko	2	15	6	-	-	88	-	88
Total	**21**	**75 to 96**	**213**	**385**	**46**	**1223**	**406**	**2060**

Catch Composition

The sampling included a total of 2060 reef-associated fish representing 93 species of 26 families (for the complete species list consult appendix I). This high diversity of the catch is representative of the multi-species nature of the Kenyan artisanal fishery. On the other hand, and despite the high number of species caught, catches were dominated by only a few species (Fig. 6 and 7). Six species accounted for over 60% of the total catch: *Siganus sutor* (Valenciennes, 1835), *Acanthurus tennenti* (Günther, 1861), *Scarus ghobban* (Forsskål, 1775), *Zanclus cornutus* (Linnaeus, 1758), *Parupeneus barberinus* (Lacepède, 1801) and *Calotomus carolinus* (Valenciennes, 1840).

All fish caught in the traps are kept and either sold to local fish dealers (commercial fish species) or are taken directly home for consumption (non-commercial species or small juvenile fish). The only fish discarded was the puffer fish (family Tetraodontidae) as it is considered poisonous. The most important local species is by far, the African white-spotted rabbitfish (*Siganus sutor*) which alone comprises 62% of the total commercial species and almost 20% of the catch taken for home use. Other important commercial species include the dash-and-dot goatfish (*Parupeneus barberinus)* (5%), the blue-barred parrotfish (*Scarus ghobban*) (4%) and the sky emperor *(Lethrinus mahsena)* (3%). In terms of non commercial species, and apart from rabbtifish, the double-banded and the brown surgeonfish (*Acanthurus tennenti and Acanthurus nigrofuscus*), the moorish idol (*Zanclus cornutus*), the one-knife and spotted unicornfish (*Naso thynnoides and Naso brevirostris*) and the threadfin butterflyfish (*Chaetodon auriga*) represented the most common species. In order to decide which fish is not sold to fish dealer ("*Kitoweo*" – *reward from the sea*), the fishermen takes in account mainly the type of fish (market price) and the size of fish (for example juveniles of high value fish, like the rabbitfish).

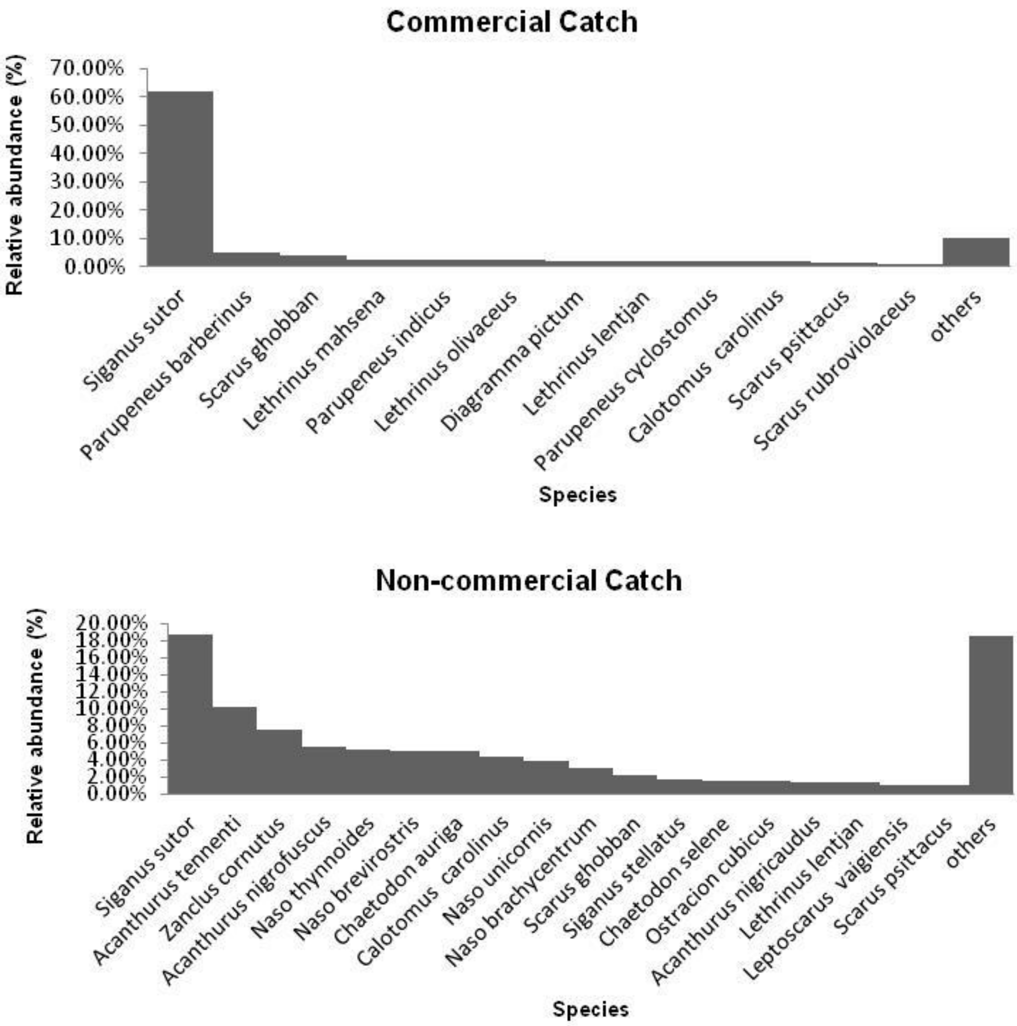

Fig.6. The relative abundance of the most common fish species based on the analysis of 2060 fish species and its final destination: market (commercial catch) or home (non commercial catch).

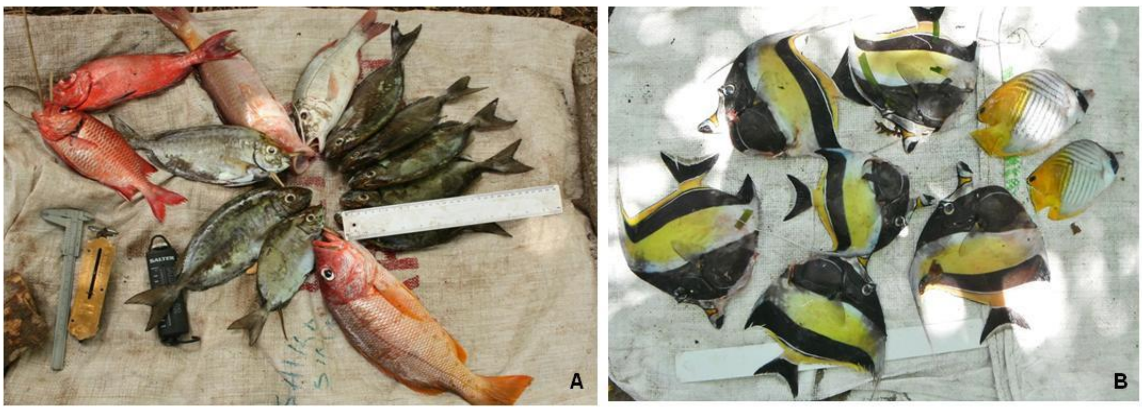

Fig.7. Example of a day's catch and separation of commercial (A) and non-commercial (B) species.

Mean Biomass of Catch per trap

We tested for differences in catch biomass, comparing it across the 2 trap types, 2 locations and 3 distance classes using a univariate general linear model, treating trap type, location and distance as fixed factors.

Table 3 – Variables and categories used to compare mean mass catch per trap (N: sample sizes). Distance to park was calculated based on the trap site distance to the park boundary.

Variables	Categories	N
Trap type	Traditional	161
	Gated	78
Location	Mpunguti ya chini	56
	Mpunguti ya juu	183
Distance to park	Close	73
	Far	63
	Medium	103

In terms of total catch per trap (total kg/trap), no significant difference was found between trap types (gated *vs.* traditional) or amongst different trap distances to the marine park (Table 4). However, there were significant differences concerning trap main fishing location (Table 4; $F=4.22$, $p\leq0.05$). Traps located in Mpunguti ya juu had a mean catch per trap of approximately 1.46 ± 0.07kg/trap and in Mpunguti ya chini of 0.985 ± 0.09kg/trap (representing a 32.8% decrease).

Table 4 –Results of ANOVAs testing differences between trap types, trap location and distance to Kisite Marine Park. Significance levels: *p < 0.05 and ***p < 0.0001.

	F	df	p	Sig.
Total Catch (kg)/trap				
Trap type	3.29	1	0.07	
Location	4.22	1	0.04	*
Distance	0.2	2	0.81	
Non-commercial species (kg) /trap				
Trap type	140.31	1	0.000	***
Location	0.18	1	0.67	
Distance	0.94	2	0.45	
Commercial species (kg) /trap				
Trap type	5.26	1	0.02	*
Location	2.81	1	0.09	
Distance	0.74	2	0.47	

On the other hand, when we analyzed the catch composition, a clear significant difference (Table 4; F=140, p≤0.0001) was established between the catch of gated and traditional traps. Gated traps caught an average of non commercial species (species with low or no marketable value, juveniles and coral reef narrow bodied species) of 0.19 ± 0.039 kg/trap (Fig. 8), contrasting with 0.30 ± 0.020 kg/trap (Fig. 8) caught by traditional traps. Therefore, gated traps decreased the catch of non-commercial species by 37.4%. On the other hand, the catch of commercial species in gated traps significantly increased (Table 4 and Fig. 8; F=5.6, p≤0.05) as to compensate the loss of smaller, low-value fish and maintain the CPUE. Mean commercial catch values were 0.97 ± 0.055 kg/trap and 1.28 ± 0.11 kg/trap (Fig. 8) for traditional and gated traps respectively.

Fig.8. Mean (±SE) mass of catch presented for each trap type (kg/trap). Catch was analyzed in terms of total biomass inside the trap and catch composition regarding non commercial and commercial (saleable fish) biomass per trap. Significant differences from controls (traditional traps) are indicated *p < 0.05 and ***p < 0.0001.

Comparative Catch Composition of gated and traditional traps

Based on the number of individuals caught for each species (and only considering species which represented more than 1% of the total catch) and the fishing ground location, correspondence analysis showed a more uniform catch of gated traps when compared to the traditional ones, especially in Mpunguti ya juu (Fig. 9). Plots that are more spread out from the rest are distinguished by differences in their species selectivity.

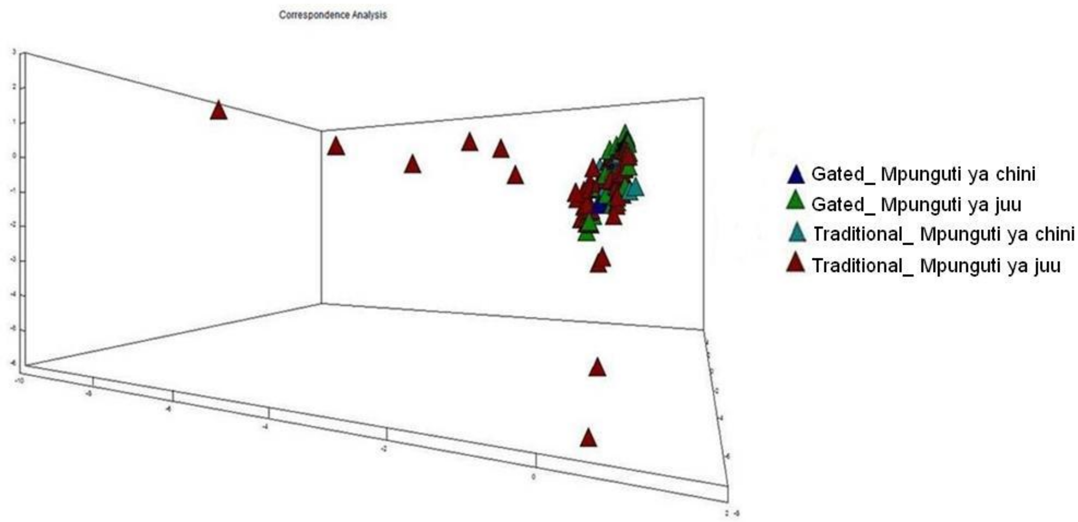

Fig.9. Correspondence analysis plot for the two gear types and two major fishing grounds, in relation to the number of individuals caught for each species accounting for more than 1% of the total catch.

Trophic Level

Both of the gear types had low mean trophic levels (2.3 - 2.5), since they mainly target herbivorous species. The mean trophic level of the catch differed significantly between traditional and gated traps, (Fig. 8; $F=19$, $p \leq 0.0001$).

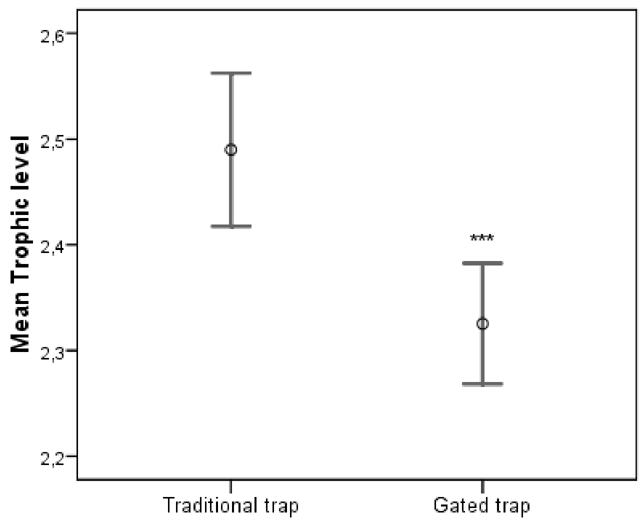

Fig.10. Plots for the mean trophic level for each of the gear types. Significant differences from controls (traditional traps) are indicated ***$p < 0.0001$. Bars indicate standard errors 95% CI.

Gated traps demonstrate a lower mean trophic level (2.33 ± 0.029) than the traditional ones (2.50 ± 0.022). This effect might be explained by the decreased catch in the gated gears of common narrow bodied coral reef species, such as butterfly fish (Chaetodontidae), Moorish Idol (Zanclidae), which are micro-invertivores (trophic level varying from 2.7 and 3.3, Appendix II) and an accentuated catch of rabbitfish (*Siganus sutor*), important grazers with a trophic level of 2 (Appendix II).

Length Frequency Distribution

We compared the length frequency distributions of the dominant species in the catch, the African white-spotted rabbitfish *Siganus sutor*, in both traditional and gated traps. We then evaluate the number of fish below Lmat (length at which 50% of the fish became mature) to find the proportion of immature individuals in the catch. Results indicated that, for this species, gated traps significantly increased the mean length (by 13.45%) and weight (by 32.2%) of captured fish (Table 5) and decreased the proportion of the catch under length at first maturity from 19.9% (traditional traps) to 3% (Table 5; Fig. 11).

Table 5 – Data by gear type for standard length, weight composition and proportion under lengths at first maturity (≤Lmat) for the African white-spotted rabbitfish (*Siganus sutor*).. Mean, standard error (SE) and percent different from control (traditional) presented for each trap type. Means significantly different from control are indicated ***p < 0.0001.

Siganus sutor	Mean Length (cm) ± SE		Mean weight (g) ± SE		≤Lmat	
	Traditional	Gated ***	Traditional	Gated ***	Traditional	Gated
	20.8 ± 0.17	23.6 ± 0.22	263.9 ± 5.52	349.1 ± 10.68	19.90%	3.00%
Difference	increase 13.45%		increase 32.2%		decrease 16.9%	

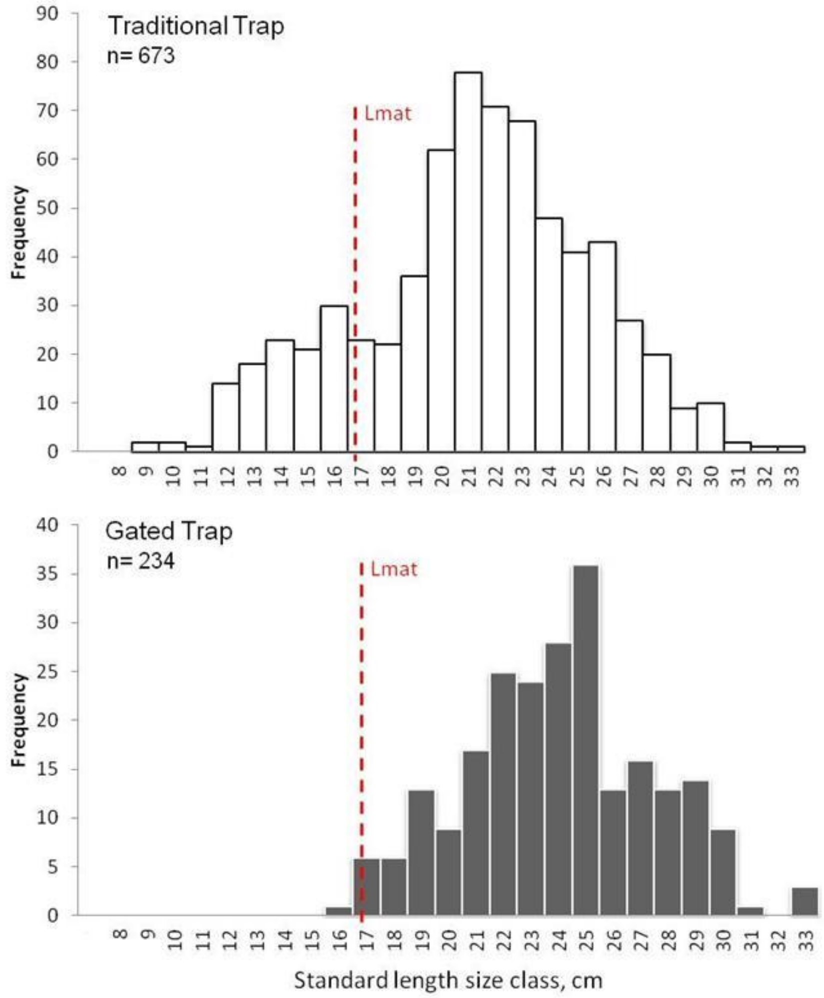

Fig.11. Standard length distribution of *Siganus sutor* in both trap types (n=number of samples). The results of a Kolmogorov-Smirnov test for comparison of two data sets showed a maximum difference between the cumulative distributions, D=0.3080 with a corresponding $p \leq 0.0001$. Lmat value taken from Hicks & McClanahan 2012 and recalculated using length-length relationship (SL=0 + 0.846 * TL) from FishBase.

Size-Selectivity Curves

We calculated a size-selectivity curve for *Siganus sutor* in the gated traps to investigate its retention properties, using width morphometric data which limits the ability of the fish to pass through the gap.

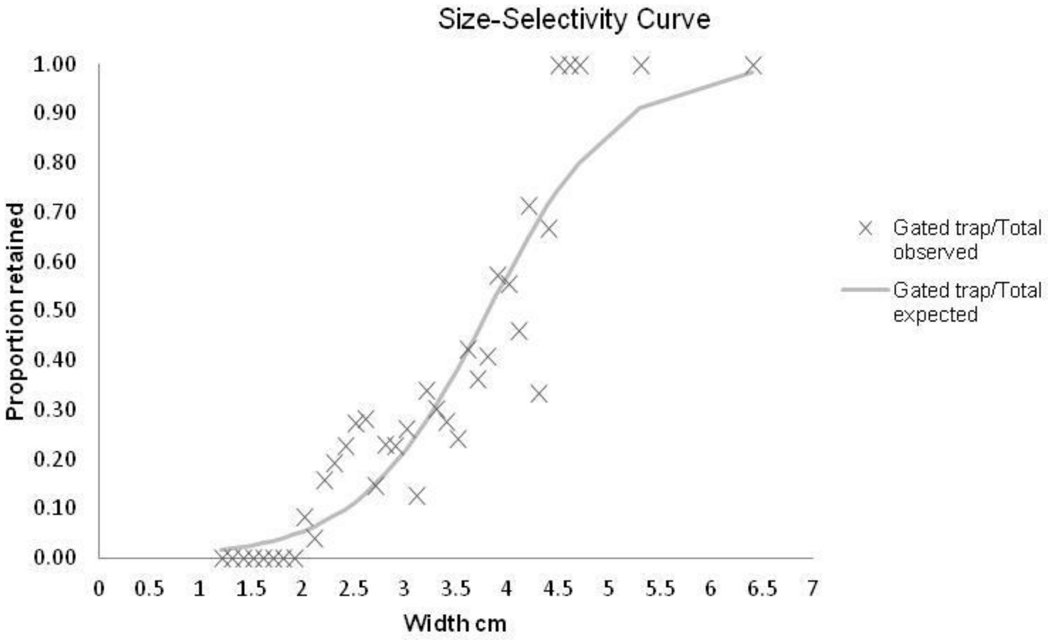

Fig.12. Gear selection ogive representing the size-selectivity curve for the escape gap size of 3cmx30cm, obtained from logistic modeling of the *Siganus sutor* width data. Curve was calculated from Eq (2) using the parameters b =1.5690 and L50 =3.8145.

The selection curves fit a logistic-type model adequately. A direct relationship was established between the fish width and the probability of being retained by the trap. L50 was 3.815 cm, meaning that 50% of the fish of this length will be retained by the trap.

Discussion

The multi-species nature of the Kenyan artisanal fishery (McClanahan & Mangi 2004) is well demonstrated here by the high diversity of species caught in the basket traps. During the experiment, 2060 species were sampled, representing 93 reef and sea grass associated species, from 26 families (for the complete species list consult Appendix I). However, and despite the great diversity, the catch was dominated by only 5 families, namely Siganidae, Acanthuridae, Scaridae, Mullidae and Lethrinidae, that constituted 85% of the total catch. These findings are typical for East Africa tropical multispecies fisheries and confirm the results of trap gear selectivity research conducted in Southern Kenya (McClanahan & Mangi 2004; Mbaru & McClanahan 2012, submitted). Fishermen keep all fish caught in the basket traps (only fish from the family Tetraodontidae were discarded as inedible) and separate the catch into marketable species (high-value fish, sold to fish dealers at the landing site) and home catch (taken home for consumption). By far, the most important commercial species in this area is the African White-Spotted rabbitfish (*Siganus sutor*), the primary species taken in the local trap fishery and much appreciated by coastal Swahili communities. It represents over 60% of all marketable species, followed by members of the families Mullidae, Scaridae and Lethrinidae (Fig. 6). On the other hand, the home consumption catch is much more diverse (comprising over 70 different species) and for the most part includes smaller sized individuals of *Siganus sutor* and narrow bodied fish, members of the Acanthuridae family (surgeon and unicornfish), Zanclidae (Moorish Idols) and Chaetodontidae (Butterflyfish) (Fig. 6). It is fundamental to include the non-commercial catch in fisheries analysis since it is not taken to local fish dealers (where the BMUs are conducting data collection) or to the markets and will not be reported in the total daily catch and weight. Consequently, it will result in an underestimation of the local CPUE and might hide the capture of low-market value but ecologically important species. This study revealed that about 25% of the total catch of the traditional African basket traps did not reach the market which is in conformity with Glaesel (1997) estimates of 30% of the total catch taken directly home.

Overall catch per unit effort (kg/trap) significantly differed among fishing ground locations (Mpunguti ya juu and Mpunguti ya chini) (Table 4; $F=4.22$, $p \leq 0.05$). Traps located in Mpunguti ya juu had a mean catch per trap of approximately 1.46 ± 0.07kg/trap and in Mpunguti ya chini of 0.985 ± 0.09kg/trap, representing a 32.8% decrease. This can be a consequence of a greater exploitation of Mpunguti ya chini, since its closer to different fishing villages, therefore having a higher fishing effort, less diverse catch and fish biomass. However, the total catch analysis in this study did not reveal any significant difference among different traps located at different distances to the marine park boundary (from 0 to ≥2000m; Table 4). Local studies developed by Watson and Ormond (1994) found strong evidence that the abundance of commercial species was significantly higher inside the park than in the reserve. The authors hypothesized that the harder fishing inside the reserve whilst fully protecting the park revealed a more productive system as it encouraged adult fish to enter the fishery by migrating into the reserve from the park (spillover effect). However, Mpunguti and Kisite reefs are patchy reefs separated by a minimum of 1.7km of deeper water and sandy bottom, restricting the movements of coral reef fish, compromising the spillover effect, which can be strongly species specific (Kaunda-Arara & Rose 2004).

Even though overall CPUE values did not significantly change between the traditional basket traps and the gated traps, significant differences were found in terms of catch composition. Gated traps caught significantly less non-commercial species (Table 4; Fig. 8; $F=140$, $^{***}p \leq 0.0001$), with a marked decrease of 37.4 %, and the catch of marketable species increased (Table 4; Fig. 8; $F=5.6$, $^{*}p \leq 0.05$) compensating for the loss of home catch species and maintaining the CPUE. This is important, since high catch rates alone will not grant the best economic profit if the trap is filled with low-valued species (Luckhurst & Ward, 1987). The overall catch per unit effort was not affected because the catch of high value wide bodied target species compensated for the loss of smaller species, resulting in an additional economic value of gated traps.

This compensation might be explained by density-dependent behavior of fish (Mbaru & McClanahan 2012, submitted). Munro (1974) and Miller (1979) recognized that catch/trap was asymptotic with time, given that the trap reached a saturation point with decreased rate of entry with increasing catch for crabs. This way, crowded traps inhibited the further entry of individuals, reducing entry rates. Moreover, studies on different selection processes in Antillean traps, carried out by Gobert (1998) hypothesized that the presence of large fish induced in small fish a “fleeing behavior”, forcing their way out through the trap mesh. Therefore, the probability of escapement of small fish in more crowded traps is higher and consequently, densely populated traps would attract larger fish in, rather than small individuals. Also, and given that traps constitute a passive mode of capture, the specific behavior of the species can play a vital role in the capture process (Fogarty & Addison 1997). Specific behavioral interactions between fish inside and outside of entrapment gear may result in conspecific attraction or heterospecific avoidance, influencing catch magnitude and composition (Munro 1971, 1974; Miller 1979; Luckhurst & Ward 1987). We then hypothesize that the "saturation point" of the traps, a "fleeing behavior" and "conspecific attraction" might be involved in explaining the higher catch of commercial species inside the gated traps. Direct in situ observations of fish ingress and escapement could give a better clarification on the selection properties of gated traps and are, therefore, recommended. Furthermore, and because trap shape and size (volume), soak time, bait type and quantity and the spacing between traps affect catch rates (Munro 1974; Miller 1979; Williams & Hill 1982) this variables should be incorporated in further controlled experiments.

Multivariate analysis was useful to determine the degree of similarity in the species selectivity of the gear. The diversity of the fish caught in gated traps declined, comprising a much more uniform catch (Fig.9) due to the reduced capture of narrow bodied non-commercial species, like surgeon fish, unicorn fish, moorish idol and butterfly fish. Again, this effect was more notorious in Mpunguti ya juu, where the catch composition of traditional traps shows greater reef fish diversity since this fishing ground appears to be less overfished than Mpunguti ya chini, of easy access and higher fishing pressure.

In terms of trophic level analysis, both traps types showed low mean trophic levels (2.3 - 2.5), probably reflecting the maturity and history of heavy fishing in the area (McClanahan & Mangi 2001). The mean trophic level of the catch differed significantly between traditional and gated traps (Fig. 10; $F=19$, $p\leq0.0001$) since gated traps presented a lower mean trophic level (2.33 ± 0.029) than the traditional ones (2.50 ± 0.022). The lower catch in the gated traps of non commercial species such as butterfly fish and Moorish idol (common narrow bodied, micro–invertivore fish, with trophic levels 2.7-3.3, Appendix II) and the increased catch of the grazer rabbitfish and scrappers parrotfish (trophic level 2, Appendix II), might explain this difference. The mean trophic level can be used as an index of sustainability in multi-species fisheries (Pauly *et al.* 2001) and management should be orientated to regulate the mix of gears to maintain a constant mean trophic (McClanahan & Mangi 2004). However, to get a full trophic composition idea of the fishery requires inclusive (sample different gears) and long length time series data.

In relation to the dominant species in the catch, *Siganus sutor*, the mean standard length and weight of fish retained in the gated traps was significantly greater than that of fish caught in traditional traps. Gated traps caught fish that were 13.45% longer and had an increase of 32.2% in the capture weight (Table 5), which represents a parallel increase in economic value, given the positive relationship between fish size and price per unit weight. Besides, as a number of the common Kenyan fisheries species grow at approximately 10 cm/year (Kaunda-Arara & Rose 2006), a one year delay in harvesting can approximately double the fish size and price (McClanahan 2010).

In addition, gated traps significantly decreased the proportion of the catch under length at first maturity for *S. sutor* from 19.9% (traditional traps) to 3% (Table 5; Fig. 11). Reducing the proportion of the catch yet to mature (length ≤ Lmat) for the most commonly landed species in the catch decreases the chances of growth overfishing (exclusion of individuals before they grow to a mature size), and potentially enhances the yield from the stock. This is particularly important as recent investigations of the life traits of *S. sutor* in Kenya south coast revealed evidence of growth overfishing and the state of the stock was considered over exploited (Hicks & McClanahan 2012).

In order to investigate the retention properties of gated traps, we calculated a size-selectivity curve for *S. sutor*, using width morphometric data, as fish width will ultimately restrain the ability to pass through the escape-gap. This gear selection ogive (Fig. 12) demonstrates that the number of fish retained increases as fish width increases, expressed by the logistic size-selectivity model (Eq. 2). $L_{50\%}$ was found to be 3.8 cm. As the escape gap size is 3 cm wide, there seems to be a divergence in the ability of wider fish to pass through the opening. However, it is important to note that this morphometric data has some limitations. Fish width was measured at the landing site, as fishermen returned from the sea and, as a general practice the fish is gutted before reaching the beach. Fishers gut the fish inside the canoes on the way back from the fishing ground, to delay the spoiling of the product, due to the lack of electricity and ice facilities on the island. It is then very likely that the width data is underestimated and that the true gear selection ogive is illustrate by a more accentuated curve and $L_{50\%}$ is reached at a smaller width (<3.88 cm) and that the width at which the maximum selectivity (selectivity = 1.0) is attained (6.5 cm) will be lower as well. Other theory relating the different "trappability" of reef fish species is supported by the work developed by Ward (1988) who proposed a "*squeezing hypothesis*", later confirmed by Robichaud *et al. (1999)*, where reef fish were able to force their way through the mesh, bending and distorting their body in their efforts to escape.

Size selectivity curves are useful for the selection of a suitable escape-gap size, helping to uncover the optimal range to provide the best compromise between low catches of undersize individuals while maintaining the retention of mature, marketable individuals. Calculation of a size-selectivity curve will then provide a universal summary of trap performance and can be further incorporated into-size based spatial models and evaluate the effect of changing escape-gap regulations on the future state of the fishery (Treble *et al.* 1998). Our results for *S. sutor* confirm that escape-gaps manifestly lowered the retention of juveniles and that the magnitude of this effect depends on the relation fish-width and escape-gap size. However, the approach used here to estimate size selectivity is considered a preliminary one and dependent on the assumption that the traditional traps are non selective. Size selectivity should be studied through experiments using traps with a series of different escape gap sizes and meshes. Comparisons of catch size frequency distributions of the different types of traps can then be used to estimate size selectivity, using for example, the SELECT model (Treble *et al.* 1998). Further studies using more accurate values of fish width, *in situ* observations, and controlled experiments using different escape gap and mesh dimensions are recommended. This will assist in finding the optimal escape gap size that fit the local ecosystem and life traits characteristics, maximizing the economic and ecological sustainability of the fishery.

Comparing our results and the results from similar studies involving the inclusion of escape gaps in tropical trap fisheries (Mbaru & McClanahan 2012, submitted, Johnson 2010) to the traditional approach of regulating trap fisheries by mesh size, escape gaps represent a key improvement in reef fishery management. Not only did they reduce the catch of juveniles, but also allowed the release of narrow bodied reef associated species (also decreasing their chances of being eaten by a predator inside the trap) without reducing the fishermen's income. The major management advantage found has to do with the absence of short-term loss in revenue for fishers when adopting the escape gap approach. This will greatly increase the likelihood of acceptance of the measure since fishermen are unlikely to adopt gears with reduced profitability without compensation (Cinner *et al.* 2009a). Consequently, the short and long-term outcome of this gear restriction practice suggests the establishment of *win-win* situations for fisher's earnings and for coral reef conservation (McClanahan pers. com.).

Given that gear selectivity will ultimately generate impacts in the structure of the ecosystem and fishery, the incorporation of escape gaps has the potential to enhance the number of mature individuals in the population and to increase local biodiversity, increasing the sustainability of the fishery. Functionally diverse and species-rich fish communities, through their distinctive feeding activities, can help to prevent or reverse phase shifts to algal-dominated reefs (Hughes *et al.* 2007), increase ecosystem productivity (Worm *et al.* 2006), promote ecosystem resilience to disturbances and climate change (Hughes *et al.* 2003, Cinner *et al.* 2009) and reduce the risk of coral disease prevalence (Raymundo *et al.* 2009). Furthermore, the escape gap management option fits well with the recently released *Kisite Mpunguti Marine Conservation Area management plan (2011-2021)* which aims to conserve the area's key ecological features through integrated and science driven management. The plan proposes improvements of the local fisheries sustainability (Fisheries Resource Management Plan, Objective 2) and increased water based tourism activities such as snorkeling and diving activities inside the Mpunguti reserve (Tourism Management Plan, Objective 4). Therefore, the addition of escape gaps in coral trap fisheries in this reserve can be a way of simultaneously increasing sustainability of fish populations and expanding eco-tourism activities as a result of higher ornamental fish biomass.

There are, however detrimental aspects of trap use that cannot be mitigated through the implementation of escape gaps. For example, fish traps when lost, continue to fish for long periods through the process of ghostfishing (Erzini *et al,.* 2008; Renchen 2011). Also, trap fisheries target herbivores that are essential to reverse coral-macroalgal phase shifts, an algal-dominated state of the reef ecosystem (McClanahan & Arthur 2001; Bellwood *et al.* 2004). Furthermore, escape gaps are not able to reduce the catch of low or no-economic value wide-bodied species, such as moray eels, trunkfish (Ostraciidae), and scorpionfish (Scorpaenidae), nor can they avoid fish being injured inside the traps (Johnson 2010). Lastly, escape gaps can only be effective when combined with other local gear restrictions, to allow for resources to be retained by different gears without compromising minimal capture size and bycatch. Limitations on gears that catch small individuals should be put into practice and management ought to promote and support a combination of gears that do not compete for similar resources or smaller fish (McClanhalan & Mangi 2004).

Finally, and in terms of execution costs and practicalities, this procedure does not require the construction of new gear, as escape gaps can be incorporated into traditional traps, and be built and attached using different materials and sizes; depending on ecological targets and financial assets. It also allows for escape gap sizes to change over time as catch rates recover and catch composition changes (Munro *et al.* 2003). Moreover, it does not imply any change in the fishing method and the installation time of the gaps in traditional traps is only about 30 min/trap. The metal gaps used in this study can be manufactured at a low price and could be provided by fishery departments or non-profit organizations and be reused when the life expectancy of the trap (3 to 6 months) comes to an end (Mbaru & McClanahan 2012, submitted).

Conclusion

A large-scale and continuing decline in coral reef health will seriously impact the livelihood of poor coastal communities and there is an urgent need to refine management strategies to effectively support the coral reef ecosystem, its fisheries and the people that depend on them. The incorporation of escape gaps in the traditional basket trap fishery represents a gear-based management approach with a vast potential to be adaptable, both in an ecological and a social context. The results of our study confirm previous findings (Munro *et al.* 2003, Johnson 2010, Mbaru & McClanahan 2012, submitted) that escape gaps are a low tech and cost-effective method of reducing bycatch and juvenile catch in trap fisheries. Furthermore, this gear based approach potentially increases social consent when compared to area closures, since it is more flexible and less intrusive for the local community (Hicks & McClanahan 2012).

To conclude, escape gaps are a significant step towards increasing the sustainability of coral reef artisanal fisheries, generating long term payoffs at different levels:

- Increasing the number of reproductively mature fish;
- Increasing biodiversity, maintaining the functional diversity of the system and protection against climate change (Hughes *et al.* 2003 , Cinner *et al.* 2009a);
- Promoting ecosystem's health, intensifying the system's resilience and protection against coral reef phase shifts and diseases (Mumby *et al.* 2006, Hughes *et al.* 2007; Raymundo *et al.* 2009);
- Sustaining fishermen's income in the short term and creating the possibility to enhance it in the long term;
- Promoting eco-tourism attractions due to high ornamental fish biomass;
- Providing the flexibility to evolve to different gap sizes as catch rates improve and catch composition changes (Munro *et al.* 2003);
- Increasing the chance of cultural acceptance and regulation compliance, as it operates on gears that are perceived to be very traditional (such as basket traps) (McClanahan & Mangi 2004);
- Offering an adaptable, low tech and low cost alternative to self-enforced community resource management (McClanahan & Mangi 2004).

Consequently, the inclusion of escape gaps is an encouraging hands-on solution to increase the sustainability of tropical reef fisheries, and the ongoing monitoring and collection of site-specific data on gear use and selectivity, will facilitate the success of an adaptive ecosystem-based management approach.

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Appendix I

Species	**English name** **(FishBase)**	**Local Kiswahili name** (Mkwiro, south coast, Kenya)
Abudefduf sexfasciatus	Scissortail sergeant	Kitata
Acanthurus dussumieri	Eyestripe surgeonfish	Ngudu
Acanthurus leucosternon	Powderblue surgeonfish	Kinanzua
Acanthurus nigricaudus	Epaulette surgeonfish	Ngudu
Acanthururus nigrofuscus	Brown surgeonfish	Ngudu
Acanthurus tennenti	Doubleband Surgeonfish	Ngudu
Acanthurus triostegus	Convict surgeonfish	Ngudu
Aethaloperca rogaa	Redmouth grouper	Hira
Balistapus undulatus	Orange-lined triggerfish	Kikande
Bodianus axillaris	Axilspot Hogfish	Pono
Caesio caerulaurea	Blue and gold fusilier	Unua
Caesio xanthonota	Yellowback fusilier	Unua
Calotomus carolinus	Stareye parrtofish	Pono
Cantherhines sandwichiensis	Sandwich isle file	Matune
Carangoides ferdau	Blue trevally	Kole kole
Caranx sexfasciatus	Bigeye trevally	Kambisi
Carcharhinus melanopterus	Blacktip reef shark	Papa
Cephalopholis spiloparaea	Strawberry hind	Kikokwe
Chaetodon auriga	Threadfin butterflyfish	Kitatange
Chaetodon kleinii	Sunburst butterflyfish	Kitatange
Chaetodon meyeri	Scrawled butterflyfish	Kitatange
Chaetodon selene	Yellow-dotted butterflyfish	Kitatange
Chaetodon trifasciatus	Melon butterflyfish	Kitatange
Cheilinus trilobatus	Tripletail wrasse	Badu
Cheilio inermis	Cigar wrasse	Ubo
Chlorurus strongycephalus	Steephead parrotfish	Pono
Coris formosa	Queen coris	Timbati
Ctenochaetus striatus	Striated surgeonfish	Kasui
Diagramma pictum	Painted sweetlips	Beha
Epinephelus caeruleopunctatus	Whitespotted grouper	Kivungwi
Epinephelus coioides	Orange-spotted grouper	Kivungwi
Epinephelus fuscoguttatus	Brown-marbled grouper	Kivungwi
Epinephelus multinotatus	White-blotched grouper	Kivungwi
Gnathodentex aureolineatus	Striped large-eye bream	Chengu
Gymnocranius grandoculis	Blue-lined large-eye bream	Chaa
Gymnothorax favagineus	Laced moray	Mkunga
Gymnothorax flavimarginatus	Yellow-edged moray	Mkunga
Heniochus acuminatus	Pennant coralfish	Kitatange
Hipposcarus harid	Candelamoa parrotfish	Kangu
Leptoscarus vaigiensis	Marbled parrotfish	Pono
Lethrinus lentjan	Pink ear emperor	Chengu

Lethrinus mahsena	Sky emperor	Tukwana
Lethrinus nebulosus	Spangled emperor	Kalua
Lethrinus obsoletus	Orange-striped emperor	Jana
Lethrinus olivaceus	Longface emperor	Ndomo
Lethrinus variegatus	Slender emperor	Sororo
Lutjanus bohar	Two-spot red snapper	Kiunga
Lutjanus boutton	Moluccan snapper	Tembo
Lutjanus fulviflamma	Dory snapper	Tembo
Lutjanus gibbus	Humpback red snapper	Chembeu
Myripristis berndti	Blotcheye soldierfish	Kijame
Naso annulatus	Whitemargin unicornfish	Sange
Naso brachycentron	Humpback unicornfish	Sange
Naso brevirostris	Spotted unicornfish	Puju
Naso hexacanthus	Sleek unicornfish	Sange
Naso lituratus	Orangespine unicornfish	Sange kitate
Naso thynnoides	Oneknife unicornfish	Kuranzi
Naso tuberosus	Humpnose unicornfis	Sange
Naso unicornis	Bluespine unicornfish	Sange
Nebrius ferrugineus	Tawny nurse shark	Papa wa usingizi
Neoglyphidodon melas	Bowtie damselfish	Nyungune
Novaculichthys taeniourus	Rockmover wrasse	Timbati
Ostracion cubicus	Yellow boxfish	Fufu
Ostracion meleagris	Whitespotted boxfish	Fufu
Paracanthurus hepatus	Palette surgeonfish	Ngudu
Parupeneus barberinus	Dash-and-dot goatfish	Mkundaji
Parupeneus cyclostomus	Gold-saddle goatfish	Mkundaji
Parupeneus indicus	Indian goatfish	Mkundaji
Parupeneus macronema	Long-barbel goatfish	Mkundaji
Plectorhinchus flavomaculatus	Lemonfish	Futa
Plectorhinchus gaterinus	Blackspotted rubberlip	Mleha
Pomacanthus imperator	Emperor angelfish	Soya
Pomacanthus semicirculatus	Semicircle angelfish	Soya
Priacanthus hamrur	Moontail bullseye	Batani
Pterois miles	Devil firefish	Chale
Scarus festivus	Festive parrotfish	Kangu
Scarus frenatus	Bridled parrotfish	Kangu
Scarus ghobban	Blue-barred parrotfish	Kangu
Scarus globiceps	Globehead parrotfish	Pono
Scarus psittacus	Common parrotfish	Kangu
Scarus rubroviolaceus	Ember parrotfish	Pono
Scarus russelii	Eclipse parrotfish	Kangu
Scolopsis bimaculata	Thumbprint monocle bream	Kanga macho
Siganus argenteus	Streamlined rabbitfish	Tafi (Tasi)
Siganus stellatus	Brown-spotted rabbitfish	Tafi (Tasi) manga
Siganus sutor	White spotted rabbitfish	Tafi (Tasi)

Sufflamen chrysopterum	Halfmoon triggerfish	Kikande
Taeniura lymma	Ribbontail stingray	Karwe
Tetrosomus gibbosus	Humpback turretfish	Kikande
Upeneus sulphureus	Sulphur goatfish	Sonyo
Upeneus tragula	Freckled goatfish	Mkundaji
Variola louti	Yellow-edged lyretail	Kikokwe
Zanclus cornutus	Moorish idol	Kibamga la mweme

Appendix II

Species	**Functional Group**	**Functional group Reference**	**Mean Trophic Level (FishBase)**
Abudefduf sexfasciatus	Grazer	FishBase	2.4
Acanthurus dussumieri	Grazer	Cinner et al 2009 J Appl Ecol 2009	2
Acanthurus leucosternon	Grazer	Cinner et al 2009 J Appl Ecol 2009	2
Acanthurus nigricaudus	Detritivore	Cinner et al 2009 J Appl Ecol 2009	3
Acanthururus nigrofuscus	Grazer	Cinner et al 2009 J Appl Ecol 2009	2
Acanthurus tennenti	Detritivore	Cinner et al 2009 J Appl Ecol 2009	2
Acanthurus triostegus	Grazer	Cinner et al 2009 J Appl Ecol 2009	1.8
Aethaloperca rogaa	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4.2
Balistapus undulatus	Pisc-Macro-Invert	FishBase	3.4
Bodianus axillaris	Invert-Micro	FishBase	3.4
Caesio caerulaurea	Invert-Micro	FishBase	3.4
Caesio xanthonota	Planktivore	Cinner et al 2009 J Appl Ecol 2009	3.4
Calotomus carolinus	Grazer-Macro	Cinner et al 2009 J Appl Ecol 2009	2
Cantherhines sandwichiensis	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	2.7
Carangoides ferdau	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4.5
Caranx sexfasciatus	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4.5
Carcharhinus melanopterus	Piscivore	Cinner et al 2009 J Appl Ecol 2009	3.9
Cephalopholis spiloparaea	Piscivore	Cinner et al 2009 J Appl Ecol 2009	4.1
Chaetodon auriga	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.2
Chaetodon kleinii	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.1
Chaetodon meyeri	Invert-Micro	FishBase	3.3
Chaetodon selene	Invert-Micro	FishBase	2.7
Chaetodon trifasciatus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.3
Cheilinus trilobatus	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.5
Cheilio inermis	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	4
Chlorurus strongycephalus	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Coris formosa	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.3
Ctenochaetus striatus	Detritivore	Cinner et al 2009 J Appl Ecol 2009	2
Diagramma pictum	Pisc-Macro-Invert	FishBase	3.5

Epinephelus caeruleopunctatus	Piscivore	Cinner et al 2009 J Appl Ecol 2009	3.7
Epinephelus coioides	Piscivore	Cinner et al 2009 J Appl Ecol 2009	3.9
Epinephelus fuscoguttatus	Piscivore	Cinner et al 2009 J Appl Ecol 2009	4.1
Epinephelus multinotatus	Piscivore	Cinner et al 2009 J Appl Ecol 2009	3.9
Gnathodentex aureolineatus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.3
Gymnocranius grandoculis	Pisc-Macro-Invert	FishBase	3.4
Gymnothorax favagineus	Piscivore	FishBase	4.2
Gymnothorax flavimarginatus	Piscivore	FishBase	4.2
Heniochus acuminatus	Planktivore	FishBase	3.5
Hipposcarus harid	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Leptoscarus vaigiensis	Grazer-Macro	Cinner et al 2009 J Appl Ecol 2009	2.3
Lethrinus lentjan	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4.2
Lethrinus mahsena	Pisc-Macro-Invert	FishBase	3.4
Lethrinus nebulosus	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	3.3
Lethrinus obsoletus	Pisc-Macro-Invert	FishBase	3.4
Lethrinus olivaceus	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.8
Lethrinus variegatus	Invert-Macro	FishBase	3.8
Lutjanus bohar	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2010	4.1
Lutjanus boutton	Pisc-Macro-Invert	FishBase	3.8
Lutjanus fulviflamma	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.8
Lutjanus gibbus	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	3.6
Myripristis berndti	Planktivore	FishBase	3.7
Naso annulatus	Planktivore	Cinner et al 2009 J Appl Ecol 2009	2.1
Naso brachycentron	Grazer	Cinner et al 2009 J Appl Ecol 2009	2.7
Naso brevirostris	Planktivore	Cinner et al 2009 J Appl Ecol 2009	2.2
Naso hexacanthus	Planktivore	Cinner et al 2009 J Appl Ecol 2009	3.3
Naso lituratus	Grazer	Cinner et al 2009 J Appl Ecol 2009	2.3
Naso thynnoides	Planktivore	FishBase	3
Naso tuberosus	Grazer	Cinner et al 2009 J Appl Ecol 2009	2
Naso unicornis	Grazer-Macro	Cinner et al 2009 J Appl Ecol 2009	2.2
Nebrius ferrugineus	Pisc-Macro-Invert	FishBase	4.1
Neoglyphidodon melas	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.5
Novaculichthys taeniourus	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.3
Ostracion cubicus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.4
Ostracion meleagris	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	2.9
Paracanthurus hepatus	Planktivore	Cinner et al 2009 J Appl Ecol 2009	3.4
Parupeneus barberinus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.2
Parupeneus cyclostomus	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4.2
Parupeneus indicus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.5
Parupeneus macronema	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.5
Plectorhinchus flavomaculatus	Pisc-Macro-Invert	Cinner et al 2009 J Appl Ecol 2009	4

Plectorhinchus gaterinus	Pisc-Macro-Invert	FishBase	4
Pomacanthus imperator	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	2.7
Pomacanthus semicirculatus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	2.5
Priacanthus hamrur	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.6
Pterois miles	Piscivore	FishBase	3.7
Scarus festivus	Scrapers/Excavators	Green, A.L. & Bellwood, D.R., 2009	2
Scarus frenatus	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Scarus ghobban	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Scarus globiceps	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Scarus psittacus	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Scarus rubroviolaceus	Scrapers/Excavators	Cinner et al 2009 J Appl Ecol 2009	2
Scarus russelii	Scrapers/Excavators	Green, A.L. & Bellwood, D.R., 2009	2
Scolopsis bimaculata	Pisc-Macro-Invert	FishBase	3.8
Siganus argenteus	Grazer	FishBase	2
Siganus stellatus	Grazer	FishBase	2.7
Siganus sutor	Grazer	FishBase	2
Sufflamen chrysopterum	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	3.5
Taeniura lymma	Invert-Macro	Cinner et al 2009 J Appl Ecol 2009	3.6
Tetrosomus gibbosus	Invert-Micro	FishBase	3.5
Upeneus sulphureus	Invert-Micro	FishBase	3.2
Upeneus tragula	Invert-Micro	FishBase	3.6
Variola louti	Piscivore	Cinner et al 2009 J Appl Ecol 2009	4.3
Zanclus cornutus	Invert-Micro	Cinner et al 2009 J Appl Ecol 2009	2.9