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Modelling spillover effects of a marine protected area in the Western Indian Ocean

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Abstract

This study estimated the abundance gradient of fishes in the waters surrounding a marine protected area (MPA) and used this information as evidence of spillover from the MPA. Fish landing data were collected from trap fishermen over a 12-month period from the Blue Bay Marine Park on the west coast of Mauritius in the Western Indian Ocean. Two indicators of abundance were used: catch per trap, and number of fish per trap. A Generalised Linear Model was used to standardise the catch data by removing the effects of individual fishermen's productivity from the abundance measurements while the negative-binomial distribution was used for the number of fish. The study found a slight declining gradient of catch beyond the MPA up to a distance of 4 km. The study also established that the individuals belonging to key species caught near the MPA were larger than those of the same species caught more than 4 km away. The negative gradient reinforces the evidence of spillover occurring from the MPA.

Keywords: marine reserve, spillovers, gradient assessment, standardisation, Indian Ocean

Introduction

In the context of this paper a marine protected area (MPA), or marine reserve, is a portion of ocean where fishing and other human activities are prohibited (Hannesson, 1998; Crowder *et al.*, 2000; Sladek-Nowlis and Roberts, 1999; Lorenzo *et al.*, 2016). When an overfished area is closed to harvesting and exploitation, its ecosystem and its resident fish populations recover, leading to so-called "reserve effects" in terms of an increase in biomass, fecundity as well as the proportions of older and larger fish (Bohnsack, 1996; Sladek-Nowlis and Roberts, 1999; Hallwood, 2005; Horta e Costa *et al.* 2013). Over time, the undisturbed area, if it is large enough, returns to a naturally bio-diverse equilibrium (Sladek-Nowlis and Roberts, 1999) and depending on density-dependent mechanisms, the carrying capacity of the protected and adjacent areas, and connectivity of suitable habitats, this translates into an export of post-settlers to the adjacent areas, commonly referred to as "spillover effects" (Chapman and Kramer, 1999; Gell and Roberts, 2003; Forcada *et al.*, 2009; Bellier *et al.*, 2013, Lorenzo *et al.*, 2016). Evidence that MPAs can lead to spillover effects provides opportunities for them to be used as fisheries management tools to sustain fishers in the adjacent areas.

This paper investigates the evidence of spillover effects of a small MPA, the Blue Bay Marine Park (BBMP), located in a heavily fished area in the southeast of Mauritius. As an attempt to conserve the marine ecosystem as a main tourist asset and to reduce fishing pressure in key sites, Mauritius began to establish MPAs around its coasts in 1983. Two marine parks and six fishing reserves have been established, while the process of establishing new protected areas and expanding existing ones is ongoing, especially as part of the marine spatial planning process (Smith, 2017).

The BBMP was declared an MPA and designated a Marine Park in June 2000 under the Fisheries and Marine Resources Act 1998 (Convention on Biological Diversity [CBD], 2013). The total area of the Marine Park is currently 353 hectares. Since the last inventory of the park carried out in 2012, a marked improvement in the fish population (biodiversity and density) has been noted (CBD, 2013). Since the proclamation

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of Blue Bay as an MPA, no fishing activities have been allowed in the conservation area (Fig. 1) while pole and line fishing for leisure is allowed from the shoreline only. No commercial fishing activities are presently being carried out in the park. Conand *et al.* (2016) concluded that biodiversity inventories show some improvements over time. Improvement of the habitats in the BBMP, despite its relatively small size, explains the rising diversity of holothurians (sea cucumbers). Without a comprehensive assessment of the reserve effect, these findings are assumed to be as a result of this phenomenon and are used as the basis to examine the spillover effects.

Fishery scientists have employed various tools to analyse the spillover effects of MPAs (Russ *et al.,* 2004). One of these is to compare variables such as fish density, biomass, size of organisms, and species diversity before and after the establishment of MPAs (Halpern, 2003). However, in many cases, these biological data are not available, and such before-and-after analyses cannot be made (Chapman and Kramer, 1999). A common alternative is therefore to assess the differences in fish population density (and other variables of interest) between sites in a reserve, and sites which have the same ecological features but are located in adjacent areas outside of the MPA. If emigration determines the distribution of fishes, fish density should be higher in the centre of the reserve and decrease gradually toward and beyond the boundaries (Rakitin and Kramer, 1996; Abesamis *et al.,* 2006). Spillover is typically observed through patterns of abundance or catch that decline with distance from reserve boundaries (Halpern *et al.*, 2009).

This, and other gradients of biological features, can be obtained by visual census and tagging of fish inside and outside the MPA (Chapman and Kramer, 1999; Abesamis *et al.,* 2006). However, this method may be costly and time consuming. Moreover, according to Chapman and Kramer (1999), the quantification of the spillover effects should take into account both the spatial and temporal variation in fish distribution. Such gradients can more feasibly be estimated using the catches made by fishers in adjacent areas (Vandeperre *et al.*, 2011). Whilst catch per unit of effort (CPUE) is a poor indicator of abundance for some species, for others it is taken as evidence of spillover when CPUE higher nearer the MPA (Chapman and Kramer, 1999; Murawski *et al.*, 2005; Goñi *et al.*, 2006; Stelzenmuller *et al.*, 2007; Forcada *et al.*, 2009; Bellier *et al.*, 2013). Such fish landing data are commonly used to measure fish abundance (Beverton and Holt, 1957; Kimura, 1981; Harley *et al.*, 2001; Pascoe and Herrero, 2004; Bordalo-Machado, 2006; Stobart *et al.*, 2009), and their use to test for a decreasing abundance gradient with distance from the MPA is commonly justified on both technical and practical grounds. Such fishery-dependent data not only offers greater coverage in space and time but are economically cheaper to collect (Ye and Dennis, 2009).

Translating data on catch rates into an abundance gradient can be an issue in that the coefficient of catchability is stable. The latter is the parameter which relates catch rates as an index of relative abundance to the stock of fish (Squires and Vestergaard, 2015). It is well established that this varies across species. However, even within a species it may not be stable. Only if catchability is constant does catch data reflect abundance. Noting the range of factors that can affect catchability, fishery scientists have adopted a statistical approach to 'standardisation' – the process through which these factors influencing catchability are 'controlled' so that the catch rate data is a truer reflection of abundance. Such standardisation typically uses the Generalised Linear Models (GLMs) or Generalised Additive Models (GAMs).

The main objective of this paper is to provide evidence on the spillover effects of the BBMP by using fish landing data which was collected from a sample of trap fishermen over a 12-month period on the east coast of Mauritius in the Indian Ocean. Two indicators were used from the data: (i) catch per trap (ii) and number of fish per trap. The data was collected through post-trip inquiries with the assistance of professional fish landing officers, and fishermen were required to indicate the location they fished for that trip on a map. Consequently, the statistical analysis had to consider the many factors which may influence catch, including fishers' characteristics, seasonality and habitat characteristics. This is captured through applying the GLM to standardise the catch by assuming a particular distribution for the indicators. Once the extent of a declining gradient was obtained, the study sought to identify the main associated fish species which could be driving the results. It was assumed that the results may be influenced by habitat, and given the limited information on this aspect, the study collected data on the depth of adjacent waters and some characteristics of the main fish species which could be the drivers of the spillover effects. The mean size of the main identified fish species in and the adjacent to the Marine

Park was also determined. No studies on potential spillover from the Marine Park had been undertaken previously, and it is anticipated that this study using spatially-collected catch data will open avenues for further research to further confirm the fisheries benefits of MPAs.

Materials and methods

Study site and data

Mauritius is located in the Indian Ocean approximately 800 km east of Madagascar. The BBMP is located on the southeast coast of Mauritius and was proclaimed a National Park in October 1997 (Fig. 1) and declared a Marine Protected Area and designated a Marine Park in June 2000 under the Fisheries and Marine Resources Act 1998 (Convention on Biological Diversity, 2016). The total area of the Marine Park is currently 353 hectares. The water depth in the park varies from 1 to 150 m (Albion Fisheries Research Centre [AFRC], 2008). Fishing activities with pole and line and basket traps are allowed in the multiple use zone lagoon). According to the CBD (2013), the level of human-induced disturbance or degradation is low, and the area harbours high coral biodiversity. Surveys carried out have revealed the presence of 72 fish species representing 41 genera and 31 families (CBD, 2013). Commercial species and many reef fish, including those that display schooling behaviour, are present in the park. The main fish families found in surveys in the Park include Acanthuridae, Labridae, Scaridae and Serranidae (AFRC, 2008).

There are approximately 350 trap fishers in the area covered by this study. No official list of fishermen was available, however, with the help from fish landing officers, a list of regular fishermen was prepared. This was supplemented by an on-site survey of regular fishermen over approximately one month (December 2014). A total of 179 regular full-time fishermen were noted, from whom 100 were randomly selected. The study attempted to record the fishing locations of this sub-sample of fishermen, as well as details of fish catch for 10 trips spread evenly over the year. The random selection was limited in that it was observed during the interviews that around 15 % of the fishermen were either unable or reluctant to provide the information needed. They were eventually replaced. The survey was conducted from January 2015 to December 2015 and 10 trips were recorded for each fisherman, creating a panel of 100 by 10 observations. For each trip the interviewer recorded the 'total catch of the fisher for the trip in kg'. To ensure that the data was collected properly, assistance was sought from experienced fish landing officers who were fully acquainted with the study sites and were known to the fishermen. The questionnaire was used to record the number of fish of each fish species as well as the weight of the fish from each trip. Fishers were given a map as shown in Figure 1 on which the reef and the waters surrounding the reserve were shown. The map was divided into grid blocks which were numbered and positioned spatially on the map using the QGIS software. Fishers indicated on the map where their traps had been

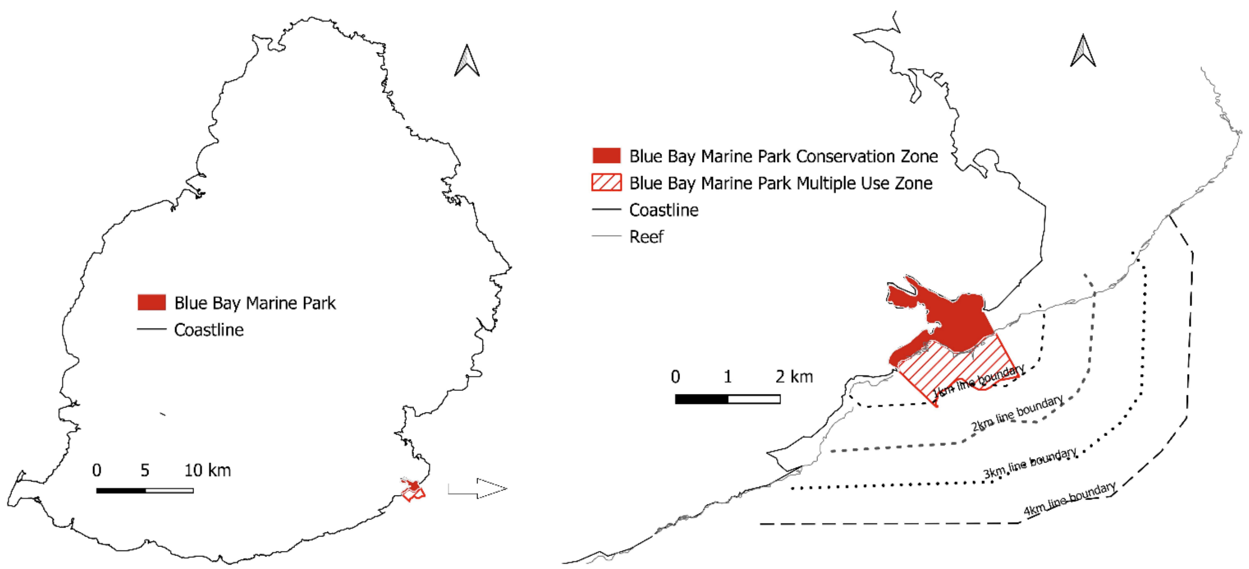

Figure 1. The study area: Blue Bay Marine Park, located on the southeast coast of Mauritius.

located and the route taken to and from those traps. Data collected also included the characteristics of fishermen and of their fishing technology, including the trap sizes and the numbers of traps used.

Conceptual framework and estimation methods

The conceptual framework assumes that a fisherman's catch is proportional to the abundance of fish. While this is a common assumption, some basic information is provided as shown by equation (1) (Maunder and Punt, 2004):

$$C_{ji} = q_{ji} E_{ji} X_i \quad (1)$$

Where C_{ji} = catch for fisher j in area i; q_{ji} = catchability coefficient for fisher j in area i; E_{ji} = effort; and X_i = population density in area i. It follows that catch per unit of effort (*CPUE*) is:

$$CPUE_{ji} = \frac{C_{ji}}{E_{ji}} = q_{ji} X_i \quad (2)$$

Changes in $CPUE_{ji}$ can therefore be due to either changes in the stock density, X_i, or changes in the catchability coefficient (q_{ji}). *Ceteris Paribus* (i.e. with q constant) spatial changes in *CPUE* may reflect other factors, such as habitat differences, rather than overall physical abundance.

In order to estimate stock abundance, statisticians standardise the *CPUE* by adding additional structure through the catchability coefficient (Maunder, 2001). The variables forming the additional structure can be continuous (e.g. sea-surface temperature, price of fish, vessel size). Once the additional structures for modelling the catchability coefficient have been incorporated, the remaining variation in *CPUE* is linked to distance from the MPA to analyse the declining gradient hypothesis. Catch per trip as well as catch per trap were both used as a measure of abundance. This conceptualisation is similar to that used by Goni *et al.* (2006) and Stelzenmuller *et al.* (2007). Following Halpern *et al.* (2009) an exponential decay relationship is given by:

$$X_i = \exp\left(-\beta_{DIS}\, DIS_{im}\right) \quad (3)$$

Where DIS_{im} is the distance from location i to the location of the marine reserve m.

Since the focus was on the artisanal fishers using traps, the number of basket traps (*NBAS*) , and the size of basket trap (*SBAS*) were used as additional structures. Seasonal effect on catches may be significant and, hence, quarterly effects were included through a categorical variable representing the four quarters of the year.

Habitat differences could lead to differences in abundance and therefore play a key role in the findings. There is currently a lack of information on the geographical characteristics of the habitats. The only accurate indicator is the depth of the water which was included as a continuous variable to capture potential habitat differences. In order to probe this issue further, the analysis was supplemented by examining the characteristics of the fish species and their associated habitats.

The predictive response indicator is specified as follows:

$$\eta = q' + quarter + NBAS + SBAS + DEPTH + (quarter \times NBAS) + (quarter \times SBAS) + (quarter \times DEPTH) + (NBAS \times SBAS) + (NBAS \times DEPTH) + (SBAS \times DEPTH) + DIS_m + DIS_m{}^2 + error \quad (4)$$

The square of the distance from the MPA is added to estimate the strength of the relationship. In particular, if there is an L-shape, the term will be redundant while a U-shape will provide a cut-off point.

To further provide insights on the spillover effects, the total individual fish per trip and per trap were also used as the response indicators. Since these data are discrete and positively skewed the response variable was modelled using a negative binomial (Bellier *et al.*, 2013). The log-linear specification is commonly used in count data models to ensure that the conditional expectation is positive (Hausman *et al.*, 1994; Delgado and Kniesner, 1997).

Results

A summary definition of the covariates used in the analysis is provided in Table 1. Distance from the MPA was measured as a linear transect from the border of the MPA to the middle of the 1×1 km grid where the fishing had taken place during the trip.

The goodness of fit was evaluated using the model's scaled deviance and two other criteria; the Akaike Information Criterion (AIC) (Akaike, 1973), and the Schwarz Bayesian Information Criterion (BIC) (Schwarz, 1978). If the selected model fits the data reasonably well, the AIC and the BIC should be low (Su *et al.* 2008; Ye and Dennis 2009).

Table 1. Summary definition of variables.

Variable	Definition	n	Mean	Standard Deviation	Minimum	Maximum
$CPBT_{ji}$	Catch per basket trap for fisher j in location i	1000	1.57	1.20	0.00	25.00
$NBAS_{ji}$	Number of baskets used in the trip for fisher j in location i	1000	8.19	1.70	1.00	13.00
$SBAS_{ji}$	Size of basket (volume) in meter cube for fisher j in location i (feet^3)	1000	21.88	28.58	1.50	216.00
DIS_{im}	Distance from location i to marine reserve m	1000	6.72	2.44	1.00	12.00
$DEPTH_i$	Depth measured in meters in location i	1000	7.06	10.12	1.00	85.00
QU_n for n = 1, 2, 3, 4	Categorical variable representing quarter: Quarter 1: January, February, March Quarter 2: April, May, June Quarter 3: July, August, September Quarter 4: October, November, December					

The analysis started with the null hypothesis that none of the covariates have any influence on the stochastic response variable (catch per fishermen per trip and catch per trap per trip). Table 2 shows the relative performances of the model.

The residual deviance, AIC and BIC decreases as covariates are added, confirming their explanatory power. For instance, adding *NBAS* and *SBAS* reduce the residual deviance (to 220.3) as expected. Correcting for seasonal factors by adding categorical variables representing quarters again reduces the residual variation substantially (and the interaction variables add further explanatory power. Adding distance from the MPA improved the model fit as can be seen from Table 2. So too did adding the square of distance. Table 3 shows a similar analysis when using catch per trap.

Using the outcomes shown in Table 2 and 3, the effect of distance from the MPA on standardised catch per trap was simulated. Figure 2 and 3 show these results. A first observation shows that there is a slight U-shaped relationship between standardised catch and distance, but the rising segment takes place beyond 4 km. Standardised catch per trip declines non-linearly consistently for 4 kms from the MPA. This finding indicates a declining gradient from the spillovers.

It is important to highlight that the analysis took catch from the boundary of the reserve which is the conservation zone (Fig. 4). The multiple use zone where pole

Table 2. Analysis GLM fitted to catch per trip.

	GLM-Normal			
	DF	Residual deviance	AIC	BIC
Null hypothesis	999	392.95	1.91	-6507.90
+ $NBAS_{ji}$	998	267.36	1.53	-6626.57
+ $SBAS_{ji}$	997	220.08	1.33	-6666.95
+ $NBAS_{ji}$ x $SBAS_{ji}$	996	211.22	1.29	-6668.90
+ QU_n + QU_n x $NBAS_{ji}$ + QU_n x $SBAS_{ji}$	987	168.91	1.08	-6649.35
+ $DEPTH_i$ + $DEPTH_i$ x $NBAS_{ji}$ + $DEPTH_i$ x $SBAS_{ji}$ + $DEPTH_i$ x QU_n	977	159.68	1.05	-6584.28
+ DIS_{im} + $DIS_{im}{}^2$	975	156.54	1.03	-6574.62

Table 3. GLM analysis fitted to catch per trap.

	GLM-Normal			
	DF	Residual deviance	AIC	BIC
Null hypothesis	999	302.45	1.64	-6598.40
$+ NBAS_{ji}$	998	291.03	1.61	-6602.82
$+ SBAS_{ji}$	997	237.99	1.41	-6649.03
$+ NBAS_{ji}$ x $SBAS_{ji}$	996	231.85	1.38	-6648.28
$+ QU_n + QU_n$ x $NBAS_{ji} + QU_n$ x $SBAS_{ji}$	987	180.87	1.15	-6637.08
$+ DEPTH_i + DEPTH_i$ x $NBAS_{ji} + DEPTH_i$ x $SBAS_{ji} + DEPTH_i$ x QU_n	977	168.11	1.10	-6576.85
$+ DIS_{im} + DIS_{im}{}^2$	975	164.15	1.08	-6567

and line, and basket trap fishing are allowed occurred one km from this zone.

The second model relates the number of individual fish per basket trap to the distance from the MPA. Table 4 shows the performance of this model.

Adding each subsequent variable reduces the residual variance significantly as well as the AIC and BIC. The simulation exercise is shown in Figure 4.

Figure 4 presents a very different picture. While the mass of fish per trap decreases with distance, the number of fish caught increases continuously with distance from the MPA to 8 km, then stabilises and falls slightly. The suggests that the reserve is contributing large fish to the catch in the adjacent waters.

Discussion

The relationship between catch per trip (in kg) and distance from the MPA observed in this study accords with the negative exponential slope typical of such studies (e.g., Bellier *et al.,* 2013). The effects of fishing characteristics and the seasonal effects in explaining variations in the catch data were expected from the literature (Stelzenmuller *et al.*, 2007). The study infers a declining fish abundance for 4 km from the MPA when using the standardised catch in kg per trip and per trap, as a measure of abundance. This may support the evidence of spillover effects observed in marine reserves as in similar studies (e.g. Roberts *et al.*, 2001; Rakitin and Kramer, 1996; Goni *et al.*, 2006). An important policy issue is the strength of the effect. In other words, when the direct spillover of adult fish is effectively at its minimum extent, by how much has

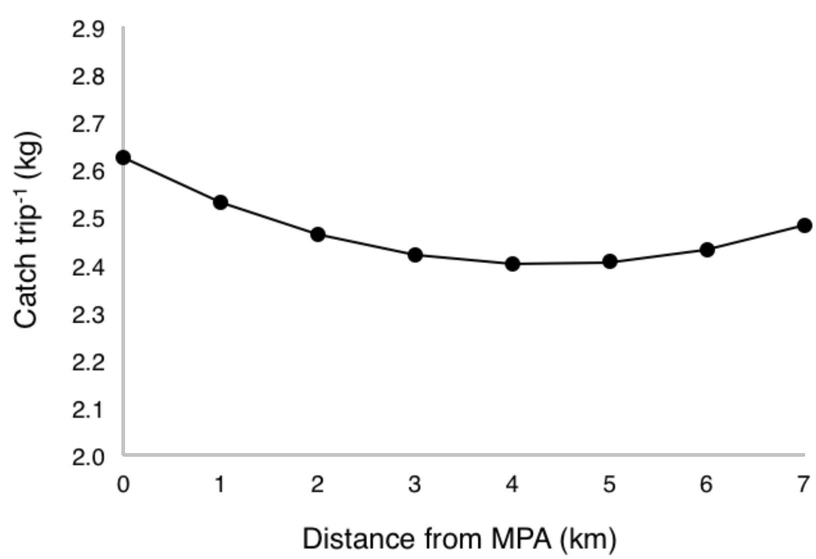

Figure 2. Standardised catch per trip and distance from MPA based on GLM.

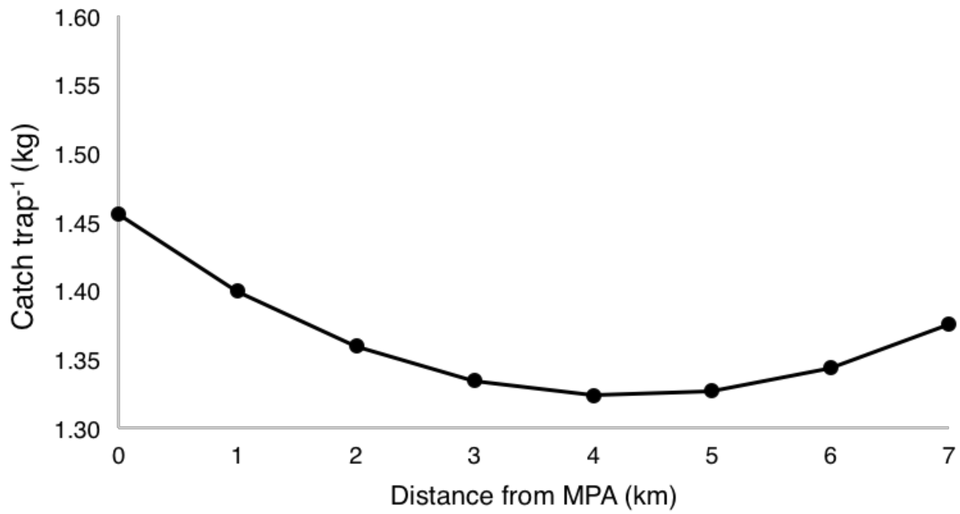

Figure 3. Standardised catch per basket trap and distance from MPA based on GLM.

the catch fallen? Catch per trap from the boundary to 4 km of the MPA shows a decline of 18.5 %.

A fundamental question is why the negative gradient prevails even after controlling for the many fishermen specific effects? If commercial species are mobile and fishermen have free access, fish yield is expected to stabilise, unless the spillovers from the MPA are continuous and systematic. These results may also reflect changes in habitat. It is important to establish that the observed gradient was caused by distance from the reserve and not an additional factor related with habitat characteristics that can change the fish community structure. Unfortunately, this is not easy as there was limited information on geographical characteristics. Some insights may be obtained from an analysis of the depth and specific fish species which exist in the adjacent areas. Figure 4 shows the depth of the waters beyond the boundary of the MPA. At 3 km, the water reaches a depth of around 325 m. Do these habitats host the fish species which may drive the results? An identification of the fish species in those waters may assist in answering this question.

Table 5 shows that there were four main fish species which were present on most of the fishing trips. The percentage of total fishing trips within 4 km of the MPA that caught Bluespine unicornfish *(Naso unicornis,* Forsskål, 1775) was 60.77 %, Spangled emperor (*Lethrinus nebulous* Forsskål, 1775) 34.4 %, Shoemaker spinefoot

Table 4. GLM analysis fitted to the number of fish per basket trap.

	Individual fish per trip				Individual fish per trap			
	DF	Residual deviance	AIC	BIC	DF	Residual deviance	AIC	BIC
Null hypothesis	999	428.48	8.40	-6472.37	999	353.56	4.56	-6547.29
+ $NBAS_{ji}$	998	423.96	8.40	-6469.99	998	315.91	4.52	-6578.03
+ $SBAS_{ji}$	997	401.16	8.37	-6485.88	997	303.23	4.51	-6583.80
+ $NBAS_{ji}$ x $SBAS_{ji}$	996	400.59	8.38	-6479.53	996	302.83	4.51	-6577.30
+ QU_n	993	3.96.68	8.38	-6462.73	993	299.26	4.51	-6560.15
+ QU_n x $NBAS_{ji}$, QU_n x $SBAS_{ji}$	987	374.79	8.37	-6443.16	987	275.79	4.51	-6542.16
DIS_{im}	986	367.10	8.37	-6443.95	986	268.28	4.50	-6542.77
$DIS_{im}{}^2$	985	362.92	8.36	-6441.22	985	265.52	4.50	-6538.62

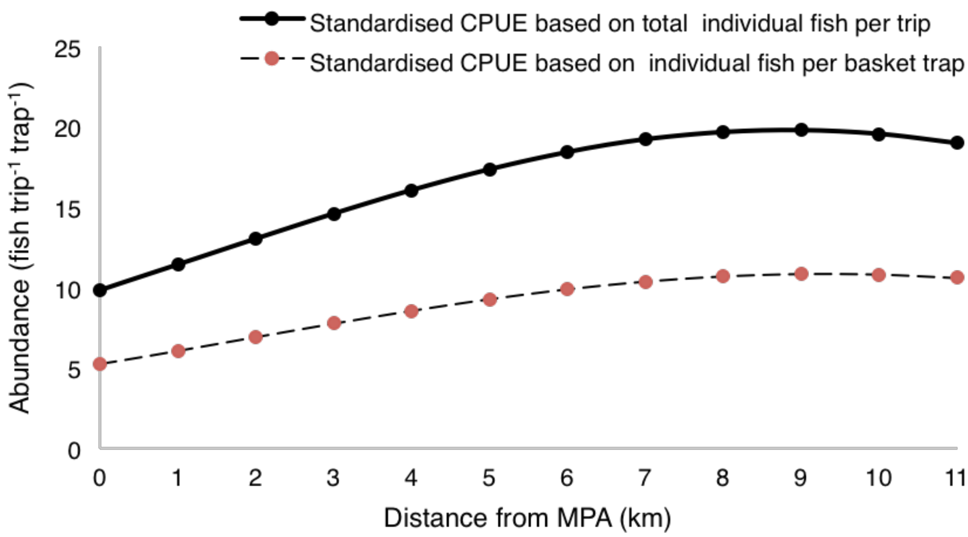

Figure 4. Number of individual fish per trip and per basket trap and distance from MPA.

(*Siganus sutor*, Valenciennes, 1835) 18.1 %, and Blue-barred parrotfish (*Scarus ghobun*, Forsskål, 1775) 13.9 %.

N. unicornis has a home range which extends a linear distance of 0.3 km to 1 km (Hardman *et al*., 2010; Marshell *et al*., 2011; Green *et al*. 2015). It is reef-associated and has been found within a depth range 1 – 180 m (Froese and Pauly, 2021). *L. nebulosus* inhabits nearshore and offshore coral reefs, coralline lagoons, seagrass beds, mangrove swamps, coastal sand and rock areas, to depths of 75 m (Froese and Pauly 2021). According to Pillans *et al*. (2014), the average home range for resident individuals is about 8 km compared to average sanctuary zone size of 30 km². *S. ghobun* is found in a depth range of 1 – 90 m while *S. sutor* inhabits seagrass beds and rocky/coral reefs with a depth of 1-50 m, but typically 1-12 m.

These variations across fish species provide relevant information on the extent of selective fishing effort targeting high value species and on the behavioural characteristics of each species. According to the literature, relatively mobile fish should exhibit a shallower gradient of abundance across the reserve boundaries in a hyperbolic shape, whereas sedentary fish should exhibit a steep linear gradient and highly mobile fish a flat gradient. Species that spend part of their life in the reserve, but then move three or four kms away include species such as *S. sutor* and *L. nebulous*. *N. unicornis* is also a highly mobile fish.

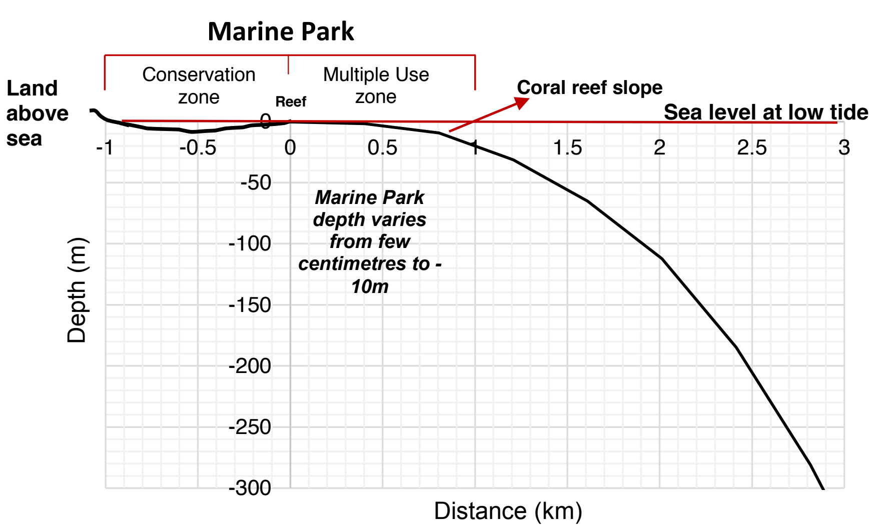

Figure 5. Depth of the Blue Bay Marine Park.

Table 5. Main fish species near the MPA.

Common names	Fish species	% of trips with the fish species	
		Within 4km from MPA	Beyond 4km from MPA
Shoemaker spinefoot	*Siganus sutor*	18.03	48.29
Bluespine unicornfish	*Naso unicornis*	**60.66**	30.87
Black grouper	*Epinephelus fasciatus*	7.38	22.32
Spangled emperor	*Lethrinus nebulous*	**34.43**	11.62
Rock flagtail	*Kuhlia rupestris*	6.56	3.53
Sky emperor	*Lethrinus mahsena*	6.56	8.66
Blue-barred parrotfish	*Scarrus ghobun*	**13.93**	26.42
Goatfishes	*Parupeneus* sp.	5.74	16.51
Doctorfish	*Acanthurus chirurgus*	3.28	6.61
Kingfishes	*Caranx* sp.	0.82	5.35

Further insights were obtained from the analysis of the weight of these fish species.

When individual number of fish per trap is considered a rising gradient of abundance is apparent; i.e., there seems to be more fish caught in the traps the further one moves from the reserve. A naïve interpretation of this is that this is inconsistent with spillover effects from an MPA. However, in waters close to the MPA the mass of fish per trap is higher even though the number of fish per trap is less; i.e., the fish caught near the MPA are larger, while the abundant juveniles are found further away. This is in fact consistent with the observed effect of MPAs; mean size should be smaller in non-reserve than in reserve areas because fishing mortality will reduce the proportion of older (hence larger) fish in the non-reserve (Rakitin and Kramer, 1996). Gell and Roberts (2003) point out that inside reserves, when the individuals of which those populations are comprised grow larger, they also develop increased reproductive potential. Reserves should serve to increase the mean sizes of sexually mature fish of each species in the community. Fish whose home range is fully located in the reserve should be bigger than those whose home range is only partly in the reserve, which in turn will be bigger than those whose home range is entirely outside the reserve. Moreover, in consequence of such growth in populations and amongst individuals, density-dependent emigration is expected to increase. This is a consequence of rising frequency of aggressive interactions between conspecifics as density and average size of targeted fish increase (Abesamis and Russ, 2005). These higher rates of aggressive interactions induce subordinate fish to relocate to home ranges outside the reserve (Kramer and Chapman, 1999). If such density-dependent aggressive interactions occur, with larger fish dominating smaller fish, a consequence is a gradient of mean sizes declining with distance from the reserve (Abesamis and Russ, 2005).

In order to examine whether mean size was higher near the MPA, the weights of the main fish species which are recorded near the MPA were collected, and the difference in their sample means was tested. These results are shown in Table 6. A clear observation from these results is that 9 out of the 11 fish species showed a higher weight within a 4 km radius of the MPA than in a zone more than 4 km from the MPA, with 6 of them having differences in means which are statistically significant. The main differences in mean weight was displayed in *N. unicornis, L. mahsena,* and *S. ghobun* and to a lesser extent *S. sutor* and *L. nebulous.* However, *L. mahsena* did not make up much of the catch. Consequently, the four other identified fish species could explain the declining gradient of individual weights with distance from the Park.

Table 6 also shows the fish species which were caught beyond the 4 km boundary. The finding that there was greater abundance of different types of fish species in those waters explains the rising segment of the standardised catches. Moving further away from the marine reserve, the fishing area is located outside the reef where

Table 6. Average weight of individual fish near the MPA.

Common names	Fish species	Average weight of individual fish (g)		Differences in mean test t-statistics (p-value)	Types of fish S=sedentary, V=vagile fish, HV=highly vagile
		Within 4km from MPA	Beyond 4km from MPA		
Shoemaker spinefoot	*Siganus sutor*	582.72	522.24	-2.03**	V
Bluespine unicornfish	*Naso unicornis*	1959.46	1484.21	-4.92***	HV
Black grouper	*Epinephelus fasciatus*	193.75	166.75	-0.766	S
Spangled emperor	*Lethrinus nebulous*	1102.33	922.88	2.44 **	V
Rock flagtail	*Kuhlia rupestris*	1116	828	-3.32 ***	HV
Sky emperor	*Lethrinus mahsena*	725	569.38	-1.77*	V
Blue-barred parrotfish	*Scarrus ghobun*	1085.29	680.03	3.75***	V
Goatfishes	*Parupeneus* sp.	285.71	292.25	0.11	V
Doctorfish	*Acanthurus* sp.	150	139.66	0.37	V
Kingfishes	*Caranx* sp.	NA	666.67	NA	HV

***=significant at 1%, **=significant at 5% and *=significant at 10%
NA: Not available

both the stock of fish and the number of fish species are relatively higher, given the depth of the waters (Fig. 4). The inclusion of the fish catch from these waters in the analysis was deemed important since the data was collected at fish landing sites and the estimation requires sufficient observations to produce an appropriate fit of the data to the degrees of freedom.

Conclusion

The results of this study show that catches were slightly higher in waters adjacent to of the Marine Park and the size of the fish near the reserve was relatively larger. These findings reinforce the available evidence of the spillover effects of marine reserves and consequent changes in fish age distribution, with a greater number of older fish within and close to the reserve. The negative gradient was most likely driven by the four fish species *N. unicornis*, *L. nebulous*, *S. sutor*, and *Scarrus ghobun*. Spatial catch data could be an effective instrument to assess the impact of MPAs on adjacent waters on a regular basis, as compared to comprehensive oceanographic assessments and visual census techniques which require greater financial and logistic resources. There is a major caveat, however. While the catch data was collected with a degree of accuracy, the findings depended solely on fishermen's responses regarding their fishing locations. The use of GPS would greatly enhance the precision of these locations. Moreover, the study only measured changes in fish abundance and size from the edge of the MPA where fishing is allowed. Theory suggests that there should be a decline from a point inside the MPA boundary. It is plausible that these results considerably understate the impacts of the reserve. Lastly, the study did not consider the likely impacts of fishing intensity on the fish population which may also influence the results. These issues open avenues for further research.

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