AQUATIC CONSERVATION: MARINE AND FRESHWATER ECOSYSTEMS
Aquatic Conserv: Mar. Freshw. Ecosyst. (2013)

Trap modification opens new gates to achieve sustainable coral reef fisheries

INES GOMES[a,*], KARIM ERZINI[b] and TIMOTHY R. MCCLANAHAN[c]
[a]*Global Vision International (GVI), Kenya*
[b]*Algarve University, Centro de Ciências do Mar (CCMAR), Portugal*
[c]*Wildlife Conservation Society (WCS), Bronx, NY, USA, 10460*

ABSTRACT

1. Innovative strategies are needed to escape the social-ecological poverty that so frequently emerges from persistent overfishing of coral reef resources.
2. This study focuses on fishing gear selectivity and its potential to increase ecosystem health and fisheries productivity without compromising the catch of profitable species.
3. An investigation into the effects of an escape gap (3 cm × 30 cm) modification to the traditional African basket trap on total catch biomass, catch composition and monetary value in two locations with different historical levels of fishing was undertaken.
4. Gated traps caught less low-value fish (juveniles and narrow-bodied coral reef species) while increasing the catch of high- and medium-value fish (wider-bodied commercially valuable species). The total monetary value of the gated trap catches was maintained in a heavily fished environment, while it increased in the less fisheries-depleted area.
5. For the most important local commercial species, the African white-spotted rabbitfish (*Siganus sutor*), the gated traps significantly increased the mean length (by 12%) and weight (by 32%) of capture and decreased the proportion of catch under length at first maturity (*Lmat*) from 56% (traditional traps) to 25% (gated traps).
6. Escape gaps have shown the potential to affect the structure of the fishery and ecosystem by enhancing the number of mature individuals, increasing reef biodiversity and promoting functionally diverse reef fish communities without compromising fisher's revenues.

Received 20 December 2012; Revised 10 May 2013; Accepted 3 July 2013

KEY WORDS: bycatch; cost–benefit; diversity; ecosystem based management; marine park and reserve; recruitment and mortality

INTRODUCTION

The present coral reef crisis is likely to affect the lives of millions of people who depend on this ecosystem for food and income (Hughes *et al.*, 2003; Bellwood *et al.*, 2004; Sadovy, 2005; Cinner *et al.*, 2012). Evaluation of the coral reef literature reveals an extensive documentation of artisanal fisheries impacts while offering few solutions apart

---

*Correspondence to: I. Gomes, Global Vision International (GVI), Kenya. E-mail: istgomes@gmail.com from fisheries closures (McClanahan, 2011; Johnson *et al.*, 2013). Although access and effort restrictions are frequently a default fisheries management tool in low-income countries (Mumby and Steneck, 2008), gear restrictions are generally more widely accepted among poor resource users and, therefore, likely to receive high compliance (McClanahan *et al.*, 2005, 2012; McClanahan and Cinner, 2008). Consequently, interdisciplinary research that includes stakeholder preferences and the cost and benefits of specific restrictions is needed to develop solutions that reduce mortality while having a high potential for adoption by resource users (McClanahan, 2012).

Gear-based management is an approach that exploits differences in selectivity among gear types with an intention to promote ecosystem health, while being adaptable to different socio-economic and ecological settings (McClanahan and Cinner, 2008; Hicks and McClanahan, 2012). A well-managed multi-species and multi-gear fishery is expected to employ gears with little overlap in selectivity, which capture target species at optimal and profitable sizes and promote sustainable harvesting, thereby avoiding resource competition among gears and emergent socio-ecological poverty traps (McClanahan and Mangi, 2004; Cinner, 2009). This sort of balanced harvesting has the potential to maximize yields and minimize ecological effects (Zhou *et al.*, 2010; Garcia *et al.*, 2012). Gear selectivity can influence the population size structure, the composition of the associated food webs and the fishery productivity. As a result, gears can be actively managed to mitigate ecosystem damage, reduce fishing mortality of juveniles in the population, alleviate fishing pressure on certain species and contribute to the recovery of key functional groups, with subsequent ecological impacts (McClanahan *et al.*, 2008; Cinner *et al.*, 2009; Hicks and McClanahan, 2012).

Mesh size can influence catch rates and size composition in fish traps (Mahon and 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 (Robichaud and Hunte, 1997; Sary *et al.*, 1997). 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. Also, increasing trap mesh size has often resulted in short-term loss in revenue for fishers, therefore, becoming a difficult measure to justify, implement, monitor, and enforce (Mahon and Hunte, 2001; Baldwin *et al.*, 2002).

Trap fisheries are among the most common traditional forms of fishing worldwide and, in Kenya, account for almost 40% of reef fish landings by weight (Fisheries Department (FiD), 2008). Fish traps are a highly non-selective form of fishing and large numbers of juveniles and low value fish can be captured (McClanahan and Mangi, 2004; Hawkins *et al.*, 2007). These fish often provide minimal profit to the fishers, which might increase if larger sizes were captured. Traps also catch a large proportion of herbivores and species that may promote the recovery of corals (reef scrapers/excavators and grazers) affected by climate and other disturbances (Cinner *et al.*, 2009). However, traps are efficient and cost-effective (Miller, 1990) and are considered by fishing elders as the most traditional and acceptable gear (McClanahan *et al.*, 1997). Regional studies have shown that, when compared with other gears, traps cause low physical direct damage to corals per unit catch and area (Mangi and Roberts, 2006) and are among the most profitable because of local availability of bait and materials for their construction (Mangi *et al.*, 2007). Consequently, modification of traditional gears, such as traps, represents a strong candidate for management intended to reduce the mortality of juveniles and to alleviate the population and ecosystem impacts of non-selective fishing.

The use of escape panels on demersal commercial trap fisheries has been studied (Stewart and Ferrell, 2003; Grandcourt *et al.*, 2011a, b), but the inclusion of escape gaps in coral reef traditional fish traps has received limited investigation and implementation. Some early efforts in heavily fished Caribbean coral reefs suggest promising results as traps retrofitted with rectangular escape gaps caught significantly fewer bycatch fish and were effective in releasing undersized individuals without significantly decrease the catchability of target species (Munro *et al.*, 2003; Johnson, 2010). It is likely, however, that the outcomes and potential long-term success of these gated traps will be context dependent and sensitive to the status and exploitation levels of the fishery. The present study had two overall objectives building on earlier results; the first one was to assess the feasibility of trap modification (incorporation of escape gaps) in providing fisheries and ecological benefits in a multi-species, multi-gear artisanal fishery in areas with different levels of fishing pressure. The second objective sought to evaluate the acceptance and social implications (fisherman’s income and community acceptance) of the practical implementation of a gear-based management approach at a community level. The study was undertaken in a remote and isolated fishing community within a government gazette marine reserve contiguous to a fisheries closure (Kisite Marine National Park) with fishing restrictions partly regulated by the government park service, and with two distinct fishing grounds close and far from the village settlement.

METHODS

Study area

The work was undertaken on the Mpunguti Marine National Reserve (established in 1978) located at the southern tip of the Kenya coastline (Kwale District) (Figure 1). Inside the reserve only traditional fishing methods (hand line – *mshipi*, and basket traps – *malema*) were allowed, but the use of gill nets and spear guns in the area have been reported by local fishers.

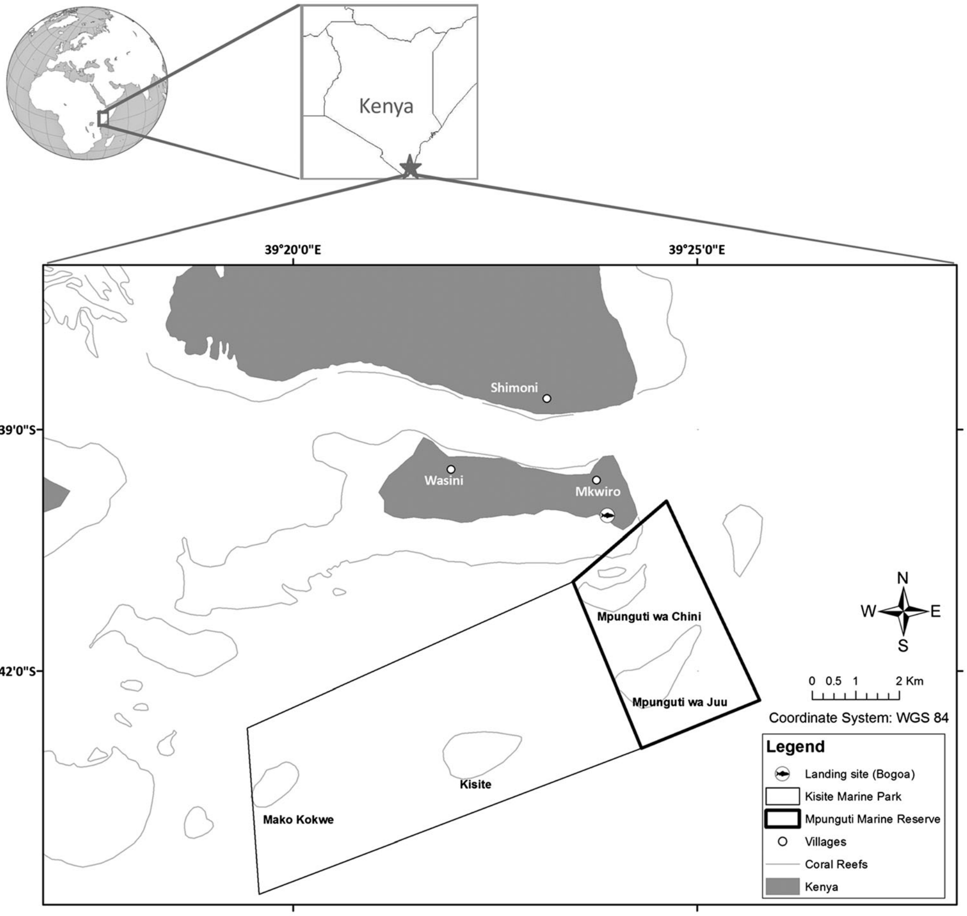

Figure 1. Map of the study area showing the location of the landing site (Bogoa), the fishing grounds in Mpunguti Marine Reserve (Mpunguti wa Chini and Mpunguti wa Juu) and the fully protected Kisite Marine National Park (Mako Kokwe and Kisite).

Traditional fishing (basket traps and hook and line) using dugout canoes inside the marine reserve is the main source of livelihood in Mkwiro village. The two major fishing grounds, Mpunguti wa Chini and Mpunguti wa Juu, have historically been subjected to different exploitation levels, since Mpunguti wa Chini is closer to the villages resulting in easier access, higher fishing effort and decreased fish biomass and diversity (Mkwiro fishers, pers. comm.). Currently, older fishers use Mpunguti wa Chini for subsistence fishing while Mpunguti wa Juu is exploited predominantly by younger fishers and seasonal migrant fishers from the neighbouring island of Pemba, Tanzania. Mkwiro inhabitants have pride in their fishing traditions and strongly disapprove of destructive fishing methods, which include seine, ring nets and spear guns.

Fishers construct their own traps and use a variety of local baits, consisting of a mixture of seaweed and green and red algae. Fish traps, composed of six panels (Figure 2) are locally constructed of wooden frames meshed with reed strips and weighted with stones on the side (Kaunda-Arara and Rose, 2004). The traps have one funnel-shaped door and an underside opening to remove the catch. The catch is checked and caught fish removed on a daily basis, during low tide, with the bait replaced and the trap reset in the same place or nearby. Traps are brought to shore for maintenance only if damaged or showing signs of algae overgrowth, and usually, can stay in the sea for 30–40 days and have a fishable life of 3–6 months (McClanahan, 2010). The study area is representative of the multi-species Kenyan artisanal coral reef fishery and the social characteristics of this fishing village, namely its high dependence on fishing, its adherence to tradition and disapproval of destructive fishing methods made it an ideal place to test the implementation of gated traps at a community level.

Trap design and data collection

Meetings were held between researchers, the local stakeholder community referred to as the Beach Management Unit (BMU), the park authorities (Kenya Wildlife Service – KWS), and trap fishers to introduce the gated trap concept and explain the aims of this project. Some fishers agreed to modify their own traditional traps with 3 cm wide × 30 cm escape gaps to evaluate the short-term results of the modified gear on their catch. These fishers fished mostly in Mpunguti wa Juu, and only one fisherman from Mpunguti wa Chini agreed to modify his trap. Modified traps consisted of two escape gaps (rebar metal gates 3 × 30 cm) inserted at either side of the V-shaped corners of the traditional fishing traps (Figure 2) Several rebar gates were purchased and distributed among the trap fishing community. Fishers used their regular fishing grounds, which varied to a certain extent in terms of reef slope, depth (around 5–15 m) and benthic substrate. No changes in the fishing method were required as gated traps were used in the same manner as traditional traps.

Data collection was carried out at Bogoa fish landing site (Figure 1) during the period from 23 January to 23 April 2012, throughout the north-east monsoon season. Standard sampling methods were

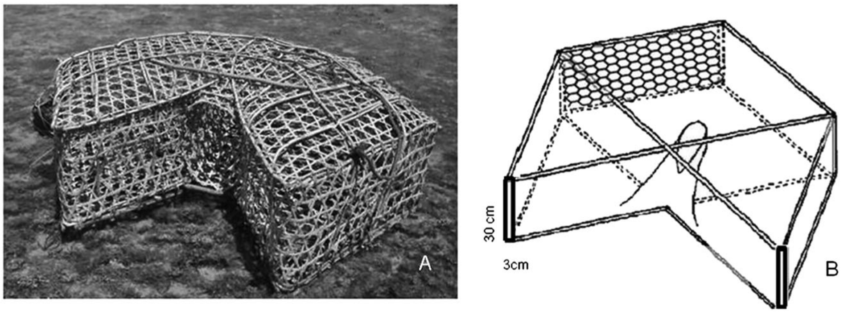

Figure 2. Two types of fish traps used in this experiment: (A) control (traditional African basket trap); and (B) gated basket trap modified with 30 × 3 cm escape gaps (adapted from image in Johnson, 2010).

used and the analysis comprised 2060 sampled fish. Fishers using both trap designs were asked to sort the catch into separate bags, separating traditional from gated trap 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), maximum body width (to the nearest 0.1 mm) and weight (to the nearest 0.1 g) were recorded. Total catch was weighed to the nearest 0.1 kg using spring scales. The entire catch was sampled whether the fish was for market (sold to local fish dealer) or household consumption, and for large catches a randomly collected subsample (around 10% of the catch) was measured. The total catch, species composition, type (traditional or gated) and number of traps, fishing ground and use of catch (market – commercial species or home use – non-commercial species) were recorded for each sampling occasion. 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.

Data handling and analysis

Local diversity of trap catches

First, and in order to have an inclusive understanding of the biological diversity and dynamics of the local trap fisheries, the relative abundance of the fish species caught in the traps (gated and traditional, all sites combined) was investigated. Species were separated into commercial and non-commercial classes depending on their use; for income generation (commercial species sold to local fish dealers) or household consumption (non-commercial catch, juveniles with low or no marketable value and coral reef narrow-bodied species).

Escape gap outcomes in catch biomass, composition and value

The outcomes of trap modification at the two fishing grounds with different exploitation levels – Mpunguti wa Chini and Mpunguti wa Juu, closer and further from the villages, with higher and lower fishing intensity, respectively – were examined. Differences in catch biomass (g per trap) and catch value (KSh per trap) between trap types (traditional and gated traps) and trap distance to the marine park boundary (no-take zone) were tested using one-way ANOVA. Data on biomass were normalized by log10 transformation. Distances to the marine park were calculated using ArcGIS 9.3 and categorized in three levels based on distance to the park boundary (Close 0–500 m, Medium 500–2000 m, Far ≥ 2000 m). The trap value concerns the total commercial biomass inside the trap (saleable fish) and fish prices were collected and confirmed locally by fishers and fish dealers. To compare catch composition, a Pearson chi-square was used to test the proportion of functional groups, fish groups and market value categories caught per gear type, for both locations. All species were categorized into different functional groups based on their diet: piscivore, invertivore, planktivore, grazer/detritivore, browser, scraper/excavator (Supplementary material, Table S1). Fish families were categorized into three market value categories; High, Medium and Low/None economic value (see Table 1 for market categorizations, fish families and socio-ecological relevance). Mean trophic level was considered at species level, based on diet composition (data compiled from FishBase – for the complete set of values consult Supplementary material, Table S1) and the mean trophic level of the catch for each gear type (K) was calculated as:

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

where Y_{ik} is the catch of species i in gear k, TL is the trophic level of species i for m fish species (Pauly *et al.*, 2001).

Escape gap outcomes – single species analysis

Finally, and for the most important and dominant species in the catch, the African white-spotted rabbitfish *Siganus sutor*, the full set of length–frequency distributions were compared for both traditional and gated traps, for all sites combined (Kolmogorov–Smirnov test). The length–frequency analysis tool from FishBase was used by inputting the full set of length–frequency data for this species to obtain estimates for the life-history parameters: length at first maturity

Table 1. Fish market value categories based on mean market value (information assembled from local fishers and fish dealers, prices in Kenyan Shillings 1 US dollars = 85 KSh, exchange rate in 2012), including fish families and the socio-ecological relevance (within the context of this study)

Market category (based on mean market value)	Fish families	Socio-ecological relevance
High 130–150 KSh kg^{-1}	Siganidae Lethrindae Lutjanidae Mullidae	Most valuable catch and much appreciated by coastal Swahili communities; highest profit for fisherman (per trap). Important for main **household income**. Most species are quite **resilient** owing to their broad habitat and diet, although some stocks (*Siganus sutor, Lethrinus lentjan*) are considered **overexploited** in southern Kenya possibly due to growth and Malthusian overfishing.
Medium 100–110 KSh kg^{-1}	Scaridae Haemulidae Serranidae Holocentridae Nemipteridae Muraenidae Caesionidae	Medium level market price. Consists of a diverse group of fish families and its associated ecological links and interactions.
Low/None 0–70 KSh kg^{-1}	Acanthuridae Pomacentridae Labridae Ostraciidae Zanclidae Chaentodontidae Priacanthidae Balistidae Dasyathidae Monacanthidae Or small juveniles from other families	Highly **biodiverse** group of fish families with low or no market value; some bycatch species or small juveniles not sold to fish dealers, but instead kept for household use. **Juveniles** are less than length at first maturity contributing to growth overfishing. Some families comprise important **herbivores** (surgeonfish) which can help decrease algae overgrowth and increase ecosystem resilience and health. Important source of protein for dietary intake and **household food security**. Most of the fish from these families are colorful, ornamental species important for **eco-tourism attraction**.

($Lmat = 26.4$ cm) and length of maximum possible yield ($Lopt = 29.3$ cm). The number of fish below the theoretical maturity length ($Lmat$) was then calculated to find the proportion of immature individuals in the catch. Furthermore, and in order to investigate retention properties of the gated traps, a logistic selectivity model was fitted to *S. sutor* width morphometric data:

$$S_i = 1/\left(1 + \mathrm{e}^{(-b(L - L50))}\right) \qquad (2)$$

where S_i is the proportion retained for width class i (Li), 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. Solver (Microsoft Office Excel 2007) was used to estimate the parameters of the logistic model by minimizing the sum of squares of the difference between the observed and expected size selectivities:

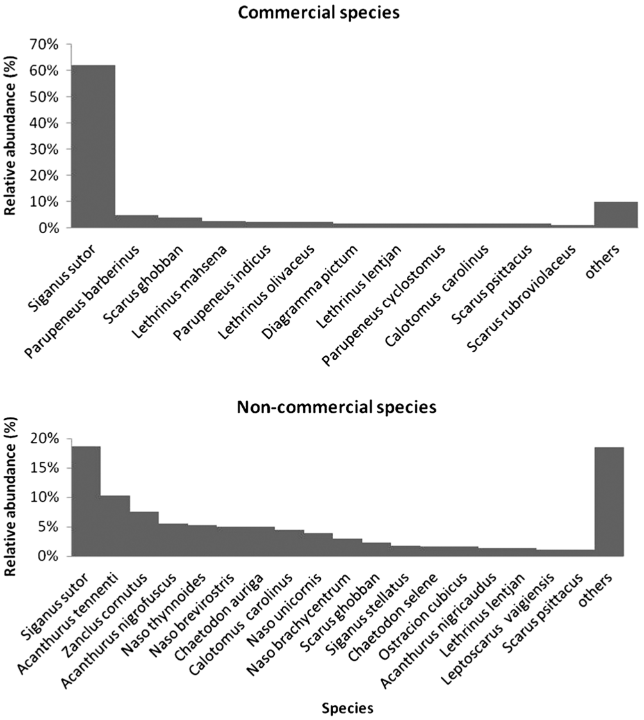

Figure 3. Relative abundance in the catch of the most common fish species, based on the analysis of 2060 sampled fish and their final destination: market (commercial catch) or household (non-commercial catch, juveniles and coral reef narrow-bodied species, with low or no marketable value).

$$\text{Min} \ \sum S_{iobs} - 1/\left(1 + \mathrm{e}^{(-b(Li-L50))}\right)^2 \qquad (3)$$

RESULTS

Local diversity of trap catches

Over the course of the study period, 65 visits to the landing site and 213 sampling occasions were made to examine the catch of 21 trap fishers from Mkwiro. The sampling included in total 2060 reef-associated fish representing 93 species of 26 families (for the complete species list consult Supplementary material, Table S2). This high diversity of the catch is characteristic of the multi-species nature of the Kenyan artisanal fishery. Despite the high number of species caught, catches were dominated by only a few species (Figure 3). 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). The most important local species by far is the African white-spotted rabbitfish (*S. sutor*), which alone comprises 62% of the total commercial species and almost 20% of the catch taken for home use. All fish caught in the traps are kept and either sold to local fish dealers (commercial fish species) or taken directly home for consumption (non-commercial species or small juvenile fish). The only fish discarded was the puffer fish (family Tetraodontidae), which is considered poisonous. For the non-commercial fish, small-sized individuals of rabbitfish (*Siganus*

Table 2. Data for biomass of fish captured per trap set (g per trap) and profit per trap (Kenyan Shillings per trap). Sample size (n), means, SE, and results of ANOVAs testing for significant differences between trap types (traditional and gated) and distance to the marine park are presented. Statistically significant differences are indicated $*P \leq 0.05$, $**P \leq 0.01$, $***P \leq 0.0001$

Fishing ground	Variables	n	Mean ± SE	F	df	p	Sig.
Mpunguti wa Chini	**Total catch (biomass g per trap)**						
	Traditional	47	1069 ± 112				
	Gated	9	541.5 ± 123	7.2	1	0.01	**
	Distance to Marine Park	56	-	0.19	1	0.66	NS
	Commercial catch (g per trap)						
	Traditional	47	826.7 ± 100				
	Gated	9	501.6 ± 133	1	1	0.33	NS
	Distance to Marine Park	56	-	0.15	1	0.7	NS
	Non-commercial catch (g per trap)						
	Traditional	47	242.3 ± 26.3				
	Gated	9	39.8 ± 24.3	32	1	0.000	***
	Distance to Marine Park	56	-	1	1	0.3	NS
	Profit per trap (Ksh per trap)						
	Traditional	47	99.2 ± 12				
	Gated	9	60.2 ± 16	1.9	1	0.18	NS
	Distance to Marine Park	56	-	0.5	1	0.48	NS
Mpunguti wa Juu	**Total catch (biomass g per trap)**						
	Traditional	114	1359.1 ± 78.8				
	Gated	68	1556.5 ± 133.4	1.4	1	0.24	NS
	Distance to Marine Park	182	-	0.42	2	0.66	NS
	Commercial catch (g per trap)						
	Traditional	114	1032.1 ± 65.3				
	Gated	68	1376.5 ± 123.9	7.2	1	0.008	**
	Distance to Marine Park	182	-	0.7	2	0.47	NS
	Non-commercial catch (g per trap)						
	Traditional	114	327.6 ± 26.8				
	Gated	68	209.9 ± 44.5	91.1	1	0.000	***
	Distance to Marine Park	182	-	0.3	2	0.2	NS
	Profit per trap (Ksh per trap)						
	Traditional	114	123 ± 7.9				
	Gated	68	165.2 ± 14.9	7.2	1	0.008	**
	Distance to Marine Park	182	-	0.45	2	0.66	NS

sutor), double-banded and brown surgeonfish (*Acanthurus tennenti* and *Acanthurus nigrofuscus*), Moorish idol (*Zanclus cornutus*), one-knife and spotted unicornfish (*Naso thynnoides* and *Naso brevirostris*) and threadfin butterflyfish (*Chaetodon auriga*) represented the most common species.

Escape gap outcomes in catch biomass, composition and value

The escape gaps had different effects on catch biomass, composition and value at the two fishing grounds, while the trap's distance to the marine park did not influence the catch rates (Table 2). In Mpunguti wa Chini, gated traps resulted in a significant decrease in total catch biomass largely due to the reduced catch of non-commercial species (with low or no marketable value, juveniles and narrow-bodied species) (Table 2, Figure 4; F = 7.2, $P \leq 0.01$). Therefore, the mean value of the catch (KSh per trap) did not significantly decrease (although gated traps did show a large variation in the mean value). In Mpunguti wa Juu, the gear modification maintained the total catch biomass and, while there was a clear significant decrease in the catch of non-commercial species (Table 2, Figure 4; F = 0.41, $P \leq 0.001$), the catch of commercial species significantly increased (Table 2, Figure 4; F = 7.2, $P \leq 0.01$), which compensated (by weight) for the loss of smaller, low-value fish. This difference in catch composition explains the increase in the mean value of the catch for the gated traps in this location (from 123 ± 7.9 KSh per trap to 165.5 ± 14.9 KSh per trap, Table 2, Figure 5; F = 7.2, $P \leq 0.01$).

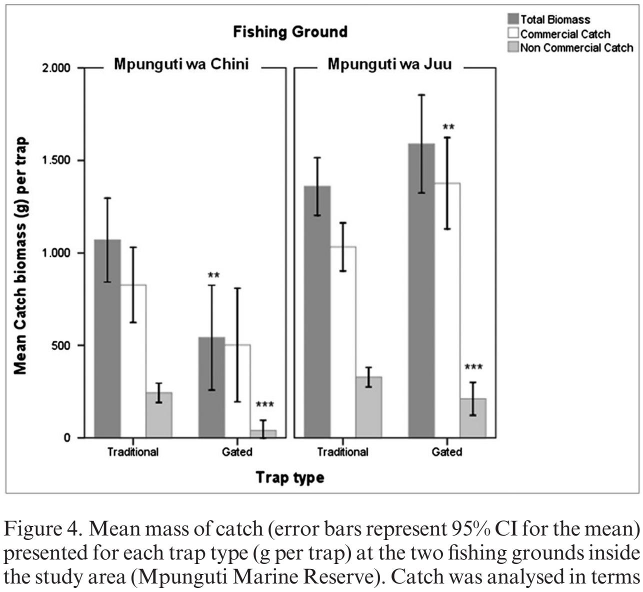

Figure 4. Mean mass of catch (error bars represent 95% CI for the mean) presented for each trap type (g per trap) at the two fishing grounds inside the study area (Mpunguti Marine Reserve). Catch was analysed in terms of total biomass, commercial (saleable fish) and non-commercial (not sold) biomass per trap. Significant differences from controls (traditional traps) are indicated $^{*}P \leq 0.05$, $^{**}P \leq 0.01$, $^{***}P \leq 0.0001$.

Escape gaps shifted the catch composition differently in the two fishing grounds (Table 3, Figure 6). However, as traps were the units of replication, sample sizes are greatly inflated and species aggregation and increased mean abundance may have increased Type I error rates on Pearson chi-square tests (Garson and Moser, 1995). In Mpunguti wa Chini, the proportion of low/no market value fish and the high-valued rabbitfish dropped, while that of parrotfish and goatfish

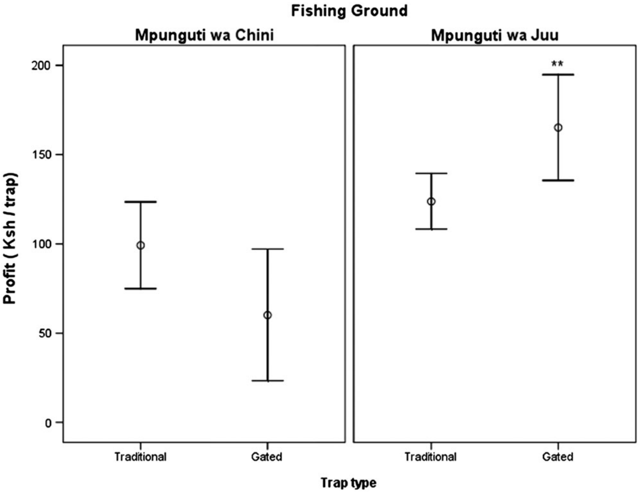

Figure 5. Error bars represent 95% CI for mean value of catch per trap (prices in Kenyan Shillings; 1 US dollar = 85 KSh, rate in 2012). Significant difference from control (traditional traps) is indicated $^{**}P \leq 0.01$.

Table 3. Results of Pearson chi-square test to compare catch composition, as a proportion of functional groups, fish groups and market value categories, caught per gear type for the two locations studied

Fishing ground	Explanatory variable: Trap type	Chi-square	df	P
Mpunguti wa Chini	Functional group	20.4	5	0.001
	Fish group	15.4	5	0.009
	Market value	39.8	2	0.000
Mpunguti wa Juu	Functional group	52.9	5	0.000
	Fish group	68.6	5	0.000
	Market value	75.8	2	0.000

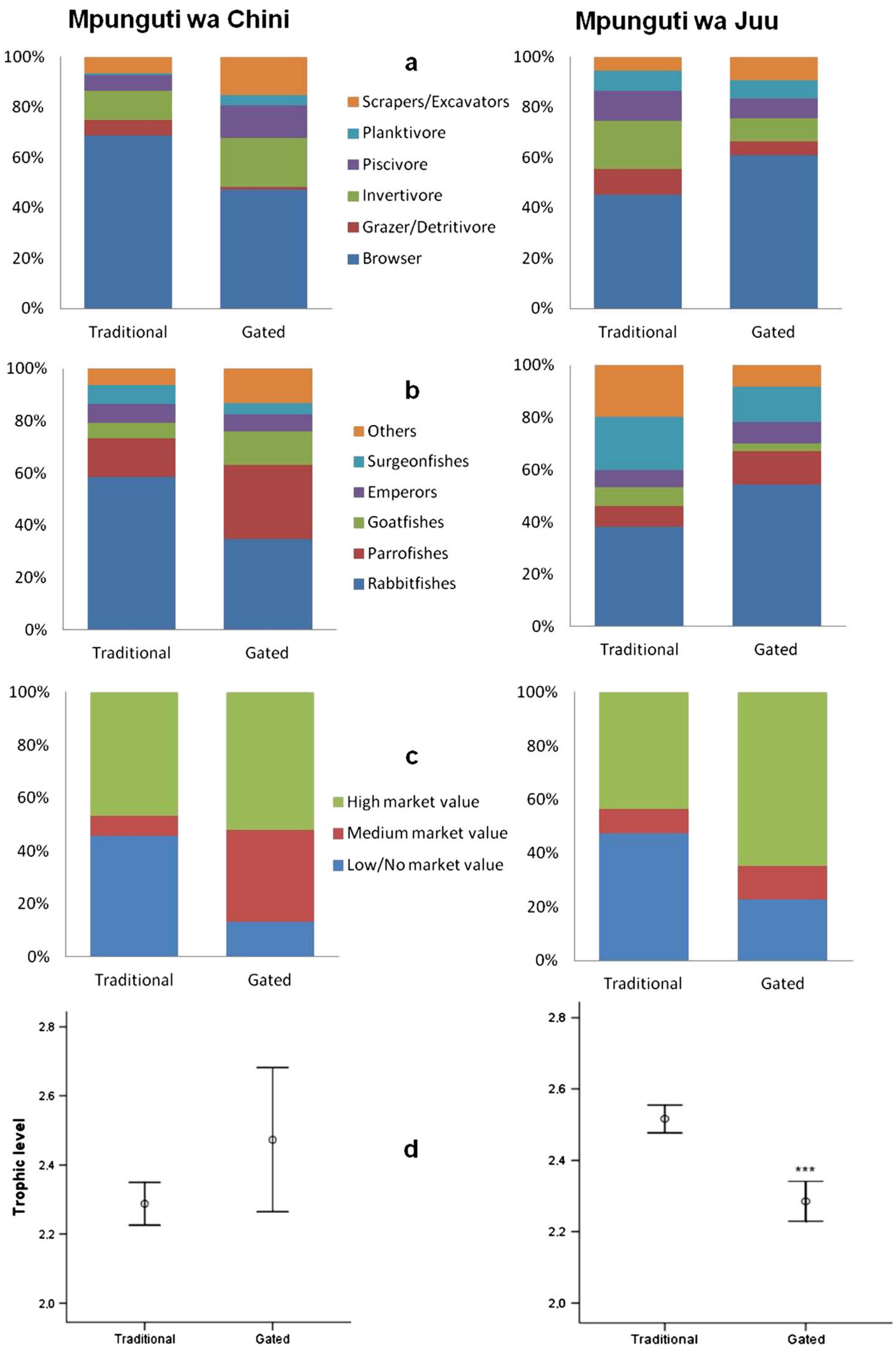

Figure 6. Catch composition by trap type and location. Bars represent the percentage of the catch that was made up by each category: (a) functional group; (b) fish groups; (c) market value (see Table 3 for chi-square test results). In (d) error bars (95% CI for mean) represent mean trophic level of catch (significant difference from control *** F = 38.0, $P < 0.0001$).

increased. This location showed an overall decrease in the fraction of grazer/detritivore and browsers caught in gated traps. In Mpunguti wa Juu, the gated trap catch also displayed a lower proportion of low/no market value fish but caught a higher proportion of rabbitfish and parrotfish. Consequently, there was a general increase in browsers and scrappers/excavators with a decrease in grazer/detritivore and invertivores. The catch of the two gear types in both locations had low mean trophic levels ($TL = 2.3$–2.6) since they mainly target herbivorous species. In Mpunguti wa Chini, variation in trophic level was high and comparisons of trap type were insignificant. In Mpunguti wa Juu, the mean trophic level of the catch differed significantly between traditional and gated traps (Figure 6; F = 38, $P \leq 0.0001$). Gated trap catch demonstrated a lower mean trophic level (2.33 ± 0.029) than the traditional ones (2.50 ± 0.022). This effect is due to fewer narrow-bodied coral reef species, such as butterflyfish (Chaetodontidae) and Moorish idols (Zanclidae) that feed on invertebrates ($TL = 2.7$–3.3, Supplementary material, Table S1) and the increased catch of rabbitfish and parrotfish ($TL = 2$, Supplementary material, Table S1) in gated traps.

Single species analysis – length–frequency distribution and size-selectivity curve

Results for all sites combined indicated that, for the African white-spotted rabbitfish *S. sutor*, gated traps significantly increased ($P < 0.0001$) the mean length (by 12.0%) and weight (by 32.2%) of captured fish and decreased the proportion of the catch under length at first maturity from 56.0% (traditional traps) to 25.2% (Table 4, Figure 7). A direct relationship was established between the width of *S. sutor* and the probability of being retained by the trap. The fitted logistic model adequately described the increasing retention with width (Figure 8), with 50% of the fish of 3.82 cm being retained ($L50$ =3.82 cm) and only 22% of fish of 3.0 cm width being retained.

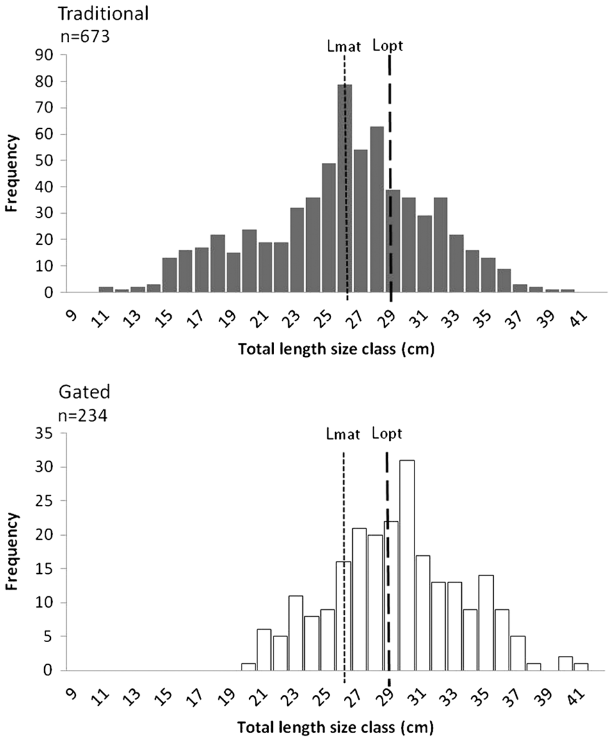

Figure 7. Total length distribution (recalculated using standard length–total length relationship $SL = 0.824TL - 0.220$ from Wambiji *et al.*, 2008) of *Siganus sutor* in both trap types for all sites combined (n = number of samples). The Kolmogorov–Smirnov test for comparison of two data sets showed a maximum difference between the cumulative distributions $D = 0.31$ with a corresponding $P \leq 0.0001$. $Lmat = 26.4$ cm, $Lopt = 29.3$ cm, calculated using the length–frequency analysis tool from FishBase.

DISCUSSION

The study finds differences in the catch biomass and composition in the two different locations that are consistent with those expected to arise from different historical levels of fishing. These differences affected the categories in ways that have implications for the home use and income of the resource users that can possibly influence their willingness to adopt escape gaps. While the escape gap has some clear net benefits in one lightly fished environment, the benefits are less clear in the more heavily fished environment. All sites showed changes in catch composition, with the

Table 4. Mean total length (recalculated using standard length–total length relationship $SL = 0.824TL - 0.220$, from Wambiji *et al.*, 2008) and weight (g) per individual fish, and proportion of individuals caught less than length at first maturity (≤ *Lmat*) for the *Siganus sutor* in both trap types for all sites combined (n = number of samples). Means, standard error (SE) and percentage different from control (traditional trap) are presented. Means significantly different from control are indicated ***$P < 0.0001$

Siganus sutor	**Traditional trap** n = 673	**Gated trap** n = 234	**Difference**
Mean *TL* length (cm) ± SE	25.5 ± 0.20	29.0 ± 0.20 ***	Increase 12.01%
Mean weight (g) ± SE	263.9 ± 5.52	349.1 ± 10.68 ***	Increase 32.20%
Individuals caught ≤ *Lmat* (%)	56.01%	25.21%	Decrease 30.80%

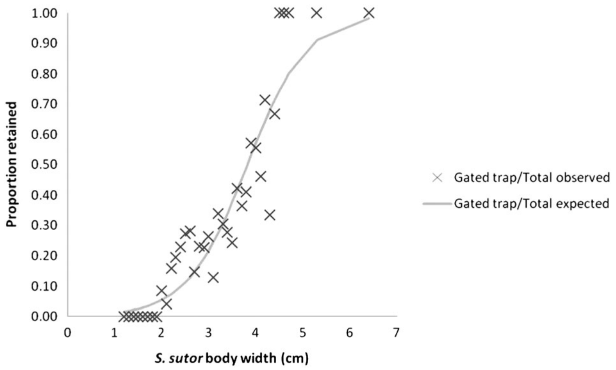

Figure 8. Gear selection ogive representing the size-selectivity curve for the escape gap size 3 cm × 30 cm, obtained by fitting the logistic selectivity model to *Siganus sutor* proportion retained by width class data. Data on the x-axis refer to *S. sutor* fish width (cm) and y-axis shows the percentage retained in the catch by body width. The fitted curve was calculated from Equation (2) using the parameters $b = 1.5690$ and $L50 = 3.8145$.

gated traps catching more target species and having reduced heterogeneity in the catch. This outcome matches the results previously described by Johnson (2010) where Caribbean fish traps retrofitted with escape gaps did not reduce the catch of high-value fish nor the total market value of the catch. More than 70 different species in the catch had no market for the species or size. These were mostly small-sized individuals of *S. sutor* and narrow bodied species in the Acanthuridae (surgeonfish and unicornfish), Zanclidae (Moorish idols) and Chaetodontidae (butterflyfish) families. Also there were no discarded fish caught in basket traps since fishers sell the marketable species and the remainders were used for household consumption. The various strengths and weaknesses of escape gaps found in this study are described briefly in Table 5.

Differences between sites

The incorporation of 3 × 30 cm escape gaps in traps fished at Mpunguti wa Chini resulted in a significant decrease in total catch biomass largely due to the reduced catch of fish with low or no marketable value, juveniles and coral reef narrow-bodied species. In this location, the gated traps showed large variation that resulted in non-significance when comparing the mean profit per trap. Gated traps caught fewer low-value fish, while increasing the catch of medium-value fish families such as parrotfish (Scaridae). Rabbitfish (Siganidae) declined in the gated trap catch owing to the reduced catch of small individuals in this location. In addition, in Mpunguti wa Chini the lower trophic level of the traditional trap catches might be a reflection of the history of heavy fishing in the area (McClanahan *et al.*, 2008).

Table 5. Summary of the ecological, social and management implications of the incorporation of escape gaps in multi-species basket trap coral reef fisheries

Approach	Potential benefits	Future research/Monitoring
Ecological	Increase the number of reproductively mature fish and avoid recruitment limits. Increase fish biomass and biodiversity, promote ecological functions and diverse reef fish communities. Promote fish biomass ecosystem health; intensifying the system's resilience and protect against coral reef phase shifts, disease and climate change.	Undertake controlled experiments testing different escape gap and mesh dimensions, including different variables such as trap shape, size (volume), mesh size, soak time, bait type and spacing between traps. Make direct *in situ* observations or underwater video surveillance of fish ingress and escapement behaviour.
Social	Sustain fishers' income in the short term and create the possibility to enhance it in the long term. Promote eco-tourism attractions due to increased ornamental fish abundance. High chance of cultural acceptance and compliance with regulations, as it does not limit fishers' access to fishing grounds.	Evaluate community and seasonal migrant's engagement and compliance. Evaluate nutritional content of different coral reef species and local household fish consumption patterns. Evaluate the long-term effects on tourism attraction and potential increase in ornamental biodiversity.
Administrative	Provide the flexibility to promote and increase gap sizes as catch rates improve and catch composition changes.	Undertake continuous site-specific monitoring of gate size adjustments on catch and income. Evaluate other local gear restrictions that do not compromise minimum capture size and bycatch and potential for competition with these gated traps.
	Offer an adaptable, low tech and low cost alternative to self-enforced community resource management.	Evaluate the ability of local fishing communities to acquire knowledge, materials and eventual capital compensation for the gear transition period.

In the more distant and lightly fished Mpunguti wa Juu, the gear modification did not result in a decline in the total catch biomass because the increase in the catch of commercial species compensated for the loss of smaller and low-value fish – thus maintaining the CPUE. This difference in catch composition explains the increase in the mean value of the catch for the gated traps in this location. Consequently, high catch rates alone will not produce the best economic revenue when traps are filled with low-value species (Luckhurst and Ward, 1987; Johnson, 2010).

In Mpunguti wa Juu, the mean trophic level of the gated traps catch was lower than of the traditional traps (2.33 ± 0.029 and 2.50 ± 0.022 respectively). It has been proposed that the mean trophic level of the catch can be used as an index of the history of fishing and sustainability in multi-species fisheries but this may be most relevant to more aggregate forms of fishing gear use and data (Pauly *et al.*, 2001). In addition, the use of multiple gears with reduced overlap in selectivity can contribute to maintaining a constant mean trophic level of the fisheries catch and promote ecosystem productivity (McClanahan *et al.*, 2008; Garcia *et al.*, 2012). While this study showed a lower trophic level for the gated trap catch, an extended time series analysis would be needed to evaluate the effect of gear transition on the trophic relationships within the ecosystem. Nevertheless, having a lower trophic level catch may increase the production potential of these gated traps as per unit area yields are expected to be higher at lower trophic levels (McClanahan, 1995).

For the dominant and most commercial species caught in the traps, *S. sutor*, mean standard lengths and weights for fish retained in the gated traps were larger than for those in traditional traps. This difference resulted in an increase in economic value because of 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 yr^{-1} (Kaunda-Arara and Rose, 2006), a 1 yr delay in harvesting can approximately double the fish size and price of these fast-growing and short-lived species (McClanahan, 2010). Escape gaps significantly decreased the proportion of the catch under the length at first maturity, thereby decreasing the chances of recruitment overfishing (high mortality of individuals before they reproduce) and potentially enhancing the stock and the stability of harvests. Baldwin *et al.* (2002) found that the decline in juvenile catch in the gated traps resulted from reduced catches overall, which was not the case in this study. This is particularly important as recent investigations of the life-history traits of *S. sutor* on Kenya's south coast revealed that the stock was over exploited (Hicks and McClanahan, 2012).

The selectivity analysis for *S. sutor* also confirmed a fish width–escape gap size relationship, with the majority ($> 78\%$) of fish with less than 3 cm body width escaping. Ward (1988) investigated the different 'trapability' of reef fish species and proposed a 'squeezing hypothesis' in which some species are able to squeeze through mesh. Robichaud *et al.* (1999) showed that some soft-bellied reef fish species (whose sizes were bigger than the actual mesh size) were able to force their way through by distorting their body shapes. Results of the present study support these findings and suggest that *S. sutor* is able to fit through escape gaps somewhat smaller than its measured body width.

Size-selectivity curves are useful for evaluating suitable escape gap sizes and to determine the optimal range to provide the best compromise between low catches of undersize fish and the retention of mature and marketable individuals. Calculation of a size-selectivity curve provides a summary of trap performance and can be further incorporated into size-based spatial models to evaluate the effect of changing escape gap regulations on the future state of the fishery (Treble *et al.*, 1998). However, the approach used here to estimate size selectivity is dependent on the assumption that the traditional traps are not selective.

Future size-selectivity experiments should use traps with variable escape gap widths and mesh sizes. This will allow comparisons of catch size frequency distributions of the different types of traps to estimate size selectivity, using for example the SELECT (Share Each Length-class's Catch Total) model, which can take into account the fishing effort and predict the proportion of the total catch in the various size classes (Treble *et al.*, 1998). A combination of *in situ* observations and the above controlled experiments are recommended for finding the optimal escape gap and mesh size for the local ecosystem and life-history traits. These studies are expected to improve the economic and ecological sustainability of the fishery.

Catch composition variation

The most consistent finding of this study is that incorporation of escape gaps into traditional fishing traps resulted in the escape of low-value and narrow-bodied species and juveniles of marketable species. This biomass loss resulted in a switch in catch composition that might be explained by density-dependent behaviour, susceptibility of capture, and interactions between and within species. Munro (1974) and Miller (1979) found that the catch/trap was asymptotic with time or the trap reached a saturation point similar to density-dependent processes. Presumably, crowded traps inhibited the entry of individuals and this is expected to influence both size and species selectivity responses. From studies of different selection processes in Antillean traps, Gobert (1998) hypothesized that the presence of large fish induced a 'fleeing behaviour' in smaller fish that may force their way out of traps. Therefore, the probability of escapement of small fish in crowded traps is higher, and densely populated traps would attract larger fish while repelling small individuals.

Given that traps constitute a passive mode of capture, the specific behaviour of the species can play a vital role in the capture process (Fogarty and Addison, 1997). Specific behavioural interactions between fish inside and outside the static gear may result in conspecific attraction or heterospecific avoidance, influencing catch magnitude and composition (Munro, 1971, 1974; Miller, 1979; Luckhurst and Ward, 1987). Renchen *et al.* (2012) conducted an underwater video surveillance of coral reef fish behaviour to evaluate interactions with traps and found that 67% of the fish approaching a trap consisted of conspecific individuals, suggesting that traps act as attractants for some species. Here, it is hypothesized that the interactions of 'saturation point' of the traps, 'fleeing behaviour', and 'conspecific attraction' produce the change in catch composition between traps – gated traps having larger and more homogeneous catches as predicted by the intraspecific attraction theory. Behavioural observations of fish movements around traps should clarify the selection properties of traps. Furthermore, we recommend evaluating these behaviours along with trap variables, such as trap shape, size, mesh size, soak time, bait type and quantity and the spacing between traps (Munro, 1974; Miller, 1979; Williams and Hill, 1982).

Gear selectivity will ultimately have impacts on the population structure and ecology of the fishery (Lokrantz *et al.*, 2009). Species-rich fish communities can contribute to prevent algal-dominated reefs, increase resilience to disturbances and reduce the risk of coral disease prevalence (Hughes *et al.*, 2003; Worm *et al.*, 2006; Hughes *et al.*, 2007; Cinner *et al.*, 2009; Raymundo *et al.*, 2009). Escape gaps will increase the size of individuals and have subsequent effects on their feeding and reproduction behaviours. Small-bodied species that are not caught will increase local biodiversity and potentially increase the prey available for larger fish. Overall, this may increase the productivity and sustainability of the fishery but future long-term investigations of this impact are needed.

Potential detriments

Findings show that it is essential to include the non-commercial catch (a result from the fisher's monetary and food costs and benefits) in the analyses of artisanal fisheries. Estimates of catch based on local fish dealers (where the BMUs are conducting data collection) will not include the total daily catch and result in an underestimation of the local CPUE. Official reports will, therefore, hide the capture and the importance of low market value but ecologically and socially important species that account for an important part of the household protein consumption. Here, it was found that 25% of the total biomass from the basket traps did not reach the market, which is similar to a 30% estimate by Glaesel (1997).

The reduction in the catch of low-value fish species of potential ecological importance was the major effect of the escape gap introduction. Consequently, the incorporation of escape gaps could affect an important source of animal protein and micronutrients in this rural fish-dependent community (Kawarazuka and Béné, 2010). However, the limited data on nutrient content in different coral reef species and local household fish consumption patterns make it difficult to evaluate the possible nutritional outcomes of the gear intervention. In Mpunguti wa Juu, where the gated traps increased fisher's income, the higher purchasing power could potentially compensate for the loss of smaller fish but this holds complex cultural and social implications, which are beyond the aims of this study. In more fishery-depleted areas, like Mpunguti wa Chini, management could consider the temporary use of smaller gates (for example 2 cm wide escape gaps) until catch composition and incomes change. Organizations involved in gear-change projects should benefit from involving fishers in the evaluations and monitoring of catch, such that they can be informed about resource user's needs and adaptations during the various phases.

A detrimental aspect of trap use that cannot be fully mitigated by the inclusion of escape gaps is 'ghostfishing' or capturing fish after the traps are lost (Erzini *et al.*, 2008). Traditional African fish traps are made of biodegradable materials and disintegrate when left under water, limiting the amount of continuous catches. The inclusion of biodegradable fasteners and panels (Selliah *et al.*, 2001; Bilkovic *et al.*, 2012) might also be included among the increasing use of metal and wire traps being introduced into the region (McClanahan, pers. obs.). In addition, trap fisheries normally target herbivores, which are essential to maintaining coral and potentially reversing coral–macroalgal phase shifts, an algal-dominated state of the reef ecosystem (Bellwood *et al.*, 2004). Herbivores are also commonly caught by spear guns and beach seines, so all of these gears would need to be managed simultaneously to be able to influence herbivore abundance (Cinner *et al.*, 2009). Furthermore, escape gaps are not able to reduce the catch or injury of all low or no-economic value species, including wide-bodied species, such as moray eels (Muraenidae), trunkfish (Ostraciidae), and scorpionfish (Scorpaenidae) (Johnson, 2010). Escape gaps can only be effective when combined with other local gear restrictions that also restrict the capture of small-bodied individuals and species (McClanahan and Mangi, 2004).

Implementation, costs and other practicalities

Gear-based solutions are popular among fishers in the region and might prove efficient in achieving societal engagement, consensus, and compliance (McClanahan *et al.*, 2008, 2012). Supportive organizations will have to consider the ease of adoption, the costs and benefits, and participate in increasing the knowledge, materials and possibly capital compensation for the gear transition period. Social consent and adoption is likely to be higher than with area-based management and closures, since gear modification is less intrusive to communities and does not promote their exclusion or changes in the fishing methods that might create a sense of social disparity or losses of income (Hicks and McClanahan, 2012; McClanahan *et al.*, 2012). Escape gaps are easily incorporated into traditional traps (installation time about 30 min per trap), and built from plant materials or locally available and affordable metal. Escape gap sizes can be changed over time as catch rates change and recover and be adaptable to local conditions (Munro *et al.*, 2003). The study here shows that the use of variable size escape gaps adapted to local conditions may be able to overcome the resistance of fishers and give them goals of increasing gap width, size of capture and income.

ACKNOWLEDGEMENTS

Erasmus Mundus Scholarship, Wildlife Conservation Society (WCS) and Global Vision International (GVI) financially supported the research. Partial support for this work was provided by a prize from RARE's Global Ocean Solutions search. Permission to work in the national reserve was provided by Kenya Wildlife Service (KWS). We are grateful for the support of Richard Lemarkat (KWS), Mwanaisha (research assistant), Emmanuel Mbaru (WCS), and GVI staff Tessa Doogue, Zeno Wijtten, Sergi Perez and all the volunteers. Finally, we thank Mkwiro village chairman Mtengo Makame and the members of the Mkwiro BMU and trap fishers for their participation in the project.

SUPPORTING INFORMATION

Supporting information can be found in the online version of this article.

REFERENCES

Allen G, Steene R, Humann P, Deloach N. 2003. *Reef Fish Identification; Tropical Pacific*. New World Publications: El Cajon, CA.

Baldwin K, Oxenford HA, Parker C, Franklin G. 2002. Comparing juvenile catch rates among conventional fish traps and traps designed to reduce fishing mortality on juvenile reef fishes. *Proceedings of the Gulf and Caribbean Fisheries Institute* **55**: 306–321.

Bellwood DR, Hughes TP, Folke C, Nystrom M. 2004. Confronting the coral reef crisis. *Nature* **429**: 827–833.

Bilkovic DM, Havens KJ, Stanhope DM, Angstadt KT. 2012. Use of fully biodegradable panels to reduce derelict pot threats to marine fauna. *Conservation Biology* **26**: 957–966.

Cinner J. 2009. Poverty and the use of destructive fishing gear near east African marine protected areas. *Environmental Conservation* **36**: 321–326.

Cinner JE, McClanahan TR, Graham NAJ, Pratchett MS, Wilson S. 2009. Gear-based fisheries management as a potential adaptive response to climate change and coral mortality. *Journal of Applied Ecology* **46**: 724–732.

Cinner JE, McClanahan TR, Graham NAJ, Daw TM, Maina J, Stead SM, Wamukota A, Brown K, Bodin Ö. 2012. Vulnerability of coastal communities to key impacts of climate change on coral reef fisheries. *Global Environmental Change* **22**: 12–20.

Erzini K, Bentes L, Coelho R, Monteiro P, Ribeiro J, Gonçalves JM. 2008. Catches in ghost-fishing octopus and fish traps in the northeastern Atlantic Ocean (Algarve, Portugal). *Fishery Bulletin* **106**: 321–327.

Fishbase World Wide Web electronic publication. 2011. http://www.fishbase.org/ [August 2011]

Fisheries Department (FiD). 2008. Marine Fisheries Frame Survey. Ministry of Fisheries Development Kenya. Provincial Headquarters, Mombasa.

Fogarty MJ, Addison JT. 1997. Modelling capture processes in individual traps: entry, escapement and soak time. *ICES Journal of Marine Science* **54**: 193–205.

Garcia SM, Kolding J, Rice J, Rochet MJ, Zhou S, Arimoto T, Beyer JE, Borges L, Bundy A, Dunn D, *et al*. 2012. Reconsidering the consequences of selective fisheries. *Science* **335**: 1045–1047.

Garson GI, Moser EB. 1995. Aggregation and the Pearson chi-square statistic for homogeneous proportions and distributions in ecology. *Ecology* **76**: 2258–2269.

Glaesel H. 1997. Fishers, parks, and power: the socio-environmental dimensions of marine resource decline and protection on the Kenya coast: PhD thesis. University of Wisconsin, USA.

Gobert B. 1998. Density dependent size selectivity in Antillean fish traps. *Fisheries Research* **38**: 159–167.

Grandcourt EM, Al Abdessalaam TZ, Hartmann SA, Al Shamsi AT, Francis F. 2011a. An evaluation of the selectivity characteristics of different juvenile fish escape panel designs for the demersal trap fishery of Abu Dhabi, United Arab Emirates. *Open Journal of Marine Science* **1**: 98–107.

Grandcourt EM, Al Abdessalaam TZ, Hartmann SA, Francis F, Al Shamsi AT. 2011b. An evaluation of the management effectiveness of a stock re-building strategy for a multi-species demersal trap fishery in Abu Dhabi, United Arab Emirates. *Open Journal of Marine Science* **1**: 82–97.

Hawkins J, Roberts C, Gell F, Dytham C. 2007. Effects of trap fishing on reef fish communities. *Aquatic Conservation: Marine and Freshwater Conservation* **17**: 111–132.

Hicks CC, McClanahan TR. 2012. Assessing gear modifications needed to optimize yields in a heavily exploited, multi-species, seagrass and coral reef fishery. *PLoS ONE* **7**: e36022.

Hughes TP, Baird A, Bellwood DR, Card M, Connolly SR, Folke C, Grosberg R, Hoegh-Guldberg O, Jackson JBC, Kleypas J, *et al*. 2003. Climate change, human impacts, and the resilience of coral reefs. *Science* **301**: 929–933.

Hughes TP, Rodrigues MJ, Bellwood DR, Ceccarelli D, Hoegh-Guldberg O, McCook L, Moltschaniwskyj N, Pratchett MS, Steneck RS, Willis B. 2007. Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Current Biology* **17**: 360–365.

Johnson AE. 2010. Reducing bycatch in coral reef trap fisheries: escape gaps as a step towards sustainability. *Marine Ecology Progress Series* **415**: 201–209.

Johnson AE, Cinner JE, Hardt MJ, Jacquet J, McClanahan TR, Sanchirico JN. 2013. Trends, current understanding, and future research priorities for artisanal coral reef fisheries research. *Fish and Fisheries* **14**: 281–292.

Kaunda-Arara B, Rose GA. 2004. Effects of marine reef national parks on fishery CPUE in coastal Kenya. *Biological Conservation* **118**: 1–13.

Kaunda-Arara B, Rose GA. 2006. Growth and survival rates of exploited coral reef fishes in Kenyan marine parks derived from tagging and length-frequency data. *Western Indian Ocean Journal of Marine Science* **5**: 17–26.

Kawarazuka N, Béné C. 2010. Linking small-scale fisheries and aquaculture to household nutritional security: a review of the literature. *Food Security* **24**: 343–357.

Lokrantz J, Nyström M, Norström AV, Folke C, Cinner JE. 2009. Impacts of artisanal fishing on key functional groups and the potential vulnerability of coral reefs. *Environmental Conservation* **36**: 327–337.

Luckhurst B, Ward J. 1987. Behavioural dynamics of coral reef fish in Antillean fish traps at Bermuda. *Proceedings of the Gulf and Caribbean Fisheries Institute* **38**: 528–546.

Mahon R, Hunte W. 2001. Trap mesh selectivity and the management of reef fishes. *Fish and Fisheries* **2**: 356–375.

Mangi SC, Roberts CM. 2006. Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. *Marine Pollution Bulletin* **52**: 1646–1660.

Mangi SC, Roberts CM, Rodwell LD. 2007. Financial comparisons of fishing gear used in Kenya's coral reef lagoons. *Ambio* **36**: 671–676.

McClanahan TR. 1995. A coral reef ecosystem fisheries model – impacts of fishing intensity and catch selection on reef structure and processes. *Ecological Modelling* **80**: 1–19.

McClanahan TR. 2010. Effects of fisheries closures and gear restrictions on fishing income in a Kenyan coral reef. *Conservation Biology* **24**: 1519–1528.

McClanahan TR. 2011. Human and coral reef use interactions: from impacts to solutions? *Journal of Experimental Marine Biology and Ecology* **408**: 3–10.

McClanahan TR. 2012. Scaling the management values divide. *Aquatic Conservation: Marine and Freshwater Ecosystems* **22**: 565–568.

McClanahan TR, Cinner JE. 2008. A framework for adaptive gear and ecosystem-based management in the artisanal coral reef fishery of Papua New Guinea. *Aquatic Conservation: Marine and Freshwater Ecosystems* **18**: 493–507.

McClanahan TR, Mangi S. 2004. Gear-based management of a tropical artisanal fishery based on species selectivity and capture size. *Fisheries Management and Ecology* **11**: 51–60.

McClanahan TR, Glaesel H, Rubens J, Kiambo R. 1997. The effects of traditional fisheries management on fisheries yields and the coral-reef ecosystems of southern Kenya. *Environmental Conservation* **24**: 105–120.

McClanahan TR, Maina J, Davies J. 2005. Perceptions of resource users and managers towards fisheries management options in Kenyan coral reefs. *Fisheries Management and Ecology* **12**: 105–112.

McClanahan TR, Cinner JE, Maina J, Graham NAJ, Daw TM, Stead SM, Wamukota A, Brown K, Ateweberhan M, Venus V, Polunin NVC. 2008. Conservation action in a changing climate. *Conservation Letters* **1**: 53–59.

McClanahan TR, Abunge C, Cinner J. 2012. Heterogeneity in fishers' and managers' preferences towards management restrictions and benefits in Kenya. *Environmental Conservation* **39**: 357–369.

Miller RJ. 1979. Saturation of crab traps: reduced entry and escapement. *Journal de le Conseil international pour le Exploracion du Mer* **38**: 338–345.

Miller RJ. 1990. Traps as a survey tool for animal density. *Coral Reef Fisheries Proceedings of the Gulf Caribbean Fisheries Institute* **39**: 331–339.

Mumby PJ, Steneck RS. 2008. Coral reef management and conservation in light of rapidly evolving ecological paradigms. *Trends in Ecology and Evolution* **23**: 555–563.

Munro JL. 1971. Dynamic factors affecting the performance of the Antillean fish trap. *Proceedings of the Gulf and Caribbean Fisheries Institute* **23**: 184–194.

Munro JL. 1974. The mode of operation of Antillean fish traps and the relationships between ingress, escapement, catch and soak. *Journal de le Conseil International pour le Exploracion du Mer* **38**: 337–350.

Munro JL, Sary Z, Gell FR. 2003. Escape gaps: an option for the management of Caribbean trap fisheries. *Annual Proceedings Gulf Caribbean Fisheries Institute* **54**: 28–40.

Pauly D, Palomares ML, Froese R, Sa P, Vakily M, Preikshot D, Wallace S. 2001. Fishing down Canadian aquatic food webs. *Canadian Journal of Fish and Aquatic Science* **58**: 1–12.

Raymundo LJ, Halford AR, Maypa AP, Kerr AM. 2009. Functionally diverse reef-fish communities ameliorate coral disease. *Proceedings of the National Academy of Sciences* **106**: 17076–17070.

Renchen GF, Pittman SJ, Brandt ME. 2012. Investigating the behavioural responses of trapped fishes using underwater video surveillance. *Journal of Fish Biology* **81**: 1611–1625.

Robichaud D, Hunte W. 1997. What factors explain reduced fishing power with increased mesh size of Antillean fish traps? *Proceedings Gulf Caribbean Fisheries Institute* **49**: 273–279.

Robichaud D, Hunte W, Oxenford HA. 1999. Effects of increased mesh size on catch and fishing power of coral reef fish traps. *Fisheries Research* **39**: 275–294.

Sadovy Y. 2005. Trouble on the reef: the imperative for managing vulnerable and valuable fisheries. *Fish and Fisheries* **6**: 167–185.

Sary Z, Oxenford HA, Woodley JD. 1997. Effects of an increase in trap mesh size on an overexploited coral reef fishery at Discovery Bay, Jamaica. *Marine Ecology Progress Series* **154**: 107–120.

Selliah N, Oxenford H, Parker C. 2001. Selecting biodegradable fasteners and testing the effects of escape panels on catch rates of fish traps. *Proceedings of the Gulf and Caribbean Fisheries Institute* **52**: 634–653.

Stewart J, Ferrell DJ. 2003. Escape panels to reduce by-catch in the New South Wales demersal trap fishery. *Marine and Freshwater Research* **53**: 1179–1188.

Treble RJ, Millar RB, Walker TI. 1998. Size-selectivity of lobster pots with escape gaps: application of the SELECT method to the southern rock lobster (*Jasus edwardsii*) fishery in Victoria, Australia. *Fisheries Research* **34**: 289–305.

Wambiji N, Ohtomi J, Fulanda B, Kimani E, Kulundu N, Hossain YM. 2008. Morphometric relationship and condition factor of *Siganus stellatus*, *S. canaliculatus* and *S. sutur* (Pisces: Siganidae) from the Western Indian Ocean Waters. *South Pacific Studies* **29**: 1–15.

Ward J. 1988. Mesh size selection in Antillean arrowhead fish traps. *FAO Fisheries Report* **389**: 455–467.

Williams MJ, Hill BJ. 1982. Factors influencing trap catches and population estimates of the portunid crab *Scylla serrata*. *Marine Biology* **71**: 187–192.

Worm B, Barbier EB, Beaumont N, Duffy JE, Folke C, Halpern BS, Jackson JBC, Lotze HK, Micheli F, Palumbi SR, *et al*. 2006. Impacts of biodiversity loss on ocean ecosystem services. *Science* **314**: 787–790.

Zhou S, Smith AD, Punt AE, Richardson AJ, Gibbs M, Fulton EA, Pascoe S, Bulman C, Bayliss P, Sainsbury K. 2010. Ecosystem-based fisheries management requires a change to the selective fishing philosophy. *Proceedings of the National Academy of Sciences* **107**: 9485–9489.