Marine Protected Areas in the Management of Artisanal Fisheries

By

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DECLARATION

I, **Mohammed Akthar Riad Sultan**, hereby declare that the work on which this dissertation/thesis is based is my original work (except where acknowledgements indicate otherwise) and that neither the whole work nor any part of it has been, is being, or is to be submitted for another degree in this or any other university.

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Abstract

This thesis examines spatial patterns of effort by fishers in waters adjoining a marine protected area, and their impacts on the reserve's ability to reduce the impacts of over-fishing. The artisanal fisheries operating near Blue Bay Marine Park, Mauritius, provided case-study data for analysis. Two forms of spatial connectivity were evaluated. The first focused on spillovers of adult and juvenile fish from the MPA. Data on catch and geographical fishing location were collected from a sample of fishers over 12 months. Assessment of spillover gradients using Generalised Linear Models and economic production functions found evidence of spillovers up to 4 km from the MPA. The second connectivity study modelled fishers' spatial behaviour, using a random utility framework and random parameters logit estimation. Fishers appeared sensitive to the expected catch, its variability, human risk and distance travelled. Expected catch and catch variability were modelled using a Just and Pope production function. The two connectivities were integrated in bioeconomic models to assess the relationship between reserve size, fishing effort and harvests. Using a Beverton and Holt recruitment function, an age-structured bioeconomic model for the dominant target species, Unicorn fish (*Naso unicornis*), was constructed from the catch data. In the hypothetical case of an initially over-exploited fishery, an inverted-U relationship between fisheries benefits and the Marine Park's size was observed. No pre-reserve data was available for use as a counterfactual; the model was therefore used to estimate the pre-reserve population. This was then used to assess the performance of the Marine Park in enhancing fish population and biomass. In the light of the findings, the thesis concludes that the extent and magnitude of fishing effort should be considered when debating the placement and size of marine reserve, and proposes a step-wise approach to integrate fishers' behaviour in the evaluation of existing and new MPAs.

1. Introduction

The Republic of Mauritius is a small island state with a total land surface of 2,040 km^2, encompassing the main island of Mauritius (1,865 km^2), and Rodrigues (109 km^2). It possesses a maritime zone of 2.3 million km^2 with an Exclusive Economic Zone of 1.96 million km^2, and a continental shelf of 396000 km^2 co-managed with the Republic of Seychelles (Government of Mauritius, 2013). Since its discovery by the Dutch in 1638, Mauritius has been heavily, and often disastrously, exploited for its natural resources; ebony trees, turtles, flightless dodos, and latterly, its fisheries. The island had soon started facing human pressures on its land and marine resources (IUCN, 1993). At independence in 1968, Mauritius embarked on an export-led industrialisation, with further development of the tourism sector in the 1980s. Its remarkable economic success over the last 50 years, however, was accompanied by increasing pressures on already profoundly modified terrestrial and marine ecosystems. The depletion of the island’s lagoon fisheries, which are important to both tourism and local artisanal fishers, soon became a concern for policy makers (Paul 1987, Harper and Zeller 2011). The symptoms of overfishing were clear, catch rates had declined as had mean sizes in the fish population (Paul, 1987: Harper & Zeller, 2011).

In 1983, in an attempt to conserve the marine ecosystem as a main tourist asset and to reduce fishing pressure in key sites, Mauritius began to establish Marine Protected Areas (MPAs) around its coasts. There are currently two marine parks and six fishing reserves. Until now, the rationale underlying the site-selection for these MPAs has been dominated by touristic value. However, the process of establishing new protected areas and expanding existing ones is ongoing, especially as an option for marine spatial planning (Kelleher, 2017: Smith, 2017).

In Mauritius, ecological parameters are primarily used as the guiding principle for the placement of MPAs (Castro de la Mata, 2012). An aspect often overlooked is that reserves also modify the behaviour and effort of artisanal fishermen. The establishment of MPAs is likely to squeeze fishers into a smaller fishing space, affecting the catch and livelihoods of fishermen. This is not helped by the fact that artisanal fishers remain largely in the lagoon of Mauritius and are the poorest in the economy. There is currently a debate on the role of MPAs in fisheries management and planning in Mauritius (Morgan et al., 2011). A problem is how to configure future MPAs so as to benefit not only tourism and recreational users, but also local artisanal fishers.

There is increasing agreement among scholars that MPAs tend to increase the size and abundance of resident fish (Rakitin & Kramer, 1996: Gell & Roberts, 2003: Gaines et al., 2010). For fishermen to benefit, these increases have to result in spillovers, i.e. movement of fish and larvae from the protected areas into the adjacent waters. However, such spillovers may induce a spatial redistribution of fishing efforts. In the case of small marine reserves as observed in Mauritius, it seems intuitively likely that fishermen would be drawn to the edges of the reserves where fishing pressure would rise (Kellner et al., 2007). Accordingly, fisheries benefits might fail to materialise.

The design of an inshore MPA; its size, its location, its effect on fishermen's travel costs etc, clearly impact on local fishermen. This thesis investigates these relationships through a study of a small MPA, the Blue Bay Marine Park (BBMP), located in a heavily fished area in the South East of Mauritius. The Park was established to meet a touristic objective rather than to generate benefits to fishermen. The study attempts to establish whether the BBMP has benefited fish stocks in the adjacent areas and examines the responses of fishers to it. It aims at seeking a scientific explanation at the current performance of the Park through an analysis of both the biological as well as economic factors. Finally, based on the findings, it attempts to develop a step-wise approach to integrate fishers' behaviour in the evaluation of existing and new MPAs.

In order to meet the above objectives, the thesis models two sets of spatial effects - the first being the movement of adult and juvenile fish from the MPA, and the second being the fishers' spatial reactions to it. The modelling is based on fishing data collected from a sample of fishers over 12 months in the study site. This included catch per unit effort by individual fish species, and locational data on fishing sites. The extent and type of spillovers, are modelled and quantified in chapter three - the spillover gradient being assessed using Generalised Linear Models and economic production functions. The reactions of local artisanal fishermen to the reserve are analysed in chapter four. This uses the random utility framework and random parameters logit estimation to model the spatial behaviours of fishers. The main objective is to examine the factors determining fishing location of fishers, including attitudes to risk.

In the fifth chapter, the effects of the reserve on fish and fishermen are integrated in an age-structured bioeconomic model for the dominant target species, Unicornfish (*Naso unicornis*), using a Beverton and Holt recruitment function. The model is constructed from catch data of artisanal fishermen in the vicinity of the BBMP, and a simulation exercise is conducted to assess the interaction between reserve size, harvests and conservation effects of the MPA. The study considers the size of reserve needed if both conservation and fisheries are to benefit. The results are particularly relevant for Mauritius in the attempt to modify the BBMP, as well as in the design of new MPAs, if the interests of artisanal fishers are to be protected.

Chapter six further analyses the process whereby the Blue Bay MPA influences fishing effort spatially, and how this in turn affects the stock of fish and harvest in the area. .

Anyone hoping to manage the lagoon fisheries that take place inside Mauritius's fringing reef faces a fundamental problem; conventional command and control instruments are not feasible. The multitude of small vessels, the low entry cost, and the difficulties of monitoring catch and effort, mean that quotas on landings and effort would be unenforceable, and taxes would be uncollectable. This has prompted fisheries managers to consider MPAs more seriously as a management tool to address widespread overfishing. However, as this thesis shows, the success of such a policy may depend critically on the spatial behaviour of fishers and the sizes and locations selected for new reserves. It is hoped that the data collected and analysed in the course of this dissertation thesis will enhance our understanding of the highly complex interactions between marine reserves, the behaviours of artisanal fishers and the benefits of the reserves to them.

2. How, when, and where are marine reserves effective? A selective review of emerging economics with biological science

Abstract

Decisions regarding the siting and size of MPAs are frequently rooted in the biophysical characteristics of competing sites. However, as reserve creation directly impacts on the welfare of many interest groups, human factors condition its costs and benefits over time. This review begins with the biological science of marine reserves, their main impacts and the key biophysical issues in their design. It then investigates the roles of economic issues in such decision. Fishers remain the apex predator in a fishery. No matter what the aims of a marine reserve are, they are likely to be influenced by fishermen – the reserve affects them, and they affect it. Consequently, MPAs are often treated as fisheries management tools.

There are two commons economic objectives in fisheries management. One is preventing the dissipation of rents through open access. It is noted that MPAs can do little in this regard. The second is to raise the sustainable yield by preserving a stock of large fecund breeding adults. If the objectives are both conservation and fishery benefits in the adjacent areas, a reserve's effectiveness depends on the (i) extent to which it helps to rebuild the population of fish species within the reserve, (ii) fishers' spatial redistribution of effort and (iii) resulting spillover effects and whether or not these offset the upward shift in fishing intensity in the remaining fishing grounds. The review concludes that economic factors that incentivize fishers' behaviour should be considered in the configuration of MPAs.

Keywords: marine reserve, biological processes, conservation effect, fisheries benefits, fisheries management

2.1.Introduction

A fishery's productivity is determined through the intersection of three inter-related systems: the aquatic biota, their habitats, and the humans that harvest them (Lackey 2005, Anderson & Seijo, 2010:11). The condition of the aquatic biota will be influenced by natural factors such as recruitment and growth rates, natural mortality and fecundity of the fish species, as well as by the habitat quality, and fishing mortality –which in turn is determined by systems of catch or effort regulation, and by available technologies and market conditions. Traditionally, fisheries managers assess and monitor both the aquatic biota and human-related activities (mainly fishing effort). Fishing effort affects the populations of target species, their preys and predators, and their habitats. Controlling fishers' effort and catch are well established methods of managing fisheries. Effort can be controlled by limiting licenses, imposing taxes, restricting gear and establishing closed season, while catch can be managed through direct limits, taxes or controlling fish size (Squires, 1987, Holland & Brazee, 1996:, Christou & Idels, 2012), or allocation of property rights(Anderson et al., 2011).

Given the difficulties of managing effort reductions, an alternative management strategy has been the establishment of marine protected areas (MPAs) or marine reserves (Halpern, 2003; Rice et al., 2012; Akpalu & Bitew, 2012: Fenner, 2016; Gallacher, 2016). As an ecosystem-based management strategy, it should take cognisance of all the three systems mentioned above. The terms, 'marine protected area (MPA)', 'marine reserve', 'closed area', 'harvest refugium', 'marine park', and 'sanctuary' are used interchangeably, though Agardy (2000a) warns that they may cause semantic difficulties given they are not perfect synonyms and may have differing implications for size, design, and management objectives.

A *refugium* is a place where exploitation of one or more species, usually of fish or shellfish is restricted. *Closed area* and *harvest refugium* are sometimes synonymous, but while 'closed area' can be a zone with a general prohibition on entry, it can also indicate an area in which extraction of non-living resources, such as oil and gas, or living resources are restricted.

A *marine reserve* is defined as a spatial area where some or all species receive long-term protection from harvesting (Grafton et al*.,* 2005). *Reserves* or *no take zones* which exist in certain locations because of their natural or physical features are often part of larger *marine protected areas*, in which there is less protection and which allow for some consumptive uses (Grafton et al*.,* 2005). However, reserves and marine protected areas are effectively synonymous in some countries, although 'reserve' can refer to a particular type of protected area such as a biosphere reserve or, as in Britain, to an area closed to all fishing (Agardy, 2000a)[1].

Over the last few decades, different rationales for the creation of reserves have emerged. Conservationists refer to the complex interactions between communities of organisms, ecological processes and ecosystems, and emphasise the loss of biodiversity due to anthropogenic activities as a main motivation for the establishment of marine reserves (Agardy, 2000b). Accordingly, marine reserves have merit in their own right.

Marine managers argue that once harvesting ceases, marine reserves lead to a replenishment of the stock of sedentary resident species, an upwards shift in the size and age distributions of the mature, and an improvement in fecundity. For species that are resident in an area, reserve formation raises the rate at which harvested fish can be replaced, i.e., they shift sustainable yields upwards. Just as theory predicts that an early indicator of overfishing is a decline in the size of the average fish caught, so a reserve raises mean sizes, thereby increasing fecundity and correcting for the genetic effects of selectively harvesting large adult fish (Rodwell & Roberts, 2004; Roberts, 2012).

Marine modellers and empiricists have taken a scientific view on the effectiveness of marine reserves. While empiricists attempt to search for evidence of the efficacy of protected areas in the world (for e.g. Roberts et al*.,* 2001; Whitmarsh et al., 2002; Gell & Roberts, 2003), modellers examine critical factors such as population dynamics, habitats, dispersal of larval and migration, to determine when, why and how protected areas should be successful (for e.g. Botsford et al*.,* 1999).

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[1] Marine reserves, marine protected areas, and no-take areas are used interchangeably in this thesis.

There are strong counter arguments presented by those that oppose marine reserves (Roberts et al., 2005; Smith et al., 2010; Voyer et al. 2014). These argue that, when areas are closed to fishing, fishers may consequently have to go farther and to more dangerous areas to fish. The same catch is targeted, but the fishing effort is concentrated into a smaller area, condensing fishing effort, and increasing the risks of local depletion and of conflict between fishers (Voyer et al., 2014; Fenner, 2016). Moreover, if general effort controls are already in place, and a fishing ground as a whole is at its maximum sustainable yield, setting aside an area of it as a reserve would mean a slowing growth rate in the stock of fish within the protected area, while transfer of effort will mean excessive harvesting, and potentially a collapse of fisheries, in the remainder of the fishing ground. This has led some fishers to oppose the creation of MPAs especially in highly accessible and preferred sites. However, this argument clearly depends on whether or not the reserve increases rates of recruitment, and if so, how fast the resulting recruits mature and spread into fishable waters.

The effectiveness of a marine reserve depends on the objectives it is set to achieve (Agardy, 2000b). However, it also depends on whether the design of the reserve is suited to those objectives, and whether the reserve is established and enforced adequately. The establishment of a reserve will impact on various stakeholders, the main one being fishers. There is clearly a human factor in the design of MPA (Agardy, 2000a; Sowman et al., 2011; 2018). While the biophysical aspects of marine reserves are well understood, the economic factors that are likely to influence its success should also be recognised in reserve design.

2.2. The biological science of marine reserves ‑ a brief review

When an overfished area is closed to harvesting and exploitation, its ecosystem and its resident fish populations recover, the biomass increases, as do the proportions of older and larger fish. Over time the undisturbed area, if it is large enough, returns to a naturally bio-diverse equilibrium (Sladek-Nowlis & Roberts, 1999; Armstrong, 2007). The implications of this recovery process have been divided into three components (Bohnsack, 1996; Sladek-Nowlis & Roberts, 1999; Hallwood, 2005): *a reserve or conservation effect* which provides improvements in spawning and exploited biomass levels and in recruitment; *a stability effect* in the form of reduced variance of fish populations and catches; and *spillover effect* as fisheries yields in adjacent areas increase.

Other benefits ascribed to marine reserves include protecting against extinction, potentially mitigating against and reducing the impacts of failures in managing effort or controlling catch, and acting as buffer against environmental shocks (Lauck et al., 1998; Sumaila, 1998; Grafton et al., 2006; Greenville & Macaulay, 2007; Kvamsdal, 2011). Hilborn et al. (2004) suggest that benefits of marine reserves should include their ability to simultaneously manage and conserve multiple species since conventional management tools such as quotas cannot be extended to all species. They also protect benthic habitats and provide scientific reference areas for future fisheries modelling.

Conservation effect of marine reserves

Kamil et al. (2017) suggest that the conservation of the ecosystem and its biophysical aspects is the main objective of marine reserves. The conservation effect occurs within the reserves when the absence of fishing activity allows degraded habitats to improve and enhances the population and biomass of targeted fish species. The change in habitats can also benefits non-targeted fish, and by-catch species. Hilborn et al. (2004) make reference to the positive effect of reserves on sedentary organisms whose movements are short-range in comparison to the spatial scale of the fishing process, or the export and dispersal of pelagic larvae. The increase in the size of mature fish also influences the recruitment effect which further improves the targeted population.

Mosqueira et al. (2000) conducted a meta-analysis of studies on the conservation effect of marine reserves and concluded that, when all species were considered, fish were on average 3.7 times more abundant inside reserves than outside. As expected, when the study differentiated between target and non-targeted species, it found that the abundance differential was greater for target species, for non-targeted ones, abundance was similar inside and outside reserves. The four commonly targeted families that benefitted most in terms of size and abundance inside reserves were *Choetodontidae, Labridae, Lethrinidae and Serranidae*. The study also shows the abundance of large individuals being up to 33 times higher inside the reserves. Thus, MPAs may benefit a commercial species if they are sedentary. However, a reserve is of little use for migratory species unless designed to protect breeding or spawning sites

Halpern's meta-analysis of marine reserves (2003) surveyed 89 studies using four biological measures: density, biomass, organism size and diversity, found that reserves increased values of all four. On average, the studies surveyed found that marine reserves, appear to double density, nearly triple biomass, and raise organism size and diversity by 20-30%, relative to comparable unprotected areas.

Empirical support for conservation effect is also observed from country-specific case studies. Roberts et al. (2001) noted rising total abundance inside and outside Soufrere Marine Management area in the Caribbean island of St Lucia. Similarly, Whitmarsh et al. (2002), investigating the effects of a local trawl ban in the Gulf of Castellammare, off Sicily, found a large positive impact on demersal stocks. Acosta (2002) found rising densities of spiny losbters and queen conch by a factor of 3 and 4.5 respectively in the Glover's Reef, Belize. Gell and Roberts (2003) report that reserves in New Zealand, Washington State in US, and Kenya's Mombasa Marine National Park, have experienced increases in egg numbers inside the reserves. Russ et al. (2004) assessed the biomass of two families of reef fish 18 years after the establishment of a no-take reserve off Apo Island in the Philippines, and observed that the fish biomass had tripled. Similarly Wielgus et al. (2008) noted enhanced biomass in a no-take marine reserve in the Gulf of California, Mexico.

Stability effect of marine reserves

The effect of the reserve on stability depends on how 'stability' is defined and measured. If stability is defined as a convergence towards an equilibrium, reserve creates a faster convergence or allows it to happen when the no reserve system does not converge at all (Pezzey et al., 2000). Given that fishing is a naturally high risk livelihood, any reduction in resource volatility will be of value to fishers. However, when effort is suddenly confined to a reduced fishing ground, reserves tend to enhance the magnitude of the first few oscillations of stock (Pezzey et al., 2000). Deeper initial oscillations mean that some support must be directed towards fishers during the transition period.

The marine environment is highly volatile and unpredictable. According to Rodwell and Roberts (2004), increases in abundance and body size translate into enhanced reproductive potential, reducing the variability in future fish catches. This adds support to Sladek-Nowlis and Roberts (1999) who show that reserves reduce variability in modelled catches when a stochastic component is added to larval survivorship to reflect natural fluctuation in the model.

Reserves can also act as a 'hedge' against negative shocks and enhance resilience against disturbances (Lauck et al., 1998; Grafton et al., 2005; Grafton et al., 2006; Grafton & Kompas, 2014). They can therefore allow for a larger harvest immediately following a shock. Grafton et al. (2006) refer to two stochastic shocks: the first may be either positive or negative and represents a temporal variation in the fish populations; the second is a negative shock which occurs randomly over time. In these terms, the larger and more frequent are such negative shocks, and the greater their effect on the harvestable stock (relative to the stock in reserves), the more valuable are marine reserves and the larger is their optimal size. However, this argument depends on the nature and magnitude of the shocks

Rodwell and Roberts (2004) added a further dimension to the role of MPAs, by describing their value in the face of errors in stocks assessments, or when the impacts of regulations are not accurately estimated. Policy based on scientific stock estimates can be damaging if the estimates are incorrect (e.g. if too great an offtake is permitted at any point in time). Even the best of modellers and researchers may miscall fluctuations in a resource, and TACs rarely parallel population fluctuations. There is thus a risk that TAC rises just as the natural population dips. Again, the more significant this risk, the more valuable the services provided by a marine reserve, and the larger the warranted reserve size.

Fisheries benefits of marine reserves

Fisheries in the areas adjacent to marine reserves can benefit through migration and/or through density-dependent dispersal (Roberts et al., 2001; Armstrong, 2007). Technically, closing an area in an exploited open access system enhances aggregate harvests whenever the system dispersal benefits to the remaining area exceed foregone harvest (Sanchirico & Wilen 2001). Such a rise in yield from the fish migrating out of the reserves is usually presented as an important benefit of a marine reserve.

Two mechanisms exist for system dispersal which may benefit adjacent fisheries: spillover effects and export effects (Sladek-Nowlis & Roberts 1999; McClanahan & Mangi 2000; Gell & Roberts 2003). Spillover effects refer to the emigration of adults and juveniles across borders, while export effects refer to increases in regional recruitment and regional catches as a result of net outward movement of pelagic eggs and larvae (Gell & Roberts 2003). While the spillover effect is easier to measure, there seem to have been far fewer estimates of the impacts of MPAs on recruitment. The distances of the spillover and export effect differ markedly. In the case of reef fish, the spillover effect is generally expected over scales of hundreds of metres to kilometres (McClanahan & Mangi, 2000) while recruitment effect is expected to operate on the scales of dispersal of pelagic larvae, that is at least tens of kilometres (Russ et al.*,* 2004).

'Spillover effect' of marine reserves

Lorenzo et al. (2015) warn that the terminology is no longer clear-cut. "Spillover" is used to describe movement of adults and juveniles from MPAs. In this respect, the movements of individual fish are either density independent or density dependent.

In the case of density independent, the dispersal flow is independent of the population densities so that in the sinks, biomass continues to flow between patches even after each population has reached its natural equilibrium. Grüss et al. (2011) divide density independent spillover into four broad spatial patterns: movement within a home range, nomadism, adult migration and ontogenetic migrations.

A *home range* is the area in which an individual spends the majority of its time and engages in most of the routine activities of foraging and resting (Kramer & Chapman, 1999; Grüss et al.*,*2011; Green et al.*,*2015). Many species also undertake regular movements to and from resident spawning aggregations, and these movements also occur within home ranges. There is a positive relationship between body-size and home range size. Larger adults within same species need more space to provide enough resources to accommodate their greater energetic requirements and range of behaviour. Species with large body size, especially reef fishes, are more likely to explore resources over larger areas than small species (Grüss et al., 2011). Hence, MPAs with larger individual fish are likely to have more density-independent spillovers.

Habitat characteristics and geometry such as coral type, intensity of algal cover, reef type, as well as structure, size and shape can also influence movement patterns (Crowder et al., 2000). Some species exhibit crepuscular movements between daytime resting areas and night-time feeding areas on a daily basis. Other species exhibit movement patterns in response to social organisation and behavioural life-history traits. Such movements may be seasonal, or influenced by time and time of day (Grüss et al., 2011).

Nomadism is a movement by large marine organisms over large areas where habitat is relatively heterogeneous (Grüss et al., 2011). This pattern of movement, being mostly random, occurs over distances ranging from a few hundred meters to some hundreds of kilometers, with long-distance movements punctuated by small-scale displacements during foraging.

Adult migrations can occur because fish or invertebrates need to reach specific remote spawning, nursery or feeding areas and/or in response to environmental changes. Protection of such areas may be important in selection of sites for reserves.

Ontogenetic migration (also known as 'ontogenetic habitat shifts') is a particular type of migration, where fish change habitats as they grow, usually over somewhat smaller spatial scales than other types of migration (Grüss et al., 2011).

Density dependent dispersal processes have biomass flows dependent upon relative population densities (Abesamis & Russ, 2005). Rising densities within an MPA increase competition between individuals, triggering a movement/transfer of individuals/biomass towards adjacent (fished) areas with lower population densities (Kramer & Chapman 1999; Lorenzo et al., 2015). Density-dependent movements have three general patterns: movement driven by local negative interactions with conspecifics or species belonging to the same guild; movement towards conspecifics or congeners; and movement resulting from predator-prey interactions (Kramer & Chapman, 1999; Grüss et al., 2011). Density-dependent effects are well established in general ecology, especially for coral reef fishes which are subject to intense competition for shelter (Hixon & Beets, 1989; Crowder et al., 2000). Empirical observations of density dependent spillovers in the Apo Island no-take reserve in the Philippines (Abesamis & Russ, 2005), revealed that aggressive interactions among adult were 3.7 times greater inside than outside the reserve, with larger individuals usually chasing away smaller ones.

It is worth noting that predator-prey interactions can cause flows in either direction: Armstrong and Skonhoft (2006) note that, when intense fishing of a predator occurs, the density of prey may be higher outside the reserve than inside, inducing a migration of predators outwards from the reserve. However, intense fishing of prey species would lower their concentration outside the reserve and promote inmigration of predators to the reserve.

Many studies of spillover demonstrated increased abundance of targeted fish inside reserves and in adjacent fished areas over time (McClanahan & Kaunda-Arara, 1996; Roberts et al., 2001; Gell & Roberts 2003). McClananan and Mangi (2000) provide evidence that spillover effects were high for fish species such as rabbit fish, emperors and surgeonfish[2] which tend to emigrate from the park to adjacent areas. Lorenzo et al. (2015) conclude that spillover was observed in 80% of the empirical studies that have been analysed. Even the 20% remaining, they argue, may show spillover since they may be subject to some biases or because they have only been established for a short period.

A caveat worth making is that there may be an implicit bias in the literature towards high commercial fish species. However, there has been comparatively little focus on migration of species with low commercial value. Of the 223 marine species mentioned in the meta-analysis by Lorenzo et al. (2015), most had high commercial value; the bulk of the studies placed little emphasis on medium and low value commercial species.

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[2] It is noted that the rabbitfish are the family Siganus, and surgeonfish are in the family Acanthuridae, both fish species are observed in the study site of this these.

Larval survival in the reserves and dispersal to adjacent areas

Botsford et al. (1999) focus on survival rates through the larval stage and the origin of recruits at each location as determinants of the effectiveness of marine reserve in enhancing fish population and catch in adjacent areas. Their study concludes that, should the number of larvae produced in the lifetime of representative individuals be so low that fishing mortality does not affect recruitment, then recruitment is not sensitive to fishing, the addition of reserves, even if fishing is prohibited, will not increase recruitment. The end result is therefore a reduction in catch. On the other hand, when larvae produced per individual is high, and fishing would reduce recruitment; consequently, expanding reserves would increase recruitment and expand catches. Hence, the impact of a new reserve in an overfished area will depend on whether the competent larvae and juveniles will be able to successfully settle and survive at the site.

Using a non-equilibrium dynamic bio-economic model which includes biological factors such as stock-recruitment and weight-fecundity relations, Guenette and Pitcher (1999) concluded that the biomass of spawners and the level of recruitment would always be higher in a reserve regime. This means that a reserve (if big enough) might be beneficial even for fast moving fish. However, while it might cause an increase in spawning biomass, a marine reserve would result in limited fishery yield as fish move across the reserve boundaries. Sladek-Nowlis and Roberts (1999) conclude that catch enhancement will follow with slow moving fish and movement larvae under the assumption that larvae will survive in reserves.

2.3.Design of marine reserves –placement and size

Sumaila et al. (2000) suggest that the effectiveness of any MPA depends on its size and location in relation to the life-history characteristics and habitat requirement of the species to be protected. A multitude of ecological and biological factors may influence the placement and size of an MPA (Gerber et al*.,* 2003). Whilst some have taken for granted that MPAs have positive effects even if randomly placed (Roberts, 2000), this view has been disputed by many scientists (Crowder et al*.,* 2000; Beattie, 2002; Rassweiler et al., 2012).

Placement of marine reserves

The objectives expected of a marine reserve over time should guide its placement. If an MPA's objective is to preserve biodiversity, with limited or no consideration for spillover benefits, then it is better to place it in an area endowed with species which have low propensities to move and/or are not targeted by fishers. Low migration would provide the highest conservation benefits, together with an improvement of the habitats and ecosystem while a non-targeted species would not attract fishers to the reserve's borders to exploit spillovers. Botsford et al. (2003). express this clearly: "The species' movement characteristics should be taken into account together with the objectives."

Smith and Wilen (2003) make a very different point about the siting of reserves whose objective is solely conservation and where spillovers are not expected. They argue that placing a protected area in an inaccessible part, isolated zones, where the adjacent areas involve high transport costs and/or risk, will provide the maximum conservation benefits.

If the objective of the marine reserve is to benefit fisheries in adjacent areas, then spillover is likely to be an important factor. The reserve is more likely to provide fisheries benefits if it is situated in an area which hosts target species and its adjacent areas are accessible to fishers.

Since habitat qualities vary, placing an MPA in an area with high and stable carrying capacity is recommended to maximise abundance (Crowder et al., 2000; Gaines et al., 2010). Schnier (2005) refers to preferred sites as biological 'hot spots' – locations that possesses greater potential to support population growth than surrounding areas.

The spawning behaviours of fish may also influence reserve location. Sites important as locations for spawning aggregations may need protection against commercial fishing. Such sites, where conspecific fishes gather specifically to spawn exhibit densities typically four or more times those found in non-reproductive periods, and are attractive to fishermen. Transient spawners, as opposed to resident spawners, often travel long distances (up to hundreds of kilometres) over days or weeks to reach specific spawning sites outside of their home range. Evidence of the reserve effect in protecting reef fish spawning aggregations can be found in Grüss et al. (2014). Further ecosystem and habitat considerations for the placement of MPA can be drawn from the fact that some coral reef fishes use different habitat types (e.g. mangroves, sea grasses, depths) as nursery grounds before moving to their adult habitat on coral reefs.

For both resident and transient aggregations, the area from which fish are drawn to reproductive sites is referred to as the catchment area and protecting such as areas is often considered necessary to fully protect the reproduction and recruitment of such species.

As a fishery management tool, it is often suggested that overfished areas be set aside as reserved areas to aid in the recovery of target species and overfished areas remain an important factor to guide the placement of MPA (Botsford et al*.,* 2004). Many studies have attempted to analyse the impact of MPAs in aiding the recovery of over-exploited fisheries. Amongst supporters of MPAs, the main conclusion is that the commercial case for reserves is strongest after fishing effort has risen so high as to reduce recruitment (Holland & Brazee, 1996; Sladek-Nowlis & Roberts, 1999; Gerber et al*.,* 2003). However, this conclusion has two weaknesses; first, the effect of redistribution of effort is commonly ignored, and second, protecting an overfished area need not lead to a recovery if the area does not contain self-persistent fish species.

Akpalu and Vondolia (2012) argue that for migratory stocks, a reserve can be effective if it is located offshore when the latter is a source. In this case, the offshore region, being a source, may serve to improve the population inshore (i.e., sink). In many coastal region, given the high concentration of artisanal fishers in the lagoon, its fisheries are often over exploited. Creating a reserve in nearby ‘off lagoon’ waters may shift the few fishers that fish there in to the lagoon, increasing the fishing pressure. In summary, the dispersal of juvenile and adult fish must be sufficient to populate the lagoon area if a reserve is to generate fisheries benefits.

Optimal size of marine reserves

The size of marine reserves has been a subject of inquiry by many scholars. Halpern’s (1993) review study showed that the proportional effect of reserves was independent of reserve size, and that a small reserve is as likely to double biomass per unit area as a large reserve. However, marine reserve size influences the rate of export of adults and juveniles (‘spillover’) to fished areas (Botsford et al*.,* 2003; Green et al*.,* 2015). Accordingly, small size of the area is ineffective with highly mobile species since dispersal rates are high (Kramer & Chapman, 1999; Botsford et al*.,* 2003; Gaines et al., 2010; Green *et al.,* 2015). In this case, Botsford et al. (2003) propose that larger fractions of coastline are required for species with greater dispersal. From a commercial fisheries perspective, much larger size is needed if the spillovers are fully exploited by fishers outside of reserves (Botsford et al*.,* 2001). The reasoning is that the size should ensure the persistence of species- if it is small and fishers are exploiting a proportion of the population in the border, recruitment into resident fish populations will be insufficient for self-sustainability.

It is self-evident that the less mobile a species is, the more it will benefit from a protected area. Mobile species that pass through a reserve as fugitive stocks, only enjoy protection for that portion of their lives spent in MPAs, and even this can be reduced if fishing effort is concentrated on the perimeters of the reserves. The conservation benefits provided by small protected areas are therefore smaller for highly mobile fish species. Marine reserve be large enough to accommodate fish species with high movement rates would be difficult to establish; however, pelagic or migratory fish species such as Pacific sardine or Pacific hake, may be partially (temporally) protected by reserves. According to Parrish (1999), their densities within the reserves should not differ greatly from those outside the reserves.

It is a commonplace that reserves can be used as supports of biodiversity. Akpalu and Bitew (2014) develop this argument and investigate its implications for desired reserve size. They suggest that the reserve should be expanded, and fishing pressure reduced if there is evidence that species diversity in adjoining fishing grounds is declining.

Home range plays an important role in the design of reserves. Whatever a reserve’s size, juveniles and adults with greater movement propensities (larger home ranges) are expected to spend relatively more time outside of the MPA, where they are vulnerable to fishing. Such fishing mortality reduces spawning biomass inside the reserves (and lowers lifetime egg production), it lowers expected recruitment in the MPA and, for small reserves, may mean a less sustainable population. This implies that the bigger the home range of a fish species, the larger the optimal size of the MPA it requires. In consequence, a small reserve's benefits accrue unevenly, favouring the species with smaller ranges.

With density dependent spillover, or when a species' home range is large, export of adults and juveniles may be high. Excessive spillover may reduce fish density and biomass inside the reserve, leading to a negative effect which could eventually decrease fisheries yield. Developing this theme, Halpern (2003) observed that, while a small reserve area may enhance biomass, size, density and diversity of organisms as effectively as a large one, small reserves are unlikely to export significantly into harvest areas. He advocated larger reserves (e.g. 10–20 km in diameter) because they serve an export function and enhance population persistence by protecting larger populations of more species. Using a different approach, Guenette and Pitcher (1999) similarly argued that larger reserves should be preferred as providing more robust recruitment and biomass of spawners.

A different view was taken by Hasting and Botsford (2003) who contended that the largest yield was obtained when reserves were as small as practically possible—thus maximizing the export of larvae from them. They recommended smaller reserves (0.5–1 km across) for fisheries management. The contention is that these protect some species and allow for the export of adults and larvae to fished areas, leading to direct benefits to fishers and potential increases in levels of recruitment (e.g. Alcala and Russ, 2006; Jones et al., 2007; Harrison et al., 2012). Green et al (2015), however, warns that, to be effective, marine reserves should include key habitats utilised by the species

Larval dispersal also has implications for marine reserve size (Green et al., 2015). For instance, Botsford et al. (2001) recommended that, for a reserve population to be self-sustaining, the radius of the reserve must be larger than the mean larval dispersal distance (at least twice the size) of the species to be protected. Since the best available empirical evidence indicates that coral reef fish larvae tend to settle on average 5–15 km from their parents, this suggests that reserves more than 10–30 km across are needed for self-sustainability of such species. While smaller reserves are more likely to be sustained by connectivity with other populations rather than by self-seeding, the available empirical evidence also shows that self-recruitment at more limited spatial scales (<1 km) is common, indicating that a certain degree of larval retention usually occurs and that some larvae have limited dispersal. Thus, smaller reserves may still provide recruitment benefits within and close to their boundaries.

Directional movement of ocean currents also play a role on the size of MPAs (Crowder et al., 2000). Small MPAs can be vulnerable to strong currents where the stock can be driven toward the boundary - where fishing mortality is high (Bourguima & Hellal, 2017). When the directional movement is higher than random movement, larger marine reserves become necessary.

Reserve size also matters when the aim is to manage uncertainty and prevent shocks to the system. According to Grafton et al. (2005), a reserve designed to buffer negative shocks to the fishery should maximize the spillovers of larvae, juveniles or adults from reserves to harvested areas. Using a theoretical model, Doyen and Bene (2003) related uncertainty in harvest rates to a reserve's ability to ensure stock sustainability. Their model suggests the higher the uncertainty, the larger the protected area must be. A similar conclusion was presented by Grafton et al. (2006) who related reserve size to ecosystem shocks. They concluded that, here the aim is to protect a fishery against such shocks, the reserve size should increase with the magnitude and frequency of negative shocks, and when the shocks are substantially felt by the harvested population.

2.4.Emerging economic issues facing marine reserves

While biological and ecological factors such as larval dispersal, adult and juvenile migration and movement of fish, initially guided the design of marine reserves, in the last few decades a growing number of studies have focused on human activities and behaviours as parameters guiding MPA design.

Economic and biological goals may diverge regarding the creation of a marine reserve (Agardy, 2000; Armstrong, 2007). For instance, conservationists with purely ecological objectives might propose large reserves to be placed in each biogeographic region, so as to give a representative sample of local and regional biodiversity (Halpern and Warner, 2003). However, fishermen might, in consequence, face increased travel distances to reach fishable grounds. If arguing for an MPA they are likely to prefer small reserves at sites that allow high transfer rates of fish.

Given the differing objectives and constraints of various interest groups, and that the costs and benefits of MPAs are not shared evenly, the avoidance of conflict is also one of the criteria to be considered for the configuration of marine reserves (Agardy, 2000). Sumaila and Armstrong (2006) make the point that if, when a reserve is established, there is a cooperative interaction, it should benefit all parties involved with the fishery, while a non-cooperative interaction creates winners and losers. Evidence of a non-cooperative situation can be found for the Hout Bay lobster sanctuary, Karbonkelberg Sanctuary and Olifants River Estuary in Cape Town, South Africa (Sowman et al., 2017). Accordingly intense conflict took place between the state having ecological objectives and the local fishing community over. The main cause was that the human dimension, referring mainly to small-scale fishers, were not integrated in the policy design of the marine reserves. The conflict was also well demonstrated by Weigul et al (2008) who showed how a reserve at site which protected the leopard groupers at Carmen Island was at the expense of local commercial fishers whilst SCUBA divers and tour operators benefitted from rising abundance and fish size. As a solution, Fraschetti et al. (2009) suggest selecting MPA sites which minimise conflict, and maximize the likelihood of local support. If the interest of stakeholders is to be taken into account, then some form of multi-criteria decision analysis, with due recognition of human factors may be necessary for the site selection and size of an MPA.

A comparison of the net benefits provided by reserves with those yielded by traditional fishery management tools can also assist in the design of MPAs. Are better options available at lower cost? Hannesson (1998) contends that the conservation effect of an appropriately sized reserve is the same as that achieved with optimal quota regulation, but that it yields a smaller catch. Botsford et al. (2003) argue the effect of reserves on yield per recruit is similar to increasing the age of first capture, and the effect of reserves on yield is similar to reducing fishing effort.

A comparison between traditional management tools and marine reserve is not straight forward and in some cases, they are not perfect substitutes. For some species, reserves will tend to work well while for others such as pelagics, they might not provide expected outcome. Similarly, quotas and allocation of rights may not be effective in managing multispecies fisheries. While the high costs of managing fishing enhance the attractiveness of marine reserve as shown by Armstrong and Reithe (2001), it is important to highlight that a marine reserve also involves enforcement and policing costs due to the risk of illegal activities inside the marine reserves. According to Hallwood (2004), the costs of policing may reduce its attractiveness, especially if the cost of catching the last illegal users is high. Thus, even if a site is biologically suitable to establish a reserve and there are important fisheries benefits through spillovers in the adjacent, the net benefits accounting for policing costs may be low. Therefore, enforcement and administrative costs are also factors that influence the design of MPA

The design of marine reserves with an economic perspective

Economic factors are particularly important if an MPA is expected to bring about both conservation and fisheries benefits. Sanchirico and Wilen (2002) were early amongst those suggesting that MPAs could be beneficial not only from an ecological but also from an economic point of view. Subsequently a number of studies have attempted to find conditions where both the biological and economic situations are improved (see for example Ami et al*.,* 2007; Armstrong, 2007). This section therefore highlights how economic factors should inform the design of MPAs.

The establishment of a marine reserve in an area which was initially an open access site, reduces the space available for fishing. If fishing effort had previously been uniformly distributed, a new MPA would concentrate effort in a smaller area and, *ceteris paribus*, would reduce yield, catch rates and welfare of fishers (Halpern & Warner, 2003). Evidence can be found in Whitmarsh et al. (2002) who observed that the effects of a trawl ban in the Gulf of Catellammare, NW Sicily led to a recovery of stock but did not improved the financial position of artisanal fishers operating in the adjacent area.

At the same time, spillovers from adult and juvenile migration, and export of larvae from the MPA should help to replenish the fish population in the fishing area. The spillover effects of the MPA are important to offset the fall in catch rates and to determine the magnitude of the fisheries benefits. Since adult spillover is common for species that respond positively to reserve protection, though at relatively small scales, (Halpern et al., 2009), an MPA can generate a double-dividend – increasing both the fishable biomass and the economic profits of the fishers (Sanchirico & Wilen, 2002). Such economic benefits depend on complex interactions between ecological processes, biological characteristic and economic factors (Sanchirico & Wilen, 2002; Ami et al., 2005; Botsford et al., 2009). Searching for such win-win situations and identifying the conditions prevail in them can eventually inform the design of an MPA (Botsford et al., 2004). This is at the heart of a pre-establishment evaluation that included an economic process.

The question is whether this replenishment effect more than counters the effects of declines in the fishing area (Attwood & Bennett, 1994; Russ & Alcala, 1996; Sanchirico & Wilen, 2001). This condition is demonstrated by Sanchirico et al. (2006) who show that a reserve is an economically viable solution when the value derived from spillover from reserve outweighs the value of fishing which is lost after its creation. The former represents the benefits of a reserve in terms of its effects on catch in the post reserve period. Smith et al. (2010) suggest that the costs should also include the financial costs of establishing the reserve.

Several factors condition the implicit cost-benefit analysis in achieving an optimality solution; these include the settlement success of the dispersing organisms, the costs of the fishing, the economic and ecological heterogeneity of the system, the discount rate, and growth characteristics of the population (Sanchirico et al.,2006). One straightforward outcome of this approach is a tendency for low-value fishing sites to be those set aside for reserves because they impose lower opportunity costs (Sanchirico, 2004).

Simulation studies by Bensenane et al. (2013) suggest that since a marine reserve always leads to an increase in total fishable biomass, an optimal size of a marine reserve can be selected to maximise the catch at equilibrium. This conclusion also maximises the fisheries benefits of a reserve if the latter is tied to its impacts on catch (Holland &and Brazee, 1996). Choosing a reserve size to maximise fisheries benefits would implicitly take into account biological factors relating to density dependent spillovers.

The implicit costs and benefits from the creation of a reserve accrue over a period of time. This led Ami et al. (2005) to further refine Sanchirico et al. (2006)'s optimality condition by expressing it in present value terms: in a perfect world, a well-designed MPA should maximise the sum of the discounted net revenues derived from the exploitation of the resource in the fishing areas in order to offset the costs associated with the establishment reserve (financial costs as well as welfare loss to fishermen). It is also suggested that the site and size to be selected as an MPA should allow for maximum spillover over the period the reserve is evaluated. However, this is not as straightforward as it appears. Support that a reserve of particular size is necessary to increase the fish population and generate sufficient spillover to raise the catch. Since the reserve also represents a lost fishing ground, as the size become larger, fishing activity is further squeezed, and each unit of fishing ground foregone represents a relatively higher opportunity cost. Consequently even if spillovers (benefits) are maximised, the future benefits may scarcely offset the opportunity costs of the foregone fishing ground beyond some point.

Conversely, a smaller size reserve leads to lower opportunity cost to fishers but it may be ineffective to achieve the required conservation benefits and spillovers. This common compromise between the objectives of yield maximization and marine biodiversity protection also influences the design of the marine reserve in relation to the optimal size (large versus small) if one wants to maximise its value (Bensenane et al., 2013) .

Since the costs and benefits arise in different periods, the discount rates to compare short-term and long-term cost benefit matter (Armstrong, 2006). Holland and Brazee (1996) argue that high discount rates reduce the value of a reserve because they increase the weight given to early losses and decrease the weighting of future gains.

To establish the optimal reserve size in an area, i.e. the set-aside area that would maximise conservation benefits and total harvest value, one also has to consider fishing costs and fish prices (Holland & Brazee, 1996). The costs relate mainly to the effect of a reserve on the distances fishing vessels have to travel: as marine reserves increase in size, so may the distances travelled to fishing areas, potentially leaving fishers worse off (Hannesson, 1998; Hilborn et al., 2004). Such rises in fuel expenditures reduces the value of the reserve. By the same token, Holland and Brazee (1996) show that since the value of a reserve is tied to its impact on the value of the total catch, the value of the reserve will fall if the demand for fish is price inelastic, i.e. if increased harvests cause a more than proportional fall in price, reducing the total value of fish sold.

An economic inquiry which focuses solely on the fisheries benefits of the reserve to inform its design overlooks the value of the conservation effect. A complete evaluation would need to consider the use and non-value of the improvement of biodiversity. Rudd et al. (2003) suggest that extractive direct use, non-extractive direct use, indirect use (maintenance of ecosystem resilience), and non-values such as option value, bequest and existence values, should all be considered. There is growing interest in this avenue among researchers. Viana et al. (2017) provide empirical evidence through a bioeconomic model which incorporates tourism benefits from the marine reserve. As expected, when tourism value rises, the optimal reserve area also increases. In the case of a small island like Mauritius, a reserve will also enhance the recreational benefits to both locals and tourists, raising its value. Moreover, if the reserve is working to populate surrounding areas, total catch may rise, leading to social benefits to locals. This may be achieved even if fishers in such a situation may not be better off since open access would eliminate any fishing rent from the reserve.

The debate is ongoing: Hilborn et al. (2004) further argue that, if fisheries are already managed through a total allowable catch, adding a marine reserve is likely to lead to a reduction of both catch and total stock abundance in the fishing area. On the contrary, when the fisheries are overexploited, the effectiveness of the marine reserve will depends on the spillovers and fishers' responses. While this suggests that the initial conditions of the fisheries are as important as the post MPA conditions, one important human factor to be considered is the spatial behaviour of fishers.

Spatial behaviour of fishers and marine reserve in the overall fisheries

So far, the fisheries benefit has been discussed with reference to an area divided into an MPA and a fishing ground. It is, however, an overly simplified representation of fishing location choice. Fishers often select one fishing site from many different options, based on a multitude of biophysical and economic characteristics (Sanchirico & Wilen 1999).

Determining the fisheries benefit of an MPA becomes particularly complicated if the establishment of the MPA affects levels of effort in adjacent area as well as other fishing sites in the fisheries (Attwood & Bennett 1995). In this regard, Smith and Wilen (2004) argue that, since a reserve removes fishing opportunities, there will be some redistribution and possibly attrition in response to diminished opportunities.

Mason et al. (2012) pointed out that, with the establishment of a marine reserve, fishing within its boundaries becomes illegal, so fishers that fished those areas must reallocate effort or face the risk of penalties. They add that there may be a displacement of effort as increased congestion in open areas changes the relative catch per unit effort across space. However, fishers may catch their quotas in less time if there is significant spillover of adult fish. Similarly they would be induced to relocate effort to make the most of any catch gradient near the boundaries of the reserve (Mason et al., 2012). The logic of this relocation parallels the marginal relocation of effort proposed in H.S. Gordon’s seminal (1954) article on fisheries economics. Indeed, empirical evidence from McClanahan and Mangi (2000) shows that the spillovers across the borders of a marine reserve induce fishers to ‘fish-the-line’, thereby hampering the evaluation of its effectiveness in terms of biomass export.

Evidence on fishers’ behaviour can also be found in Mason et al. (2012) who stress that non-uniform redistribution of effort after a closure can alter the fisheries benefits of marine reserves. Examining the effects of a marine reserve off the California coast on the level and spatial distribution of commercial fishing effort, they found evidence (albeit weak) that high intensity users of the area closed were more likely to exit the fishery. Their mapping analysis revealed some clustering of effort near the boundaries of the closed area. They concluded that some effort was probably re-directed toward alternative species, and that the desire to fish near the closed area may have been motivated more by considerations of distance from port and associated fuel costs than by any desire to ‘‘fish-the-line’’.

Any analysis of an MPA based solely on biological mechanisms is open to criticism if the assumptions governing behaviour of fishers (an apex predator in the ecosystem) are unduly simplistic. Not only does a new marine reserve have a direct displacement effect, it also has indirect effects via adaptations in fishers' behaviour as Mason et al. (2012) showed.

The economic decision, 'where to fish?' has been a subject of inquiry since the work of Wilen (1976) on entry/exit and Bockstael and Opaluch (1983). Wilen (1976) argued that fishermen respond to variations in the profitability of fisheries. Bockstael and Opaluch (1983) developed this view, concluding that, on average, a representative fisherman reacts positively to expected profits and negatively to risk – a conclusion which is further substantiated by Eales and Wilen's (1986) study of the Pink Shrimp Fishery in Alaska.

Exogenous factors such as the prices received for fish can also affect the location of fishing effort. Dupont (1993) argues that price uncertainties affect fishers' location choices through their implications for the expected mean and variance of seasonal profit wealth. While this is consistent with Holland and Brazee's (1996) view that the benefits of the marine reserve depends also on price elasticity of demand for target species, fishers' spatial behaviour in the post reserve period is likely to influence the supply of fish species spatially, adding a further dimension to the price endogeneity issue.

Dupont (1993) also suggested that a stock effect could influence fishers' locational decisions. For species where CPUE (and therefore costs) are tied to population density, overflows from a reserve will modify the cost structure of fishing, again altering fisher effort which in turn affects the expected benefits of the reserve.

In case of inertia on fishing location choice, and slow adjustment by fishers as observed by Eales and Wilen (1986), the marine reserve is likely to impose a higher cost on fishers in the early stages. The discount rate is likely to play a role in determining the effectiveness of the reserve by influencing the relative weightings of early losses and future benefits.

Fishing location choice is also influenced by fishers' attitude towards risk (Mistiaen & Strand, 2000; Eggert & Tveteras, 2004; Eggert & Lokina, 2007). In relation to the expected yield and associated average catch variability, the extent to which fishers are risk-loving and risk-averse would determine whether fishing sites are complementary and substitute to the MPA. Smith and Wilen (2005) further argue that not only the financial risk that matters, but also physical risk. Risk-loving fishers are more likely than risk-averse ones to venture in sites which are financially and physically risky. This important information is key to predict the spatial relocation of effort in other fishing sites within the fisheries, following the establishment of a marine reserve.

In their discussion of spatial effort redistribution, Smith and Wilen (2004) asserted that economic research challenged the biological science of marine reserves by the very fact that fishing effort is observed to respond to economic incentives over time and space. Whilst this seems intuitively obvious, many factors could influence the spatial allocation of fishing effort, including spatial distribution of the fishery stocks, the differential value of various target species, weather conditions, social factors such as local traditions or agreements among stakeholders and managers, and the location of the MPA with respect to fishing ports.

As a result, models which analyse the impacts of marine reserves when fishing effort is endogenous differ substantially from those where effort is exogenous. The importance of accounting for spatial behaviour in the post reserve period is emphasised by many scholars such as Holland and Brazee (1996), Hannesson (1998), Sanchirico and Wilen (1999, 2001, 2002) and Smith and Wilen (2003). If the decision maker's objective function specifies maximisation of net benefit to fishermen, then the location and size of marine reserve should be determined so that it increases the fisheries benefit after fishers have changed their behaviour, i.e., after displaced efforts, fishing the line, changes in targeted species, etc. Sanchirico and Wilen (2001) point out that, if a location is chosen in a manner that eliminates spatial arbitrage opportunities, context-specific double pay-off conditions which improve aggregate biomass as well as aggregate harvest may exist: the closed area is part of the source in a sink source system, the gain from dispersal exceeds the harvest loss from the reduction of the source area. Studies which integrate fishers' behaviour with biological and ecological factors may offer guiding principles in this respect. For instance, Smith et al.(2009) build spatial-dynamic processes and conclude that for settlement-limited fisheries, the increase of production within reserves may compensate for greater fishing pressure outside the reserve for habitat closures up to 50% of local fishable waters.

Sanchirico and Wilen (1999) go further and suggest that, whether a particular area is a source or sink in equilibrium may depend on economic parameters as well as biological ones. In a fishable area, an increase in the relative costs of fishing implies an increase in the equilibrium density of biomass, changing its characteristic towards a source. Smith and Wilen (2004) found extreme examples of this in their analysis of the California Sea urchin fishery, where some patches play a role as de facto reserves because they are so costly to access.

If biological dispersal linkages are added, dispersal takes place from the high density patch (where low fishing effort is due to high fishing costs) to the low density patch, attracting efforts to the low density patch. Sanchirico and Wilen (1999) concluded that the way patches are linked eventually determines the displacement of efforts.

This combination suggests that, following the establishment of marine reserve, there is an endogenous "economic gradient" the results from incentives to reallocate effort toward zones that provide higher relative rents. There is also a "biological gradient" along which biomass moves to areas of lower density or zones that are natural sinks. These two forces operate together to determine the process of bioeconomic convergence over space and time. Whilst the economic and biological gradients can be complementary, they may also work against each other, depending on the kind of connectivity between patches (closed, sink source or multiple source, etc.). In these terms, the ideal reserve is one situated so as to maximise the overall system biomass and catch through the interaction of these two forces.

Fraschetti et al.(2009) showed how the economic gradient may affect biological conditions. Their study in southern Italy showed how differential habitat quality may have been caused by human activities. Concentration of efforts and intense fishing can change the situation and success of spawning aggregations, and can move resident fish to other sites. The cessation of such fishing activities, and the rehabilitation of the seabed, might see fish moving to their formerly preferred sites. In such cases, reserve location should be based on records of historic rather than present abundance.

2.5.Conclusion

The performance of an MPA is measured against a set of desired outcomes, and the attainment of these outcome is conditioned by many human factors. Among these, fishermen as an important interest group in the fisheries cannot be ignored. No matter what these targets are, they are likely to be influenced by fishermen – the reserve affects them, and they affect it. This is among the main reasons that while there is a consensus on the ecological merits of marine reserves, their benefits for fisheries remain controversial.

Whether or not the inter-relationships between fishers and reserves are mutually beneficial will depend on a multiplicity of factors. Key amongst these is the behaviour of fishers after the establishment of a marine reserve. This determines the extent to which the reserve helps to rebuild the population of fish species within the reserve, the resulting spillover effects, and whether or not these offset any upward shift in fishing intensity in the remaining fishing grounds.

A reserve changes the incentive structure of fishers, redistributing effort non-uniformly. This relocation effort, together with the fishers’ adaptation to the effects of the marine reserves, and their behaviour are key factors in a reserve’s success. The key lesson is that even where the location of an MPA is not simply intended as a way to control fishing effort, (such as MPAs that aim to preserve the recreational value of a site), recognising the likely subsequent induced behaviours of fishers may be important.. The economic factors that incentivize fishers’ behaviour will also be significant in determining the effectiveness of marine reserves, including the costs relating to displacement of fishers’ efforts, their opportunity costs, changes in distance travelled, and differing levels of risk in substitute fishing areas.

Where the MPA is intended to benefit a commercial fishing resource, then economic aspects that induce changes in fisher behaviour should be incorporated into ecological models simply because mankind is the apex predator of these commercial species.

Few studies have been able to compare biological processes and effort distribution between pre- and post-reserve periods. In many cases, data were unavailable, hence spillover effects on adjacent areas have been modelled through the relationship between measures of catch and distance from reserves to areas or other space-static formulations of the research objectives. Gradient assessment within and between reserves are very often limited to an analysis of abundance either through visual census or tag-and-recapture of fish or standardised CPUE as a measure of abundance. Integrating the fishers' spatial behaviour with distance-based studies on adjacent areas can provide important insights on whether the benefits are sustainable across time and space and enhance our understanding of potential changes following the establishment of a marine reserve.

This review suggests that economic factors should complement biological and ecological factors in the design of marine reserves. Fishers' location choice, fish targeting, responses to changes in travel cost, revenue, risk (financial and human) and regulatory practices, expectations formation, adaptive and survival strategies, and attitudes towards risk, are factors to be considered for the configuration of MPAs.

3. Evidence of spillover effects of a marine protected area

Abstract

The study estimates the abundance gradient in water surrounding a marine reserve before using it as evidence of spillovers from a marine protected area. Four indicators of abundance are used – (i) catch per trip, (ii) catch per trap, (iii) number of fish per trip and (iv) number of fish per trap. The gradient is estimated using both a statistical and economic approach. Fish landing data comes from a sample of trap fishers drawn over a 12 month period, all of whom fish the Blue Bay Marine reserve on the west coast of Mauritius, in the Indian Ocean. The Generalised Linear Models (GLM) is used to standardise the catch data by removing the effects of individual fisher' productivity from the recorded measures of abundance. The GLM normal and gamma distribution are estimated for indicators (i) and (ii), respectively and the negative-binomial distribution for (iii) and (iv), respectively. The statistical approach concludes that catch per trip and catch per trap decline for 4km from the MPA boundary. The stochastic frontier analysis confirms evidence of spillover effects, indicating that fishers catch more per unit of effort near the MPA. The study also establishes that the fish of key species caught near the MPA are larger than those of the same species caught more than 4km away from it.

Keywords: marine reserve, spillovers, gradient assessment, standardisation, production function

3.1.Introduction

An argument advanced by proponents of marine reserves is the way they help to increase the abundance of fish, which spills over to adjacent areas (Chapman & Kramer, 1999; Gell & Roberts, 2003; Forcada et al., 2009; Bellier et al., 2013). However, providing evidence is not a trivial task. Although the directional movements of fish from inside reserves to outside may be due to either density dependent or density independent effects (Abesamis & Russ, 2005), increases in fishery yields are typically the consequences of two mechanisms: the emigration of juvenile and adult fish to surrounding non-protected areas and the export of larvae (Roberts et al., 2001; Russ et al., 2004). The relative magnitudes of the recruitment effect and the adult spillover effect have been of great interest among marine scientists in recent years (Halpern, 2003; Hilborn et al., 2004; Stelzenmuller et al., 2009). Since the recruitment effect is difficult to detect, spillovers from MPAs provide a tangible indicator of potential benefits from their establishment. Spillover is typically observed through patterns of abundance or catch that decline with distance from reserve boundaries (Halpern et al., 2009).

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 & 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 it. If emigration determines the distribution of fishes, fish density should be higher in the centre of the reserve and be decreasing gradually toward and beyond the boundaries (Rakitin & Kramer, 1996; Abesamis et al., 2006).

This, and other gradients of biological features, can be obtained by visual census and tagging of fish inside and outside the MPA (Chapman & Kramer, 1999; Abesamis et al., 2006). However, this method may be costly and time consuming. Moreover, according to Chapman & Kramer (1999), the quantification of the spillover effects should control for both the spatial and temporal variation in fish distribution.

More feasibly, such gradients can also be estimated using the catches made by fishers in adjacent areas (Vandeperre et al*.,* 2011). Whilst catch per unit of effort (CPUE) is a poorer indicator of abundance for some species, for others when it is higher near MPAs, it is taken as evidence of spillover (Chapman & 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 common measures of fish abundance ((Gulland et al., 1956; Beverton and Holt, 1957; Kirmura, 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 is commonly justified on both technical and practical grounds. Since the 1950s, catch and effort data have been linked with abundance either as an input in calculations of fish stock (Shaefer, 1957) or as an index of abundance (Bordalo-Machado, 2006). Such industry-dependent data not only offer greater coverage in space and time, but are economically cheaper to collect (Ye & Dennis, 2009).

However, the use of CPUEs as proxies of abundance in order to estimate gradients of abundance may not only be confounded by differences in habitat quality between the two sites, but also by the difficulties in measuring effort. Effort is a non-homogeneous combination of measurable inputs, such as the physical capacity of fishing boats, or the gear in use, with less measurable factors such as the skills of fishers (Pascoe and Robinson, 1998). Higher total catches need not mean that the reserve is effective. A declining gradient of catches could merely indicate that more highly skilled fishermen or larger boats are the ones locating near the marine reserve (Smith et al., 2006).

CPUE is an imperfect measure of abundance for a number of other reasons. It can be influenced by inter-temporal factors (weather, season etc) which can be significant sources of error unless smoothed or averaged over time. More importantly, it is particularly poor measure of abundance for species that shoal, migrate along fixed routes, or form breeding aggregations. There may be very few members of such species left, but a fisher who knows their migration or aggregation patterns can harvest a set quota from the few remaining with no more effort than if they were abundant.

Another issue in translating data on CPUE into an abundance gradient that can be used to test for “spillover” of fish from the boundary of the MPA is the different species’ coefficients of catchability. The latter is the parameter which relates CPUE as an index of relative abundance to the stock of fish (Squires & 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.

When improvements in fishing technology increase catchability, the relationship between CPUE and abundance is clearly affected. However, catchability (and hence CPUE) can also be affected by exogenous factors such as habitat differences over space, and environmental fluctuation over time. 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 CPUE data is a truer reflection of abundance. Such standardisation typically uses the Generalised Linear Models (GLM) or Generalised Additive Models (GAMs).

From an economic point of view, the fishers’ catch can be seen as the output of a production function. In this way, catchability is the proportionality parameter between fishing effort and fishing mortality, or the portion of the stock captured by one unit of effort (Squires & Vestergaard, 2015). By controlling for effects other than those caused by changes in population densities, standardisation allows the catchability coefficient to be treated as if constant, (Squires & Vestergaard, 2015). In this way, if abundance is higher near the reserve, it should be reflected in a CPUE that falls with distance from the reserve.

The two approaches (statistical & economic) both borrow from the methods developed for stock assessment. They both aim at removing the effects of changes in fishing technology from the CPUE – allowing the latter to be used to estimate both the stock and the resource’s productivity. The estimating issues are well elaborated by Squires & Vestergaard (2015).

This study tests for evidence of fish abundance spillovers from an MPA to adjacent areas off the Mauritian coast, using both the statistical approach as well as the economic approach applied to catch data from artisanal trap fishers.

The statistical approach applies the Generalised Linear Model (GLM) using four response variables: catch per trip (in weight), catch per trap (in weight), total individual fish per trip and total individual fish per trap. The first two response variables assume the normal and gamma distribution while the remaining ones, being count data, assume a negative binomial distribution, all with a log-link function. Following Bettaile and Quinn (2004), the GLM is used in this study, not to provide an index of abundance across time and space, but to eliminate other pre-study spatial and temporal effects related to fishers.

The economic approach relies on a *harvest production function*. Commonly a Cobb-Douglas or translog production function is used to examine the determinants of harvests, including technical differences, fisher skill and socio-economic characteristics of fishers. A concern is that the data on catch at a particular site may be affected by technical inefficiencies. Thus, a proper production analysis must account for technical efficiencies, that is, the best practice technology represented by a production frontier. The catch rates when evaluated using the production frontier proxy fish density which can then be used to assess the spillovers of marine reserves. The economic approach in this study explicitly treats the catch per trip as the output of a production function, and estimates both an average production frontier as well as an efficient production frontier by removing the influence of technical inefficiencies.

An attempt is also made to establish the mechanism through which the spillover may occur. The different fish species, as well as their average weights, are further analysed before drawing any conclusion on the effect of the MPA.

3.2.A brief review of evidence on spillovers of marine reserves

If a reserve has been functioning effectively, and areas outside it are fished, the observed abundance of resident fish is likely to decrease from the inside to the outside; the gradient being determined by three major factors: population growth effects, movement (diffusion) of fishes (emigration) and harvesting (Abesamis et al., 2006, Bellier et al., 2013).

Emigration can occur in many ways and for many reasons, however, they are commonly sorted into two sets, density dependent and density independent.

A density dependent gradient occurs when the spillovers are the result of an increase in stock of fish inside the reserve (Rakitin & Kramer, 1996; Abesamis et al., 2006). Frequency-dependent models of animal distribution predict that animals should prefer to move toward areas where density is low relative to available resources as this is beneficial to their fitness (Goni et al., 2006). As population increases fish density in the reserve, there is an increase competition of resources. In consequence of the combination of fishing pressure outside the reserve and population pressure in it, the biomass of relocated fish should decrease further away from the reserve boundary. Russ (2002) points out that spillover can vary from hundreds of metres to a few kilometres from the reserve boundary, depending on the species. Density-dependent cross boundary movements accord even more closely with the notion that the waters surrounding a successful MPA will exhibit a decreasing abundance gradient.

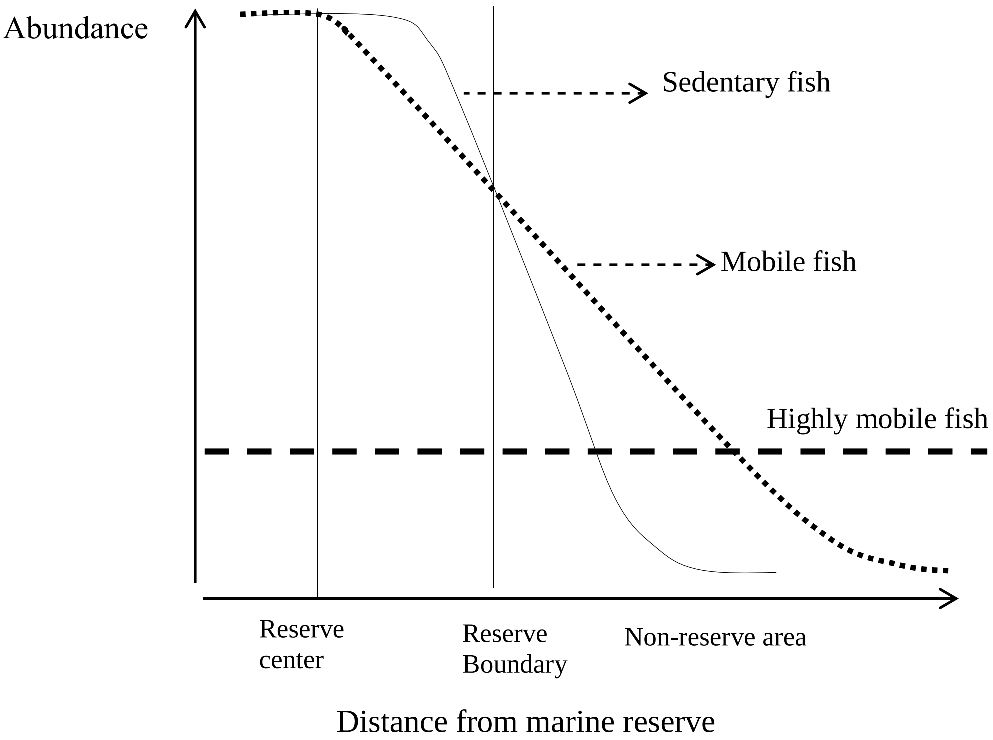
Density-independent drivers of fish movements across the borders of MPAs are not merely random movements of individuals; they may include directed movements over a large home range as well as ontogenic movements across habitats (Rakitin & Kramer, 1996; Forcada et al., 2009).

Each fish species will have a different abundance gradient in the fishable waters surrounding a reserve. The slope of gradients of abundance across reserve boundaries would depend on the fish species and catch rates (Rakitin & Kramer 1996; Kaunda-Arara & Rose, 2004; Halpern et al., 2009; Bellier et al. 2013). Figure 3.1 relates different shapes of the abundance gradients to species' characteristics. The stylised shapes have been adapted from Rakitin and Framer (1996), Chapman and Kramer (1999), Halpern et al (2009) and Bellier et al (2013).

Coral reef fish are generally described as sedentary once they settle on a reef after a pelagic larval stage (Rakitin & Kramer, 1996). The gradient for sedentary fish is hypothesised to be linear as shown in figure 3.1. Sedentary organisms are those whose movements are short-range when compared to the spatial scale of the fishing process and/or pelagic larval dispersal (Hilborn et al., 2004).

Fish whose home ranges have centres close to the boundary are more likely to relocate outside the reserve but near its boundary. This relocation should decrease farther from the reserve boundary due to their low mobility and fishing mortality. Hence, mobile fish should exhibit a shallower gradient of abundance across the reserve boundaries than sedentary fish (Rakitin & Kramer 1996), while the gradients are expected to be sharper for low-mobility species (Harmelin-Vivian, 2008).

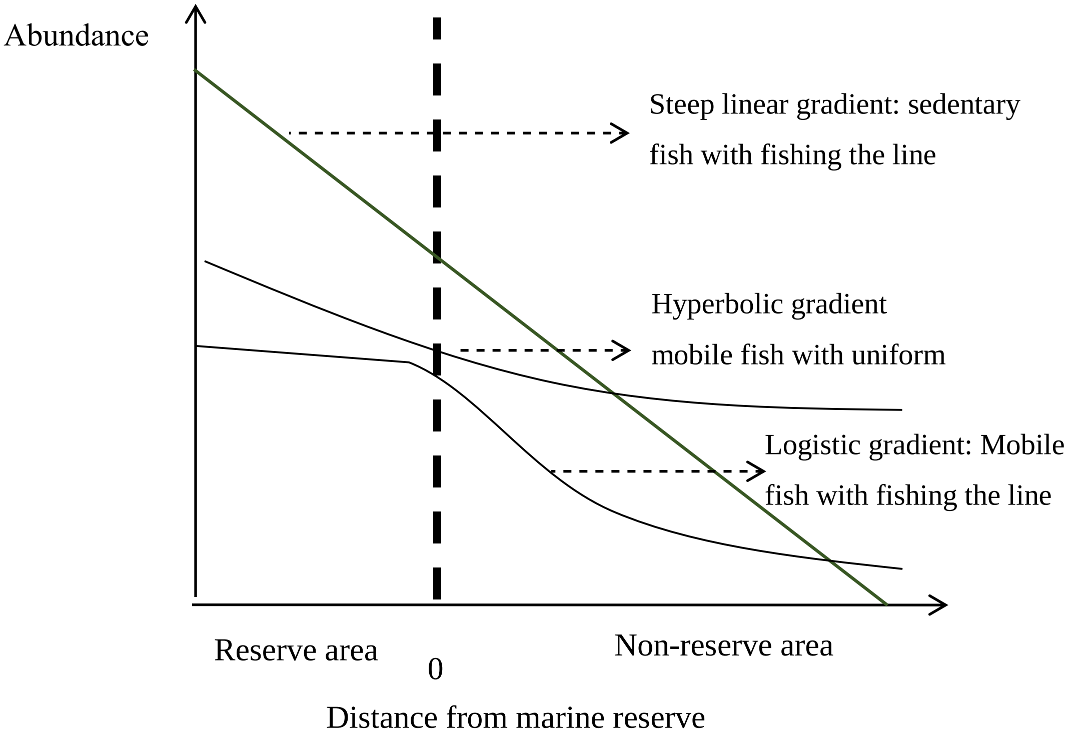
Figure 3.1: Gradients of spillovers with fish species



Highly mobile fish could easily relocate from areas close to the centre of a reserve to areas far outside the reserve. In this respect, fish that are highly mobile and/or not vulnerable to fishing should not be directly affected by the reserve, unless other factors such as trophic and competitive relationships are considered.

If equal mobility is assumed, gradients of abundance between the reserve and the fishery areas should be steeper for highly catchable fish and those which are more vulnerable to fishing gears than for moderately catchable fish. The gradient is therefore affected by the fishing intensity of any species. Figure 3.2 shows the difference in spillovers between fishing the line and uniform fishing.

Figure 3.2: Gradient of spillovers with mobile fish and fishing intensity



Source: Adapted from Chapman and Kramer (1999), Halpern et al. (2009), Bellier et al. (2013)

For modelling purposes, Halpern et al. (2009) provide three different mathematical forms of gradient shape: linear decay, exponential decay and logistic decay. Mathematically, the three types can be modelled as follows.

$$X_i = \beta_0 - \beta_{DIS} DIS_{im} \quad \text{(Linear gradient)} \tag{3.1}$$

$$X_i = \exp(-\beta_{DIS} DIS_{im}) \quad \text{(Exponential gradient)} \tag{3.2}$$

$$X_i = \frac{\beta_0}{1 + \exp(-\beta_{DIS} DIS_{im})} \quad \text{(Logistic gradient)} \tag{3.3}$$

X_i is a measure of abundance at location i and DIS_{im} is distance from location i to the location of the marine reserve m.

In their quest to quantify spillover effects from marine reserves, fishery scientists, have used a range of assessment methods. One is in-*situ* estimations using visual censuses. This was used by Rakitin & Kramer (1996) in Barbados, Roberts et al. (2001) in Florida (US) and St Lucia, and Harmelin-Vivian et al (2008) in six Mediterranean MPAs. Harmelin – Vivian et al (2008) found a fish spillover mean distance of 0.5km to 0.6km from the six MPAs studied, ranging from a minimum of 0.11km to a maximum of 2km. Using visual census, Abesamis et al. (2006) express fish density of targeted fish and non-targeted fish as number of fish observed per 500m^2 and 250m^2 respectively. They observe gradients of decreasing abundance for sedentary and vagile targeted fish across the northern boundary of Apo Reserve in the Philippines, which may have been due to spillover.

Tagging of adult fish can be used to check for movement from a reserve to adjoining fishable areas (Holland et al.,1993; McClanahan and Mangi, 2000; Roberts et al., 2001; Abesamis et al., 2006). Goni et al (2010) use tag-recapture data for lobster from the Columbretes Islands Marine Reserve in Barbados and estimate around 7% of protected lobster emigrated annually to the adjacent fished grounds. Follesa et al. (2011) using tag-recapture data observe movement of fish species *Palinurus elephas* up to 50km from the centre of the reserve In research with specific relevance to this study, Hardman et al. (2010) analyse movement of *Naso unicornis* in and around a marine reserve of the island of Rodrigues, East of Mauritius in the western Indian Ocean. They track the tagged fish 21 times to quantify the home range, the fish's home rage being a convex polygon with radius 56.4m to 295.3m.

Visual censuses and tagging of fish require considerable logistics arrangements (Gell and Roberts, 2003; Abesamis et al., 2006). Therefore evidence of spillover comes mainly from catch data from an experimental survey or actual fish landing data from fishers that shows increasing captures in fisheries adjacent to MPAs (Bertrand et al., 2008; Forcada et al., 2009).

Standardisation does not arise when catch data comes from an experimental design. For instance, Rakitin and Kramer (1996) collected catch from traps places on the sandy bottom on the seaward edge of the reef. In the design, the traps followed same specification, hence, the fishing effects would be constant during the survey. The study observed gradual decrease of coral reef fish with increasing distance from the centre of the Reserve and they concluded that emigration of fish was limited due to discontinuity in habitat quality. Similarly, fish trap catches were collected by Kaunda-Arara and Rose (2004) across the boundaries of two Kenyan MPAs. They found that abundance of most species were higher (by up to an order of magnitude) inside both parks than outside them. Spillovers were observed for a few sedentary species (e.g., *seagrass parrotfish, Leptoscarus vaigensis, whitespotted rabbitfish, Siganus sutor*). In turn, data of biomass per trap net was collected specifically for the purpose of the study by Forcada et al. (2009) who observed that even if spillover was localised due to habitat quality, it was sufficient to provide benefit to artisanal fisheries. The experimental design adopted by Forcada et al. (2009) in their assessment of three Mediterranean Marine Reserves in Spain and France, namely Tabarca Marine Reserve, Carry-le-Rouet Marine Reserve and Cerbere-Banyuls Marine reserve, also observed that, for some species, catches were significantly higher in fishable waters near the borders of the MPA.

Many studies have attempted to find evidence of spillover effects from fish catch and catch per unit of effort (CPUE) of artisanal and commercial fishers but such measure was rarely standardised. Roberts et al (2001) collected data from trap-fishing methods to construct a measure of CPUE and observed that catch from both large traps and small traps increased after the establishment of a reserve in the Soufriere Marine Management Area in St. Lucia. The fish species being sedentary and higher weight of the average individual fish explained the rising total landings. Russ et al. (2003) uses raw hook-and-line catch from fishers in the adjacent of the marine reserve from Apo Island, the Philippines, to demonstrate spillover. Follesa et al. (2011) also analysed raw CPUE from commercial survey in the adjacent areas of the Su Pallosu marine reserve in the Western Mediterranean to show the negative gradient in lobster CPUE with distance from the MPA. It is important to note that, since no standardisation method was applied, the catch data in this study was influenced by the characteristics of the fishers.

In contrast, the study of Goni et al. (2006) collected catch and effort data onboard commercial fishing boats in Columbretes Islands Marine Reserve (CIMR, Western

Mediterranean) to construct the catch per unit of area. The latter was analysed using the Generalised Additive Model which controlled for the effect of depth and confirmed a density gradient caused by lobsters emigrating from the reserve. Attempt to control for the effect of fishers can also be found in Bellier et al's (2013) who carried a study of the Carry-le-Rouet MPA in France using both visual census data and fish landings by artisanal fishers. An exponential function was used which controlled for the different types of fishing technologies such as trammel nets, gillnets and combined nets. They observed that landings decreased as distance from the MPA rose, although the visual census showed a slight increase for distances above 900m.

A more comprehensive standardisation approach, similar to the one in this study, can be observed in Stelzenmuller et al. (2009). The Generalised Linear Model (GLM) was used to standardise biomass per boat with explanatory variables such as fishing gear, year, month, quarter, and type of bottom (sand, mud, gravel). The study concluded that there were no discontinuities in habitats of the Medes Island Marine Reserve (northwestern Mediterranean) outwards and that biomass per boat decreases with increasing distance from the border of the MPA.

Each of these studies used data from fishers. They therefore had to deal with a simple reality, that CPUE is an imperfect proxy for abundance or for the health of a resource. Still, the raw data from CPUE was used for the analysis. In the next section, catch per trip and per trap are used to measure spillovers and several modelling approaches are adopted in order for these two measures to reflect abundance. Nevertheless, other challenges such as movement of shoaling species, migratory species, characteristics of the sea bed, among others, remain. These are standard problems that have to be addressed before the costs and benefits of a proposed (or existing) reserve can be assessed.

3.3.Modelling approaches

Any analysis using catch rates to test for decreasing abundance with distance from a MPA must account for factors such as fishers' characteristics, fishing technology, and seasonal variations, which can affect it. This section identifies two methods of controlling for these influences: the statistical approach and the economic approach. Both use CPUE as the response variable and hypothesise a declining gradient of fish abundance after controlling for specific effects.

The primary assumption underpinning all of these approaches is that a fisher's catch is proportional to the abundance of fish as shown by equation (3.4) (Campbell, 2004; Maunder & Punt, 2004):

$$C_{ji} = q_{ji} E_{ji} X_i \tag{3.4}$$

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 is:

$$\mathrm{CPUE}_{\mathrm{ji}} = \frac{C_{ji}}{E_{ji}} = q_{ji} X_i \tag{3.5}$$

Changes in CPUE 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, and there are two fundamentally different ways of accounting for such 'other differences': the statistical approach and the economic approach.

The statistical approach

In order to estimate stock abundance, statisticians standardise the CPUE by adding additional structure through the catchability coefficient (Maunder, 2001). Such standardisation allows the comparison of catch rates across space and time. The variables forming the additional structure can be continuous (e.g. sea-surface temperature, price of fish, vessel size, etc.) or categorical (e.g. capture area or gear type). The catchability coefficient for all combination of categorical-continuous variables is calculated relative to the bases. The formulation by Maunder (2001) illustrates this clearly. Let $p_A, p_B, .., p_N$ represent the categorical variables for attributes $A, B, ..., N$:

$$q_{A=a,B=b,\ldots,N=n} = qp_a p_b \ldots p_n \tag{3.6}$$

for $a,b,\ldots.,n = 0,1$

Let the p 's corresponding to the base equal one such that

$$p_{A=1} = p_{B=1} = \ldots p_{N=1} = 1 \tag{3.7}$$

then, all the other categorical variables can be estimated as parameters in the model.

Any continuous variable is modelled by including a parameter β to scale the relationship between catchability and the continuous variable. This relationship can be modelled as a multiplicative using the exponential form. Let the value of a continuous variable, k, be represented by I_{ki} in area i. Then:

$$q_{ji} = q\exp\left(\sum_{k=1}^{K} \beta_k I_{ki}\right) \tag{3.8}$$

The catchability coefficient with both categorical and continuous variables is then:

$$q_{ji} = qp_{a_i} p_{b_i} \ldots p_{n_i} \exp\left(\sum_{k=1}^{K} \beta_k I_{ki}\right) \tag{3.9}$$

Adding a log-normal error term to 3.9, and with equation (3.5),

$$\frac{C_{ji}}{E_{ji}} = q_{ji} X_i \exp^{\varepsilon_{ji}} = X_i qp_{a_i} p_{b_i} \ldots p_{n_i} \exp\left(\sum_{k=1}^{K} \beta_k I_{ki}\right) \exp^{\varepsilon_{ji}} \tag{3.10}$$

Equation (3.10) can be estimated using the GLMs defined by the statistical distribution of the response variable and by the way in which some linear combination of explanatory variables relates to the expected value of the response variable (Maunder & Punt, 2004).

In order to apply the GLM, it is necessary to (a) choose a stochastic response variable y_{ji} or z_{ji} for $z_{ji} = \ln(y_{ji})$, (b) select a sampling distribution for the stochastic response variable from the exponential family (for example, normal, exponential, Poisson, binomial, gamma), (c) chose a *link function* appropriate to the distribution and (d) select as a single linear predictor, η, a set of variables that are explanatory to the distribution or stimulus (Venables & Dichmont, 2004):

$$\eta = qp_{a_1} p_{b_i} \ldots p_{n_i} \exp\left(\sum_{k=1}^{K} \beta_k I_{ki} \right) \qquad (3.11)$$

The mean of z_{ji} is related to η by a known function called the *link function*, that is,

$$E(z_{ji}) = \ell^{-1}(\eta), \eta = \ell(\mu) \qquad (3.12)$$

Natural canonical link functions exist for each of the distributions in the exponential family. For the normal distribution, the natural link is the identity link, $\eta = \mu$, the variance function is the constant (equal to one) and the scale parameter is the variance.

The choice of a statistical distribution for the response variable should take account of the nature of the process that generated the data. If the catch is recorded in numbers of individual fish caught, a discrete distribution, such as the Poisson or the negative binomial may be the most appropriate (Maunder & Punt, 2004). If the catch is in weight, catch rate is modelled using a continuous distribution such as the log-normal, gamma and inverse Gaussian (Dick, 2004).

The linear equation can be expressed as follows:

$$E(z_{ji}) = q + p_{a_i} + p_{b_i} + \ldots + p_{N_i} + \sum_{k=1}^{K} \beta_k I_{ki} + \varepsilon_{ji} \qquad (3.13)$$

Modelling distance-density relationship

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. This conceptualisation is similar to Goni et al.

(2006) and Stelzenmuller et al. (2009). The exponential relationship is selected for the analysis and is represented as follows:

$$\eta = qp_{a_i} p_{b_i} \ldots p_{n_i} \exp\left(\sum_{k=1}^{K} \beta_k I_{ki} \right) \exp(\beta_{DIS} DIS_{im}) \exp^{\varepsilon_{ji}} \tag{3.14}$$

Operationalising the estimation equation

Model 1. Stochastic response variable-Total Catch per Trip $(TCPT)$ and Catch per Basket Trap $(CPBT)$

The first model uses Total Catch per Trip $(TCPT)$ and Catch per Basket Trap $(CPBT)$ as the stochastic responses. Since the focus is laid upon the artisanal fishers using basket traps, the response variables included the number of baskets, $NBAS$, and of the size of baskets, $SBAS$. Seasonal effect on catches may be significant and, following the literature, quarterly effects were included through a categorical variables representing the four quarters of the year. The predictive η' is specified as follows:

$$\begin{gathered} \eta' = q' + quarter + NBAS + SBAS + (NBAS \times SBAS) + (quarter \times NBAS) + (quarter \times SBAS) \\ + DIS_{im} + DIS_{im}{}^{2} + \varepsilon rror \end{gathered} \tag{3.15}$$

For each of the response variable, two GLM models are implemented (see Marchal et al., 2006) for further details on the statistical properties). The response variables, $\ln(TCPT)$ and $\ln(CPBT)$ are assumed to follow both a normal distribution (i.e. the link function is identity) and a gamma distribution (i.e. a logarithmic link function).

Model 2: Stochastic response variable: Total Individual Fish per Trip $(TIFT)$ and Total Individual Fish per Basket Trap $(TIFB)$.

The second set of response variables are the total number of fish per trip $(TIFT)$ and total number of fish per basket trap $(TIFB)$. The later cannot be modelled directly by a normal distribution since these data are discrete and positively skewed (Bellier et al., 2013). In this case, the response variable is modelled using either the Poisson process or a negative binomial. The log-linear specification is commonly used in count data models to ensure that the conditional expectation is positive (Hausman et al., 1994; Delgado & Kniesner, 1997). With equi-dispersion the model reduces to the Poisson model and the dispersion parameter goes to zero. When it is greater than zero, the negative binomial model represents count data with over-dispersion.

Model selection

The goodness of fit is 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 (Marchal et al., 2006).

The economic approach

Catch rate and its relationship with inputs form the basis of the neoclassical production analysis of fishing (Anderson, 1976; Padilla & Trinidad, 1995). Economic production functions have long been applied to the fisheries sector, using a measure of catch per unit effort. The fixed factor, capital, can be proxied by variables such as vessel length, tonnage, engine size and gear type, while the variable inputs can include the number of fishers or fishing time (Hannesson, 1993; Padilla & Trinidad, 1995). The production process is commonly represented using a Cobb-Douglas or a translog production function. In equation 3.16, effort is assumed to be determined by a Cobb-Douglas production function:

$$E_{ji} = A(NBAS_{ji})^{\alpha_1}(SBAS_{ji})^{\alpha_2} \exp \xi_{ji} \tag{3.16}$$

α_1 and α_2 are the output elasticity with respect to $NBAS_{ji}$ and $SBAS_{ji}$ respectively, and ξ_{ji} is the error term assumed to follow a normal distribution.

The catch equation from 3.2 is modelled as a function of the inputs as well as the stock of fish as follows:

$$C_{ji} = f(X_i, NBAS_{ji}, SBAS_{ji} : \beta')\exp\varepsilon'_{ji} \tag{3.17}$$

Where X_i is the stock of fish in region i and β' are the coefficients. The error term ε'_{ji} in equation captures not only random factors associated with effort, but also the fishing conditions in the area.

Operationalising the model

Assuming that the resource (X_i) remains constant, but spatially distributed, the production function becomes weakly separable. In logarithmic form, equation 3.16 is written as follows:

$$\ln E_{ji} = \ln A + \alpha_1 \ln(NBAS_{ji}) + \alpha_2 \ln(SBAS_{ji}) + \xi_{ji} \tag{3.18}$$

Equation 3.18 is introduced in the logarithmic catch equation. Consequently, the reduced form equation ends with $\ln X_i$. To introduce the stock effect from the distance from the MPA, equation 3.2 (section 3.2) is used $(X_i = \exp(-\beta_{DIS} DIS_{im}))$ which is equivalent to $\ln X_i = -\beta_{DIS} DIS_{im}$. In order to estimate the extent of the spillover effect with cut-off point, the square term is added such that $\ln X_i = \beta_{10} DIS_{im} + \beta_{11} DIS_{im}{}^2$. β_{10} is expected to be negative while β_{11} can be negative, positive or zero. The relevant statistical test can be eventually be applied.

$$\begin{aligned}\ln(C_{ji}) = \beta_0 + \beta'_1 \ln(NBAS_{ji}) + \beta'_2 \ln(SBAS_{ji}) \\ + \beta'_3 (\ln(NBAS_{ji}))^2 + \beta'_4 (\ln(SBAS_{ji}))^2 \\ + \beta'_5 \ln(NBAS_{ji}) \ln(SBAS_{ji}) \\ + \sum_{n=2}^{n=4} \beta_{4+n} QU_n + \beta_{10} DIS_{im} + \beta_{11} DIS_{im}^2 + \varepsilon'_{ji}\end{aligned} \tag{3.19}$$

Equation 3.19 also accommodates the seasonal effects through the quarterly dummy variables QU_n.

Equation (3.19) can be estimated using Ordinary Least Squares (OLS), or using frontier analysis. Frontier analysis attempts to compare production units with the 'best practice' efficiency frontier that could be formed by an efficient production unit within the industry. The best practice frontier is estimated using data from a sample of observed units. The fishers on the efficient frontier are identified as those who are not outperformed by any others who have similar characteristics. The estimated efficiency frontier then provides the reference points needed to assess the efficiency levels of firms.

The estimation of equation (3.19) requires two steps: first, an appropriate production function is estimated using ordinary least squares. The second step separates the error terms into two components, usually a two-sided random error component and a one-sided inefficiency component. In this case, the ε' is a composite error term specified as follows:

$$\varepsilon'_{ji} = v_{ji} - \psi_{ji} \tag{3.20}$$

v_{ji} is the conventional stochastic error term which is assumed to be an independently and identically distributed normal random variable with mean equal to zero and constant variance $N(0,\delta_v^2)$; ψ_{ji} is a non-negative random error term, independently and identically distributed as $N(0,\delta_\psi^2)$. The translog stochastic production function is:

$$\begin{aligned}\ln(C_{ji}) = b_0 + b_1\ln(NBAS_{ji}) + b_2\ln(SBAS_{ji}) + b_3(\ln(NBAS_{ji}))^2 + b_4(\ln(SBAS_{ji}))^2 \\ + b_5\ln(NBAS_{ji})\ln(SBAS_{ji}) \\ + \sum_{n=2}^{n=4} b_{4+n}QU_n + b_{10}DIS_{im} + b_{11}DIS_{im}^2 + v_{ji} - \psi_{ji}\end{aligned} \tag{3.21}$$

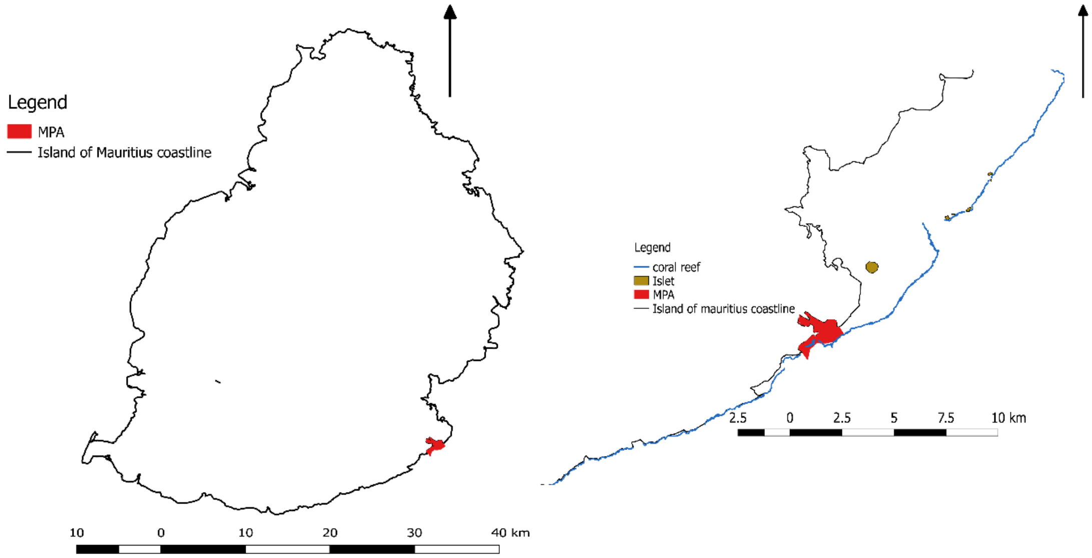
The following section shows the estimation process for each of the four models.

3.4.Study site and data

Study site

Mauritius is located in the Indian Ocean approximately 800km east of Madagascar. The Blue Bay Marine Park is located on the south east coast of Mauritius and was proclaimed a National Park in October 1997 (Figure 3.3) 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). In 2008, it was officially nominated as the second Wetland of International Importance (RAMSAR Site) for Mauritius. The total area of the Marine Park currently has an area of 353 hectares (3.53Km^2), most of it in the lagoon between Pointe Corps de Garde (its northernmost point) and Pointe Vacoas (its southernmost point), it extends about one kilometre seaward from the reef crest. The water depth in the park varies from 1 to 150m metres (Albion Fisheries Research Centre[AFRC], 2008).

Figure 3.3: The study area: Blue Bay Marine Park



Source: author from QGIS

The biodiversity is high with 108 species of coral, from 33 genera. These include a number of commercially important species and many reef fish, including 233 species of fish, some that exhibit schooling behaviour, and 201 types of mollusc. Blue Bay is a popular tourist spot. Its beach is extensively used for recreational purposes.

The Blue Bay Marine Park has been demarcated into zones with specific coloured buoys in order to provide protection to critical habitats, ecosystems and ecological processes, conserve biological diversity, cater for various permissible activities, and separate conflicting human activities. Strict Conservation Zones (A & B) – for the conservation of sensitive and special ecosystems in which a limited number of recreational activities is permitted, such as glass bottom boating, snorkelling and diving. Fishing is not allowed in Strict Conservation Zone A, although line fishing is allowed from the shore in the Strict Conservation Zone B. These zones are demarcated with green buoys.

- Conservation Zone – this places emphasis on the conservation of biological resources. Most recreational activities are permitted, but fishing is not, except line fishing from the shore in a designated area.
- Multiple Use Zone – this allows for recreational activities, line fishing and basket trap fishing.
- Swimming Zone – this is designated for swimming only and demarcated with yellow buoys and floats. Boating and fishing are not allowed in the swimming zone.

A traffic lane provides for entry into or passage through the park by motorised boat with a speed not exceeding 3 knots. Buoys with red and white vertical stripes demarcate the traffic lane. Use of non-motorised boats, fishing, snorkelling, swimming and diving are not allowed in the traffic lane. A ski lane is designated for water skiing and no other activity is permissible. It is demarcated by orange buoys. Mooring zones for the mooring of boats are demarcated by white buoys.

Data requirements and collection

The analysis requires four response variables: (1) total catch per trip, defined as the total weight of fish caught during a trip, (2) catch per trap during a trip in weight, (3) total number of fish captured per trip and (4) number of individual fish captured per trap during a trip. To collect the four variables, a questionnaire was prepared and administered to a sample of fishers.

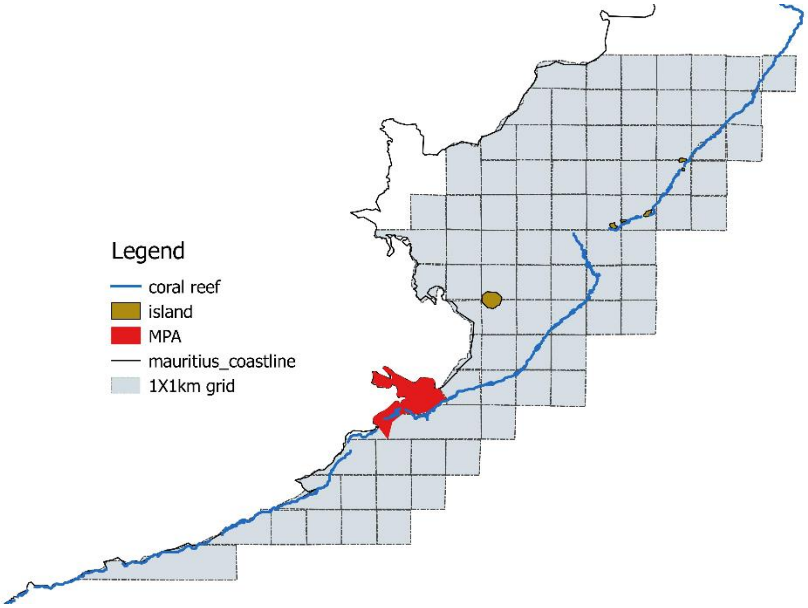
There are approximately 350 trap fishers in the region of the study. No official list of fishermen was available, however, with help from fish landing officers, a list of regular fishers was prepared. This was supplemented by an on-site survey of fishers who were regular for approximately one month (December 2014). 179 regular full time fishers were noted, from whom 100 were selected at random (each fisher was given a number from which 100 were chosen randomly). The study attempted to record the fishing locations of these hundred fishermen, as well as details of fish catch, for 10 trips spread evenly over the year. The random selection was limited in one important respect: it was observed during the interviews, that around 15% fishers 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 for each fisher, 10 trips were recorded, creating a panel of 100 by 10 observations. For each trip, the interviewer recorded the catch as soon as the fishers reached the fish landing stations. The questionnaire recorded the '*total catch of the fisher for the trip in kg'*. For each trip, the questionnaire collects the number of fish of each fish species as well as the weight of the fish.

Table 3.1: Data collection on catch

Name of fish species	Weight of individual fish	Price of fish species per kg
1 e.g. *Siganus Sutor*	…	…
2 e.g. *Naso unicornis*	…	…

The fishers were given a map as shown in figure 3.4 on which the reef and the waters surrounding the reserve are shown. The space is divided into $1 \times 1\,\text{km}$ grid blocks numbered from 1 to 82. On it fishers indicated where their traps had been located and the route taken to and from those traps. Data collected also included the characteristics of fishers and of their fishing technology including the trap sizes and the numbers of traps used.

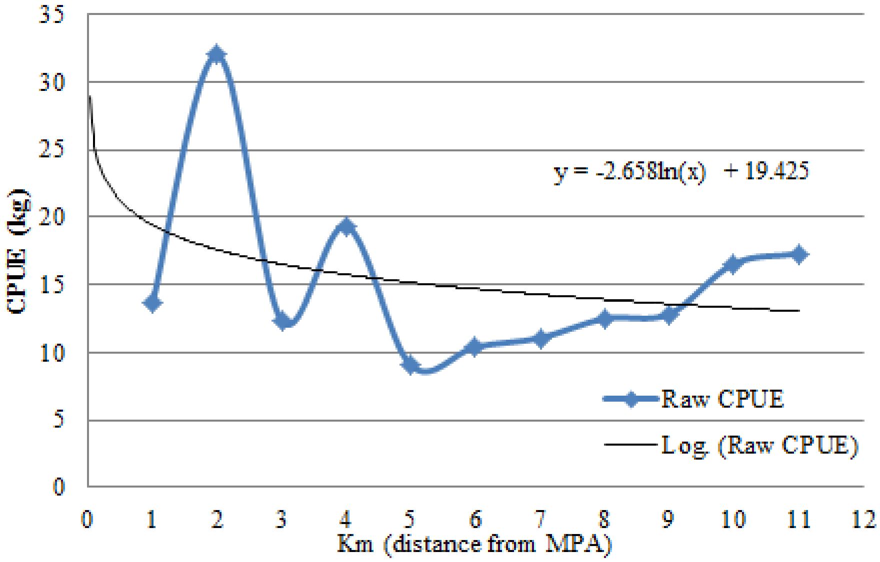
Figure 3.4: Fishing location grids near the MPA



3.5.Results

The survey was conducted during the year January to December 2015. For each fisherman, there are data on 10 trips, together with their spatial locations. Each fisher's catch per trip was recorded in kilograms (Kg). The data on $TCPT_{ij}$ were also positioned spatially on the map using the *QGIS* software. A first attempt is made by estimating the average $TCPT_{ij}$ at an interval of 1 km from the MPA. Figure 3.5 shows the results. The high level of catch near the MPA up to 5km from the MPA is quite visible from the figure. Using a logarithm transformation of $TCPT_{ij}$ against distance, a non-linear relationship is observed, showing a fast declining gradient in the proximity of the MPA and a flat tail after 4km.

Figure 3.5: Graphical illustration of gradient of $TCPT$



A summary definition of the variables used in the analysis is provided in table 3.2. The two variables which are associated with fishing technology are $NBAS_{ji}$ and $SBAS_{ji}$, while the seasonal nature of the data is captured by categorical variables representing quarters. Distance from the MPA is measure as a linear transect from the border of the MPA to the middle of the 1X1 km grid where the fishing has taken place during the trip.

Table 3.2: Summary definition of variables

Variable	**Definition**
$NBAS_{ji}$	Number of baskets used in the trip for fisher j in location i
$SBAS_{ji}$	Size of basket (volume) in meter cube for fisher j in location i
QU_n for n =1, 2, 3, 4	Categorical variable representing quarter: Quarter 1: January, February, March Quarter 2: April, May, June, July Quarter 3: July, August, September Quarter 4: October, November, December
DIS_{im}	Distance from location i to marine reserve m

The GLM uses a set of indicators to test the performances of the models. Following Su et al. (2008) and Ye and Dennis (2009), the model selection process considers the total deviance explained, the change in deviance by adding additional variables, the AIC and BIC. Table 3.3 shows the outcome of the model selection.

The analysis starts with the null hypothesis that none of the covariates have any influence on the stochastic response variable (catch per fisher per trip). Table 3.3 show the relative performances of the models, first if the response variable is assumed to follow a normal distribution, and then if it follows a gamma distribution.

Table 3.3: Analysis GLM fitted to $TCPT$

	GLM-Normal				GLM-gamma			
	DF	Residual deviance	AIC	BIC	DF	Residual deviance	AIC	BIC
Null hypothesis	999	392.95	1.91	-6507.90		343.02	7.04	-6557.82
$+NBAS_{ji}$	998	267.36	1.53	-6626.57	998	252.30	6.96	-6642.64
$+SBAS_{ji}$	997	220.08	1.33	-6666.95	997	203.45	6.91	-6683.59
$+NBAS_{ji}, SBAS_{ji}$	996	211.22	1.29	-6668.90	996	195.62	6.91	-6684.50
$+QU_n$	993	187.54	1.18	-6671.86	993	171.08	6.89	-6688.32
$+ \begin{matrix} QU_n \times NBAS_{ji}, \\ QU_n \times SBAS_{ji} \end{matrix}$	987	168.91	1.08	-6649.35	987	150.54	6.88	-6667.42
DIS_{im}	986	165.84	1.07	-6645.21	986	1.49.12	6.88	-6661.93
DIS_{im}^2	985	163.27	1.05	-6640.87	985	146.97	6.88	-6657.17

Assuming a normal distribution, the residual deviance stands at 393, with an AIC and BIC of 1.91 and -6508, respectively. Adding $NBAS_{ji}$ reduces the residual deviance to 267.4 and subsequently adding $SBAS_{ji}$ further reduces the residual deviance (to 220.3) as expected. Correcting for seasonal factors by adding categorical variables representing quarters again reduces the residual variation substantially (from 211.2 to 187.6) and the interaction variables add further explanatory power. The effects of fishing characteristics and the seasonal effects in explaining variations in the dependent variables were expected from the literature (Stelzenmuller et al., 2007). Adding distance from MPA does improve the model fit as can be seen from table 3.3 as well as figure 3.6 and 3.7. So too does adding the square of distance. This formulation stems from the graphical plot of the two variables.

When the stochastic response variable is assumed to follow a gamma distribution, the residual deviance of the null hypothesis of covariates is reduced. However, the AIC starts with a much higher figure while the BIC is slightly lower. The fall in residual deviance, AIC and BIC follows the same structure as the normal distribution GLM model.

Using the outcome of table 3.3, the effect of distance from the MPA on standardised catch per trip is simulated. Figure 3.6 shows the result. A first observation shows that there is a U-shaped relationship between standardised catch and distance. Standardised catch per trip declines non-linearly consistently for 5kms from the MPA, then rises. Both the Normal and Gamma models provide the same conclusion.

Figure 3.6: Standardised *TCPT* and distance from MPA based on GLM

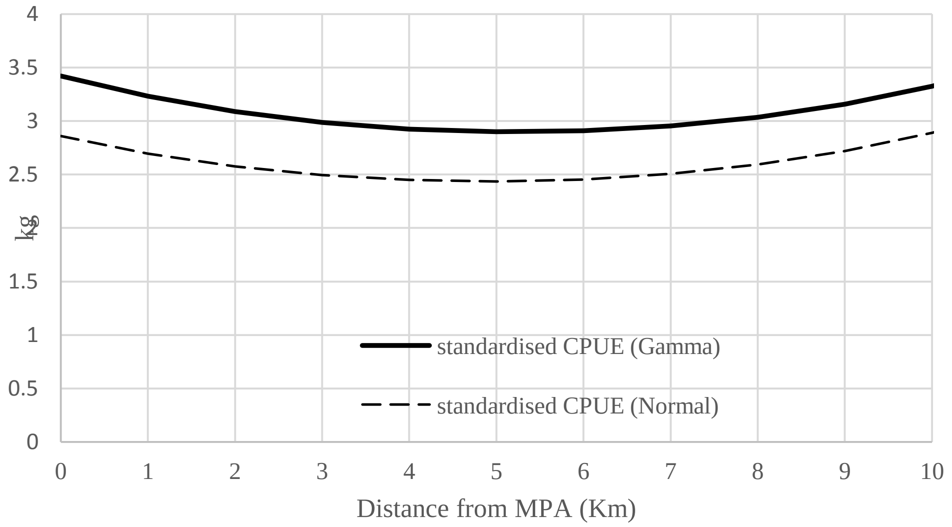


Table 3.4 parallels the previous analysis, but uses catch per basket trap rather than catch per trip. The conclusion from normal distribution and gamma distribution of the catch per basket trap provides similar results to the catch per trip.

Table 3.4: Analysis GLM fitted to $CPBT$

	GLM-Normal				GLM-gamma			
	DF	Residual deviance	AIC	BIC	DF	Residual deviance	AIC	BIC
Null hypothesis	999	307.38	2.86	-6593.47	999	302.45	1.64	-6598.40
$+NBAS_{ji}$	998	305.94	2.86	-6588.00	998	291.02	1.61	-6602.92
$+SBAS$	997	238.80	280	-6648.24	997	238.00	1.41	-6649.03
$+NBAS_{ji} \times SBAS_{ji}$	996	233.65	2.79	-6646.48	996	231.85	1.38	-6648.28
$+QU_n$	993	211.49	2.78	-6647.92	993	207.95	1.28	-6651.45
$+ QU_n \times NBAS_{ji}, QU_n \times SBAS_{ji}$	987	167.11	2.74	-6650.85	987	180.87	1.15	-6637.08
DIS_{im}	986	164.88	2.74	-6646.17	986	177.44	1.14	-6633.61
$DIS_{im}{}^{2}$	985	162.28	2.74	-6641.86	985	174.44	1.12	-6629.71

Using the catch per basket trap as the response variable, the relationship between standardised $CPBT$ and distance from MPA is simulated. A declining gradient near the MPA and rising $CPBT$ after 5km is also observed.

Figure 3.7: Standardised $CPBT$ and distance from MPA based on GLM

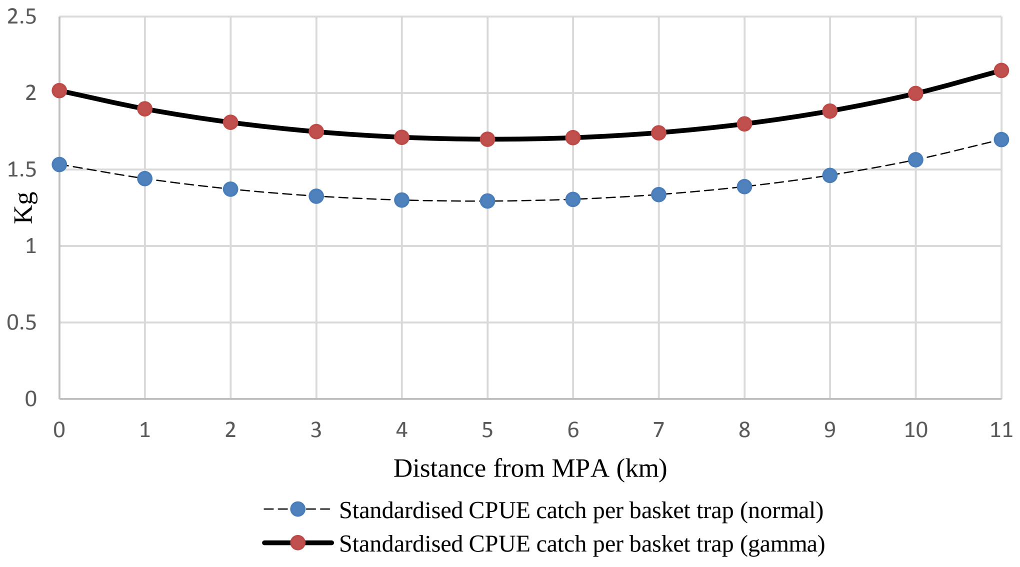


Table 3.5 shows the results of the economic models, that is, the average production frontier and the efficient production frontier.

Table 3.5: Standardised $TCPT$ based on economic models: the average production and stochastic frontier (translog production)

Covariates	**OLS**	**SF**
$\ln(NBAS_{ji})$	-0.958 (0.157)***	-0.675(0.169)***
$\ln(SBAS_{ji})$	0.724 (0.104)***	0.837(0.083)***
$(\ln(NBAS_{ji}))^2$	0.560 ().056)***	0.493(0.047)***
$(\ln(SBAS_{ji}))^2$	-0.030 (0.015)**	-0.027(0.011)***
$\log(NBAS_{ji})\times\log(SBAS_{ji})$	-0.083(0.069)	-0.144(0.0514)***
QU_2	-0.341(0.057)***	-0.409 (0.041)***
QU_3	-0.441 (0.578)***	-0.494 (0.041)***
QU_4	-0.570 (0.062)***	-0.635 (0.047)***
DIS_{im}	-0.070(0.021)***	-0.074 (0.018)***
$DIS_{im}{}^2$	0.007(0.001)***	0.006(0.001)***
Constant	1.222(0.222)***	1.323(0.222)***
N	1000	1000
F test	200.35	
Prob>F	0.000	
R-squared	0.644	
Wald chi square		2080.88
Prob>chi square		0.00
Log likelihood		-361.48
Sigma v		0.231 (0.011)
Sigma u		0.284 (0.018)
Sigma2		0.134 (0.008)
lamda		1.232 (0.255)

The U-shaped relationship between standardised $TCPT$ and distance from MPA is also observed when using the economic models.

Figure 3.8: Standardised $TCPT$- average versus efficient frontier production function

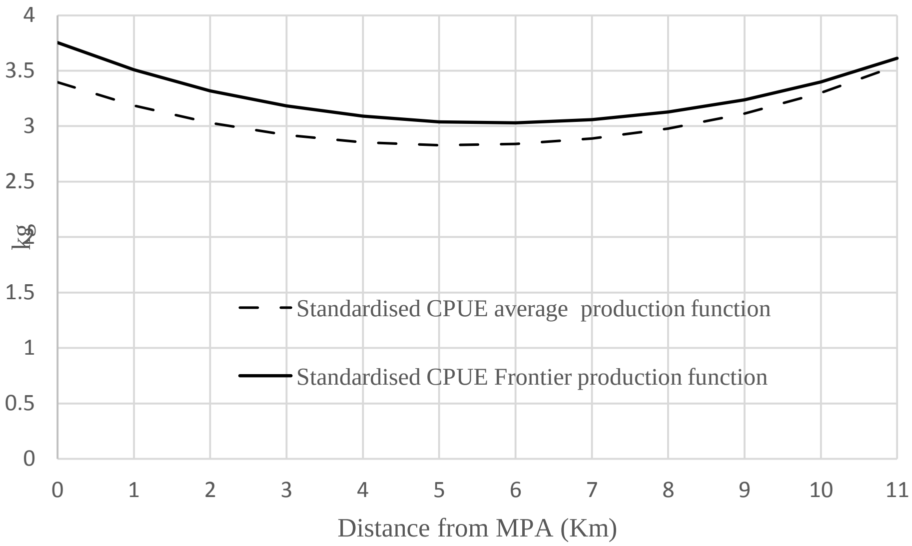
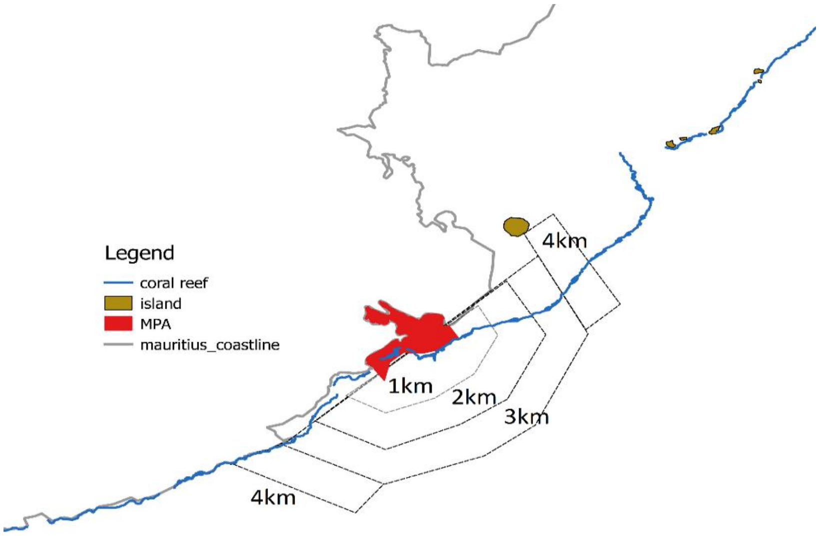


Figure 3.9 shows the extent of declining gradient on the map near the MPA. The two horizontal lines show the decreasing in abundance as measured by the standardised $TCPT$ after which it starts rising again. The figure shows that the declining gradient includes waters inside and outside the reef.

Figure 3.9: Extent of declining gradient of catch near the MPA



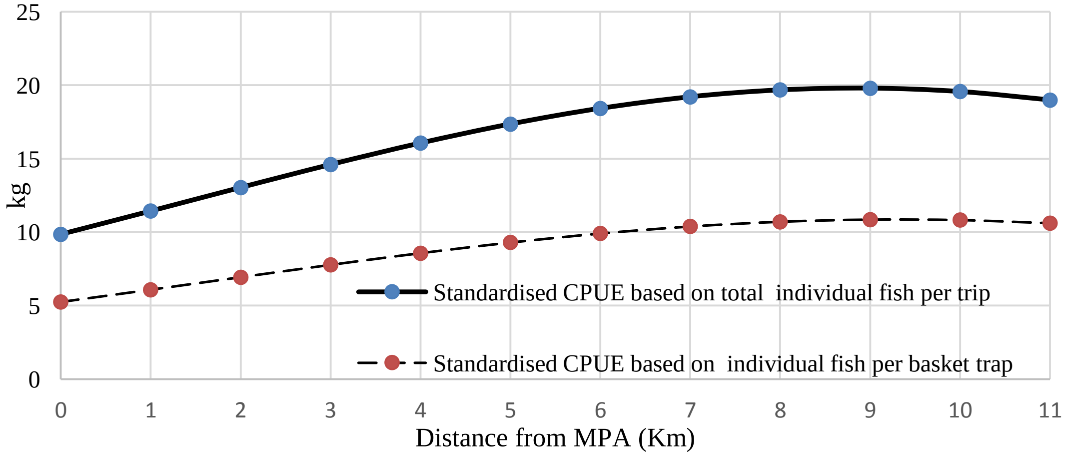
The last model relates the number of individual fish per trip and per basket trap to the distance from MPA. Table 3.6 shows the performance of the model.

Table 3.6: Analysis GLM fitted to $TIFT$ and $TIFB$

	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
$+ NBAS_{ji} \times SBAS_{ji}$	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
$+ \begin{matrix} QU_n \times NBAS_{ji}, \\ QU_n \times SBAS_{ji} \end{matrix}$	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

Adding each subsequent variable reduces the residual variance significantly as well as the AIC and BIC. The simulation exercise is shown in figure 3.10.

Figure 3.10: Number of individual fish and distance from MPA



This presents a very different picture. While the mass of fish per trip and per trap *de*creases with distance, the number of fish caught increases continuously as we move away from the MPA to 8km, then stabilises and falls slightly. This is true for both the total number of fish per trip and the total number per basket trap. The reserve is found to contribute large fish to the catch.

3.6. 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 (for e.g. Bellier *et al.,* 2013). The study infers a declining fish abundance for a distance of 4km from the MPA when using the first two measures, standardised catch in Kg per trip and mass of fish per trap, as a measure of abundance. The results are consistent whether using statistical approaches, that is, GLM normal and GLM gamma, or economic approaches i.e. the average and efficient production frontiers. This supports the evidence of spillover effects observed in marine reserves as in similar studies e.g. (Roberts et al., 2001; Rakitin & Kramer, 1996; Goni et al., 2006).

The rising catch per trip as well as catch per trap after 4km corresponds to the abundance in the off-lagoon areas. As one moves further away from the marine reserve, the fishing area is located outside the reef where stock of fish is relatively higher.

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 the catch fallen. Table 3.7 presents estimates of the percentage change in CPUE from the MPA boundary to points 4 km away using the statistical and economic (production function) approaches.

The GLM normal and GLM gamma estimate a decline up to 17-18% from the boundary to 4km of the MPA using the total catch per unit while the GLM normal and GLM gamma for catch per basket show a decline of 18.5 and 18.7% respectively. The economic approach, that is, the average production frontier and efficient production frontier estimates a decline up to 20.2% and 23.5% respectively.

Table 3.7: Summary of findings

Response variable	Approach	Model	Distribution of response variable	Link function	% change of response variable from MPA to 4 km from MPA
Total Catch per Trip	Statistical	GLM	Normal distribution	log-link	-18
	Statistical	GLM	Gamma distribution	log-link	-17.43
Catch per Trap per Trip	Statistical	GLM	Normal distribution	log-link	-18.5
	Statistical	GLM	Gamma distribution	log-link	-18.7
Total Catch per Trip	Economic	OLS	Average production function	log-log	-20.11
	Economic	OLS	Efficient production frontier	log-log	-23.49
Number of fish per trip	Statistical	GLM	Negative binomial	log-link	76.06
Number of fish per trap per trip	Statistical	GLM	Negative binomial	log-link	77.14

A fundamental question is why the negative gradient prevails even after controlling for the many fishers' specific effects? If commercial species are too mobile and fishers have free access, fish yield is expected to stabilise, unless the spillovers of the MPA is continuous and systematic. A possible explanation may be obtained from an analysis of specific fish species which exist in the adjacent areas. Table 3.8 shows that there are four main fish species which are present in most of the fishing trips: The proportion in the catch per trip of *Naso unicornis* is 60.77%, *Lethrinus nebulous* is 34.4%, *Siganus sutor,* 18.1% and *Scarrus ghobun* is 13.9%.

Table 3.8: Main fish species near the MPA

Local names	Fish species	% of trips with the fish species		% of the fish species in total catch from fishers' trip (kg)	
		Within 4km from MPA	Beyond 4km from MPA	Within 4 km from MPA	Beyond 4km from MPA
Cordonnier	*Siganus sutor*	18.03	48.29	64.77	55.95
Licorne	*Naso unicornis*	60.66	30.87	65.87	61.3
Vielle	*E pinephelus fasciatus*	7.38	22.32	36.11	33.29
Capitaine	*Lethrinus nebulous*	34.43	11.62	40.47	43.38
Carp	*Kyphosus*	6.56	3.53	37.5	37.1
Dame Berry	*Lethrinus mahsena*	6.56	8.66	34.4	35.2
Cateaux	*Scarrus ghobun*	13.93	26.42	57.35	45.37
Rouget	*Parupeneus Sp.*	5.74	16.51	32.14	29.66
Chirurgien	*Acanthurus Sp.*	3.28	6.61	43.75	31.03
Carangue	*Caranx Sp.*	0.82	5.35	25.3	42.55
Cabot	*Hypseleotris cyprinoides*	9.02	6.15	54.54	37.96

The 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 such as a hyperbolic shape, whereas sedentary fish should exhibit a steep linear gradient and highly mobile fish a flat gradient. The species that spends part of its life in the reserve, but then wanders three or four kms away would be the sort of vagile species such as the *Siganus Sutor, Lethrinus nebulous* and *Lethrinus mahsena*. The Naso *Unicornfish* is highly vagile fish and has home range which extends from a linear distance of 0.3km to 1 km (Hardman et al., 2010; Marshel et al., 2011; Green et al. 2015). Its spillover is likely to be less than 4km.

Using the individual number of fish as response variable (total and per trap) shows a rising gradient of abundance, i.e. there seem to be more fish the further one moves from the reserve. A naïve interpretation 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 near the MPA are larger, while the abundant juveniles are found further way.

Reserves should increase the mean sizes of sexually mature fish of each species in the population. Fish whose home range is fully located in the reserve should be bigger that those whose home range is only partly in the reserve, which in turn will be bigger than those whose home range is wholly outside the reserve. Therefore, 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 & 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.

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 & Russ, 2005). These higher rates of aggressive interactions induce subordinate fish to relocate to home ranges outside the reserve (Kramer & Chapman, 1999). If such density-dependent aggressive interactions occur, with larger fish dominating smaller fish, a consequence is a gradient of mean sizes declines with distance from the reserve (Abesamis & Russ, 2005).

In order to examine whether mean size is higher near the MPA, the study further collects the weights of the main fish species which are recorded near the MPA and tests the difference in their sample means. Table 3.9 shows the results. A clear observation from the table is that 9 out of the 11 fish species show a higher weight within a 4km 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 comes from the *Naso unicornis, Lethrinus mahsena,* and *Scarrus ghobun* and to a lesser extent *Siganus sutor* and *Lethrinus nebulous.*

Table 3.9: Average weight of individual fish near the MPA

Local name	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		
Cordonnier	*Siganus sutor*	582.72	522.24	-2.03**	V
Licorne	*Naso unicornis*	1959.46	1484.21	-4.92***	HV
Vielle	*E pinephelus fasciatus*	193.75	166.75	-0.766	S
Capitaine	*Lethrinus nebulous*	1102.33	922.88	2.44 **	V
Carp	*Kyphosus*	1116	828	-3.32 ***	HV
Dame Berry	*Lethrinus mahsena*	725	569.38	-1.77*	V
Cateaux	*Scarrus ghobun*	1085.29	680.03	3.75***	V
Rouget	*Parupeneus Sp.*	285.71	292.25	0.11	V
Chirurgien	*Acanthurus Sp.*	150	139.66	0.37	V
Carangue	*Caranx Sp.*	NA	666.67	NA	HV
Cabot	*Hypseleotris cyprinoides*	181.81	160.19	0.83	NA
***=significant at 1%, **=significant at 5% and *=significant at 10% NA: Not available					

3.7.Conclusion

The result of this study reinforces the evidence of spillover effects of marine reserves, with higher CPUE adjacent to reserve borders. An exponential gradient is observed. Eventually the negative gradient is influenced by four fish species, namely. Naso *unicornis*, *Lethrinus nebulous*, *Siganus Sutor*, and *Scarrus ghobun*. There is also evidence that the MPA changes the age distribution with a greater number of older fish. The size of the fish near the reserve is relatively larger. This is an important finding which explains the decreasing gradient of spillovers and provides useful insight on the effectiveness of MPA.

The study only measures the decline from the edge of the MPA. Theory suggests that the density should be declining from a point inside the MPA boundary. It is plausible that these figures considerably understate the impacts of the reserve. An important issue which arises is whether effort is concentrated where fish is concentrated. An analysis of drivers of spatial distribution of effort will provide insight on the fishing the line hypothesis that is, fishers concentrate effort at the border of the MPA and their effects on spillovers.

4. Modelling fishing location choice and spatial behaviour of fishers near a Marine Protected Area

Abstract

This study investigates whether higher catch rates near an MPA, and/or in other fishing grounds within a choice set, attract more fishers. A survey conducted in the fishing grounds near an MPA in the Republic of Mauritius, in the Indian Ocean, shows concentration of fishers in regions with lower catch rates. This contrasts with the predictions of the 'fishing-the-line' hypothesis and the ideal free distribution that fishers are likely to be attracted near the MPA with higher resource abundance. Expected catch and catch variability are modelled using the Just and Pope production function. Using the random utility model as framework and the random parameters logit model, the study attempts to explain such behaviour and finds that the drivers of spatial behaviour include expected catch, catch variability, distance from home port to fishing ground, potential physical risk and attitudes towards risk of fishers. The paper concludes that higher catch does attract fishers, but is a partial and very restrictive explanation of fishers' behaviour. The 'fishing-the-line' hypothesis does hold to some extent, but it should not be taken for granted that rising catch rates in adjacent waters will increase fishing pressure. The paper concludes that factors affecting spatial behaviour of fishers should be considered in the design of MPAs.

Keywords: marine protected areas, fishers' behaviour, spatial analysis, location choice, random parameters logit

4.1.Introduction

There is widespread agreement that protected areas or marine reserves generate conservation benefits (Smith et al., 2005; Lorenzen et al., 2010; Norse et al., 2010; Smith, 2010, Raphael et al., 2017). However, there is ongoing debate as to whether they improve long-term harvests for fisheries (Smith et al., 2005). In the years that follow its creation, it is expected that a marine protected area will increase the abundance of fish, and that this increase will lead to spillovers in the forms of adult and juvenile migration, and the export of larvae to adjacent areas. These are important benefits to fisheries and are used to assess the effectiveness of marine reserves over a defined period of time (Chapman & Kramer, 1999; Gell & Roberts, 2003; Forcada et al., 2009; Norse, 2010, Bellier et al., 2013).

One of the key arguments against the use of MPAs as a fisheries management tool is that the spillovers simply draw fishing effort to the reserve boundaries (Barkai & Bergh, 2010:221). Such behaviour may impact negatively on marine ecosystems by inducing localised overfishing and altering the habitat in the over-concentrated areas. This effort redistribution has been referred to as ‘fishing the line’ (Kellner et al., 2007; van der Lee et al., 2013). The ‘fishing the line’ hypothesis inevitably throws doubt on the effectiveness of marine reserves as remedies to fisheries degradation, since it suggests that fishing pressure is likely to increase in areas near reserves (Smith, 2004, 2005; Smith & Wilen, 2003; Daw 2008). Importantly, it suggests that fisheries may fall back to the same condition they were in before the reserve was created.

This ‘fishing the line’ behaviour is often simply assumed. Even if economic theory predicts that potential rents are squandered in an open access fishery, and that fishers will earn normal profits only, fishers’ choice of fishing grounds is far more complex than suggested by the simple assumption that rising catch will attract more fishers.

Expectations that fishing effort will be relocated may be important for the design and site selection of a marine reserve. If the aim of the MPA is to protect a breeding site or a rare species, or a recreational site for divers, and the area protected is fairly small, then it may fail in its objectives if fishers target its boundaries. On the other hand, if the reserved area is large relative to the fishable area, a reserve can reduce fishing opportunities, forcing people with limited alternative occupations to abandon fishing altogether. The details of fisher effort relocation can therefore be crucial since it affects the broader costs and benefits of a reserve: in particular its impacts on expected catch, catch variance, targeted species and fishing costs. Valcic (2009) makes the point that one may design the appropriate regulations and incentive structures to minimize the impacts of such effort displacement. The starting point is to determine the factors, trip-related and other, influencing fishing location choice.

This study tests the hypothesis that rising catch rates will attract fishers at the border of a marine reserve. It also examines other determinants of fishers' location choices, building on work done elsewhere by Bockstael and Opaluch (1983), Eales and Wilen (1986), Campbell and Hand (1999), Bingham et al. (2011), Andersen et al.(2012), and van Putten et al. (2012). Its aim is to analyse the behaviour of fishers and from the analysis to draw inferences for use by planners seeking to establish MPAs in similar sites used by artisanal fishers under conditions of open access.

Once the socioeconomics and trip-related determinants have been established, the study conducts a simulation exercise. This is to calculate the responsiveness of fishing effort with respect to changes in the factors identified as drivers of fishing location choice. The simulation also separates the waters surrounding the MPA into those that are complements to it, and those fishing areas that are its substitutes, as well as the marginal rate of substitution. This process establishes the opportunity cost of the reserve's creation. Finally, a simulation exercise is undertaken to estimate the expected change in fishing effort should the area of the MPA be increased. Effort displacement is also simulated for the closure of other fishing areas, to showcase the issue of spatial connectivity across fishing areas.

4.2.A brief review of fishing literature on spatial choice

From both a theoretical and an empirical perspective, fishing is non-random (Gillis et al., 1992). Economic theory goes beyond a simple analysis of effort distribution and adopts a perspective which is consistent with the ‘ideal free distribution’ (IFD) of behavioural ecology (Gillis et al., 1992; Abernethy et al., 2007; Gillis & van der Lee, 2012).

The IFD predicts that, assuming free movement and perfect knowledge, fishing pressure should increase with resource abundance. Ultimately efforts will be adjusted among fishing areas so as to equalize catch per unit of effort (CPUE) across all areas. In reality, CPUE frequently varies across neighbouring fishing grounds. When there are differential catch rates, factors other than expected catch or revenue may be affecting the fishers’ choice of fishing ground (Holland, 2000).

H. S. Gordon’s (1954) pioneering paper showed how economic theory be used to inform fisheries management. It was followed by numbers of studies evaluating fishers’ behaviours in terms of economic theory (for example, Wilen, 1976; Eales &Wilen, 1986; Curtis & McConnell, 2004). The economic framework, typically using micro-level analysis of fishers’ behaviours and discrete choice models, provided both the theoretical underpinnings and quantitative techniques for the analysis of spatial behaviour in fisheries (Davies *et al.*, 2014). Fishermen are assumed to choose between finite discrete alternatives, each having a set of possible outputs or closely substitutable inputs. In this respect, there are several key factors which determine site selection.

There have been numerous examples: Wilen (1976) in a study of aggregate behavioural responses concludes that fishermen respond to profitability in their long-term decision to enter or exit a fisheries. Similarly, Bockstael and Opaluch (1983), in a study of location and species choice of New England fishers, used a utility-based framework to model the micro-level behaviour of fishers and also found that fishermen react positively to expected profits. Eales and Wilen (1986) found that expected catch explained the choice of fishing site in the Alaskan Pink Shrimp Fishery, while Campbell and Hand (1999) made a similar observation in the western Pacific tuna fishery.

Bockstael and Opaluch (1983) broadened these insights into fisher behaviour by adding risk as an explanatory variable. Their analysis noted that fishing location choices are negatively affected by risk, modelled by the variance of output. Dupont (1993) expanded this approach further in the British Columbia salmon fishery. After generating expected profit through price forecasts derived from an ARIMA model she found that not only expected seasonal profit and its variability, but also expected wealth and its variability, were determinants of fishing location choice. Mistiaen and Strand (2001) considered price as a deterministic variable and harvest as a stochastic variable to construct expected profit: using a multinomial logit as well as a random parameters logit model for fishers in the North Atlantic fishery, they also found that expected profit and its variability affect locational decisions by fishers.

Since profit is not driven by revenue alone, the economic approach also treats fishing cost as an important driver of fisher behaviour (Raphael et al., 2017). Fishing cost in turn is strongly linked to distance from home port to fishing location. Distance travelled to a site is found to negatively influence the likelihood of its selection by Campbell and Hand (1999), Mistiaen and Strand (2001) Berman (2007), Bingham et al. (2011) and Andersen et al. (2012). Campbell and Hand (1999) further noted an elastic response to expected value of catch, and an inelastic response in respect of travel cost.

In an uncertain environment, if a particular location is high yielding, but has higher than average catch variability, the extent to which it is used will, to a large extent, depend on whether fishers are risk-averse or risk-loving (Mistiaen & Strand, 2000; Eggert & Tveteras, 2004; Eggert & Lokina, 2007). According to Smith and Wilen (2005) commercial fishermen are inherently risk-loving; however, this generalisation is clearly an oversimplification. Salas and Gaertner (2004) pointed out that some fishers are willing to sacrifice high catches to minimise personal and economic risk, and a number of empirical studies have concluded that fishers are risk-averse (Dupont, 1993; Mistian & Strand, 2000; Eggert & Lokina, 2007). Fishers' attitude towards risk is an important factor influencing fishers' decisions (Raphael et al., 2017). Discrete choice models, usually modelled by a parametrisation of mean and standard deviation of revenue[3], are one way to infer the degree of risk aversion amongst fishers. However, this can also be obtained via separate experiments that provide information on risk preferences (Eggert & Martinsson, 2004, Eggert & Lokina, 2007; Brick et al., 2012).

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[3] This technique is used in this study and further information is provided in the next section.

Fishers' individual characteristics, including their fishing practices and vessel characteristics, have also been found to play an important role in fishing location choice (van Putten et al., 2012). Bingham et al. (2011) analyse the relationships between site quality, angler characteristics, and urban angling behaviour in five north eastern New Jersey counties. Using a repeated nested logit model, the authors find that a fisher's age and race also affect the likelihood that they will visit a specific fishing site. Holland and Sutinen (2000), Mistianen and Strand (2000), and Eggert and Tveteras (2004) capture individual heterogeneity in their estimation models and argue that it improves the predictive power of fishing behaviour models.

There are many other factors influencing the rate at which fishers change their sites including technical considerations, site characteristics, and social factors. Technical factors such as fishing equipment (Eales & Wilen 1986), tactical decisions (Salas & Gaertner, 2004; Christensen & Raakjaer, 2006) information sharing (Curtis & McConnell, 2004; Gillis et al., 2005) are important determinants of fishing behaviour. Sites characteristics, including the biophysical ocean and weather conditions (Bingham et al. 2011, Lopes & Begossi, 2011; Andersen et al., 2012; van Putten et al., 2012; Davies et al., 2014), often combine with site specific experience and knowledge (Andersen et al., 2012) to influence fishers. Social factors such as rules, institutions, local traditions, culture, habits, (Dow, 2008; van Putten et al., 2012) and management practices and regulation (Andersen et al., 2012) are also important. The complex interaction of these factors may translate into inertia and slow adjustment between species/gear combinations from year to year (Bockstael & Opaluch, 1983; Davies et al., 2014).

4.3.Conceptual framework, estimation method and data

The conceptual framework of this paper is based on the random utility model using a two-stage estimation method. The random utility model assumes that a decision maker's utility function is deterministic, but contains some components which are unobservable (Pradhan and Leung, 2004). Such unobserved components are then treated as random variables related to specific characteristics of the decision-maker or attributes of the choices. The approach allows the variations in tastes across the individuals in a population to be combined with unobserved variables in an econometric model (Hanemann, 1984).

Three main drivers of fishers' behaviour in a random utility model are analysed: the expected catch, the distance travelled, and the variability in catch rates. A number of authors (e.g. Eales & Wilen, 1986; Campbell & Hand, 1999; Eggert & Tveteras, 2004), suggest using a sub-model for expected catch rate. This study uses a Just and Pope production function (Just and Pope, 1974), as suggested by Holland and Sutinen (2000), Mistianen and Strand (2000), and Eggert and Tveteras (2004), to estimate expected catch and its variability. The Just and Pope method also allows an estimate of the catch variance facing each fisher and, when combined with the random parameters logit mode, accounts for individual heterogeneity in relation to variability in catch. The technique can be used to classify fishers as risk-loving, risk-neutral and risk-averse.

Random utility model and location choice

Following Bockstael and Opaluch (1982), Smith (2000) and Eggert and Tveteras (2004), the location choice model used in this study is a random utility model based on expected utility. The decision maker i has to choose a fishing location *j* from a finite and exhaustive set of mutually exclusive alternatives J. He chooses alternative j if and only if his utility $U_{ij} > U_{il}$ for $l \neq j$. Preferences are described by a well-behaved utility function whose arguments include a vector of exogenous constraints on current decision-making. Following Pradhan and Leung (2004), the probability that a given individual i, will make a choice j within the choice set C can be expressed as:

$$P_C^i(j) = P\left[U_{ij} = \max_{l \in C} U_{il}\right] \qquad \forall\, j, l \in C,\, j \neq l \tag{4.1}$$

where U_{ij} is the maximum utility attainable by individual i if he selects option j from $C, [j = 1,..., J]$. A linear utility function is specified as the function of observable variables that are assumed to impact the relative utility of alternative choices. The linearity implies that the random utility function can be decomposed into a systematic (deterministic) term (V_{ij}) and a stochastic component (ε_{ij}) as follows:

$$U_{ij} = V_{ij} + \varepsilon_{ij} \tag{4.2}$$

V_{ij} is the utility the individual can expect to obtain, while the random component, ε_{ij}, represents unobservable factors, measurement errors, and unobservable variations in preferences and/or in random individual behaviour. The error term is assumed to be uncorrelated across choices, allowing the independence of irrelevant alternatives (IIA) property in the choice model. This means that outcome categories can be plausibly assumed to be distinct in the eyes of each decision-maker. The unobserved component of the utility is assumed, through extreme value distribution, to have a zero mean; the observed part of the utility is the expected or average utility. The econometric model is driven by the probability that choice j is made as follows:

$$P_{ij} = P(V_{ij} - V_{il} > \varepsilon_{il} - \varepsilon_{ij}) \quad \forall \; j \neq l \tag{4.3}$$

Since ε_{ij} and ε_{il} are random variables, the difference between them is also a random variable. The probability that the fisher will choose alternative j is given as

$$P(Y_i = j \mid X_{ij}, W_i) = \frac{\mathrm{e}^{V_{ij}}}{\sum_{j=1}^{j} \mathrm{e}^{V_{ij}}} \tag{4.4}$$

V_{ij} can be specified as follows:

$$V_{ij} = \theta Z(X_{ij}, W_i) = X_{ij}\beta + W_i\alpha \tag{4.5}$$

Where X_{ij} are the attributes of the choices for which the variables vary across choices, W_i contains the characteristics of the individual or factors whose values are invariant to a choice a fisher makes, and θ, β and α are the vectors of coefficients providing information on the marginal utilities with respect to their relevant characteristics.

Assume the indirect utility is a linear function of mean profits and variance as follows:

$$V_{ij} = \alpha E[W_0 + REV_{ij} - COS_{ij}] + \beta \mathrm{Var}[W_0 + REV_{ij} - COS_{ij}] \tag{4.6}$$

Where W_0 is initial wealth, REV_{ij} is revenue of fisher ith trip to the jth fishing ground and COS_{ij} is the cost. Revenue is defined as:

$$REV_{ij} = P \times CAT_{ij} \tag{4.7}$$

where P is a composite price index in a multi-fish species environment, assumed to be constant for the period. In reality some species are more desirable than others and probably sell at better prices: it could be that fishermen will base their locational decisions on the abundance of a single desirable species rather than of all fish in general as the use of a composite price index would suggest.

By using a standard fish price, equation (4.7) can be converted to a model in which utility is effectively driven by the catch per unit effort rather than by short term variations in the prices fetched by individual species. The use of catch is justified on several grounds. First, the aim of the study is to connect fishers' behaviour with the change in catch rates as a result of spillovers from the marine reserve. Using expected catch is consistent with the fact that spillovers are often evaluated in terms of catch per unit of effort as a measure of abundance, or catch arising from adult and juvenile migration. Price effects can obscure changes in catch rate following changes in conditions in the fishing grounds. Secondly, in many fishing communities, where the most visible measure of output is the catch, there is also a perception that many fishermen are not profit maximisers but prefer catch weight maximisation (Herrero & Pascoe, 2003). On the other hand, Abbott and Wilen (2011) argue that simply lumping species together into a single revenue index is not realistic when fishers' behaviour has much to do with catching different species. Price variability is also a factor in the decision, but according to Eggert and Tveteras (2004), it is a less important source of risk, as fishers usually have information on prevailing market prices and changes occur over a longer time period. For these reasons, prices are assumed to be non-stochastic and exogenous, and expected catch is modelled as the main motivation to fish in a particular site.

The cost function used is a simple one. The cost per trip is simply taken as a linear function of distance that the fisher must travel from their home port to the point at which the traps are laid, such that

$$COS_{ij} = cDIS_{ij} \tag{4.8}$$

where c is the unit cost of traveling. Equation 4.6 reduces to

$$V_{ij} = \alpha W_0 + \alpha E(CAT_{ij}) + \alpha DIS_{ij} + \beta \operatorname{var}(CAT_{ij}) \qquad (4.9)$$

A fisherman's decision to set his traps a specific site is based on the catch expected, not the catch actually achieved. Consequently, they form expectation about the weight, species and number of fish which they can catch at each fishing site. Hence, equation 4.9 posits that the utility is positively related to expected catch measured in weight, $E(CAT_{ij})$, and negatively to distance from home port, DIS_{ij}, and variance of catch, $\operatorname{var}(CAT_{ij})$.

The fishing-the-line hypothesis implies that $\frac{\partial U}{\partial ECAT} > 0$. Equation (4.9) can be used to estimate the sensitivity of site choice, with respect to variability of catch and fishers' risk preferences. If fishers are risk-neutral, then only the mean matters, while risk-averse and risk-loving fishers will make a trade-off between expected catch and variance of catch (Eggert & Martinsson 2004). In the case of the risk-averse, $\frac{\partial U}{\partial \operatorname{Var}(.)} < 0$ while for risk seekers, $\frac{\partial U}{\partial \operatorname{Var}(.)} > 0$.

Estimation method: The random parameters logit model

Random Utility Models can be run in a number of ways; Raphael (2017) mentions conditional logit, multinomial logit, nested logit, and random parameter (mixed) logit, as the most common. Conditional logit initially appears to be the appropriate model in a case like this where the independent variables are all choice-specific attributes whose values vary across alternatives, but where a single parameter is estimated for the effect of the variable (McFadden, 1974). However, the use of conditional logit has been criticised because it requires the independence of irrelevant alternatives (IIA) (Wilen et al., 2002; Hutton et al., 2004; Tidd et al., 2012). The IIA property assumes that the random error component is independent across choices for each decision-maker, and that the unmeasured attributes of choice are uncorrelated. This implies that a change in one attribute of an alternative, or the introduction of a new alternative, or the elimination of an existing alternative, would cause proportional changes in the probabilities of the other alternatives, i.e. the ratios of the probabilities would remain unchanged.

The IIA is clearly unrealistic. The ratio of probabilities between any two choices depends on their respective attribute vectors, but any single probability depends on the attributes of all choices. For this reason, the mixed logit model, also known as the random parameters logit (RPL) (Train, 2009), is used. The RPL is a modified conditional logit model that combines the conditional and multinomial logit (unordered) models into a single model using data with choice- and individual-specific attributes. This allows it to avoid specification error caused by the omission of relevant variables. The RPL relaxes the IIA property because it assumes heterogeneity among alternatives at the population level. It differs from the conditional logit (McFadden, 1974) in that the coefficient of the random variable (variance of catch in this case) varies in a population across individuals. Instead of estimating one parameter for all individuals; the mean and its standard deviation are used to represent the preference distribution in the population of fishers (Train, 2009; Abott & Wilen, 2011). The explanatory power is tested using the likelihood ratio statistic, which is similar to an R^2 in a standard least squares application (Pradhan & Leung, 2004).

Operationalising the model: the Just and Pope Production function

One of the main problems facing location choice models is how to replicate the process of catch expectations formation. It is assumed that fishers' implicitly model expected catch on the basis of past catch rates when they are choosing between different fishing grounds.

This study follows Eggert and Tveteras (2004) and models expected catch and variance using the Just and Pope production function which integrates output risk into the analysis.

Consider the following relationship between input and output:

$$y = g(\mathbf{x}) + u = g(\mathbf{x}) + h(\mathbf{x})^{1/2}\varepsilon \tag{4.10}$$

Where $\mathbf{x}$ is a vector of K inputs, *g(.)* is the mean function (or deterministic portion of the production function), $h(.)$ is the variance function or risk portion, and ε is an exogenous shock with $E(\varepsilon)=0$ and $Var(\varepsilon)=\sigma_{\varepsilon}^{2}$. The vector of inputs, $\mathbf{x}$, influences both mean catch and output risk, because $\operatorname{var}(y)=\operatorname{var}(u)=h(\mathbf{x})\sigma_{\varepsilon}^{2}$. A key requirement of the JP method is that there should be no *a priori* restrictions on the risk effects of inputs, that is, the inputs may increase or decrease the variance of the output. In other words, the production function should be general enough to accommodate both increasing and decreasing risks.

Each fisher has to choose between eight possible fishing grounds, but can choose only one per trip. For the chosen ground, the choice-specific attributes take the expected values of the fisher as measured by equation 4.11. The mean catch $(ECAT_{ij})$ and standard deviation of catch $(SDCAT_{ij})$ take the values estimated by using the Just-Pope production function. This is the first stage 1 of the estimation.

The mean function is estimated using ordinary least squares as follows:

$$CAT_{it}=\alpha_0+\alpha_1 HRS_{it}+\alpha_2(HRS_{it})^2+\alpha_3 NBAS_{it}+\alpha_4(NBAS_{it})^2 + \sum_{m=1,9}\alpha_m DUM_m+\sum_{i=1,99}\alpha_i DUM_i+u_{it} \tag{4.11}$$

CAT_{it} is the catch per trip for fisher i at time t, HRS_{it} is hours fished for fisher i at time t, and $NBAS_{it}$ is the number of basket traps for fisher i at time t. DUM_m is a dummy variable representing the month and DUM_i is a dummy variable for each fisherman.

The parameters of the variance function are estimated using the predicted residuals from the following equation:

$$\operatorname{var}(u_{it})=\exp(\delta_1 HRS_{it}+\delta_2(HRS_{it})^2+\delta_3 NBAS_{it}+\delta_4(NBAS_{it})^2 + \sum_{m=1,9}\delta_m DUM_m+\sum_{i=1,99}\delta_i DUM_i) \tag{4.12}$$

The second stage is to operationalise the mixed logit model. The expected catch and standard deviation of catch for each fisher are calculated from equation 4.11 and 4.12. However, expected values for the explanatory variables must be assigned to the non-chosen alternatives, assuming that those alternatives were also available to the fishers (Pradhan & Leung, 2004). That is, for those sites which are not chosen by the fisher, a proxy is selected. The proxy is the average of the predicted catch of all fishers in a particular region, as estimated by equation 4.11. Eight regions are selected for this study (see figure 4.2). Pradhan and Leung (2004) use the means of expected values of similar size and trip type. In this study, the proxies are taken to be the average an catch of a trip.

The third region-specific attribute is the depth at the fishing site in metres $(DEPTH_j)$, measured where the trap is laid . The data are taken from the survey of fishers.

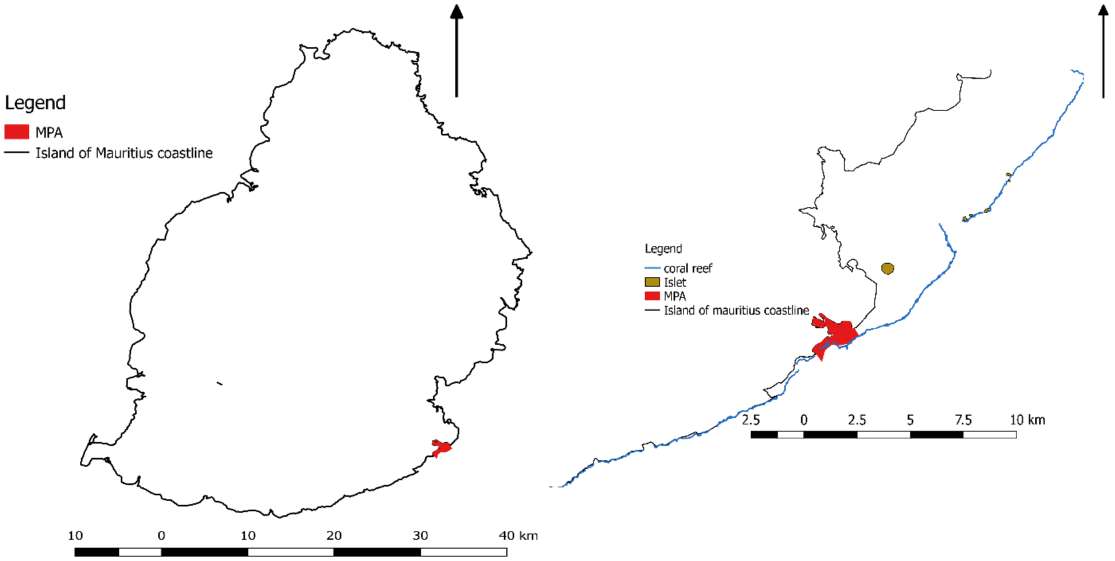
An important region-specific factor is the distance travelled from home port to the fishing locations (Campbell & Hand, 1999). The survey captures data for each fisher, and the variables are constructed by measuring the distance between the home port and the fishing location. An important consideration is that fishers sometimes follow different routes to the same location. Care is thus taken to measure the distance along the route which each fisher actually took to the destination. Where a fisher did not stipulate a particular route, the path used by another fisher from a nearby port has been used as a proxy.

Data

The study area comprises roughly 90km^2 of waters off the south-east coast of Mauritius. The site is one of the biggest lagoon areas of the island and is home to 40% of the island's artisanal fishers). The area also contains the Blue Bay Marine Park (see figure 4.1).

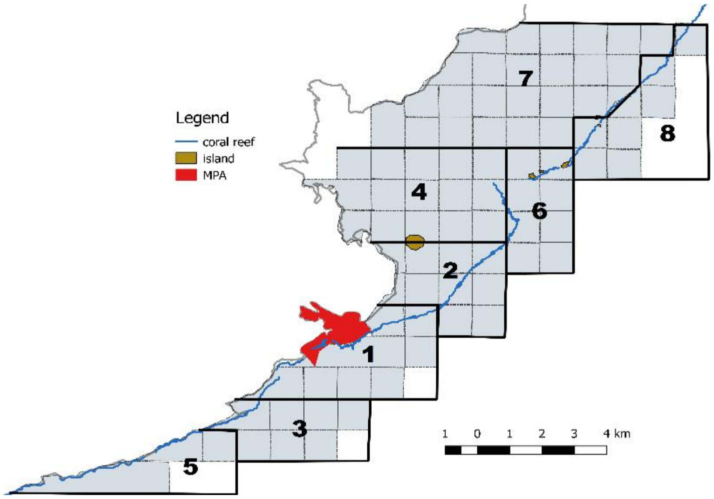
To collect and construct the variables, a questionnaire was prepared and was administered to a sample group of fishers. There are approximately 350 trap fishers in the region of the study, from whom 100 fishers, registered with the fishing authority in Mauritius, were randomly selected. For each fisher, data was collected for 10 trips per year (averaging one trip per five weeks). For each trip, the interviewer recorded the catch as soon as the fisher reached the fish landing stations. The questionnaire recorded (among other data): (i) total catch in kg; (ii) species caught and in what proportion; (iii) number of traps; (iv) length and width of the trap; (v) number of hours spent fishing; and (vi) number of fishers on the boat.

Figure 4.1: Map of Mauritius and study site



Source: author from QGIS

Figure 4.2: Fishing location grids near the MPA



Source: author from QGIS

The fishers were also asked to indicate the locations of their traps. To this end they were provided with the map, as shown in figure 4.2, on which the space is divided into 1×1 km grid blocks, numbered from 1 to 82. The interviewer recorded the number and the position on the map. The route which each fisher took was also traced on the map. Data collected also included the characteristics of fishers and of their fishing technology, including trap size and the numbers of traps used.

One important consideration is that spatial aggregation which leads to a set of geographical areas eventually became one of the objects of choice in the fishers' decision-making process the degree of spatial aggregation needed to analyse the fishing decision. Following Bockstael and Opaluch (1983), Eales and Wilen (1986), Dupont 1993) and Campbell and Hand (1999), the aggregation was done so as to ensure continuity of fishing effort in each fishing area. Curtis and McConnell (2004) point out that a smaller spatial aggregation corresponds to the short run decision and vice versa. The smaller the spatial aggregation, and therefore the area, the less aggregation bias and the greater the ability to model realistic policy measures. However, the smaller the site, the less likely it is that the spatial data will provide much information about the site (Curtis & McConnell 2004). When the individual areas available becomes smaller, the number of choices increases. According to the Curtis and McConnell (2004), this corresponds to the short-run decision of fishers. As spatial areas are aggregated into larger blocks, so the choice eventually shifts from being a short run decision to being a medium to long-term fishing location choice.

4.4.Results

Random parameters logit estimation

Table 4.1 shows the calculated distribution of trips, average catch per trip and other characteristics across the eight fishing areas (refer to figure 4.2. for the geographical location). Fishing area 1 is nearest to the MPA, and exhibits an average catch of 13.7kg per trip. However, the bulk of it is located outside of the reef and therefore involves higher human risk. Despite the relatively high catch rate compared to fishing ground 4 and 7, only 3.7% of trips are made to this area. Fishing the line in this respect appears not to hold. Area 3 and 5 share similar characteristics as area 1 since they are both located in the off-lagoon. However, they are also very far from the fishers whose home port is situated in the north. Fishing efforts are also very low.

Table 4.1: Distribution of trips and average catch per trip from survey

Fishing areas	Distribution of trips (%)	Average catch per trip	Standard deviation of catch rates (kg)	Other characteristics
Area 1	3.7	13.7	8.55	Adjacent to the MPA and off lagoon, high human risk
Area 2	6.1	11.97	7.9	Situated partly in the lagoon and partly off it, high human risk in waters off the lagoon
Area 3	2.5	17.88	9.08	High travelling cost, high human risk
Area 4	27.3	8.2	3	Low traveling cost, low human risk
Area 5	7.9	15.62	11.75	High travelling cost, high human risk
Area 6	26.8	12.81	5.02	Off lagoon, but very productive reef area
Area 7	13.8	10.88	3.6	Low traveling cost, low human risk
Area 8	11.9	19.82	6.11	Off lagoon, high travel cost

Fishing ground 2 includes a lagoon area, but with a larger proportion beyond the reef. It is also situated comparatively far from local home ports. Although adjacent to the MPA and enjoying a catch per trip of 12kg, it attracts only 6.1% of trips. In contrast, fishing grounds 4, 6 and 7, show respectively catches per trip of 8.2kg, 12.8kg and 10.9kg, much lower than the adjacent areas to the MPA, but they attract 27.3%, 26.8% and 13.8% of trips respectively, and make up 67.9% of total trips in the region. Areas 4 and 7 are the most accessible of the areas and also the most reliable, having lowest catch variances.

This section uses a random parameters logit model to explain such behaviour. To facilitate interpretation, table 4.2 defines and characterises the variables.

Table 4.2: Variables and definitions

Variables	Definition	Mean	Standard Deviation	Minimum	Maximum
$ECAT_{ij}$	Expected catch of fisher *i* in fishing ground *j* (kg)	13.77	3.54	0	31.60
DIS_{ij}	Distance from home port of fisher *i* to fishing ground *j* (km)	8.38	3.37	1	18
$DEPTH_{ij}$	Average depth in meters in fishing ground *j*	16.17	7.80	5.30	26.6
$SDCAT_{ij}$	Standard deviation of catch facing fisher *i* at fishing ground *j* (kg)	2.19	0.47	0.36	6.10
$NBAS_{ij}$	Number of traps of fisher *i* used in fishing ground *j*	8.19	1.70	1	13
$VBAS_{ij}$	Size of traps of fisher *i* used in fishing ground *j*	21.88	28.57	1.5	216

Model 1 in table 4.3 shows the calculated expectations of catch and distance as covariates. The standard deviation of catch is treated as the random variable. Importantly, the expected catch in an area is positively related to the likelihood that it is fished while the negative sign of the coefficient of the distance variable shows that the probability of choosing an area falls with distance from a fisher's home port.

Table 4.3 shows the RPL estimation. The random parameter ($SDCAT_{ij}$) has a negative coefficient and is highly statistically significant ($p<0.01$). The negative coefficient is an indicator that fishers are typically risk-averse; i.e. the likelihood that an area will be fished falls with the variability of catches in it. The standard deviation of the variability of catch is also statistically significant supporting the hypothesis that the sensitivity of the catch variability varies amongst fishers. The most commonly used goodness of fit measure for DCMs is the likelihood ratio test for the null hypothesis that all parameters are equal to zero; the results show this can be rejected at 99% confidence.

Table 4.3: Random parameters logit model 1 & 2

Variables	Coefficient (SE)	Coefficient (SE)
	Model 1	Model 2
$ECAT_{ij}$	0.038(0.020)***	0.202(0.024)***
DIS_{ij}	-0.167(0.0239)***	-0.175(0.028)***
$DEPTH_{ij}$		-0.144(0.011)***
Random Parameter		
$SDCAT_{ij}$	-2.702(0.757)***	-2.953(0.826)***
SD of $SDCAT_{ij}$	7.926(0.899)***	9.512(1.137)***
Number of observations	8000	8000
LR Chi^2(1)	1828.00	1903.61
Prob > Chi^2	0.00	0.000
Log likelihood	-1015.63	-919.95
SE=Standard error; ***=significant at 1%, **=significant at 5% and *=significant at 10%		

In model 2, the variable $DEPTH_{ij}$ is added. A negative coefficient is observed with 99% significance. One plausible explanation is that the deeper waters outside the reef that bounds the lagoon here, are regarded as riskier by artisanal fishers in small boats.

If fish density is given, two important determinants of the catch per trip are the number and size of the traps used. Accordingly fishers' location choice is also influenced by these two factors which, consequently have to be controlled for in the analysis. Table 4.4 shows the regression results after including the numbers of baskets and the volumes of baskets as attributes of individual fishermen. This construct of individual specific attributes follows Hoffman and Duncan (1988). A brief explanation is given as follows: consider an attribute, Ws, of fisher, i, which is invariant across choices. Let DUM_k for k=1 to 7 are the dummy variables for region 1 to 7 respectively. The attribute enters in the mixed logit model as follows: $W_s DUM_k$, respectively for the seven regions. Area 8 is the omitted variable. The coefficient gives the effect of attribute in each region relative to the omitted category.

Table 4.4 (page 99) shows the results. The coefficient of ‘expected catch’ is positive and highly significant in all models, i.e. a rise in expected catch increases the likelihood that a site will be fished. This is consistent with studies such as Curtis and Hicks (2000), Mistiaen and Strand (2000), and Eggert and Tveteras (2004) which use expected revenue instead of expected catch. Model 4 treats distance as the random variable. From the lower part of the table, the coefficients of $SDCAT_j$, and DIS_{ij} are highly significant and show the expected signs. The standard deviation of the two variables is also statistically significant, justifying their treatment as random variables in the regression.

To gauge the model’s predictive ability, the estimated percentage of fishing effort in each fishing area is compared to the actual percentage shares (table 4.5). The RPL estimates in table 4.4 (model 4) are used for estimating the expected effort distribution. The shares differ by less than 0.03 points in 7 of the 8 fishing areas.

Table 4.4: Random parameters logit model 4 & 5

Variables	**Coefficient (SE)**	**Coefficient (SE)**
	Model 3	Model 4
$ECAT_{ij}$	0.189(0.0269)***	0.236(0.030)***
DIS_{ij}	-0.327(0.038)***	
$DEPTH_{j}$	-0.067(0.066)***	-0.178(0.049)***
$R1 \times NBAS_{iJ}$	-0.067 (0.0657)	-0.115(0.074)
$R2 \times NBAS_{iJ}$	-0.091 (0.069)	-0.166(0.074)**
$R3 \times NBAS_{iJ}$	-0.251 (0.059)***	-0.334(0.070)***
$R4 \times NBAS_{iJ}$	0.1017 (0.107)	0.039(0.119)
$R5 \times NBAS_{iJ}$	0.076 (0.052)	-0.024(0.059)
$R6 \times NBAS_{iJ}$	0.131 (0.080)*	0.070 (.0876)
$R7 \times NBAS_{iJ}$	0.065 (0.108)	-0.019 (.119)
$R1 \times VBAS_{iJ}$	-0.031 (0.012)***	-0.021 (.0139)
$R2 \times VBAS_{iJ}$	-0.025 (0.001)***	-0.011 (.0102)
$R3 \times VBAS_{iJ}$	0.0015 (0.009)	0.007 (.00954)
$R4 \times VBAS_{iJ}$	-0.0886 (0.013)***	-0.070 (.0126)***
$R5 \times VBAS_{iJ}$	-0.0080 (0.007)	-0.003 (.0076)
$R6 \times VBAS_{iJ}$	-0.0390 (0.007)***	-0.029(.0073)***
$R7 \times VBAS_{iJ}$	-0.094 (0.017)***	-0.093(.0202)***
Random Parameter		
$SDCAT_{j}$	-3.140(0.952)***	-3.143 (.8323)***
DIS_{ij}		-.4128 (.0633)***
SD of $SDCAT_{j}$	9.200 (1.12)***	10.288 (.9848)
SD of DIS_{ij}		.556 (.0722)
Number of observations	8000	8000
LR Chi^2(1)	1602.15	1720.78
Prob > Chi^2	0.00	0.000
Log likelihood	-773.529	-714.21

Table 4.5: Effort distribution actual versus simulated

	Actual proportion of trip	Simulated proportion of trip
Area 1	0.04	0.04
Area 2	0.06	0.04
Area 3	0.03	0.05
Area 4	0.27	0.30
Area 5	0.08	0.05
Area 6	0.27	0.21
Area 7	0.14	0.17
Area 8	0.12	0.15

Elasticities to model parameters

Table 4.6 shows the sensitivity of decisions regarding location to a 1% increase in expected catch in each region. The bold figure in the table shows the elasticity of own effort to rising catch rate expectation; thus a 1% increase in the expected catch rate in fishing grounds 1, 2 and 3 (the adjacent to the MPA) leads to increases of 1.8%, 1.4% and 1.9% respectively in the probabilities of fishers moving to these areas.

Table 4.6: Elasticities of probability of location choices with respect to expected catch

1% rise in expected catch in	**% change in probability that area will be selected**							
	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Area 7	Area 8
Area 1	**1.79**	-0.17	-0.12	-0.06	-0.11	-0.08	-0.05	-0.02
Area 2	-0.18	**1.35**	-0.11	-0.06	-0.05	-0.09	-0.04	-0.06
Area 3	-0.18	-0.14	1.89	-0.01	-0.11	-0.06	-0.01	-0.24
Area 4	-0.34	-0.30	-0.05	**0.54**	-0.14	-0.31	-0.44	-0.07
Area 5	-0.17	-0.09	-0.15	-0.05	**1.34**	-0.13	-0.07	-0.12
Area 6	-0.34	-0.37	-0.18	-0.26	-0.30	**1.22**	-0.31	-0.23
Area 7	-0.17	-0.11	-0.03	-0.28	-0.14	-0.24	**1.11**	-0.08
Area 8	-0.13	-0.31	-0.94	-0.07	-0.34	-0.28	-0.13	**1.06**

In contrast, the own elasticity in area 4 stands at 0.5. The remaining fishing areas, 5, 6, 7 and 8 respond positively to rises in their own expected catch rate. These effort elasticities warrant further explanation.

Areas 1, 2, and 3 are adjacent to the MPA, and are close to the shore, only 10% of fishing effort is directed to them. Why then, does the model suggest that fishing patterns are so responsive to yields in these areas? The travel cost has to be incurred before any catch is made. It is only when fishers are sure that they can cover the travel cost that they will decide to go to a particular site. Hence, a site may be highly preferred as soon as there is a slight increase in expected catch rate which allows fishers to cover the travel cost. Therefore, a relatively high positive change in fisher effort may be observed. The estimates also suggest that effort at a site which is easily accessible will be less responsive to a change in expected catch rate than the effort expended on a site where access entails high travelling costs.

The survey found that fishing area 4 attracted almost 27% of fishing effort. It has the lowest travel cost (see table 4.7) because many fishers have their home port to the north of the MPA and access is easy. However, fishing effort is not responsive to changes in the expected catch. This may be explained by congestion externalities. Numbers in the area are already high, and the impacts of such congestion on effort displacement are well documented (e.g. Curtis & Hicks, 2000; Poos *et al.,* 2010; Raphael *et al.,* 2017).

As the expected catch in one area rises, ceteris paribus, so the probability of fishing in other regions will fall. The cross-elasticities are low, varying between 0.1 and 0.2, indicating that a rise in expected catch rate does not lead to a significant fall in fishing effort in other fishing areas. However, there is a certain degree of substitutability between areas 1, 2, and 3, all of which are adjacent to the MPA.

A rise in the expected catch in fishing area 4 attracts fishers from adjacent areas 1, 2, 6 and 7. Effort in all fishing areas appears highly responsive to the catch in fishing area 6. One explanation is that area 6 is situated close to most of the other fishing areas. Fishing area 7 is a substitute to neighbouring areas 4 and 6. Efforts in areas 4 and 6 are also highly responsive to catches in fishing area 8 which is adjacent to them. The closer are two areas the higher the degree of substitutability between them.

The distance from the home port to a fishing area has a negative effect on the probability that it will be fished. This relationship is consistent with similar studies, such as Stelzenmüller et al. (2008) and Valcic (2009), and is shown graphically in figure 4.3.

Figure 4.3: Distance and location choices from the RPL estimates

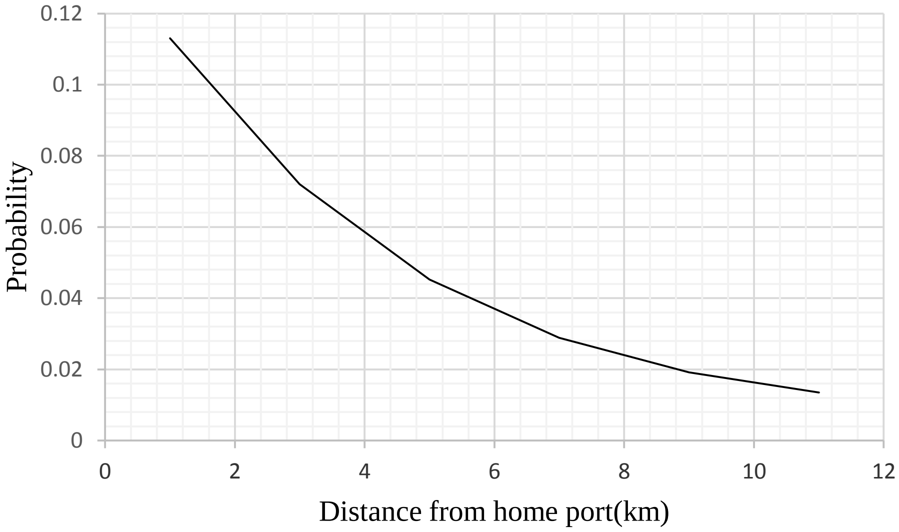


Table 4.7 shows the average distances travelled to each fishing ground by fishers. The average distance to fishing area 1, which is the nearest to the MPA region (region 1), is 4.8km. Fishing ground 4, which has the lowest catch rates but attract most fishers, has an average distance travelled of 5km. The difference between the two alternatives can be drawn from the other factors influencing fishing location choices, for example, the variability of catch.

Table 4.7: Average distance travelled to fishing grounds

	Average distance travelled (km)	Standard deviation	Minimum	Maximum
Area 1	4.81	2.88	1	12.5
Area 2	6.44	2.57	3	14
Area 3	6.52	2.71	1	10.7
Area 4	5.05	1.26	2	10.5
Area 5	7.96	3.69	2.1	17.8
Area 6	7.37	1.81	1	12
Area 7	6.33	2.15	2	12.8
Area 8	10.01	3.01	1.1	16.8

Table 4.8 shows the change in the likelihood of choosing a particular fishing ground with respect to a change in risk as measured by the standard deviation of catch.

Table 4.8: Elasticities of probability of location choices with respect to catch variability

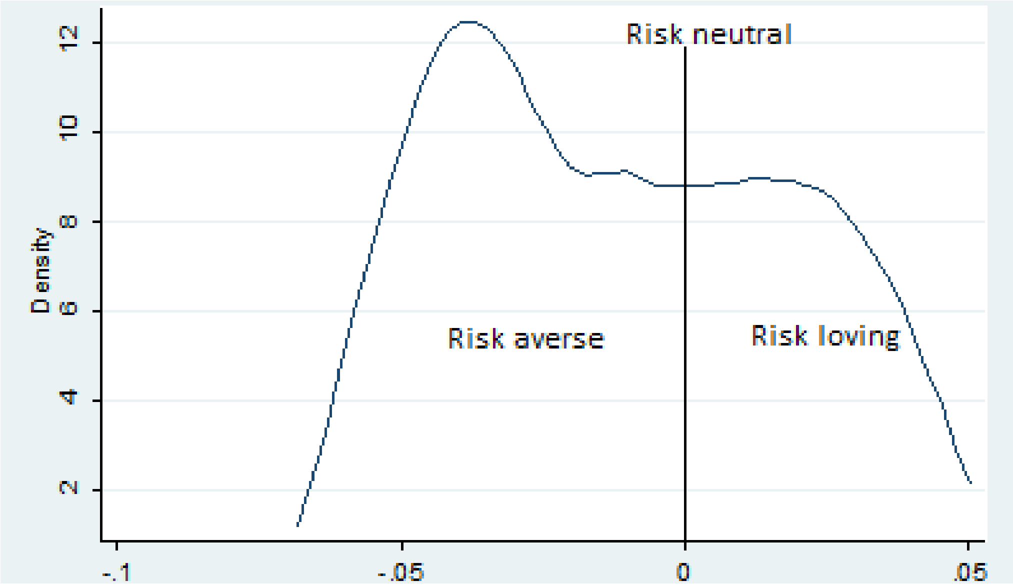
1% rise in SD of catch	% change in probability to choose respective fishing ground							
	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Area 7	Area 8
Area 1	**-2.57**	0.02	-0.29	1.40	0.39	0.51	0.70	-0.26
Area 2	0.01	**1.30**	-0.66	0.42	0.04	-0.26	0.12	-1.09
Area 3	-0.21	-0.69	**10.61**	-0.17	-0.49	-0.82	-0.09	-6.47
Area 4	0.16	0.06	-0.03	**-2.31**	0.16	0.62	1.36	-0.09
Area 5	0.25	0.03	-0.36	0.73	**-1.14**	0.38	0.40	-0.55
Area 6	0.09	-0.08	-0.22	1.04	0.14	**-1.03**	0.55	-0.71
Area 7	0.17	0.03	-0.03	3.02	0.19	0.68	**-3.77**	-0.19
Area 8	-0.05	-0.30	-1.87	-0.11	-0.18	-0.68	-0.13	**3.35**

Importantly, the effects of risk are not consistent across the eight areas. A rise in catch variability does not necessarily reduce the probability of fishing in any region. A 1% rise in catch variability in fishing area 1 leads to a fall in effort, not only in that area but also in areas 3 and 8. When the same analysis is conducted for area 2, a reduction in effort in areas 6 and 8 is observed. However, when the standard deviation of the catch widens in areas 2, 3, and 8, the number of fishers to these areas increases.

The explanation for the ambiguous result on the sensitivity of effort to standard deviation comes from the different attitude to risk of individual fishermen. From table 4.8, it is observed that, while the other areas attract risk-averse fishers, a 1% rise in the standard deviation of catch increases the probability that fishers will visit areas 2, 3, and 8 by 1.3%, 10.6% and 3.4% respectively. In other words, areas 2, 3 and 8 tend to attract risk-loving fishers. While the other fishing areas are in the quiet waters protected by the Island's fringing reef, these areas are situated outside it, in the off-shore part of the fishing ground, an area that also carries greater physical risk.

Using the RPL model, the coefficient of the standard deviation of the expected catch can be used to differentiate fishers according to their risk preference. Almost 33% of the fishers are classified as risk-loving (figure 4.4). The fishers' attitudes have implications for the displacement of effort, since the reactions of risk-averse fishers differ from those of risk-lovers.

Figure 4.4: Risk preference of fishers



Marginal rates of substitution between drivers of fishing location and catch rate expectation

Since the underlying utility function assumed in his study is linear in its explanatory variables, the marginal rates of substitution (MRS) between the drivers of fishing effort appear in terms of catch rate (Buracam et al., 2013). The MRS for distance quantifies how much catch (in kg) would be needed to maintain the same level of fishing activity if there is an increase in the average travel distance of 1km. The result shows that it will need 1.7kg[4]. The MRS with respect to depth measured in meters stands at 0.8kg.

When fishers face a one unit increase in standard deviation of catch per trip in a particular location, they will require a compensation of 13.3kg on average to keep fishing there. However, the rate of substitution is an average that ignores the variation in risk aversion between fishermen.

Modelling the impacts of marine closure on spatial effort distribution

A key issue in the design of marine reserves is effort displacement. Where do fishers go when an area is closed to them? Using the estimates from the RPL, it can be shown where displaced fishers would go if any of the eight areas studies were closed to fishing. In order to simulate the closure of each fishing area, the study follows Bucaram et al., (2012) who suggest that the indirect utility of each closed patch is set to a very negative value and the expected distribution of effort is then recalculated.

Table 4.9 shows that closing a high fishing intensity area such as area 4 leads to a significant increase in fishing effort in areas 6 and 7, while the closure of area with low fishing pressure (area 1,2 3 and 4) leads to a marginal change in other fishing areas.

The welfare loss resulting from area closure is not within the scope of this study. However, the analysis can be extended to estimate such loss following the approach of Curtis and Hicks (2000).

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[4] The MRS between expected catch and distance, $\frac{\partial ECAT}{\partial DIS}$, is calculated as follows: $-\frac{\partial U}{\partial DIS} \Big/ \frac{\partial U}{\partial ECAT}$. The coefficient of the covariate DIS (-0.33) and $ECAT$ (0.19) are used. This gives 1.7.

Table 4.9: Redistribution of fishing effort with marine closure

	Status quo	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Area 7	Area 8
Area 1	0.04	0	0.4	0.04	0.06	0.05	0.04	0.04	0.04
Area 2	0.05	0.05	0	0.05	0.08	0.05	0.06	0.05	0.06
Area 3	0.04	0.04	0.04	0	0.04	0.05	0.05	0.04	0.10
Area 4	0.32	0.33	0.34	0.32	0	0.33	0.40	0.40	0.33
Area 5	0.06	0.07	0.07	0.07	0.07	0	0.08	0.07	0.08
Area 6	0.18	0.19	0.19	0.19	0.26	0.20	0	0.22	0.22
Area 7	0.15	0.15	0.15	0.15	0.31	0.16	0.19	0	0.16
Area 8	0.16	0.16	0.17	0.18	0.18	0.18	0.20	0.17	0

As shown earlier, there was little evidence of fishermen clustering around the border of the Blue Bay reserve and "fishing the line". The question is how and where do they locate themselves, and can this inform policy makers considering the closure of another area. The analysis shows that expanding the locally reserved area by closing any of the eight areas studied, would tend to shift effort in the direction of the areas closer to the local fishing port inshore i.e., 4, 6, 7 and 8.

4.5. Conclusion

The main objective of this study was to provide insights into fishing behaviour in the fisheries which surround the Blue Bay MPA. Such fishing takes place in the inshore lagoon area, over offshore reefs and in offshore deeper waters. The problem is how to predict the locations to which effort would be relocated if there were an increase in the area closed to fishing.

As fisheries policies have increased in complexity, and as the durations of property rights extended have increased, so the ability to model and predict fishers' behaviour in a systematic and scientific manner has become increasingly important (Davies et al., 2014). This is especially true when the policy changes include new or expanded marine reserves (Smith, 2010). Ramirez et al. (2012) and Bucaram et al. (2013) both argued that successful long term fisheries management requires scientific information about the distribution and spatial dynamics of fishing effort - this study attempts to provide such information.

Marine reserves are believed to affect the expected catches in surrounding areas. They also affect the variability in these catches, and the distances fishers may have to travel in order to reach their preferred grounds. With these issues in mind this study models fishing effort in eight contiguous areas believed to have been affected by the Blue Bay Reserve. The data for analysis was obtained from a survey of a sample of fishers whose location of fishing was recorded for 10 trips during a year. Eight distinct fishing areas were analysed. Two were fishing areas with low travelling distance from home port, located inside the lagoon (lower associated human risk), and had low expected catches. Both accounted for 40% of total trip, implying high fishers' concentration of effort. Three areas were adjacent to the MPA, located in the off lagoon far from many fishers and are characterised as having high human risk. Both had very high expected catch rates, but lower fishing efforts. Three areas are located in the off lagoon with high human risk but they differ according to expected catch and distance travelled by fishers from home port.

The analysis supports the view that high expected catch rates attract fishers to an area. However, the elasticity of fishers' effort with respect to expected catch differs across the eight areas. Importantly, the responsiveness of effort to catch rate is lowest in areas which are already heavily fished. One explanation which can borrowed from the literature congestion arising from excessive fishing effort.

This study is based on the premise that fishers respond to economic opportunities as rational economic agents. The assumption that fishers react positively to an increase in expected catch rates means that spillovers from a marine reserve will pull fishers towards its border. This is consistent with Gillis et al's (1992) notion of an 'ideal free distribution' in which fishers move towards locations with higher abundance, and also with the hypothesis that they will 'fish-the-line'. However, although there is clear evidence that the Blue Bay Reserve increases the density of the fish stock, and provides spillovers into adjoining waters, there is little evidence that fishers have been fishing the line. Only 12%

of local fishing trips are in the vicinity even if the catch rates are high relative to other fishing grounds. Thus, whilst higher catches do attract fishers, they are only a partial and very restrictive explanation of fishers' behaviour. The fishing-the-line hypothesis does hold to some extent, but it should not be taken for granted that rising catch rates in adjacent waters will lead to intense pressure. Fishers evaluate several factors before choosing their preferred fishing sites. One of them suggests that distance from home port plays an important role in determining fishers' effort. The further fishers have to travel to reach a site, the greater the relative increase in catch needed to induce relocation to it. When the creation of a marine reserve increases the distance travelled by fishers to some fishing areas, some effort is likely to be relocated to sites nearer to their home port.

The variance of catch at a site is another factor influencing spatial distribution of effort. Fishing grounds, 4, 6 and 7, had the lowest catch variabilities (with standard deviations of 3.0, 5.02 and 3.6 respectively) and attracted almost 76% of trips. It appears from the finding that lower catch variance increases the likelihood that a particular area will be fished.

A finer analysis incorporates fishers' attitudes to risk. The study found that risk-averse fishers are more likely to fish areas that entail lower travel costs and lower variances, even if these mean lower expected catch rates. In contrast, risk-loving fishers are willing to target an areas with high expected catches but high variance.

Low human risk also forms part of the risk-averse fishers' selection of sites. It is observed that 40% of trips are made by risk-averse fishermen to the fishing areas having low potential physical risk. Risk-loving fishers are also those that would be fish the higher physical risk offshore areas.

The RPL model generates the marginal rate of substitution between the drivers of fishers' effort and catch rate expectations. This enables the estimation of costs that fishers would incur following closure. An indifference curve drawn with distance travelled on one axis and expected catch on the other, would have a positive slope of 1.7, representing the compensation of fishers following a rise in distance so that they are not worst-off.

In discussion with fisheries scientists, a comment often made is that Blue Bay is too small to make a real difference to fisheries, and that at its current size it is merely a tourist amenity. The implicit suggestion is that it should be enlarged. Such an increase in reserve size might, however, have unexpected consequences. In the relocation of effort that follows the closure of a portion of a region's water to fishing, effort does not increase uniformly across the remaining space. Indeed some areas may see a decline in effort (complementary areas) whilst others see differing degrees of increase (substitutes of differing strengths). When there are substitution, the new fishing activities may exert a different kind of pressures and interaction in the overall fisheries. Such changes in effort are likely to have differing effects by species across the area as a whole. These species interact with others at different stages in their life histories suggesting that the ecosystem consequences of a reserve may be broader than expected by a naïve policy maker. As shown in this study, the Blue Bay region has several fishing areas which would be especially sensitive to further marine closure in terms of effort distribution, particularly vulnerable ones being those that provide stable catches and are close to fishers' home ports.

4.6.Limitations of the study

The simulations of spatial fishery closure in this study used static economic analysis which neither allowed for the dynamic increases or decreases in the fishable stock following spatial closure, nor for potential changes in fish prices. The results can be seen as worst-case economic outcomes. Drivers such as spatial behaviour such as past behaviour, tradition, and information sharing were also not treated in this analysis. The analysis can be improved through a much in-depth analysis of fishers' location choice, combining several research design quantitative as well as qualitative and replicated in many small scale fishers.

5. Spillover from a marine protected area in the coast of Mauritius - a bioeconomic model of the Bluespine Unicornfish (*Naso unicornis*)

Abstract

Whether an MPA can simultaneously benefit both conservation and fisheries depends on a complex interaction between ecology, the economy, and the spatio-temporal dynamics of the fish population. A key factor is the size of the MPA relative to the fishing ground. The study uses available biological information on Unicornfish (*Naso unicornis*) together with fisheries data from the area adjacent to the Blue Bay Marine Park in Mauritius (Indian Ocean), to construct an age-structured bioeconomic model using a Beverton and Holt recruitment function. The study models an increase in reserve size from 3.5km^2 to 11km^2, and then tests for sensitivity to background assumptions. Three different migration rates are tested, corresponding to a low, moderate and fast movements of fish from the MPA to the adjacent area. The impacts of the reserve are also modelled for two different hypothetical pre-MPA conditions – an originally well-managed fishery with optimal effort, and an over-exploited fishery with fishing effort beyond MSY. The reserve is found to enhance fish population and biomass under both pre-MPA conditions, the growth being highest with low migration. Benefits to fishermen come in the form of an inverted-U-relationship between yield and size of the MPA, but only for an initially over-exploited population i.e. the reserve could be expanded while maintaining or enhancing fisheries yield in an over-exploited fishery. However, expanding the MPA leads to a fall in catch in a well-managed fishery. A caveat is that the steady state is only reached after roughly a decade: a bigger MPA within an over-exploited fishery initially reduces the total catch. This may affect fishers in the short run.

Keywords: reserve size, density dependent spillovers, fisheries benefits, conservation effect

5.1.Introduction

A marine protected area (MPA), or marine reserve, is a portion of oceans where fishing and other human activities are prohibited (Hannesson, 1998; Crowder et al., 2000; Lorenzo et al., 2016). MPAs are established for many different reasons – preservation of sites with unusual biodiversity, conservation of “at risk” species, and enhancing tourism or recreation activities, among others. The question is whether these benefits come at the expense of local fishermen? Does an MPA improve the resource sufficiently for fishermen to benefit from the spillover?

Although many studies have provided evidence of the conservation benefits produced within MPAs, some authorities contend that the overall benefits to local fisheries, especially via spillovers, remain unclear (Leo & Micheli, 2015; Lorenzo et al., 2016) while others such as Bergh and Barkai (2010:231) claim that there are no benefits at all. In part, this is because modelling the replenishment of the fish population and the subsequent increase in reproductive output inside the MPA require an assessment of the within-refuge dynamics (Carr & Reed, 1993). At the same time, it has to be established whether or not the spillover of adults juveniles and larvae, is likely to offset the increased fishing pressure in areas still open to local fishing (Attwood & Bennett, 1994; Russ & Alcala, 1996; Smith & Wilen, 2003; Lorenzo et al., 2016). Previous studies have concluded that the conservation and fisheries benefits of MPAs depend on the initial conditions prevailing prior to the reserve’s creation (Holland & Brazee, 1996), and on the degree of fishing effort control (Sladek Nowlis & Roberts, 1999). Others, however, have a stronger view, concluding that MPAs are beneficial only for overexploited fisheries with low mobility fish (Guenette et al., 1998; Guenette & Pitcher, 1999).

If the aim is to benefit local fisheries (or not to harm them) then a fundamental question is, given the migration rates of the key species affected, what proportion of the local fishing grounds should be set aside as protected areas? The case of the Blue Bay Marine Park in the Mauritian lagoon (Indian Ocean) is an example. Covering area of 353 hectares, and located in the largest lagoon of the island of Mauritius, there is discussion as whether this is an optimal size given the ecological and economic conditions in the fisheries. There is also debate as to whether its design should be replicated elsewhere along the coast.

To answer these questions, the study examines the effects of MPA size on its conservation and fisheries benefits, using fishery data and biological information from the area adjacent to the Blue Bay Marine Park. The study initially constructs a spatially explicit bioeconomic model of a reef fish, the Unicornfish (*Naso unicornis*), one of the Acanthuridas family, and a species locally sought-after by artisanal fishers. The model uses age-structured population dynamics in a Beverton and Holt stock-recruitment function (cf. Holland & Brazee, 1996). Fish landing data, which were collected from the fishing grounds over a year (January 2015 to December 2015), are used to estimate the catch at age of the fish, the fishing mortality rate and natural mortality rates. These together with other biological parameters (length-weight relationship, von Bertalanffy growth and recruitment parameters), are used to estimate the dynamics of the fishery and to evaluate different scenarios relating to the size of the MPA.

It is important to stress that the fishable area in these waters is finite. Since the fishermen are using traps in shallow water, they are largely confined to the area within the fringing reef, together with a few areas near offshore reefs. The limits of the area are set by these reefs and by travel costs of reaching more distant sites.

The study simulates the effects of changes to the proportion of the fishable area set aside as a no-take zone. In particular it models its effects on the resource stocks inside the reserve and in adjacent areas, on yield per recruit and on catch per trip in adjacent areas. The catch is measured in terms of both number of fish and of biomass. Given the very limited information on larval dispersal in the study site, as in other places worldwide (Gerber et al., 2003), only adult and juvenile spillovers are modeled. The migration of fish between sites is density-dependent based on the difference between population densities and is treated as uni-directional from the reserve to the adjacent area. Following Bensermane et al. (2013), three time based scenarios are considered: a fast migration rate where a large proportion of fish (80%) of differential population densities migrate to adjacent area within a year; a moderate migration where around 40% migrate to adjacent areas and a slow migration with a migration parameter of 10%.

A second objective of the study is to consider the effects of conditions prior to the establishment of a reserve. Two different hypothetical pre-MPA conditions are considered – a well-managed fishery with optimum effort, and an over-exploited fishery (i.e. one with effort beyond E_{MSY}). In each case the model addresses the change in fish population and biomass in both the reserve area and the adjacent fishing ground, the yield, and the catch per unit of effort (catch per trip in number and weight) following changes in the proportion of the area set aside as reserve. The study also presents the path of the above mentioned indicators towards their steady states. Finally, the model is used to estimate the reduction in effort that would engender the same volume and rate of increase in biomass as that generated by the reserve.

The chapter seeks to determine the critical size beyond which the MPA imposes costs on local fishermen. This allows questions regarding the trade-off between conservation/tourism and fisheries to be evaluated. Many studies have constructed theoretical models to provide insights on the effectiveness of MPAs (for example Armstrong, 2007; Greenville & Macaulay, 2006; 2007). However, empirical evidence based on specific case studies with real-world parameters and representation has been limited (Wielgus et al*.,* 2008; Takashina & Baskett, 2016), a gap which this study aims to fill.

5.2. A Review of bioeconomic models of marine reserve

The bioeconomic modelling of a fishery uses a representation of biological processes and combines it with the behavior of economic agents (Clark, 1980, 1990; Pelletier & Mahévas, 2005; Punt et al., 2011; Pascoe et al., 2016). Conventional fisheries models simply treat man as another predator, and try to model the proportion of the stocks which he can harvest sustainably. Bioeconomics goes further and investigates the impact of human motivations, technologies and policies on the resource, its health and its economic value. Although the standard bio-physical model of a fishery necessarily involves time (in terms of rates of recruitment and mortality), the bioeconomic models built on them may be static (treating efficiency and yield at a point in time) or dynamic (treating the economic yields of the fishery over time with due consideration to discounting over time and temporal flows of benefits and costs). Sumaila and Charles (2002) suggest that applied bioeconomic models should use data and parameters from actual fisheries, and analyze specific case studies. Not only can such models inform the management and design of MPAs, they can also provide counterfactuals that indicate what would have followed given a different sequence of events or circumstances (Fulton et al., 2015).

Following the work of Gordon (1954) and Scott (1955), much of the early literature on bioeconomic models by fishery economists focused on processes such as price setting, effort and cost dynamics. Gordon used the static economic theory of production to describe the behaviour of fishermen in an open access competitive environment and to demonstrate how overfishing is rooted in that behaviour. The basic fisheries model was subsequently dynamised by including interest rates and discounted future earnings. In particular, Clark (1976, 1985) extended Gordon's model into a dynamic formulation, but the substantive result, that open access fisheries with low entry costs will squander fishing rents, was unaffected.

Unlike the economists, fishery scientists have consistently used detailed age structured models building on recruitment dynamics. However, few considered economic processes, even if over the last decade or so, there has been higher integration of both disciplines to produce truly integrated bioeconomic models (Grafton et al., 2005).

Whilst economic models tend to focus on revenues, costs and property rights, the major components of biological models relate to population dynamics. These include recruitment and mortality functions, individual growth rates and rates of transfer of eggs, larvae, juvenile and adult fish.

Population dynamics

The Schaefer model, one of the most common mathematical formulations of stock, assumes that recruitment, individual growth and natural mortality can be represented simultaneously by a logistic growth equation (Schaefer, 1954). On the other hand, age structured fishery models such as Beverton and Holt (1957) divide the population into age cohorts. Gerber et al. (2003) provide a comprehensive survey of population models of marine reserves. Early contributors in the field of economics on population models include Clark (1976) and Reed (1979). A higher level of biological sophisticated model, the ‘metapopulation model’, has been used by Sanchirico and Wilen (2001), Sanchirico (2004), Smith and Wilen (2003) and Smith (2004). Such models treat the fishery for a species as a set of discrete age- and size-structure sub-populations linked by a dispersal matrix. Each patch has a number of adults at age class and each separate sub-population has a size structure described by a von Bertalanffy equation.

Single-species models are naturally limited and many attempts have been made to develop multi-species, multi-fishery bioeconomic models (see Ströbele & Wacker, 1991; Holland, 2000; Ulrich et al., 2002; Thøgersen et al., 2012). In a multi-species dynamic model, the relationships between the species have to be specified, one of the simplest being predator-prey relationships (Greenville & MacAulay, 2006; Chakraborty & Kar, 2012).

Stock-recruitment function

The stock-recruitment function plays an important role in any age-structured population model. Recruitment can be defined as the number of individuals in a population reaching a specified stage in life cycle during a defined time interval (Arnott & Ruxton, 2002). It is the result of many factors which affect survival between spawning and the individuals’ recruitment to the stock (Subbey et al*.,* 2014).

Recruitment is not only driven by the spawning-stock biomass (SSB) which is often used as a proxy for reproductive potential or egg production but also by factors that affect fecundity such as parental age and size, survival rates, growth history and environment of the individual. These are particularly influenced by MPAs since no-take zones tend to increase the proportion of old mature fish in a population. The SSB is a measure of the stock’s reproductive capacity and its simplest definition is the weight of sexually mature females at the time of spawning, or the integral of that weight over the spawning season or life span (Laurec & Le Guen, 1981; Munyandorero, 2001). This definition means that, within a stock, the spawning biomass developed by a cohort of fish over its lifetime can be used in the analytical modelling approach

When stock size and recruitment are used in a single model, it is implicit that the parental stock determines the numbers of young that will join it in future periods (Subbey et al., 2014). At low stock sizes, recruitment is primarily driven by density-independent factors and thus recruitment increases monotonically with stock size. However, at large stock sizes, other density-dependent factors become increasingly influential. This method of estimating recruitment levels into the future is primarily for the purpose of modelling and not to illuminate the early life-history dynamics of these species. However, parameters characterizing recruitment are of particular interest in establishing reference points to determine the scale and productivity of the population.

Transfer function

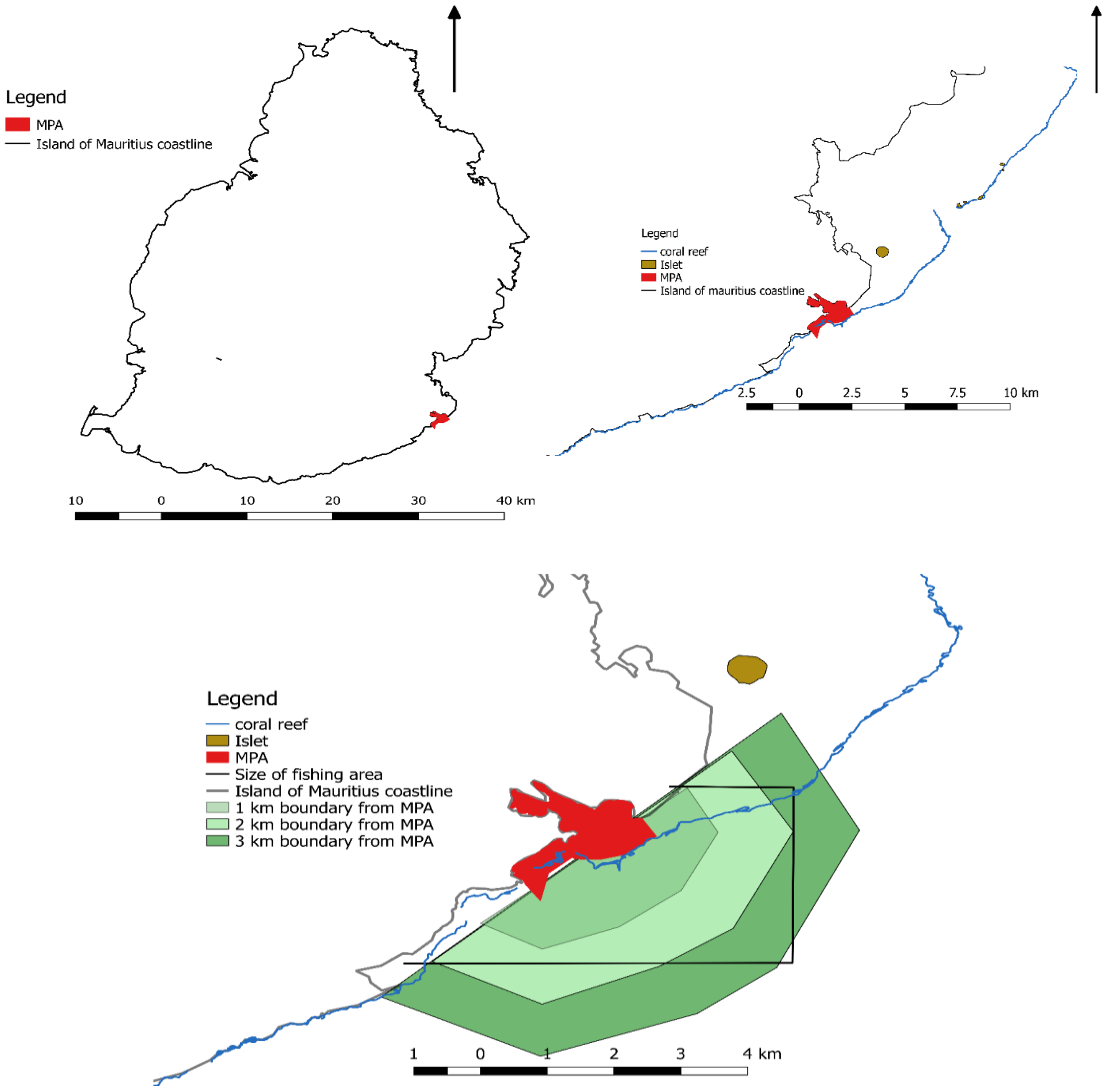
A reserve and its surrounding harvest areas are also linked through the dispersal of eggs, larvae, juveniles and adult fish. Whilst it is simpler to model such movements as uniform random diffusion, independent of location or age (Guenette et al*.,* 1998), they are also regularly treated as functions of the difference between the biomass density in each area assuming that the areas are identical in size (Grafton et al., 2005; Kar & Matsuda, 2007; Armstrong, 2007). In most bioeconomic models, migration between the two areas is usually captured by a density-dependent migration coefficient or dispersal parameter, although some use a unidirectional (independent) coefficient.

The spillover of juvenile and adult fish from a reserve are typically presumed to depend on its size, the movement patterns of the fish and the gradient of population densities between the MPA and the fishing grounds surrounding it (Chakraborty & Kar, 2002). In such models, the net transfer from the protected area to the unprotected area is assumed to be positive. In contrast, ‘sink-source’ models characterize the dispersal flow of fish as independent of population densities. Such models show biomass continuing to flow between patches even after each population has reached its natural equilibrium (Sanchirico and Wilen, 1999; 2001).

5.3.Material and method

The age structured population model in this study has four components: the first is the number of regions which would form part of the connectivity and the modelling. The second is the dynamics of the population, in particular the rates of natural and fishing mortality in each region, including the catch equation. The third is the stock-recruitment function in each region, and the fourth is the flow of the population between regions.

Figure 5.1: Marine Protected Area in south east coast of Mauritius



Source: Author from QGIS 2.18.16

Study site and area classification

The study site is the Blue Bay Marine Park located in the south east of the island of Mauritius. Figure 5.1 shows the location of the MPA. Details of the site are given in chapter 3 of this thesis. The green shaded areas indicate various plausible estimates of the home range of fish species (see section 5.4). Green et al. (2015) estimated the linear scale of movement of *Naso unicornis* at 0.3 to 1km. Botsford et al. (2009) suggest a far smaller home range of 35meters in any direction to and from home point. In this study, it is assumed that the MPA would yield spillover from the boundary of the MPA to 1km^2 in the off-lagoon area.

Population dynamics

Population abundance in an area is measured by the number of survivors from each recruited age class at the beginning of the year and is denoted by the vector $N_{i,t}$ with elements $N_{a,i,t}$ where "a" indexes the age class, "i" is the i^{th} area and "t" indexes the year in the projection horizon. The oldest age class is a plus-group composed of all fish older than a chosen cutoff age (A). Population survival at age (from year to year) is calculated using instantaneous mortality rates. Total mortality ($Z_{a,t}$) is decomposed into natural mortality and fishing mortality. The instantaneous mortality rate on fish of age a is denoted by $M_{a,t}$ and it is assumed constant across areas. Since age-specific natural mortality data were unavailable, this parameter has been assumed be constant across age classes commonly suggested in the fisheries literature. The instantaneous fishing mortality rate ($F_{a,i,t}$) varies according to fishing intensity in the different areas.

Equation 5.1 introduces selectivity, defined as the probability of capturing an animal if it encounters the gear (Punt, 2014; Butterworth et al., 2014). Selectivity, $Sel_{a,i,t}$, changes the fishing mortality rate as follows:

$$\overline{F}_{a,i,t} = Sel_{a,i,t} \times F_{a,i,t} \tag{5.1}$$

The fishing mortality rate is function of fishing effort (*trips*) and the catchability coefficient (q).

$$\overline{F}_{a,t,t} = q_i \times Trips_{it} \tag{5.2}$$

The total number of trips per annum, $Trips_{it}$, is approximated by the number fishers and trips that each fisher make over a year. In many studies, the catchability coefficient q_i is assumed constant across the assessment areas. Here, the coefficient varies across areas because these vary in such characteristics as depth, and whether the location lies within the reef or beyond it.

Given the mortality rates, the size of age cohort a in each area i , and the total population under cut-off age A, follow ‘exponential decay’ with a constant natural mortality rate in each year (Sparre & Venema, 1998):

$$N_{a,i,t} = N_{a-1,i,t-1} e^{-M_{a-1,t-1} - F_{a-1,i,t-1}} \text{ for } a = 1 \text{ to } A-1 \tag{5.3a}$$

And

$$N_{A,i,t} = N_{A,i,t} e^{-M_{A-1,t} - F_{A,i,}} + N_{A-1,i,t-1} \times e^{-M_{A-1,t-1} - F_{A-1,i,t-1}} \tag{5.3b}$$

Annual catch at age in each area $C_{a,i,t}$ is given by the standard catch equation:

$$C_{a,i,t} = \frac{\bar{F}_{a,i,t}}{M + \bar{F}_{a,i,t}} (1 - e^{-(M + \bar{F}_{a,i,t})}) N_{a,i,t} \tag{5.4}$$

The biomass of fish caught during time period t, is the sum of the numbers of fish caught from each size category multiplied by their corresponding mean weights (Quinn & Deriso, 1999; Apostolaki et al., 2002)

$$C_{a,i,t} = \frac{\bar{F}_{a,i,t}}{M + \bar{F}_{a,i,t}} [1 - e^{-(M + \bar{F}_{a,i,t})}] N_{a,i,t} w_{a,t} \tag{5.5}$$

Total population and catch are respectively:

$$N_{i,t} = \sum\nolimits_1^A N_{a,i,t} \tag{5.6}$$

$$C_{i,t} = \sum\nolimits_2^A C_{a,i,t} \tag{5.7}$$

Introducing reserve effect and transfer function

Introducing a marine reserve into the area described above has two impacts of direct relevance: the first is that the size of the fishing area decreases, and the second is that there is a transfer of adult and juvenile fish to adjacent areas. It is assumed that a site's carrying capacity, is proportionate to its area (Kar & Matsuda, 2008; Yamazaki et al*.,* 2015) and that spillover is strongly density dependent giving a uni-directional flow from the MPA to adjacent fished areas as in Grafton et al. (2006).

Assume the size of the area is K and the fraction of it designated for the marine reserve is τ, so that $(1-\tau)$ is the proportion still open for fishing. The population of fish under cut-off age A in the marine reserve will be $N_{MR,t} = \sum_1^A N_{a,MR,t}$ and the population in the fishing ground will be $N_{FA,t} = \sum_1^A N_{a,FA,t}$. The populations per unit area in the marine reserve (n_{MR}) and fishing ground (n_{FA}) will be: $n_{MR,t} = N_{MR,t}/(\tau K)$ and $n_{FA,t} = N_{FA,t}/((1-\tau)K)$, where $n_{MR,t} > n_{FA,t}$ if there is no fishing mortality in the marine reserve.

Before the creation of the reserve, τ is 0, and the population is $n_{FA,t}K$. As the area set aside as a reserve increases (i.e. τ increased) so total fish numbers grow such that: $n_{MR,i}\tau K + n_{FA,i}(1-\tau)K > n_{FA,t}K$

If there are no spillover effect, the loss of fishing ground will reduce catch rates. Given the assumptions that movements of fish are density dependent, and that the migration coefficient (*d*) falls between 0 and 1, total spillover becomes:

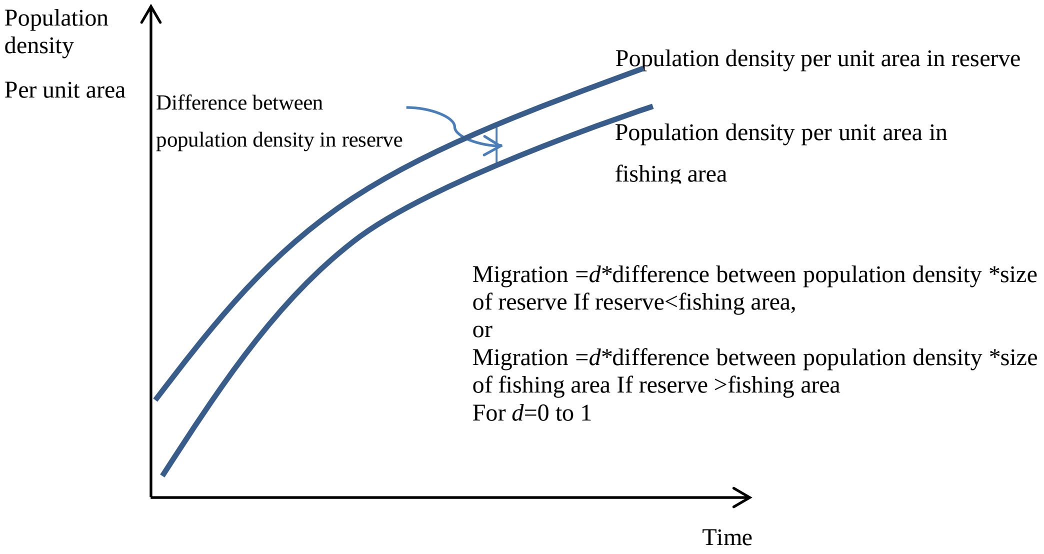
$d[(n_{MR,t} - n_{FA,t}) \times \tau K]$ if $\tau \leq 0.5$ (i.e. if the reserve area is less than the fishing ground)

(5.8a)

$d[(n_{MR,t} - n_{FA,t}) \times (1-\tau)K]$ if $\tau > 0.5$[5] (i.e. if the reserve area is larger than the fishing ground) (5.8b)

According to Grafton et al. (2006), this approach is also compatible with diffusion models that suggest reserve size directly influences dispersal and that spillover is also strongly density dependent. A graphical illustration of migration rate in this model is provided in below.

Figure 5.2: Transfer function –graphical illustration



Source: author

The population in the fishing ground would be given by

$$N_{FA,t} = [N_{FA,t-1} + d(n_{MR,t-1} - n_{FA,t-1})\tau K]e^{-M-\bar{F}} \tag{5.9}$$

$$N_{MR,t} = [N_{MR,t} - d(n_{MR,t-1} - n_{FA,t-1})\tau K \times e^{-M} \tag{5.10}$$

Stock-recruitment relation (S-R)

Before fish can move more plentifully from the reserve to the less densely populated fishing areas, the stock in the reserve needs to grow. This brings in the next step of the

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[5] This condition prevents the area adjacent to the MPA to be overpopulated and eventually prevents the unrealistic density dependent movement of fish from adjacent to the MPA.

model, the S-R relationship. The Beverton and Holt (1957) S-R relationship is given as follows:

$$R(t) = \frac{SBB}{\alpha + (\beta \times SBB)} \tag{5.11}$$

Where $R(t)$ is recruitment, SBB is the spawning biomass at time of spawning, α and β are mathematical parameters with limited biological meaning, α being the inverse of the curve's initial slope and β being the inverse of asymptotic recruitment. Steepness (h) is defined as the fraction of virgin number of recruits expected when the spawners stock size is reduced to 20% of its unexploited (virgin) biomass (Mace & Doonan, 1988). h is the ratio of two recruitment levels: that when the spawning stock is at 20% of its pristine level, and the recruitment when the stock is at its pristine level. It is also an index of resilience - the higher it is (the closer to its upper limit of 1) the faster the stock recovers from overfishing.

The above two parameters are therefore as follows:

$$\alpha = \frac{(1-h)SPR_{F=0}}{4h} \tag{5.12}$$

$$\beta = \frac{5h-1}{4hR_0} \tag{5.13}$$

In the context of commercial fisheries, a target species' recruitment is usually assessed as the number of individuals alive at the age when they first appear within the catches (Arnott & Ruxton, 2002). Recruitment depends upon the number of eggs that are produced by spawning adults, and the natural mortality rate of the eggs, larvae and juveniles between being spawned and reaching an age or size at which they are susceptible to capture.

Application of the model and base line calibration

Biological Characteristics of Naso *unicornis* (Forsskål 1775)[6]

The Bluespine Unicornfish (*Naso unicornis)* is a large-bodied herbivore that plays a key role in structuring the algal assemblage of nearshore benthic communities on tropical coral reefs and where it functions as a principal fish among browsing fishes (Andrews et al., 2016). It is a typically an inshore species that regularly moves into very shallow water on reefs to browse but has been recorded down to depths of 200m. In being one of the *Acanthuridae*, unicornfish exhibit a distinctive 'square' growth pattern during the course of their long lives. This growth pattern is characterized by a rapid growth spurt during the first year or two of the fish's life followed by a quick cessation in somatic growth over the remaining decades of the fish's life, yielding a nearly right-angled growth 'curve' (Trip et al*.,* 2014).

Some of the characteristics of the fish species are: (1) settlement in relatively large number and in discrete, identifiable pulses within a narrow reproductive season, (2) spawning aggregations form at dusk, where currents flow off the reef, (4) after hatching, larvae spend between 60 to 80 days in the plankton before returning to adult habitats (Planes *et al.,* 2002). *Naso unicornis* is widely distributed throughout the Indo-Pacific on both side of the equator (Ford et al., 2016). Importantly, Andrews cites Williams (2008) as saying, 'There is no clear relationship between the abundance of Bluespine Unicornfish and fishing pressure or other factors related to human population density'; although it was identified as one of six species most likely to be experiencing overfishing in Hawaii (Nadon et al*.,* 2015).

According to Andrews et al (2016), the growth parameter for males is 0.46 year^{-1} and for females, 0.43 year^{-1}. Maximum length is 48.4cm for males and 47.8cm for females. However, Ford et al. (2016) observe that from available data, *Naso unicornis* generally display slow to medium growth rates with a growth coefficient of 0.15-0.3. Spawning periodicity is highly seasonal in Hawaii, peaking in May and June (DeMartini *et al.,* 2014;

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[6] The following section details the biological characteristics of this species. It may be omitted without loss of meaning by a reader familiar with the species

Ford et al*.,* 2016). Using observations of the species in Mauritius, Rathacharen *et al.* (1999) find a growth rate of 0.106cm year^{-1} to 0.188cm year^{-1}, with an average rate of 0.163cm year^{-1}. This is (again) consistent with the observation that *Naso unicornis* grows asymptotically slowly in length.

Mortality rates of *Naso unicornis* remain largely unknown. The total mortality and natural mortality coefficients were estimated at 0.543 and 0.40 respectively in Rathacharan *et al.* (1999). The value of the length at first capture was estimated, at 27.42cm at age of 1.9year. L_{50} was estimated at 28.5cm at age 2.03. According to Andrews *et al.* (2016), it can live for more than 50 years but Rathacharan et al (1999) show a much longer life time of 81 years The rapid initial growth rate, and subsequent tapering of growth suggest that unicornfish, if farmed, would have a rapid optimal rotation rate. Harvested in the wild, it seems a species that could support a sustainable harvest if well managed.

Unicornfish regularly move to and from resident spawning aggregations that fall within individual's home ranges (Green et al*.,* 2015). Abesamis and Russ (2005) find they are driven by positive or negative interactions with conspecifics or species belonging to the same guild. Direct measurement of home range sizes involves marking individuals visually or acoustically and following or relocating them after a biologically relevant time step (Botsford et al*.,* 2009). *Naso unicornis* are described to as having a home range consistent with shallow areas (Meyer et al*.,* 2005, Hardman et al*.,* 2010). Home ranges of radius 0.5km to 1km have been observed (Green et al., 2015; Weeks et al., 2017). This information is used to design the fishing area in section 5.2.

Base line data for calibration

The following analysis uses catch data collected from Mauritian artisanal trap fishers. There are roughly 350 fishers harvesting in the area, averaging two trips per week, i.e. 36400 trips per annum are made in the area. A sample of 100 fishers was selected randomly from those regularly fishing in the study area. The process involved making a list of full time fishers who are registered with the authorities and had genuinely adopted fishing as their profession (Fish landing officers assisted with this process). The sample was then dawn randomly from this set. The 100 fishers were followed over the course of a year. At four to six weekly intervals, their fishing locations were recorded with details of fish species caught and the weights of each fish in the trap. For each fisher, 10 trips were recorded over the period January 2015 to December 2015. Of the 1000 trips recorded, 339 yielded catches of *Naso unicornis*.

Weight-length relationship

Aging fish is ideally done using sections cut through otoliths or other bony structures that contain growth rings. This is, however, expensive and time consuming. Since fish continue growing over their lives (albeit sometimes very slowly) one can instead use standardised estimates that convert size into age. A common example (which is used here) is the von Bertalanffy growth function. The fish size is often captured using length, but weight/length relationships are also stable, so weight can also be used. The weight of each fish caught is used to provide an initial rough estimate of the age of the fish and to proxy the length of each fish (done using a weight-length relation). There have been several studies on the weight-length relation for the *Naso Uncornis* including, Peyton et al. (2016) $(w = 0.0405L^{2.89})$; Kulbicki et al. (1993) $(w = 0.0222L^{2.98})$; and DeMartini (1993) $(w = 0.0111L^{3.10})$.These estimates mean that the size and growth rates of this fish species seems to be different across the globe. Given that Rathacharan et al (1999) was a study conducted in the site of the study and the estimates are similar to Edwards et al. (2011), the following weight-length relationship is used.

$$w = 0.0321L^{2.822} \tag{5.14}$$

Where w =weight and L =length

Von Bertalanffy growth equation

The Von Bertalanffy growth equation for the *Naso unicornis* was obtained from Rathacharen et al. (1998). The length to infinity (L_∞) was 81.03, the growth coefficient (k) was 0.16, and the theoretical age at zero length (t_0) was taken to be -0.5. Thus, the age of each fish was estimated using the equation:

$$t = t_0 - \frac{1}{k}\ln(\frac{L_\infty - L_t}{L_\infty}) \tag{5.15}$$

The total catch and the number of trips undertaken, were used to estimate the CPUE for each age class.

Natural and fishing mortality rates

Any attempt to assess the impact of a no take zone needs to estimate a counter-factual: i.e. what would the condition of the resource be if the reserve area was instead open to fishing. The populations in neighbouring areas do not provide relevant information, since these have been bolstered by the migration from the reserve which is being studied. Hence, in order to assess the impact of a no take zone, it is necessary to produce the fisheries under a set conditions which were prevailing before based on a set of parameters. The natural and fishing mortality rates are two parameters to simulate the population.

There are different methods to calculate natural mortality rates when experimental studies have not been conducted. This study refers to Then et al. (2015) who evaluated 12 estimation approaches for predicting natural mortality. Based on the meta-analysis, the use of the Pauly estimator was recommended by Then et al. (2015), especially when Tmax is not known[7]. This suggests that the issue of maximum size could matter for the analysis[8].

$$M = 4.118K^{0.75}L_{\infty}^{-0.555} \qquad (5.16)$$

The literature offers several methods for the estimation of total fishing mortality. One standard is the Z-Beverton Holt equation (Beverton and Holt, 1956). The Z-Beverton Holt equation looks at the size distribution of the catch relative to the maximum for the species.

$$Z = \frac{K(L_{\infty} - \bar{L})}{\bar{L} - L'} \qquad (5.17)$$

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[7] In order to estimate the natural mortality rates, several methods were also employed such as the 'one-parameter Tmax', Hoenig non linear least squares, Hoenig geometric mean, and several variations of Pauly estimator (see Then et al., 2015).

[8] The difference in the length to infinity according to the above-mentioned authors may influence the results of the study. Hence, to test for the sensitivity of the results, the two estimates of 81 and 48 were used for the calculation of natural mortality rates. The results are respectively 0.25 and 0.30. When inserted in the mathematical equations of the model, the difference in the population is around 8% at the steady state (population at 85000 versus 91000). As the population rises, the gap decreases. For a population of 500000, the gap is 5.5%. Still, since the natural mortality is constant for the two areas (reserve and adjacent), no significant change in the final conclusion of the study is observed.

Where Z is the mortality rate, $\overline{L}$ is the mean length in the catch, L' is the smallest size of fish in the catch samples.

A linear relationship between Z and $T_{\max}$ has been suggested by Beverton (1963). This was improved by Hoenig (1983) who offered two alternative estimation approaches:

$$Z = 4.31T_{\max}^{-1.01} \tag{5.18}$$

$$Z = 6.99T_{\max}^{-1.22} \text{ (geometric)} \tag{5.19}$$

The estimates using the three methods are shown in table 5.1.

Table 5.1: Estimates of total mortality rate

Z-total mortality rate	**Estimates**
Z Hoenig	0.60
Z Hoenig geometric	0.65
Z-Beverton Holt model	0.71

NB: In a model using discreet time the fishing mortality parameter has to lie between 0 and 1. However, in a continuous and instantaneous model the F and stock are adjusting at the same time continuously; in consequence the combination of natural and fishing mortality can exceed 1. In these terms a Z of 1.5 would correspond to an annual Z of 0.77 (see Anderson and Seijo p85).

The Beverton-Holt model, which relates mean length to Z has a known bias associated with the assumption of an infinite lifespan (Nadon et al., 2015). Since our aim is not stock assessment but to compare the status quo with a counterfactual, i.e., the fishery as it is with the fishery as it would be if the Blue Bay MPA had never existed, the simplest method, the Z- Hoenig, is used.

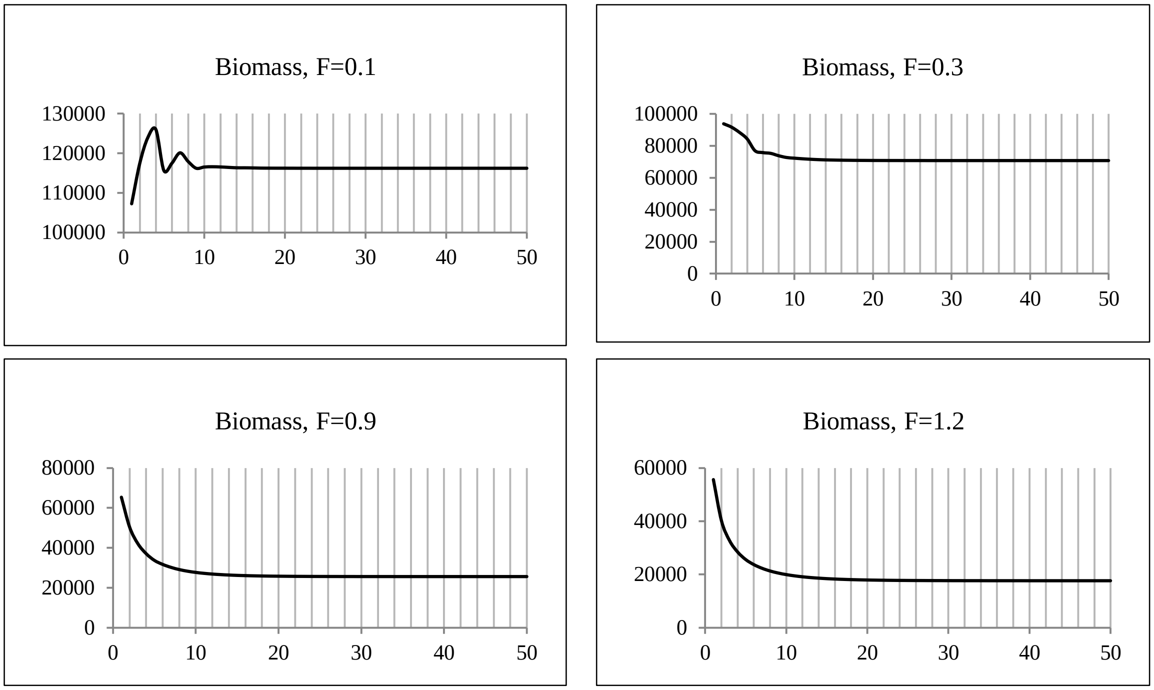
5.4.Results

Fisheries counterfactual reference points in the absence of the MPA

Using the bioeconomic model to simulate the fishery in the absence of the MPA, equation 5.16, the natural mortality rate (M) is estimated at 0.25, while the fishing mortality (calculated using the Z-Hoenig and Z-Hoenig geometric in table 5.1), is taken to be 0.4 in the fishing area adjacent to the MPA..

The analysis begins with the construction of reference points to characterise the fisheries in the absence of an MPA. The population density (fish per km^2) which is estimated from the survey data in the fishing area adjacent to the MPA is simply imposed on a hypothetical area equivalent in size of the MPA. Using the calibrated model from the survey data, figure 5.1 shows the dynamics of the biomass as it approaches the steady state assuming each of four different fishing mortality rates, 0.1, 0.3, 0.9 and 1.2.

Figure 5.3: Biomass at the steady state



The vertical axis and horizontal axis represents biomass (kg) and year respectively

Source: author's calculations

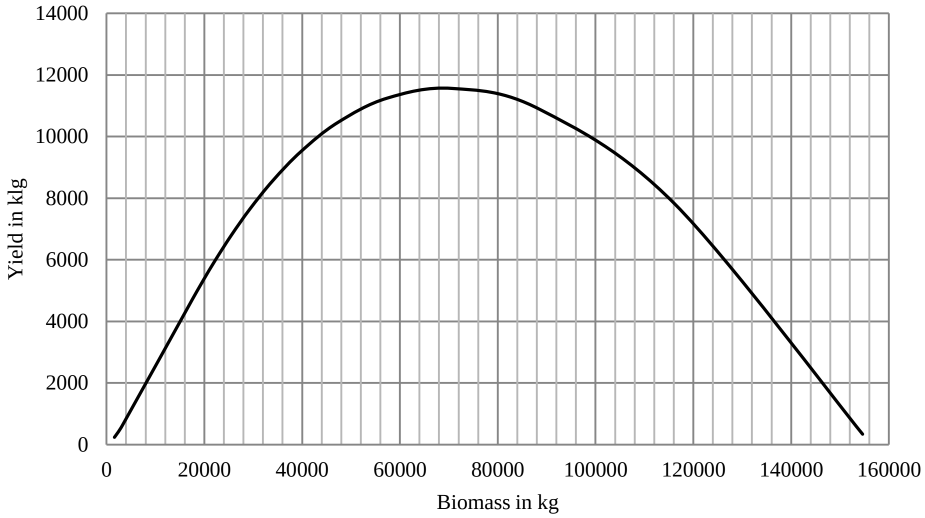
The model is used to estimate population, biomass, yield per effort, catch per trip in number of fish and weight, and yield per recruit at different fishing mortality rates at the equilibrium level (table 5.2). The steady state is achieved over a period of 15 to 20 years but to be prudent these metrics are estimated in the 30th year. As fishing mortality rate increases, population and catch per trip (whether measured as numbers of fish or as mass of fish landed) fall as shown in table 5.2 and figure 5.4. However, the yield per recruit is maximised at a fishing mortality rate of 0.4, the total yield then being approximately 12000kg per annum, from a biomass of 74000kg.

Table 5.2: Fisheries in the absence of MPA

F	Population	Biomass (kg)	Catch per trip number	Catch per trip (g)
0.1	105893	116237	15.0	23324
0.2	92369	88613	11.7	16008
0.3	82354	70740	9.5	11565
0.4	73815	57398	7.8	8459
0.5	66411	47268	6.5	6262
0.6	60247	39826	5.5	4763
0.7	54870	34027	4.7	3681
0.8	49987	29280	4.0	2862
0.9	45769	25544	3.4	2268
1	41870	22364	3	1802
1.2	38451	19773	2.6	1454
1.3	35342	17563	2.3	1180

Source: author's calculations

Figure 5.4: Biomass versus yield



Source: author's calculations

Size of MPA and spillovers to adjacent fishing areas

Increasing the size of marine reserve has two effects: a squeezing effect as existing fishers are squeezed into a smaller area, and a compensation effect as enhanced production of adults and juveniles spills out of the reserve into the adjacent area (Halpern et al*.*, 2004). A longer-distance compensatory effect also occurs as increasing numbers of mature adults in the reserve enhance the export of larvae.

In order to show the two effects of the MPA a sensitivity analysis was conducted. This involved increasing the proportion of the study area from its actual size (30% of the area) to a hypothetical 90%, assuming three different migration rates: 0.1, 0.4 and 0.8, corresponding to low, moderate and rapid migration of fish from the MPA. The area (see map - fig 5.1), can be treated as homogeneous because the same fish species are caught consistently across it

The benefits that a reserve generates depend on the fishing mortality rates in the pre-reserve period - the more heavily fished the area was beforehand, the greater the improvement in the resource. Accordingly the model is tested using two different fishing mortality rates. The first would generate the maximum sustainable yield and uses the optimum fishing mortality rate of 0.3; the other reflects an over-exploited fishery with a fishing mortality rate of 1.2 [9].

The MPA in an over-exploited fishery

In a fishery where there is over-fishing, Beverton and Holt (1957) and Bostford *et al.* (2003) show that increasing the proportion of the area set aside as a no-take zone has the same effect as increasing the age at which fish can first be captured.

Table 5.3 shows the results for a fishing mortality rate of 1.2 in the fishing area without the MPA. The population in the pre-reserve situation stands at 35000, with a catch per trip of 2.28 and 1188kg. With a migration rate of 0.1 (lowest) and an MPA of size 0.1 (10% of the study area), the population rises by 27% compared to the no-reserve scenario, it increases by 200% with a size of 0.9. As the reserve size increases, effort is shifted to a smaller area. The migration of fish is therefore crucial to ensure benefits to fishers.

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[9] A fishing mortality rate of 1.2 might seem unrealistic since it appeared that the fish species would go extinct. However, this figure is used in a continuous time modelling of the fisheries. The relationship between a total mortality rate in continuous time (Z) and discrete time Z' is as follows:

$$Z' = 1 - \exp^{(-Z)}$$

A fishing mortality rate of 1.2 and natural mortality rate of 0.25, it implies that the corresponding total mortality rate in discrete time is 0.77. Therefore, the over-exploited fisheries simply relates to a total mortality rate in discrete time at this level in the subsequent analysis of this study. Similarly, a fishing mortality of 0.3 and natural mortality rate of 0.25 would correspond to a total mortality rate in discrete time of 0.4.

Table 5.3: MPA Size and Performance with F=1.2

MPA as proportion of total area	**Population reserves**	**Population adjacent**	**Total Population**	**Catch per trip adjacent area (number of fish)**	**Catch per trip adjacent area (gram)**
0		**35432**	**35432**	**2.28**	**1188**
Migration rate=0.1					
0.1	9739	35425	45164	2.32	1308
0.2	19726	34536	54262	2.30	1383
0.3	29966	32943	62909	2.23	1425
0.4	40491	30734	71225	2.12	1436
0.5	51357	27939	79297	1.97	1414
0.6	65114	22939	88053	1.63	1200
0.7	79248	17552	96800	1.25	942
0.8	93657	11889	105546	0.85	651
0.9	108271	6022	114293	0.43	336
Migration rate=0.4					
0.1	6844	37681	44525	2.49	1441
0.2	14564	38147	52711	2.59	1619
0.3	23097	37357	60455	2.60	1744
0.4	32475	35532	68006	2.53	1817
0.5	42798	32753	75552	2.39	1834
0.6	56461	27966	84427	2.08	1686
0.7	71434	22172	93606	1.68	1427
0.8	84989	16605	101594	1.27	1123
0.9	104826	8089	112915	0.63	581
Migration rate=0.8					
0.1	5694	38076	43771	2.52	1453
0.2	12541	38935	51476	2.65	1655
0.3	20464	38447	58911	2.69	1806
0.4	29493	36805	66298	2.64	1903
0.5	39977	33831	73808	2.48	1896
0.6	52930	29611	82541	2.23	1838
0.7	67835	23918	91754	1.84	1611
0.8	84474	17059	101533	1.34	1241
0.9	102867	9076	111943	0.73	710

Source: author

The population of fish in numbers is always highest with a low migration rates. This is consistent with Bostford et al.'s (2003) view that MPAs' contribute most to fishery management where target species have low rates of juvenile and adult movement.

The higher the migration rate, the greater is the population in the adjacent area and the higher the catch per trip. Thus, when migration rate is low, (migration rate=0.1), there is a mild increase in catch per trip for a reserve size of 0.1. With a higher migration rate of 0.4 and 0.8 as shown in the table 5.3, catch rate rises slightly until the reserve size reached

50% and 40% of the available area respectively. The findings in this analysis of an over-exploited fisheries are consistent with those of Quinn *et al.* (1993), Holland and Brazee (1996) and Sladek Nowlis and Roberts (1999) that reserves would increase yield only if the population has already been fished beyond the point of maximum yield. Rodwell and Roberts (2004) acknowledge that overexploited stocks require increased area closure to attain target stock levels, and advocate 60% closure for overexploited stocks while Pitcher et al. (2000) show that a reserve of 20% can significantly increase returns in the Hong Kong marine fisheries. Clark (1996) and Lauck et al. (1998) suggest that reserve size needs to be extremely large — 50–90% of total habitat and argue that there is an optimal size of the MPA if fishing yield is to be maximized in the long term. The catch per trip per fisher rises with the reserve size to peak at 1.5kg when d=0.1, 2kg when d=0.4 and 2.2kg when d=0.8. The double dividend hypothesis is clearly visible in an over-exploited fisheries.

The relationship between the size of an MPA and fishing yields nearby may be an inverted-U shape. This non-linearity means that, when the MPA is small (less than 40% of the area), it is beneficial to increase its size because it enhances the fish population available to be caught by fishers. However, if the proportion of a fishing ground set aside as a reserve is too large, reducing it improves access to fishing grounds and increases yields.

Part of the double dividend is explained by the effect of MPA on the size of the fish. In order to investigate this point further, table 5.4 compares the percentage rise in biomass and with percentage rise in population. It is clear from the table that the rise in biomass is more than twice the rise in the population of fish following the establishment of the reserve in an overfished area (with fishing mortality F=1.2).

Table 5.4: Percentage change in population and biomass (F=1.2)

	Migration rate=0.1		Migration rate=0.4		Migration rate=0.8	
MPA as proportion of total area	% rise in population	% rise in biomass	% rise in population	% rise in biomass	% rise in population	% rise in biomass
0.1	27.5	66.6	25.7	49.4	23.5	41.9
0.2	53.1	132.2	48.8	100.0	45.3	86.7
0.3	77.6	198.1	70.6	153.6	66.3	136.0
0.4	101.0	264.6	91.9	211.1	87.1	190.9
0.5	123.8	332.4	113.2	273.8	108.3	252.5
0.6	148.5	419.1	138.3	359.0	133.0	334.1
0.7	173.2	508.4	164.2	453.3	159.0	427.4
0.8	197.9	599.4	186.7	539.5	186.6	533.4
0.9	222.6	691.9	218.7	667.1	215.9	652.5

Source: author

Figure 5.5 shows the age distribution in a fisheries which was over-exploited at a fishing mortality rate of 1.2 in the absence of the MPA. It then compares the results when the MPA of size 0.3 is established in the same fisheries. This is particularly important for long lived resident species, where fecundity increases with individual age. Figure 5.6 shows the age distribution of fish caught when the MPA is established.

Figure 5.5: Population with and without MPA (F=1.2, MPA=0.3)

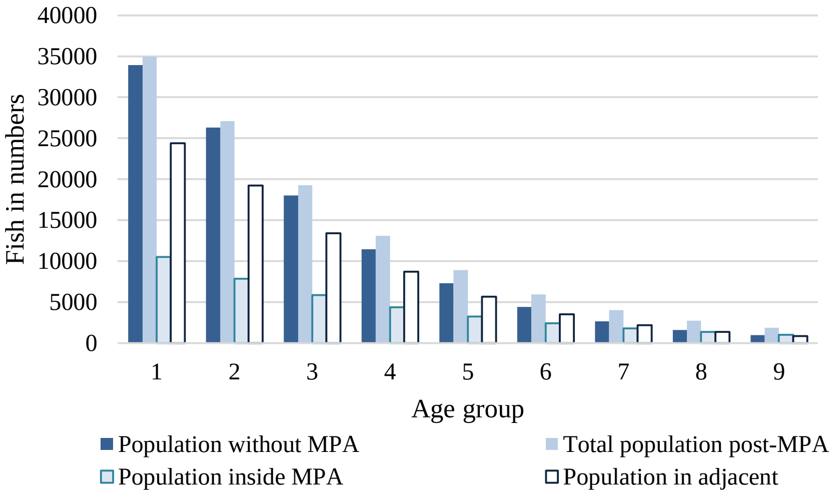
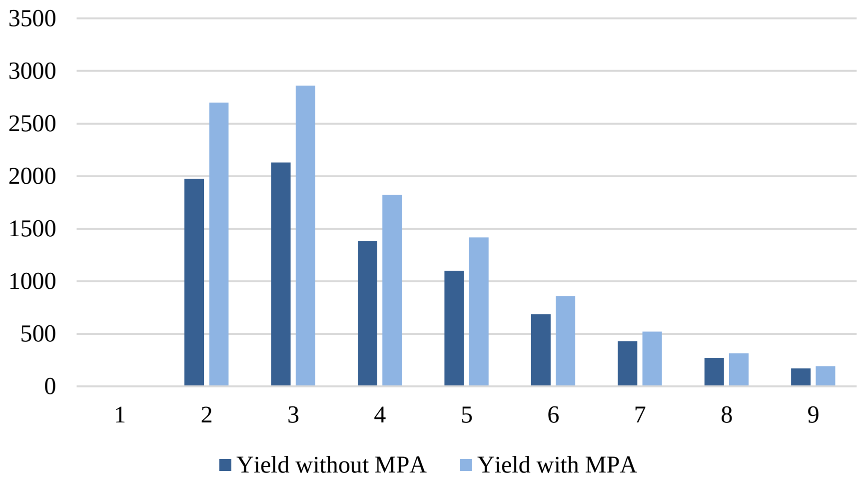


Figure 5.6: Age distribution of yield (F=1.2, MPA=0.3)



The MPA in a well-managed fishery

A different picture would be observed if the effort in the fisheries were already optimal (i.e. E_{MSY}). In such a situation the population would still rise inside the MPA, but catch per trip (in both number and weight of fish) in the fishable area would fall sharply, regardless of the fish migration rate, as the reserve size increases.

An important question relating to marine reserve is why it should be established if the fisheries is at the maximum sustainable yield. In this case, a marine reserve is justified if it can enhance tourism value and recreational activities. As in the case of Mauritius, the conservation effect may benefit local residents as well as tourists. However, there is a trade-off between rising population and fall in catch per trip in a fisheries. The conservation effect comes at a cost to artisanal fishers. In a well-managed fishery, a reserve of any size will reduce fishing yields. This is shown in table 5.5.

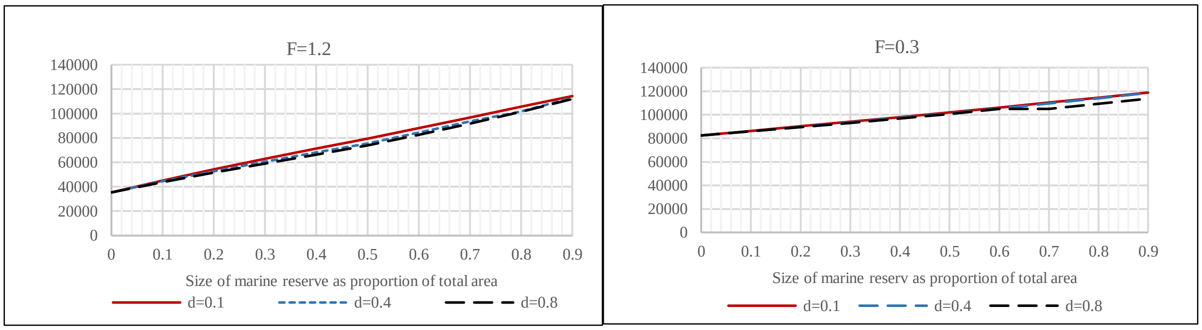
Table 5.5: Performance of the size of the MPA with F=0.3

Size of MPA	**Population in MPA**	**Population in adjacent waters**	**Total**	**Catch per trip adjacent area (number of fish)**	**Catch per trip adjacent area (gram)**
0		**82354**	**82354**	**9.53**	**11565**
Migration rate=0.1					
0.1	11132	75162	86294	8.74	10692
0.2	22345	67882	90227	7.94	9795
0.3	33660	60496	94156	7.12	8872
0.4	45107	52977	98084	6.28	7916
0.5	56728	45288	102016	5.41	6918
0.6	69688	36503	106191	4.37	5617
0.7	82843	27544	110386	3.31	4264
0.8	96142	18454	114596	2.22	2871
0.9	109551	9266	118816	1.12	1447
Migration rate=0.4					
0.1	9855	76192	86047	8.90	10956
0.2	20038	69717	89755	8.22	10272
0.3	30590	62902	93492	7.49	9508
0.4	41566	55711	97277	6.70	8652
0.5	53037	48094	101131	5.85	7688
0.6	65975	39350	105325	4.83	6416
0.7	79504	30120	109624	3.72	5001
0.8	93563	20454	114017	2.54	3453
0.9	108092	10401	118493	1.30	1782
Migration rate=0.8					
0.1	9334	76576	85910	8.96	11050
0.2	19131	70380	89511	8.32	10438
0.3	29434	63740	93174	7.62	9724
0.4	40298	56623	96921	6.85	8893
0.5	51797	48978	100774	5.99	7928
0.6	64585	40352	104937	4.99	6696
0.7	78134	31115	109249	3.88	5287
0.8	92411	21295	113706	2.68	3701
0.9	107390	10916	118306	1.38	1938

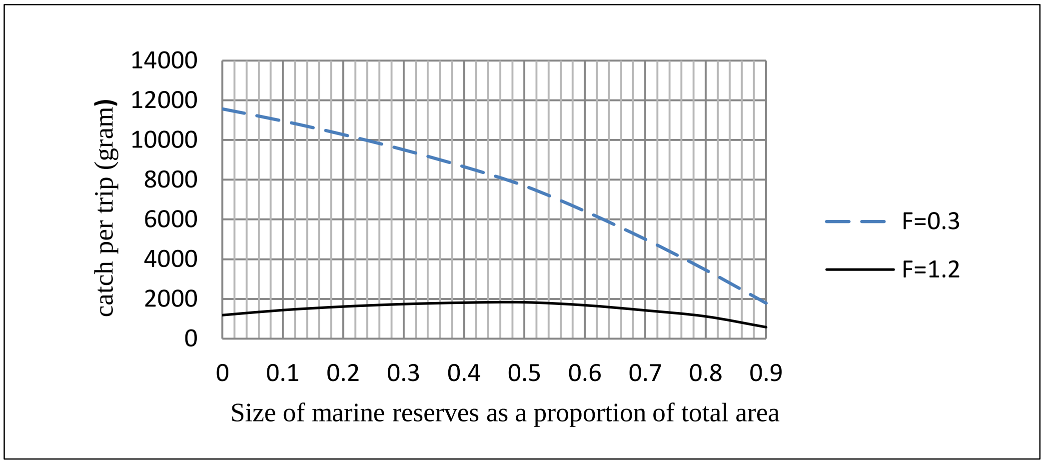
Figure 5.7 compares effects of increasing the reserve size on the biomass and catch per unit effort of unicornfish under two different levels of fishing pressure (fishing mortalities F=1.2 and F=0.3) at migration rate of 0.1, 0.4 and 0.8. At time t=0, there is no reserve i.e. the entire area is a single open access fishery. In such a situation the biomass is 35 000kg, if F=1.2, but a far larger 85 000kg. with F=0.3, As the reserve size increases, population rises in both situations, but significantly more in the over-exploited situation; however, as the reserve size rises there is a convergence of stock outcome for the two pre-reserve conditions, as expected. If the reserve is a really large proportion of the total area, it does not matter what the fishing mortality is in the remaining area since the zone open to fishing

is then too small to make much difference. But when the reserve is only a small proportion of the area, then the intensity with which the resource is fished is a key issue.

Figure 5.7: Effect of MPA in population and catch rates with different fishing mortality rates

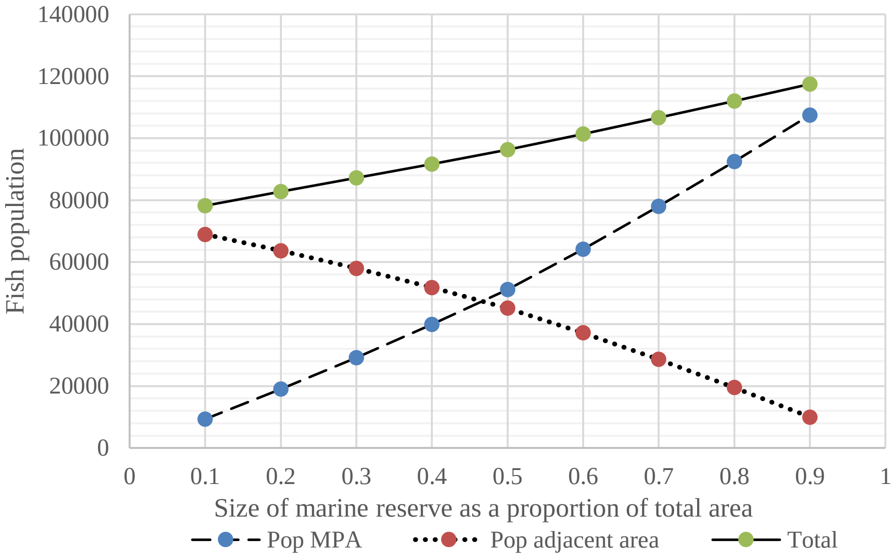


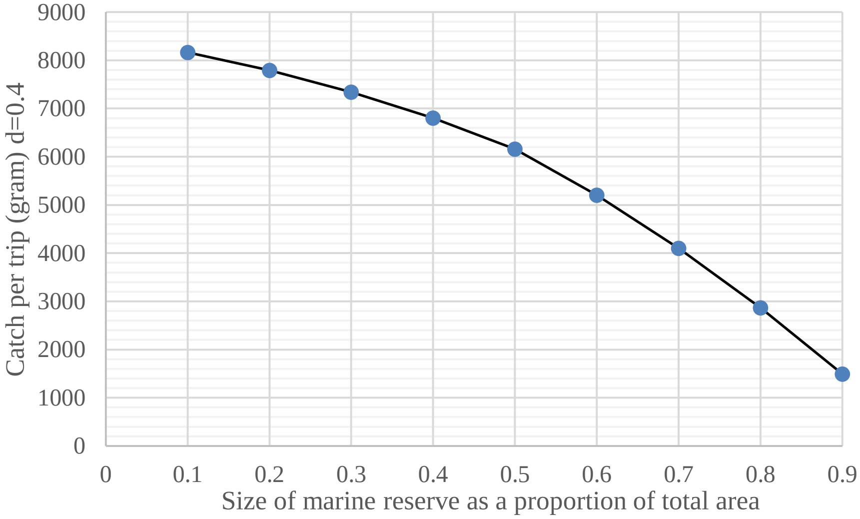
Population on vertical axis



Increasing the size of the Blue Bay Marine Park assuming a migration of 0.4 and fishing mortality stands at 0.4 will decrease catch per trip but will increase the population. The scenarios from increasing the size from 10% to 90% are shown in figure 5.8.

Figure 5.8: Reserve size, population and catch per trip at the adjacent of the Blue Bay Marine Reserve (F=0.4, d=0.4)





The lower panel shows the projected changes in catch per trip if Blue Bay Reserve were to expand.

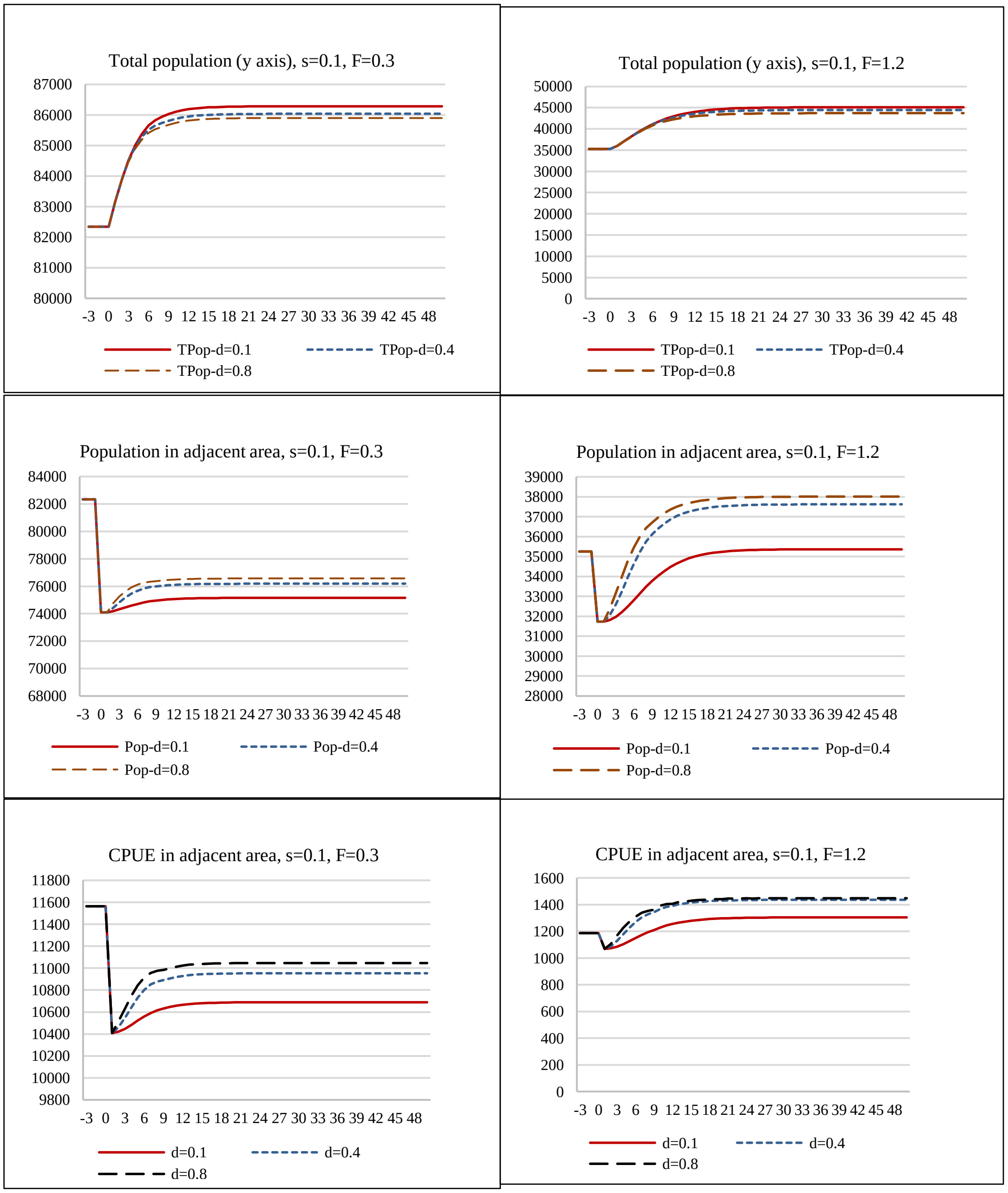
Impacts of MPA on the dynamics of biomass and CPUE towards the steady state

The previous section analysed the long run impacts of reserves. However, following the creation of a reserve, it takes time for a fishery to reach a steady state. This section addressed the dynamics of stocks and other metrics (catch per trip) during that adjustment period. Three reserve sizes are considered, each as a proportion of the total area (assuming the entire area had been previously fished): 0.1 (10% set aside as MPA) (Figure 5.9), 0.3 (30% set aside) (Figure 5.9) and 0.6 (60%) (Figure 5.10). Two scenarios are considered: 1) when effort is at a level closer to E_{MSY} (F=0.3) and 2) when the fisheries are over-exploited (F=1.2). Three transfer rates of fish from the reserve to adjacent waters are shown, corresponding to low, medium and high.

Comparing the short run impacts of the three reserve size, it is observed that it takes more than a decade for the system to approach the steady states. In the first year after the reserve is created, the population in the adjacent area and catch per trip falls for both an over exploited and well managed fisheries. Although an over-exploited fishery may enjoy a long run double dividend, the short-run dynamics suggest that fishers will be worse-off initially. In a well-managed fishery, the population in the adjacent area and catch per trip fall and never return back to the pre-reserve level.

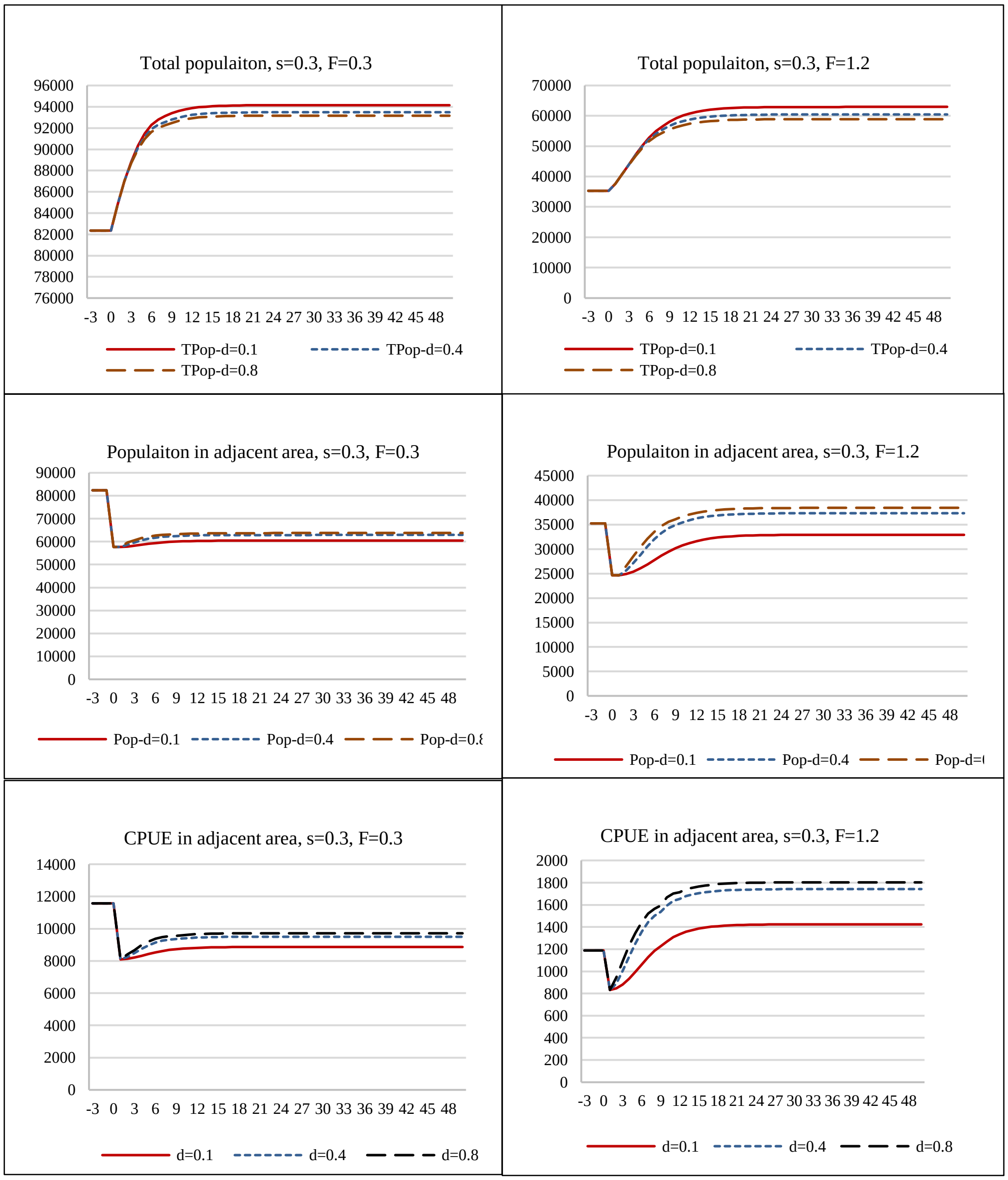
The initial fall in calculated catch per trip should be a serious concern for policy makers, suggesting that additional measures must accompany the establishment of reserve for the well-being of fishers.

Figure 5.9: Dynamics of population and catch per trip after creation of reserve-size 10%



Dynamics with reserve size=0.1 of total area, fishing mortality=0.3 and 1.2, and migration rate=0.1, 0.4, 0.8

Figure 5.10: Dynamics of population and catch per trip after creation of reserve-size 30%


Total populaiton, s=0.3, F=0.3

Total populaiton, s=0.3, F=1.2

Populaiton in adjacent area, s=0.3, F=0.3

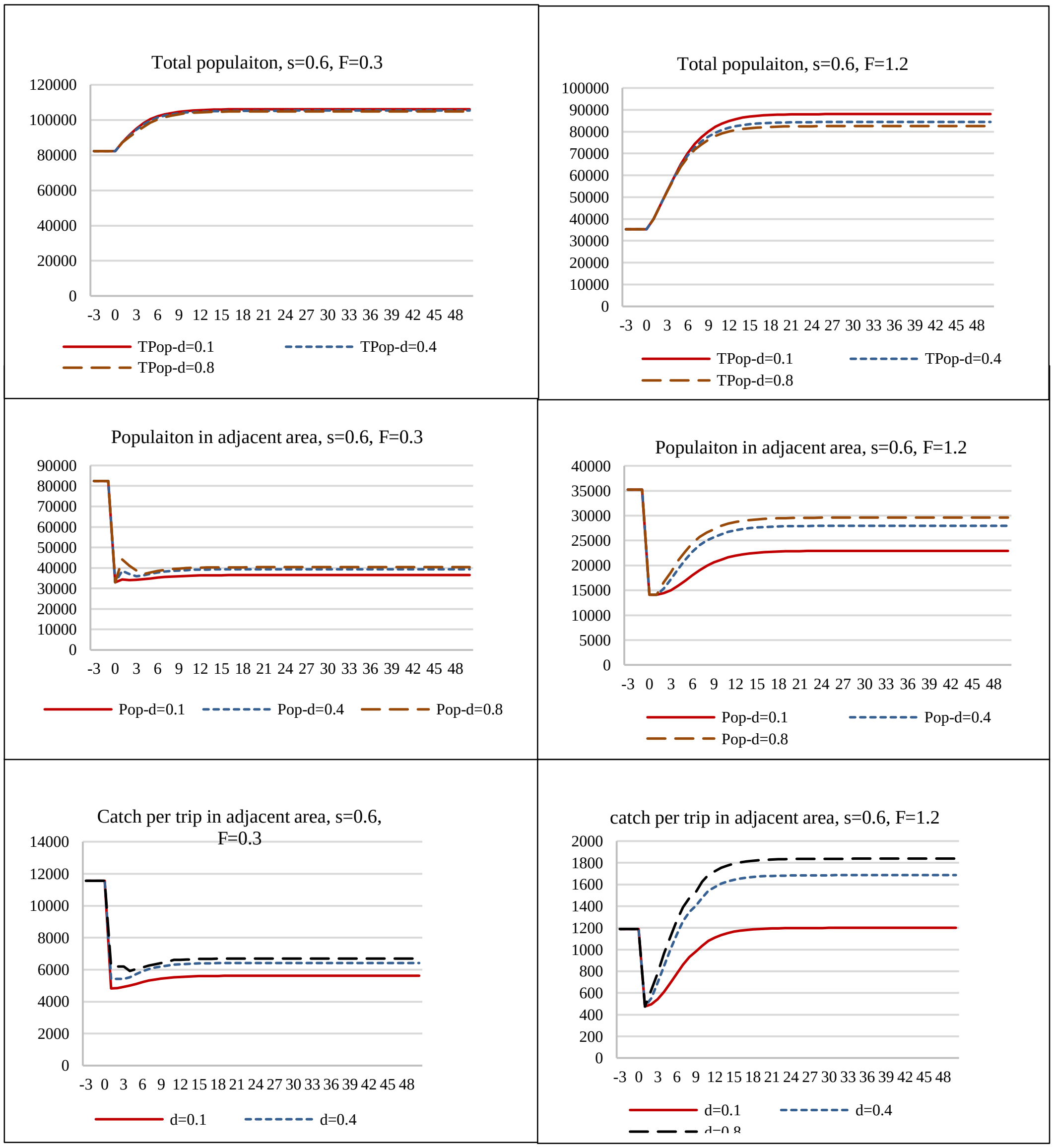
Populaiton in adjacent area, s=0.3, F=1.2

CPUE in adjacent area, s=0.3, F=0.3

CPUE in adjacent area, s=0.3, F=1.2

Dynamics with reserve size=0.3 of total area, fishing mortality=0.3 and 1.2, and migration rate=0.1, 0.4, 0.8

Figure 5.11: Dynamics of population and catch per trip after creation of reserve-size 60%



Dynamics with reserve size=0.6 of total area, fishing mortality=0.3 and 1.2, and migration rate=0.1, 0.4, 0.8

5.5.Discussion and conclusion

Before discussing the findings of the study, it is important to note that the results are conditional on the ecological and biological dynamics of the fishing areas adjacent to the Blue Bay Marine Park, they are in no sense universal.

A main contribution of the study is that it uses case-study data from an existing fisheries with a marine reserve to enlighten the effects of reserve size. There are three major findings from this study. In particular it shows how the population of the MPA at Blue Bay is affected by the pre-MPA conditions, the spillovers of the reserve, and the way they are managed (which contribute to the fishing mortality in unrestricted waters). Second, it shows the role of the size of the reserve in determining the enhancement of the population (conservation effect) and fisheries benefits to fishers in the fisheries, that is, the relationship between reserve size and catch rates (in an over-exploited fishery versus a well-managed fishery). Third, it demonstrates the dynamics of the fisheries benefits as when the reserve is established to the steady-state situations of the fisheries.

Based on the ecological and biological parameters of the fisheries at the Blue Bay Marine Park, the analysis concludes that if the pre-reserve condition of the Blue Bay Marine Park been a managed fishery with effort close to E_{MSY} (or a fishing mortality of F=0.3), the establishment of a reserve would always reduce catch per trip. As the reserve size increases, the catch per trip falls, even though the population of fish rises in the area as a whole. Fishers are thus worse-off although the tourism sector may gain from the greater density of fish in the protected areas.

If the fishery had initially been over-exploited, i.e. with effort sustained at levels well in excess of E_{MSY}, then the model concludes that there is an inverted-U relationship between catch per trip and the proportion of the area set aside as an MPA. The maximum catch per trip is obtained when the reserve covers half of the fishable study area. The fish population rises significantly as the reserve size increases. This information can be used to establish marine reserves in other part of the island where the ecological conditions and the fish species share common characteristics with those in this study.

The study also informs the debate between those advocating a small number of large reserves, and those promoting multiple smaller ones over and above the biological issues. The simulation exercise concludes that, depending on the pre-reserve conditions, with a moderate migration effect, assuming there is no change in fishing morality, it can take up to a decade before the population and catch per trip reach their steady states. It the biological assessment recommends a bigger reserve size, then catch per trip falls significantly as soon as an MPA is established. A smaller reserve size induces a relatively slight decline in catch per trip. This means that a bigger reserve size may leave fishers relatively worse-off in the short run. The cost of a big MPA is to some extent, borne by fishers, in terms of lower catches in the years following the creation of the reserve.

It does not consider the production of eggs or larval settlement outside the MPA, or a predator-prey relationship between fish species. The ability of reserves to increase spawning biomass and the proportion of larger fish in the population is likely to increase reproductive potential and egg production, while the predator-prey relationship may increase the complexity of the analysis.

An important caveat follows. In a multi species fisheries, the relationship between the fish species becomes an important component to determine the overall impact of the marine reserves. The current model can be extended to accommodate such features.

Fishing effort is assumed to have been uniformly distributed across the study area. This is not usually the case, as fishing grounds are heterogeneous and patchy. The heterogeneity between patches, in terms of different mortality rates or varying productivity, may affect the optimal size and location of an MPA. Substitute fishing areas may be unaffected by the biological dynamics of an MPA, but they are not insensitive to fishers' spatial behavior. The aim has been to show how bioeconomic modelling which integrates a fishing effort component that is endogenously determined by the costs and payoffs of the different fishing areas, provides insights into the conservation effects, fisheries benefits, optimal size and economic value of a new MPA.

6. An integrated bioeconomic analysis of a marine protected area with endogenous fishing efforts

Abstract

The study assesses the conservation and fisheries benefits of the Blue Bay Marine Park, located in the coast of the island of Mauritius, and answers an important question - are the higher catch rates in its vicinity a result of spillovers or of changing incentives which alter patterns of fishing effort? Unfortunately there are few records of catches and fishing effort prior to the reserve's establishment. In the absence of a true counterfactual, a bioeconomic model is used to generate a hypothetical one. The area's fish populations are modelled using a dynamic age-structured model with a Beverton-Holt recruitment function, while fishing effort is predicted using a random utility model and random parameters logit estimation. The counterfactual analysis is characterised by two-way feedback loops between fish stocks and the geographic redistribution of fishing effort. A comparison of fish population, biomass, and catch rates in the fisheries, i.e. with and without the Marine Park, demonstrates the extents to which relocation of fishing effort as opposed to spillover has determined the impacts of the Marine Park. The analysis concludes that the extent and magnitude of likely effort redistribution can, and should, be used to inform decisions on the placement and size of marine reserves.

Keywords: marine reserve, spatial effort, bioeconomic model, counterfactual analysis

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6.1. Introduction

Whilst MPAs have often been proposed as means to support inshore ecosystems, many came into existence to protect areas of high amenity value. The Blue Bay Marine Park (BBMP) in Mauritius is one of these MPAs. Being small in size (353ha), it lies in the otherwise heavily fished lagoon area between the shoreline of Mauritius and the Island's fringing reef. In addition to amenity value, such MPAs can offer a range of ecological and economic benefits which include the conservation of biodiversity, increased levels of biomass, improvements in fishery yield, and a buffer against environmental shocks and management failures (Allison et al*.,* 1998; Sumaila, 1998; Hannesson, 1998; Sanchirico et al*.,* 2006). These benefits often depend on a complex interactions between many biological and economic factors (Halpern, 2003). However, many authors (e.g. Salas & Gaertner, 2004; Anderson et al*.,* 2012; Benson & Stephenson, 2017) have commented that fisheries managers and reserve designers have tended to focus on biological and ecological considerations whilst downplaying socio-economic issues.

The biological factors which influence the outcome of an MPA such as the BBMP are multi-fold. They include the growth of fish populations inside the reserve, rates of recruitment, natural and fishing mortality, movement of fish, and dispersal of larvae to other areas (Polachek, 1990). These biological aspects are, however, also influenced by an important human factor, and one which should be considered in the design of MPAs, the spatial behaviour of fishers.

The importance of fishers' behaviour after the establishment of an MPA has long been realized (Wilen, 1979; Smith et al., 2006). However, in many ecological models, it is rarely based on changing economic incentives (Mann et al*.,* 1995) or on utility maximisation (Fenichel et al., 2013; Lee et al*.,* 2017).

In more general terms, a reserve may redistribute effort unevenly if the costs of fishing are higher in some fishing grounds than others, or if some are riskier to fish. These issues are important because, if fish stocks increase inside the MPA and in the adjacent fishing areas, the question arises; are the higher catch rates in the vicinity of the MPA a result of spillovers or of changing incentives which alter patterns of fishing effort.

The aim of the study is to assess the effect of spatial redistribution of fishing effort on the conservation and fisheries benefits of the BBMP. The study constructs a counterfactual analysis to compare several metrics - fish population, biomass, catch per trip and average of size of fish- with and without the Marine Park taking in account fishers' reaction to both situations. In this case study, the fishing grounds available to local fishers are divided into eight contiguous areas, one of which hosts the BBMP. Fishing effort is predicted using a random utility model (RUM) similar to that used by Smith (2005). This is a utility theoretic model of fishers' fishing location choice, estimated through mixed logit - also known as a random parameters logit (RPL). The approach is to model fishers' location choice and predict effort in different fishing areas, with and without BBMP. The BBMP makes access to some areas more difficult and therefore prompts fishers to relocate their efforts. Such relocation may entail changing costs and changing risks (if they move over the reefs into deeper water). In the counterfactual analysis, the effect of spatial distribution of effort with the BBMP is compared to a situation where the BBMP becomes a fishing area.

The counterfactual analysis is made possible through the construction of a bioeconomic model. An advantage of bioeconomic models that incorporate fishers' behaviours is that they can be used to correct for the effects of fishing effort relocation. Thus, an accurate picture of whether the marine reserve benefits fishers can be estimated. The previous chapter used a bioeconomic model with exogenous fishing effort to analyse the configuration of the Blue Bay Marine Park which gives the highest conservation and fisheries benefits. Similarly, this chapter assesses the effectiveness of the Blue Bay Protected Area as fishing effort responds to changing incentive patterns.

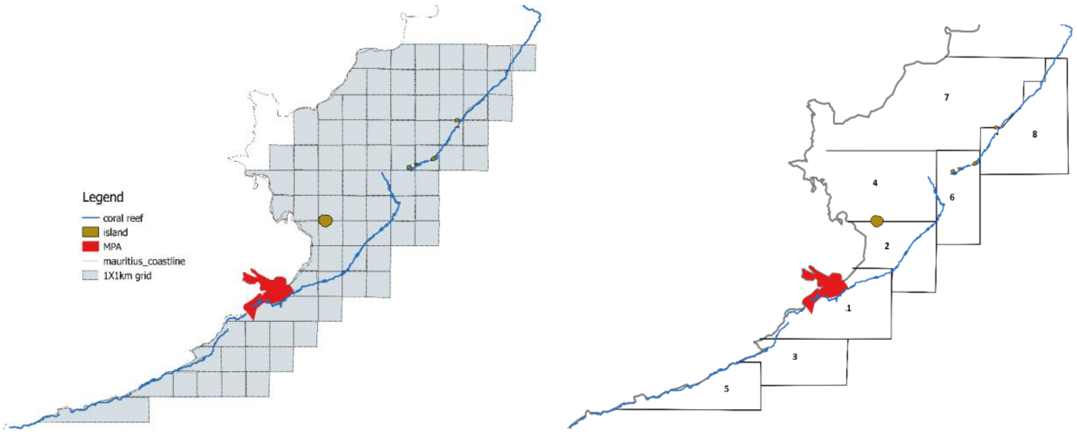
The fish populations are modelled using a dynamic age-structured model characterised by two-way feedback loops between fish stocks and fishers' participation. The bioeconomic simulation follows methods used by Hannesson (1998), Sanchirico and Wilen (2001), Yamakazi et al (2015), and Lee (2017). As in Kahui and Alexander (2008), the bioeconomic model only analyses first round effects.

This paper distinguishes the benefits to the fisheries generated by spillovers from the MPA from those which arise when fishers relocate efforts as a result of changes in incentives driven by the MPA. It also demonstrates how the extent and magnitude of effort redistribution can influence the outcome of MPAs and should therefore be used to inform decisions on the placement and size of marine reserves.

6.2. Material and method

The study site is the Blue Bay Marine Park located in the south-east of the island of Mauritius (figure 6.1). The size of the fisheries as shown in the figure reflects the extent to which artisanal fishers are free to move around. For the purpose of this analysis, the fishing ground is divided into eight areas.

Figure 6.1: Study site – Blue Bay Marine Park



Conceptual framework

The research strategy is to estimate the population and biomass of one fish species (*Naso unicornis*) in each of the fishing area, through a biological sub-model. The estimates include the effects of BBMP on the adjacent fishing area through its adult and juvenile spillover. It then constructs an economic sub-model to estimate and predict the distribution of effort. A link function with fishers' effort is used to connect the two sub-models and this enables the model to show the effects of increase (or decrease) in biological or economic parameters of the fisheries with and without the MPA. Similar approaches can observed made by Smith (2005) for the commercial sea urchin divers in

California, Kahui and Alexander (2008) for the Marine Reserves at Stewart Island New Zealand, and Lee *et al.* (2017) for the groundfish fisheries of the North-eastern United States.

Biological sub-model[10]

The variables and parameters of the biological sub-model are described in tables 6.1 and 6.2. Starting with the population dynamics: the population of fish in age class a in fishing area i for i=1 to 8 at time t is modelled by equation $N_{a,i,t}$ (v1, v2 in table 6.1, and e1,e2 in table 6.2). The population decays exponentially and is partitioned into instantaneous, natural and fishing mortality rates (v8, v9 in table 6.1 and e9,e10 in table 6.2). Assuming fishers are selective, the rate of fishing mortality is dictated by variable v11.

The dynamics of the total population inside the MPA, i.e. with no fishing mortality in the equation, are provided in e2. The population in the adjacent area is augmented via a spillover function v16, characterising the transfer of adult fish and juveniles from the MPA. It is governed by e5, which captures the gradient of the fish population between the MPA and the adjacent area (fishing area 1 in figure 6.1), and the transfer parameter, d. The last mentioned reflects the speed of the population density adjustment between the two areas and can range between 0 and 1. A transfer parameter of 1 indicates that the population density equalises completely within a year and represents the extreme transfer rate, while a figure of 0.5, for example, would represent a 50% adjustment between population densities in the course of a year.

The recruitment function follows the Beverton and Holt formulation and is shown in e4. The recruitment function is based on the biomass and recruitment in an unfished population. There was no information available in the Mauritian context and hence, it was assumed in all the fishing areas that the biomass in the unfished population was twice the current biomass of the fisheries. Consequently, the biomass per recruit has been adjusted so that it calibrates the fisheries according to the catch data. In the absence of data for

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[10] The biological sub-model is detailed in chapter 5.

steepness, h, a figure of 0.75 has been adopted. This is commonly used by fisheries scientists when steepness data are not available[11].

Estimates of population, biomass, catch per trip and average size of individual fish are used as the metrics of analysis. Catch data by weight was collected from artisanal fishers, and using the weight-length relationship (e7) and the von Bertalanffy equation (e8), the age of the fish and the number in each age class were estimated. Using the fishing mortality and natural mortality rates, the fish population in each area from 1 to 8 was estimated as shown in figure 6.1. A representative fish distribution for the MPA was created by extrapolating the data on the fish species in area 1 and assuming a zero fishing mortality rate. The transfer rate of 0.4 was derived from the analysis in the previous chapter.

The estimates on the population and biomass are based on the parameters of the model. In Mauritius, there has only been limited scientific study of these parameters. Stock assessment of the lagoon fisheries has been practically non-existent. Certain parameters have simply been assumed; these include the biomass, recruitment levels in the unfished population, and the Beverton-Holt steepness. The current biomass estimated from the catch data is assumed to be 50% of the unfished biomass; this is clearly optimistic, and is only done to provide initial estimates for the simulation. A sensitivity analysis was undertaken by varying these parameters. Applying the same parameters to other fishing areas did not change the results of the study. Nevertheless the estimates may not necessarily reflect a realistic stock level at these fishing areas since this simulation exercise was not to assess stock of fish *per se*.

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[11] Professor D. Butterworth, University of Cape Town (personal communication, 2017).

Table 6.1: Variables and parameters in the biological system

Parameters	Descriptions	Reference
$N_{a,i,t}$	Population of age a class in fishing area i for i =1 to 8 at time t, area 1 is the adjacent to the MPA	v1
$N_{a,MR,i}$	Population in marine reserve of age a class at time t	v2
$n_{MR,i}$	Population density in marine reserve at time t $n_{MR,i} = \frac{\sum_1^A N_{a,MR,t}}{\tau K}$	v3
$n_{1,t}$	Population density in adjacent area to MPA (area 1) $n_{1,t} = \frac{\sum_1^A N_{a,1,t}}{(1-\tau)K}$	v4
K	Total fishing area with MPA, excluding area 2 to 8.	v5
τ	Proportion of K attributed to MPA	v6
d	Transfer rate of adult and juvenile fish	v7
M, M_a	Natural mortality rate of age a class, assumed constant at M	v8
F_a, F	Fishing mortality rate of age a, assumed to be constant at F	v9
Sel_a	Selectivity at age a	v10
$\bar{F}_a$	$F_a \times Sel_a$	v11
R_t	Recruitment at time t	v12
SBB	Spawning biomass	v13
$SPR_{F=0}$	Biomass per recruit at F=0	v14
h	'Steepness' of the recruitment function	v15
TR_a	Transfer of adult and juvenile at age a to adjacent area 1, total transfer is $TR = \sum_1^A TR_a$	v16
w	Weight of fish	v17
L	Length of fish	v18
L_∞	Length to infinity	v19
$\bar{L}$	Mean length in the catch	v20
L'	Smallest size of fish in the catch samples	v21
k	Growth coefficient of Von Bertalanffy growth equation	v22
t_0	Theoretical age at zero length	v23

Table 6.2: Biological system equations

Description	Equation	Reference	Data source
Population dynamics of age a class in area i	$N_{a,i,t} = N_{a-1,i,t-1} e^{-M_{a-1,t-1} - \bar{F}_{a-1,t-1}}$ $N_{i,t} = \sum_1^A N_{a,t,t}$	e1	Estimate from survey
Population dynamics in MPA of age a class	$N_{MR,t} = \sum_1^A N_{a-1,MR,t-1} e^{-M_{a-1,t-1}} - TR_t$	e2	Estimate from survey
Population dynamic in the adjacent area	$N_{1,t} = \sum_1^A N_{a-1,1,t-1} e^{-M_{a-1,t-1} - \bar{F}_{a-1,t-1}} + TR_t$	e3	Estimate from survey
Recruitment function	$R(t) = \frac{SBB}{\alpha + (\beta \times SBB)}$ $\alpha = \frac{(1-h)SPR_{F=0}}{4h}$ $\beta = \frac{5h-1}{4hR_0}$	e4	'Guesstimates' from marine science (see section 6.2)
Transfer of adult and juvenile fish, TR	$d\lfloor (n_{MR,t} - n_{1,t}) \times \tau K \rfloor$ if $\tau \le 0.5$ $d\lfloor (n_{MR,t} - n_{1,t}) \times (1-\tau) K \rfloor$ if $\tau > 0.5$	e5	Author's calculation
Catch equation	$C_{i,t} = \sum_2^A C_{a,i,t} = \frac{F}{M+F}(1-e^{-M-F}) \times N_{a,i,t}$	e6	Author's calculation
Weight-length relationship	$w = 0.032L^{2.82}$	e7	Source: Rathacharen et al. (1998) and Edwards et al. (2011)
Von Bertalanffy growth equation	$t = t_0 - \frac{1}{k}\ln(\frac{L_\infty - L_t}{L_\infty})$	e8	Rathacharen *et al.* (1998)
Natural mortality rate from the Pauly estimator	$M = 4.118k^{0.75}L_\infty^{-0.555}$	e9	Then et al. (2015) updated parameter estimates
Total fishing mortality	$Z = 4.31T_{\max}^{-1.01}$	e10	Author's estimate from survey data

Economic sub-model: fishers' decision o*n* where to fish

The conceptual framework is based on a two-stage estimation method which is built on the foundation provided by a random utility model. The random utility model assumes that when a fisher j makes a choice to fish in a location i from a finite set of I mutually exclusive fishing alternatives $[I = 1,\ldots,i]$, her maximum utility (U_{ji}) has two components: observed (V_{ji}) and unobserved (ε_{ji}). In additive form, this is represented as follows:

$$U_{ji} = V_{ji} + \varepsilon_{ji} \tag{6.1}$$

The observed (deterministic) component can be specified as a function of measured attributes of the individual and/or alternatives, and can be taken as an expected utility.

The unobserved component reflects the uncertainty regarding the choice and may include measurement errors and functional misspecification. The lack of information on ε_{ji} compels the analyst to treat that part of utility as a random variable with a zero mean.

When faced with alternatives l and i, the fisher will choose alternative i if and only if his utility $U_{ji} > U_{jl}$ for $l \neq i$. Technically, this is equivalent to $V_{ji} - V_{jl} > \varepsilon_{ji} - \varepsilon_{jl}$ for $\forall\ i \neq l$. The probability that choice i is made can then be represented by a probability as follows:

$$P_{ji} = P(V_{ji} - V_{jl} > \varepsilon_{ji} - \varepsilon_{jl}) \qquad \forall\ i \neq l \tag{6.2}$$

Since ε_{ji} and ε_{jl} are random variables, the difference between them is also a random variable. Hence, the probability that the fisher will choose alternative i can be written as:

$$P(Y_j = j \mid X_{ji}, W_j) = \frac{\mathrm{e}^{V_{ji}}}{\sum_{i=1}^{I} \mathrm{e}^{V_{ji}}} \tag{6.3}$$

Assuming that V_{ji} has an additively separable linear form with exogenous variables and parameters, it can be specified as follows:

$$V_{ji} = \theta f(X_{ji}, W_i) = X_{ji}\beta + W_j \alpha_i \tag{6.4}$$

where X_{ji} represents the variables which vary across choices, W_j is a set of characteristics of the individual or factors whose values are unaffected by the choice a fisher makes, and θ, β and α_i are the vectors of coefficients providing information on their marginal utilities with respect to the relevant characteristics.

The link function: connecting the economic and biological system

Before integrating the economic with the biological sub component, the change in the number of trips predicted by the econometric model (described in the next section) in each area has to be translated into a change in the predicted fishing mortality rate (*F*). The link function, similar to that used by Smith and Wilen (2003;2004) is:

$$F_i(t) = q_i \times Trips_i \tag{6.5}$$

q_i is the catchability coefficient in area i. The fishing mortality rate thus depends on the effort expended, the number of fish present, and the participation of the fisherman.

The revenue from a harvest will be an injection into the local economy's circular flow. Just as fish left in the sea continue to multiply, so such an economic injection will go through subsequent rounds in a multiplier process. However, the convention in cost-benefit analysis focuses on 'first round' impacts only, as in Kahui and Alexander (2008). The main reason is that subsequent multiplier effects will depend on the propensity to withdraw from the circular flow, which is often influenced by exogenous events.

6.3. Operationalising the economic system

Estimation method: the Random parameters logit model

The random parameters logit model (RPL) is used to estimate probability that a fisher *j* will choose fishing area *i*. A detailed explanation of RPL can be found in Train (2009). When data consists of both choice- and individual-specific attributes, the conditional and multinomial logit can be combined in a single random parameter or mixed logit model, estimated by modifying the conditional logit model. Hence, the probability (P_j) that individual *j* will choose alternative *i* is estimated using:

$$P_j = \int L_j(\theta') f(\theta') d\theta' \qquad (6.6)$$

L_j is the logit function for parameters, θ' and $f(.)$ is the density function of the parameters θ'.The RPL generalises the conditional logit model by allowing the estimated coefficient θ' to be random rather than fixed. A further advantage of the mixed model is that it avoids specification error caused by the omission of relevant variables. The model's overall explanatory power is given by the likelihood ratio statistic which is similar to the R^2 statistic in an ordinary least squares application (Pradhan & Leung, 2004).

Data[12]

A questionnaire was administered to 100 fishers in connection with each of 10 fishing trips which they undertook in the course of a year from January to December. For each trip, fishers were asked to show on the map the locations of their traps as well as the routes taken to these sites. The information was recorded on a map and has been entered using QGIS software. The species distribution of the catch was also collected using the weight of each fish species; and was similarly recorded.

Location-specific and individual-specific characteristics

The fishermen's choices - whether or not to go fishing, and where to do so, depend on a series of factors. For this study, site-specific factors include distance travelled (DIS_{ji}), depth of the fishing area, $(DEPTH_i)$, whether the fishing area is located in or off the lagoon $(LAGOON_i)$, and most importantly the expected abundance of *Naso unicornis*, measured in terms of expected catch per trip of the fish species, $(ECATN_{ji})$. In order to control for abundance of other frequently targeted fish species, expected catches of two other species are included as variables $ECATB_{ji}$ and $ECATC_{ji}$.

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[12] Details on the survey are provided in chapter 3 and 4.

An important area-specific variable is the distance travelled from the home port to the fishing locations (Campbell & Hand, 1999). As details of fuel expenditure per trip are rarely available, distance has traditionally been used as a proxy for fuel cost (Holland & Sutinen, 1999; Wilen et al*.,* 2002; Smith, 2005; Anderson et al*.,* 2012). Accordingly, data on home port was collected for each fisher participating in the survey and the distance between the home port and the fishing area was estimated. Since fishers may take different routes to their chosen locations, care was taken to measure the distance via the route each fisher had indicated. These routes were not linear extrapolations from home port too fishing area, the paths being based on local ocean geography (currents, depth, etc). The RPL model requires data, not just on the trips that each fisherman did make, but also on the distances to other locations that were available to them but were not selected. Such information was proxied by using actual routes taken to these locations by other fishermen from the same home port

In order to control for individual characteristics, the number of basket traps used by the fisher during the fishing trip in area, $i, NBAS_{ji}$, is taken as a covariate in the estimation. The construction of this individual specific attribute follows Hoffman and Duncan (1988). Consider an attribute, W_j, of fisher, j, which is invariant across choices (e.g. age, boat size, etc). If area 1 is the omitted variable, then if $DUM_2, DUM_3, ..., DUM_8$ are dummy variables for areas 2 to 8 respectively, the attribute enters in the mixed logit model as, $W_j DUM_2, W_j DUM_3, ..., W_j DUM_8$, respectively for areas 2 to 8. The coefficient for each attribute then shows its effect on the likelihood that a location will be chosen, relative to area 1 (the omitted category).

Finally, since the measure of success is the catch of preferred species, the standard deviation of the catch of *Naso unicornis* per trip is used as a measure of risk.

6.4. Results

Fisheries with BBMP

The construction of total catch by age group (table 6.3) was based on a total of 36500 trips made by fishers during the year 2015.

Table 6.3. Catch from the survey of artisanal fishers

Age	R1	R2	R3	R4	R5	R6	R7	R8
0	0	0	0	0	0	0	0	0
1	0	0	0	0	0	0	0	0
2	673	1383	364	994	1056	9755	3516	2657
3	1347	1554	819	2186	2512	7804	1005	4765
4	1616	1776	1092	1987	2588	5853	603	6064
5	4040	2887	2912	994	8052	4878	502	12995
6	673	244	455	0	1438	293	0	2166
7	135	178	100	0	288	195	0	433
8	0	111	0	0	58	0	0	0
9	0	44	0	0	58	0	0	0

Source: Survey

Table 6.4. Fisheries with BBMP

	Fishing mortality rate Z-Hoenig	Population (000)	Biomass (tonnes)	Average size of individual fish
BBMP	0.25	65.4	63.2	968
Area 1	0.66	95.0	86.2	907
Area 2	0.59	262.7	194.5	741
Area 3	0.89	62.8	44.4	707
Area 4	0.85	396.5	291.5	735
Area 5	0.54	364.2	296.7	815
Area 6	0.74	517.1	408.0	789
Area 7	0.76	194.3	148.5	764
Area 8	0.69	403.5	315.5	782
Total	-	2361.5	63.3	-

In order to calculate the fish population (equation e1) in the eight fishing areas, estimates of fishing mortality and natural mortality rates were required. The Z-Hoenig (1983) was adopted (equation e10) and the results are shown in table 6.4. The parameters for the equation were obtained from the survey data. The natural mortality rate (0.25) was estimated using the Pauly estimator (equation e9) as was recommended by Then et al. (2015). The average size of an individual fish is relatively higher inside the BBMP and in the adjacent area.

Modelling effort distribution with the BBMP: random parameters logit estimates

A summary statistics of the choice- or area-specific attributes of fisheries for the RPL estimation is shown in table 6.5.

Table 6.5: Variables and definitions

Variables	Definition	Mean	Standard deviation	Minimum	Maximum
$ECATN_{ji}$	Expected catch of *Naso unicornis* per trip for fisher j (g)	2089.97	1375.34	0	8855
$ECATB_{ji}$	Expected catch of *Lethrinus mahsena* per trip for fisher j (g)	1200.3	600	0	6000
$ECATC_{ji}$	Expected catch of *Caranx sp* per trip for fisher j (g)	1500	850	0	5500
$NBAS_{ji}$	Number of basket traps for fisher j	8.187	1.70	1	13
DIS_{ji}	Distance travel of fisher j	6.79	2.72	1	17.8
$DEPTH_i$	Depth in area *i* (meters)	12.09	11.012	1	35
$LAGOON_i$	Dummy variable: 1 if fishing area in located in the lagoon; 0 otherwise			0	1
$SDCATN_i$	Standard Deviation of revenue in area *i* using the JP estimation	412.07	197.78	86.99	1216.89

In table 6.6, the econometric estimation shows that the higher the expected catch rates of the three fish species in a particular fishing area, the greater the likelihood that the latter will be chosen. Also as expected, the coefficient of the variable standard deviation of catch rates is negative, i.e. fishers are risk-averse and respond negatively to the variance in the catch. Depth and area status (lagoon versus off-lagoon) and distance from home port are all factors influencing fishing location choice. The standard deviation of catch rates and distance are treated as random parameters. In column 2, fisher specific characteristics are introduced, i.e., the number of basket traps. Both positive coefficients and negative coefficients are observed. The Chi-square statistics and the log-likelihood show improvements of model 2 over model 1. Hence, model 2 is used for further analysis.

The current distribution of effort across the eight fishing areas, and the simulated ones from the RPL estimates are shown in table 6.7. Area 1 is adjacent to the MPA, and areas 2 and 3 adjoin area 1. Area 8 is far from the MPA, and is located outside the lagoon. Area 4 has the lowest catch per trip for *Naso unicornis*, but at the same time has the highest level of fishing effort. Located in the lagoon, the distance to it is shorter, making it is more accessible to the fishers. The right-hand column shows the distribution according to the RPL econometric estimates. With the exception of areas 4 and 5 which are marginally different, all other estimates accord fairly well with the information collected in the surveys.

Table 6.6: Random parameters logit regression estimates

Variables	Coefficient (SE)	Coefficient (SE)
	Model 1	Model 2
$ECATN_{ij}$	0.170(0.080)**	0.719(0.294)**
$ECATB_{ij}$	0.012(0.033)***	0.316(0.131)**
$ECATC_{ij}$	0.022(0.030)***	0.390(0.138)***
$DEPTH_{j}$	-0.127(0.033)***	-0.310(0.122)**
$LAGOON_{j}$	-0.835(0.140)***	1.304(0.966)**
$R2 \times NBAS_{it}$		-0.056(0.117)
$R3 \times NBAS_{it}$		-0.083(-0.083)
$R4 \times NBAS_{it}$		0.349(0.349)**
$R5 \times NBAS_{it}$		-0.264(-0.264)***
$R6 \times NBAS_{it}$		0.001(0.001)
$R7 \times NBAS_{it}$		0.021(0.022)
$R8 \times NBAS_{it}$		-0.200(0.200)**
Random Parameter		
$SDCATN_{ij}$	-0.116(0.112)	-0.707(0.345)**
DIS_{ij}	-0.320(0.056)***	-0.290(0.580)***
SD of $SDCATN_{ij}$	0.873(0.094)***	0.912(0.93)***
SD of DIS_{ij}	0.493(0.072)***	0.526(0.059)***
Number of observations	8000	8000
LR Chi2(1)	344.36	336.58
Prob > Chi square	0.000	0.000
Log likelihood	-1535.10	-1493.30
SE=Standard error; ***=significant at 1%, **=significant at 5% and *=significant at 10%		

Table 6.7: Summary data from catch and effort distribution from the RPL

	Expected catch rate per trip	Distribution of fishing from survey	Distribution of fishing effort from RPL
Area 1	13.76	0.04	0.038
Area 2	13.54	0.08	0.083
Area 3	13.61	0.02	0.020
Area 4	7.24	0.29	0.261
Area 5	14.4	0.08	0.103
Area 6	9.44	0.24	0.237
Area 7	6.54	0.13	0.135
Area 8	14.41	0.11	0.123

Fisheries without the BBMP

Relationship between fishing effort and expected catch per trip from the RPL estimates

What would be the population, biomass, average size of individual fish in the absence of the BBMP? The answer to the question would depend on a set of specific conditions characterising the fisheries without the BBMP which would determine the number of fishers and trips made to the fishing area.

With the creation and/or expansion of the MPA, there would be a change in the area available to the fishers, and in the proportion of the stock available for harvest. In the absence of the Park, since area 1 is bigger, this would drive up the 'expected' catch per trip. Over time, more fishers would be likely to enter the fishing area. It is therefore important to predict the number of trips that would have been made as expected catch per trip increased.

The probability of a fisher to choose a fishing area 1 at a given level of expected catch per trip can be estimated from the coefficients of the covariates from the RPL[13] . In figure 6.2,

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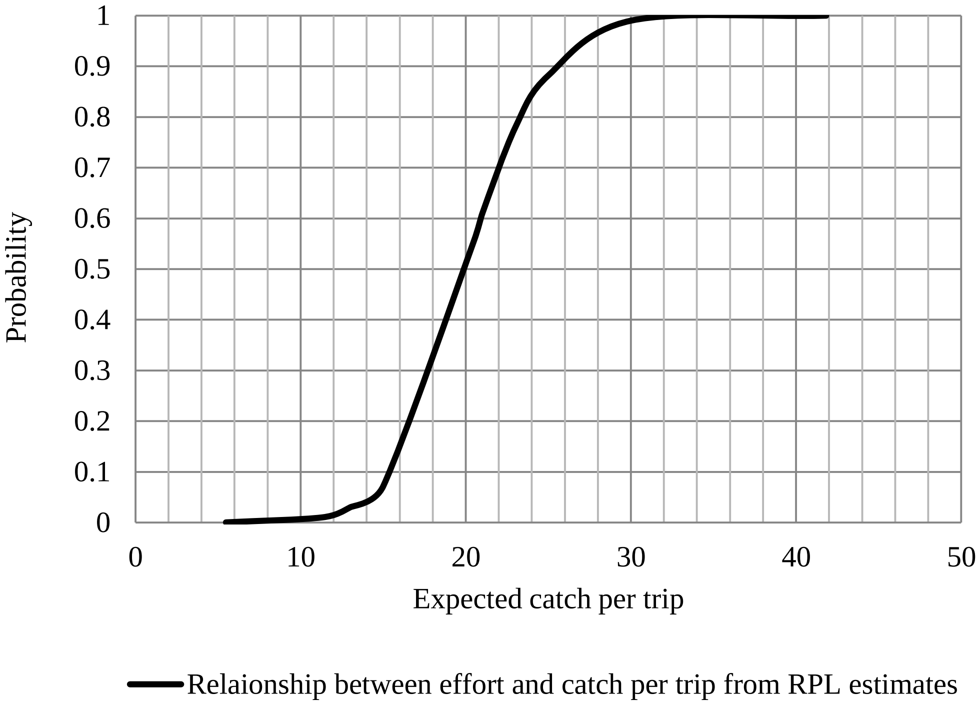
[13] The simulation makes use of the Stata program *mixlpred* developed by Hole (2007) .

when the expected catch per trip is around to 5 to 10 kg, the probability that fishers will choose area 1 is very low, approximating zero. However, as expected catch per trip rises, the probability rises and approaches one when the expectation reaches 30kg. At this level, the area becomes most profitable relative to any other areas in the fisheries.

Figure 6.2. Relationship between expected catch per trip and probability that fishing area 1 is chosen with the BBMP

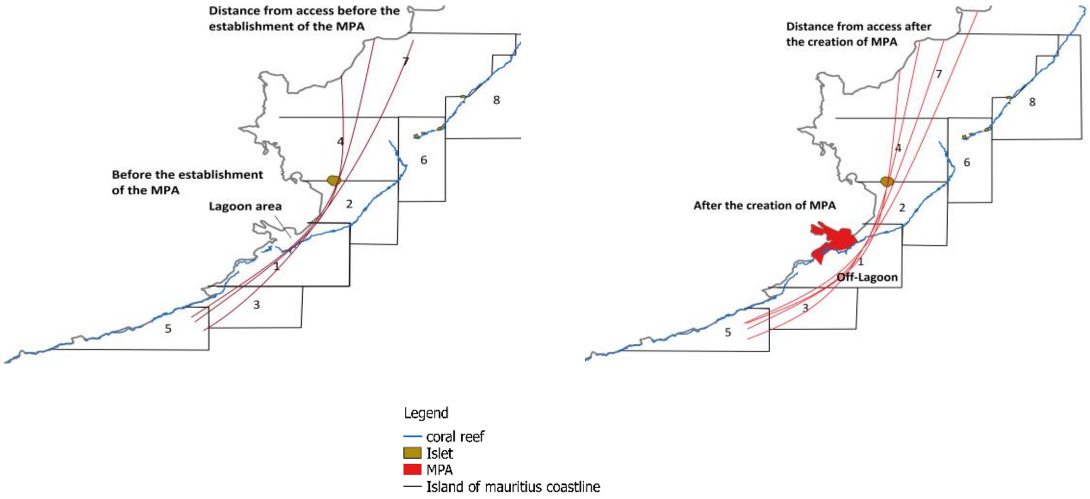


The probability that fishing area 1 is chosen as expected catch rises will be different without the BBMP, because the altered fisheries conditions affecting fishers' behaviour. In order to estimate such probability at each level of expected catch per trip, there was a need to identify carefully those fisheries conditions which would be different in the absence of the BBMP.

The BBMP is located inside the lagoon area and covers the lagoon completely. It therefore makes the fishing in area 1 completely off-lagoon. Without the BBMP, fishing can be undertaken inside the lagoon. Consequently, the depth of the water would also be lower at around 8 meters compared to almost 20 meters in the off-lagoon area. Fishers face a potentially more dangerous environment in the off-lagoon area and hence, in the absence of the Park, fishers face a lower physical risk. According to Smith and Wilen (2005), fishers are attracted to a fishing area when the potential physical risk is relatively lower.

Fishers who used to travel to the adjacent area because of the BBMP (fishing area 1), may now shift inside the lagoon at a shorter distance. Thus, distance of those fishers travelling to fishing area 1 reduces marginally. It is assumed that it decreases by 1km. A graphical illustration is shown in figure 6.3.

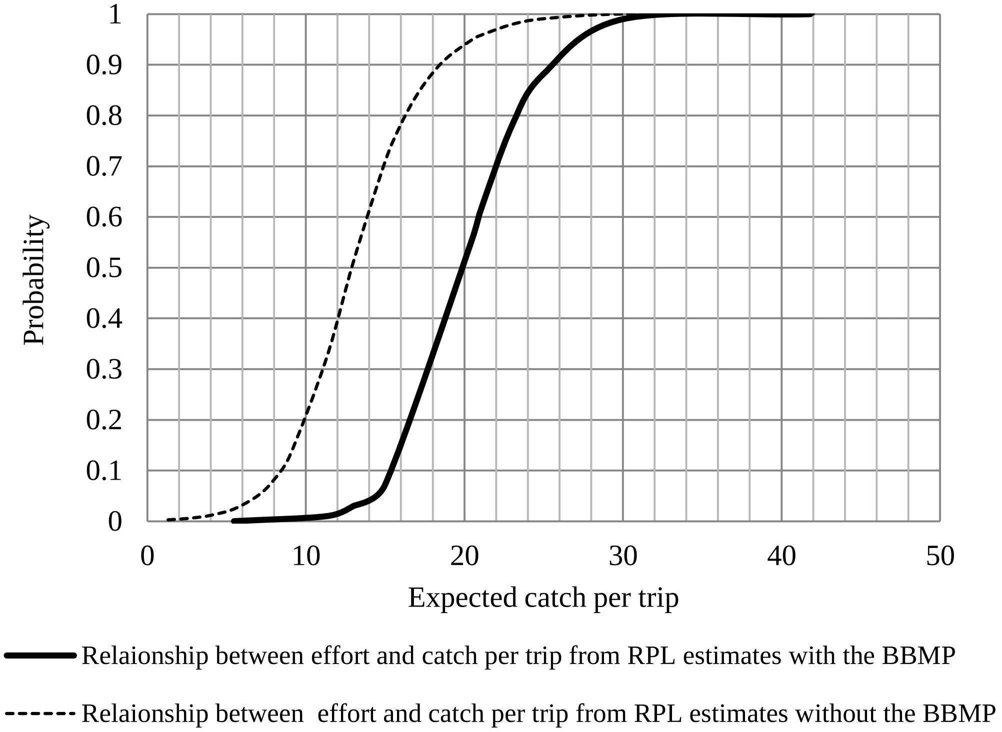
Figure 6.3: Distance from access – before and after BBMP



Source: author from QGIS 2.28.26

As shown in figure 6.4, the probability to choose fishing area 1 without the BBMP is higher compared to the fisheries with the BBMP.

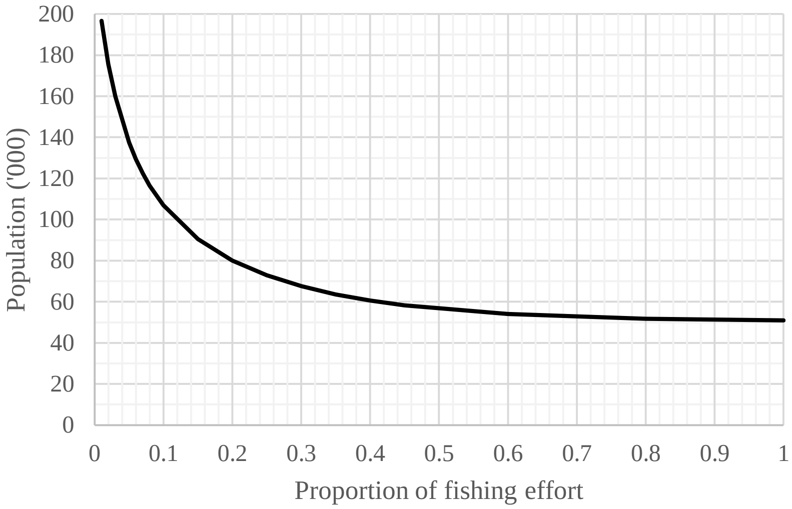
Figure 6.4. Probability that fishing area 1 is chosen and expected catch per trip with and without the BBMP



Relationship between fishing effort and catch per trip from the biological sub-model

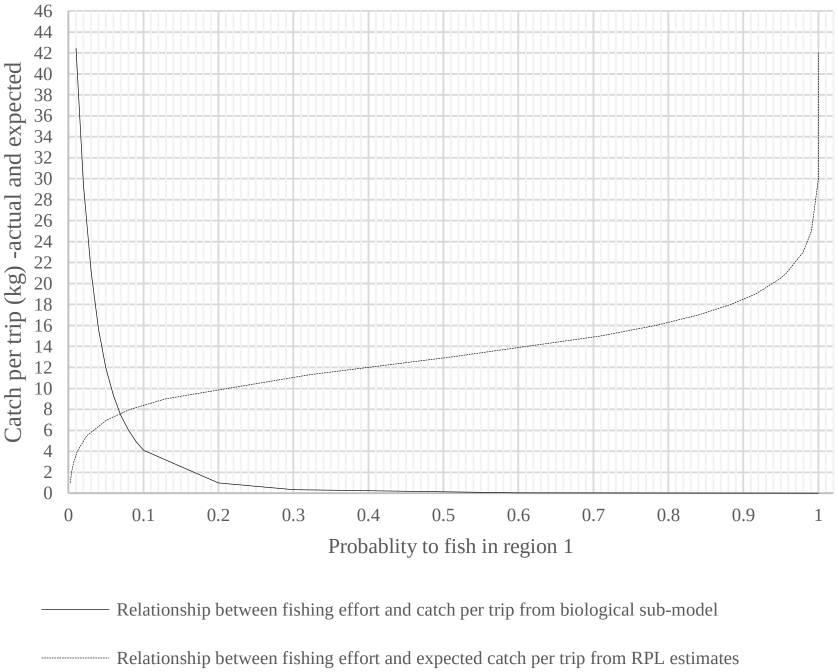
The conditions which characterise the fisheries and fish species (*Naso unicornis*) imply that there is also a unique biological relationship between fishing effort and fish population (and catch per trip) in the fishing area 1 without the BBMP. In comparison to the economic-sub model where fishing effort with rising catch rates, the biological processes of the fisheries imply that as fishing effort increases, fish stock and catch rates will decline. The biological sub-model provides a basis for establishing this relationship as shown in figure 6.5. As the proportion of fishing effort rises, the population falls, and catch per trip goes down.

Figure 6.5. Population in fishing area without the BBMP as fishing effort rises



At the steady state, the expectation of catch per trip at a given level of fishing effort must equal the actual catch per trip prevailing in the fisheries at that level of fishing effort. Technically, this is the interaction of the two curves (figure 6.6.) To better understand underlying mechanism, suppose fishing effort is lower than the one at the steady state, for example, 6% of trips are made to area 1. The biological sub-model shows that the population is approximately 130000 (figure 6.5). At this level of fishing effort and biomass, the actual catch per trip will stand at 9kg (figure 6.6). Fishers will rapidly adjust their expectation, but the high expectation will attract more fishers to that area of the fisheries. As effort increases, the actual catch per trip falls, so does the expectation from fishers. Fishers keep on entering the fisheries until the actual is equal to the expected catch per trip.

Figure 6.6. Fishing effort and catch per trip: interaction between biological and economic relationships



The above analysis provides a basis for estimating fishing effort in fishing area 1 in the absence of the Marine Park. The percentage of total effort that goes to the fishing area according to figure 6.6 is 6.9% and catch per trip is estimated at 7.36kg.

Comparison of the fisheries with and without the BBMP

Without the Blue Bay Marine Park, the number of trips to the fishing area 1 would be 2512, representing 6.9% of total trips in the fisheries, while with the Park, it was 1382 representing 3.8% (a difference of 80%). A fishing area which was inside the lagoon, with lesser depth and involved lower travel costs would attract relatively many fishers. These findings are consistent to Stelzenmüller et al. (2008). Fishing areas 4 and 7 have both these characteristics; the proportions of fishing effort are respectively 26% and 14% while catch per trips are 7.2kg and 6.5kg (table 6.7). The counterfactual analysis concluded that catch per trip without the BBMP was close to area 4 and 7 (figure 6.11).

With the BBMP, the catch per trip of 13.8kg. The difference in catch per trip (and consequently, population) in the fishing area with and without the BBMP are caused by an altered distribution of fishing effort as well as the spillover of the Park.

What would have been the effect of spillovers on the adjacent area 1 of the Park had the number of trips remained at its level without the BBMP (i.e 6.9%). As soon as the BBMP would be established, the size of the fishing area would decrease by 30% and catch per trip would fall from 7.4kg to 3.5kg as estimated from the biological-sub model. This is shown in figure 6.7 (dotted curve) coinciding to a situation with no spillover effect (Spillover from BBMP=0). The drastic fall in catch per trip is the outcome of a squeezing effect where the level of effort prevailing in the absence of the Marine Park has shifted to a smaller fishing area.

The spillover parameter in figure 6.7 ranges from 0.1 to 1 and is density-dependent. Relevant explanation on the modelling is provided in section 2. The analysis attempts to simulate the effects on catch per trip at different levels of spillover. With spillover, the fish population inside the Park would fall (figure 6.8). However, the fish population in the adjacent area would increase (figure 6.9), leading to a rise in catch per trip (figure 6.7). The model concluded that at a level of spillover of 0.1, catch per trip rose to 5.8kg, and increased significantly to 7.4kg when spillover was 0.6. The counterfactual analysis concluded that the Park would have kept catch per trip more or less the same level if spillover was relatively high (>0.6). A lower level of spillover would have made fishers worst-off. The model estimated that the latter reached 7.5kg with maximum spillover. When the catch per trip with varying spillover from the Park was compared to a situation characterising absence of the Park, there was no evidence of significant benefits emanating from the Park.

A different picture emerged when the bioeconomic analysis was conducted with the lower participation of fishers caused by the Park in fishing area 1. With a fishing effort of 3.8% of total trips, the fish population in both the Park and the adjacent area were higher (figure 6.8 and 6.9). More importantly, at that level of fishing effort, catch per trip was much higher than in the absence of the Park for any level of spillover (figure 6.7), showing significant fisheries benefits.

From this analysis, it can be concluded that the Marine Park caused fishers to move away from it, leading to a lower fishing mortality rate in fishing area 1. The fish population was therefore higher in the adjacent area. Since spillover was based on differential population densities, less fish migrated from the Marine Park and consequently, the fish stock was higher inside the Park.

Figure 6.7. Catch per trip with spillovers in fishing area when reserve is 30%

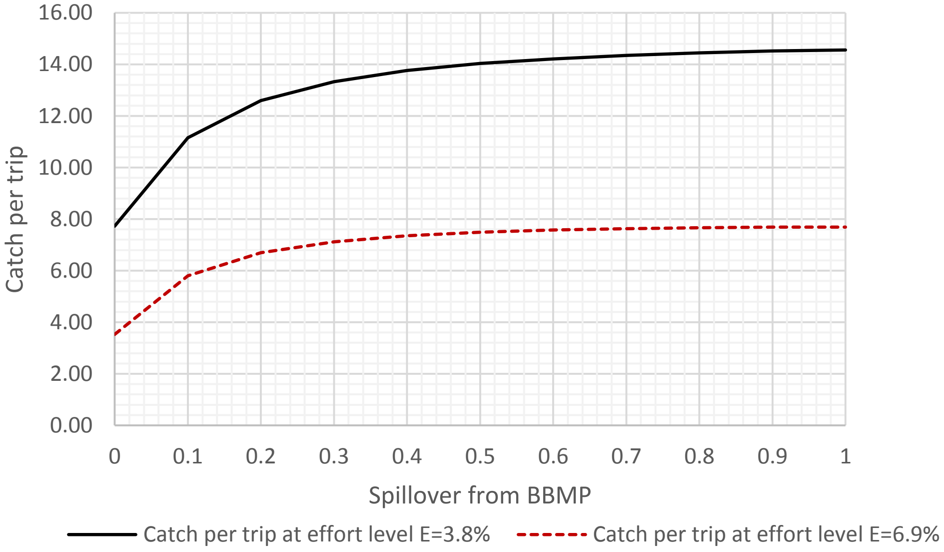


Figure 6.8. Population inside the BBMP with and without effort redistribution

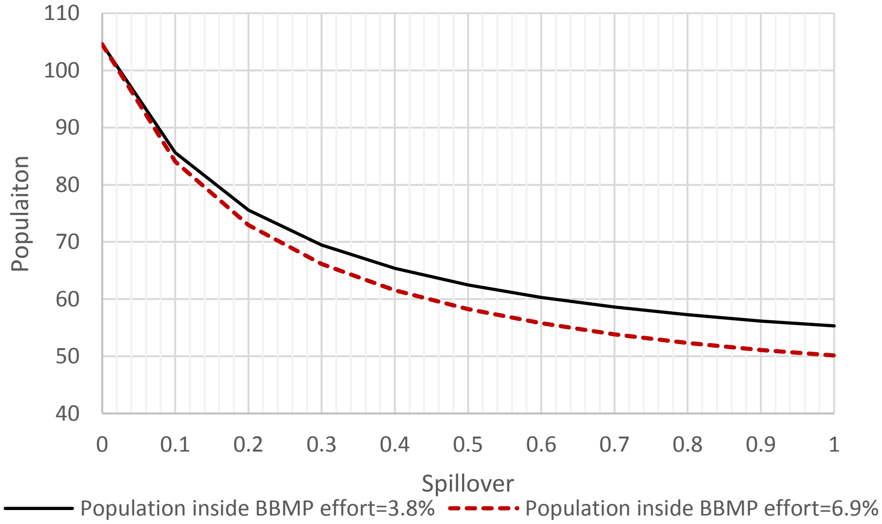
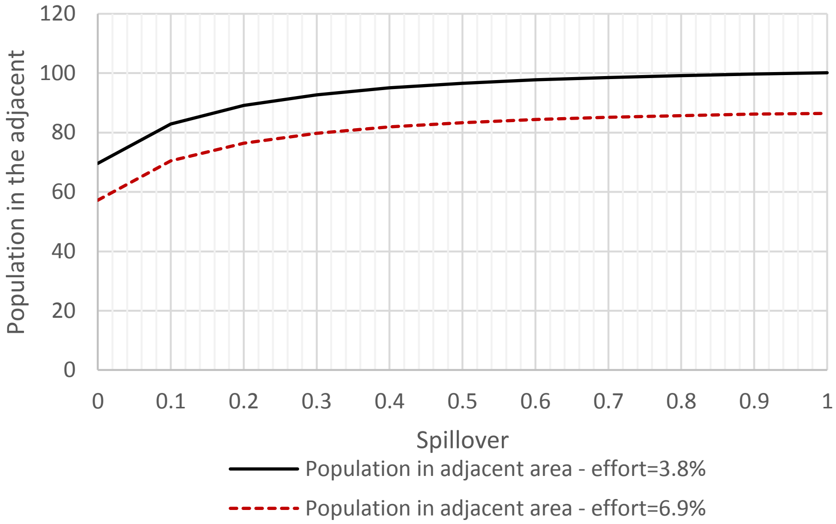


Figure 6.9. Population in adjacent area to the BBMP



Impacts of the BBMP on other fishing areas

The fisheries is much bigger when considering the extent to which fishers can move from their home ports. The effects of the Park go way beyond the fishing area 1. Figure 6.10 shows the distribution of fishing effort in the rest of fisheries with and without the BBMP. With the Park, fishing effort was higher in the remaining 7 fisheries areas in comparison to the distribution of effort without it (table 6.8). There was evidence that fishers settled in the other fishing areas in the fisheries when the Park was created.

Figure 6.10. Redistribution of fishing effort in the overall fisheries (%)

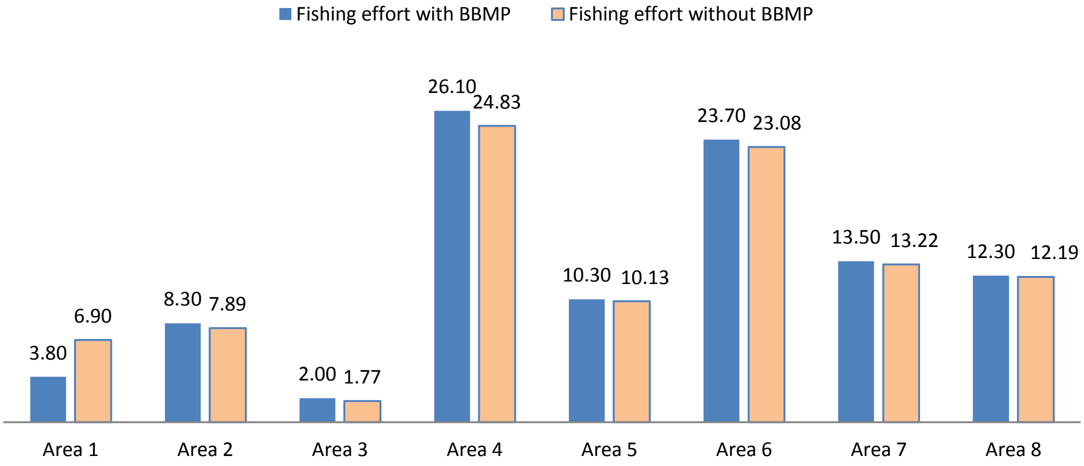


Table 6.8. Number of trips in the fisheries with and without the BBMP

	With the BBMP	Without the BBMP	Percentage change
Area 1	1383	2512	81.5
Area 2	3021	2874	-4.9
Area 3	728	643	-11.7
Area 4	9500	9038	-4.9
Area 5	3749	3687	-1.7
Area 6	8627	8400	-2.6
Area 7	4914	4813	-2.1
Area 8	4477	4435	-0.9

The different distribution of fishing effort in turn would affect the population structure of the overall fisheries. Figure 6.11 shows the difference in catch per trip. Without the BBMP, catch per trip in area 1 would be close to the area 4 and 7. These three areas were all located inside the lagoon. The economic reasoning stipulates that fishers would participate in the lagoon areas until catch per trips were equalised. With the BBMP, fishers shifted to other areas, and those who continued fishing in the adjacent of the BBMP (area 1) observe a catch per trip which paralleled the off lagoon area, that is, area 3, 5, 6 and 8.

Figure 6.12 and 6.13 illustrates the difference in fish population and biomass. The fisheries with the Marine Park led to a fall in population and biomass in other fishing areas. When accounting for this decline in fish stock, the conservation benefit of the Park are lower. For instance, the difference in fish population with and without the BBMP stood at 37000, representing a conservation benefit of 30%. However, the reduction in the population and biomass in area 2 to 8 following the redistribution of effort with the BBMP was estimated at 15000, implying that the net effect on the population with and without the BBMP was only 18%. Similarly, the change in biomass was 102% with the BBMP compared to a situation without it while the net effect was estimated at 66%, when accounting for the fall in biomass in other areas. It is also important to emphasise that the relatively higher biomass was due to the change in age distribution of the fish inside the Park and in the adjacent area towards older and bigger fish. This is confirmed by the figures of average weight of fish in the counterfactual analysis without the Park in table 6.9 (600g without the Park versus 900g with it).

Figure 6.11. Catch per trip in the fisheries with and without the BBMP (kg)

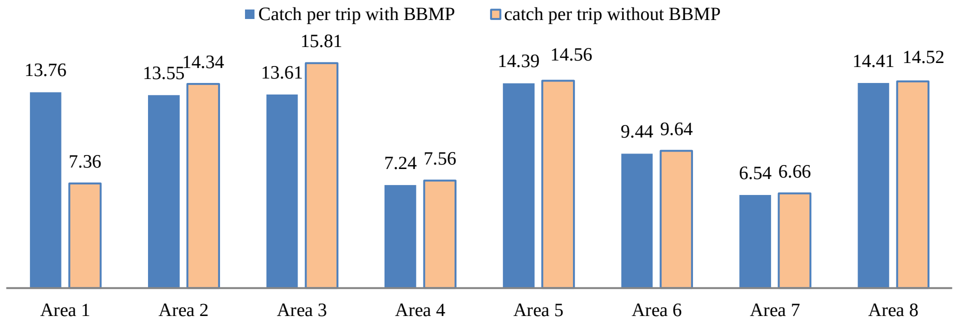


Figure 6.12. Population in the fisheries with and without the BBMP (000's)

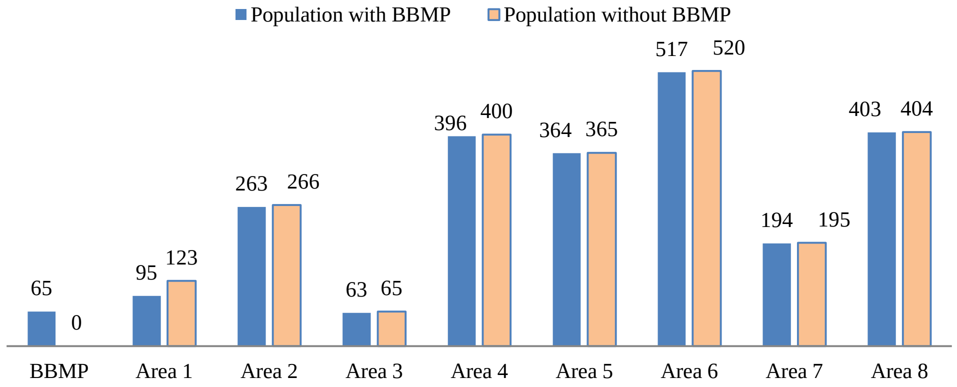
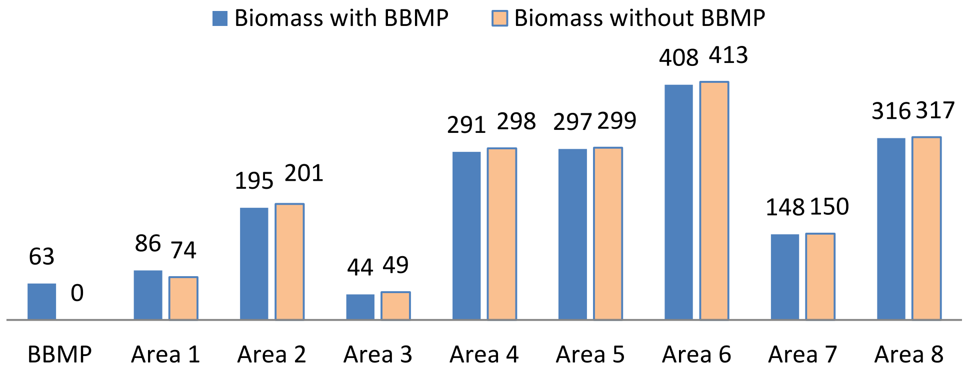


Figure 6.13. Biomass in the overall fisheries with and without the BBMP (tonnes)



Given the small size of the Park, the redistribution of effort did not significantly alter the fish population in the rest of the fisheries. This could be taken as a positive attribute of the Mauritian case. Are smaller marine reserve recommended when fishers spatial behaviour matter? If the marine reserve was larger, covering more space of the lagoon, the impacts of redistribution of fishing efforts would also be significant, especially in substitutes fishing areas.

Table 6.9. Average weight of individual fish with and without the BBMP

	Average weight of individual fish with BBMP (g)	Average weight of individual fish without BBMP (g)
BBMP	968	
Area 1	907	601.7
Area 2	741	755.4
Area 3	707	742.9
Area 4	735	745.0
Area 5	815	817.6
Area 6	789	793.9
Area 7	764	768.2
Area 8	782	783.7

6.5. Lesson learned

The study estimated the distribution of fishing effort with and without the BBMP, through a careful identification of changes in the conditions brought by the Park in the fisheries. When the Blue Bay Reserve was declared, it had two strictly local effects, both of which are also treated. Setting aside the lagoon section of the area as a reserve means that the remaining fishable area of it is in the off-lagoon area (since the Blue Bay Reserve extends from the shore to the reef). The displacement of effort from the lagoon area to deeper unprotected waters beyond the reef, adds human risk to the conventional catch risk (which was proxied by catch variance). It is observed through the econometric analysis that Mauritian artisanal fishers are attracted to fishing areas which have less physical risk. The Park also increased the travel distance between home ports lying north of the Blue Bay reserve and fishing grounds in the remainder of the area and south of it.

The model predicts that, without the Reserve, involving the absence of risk issues and changes in travel cost, the proportion of fishing effort directed to area 1 would have been 6.9% (around 2500 trips annually). With the BBMP, the proportion of fishing effort stands at 3.8% (around 1400 trips annually). In the light of these results and further analysis from the study, several questions relating the Blue Bay Marine Park can be answered.

Has the BBMP been successful in achieving the conservation effects?

The counterfactual analysis concluded that if the Park was open for fishing, the area would attract nearly twice the number of trips made by fishers. At this level of fishing effort, the area would be over-exploited, leading to lower fish population. The biological sub-model provided an estimate of fish population of 123000 and biomass of 74tonnes.

With the BBMP and associated fishing effort, the fish population (inside the reserve and adjacent area 1) is estimated at 160000 and a biomass of around 149 tonnes. The Blue Bay Marine Reserve had thus a highly conservation effect of increasing the fish population by 30%, despite being small.

However, this conservation effect has to be weighed against the fall in population in other fishing areas dues to effort distribution. Indeed, it is the overall population and catch that matter, not merely those in the Park and in the adjacent area 1. When accounting for the redistribution of effort to other fishing areas, the conservation effect is much smaller (18%)

Has the BBMP ensured sufficient fisheries benefits to fishers?

An important issue in the configuration of a marine reserve is whether it ensures sufficient spillover to the adjacent area to sustain the livelihoods of fishers. In the biological sub-model, spillover of fish from the Park to the adjacent area depended on the relative population densities of the two areas.

Comparing catch per trip in area 1 with and without the Marine Park did not provide a straight forward answer as to whether it provided effective fisheries benefits because there was a different distribution of fishing effort in the fisheries. A counter-factual situation is modelled and used to establish how much more a particular fisher, deploying a particular type of gear, would have caught in an area near the reserve before and after the reserve's establishment (an approach suggested by Smith et al. 2006). This approach suggested that if effort would have remained at the same level as it was without the Park, did the spillover from the reserve ensure fisheries benefits to fishers?

The counterfactual analysis concluded that the Park would have simply kept catch per trip to the level prevailing in its absence if spillover was relatively high. A lower level of spillover would have made fishers worst-off. Still, fishers would not experience a major rise in catch per trip even with maximum spillover. The configuration of the BBMP did not provide sufficient migration of adult and juvenile fish to the adjacent area to produce substantial fisheries benefits. This result is attributed to the size of the reserve. Being small, it did not produce sufficiently high population inside the Park for spillovers.

What has been the role of effort redistribution to the effectiveness of BBMP?

It is important to understand the mechanism through which fishing effort determines the magnitude of both the conservation effect and fisheries benefits. When fishing effort is relatively higher in the adjacent area 1, the fish population is consistently lower. The lower population density attracts more fish from the reserve, reducing the latter's conservation benefits. The opposite effects will prevail if fishing effort is low in area 1.

In the case of the BBMP, the changes it brought to the conditions of the fisheries meant that fishing effort has shifted away from the adjacent area (area 1). The lower fishing effort consequently led to a higher fish population in the adjacent area. Given that the difference between the population densities of the two areas were lower, less fish migrated, enhancing the conservation benefits. The situation also led to a higher catch per trip in the adjacent area of the Park.

In the Mauritian case, fishers' effort redistribution in the fisheries has been a critical factor in determining the conservation effect of the BBMP as well as the associated fisheries benefits. The siting of the BBMP by shifting the fishing area 1 to off lagoon also implied that it prevented entry even if there was a rise in catch per trip. The bioeconomic simulation with the Mixed Logit model showed that a rise in human risk and distance cost affected the spatial redistribution of effort significantly. From this perspective, the Park had been successful in reducing the pressure in an area which was intensely exploited by fishers. Despite being small, it led to a recovery of fish stock inside the Park as well as in the adjacent area 1 through its effect on fishers' behaviour.

The conservation effect must be analysed by considering the effect of fishers' effort in other parts of the fisheries. The RPL estimation predicted that as fishers shifted away from the adjacent area of the BBMP, effort rose in other fishing areas. The area which hosted the BBMP was perceived by fishers as a substitute to those which were mainly situated inside the lagoon. Effort redistribution impacted on the population, biomass and catch per trip in these areas as well. Given the small size of the BBMP, the changes were not significant in all fishing areas, ranging from 1% to 11%. Since the effect of the Park was mainly to redistribute effort, its small size became an important attribute. Had it been larger, more fishers would have displaced, perhaps shifted to another over-exploited areas. This would have had more detrimental effect on the fisheries.

6.6. Conclusion

The present study analysed some of the economic and biological impacts of an MPA by relaxing the assumption that fishing effort is fixed and uniformly distributed, and by allowing for endogenously determined fishing effort that responds to economic incentives. The modelling of the biological system and the human system in the study site illustrates how the Marine Park has had both some successful and unsuccessful outcomes and how fishers' behaviour can influence the impacts of an MPA on a fishery.

The analysis concluded that the current performance of the BBMP in enhancing the fish population was determined by a reduction in fishing effort away from the adjacent area to other fishing areas in the fisheries. From that perspective, the BBMP had been successful in the recovery of the fish stock in a heavily fished area. Without the BBMP, that part of the fisheries would have been highly exploited by fishers. Given its size, it was unsuccessful to ensure fisheries benefits to fishers.

The extent and magnitude of fishers' effort is an important factor to consider for the placement and size of marine reserve. A bigger reserve in an over-exploited fishery is likely to redistribute effort substantially to other fishing areas. While the population inside the reserve recovers, the change in effort may be detrimental to other fishing areas, especially those which are already facing intense fishing pressure. A smaller reserve, instead, such as the BBMP, does not lead to significant pressure on other fisheries areas. This is a positive attribute of the Park. As a fishery management policy, marine managers may attempt to configure a marine reserve or a network of them so that the level of fishing effort is optimally distributed across the fisheries to lead to a recovery of over-exploited areas, maximise the overall fish population of the fisheries and more importantly, ensures appropriate fisheries benefits to sustain the livelihoods of fishers.

7. Conclusion

7.1. Summary of findings

This dissertation thesis makes use of fishery-dependent catch information and data on biological processes to evaluate the performance of a marine protected area. Details of catches per trip and per trap were collected from fishermen using the waters adjacent of the Blue Bay Marine Park. After being standardised these were analysed and revealed a declining gradient in the populations of targeted fish species for 4km from the edges of the Marine Park. It also found that the different fish species inside the reserve and close to it were larger.

A priori, spillovers that increase expected catches near an MPA are likely to attract fishers to the area. However, only 4% of trips by fishermen in the general area are made to the vicinity of the Blue Bay Marine Park. In attempting to resolve this apparent conundrum, the analysis argues that the choice of a fishing area is highly sensitive to travel cost, proxied here by the distance travelled from the home port, and to the potential human risk associated with the water. Fishers' spatial connectivity is also influenced by the variance of catch and their attitudes towards risk.

A crucial question is whether the size of the Blue Bay Marine Park matters. If, prior to the reserve's establishment, the resource had been in good condition, and the stock above MSY, setting aside a no-take zone would necessarily have reduced the total catch per trip in its adjacent. However, should the resource have been heavily over-exploited (as was the case in Mauritius), the bioeconomic analysis concludes that there is an inverted-U shape relationship between catch per trip and the MPA's size.

A no-take zone could improve catches when an optimal proportion of the fishable waters set aside. It implies that the Marine Park in Mauritian waters can bring both conservation and fisheries benefits with an optimal size. From a fisheries perspective, the optimal size of a marine reserve will be one which maximises the average catch per trip.

However, in an open access fishery with high unemployment, barriers to entry would still be needed to keep catch per trip and incomes of individual fishers to desirable levels.

The decreasing gradient of catch per trip up to 4km reflects the two effects: spillovers and spatial redistribution of effort due to increases in travel costs and human risks at the fishing sites. To further provide on these two effects, the study conducts a counterfactual analysis comparing the fisheries with and without the Marine Park through a bioeconomic model and simulating fishers' behaviour using a spatial random utility model.

The counterfactual analysis concludes that the current size of the BBMP would not provide sufficient spillover to the adjacent area to produce fisheries benefits had fishing effort remain the same level it was without the Park. The high catches in the adjacent areas is therefore determined by a redistribution of fishers' effort, caused by the placement of the reserve. The reserve extends from the shoreline to the reef and therefore forms a barrier between most local fishers and lagoon waters south of the Park. The fishable waters in the vicinity of the BBMP are no longer in the lagoon, but lie beyond the reef, raising the human risk and distance from home port. The motivation of fishers is still to make profit, however the conditions that govern attempts to make such profit safely clearly have changed.

7.2. Policy implications

The placement of the reserve made it a barrier for many fishers, raising the cost of accessing lagoon waters south of the park. It reduced the attractiveness of the adjacent area to fishermen and therefore effort had shifted to other areas. And requiring those who wanted to 'fish-the-line' to do so beyond the reef, i.e. by raising the cost of fishing in the area. It therefore induced a reduction in local effort and a recovery of fish stock in the adjacent area. This attribute is particularly relevant for other inshore multi-species fisheries around the island, where effort controls are constrained to close seasons and gear controls such as the prohibition of nets.

The identification of fishing areas as complements (when fishing in one area increases fishing effort in another) and substitutes is valuable information for the placement of an MPA. In the case of Blue Bay Marine Park, the analysis suggested that the zones most likely to have suffered from the area's closure would have been those substitute areas that offered fishers low travel costs, stable catches, and a lower potential human risk.

If a reserve is placed in a preferred fishing area, the resulting spatial redistribution of fishers may have profound effects on fish populations in other areas, as well as on the welfare of fishers. A larger reserve is likely to have more displacement effect in this case than a smaller one. Therefore, a small or a network of a small reserve may be appropriate in such area. The small size of the Blue Bay Marine Park is a positive attribute in this respect.

Even if a fishing ground is spatially homogeneous, a marine reserve still can be located to maximize the fisheries benefits. The biomass in the reserve will rise, and spill over into surrounding waters. This reduces fishing pressure in substitute sites elsewhere, and increases it locally; but the changes are not even, despite the fishing ground being homogeneous. In a competitive fishery with full information, effort is located so as to equalize the net and not the gross revenue. i.e. effort increases in low cost areas, and decreases in more distant (or riskier) high cost waters. In this respect, the geographical location of the reserve can be selected in a manner that it redistributes effort spatially to minimise pressures and maximise fish population and catches in the overall fisheries.

Whether fishers as a whole benefit or lose from a reserve depends on its effect on total catch and on total costs. Both will be affected by the reserve's geographic situation, and its influence on the location of effort and patterns of travel to fishing grounds.

If, as in the case of the BBMP, the aim is to conserve the resource for touristic purposes, without diminishing the total catch in the remaining waters, then both the size and location are again critical. The reserve must be designed such that the spillover is sufficient to compensate for the decrease in extent of the fishable

Biological considerations have hitherto provided the principles for siting of MPAs. Integrating the economic aspect would strengthen the process. Gotz et al. (2012) actually offer the following:

Step I: Assessing differences in exploitation pressure

Step II: Detecting a difference in population parameters

Step III: Detecting ecosystem effects

Step IV: Evaluating the potential for spillover of adult fishes

Step V: Evaluating the potential for larval spillover

Step VI: Detecting effects on the fishery

In describing step VI, Gotz et al. (2012) mention that, "spatial information on fisheries catches before and after MPA implementation are most useful. For MPAs, this information is typically either not available or derived from fishery-dependent catch information. These are of varying spatial resolution and quality". This thesis is an attempt to circumvent these deficiencies through the modelling of fishery-dependent catch information from fishers. In this respect, box 7.1 expands on Gotz et al.'s approach and proposes to integrate spatial fishers' behaviour in bioeconomic models. The approach presented here also takes into account the assessment proposition by Clark (2002) for the Cape Peninsula National Park in South Africa. He proposed three phases when evaluating the existing MPA network and for selecting new areas for conservation: the first one is an assessment of broad areas of conservation worthy marine areas; the second one is to identify all existing and possible future conflicts to the proclamation of new MPAs at a particular location with due consideration to its type and intensity; and the third one is to weight the environmental significance, recreational use intensity, fishing intensity, and other attributes of the location to one another.

Biological and ecological processes of the marine environment and behavioural dynamics of fishers are very complex and their interaction cannot simply be assumed when establishing MPAs. The dissertation has attempted to provide some insights on the ecological and economic connectivities in a fisheries to enlighten the interactions between marine reserves, the behaviours of artisanal fishers and the benefits of the reserves to them. It concluded that fishers' spatial behaviour is a key factor in the design of MPAs. Future research is warranted which takes into account a more elaborated representation of the ecological processes of a fishery and the behaviour of agents. Such research would inform marine managers and the scientific community on the design of MPAs. Applied bioeconomic model in this respect is a promising avenue for a research agenda for those hoping to inform marine reserve formation and location.

Box 7.1. Step-wise approach to integrate fishers' effort in the design of a marine reserve

Step 1: Conduct an 'area selection' to identify those sites with environmental significance and valuable for conservation. This step relates to Clark (2002)'s first proposition and Gotz et al. (2012), i.e. to address species and ecosystem specific issues (composition and conformation, and depth of seabed, types of coral, water quality etc). It includes biological and ecological assessment of the characteristics of species, home range, competitors and non-competitors, and other parameters of the sites being potential for conservation.

Step 2: Identify intensity of effort and fishing technologies in the overall region as well as in areas hosting target species. This includes an in-depth analysis of areas considered locational complements and substitutes, with an estimation of effort elasticities. The Random Utility Model to be used to simulate changes in fishing efforts with respect to changes in the characteristics of the fishing areas. This step can be extended to include Clark' second proposition to include an identification of all existing and possible future threats to the proclamation of new MPAs at any particular position including human activities which can cause conflict and render the MPA ineffective, for e.g. recreational activities, heavy industries, etc.

Step 3 Using available information on key commercial species and ecological processes in step 1, construct a biological sub-model and simulate the immediate and long-run effects of reserve size at each potential sites on patterns of fishing effort, fish population and catches. The modelling would enlighten the immediate projected harvests across the affected areas *ceteris paribus*, as well as the long run effects taking into account the recovery of the fisheries, net migration of juvenile and adult fish into fished areas as well as spawning biomass, export of fertilised eggs, larvae and movements of juvenile and adult fish.

Step 4: Identify configuration (size and placement) that would most benefit the fishable population of each of these species, and which optimises the conservation benefit and fish population in the overall fisheries accounting for fishers' spatial patterns in the short and long term. The selection of the site as well as the size may involve trade-off between different objectives. Clark (2002)'s proposition may be adopted which entails overlaying the environmental significance, and other human activities (recreational use intensity, principal fishing grounds' layouts) on top of one another. This will enable the identification of areas that are highly rated in terms of the environmental significance but with lowest impacts on spatial fishing intensity in the fisheries.

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