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Economic analysis of Marine Protected Areas:

Bioeconomic Modeling and Economic Valuation Approaches

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Summary

Marine protected areas (MPAs) are often established for conservation objectives. Benefits provided by MPAs exceed pure biodiversity conservation as they may include contributions to social and economic benefits of local communities. And though it is still debated, MPAs may provide a management tool for sustainable fisheries and/or solving conflicts of interests between users of marine resources. It is of value to analyze and understand how implementation of an MPA can give different benefits to the economy and society. This thesis attempts to analyze some of the benefits of MPAs in specific situations. The thesis includes two parts; part 1 presents the general introduction of the thesis and part 2 consists of three papers. There are three main sections in the introduction. The first section presents the basic literature on the use of natural resources and MPAs, as well a description of the Nha Trang Bay marine protected area (NTB MPA) as an empirical case study. The second section presents research objectives and the summary of three topics dealing with these objectives. The first topic describes how an MPA can be used as a management tool to solve economic conflicts between ocean users, more specifically aquaculture and wild commercial fisheries competing for the use of the same species. An integrated bioeconomic model is developed for analyzing the impacts of an MPA on aquaculture-fisheries interactions. In the second topic, benefits from MPA-based tourism activities are derived using the discrete choice experiment method. The empirical analysis is applied to the NTB MPA in Vietnam. The total benefits of the coexistence of multiple activities, i.e. fisheries and tourism, affected by MPAs is analyzed and discussed in the third topic. The combination of a bioeconomic model and non-market valuation techniques (i.e. discrete choice experiment) is the approach for this study. Data from the anchovy purse seine fishery in Khanh Hoa province and tourism activities related

to the NTB MPA are applied for the empirical analysis. The final section presents overall conclusions of the thesis.

PART 1. INTRODUCTION

1. Background

1.1. Literature overview and research motivation

According to the IUCN (2008), coastal resources and marine ecosystems are in decline worldwide. Many of them have collapsed due to the impacts of overfishing, pollution, habitat degradation, and climate change. More than half of global fish stocks are fully exploited, while more than one fourth are either overexploited or depleted, and about 70% of coral reefs worldwide are threatened or destroyed. Implementation of MPAs is suggested as a key management strategy to address the issues that have impacts on marine ecosystems and resources. As a result, there has been a remarkable growth of MPAs worldwide from 0.9% to 8.4% of areas under national jurisdiction during the period of 1990 – 2014 (Juffe-Bignoli et al. 2014). Despite the increase in the number of MPAs worldwide, and the target of at least 10% of the world's marine and coastal regions by 2012 made at the Convention of Biological Diversity in 2006 (Jentoft et al. 2012), only 3.4% of the global ocean area was protected by 2014 (Juffe-Bignoli et al. 2014). This consists of 10.9% of all coastal waters, but only 0.25% of marine areas beyond national jurisdiction are protected (Juffe-Bignoli et al. 2014).

Together with the growth of global marine protected area coverage, the literature on MPAs has rapidly increased during recent decades. However, economic analysis is only a small share of this literature (Alban et al. 2008). MPAs are known as an effective fisheries management tool to recover over-exploited fish stocks, though the economic benefits in terms of fisheries management are still controversial (Merino et al. 2009). Some studies show that MPAs combined with optimal harvesting outside the reserves is less beneficial to fishers compared to conventional management tools (Hannesson 1998; Conrad 1999). Other research indicates that optimal harvest combined with a certain size of MPA can under some circumstances generate more resource rents than optimal harvest without an MPA (Sanchirico & Wilen 2001; Grafton et al. 2009; Punt et al. 2010; Punt et al. 2013; Schnier 2005b; Schnier 2005a). Moreover, MPAs have also been considered as a management tool for reducing the economic conflicts between ocean users (Bohnsack 1993).

Similar to terrestrial resource use, where there are potential conflicts of interests as regards land use and species conservation (Schulz & Skonhoft 1996; Skonhoft 2007), there are potential conflicts of interests between ocean users related to marine resources. A few studies have discussed issues of economic conflicts related to marine resources use. Ottolenghi (2008) indicated that tuna capture-based aquaculture has negative impacts on wild resources and hence conflicts in interests with other resource users. For example, the activity of tug boats towing tuna cages disturb the traditional longline fisheries and reduce tuna catches in many countries (e.g. Italy, Malta, and Tunisia). Bluefin tuna capture-based aquaculture relying entirely on wild-caught seed is shown to be the main cause of the reduction in the spawning stock, and the rapid increase in fishing mortality is the reason behind conflicts with the fisheries sector. In addition, bluefin tuna farmers in Croatia have serious conflicts with tourism activities in the use of the coastal zone (Ottolenghi 2008). Lee and Iwasa (2011; 2014) demonstrated the potential economic conflicts in natural resource use involving tourists as recreational anglers in competition with fishers as traditional divers. Liu et al. (2014) used a bioeconomic model to analyze the interactions between escaped farmed and wild Atlantic salmon in Norway. They show that both harvests and profits of wild commercial and recreational fisheries may decline after the escape of cultured fish from a marine aquaculture facility, but the total profits from the harvest of both wild and farmed stocks may either increase or decrease, depending on the change in values of parameters in the model. The analysis of the economic conflicts between aquaculture and wild fisheries, that is, the presence of aquaculture having negative effects on wild fisheries and hence reducing wild stocks, harvest and profits, was also carried out by Hoagland et al. (2003) and Mikkelsen (2007).

Marine resources are abundant and diverse. They provide various goods and services to different users (Cicin-sain & Belfiore 2005). Conflicts over resource use among users are common and expected to increase in extent and severity (Armstrong 2007; Mikkelsen 2007). In this thesis, the conflicts between capture-based aquaculture and wild fisheries is considered as the case study. A question is therefore addressed: how to solve the economic conflicts between ocean users, fishers and farmers?

Via zoning and separating different interest groups, implementation of an MPA is suggested as a solution to reduce the conflicts of interests at sea (Bohnsack 1993; Lee & Iwasa 2011; Ngoc & Flaaten 2010). Some studies, however, indicated that implementation of an MPA may not be an effective solution for reducing the economic conflicts between groups in marine resource use. Holland (2000) used a fleet dynamics model integrated with an age-structured model of a multi-species fishery to explore how an MPA implementation might affect the efficiency and distribution of benefits among different fishing groups. He indicated that introducing an MPA may have little effect in overall revenues but have different impacts on fishing groups from different ports, i.e. there will be winners and losers. Sumaila & Armstrong (2006) used a two-cohort model for a single species harvested by two groups and showed that the economic rents of cooperating fishing groups can be increased with the implementation of an MPA. However if fishers do not cooperate, introducing an MPA may not ensure rents to all the involved fishers. The question asked in this thesis is whether it is possible to achieve a win-win management strategy with an MPA implementation, which allows an increase in economic benefits for both competitors in marine resource use, fishers and farmers, while at the same time achieving biodiversity conservation objectives.

Besides the benefits of an MPA as a management tool for sustainable fisheries and solving economic conflicts between groups in marine resource use, an MPA may provide resources for tourism development. The recovery of marine biodiversity and degraded habitats due to protecting marine areas rapidly makes the areas become attractive for tourism (Alban et al. 2008). Some studies have underlined the attractiveness of MPAs for tourism, and indicated that tourists are willing to pay a premium price for an increase in the quality of the ecosystems (Can & Alp 2012; Parsons & Thur 2008; Schuhmann et al. 2013). As a result, developing tourism associated with MPAs might be regarded as a way of translating benefits of ecosystem conservation into economic terms (Alban et al. 2008).

In the economic literature on MPAs, the two benefits of MPAs, fisheries management and tourism opportunities, are often evaluated separately. However, as for management of common marine resource use, the economic values associated with different users are important for determining the optimal size of MPAs. A few studies have taken into account the combination of either non-extractive and extractive values or non-use and use values for a maximization of social welfare in connection to natural resources (Alexander 2000; Bulte et al. 1998; Clark et al. 2010; Moyle & Evans 2008; Rondeau 2001). In terms of MPAs, there are very few studies on the multiple benefits provided by MPAs, such as in work by Boncoeur et al. (2002) and Merino et al. (2009), who use bioeconomic models to illustrate the benefits of the coexistence of fishing and tourism activities under circumstances of different area-size distributions and fishing-effort levels. This thesis investigates the question of optimal size of an MPA, while integrating various values in order to maximize total value over all relevant stakeholders, i.e. in fisheries and tourism.

MPAs, although they provide various potential benefits, they also involve costs (Balmford et al. 2004). The issue of management costs of MPAs, especially enforcement costs, is still controversial (Alban et al. 2008). Some authors suggest that costs may be lower with an MPA than with conventional management tools (Armstrong & Reithe 2001), while others have the opposite point of view (Sanchirico et al. 2002; Cullis-Suzuki & Pauly 2010; Parrish et al. 2001). It is shown that about 70% – 80% of MPAs worldwide are protected only in name and are not effectively managed and hence the objectives of conservation and fisheries management of the MPAs are not ensured. The lack of ability to secure funds for running and managing MPAs has been indicated as the main obstacle to the success of MPA implementation (Depondt & Green 2006). It is also indicated that funds for maintaining MPAs often come from very limited public budgets, resulting in problems for the managers of MPAs (Cullis-Suzuki & Pauly 2010).

Tourist payments in the form of user fees for entering and using protected areas is a way to ensure the sustainable financial source funding for management of MPAs. It is well known that the use of terrestrial protected areas (e.g. national parks) for non-extractive commercial activities, such as eco-tourism, yields a price-premium (Bandara 2004; Baral et al. 2008; Birol et al. 2006; Wang & Jia 2012). Corresponding effects of MPAs are less explored (Jacobsen & Thorsen, 2010). A few studies underline the attractiveness of MPAs for tourists through recreational activities such as diving (Parsons & Thur, 2008; Sorice, Oh, & Ditton, 2007) and whale watching (Wilson & Tisdell, 2003), and attractive sea scenery as well as other tourist activities (Can & Alp, 2012, Hall & Hall, 2002). They show that tourists are willing to pay more than the current fees for improved biodiversity and environmental quality within the MPA. The increased income may be a sustainable financial source supporting the management costs of MPAs.

In this thesis, Nha Trang Bay MPA (NTB MPA) is used as an empirical case study to investigate the preferences of tourists visiting the MPA as regards the improvements of biodiversity and environmental quality within the MPA. The results of this study give relevant information for MPA managers in relation to potential sustainable source funding of ongoing MPA management costs as well as additional costs associated with environmental improvements. This study therefore contributes to the existing literature on assessing the potential use of tourism fees as a solution for sustainable financial sources of MPAs. Furthermore, the NTB MPA is used for the empirical analysis of multiple services provided by MPAs in this study.

1.2. Nha Trang Bay MPA and its total economic value

An international definition of MPAs is "a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (IUCN, 2008, page 3). MPAs are therefore acknowledged to be the cornerstone for promoting biodiversity, ecosystem services, and human well-being (Kemsey et al. 2012). If well-managed, MPAs may provide various benefits such as conserving biodiversity and ecosystems, protecting important habitats for fish, providing opportunities for nature-based recreation and tourism, and providing focal points for education, training, heritage and culture, etc. (Toropova et al. 2010).

In this thesis, the Nha Trang Bay (NTB) MPA is used as an empirical case study to investigate a part of the various quantifiable values which can be generated by an MPA. The NTB MPA was established in 2002, with two main purposes: to conserve marine biodiversity and to provide sustainable uses of natural resources, of which the former is considered to be the most important goal (Vo et al. 2002). The biodiversity in NTB was well known as the highest in Vietnamese coastal waters and relatively high for the overall Pacific Ocean, with 350 species of hard coral (accounting for over 40% of all reef-building coral species in the world), 220 species of demersal fish, 160 species of mollusks, 18 species of echinoderms, and 62 species of algae and seagrass (Vo et al. 2002; Nguyen & Phan 2008). This marine area therefore does not only support a variety of important habitats and ecosystems (i.e. coral reefs, mangrove forests, seagrass beds, sandy-muddy bottom areas, and rocky shores), but it is also considered a major nursery ground supplying fish larvae to other Vietnamese ocean areas and possibly also to Cambodian waters (Dung, 2009).

For management purposes, a zoning scheme for the NTB MPA was applied. The total protected area is 160 km^2, consisting of nine islands and their surrounding waters, and regulated into three zones with different levels of use and protection (see Figure 1). The core zone included four islands, i.e. Hon Mun, Hon Noc, Hon Cau, and Hon Vung, and the areas from the water's edge out to 300 meters around these islands, where only tourism is allowed. The buffer zone included the remaining islands and waters within 300 meters of these islands, and additional waters of 300 meters surrounding the core zones. Tourism, fishing, and marine farming were allowed in this zone, but no trawling. The transition zone opens to all activities, though limiting bottom trawl with regards to mesh size and engine power (Vo et al. 2002). In 2014, the names of the three zones, their boundaries, and the regulations related to the use and the protection were changed. The three regulated zones are currently renamed as the strictly protected area; the ecological rehabilitation zone; and the development zone, respectively. The strictly protected zone is now extended northward from Hon Mun and the whole area east and the southeast of Hon Tre. Fishing is not allowed in both the strictly protected area and the ecological rehabilitation zone.[1]

[1] Nha Trang Bay regulations. http://thuvienphapluat.vn/van-ban/Tai-nguyen-Moi-truong/Quyet-dinh-3363-QD-UBND-2014-Quy-che-quan-ly-vinh-Nha-Trang-Khanh-Hoa-263937.aspx.

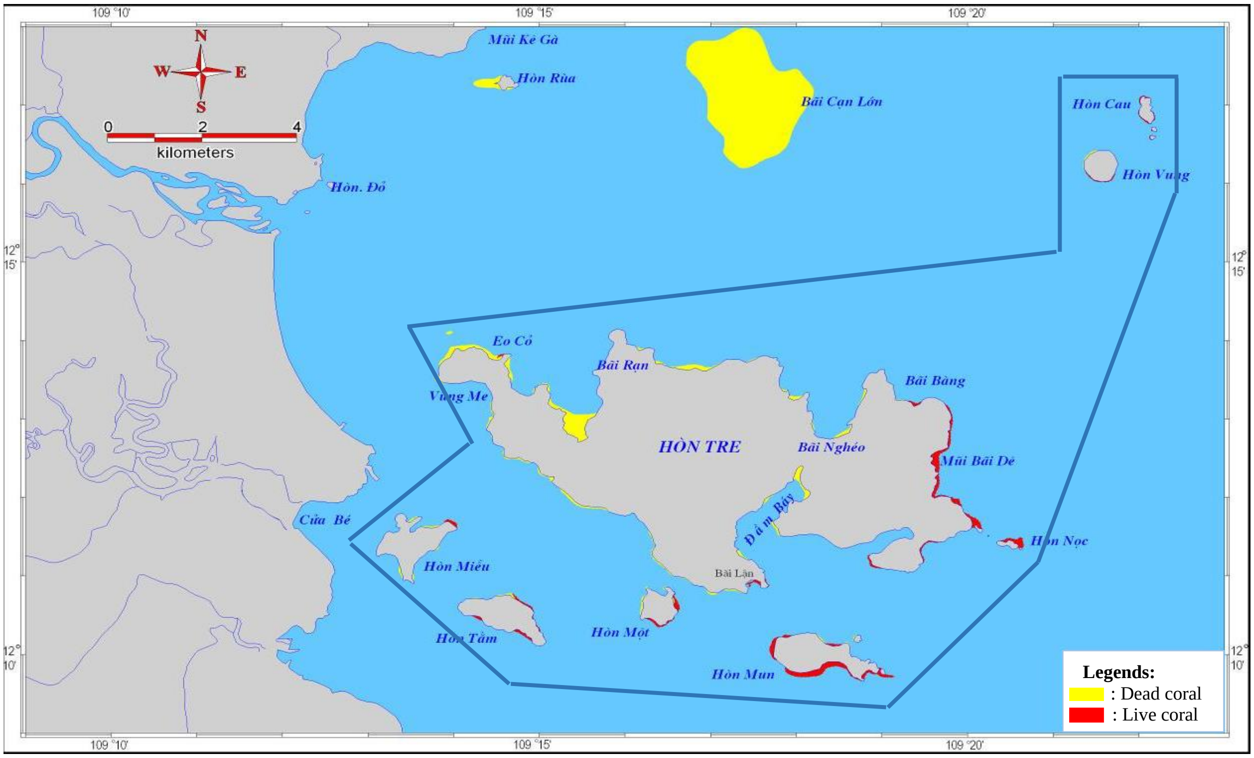

Figure 1. Map of the NTB MPA.

Source: Van (2013)

Figure 2 presents a summary of potential total economic value (TEV) of the NTB MPA, constructed based on both the general framework of the TEV of MPAs presented by Grafton et al. (2011) and the particular objectives of the NTB MPA project as indicated in Vo et al. (2002). The TEV of the NTB MPA includes the two components, use and non-use values. The use values consist of direct, indirect, and option values. The direct use values include consumptive use (e.g. fishing) and non-consumptive use. Non-consumptive use values are obtained from direct uses of natural resources for recreational purposes, i.e. diving/snorkeling or sea mammal-watching, and an increase in knowledge of marine system (i.e. education and research). Indirect use values are the values of ecosystem services with regards to biodiversity conservation and habitat protection. Option values are the current values of potential future direct and indirect uses of marine ecosystem.

Another important contribution of MPAs is non-use value, being the benefits obtained from conserving threatened, endangered and rare marine species. It consists of two components, i.e. existence value and bequest value. The former is the benefit of the knowledge about the species protected by an existing MPA, and the latter refers to benefits from ensuring the availability of the ecosystem services of MPAs to the coming generations (Grafton et al. 2011).

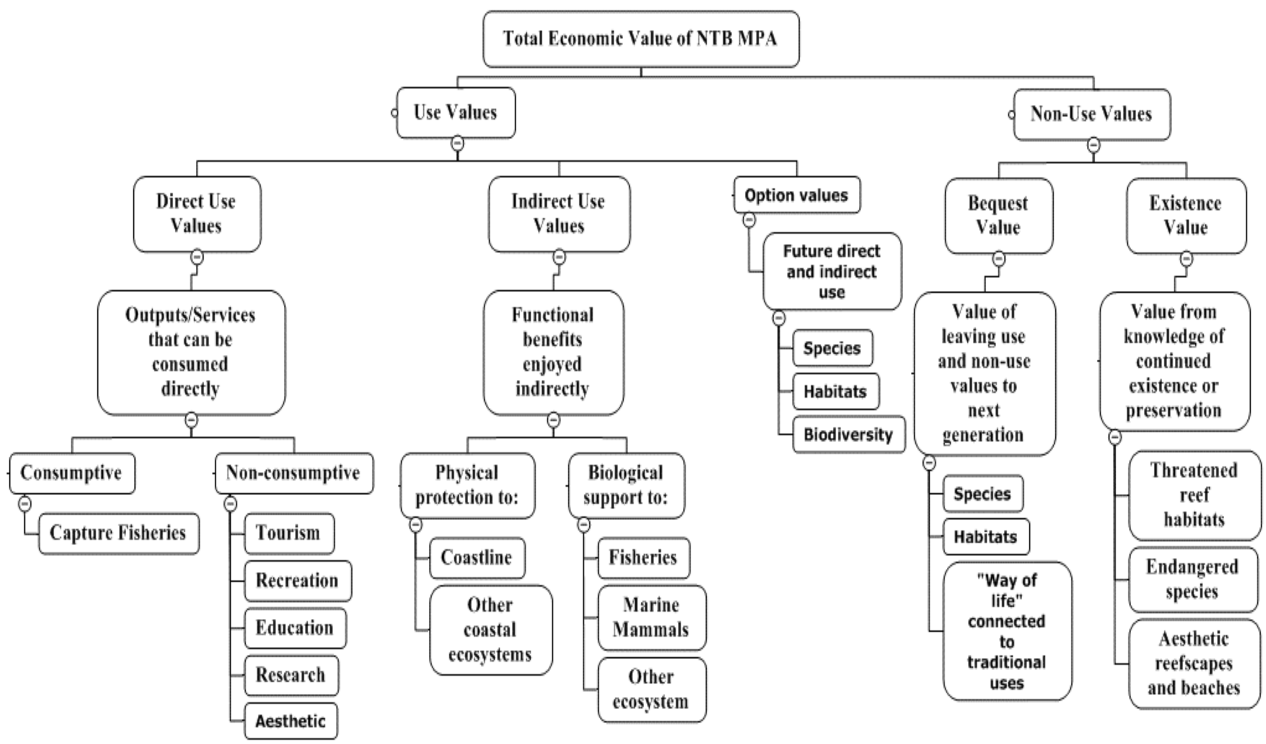

Figure 2. Total Economic Value of the NTB MPA.

Based on the potential values provided by NTB MPA, these can be grouped into two main economic topics. One is the valuation of ecosystem services provided by MPAs, the other is the analysis of the economic impact of an MPA as a management tool on these services. The former is conducted with the help of some assessment methods such as revealed preference (i.e. travel cost and hedonic pricing) and stated preference techniques (i.e. contingent valuation and discrete choice experiments), while the latter is often performed using bioeconomic modelling (Alban et al. 2008).

In this thesis, I focus on both economic topics of MPAs, one is the use of MPAs as a management tool to reduce the economic conflicts between ocean users, using an integrated bioeconomic model. This topic is presented in the first paper. Another topic is the valuation of the direct use values of the NTB MPA, i.e. from fisheries and tourism. A discrete choice experiment (DCE) method is used to evaluate the tourism value provided by the NTB MPA, which is presented in the second paper. A combination of both approaches, bioeconomic modelling and DCE, is used to analyze the multiple benefits provided by MPAs (e.g. fisheries and tourism). This is presented in paper three. Although other values of the NTB MPA such as option values and non-use values are not included here, the approach used in this thesis to evaluate non-consumptive value can be applied for the valuation of the remaining values of the NTB MPA.

2. Research objectives

The overall theme of this thesis is the economic benefits provided by MPAs. Three main topics have been analyzed and discussed:

(1) Can an MPA be considered to reduce the potential economic conflicts among ocean users, i.e. fishers and farmers?

(2) May the willingness to pay of national tourists for an improvement of biodiversity and environmental quality within an MPA be sufficient to cover the costs connected to this improvement?

(3) What is the optimal management for the multiple use provided by MPAs, i.e. fisheries and tourism?

Each topic is studied in a separate paper, presented below.

Topic 1: MPAs – a potential solution for solving the conflicts of interests between fishers and marine farmers.

In the economic literature, there are many studies on the interactions between wild fisheries and marine aquaculture (Anderson 1985; Hannesson 2003; Naylor et al. 2000; Valderrama & Anderson 2010; Ye & Beddington 1996). These studies present the two main classes of interactions, one being the market interaction which may indirectly provide positive effects of aquaculture on wild fisheries by increasing total supply to the market and thereby reducing fish prices which may result in reduced fishing effort and hence a potential increase in wild fish stocks and harvest in open access fisheries. The other is the interaction related to environmental issues and wild resource uses (e.g. wild seeds and feeds), which may show the opposite effects, that is, the growth of aquaculture may reduce wild fish stocks and hence wild catch. The latter is considered to be one of the reasons for the conflict of interests between fishers and marine farmers. These conflicts have been analyzed and discussed in the literature by Hoagland et al. (2003), Mikkelsen (2007) and Liu et al. (2014), though the authors do not include a solution to reduce these conflicts.

Hoagland et al. (2003) present a bioeconomic model to analyze the external negative impacts of aquaculture on fish population dynamics and hence wild fisheries production. By letting carrying capacity of fish stock be a linear function of aquaculture area, they investigate the impacts of aquaculture on wild fisheries via different management scenarios, i.e. under an open access fishery, an optimally managed fishery with individual quotas, and an optimal management of both industries competing in the market of fish production. In the first two scenarios, they show that the presence of aquaculture reduces fishing effort or the equilibrium value of quota. This may induce the fishers to oppose the introduction or expansion of marine aquaculture. In the last scenario, they look for the optimal solution of the coexistence of aquaculture and wild fisheries in an ocean area. They show that economic optimum is often related to a corner solution, i.e. either wild fishery or aquaculture should be carried out exclusively in the region, and the coexistence of the two users is sub-optimal. A counter-intuitive optimal outcome is also indicated; that higher unit cost of aquaculture results in a larger area allocated to aquaculture (and with a contraction of wild fishing effort), implying a tradeoff between aquaculture and wild fisheries.

In a similar vein, Mikkelsen (2007) analyzes the potential impacts of aquaculture on wild fisheries by modelling negative effects of aquaculture on aspects of wild fish: 1) carrying capacity, 2) intrinsic growth rate, and 3) catchability, depending on aquaculture production volume. These effects are investigated under both open access and sole ownership regimes. The author shows that the steady state fish stocks, fishing effort, and fishing yield, both under open access and sole-owner fisheries, vary depending on whether the impact of aquaculture on fisheries is via carrying capacity, intrinsic growth rate or the catchability coefficient. Despite the varying values in optimal variables, the equilibrium fishing rents decline for all three types of negative effects of aquaculture on fisheries, inducing the potential economic conflicts between fisher and marine farmers.

Similarly, Liu et al. (2014) present a bioeconomic model to analyze the impacts of escaped farmed fish (i.e. a type of biological invasion) on wild fish stock and harvest. Escapees have negative ecological effects, but positive economic effects on wild fisheries because escaped fishes increase the stock available for harvest. The authors suggest that stock, growth, and harvest of wild fish may decline after an invasion, and hence also the profitability of the wild fishery, but the total profits from fishing both wild and escapee stocks decrease only slightly. In some cases, the total profits can be improved compared to solely catching wild fish. This is due to the assumption that there is no difference between wild and farmed fish to fishers, the escaped farmed fish therefore contribute to the available stock for harvest. However, this may not always be the case as indicated by Olaussen & Liu (2011), that is, anglers are willing to pay substantially more fishing for wild than escaped farmed salmon.

In this thesis, the external effect of aquaculture requiring wild juveniles, on the commercial wild fisheries of the same species, has been applied particularly in model analysis. Aquaculture that relies on the collection of live material from the wild is of concern, as it is one reason for the reduction in juvenile availability with resulting impacts on capture fisheries, potentially creating economic conflicts between users (Sadovy de Mitcheson and Liu [2008]). When wild seed fisheries are poorly managed, growth of aquaculture may contribute to the threat of overfishing for some species (FAO 2011) or even cause fishery collapse, of which the lobster fishery in Vietnam is an example (Thuy & Ngoc 2004). Implementing an MPA is not only expected to mitigate these negative effects of aquaculture on wild fisheries, but may increase the harvests and hence profits for both fishers and farmers as well, and thereby reduce the conflict of interests between them. This topic is analyzed and discussed in paper 1.

With regards to the role of MPAs in fisheries management in general, and in relation to solving the conflicts of interests between users in particular, the benefits of MPAs include both biological (i.e. increase in fish abundance) and economic (i.e. positive spillovers to adjacent fisheries) perspectives (Kompas & Schneider 2005; Lee & Iwasa 2011; Ngoc & Flaaten 2010; Sumaila & Armstrong 2006). Bioeconomic models are the major approach, used to capture the economic impacts of MPAs. Bioeconomic models of MPAs are built upon the basic bioeconomic fisheries models and usually based on the key assumption of density-dependent dispersal. The models can be categorized into three major types: the logistic Schaefer model (see Conrad 1999; Hannesson 1998; Flaaten & Mjølhus 2010), the age- and size-structured population model (see Holland &

Brazee 1996; Holland 2000; U.R. Sumaila 2002), and spatially explicit bioeconomic models (see Sanchirico & Wilen 1999; Sanchirico 2004). In paper 1, we apply the first type of bioeconomic MPA model in order to analyze the impacts of an MPA on the interaction between wild fisheries and aquaculture.

In this paper, the presence of aquaculture that relies on wild caught juveniles is assumed to reduce the intrinsic growth rate of wild fish stock, and hence wild stock and harvest. The results of this study show that introducing an MPA of a certain size improves the results from both a biological and an economic perspective, compared to without an MPA under both open access and optimal management regimes.

Particularly, under open access, the equilibrium wild stock size increases with increasing MPA size and is larger than without an MPA, given aquaculture production. For a certain size of MPA, the equilibrium wild harvest is larger than without an MPA. These results are somewhat similar to existing studies, i.e. Hannesson 1998; Sanchirico & Wilen 2001, despite the fact that the underlying model in the first paper includes an aquaculture effect on wild fisheries outside the MPA. The combination of a certain size of MPA and optimal harvesting outside the MPA gives better results compared to conventional management tools. The optimal wild stock size increases with an increasing MPA size and is larger that without an MPA. The total profit of both industries as well as the profit of each with an MPA is larger than without an MPA. The results of this study are different to those indicated by Holland (2000) and Sumaila & Armstrong (2006), that is in the competitive environment of resource use there may be losers and winners when introducing an MPA.

Topic 2: Informing Management Strategies for a Reserve: Results from a Discrete Choice Experiment Survey

The second topic in this thesis is to examine whether information from a study of MPA valuation can help to design management strategies adequate to conserve and improve the poor state of biodiversity and environment within the MPA. For this topic, the NTB MPA in Vietnam is used as the empirical case study. This topic is presented in the second paper of the thesis.

The purpose of an MPA creation is often to conserve marine biodiversity and ecosystems, and hence ensure social and economic development (IUCN 2008). Many studies have focused on the effectiveness of MPAs and indicate that insufficient funding is one of the main obstacles to successful MPA implementation (Depondt & Green 2006). A few studies show that revenue from MPA-based tourism can be a sustainable financial source to cover the costs of managing an MPA (Grafeld et al. 2016; Depondt & Green 2006; Gelcich et al. 2013; Emang et al. 2016; Terk & Knowlton 2010; Thur 2010).

To assess the economic values of ecosystem goods and services which cannot be directly observedin markets, one often uses non-market valuation techniques (Grafton et al. 2011). Non-market valuation techniques can be divided into two main types: revealed and stated preference. Revealed preference methods (i.e. travel cost and hedonic pricing method) are often used to assess non-consumptive use values of marine resources, based on observations of actual choices or travel behavior of the visitors. Stated preference methods (i.e. contingent valuation, discrete choice experiment, and best-worst scaling) are often used to estimate monetary values of ecosystem goods and services, based on public surveys asking respondents about their willingness to pay to protect or improve the quality of the ecosystem, which is often constructed in a hypothetical referendum.

In evaluating ecosystem quality, the stated preference techniques are more commonly used (Grafeld et al. 2016). Of these, discrete choice experiments (DCEs) have been shown to have some distinct advantages, of which the ability to disaggregate policies/resources into appropriate characteristic sets and levels is the key feature (Adamowicz et al. 1998; Hanley et al. 1998). The second paper in this thesis therefore applies the DCE method to elicit national tourists' willingness to pay (WTP) an additional premium for boat trips within the NTB MPA, in relation to improvements in the environmental quality and increased biodiversity within the MPA.

As mentioned in the previous section, one of the most important objectives of establishing the NTB MPA is to conserve marine biodiversity. However, after more than 10 years of protection from 2002 to 2015, it is indicated that overall the status of biodiversity in Nha Trang Bay has not changed, though it does include areas with improvement as well as deterioration (Ben et al. 2015). The increases in live coral cover as well as diversity and abundance of fish were recorded mostly in the core zones of the MPA, i.e. Hon Mun, while declines were observed in the buffer zones, i.e. Hon Mieu and Hon Tam. Coral reefs at some sites are shown to be in such a degraded condition that they will not recover naturally (Ben et al. 2015).

Two main reasons have been suggested for why the MPA has failed to achieve the desired increase in biodiversity. One is the unplanned and unregulated human activities existing within the MPA, i.e. overfishing, aquaculture, tourism and urban run-off. These human activities have negative effects on the recovery of coral reefs and reef fish abundance which were heavily degraded prior to 2002 (Tuan 2011). Another reason is the core zone of the NTB MPA is believed to be too narrow to ensure biodiversity restoration and prevent marine environmental pollution (Dung 2009; Tuan 2011). Particularly, most protection has been focused on Hon Mun which is a small area, so the potentially positive dispersal effects of the MPA core zone may not be really effectively promoted (Tuan 2011).

Expanding the core zone of the NTB MPA and changing management policies may improve marine biodiversity, coral reef cover and coastal environmental quality. However, the funding for sustaining and running the NTB MPA at present is indicated to be one of the greatest challenges (Dung, 2009) and expanding the core zone of the MPA is expected to incur even more management costs. The second paper therefore focuses on the assessment of whether MPA-based tourism development can generate a sustainable source funding for managing the NTB MPA. The results of the second paper show that Vietnamese tourists are willing to pay a price premium for boat trips visiting the NTB MPA and confirms there is a potential sustainable financial source to fund the improvements in biodiversity and environmental quality within the MPA.

The majority of studies on tourism values of MPAs using stated preference methods focus on the attractiveness of MPAs for tourists, such as ecological characteristics (i.e. biodiversity and environment) (Grafeld et al. 2016; Emang et al. 2016; Wang & Jia 2012). However, implementing an MPA may have employment effects on the local fishers, as some may lose their jobs due to unavailable fishing grounds. Tourists visiting the MPA may be concerned about the potential job losses of the local fishers when they make choices regarding the alternatives of the MPA management plan. There are apparently no studies on MPA-based tourism valuation taking into account an employment effect of the MPA creation, though it is included as an attribute for tradeoffs in several studies of wetland valuation (see Birol & Cox 2007; Morrison 2002; Morrison et al. 1999; Othman et al. 2004 for more discussion).

In the second paper, an employment effect of an expansion of the NTB MPA on the local fishers is included together with the ecological aspects (i.e. coral cover and environmental quality) in the

DCE design to capture the benefits preferred by tourists. The results show that the tourism values of an MPA can be provided by both the social attributes (i.e. employment opportunities or losses) and the ecological attributes. The latter is normally presented in the literature of MPA valuation, while the former is not.

Topic 3: Extractive and non-extractive values of a marine protected area

The creation of an MPA can be considered an investment in natural capital. Natural assets in general, however, are often providers of multiple services, and hence multiple benefits (Alban et al. 2008). In the literature, the economic values of these services are often evaluated separately and by applying different methods. A few studies combine extractive and non-extractive value or use and non-use value to capture multiple benefits accrued from wildlife both in terrestrial and marine resources. For instance, Alexander (2000) presents a bioeconomic model, using the African elephant as an example, in order to estimate the use and non-use value of endangered species. He shows that non-consumptive value (i.e. tourism revenue) and non-use value (i.e. existence value) can be used to support elephant conservation, and hence play an important role in slowing the population decline. Skonhoft (2007) uses a bioeconomic model taking into account both consumptive and non-consumptive tourism value to analyze the conflict of interests between a park agency and local people related to terrestrial wildlife conservation. Bulte et al. (1998) and Horan & Shortle (1999) integrated non-use values of Minke whale stocks in a fisheries bioeconomic model in order to study the optimal management of whale resources. The results of the two studies show that the Minke whale moratorium was inefficient when only market values (i.e. whale hunting) were considered. However when there existed a significant non-use value, a moratorium could be optimal. Moyle & Evans (2008) included non-extractive or tourism values of whale watching in a model to inform policy and discuss issues related to the economic benefits of switching from whale hunting to watching. They show that there exists a steady state equilibrium for maximizing the total returns from both consumptive and non-consumptive values of whale populations. Armstrong et al. (2015) use an expanded bioeconomic model to show how non-use value of natural habitats impacts on the optimal fishing activities, using cold water corals in Norway as an example. Hence a combination of multiple benefits accruing from natural resources use may give different economic implications than predicted in the studies solely focusing on commercial harvest.

Nonetheless, very few studies have been carried out in order to investigate multiple benefits provided by MPAs (Boncoeur et al. 2002; Lee & Iwasa 2011), especially using empirical data for model application (see one exception in Merino et al. (2009)). Boncoeur et al. (2002) and Merino et al. (2009) use a bioeconomic model of MPAs to analyze the impacts of an MPA creation on both fishing and ecotourism. Their results show that implementing an MPA does not only increase the benefits for the fisheries, but it also generates additional income through tourism activities. Lee & Iwasa (2011) also use bioeconomic models to analyze and discuss the conflicts of interests between tourists and local fishers in marine resource use, as well as how to reduce these conflicts. The third topic and also the third paper of this thesis, therefore, is about multi-service benefits (i.e. fisheries and tourism) generated by MPAs. A bioeconomic model of MPAs combined with an estimation of tourism values as regards to the MPA is used for an investigation of the optimal management of the two activities; fisheries and tourism. Differing from earlier studies on multi-benefit provided by natural resources, where non-consumptive values and non-use values are modelled as a function depending on either the size of the stocks or the size of harvest (Alexander 2000; Boncoeur et al. 2002; Bulte et al. 1998; Merino et al. 2009), tourism value in this paper is formulated as depending on the size of the MPA. In this paper, implementation of an MPA does not only address the potential positive effects on fisheries (i.e. spillover effect) and tourism activities, but it is also considered as a controlling factor for a maximization of total values of the two activities.

Empirical data from both the anchovy purse seine fishery and tourism activities related to the Nha Trang Bay marine protected area (NTB MPA), located in the south-central of Vietnam, is applied into the model. It is assumed that the managers want to maximize the total welfare from fisheries and tourism. The tourism value function for the NTB MPA is estimated based on the primary data which is used for estimating tourists' willingness to pay for a hypothetical management policy of the NTB MPA in the second paper. Fishery data are secondary data, the biological and economic parameter values of the anchovy purse seine fishery are borrowed and developed from results of studies carried out by Thi et al. (2007) and Thuy & Flaaten (2013).

The results of this paper suggest that implementing an MPA is not only a good policy for biodiversity conservation but it also is a good economic policy. Although the fishery bioeconomic model of MPAs in this paper is based on the Conrad (1999) model, which is indicated as giving less benefit to fishers compared to conventional management tools, the inclusion of a tourism value of the MPA in the model highlights a broader picture of the actual reasons for MPA implementation. In this study, optimal management requires an expansion of the MPA and a reduction in fishing effort. This secures the fish population from overexploitation but may have short-term negative effects on local fishers. However, the additional income through MPA-based tourism activities may be used partially or totally to compensate the loss of fishery rent due to the MPA expansion.

The use of MPAs as a fishery management tool is of interest from a bioeconomic modelling point of view. The inclusion of tourism value related to the MPA in the basic fisheries bioeconomic model strengthens the model's illustration of the multiple benefits of MPAs. Moreover, for management of common marine resources that support various goods and services to different users, the economic values associated with different users are important for determining the optimal size of MPAs. This in turn allows for maximizing the total value over the stakeholders of natural resources. This study, therefore, contributes to the important literature of MPAs as a multi-service provider of common natural resources.

3. Conclusions

The main subject of this thesis is the benefit provided by MPAs. The thesis, therefore, focuses on analyzing the economic values of MPAs, specifically fisheries and tourism, as well as the role of MPAs as a management tool for solving economic conflicts among natural resource users, i.e. fishers and marine farmers. Both the approaches of bioeconomic modeling and economic valuation of non-market goods (e.g. a discrete choice experiment), are used in this thesis. Data from the anchovy purse seine fishery and tourism related to the NTB MPA in Khanh Hoa province in Vietnam are used in the model applications.

Though still hotly debated, MPAs are argued to supply a management tool for sustainable fisheries and to solve the conflicts of interests between oceans users. This thesis adds to the literature a positive effect of an MPA creation in terms of mitigating the economic conflicts between ocean resource users. In this thesis, an integrated bioeconomic model is presented to show that the implementation of an MPA of a certain size may increase both biological (i.e. wild fish stock size) and economic benefits (i.e. harvest and profit) for both fishers and marine farmers who are competing in the use of the same species. Hence the economic conflicts may be resolved. This result implies that implementation of an MPA can be considered to be a win-win management strategy, where both conservation and economic objectives are ensured.

Implementation of an MPA is not only considered as a management tool, but it is also regarded as an investment in natural assets which provide multiple service benefits for people. The literature on MPAs to date however mostly considers the biological significance and increase in fish yields. There is a shortage of the social perceptions of MPAs and economic valuations of activities in MPAs, for example, recreational fishing, diving, ecotourism, and research (Christie 2004) which may contribute to higher values of MPAs. Moreover, in developing countries, with overexploited fisheries and limited funding for monitoring and enforcement, community awareness and their support towards MPAs are crucial for the success of an MPA (Kompas & Schneider 2005). In this thesis, the tourism service benefit is evaluated as an alternative benefit of MPAs. The information derived from this study is relevant for the NTB MPA managers in terms of both management strategies for biodiversity conservation and environmental protection, as well as sustainable financial source funding for maintaining and running the MPA.

MPAs are often established for multiple goals (Kompas & Schneider 2005), and involving different stakeholders. It is necessary to incorporate the benefits of relevant stakeholders for an optimal management of natural resources. The inclusion of multiple benefits provided by MPAs in the model may give more complex economic implications than predicted in studies that solely focus on one benefit of an MPA. However, there are relatively few studies that combine multiple benefits for a better use of MPAs. To fill this knowledge gap regarding MPAs, this thesis presents a bioeconomic model that allows for the incorporation of the benefits from fisheries, aquaculture and tourism activities related to an MPA in order to determine the optimal MPA size for a maximization of total welfare of the two activities. Although the model and its' application are represented by the two activities, fisheries and tourism, the approaches used in this thesis allow for the inclusion of alternative values of MPAs in the model for a broader illustration of multiple benefits generated by MPAs.

Multiple goals of MPA establishment, which involves different stakeholders, are often connected to conflicts of interests (Jentoft et al. 2012). Potential economic conflicts can occur among conservationists and fishers (Francis et al. 2002), tourists and fishers (Lee & Iwasa 2011; Milazzo et al. 2002), or different fisher groups (Ngoc & Flaaten 2010). Though the conflicts of interest in relation to MPA implementation is discussed in the third paper of this thesis, it is not included in the model as clearly as the interaction effects among groups. Therefore, it is of interest to future research to further take into account these interaction effects for an optimization of common resource use.

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PART 2. PAPERS

PAPER I

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Marine reserve creation and interactions between fisheries and capture-based aquaculture: A bio-economic model analysis

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Abstract

The rapid growth of aquaculture affects wild fisheries in several ways. We present a bioeconomic model of the interaction between a commercial wild fishery and capture-based aquaculture that depends on harvest of wild juveniles. We assume that aquaculture reduces the intrinsic growth rate of the wild fish stock due to wild caught juveniles used as seeds, influencing wild stock size and commercial harvest. This may increase the economic conflicts between fishers and farmers. Introducing a marine protected area is expected to reduce these conflicts. The model considers both open access and economically optimal management regimes outside the marine protected area using numerical simulation. The numerical results show that a marine protected area of a certain size may increase economic benefits of both fishers and farmers in the case of optimal management, and enhance wild catch outside the reserve when there is open access.

KEYWORDS

aquaculture, bioeconomic model, management, marine protected areas, wild fishery

1 | INTRODUCTION

Total world fish production increased dramatically from 19.3 million tons in 1950 to 148.5 million tons in 2010, with the largest contribution coming from marine capture fisheries (FAO, 2012). However, fish supply from wild fisheries reached a peak and has been fairly stable at about 90 million tons since the late 1980s. Since then, the increase in total fish production has come from aquaculture (Diana, 2009). Aquaculture production reached 59 million tons in 2010, compared to 4.7 million tons in 1980, which is equivalent to an average annual rate of increase of 8.45% (FAO, 2012). As the share of aquaculture contribution in seafood supply increases, the fisheries–aquaculture interactions become important and are receiving ever more attention from researchers (Natale, Hofherr, Fiore, & Virtanen, 2013).

Wild fisheries–aquaculture interactions can be divided into two classes: the market interactions (between wild and farmed fish that are sold as food) and the interactions related to environmental efficiency and wild resource uses (i.e., seed and feed). The former type of interaction may provide positive effects of aquaculture on wild fisheries such as the increase in total supply and reduction in fish price, resulting in a decline in fishing effort and hence an increase in wild fish stocks and harvest in open access fisheries (Anderson, 1985; Valderrama & Anderson, 2010; Ye & Beddington, 1996). The positive market effects of aquaculture on wild fisheries, however, have only been identified for some species such as sea bream, sea bass, salmon, shrimp, tilapia, and pangasius (Asche, Bjørndal, & Young, 2001; Asche, 2008). For the majority of aquaculture species that are low-intensity, small-scale, and wild seed dependent, productivity-enhancing innovation effects are limited (Asche, 2008). The second type of interaction shows the opposite result; the growth of aquaculture that relies on natural resources has a negative effect on wild fish stocks and hence diminishes wild catch (Hannesson, 2003; Naylor et al., 2000). In addition, the growth of aquaculture creates ecological and environmental impacts such as water and genetic pollution, destruction of coastal environments such as mangrove forests, and results in disease being spread from cultured fish to wild fish (Chopin et al., 2001; Naylor et al., 2000). Though there exist a variety of interactions between wild fisheries and aquaculture, in this study we focus on the external effect of aquaculture requiring juveniles of the same species that is harvested in a wild commercial fishery.

Aquaculture that relies on the collection of live material from the wild is defined as capture-based aquaculture (CBA) (FAO, 2011). A wide range of representative marine and freshwater, vertebrate and invertebrate species are used in CBA practices as seeds and selected from four major groups: mollusks (i.e., mussels, oysters, scallops, cockles), crustaceans (i.e., shrimps, spiny lobsters, craps), echinoderms (i.e., sea cucumbers), and finfishes (i.e., freshwater eels, milkfishes, air-breathing catfishes, cods, mullets, temperate basses, jacks and pompanos, grouper, snappers, porgies, wrasses, rabbit-fishes, mackerels and tunas, snakeheads) (Sadovy de Mitcheson & Liu, 2008). Though we could find no current CBA coverage data, it was estimated to be at least 20% of the total annual fish aquaculture production in 1997 (Hermansen & Eide, 2013), indicating a substantial industry worldwide. CBA therefore is of concern as it is yet another cause of reduction in juvenile availability with impacts on capture fisheries, and potentially creating economic conflict between users (Sadovy de Mitcheson & Liu, 2008). With the exception of some developed countries where CBA is carried out alongside managed fisheries (i.e., cod in Norway, crab in Australia, yellowtail in Japan), most fisheries around the world, including the wild seed fisheries for CBA, are typically not managed or controlled effectively (FAO, 2011). When wild seed fisheries are poorly managed, growth of CBA may contribute to the threat of overfishing for some species, for example, milkfish in the Philippines; shrimp in Bangladesh; seahorses, humphead wrasse, and grouper in Southeast Asia; European eels; mullet in Egypt; Atlantic Bluefin tuna (FAO, 2011); or even cause fishery collapse, of which the lobster fishery in Vietnam is an example (Thuy & Ngoc, 2004). CBA can be replaced by hatchery-based aquaculture (HBA) in the future, but full conversions to HBA are rare, and CBA can therefore be expected to continue for many species due to biological, social and economic reasons (FAO, 2011). As stated by FAO (2011), such CBA practices require responsible development and management.

Regulations and enforcement of fisheries exploitation face challenges worldwide. Marine protected areas (MPAs) are seen as an alternative fisheries management tool for securing the recovery of overexploited fish stocks and for ensuring sustainable fisheries (Alban et al., 2008). MPAs are usually closed areas for fishing, and as such they therefore not only protect part of the fish stock, but they may also provide protected grounds for spawning, juvenile settlement, nurseries and feeding. Though there are a number of ecological studies showing the recovery of fish stocks both inside and outside MPAs, these effects seem to be largest within the MPA (Halpern, 2003; Halpern, Lester, & Kellner, 2009). The early research on systems consisting of a marine sanctuary and a fishing ground showed that the catch with an MPA might be greater than under solely open access due to migration to fishable areas from a more plentiful stock in the MPA. However, these studies underlined that reserves give less benefits than optimal management (Hannesson, 1998; Holland & Brazee, 1996; Conrad, 1999). Newer research has qualified this aspect, where, for instance, Sanchirico and Wilen (2001) develop a dynamic, spatial bioeconomic model that suggests both total fish biomass and harvest may under certain circumstances increase after closing and protecting an area from exploitation. Grafton, Kompas, and Van Ha (2009) use northern cod fishery data and counterfactual analysis with stochasticity to show that optimal harvest and an optimal-sized marine reserve could generate more resource rents than optimal harvest without a reserve. MPAs have also been shown to potentially outperform conventional fishery management (i.e., TACs, quotas) by increasing the average intrinsic growth rate of the stock, making it more resilient in the face of large harvest pressure (Punt, Weikard, Groeneveld, Van Ierland, & Stel, 2010; Punt, Weikard, & Van Ierland, 2013; Schnier, 2005a,b).

In this study, we will use a bioeconomic model to analyze the interaction between a commercial adult fishery and aquaculture dependent on wild juveniles, including an MPA effect.[1] Examples of general interactions between aquaculture and wild fisheries are few, but do exist in the literature. For instance, Hoagland, Jin, and Kite-Powell (2003) presented a bioeconomic model in which aquaculture activities reduce the environmental carrying capacity of a wild fish stock, resulting in smaller stock size and harvest from the wild fishery; Mikkelsen (2007) also found that wild fishery production varied dependent upon both aquaculture impacts (i.e., impact on intrinsic growth rate, carrying capacity or catchability) and the choice of management regimes. There is, however, to the best of our knowledge, no study showing the negative effects of CBA on wild adult fisheries while including the introduction of an MPA.[2]

The presence of CBA can increase natural mortality of the wild fish stock in early life stages, hence reducing the intrinsic growth rate. This, again, reduces the natural growth of the wild stock, impacting the equilibrium wild stock, harvest, and profit. In this vein, Simon et al. (2012) analyzed a dynamic biomass model using all available biological and ecological information regarding Atlantic bluefin tuna to show that the natural mortality of prerecruit stages (larvae and juveniles) and the intrinsic growth rate are indeed strongly and negatively correlated. Similarly, Gedamke, Hoenig, Musick, Dupaul, and Gruber (2007) argue the positive connection between first year survival (albeit without fishing) and the intrinsic growth rate, underlining the connection between juvenile survival and the intrinsic growth rate of fish.

Our study is based on Hoagland et al. (2003), but we assume that CBA reduces the intrinsic growth rate of a specific wild fish stock, rather than the carrying capacity, and expand the bioeconomic model with an MPA. Though many economic studies have questioned the overall benefits of MPAs (Anderson, 2002; Hannesson, 1998), we ask whether MPAs can reduce the economic conflicts between ocean users: fishermen and farmers.

This study makes two main contributions to the literature. The first is an expansion of the Hoagland et al. (2003) model in order to show the effects of the introduction of an MPA on interactions between a wild fishery and CBA. Furthermore, we develop the model to include the impact of CBA on intrinsic growth of the wild stock in a nonlinear functional form.[3] This is a novel contribution as to the best of our knowledge, there exists no earlier study of the impact of an MPA on the interaction between aquaculture and wild fisheries. Secondly, we present MPAs as an integrated management tool to reduce potential conflicts between different users of the sea. This is an additional contribution to the scarce but important literature on resolving conflicts between ocean users via zoning, thereby separating conflicting interest groups (Bishop & Samples, 1980; Hoagland et al., 2003; Jin, Kite-Powell, & Hoagland, 2007; Laukkanen, 2001; Lee & Iwasa, 2011). MPAs may however not be conducive for reducing conflict because some fishers may experience an increase in yields from the implementation of MPAs, while others get reductions in harvests (Holland, 2000; Ngoc & Flaaten, 2010; Sumaila, 2002; Sumaila & Armstrong, 2006). Our study shows that introducing an MPA, with optimal harvesting, may not only mitigate the negative external effect of CBA on the wild fishery, but it also increases the economic benefits for both users, contributing to reduce conflicts between resource users.

The remainder of the article is organized as follows. In the next section, we present two CBA–wild fishery interaction models. In the first model, we assume that CBA has an external effect reducing the intrinsic growth rate of the wild fish. In the second model, an MPA is established in a portion of the coastal area, which is protected from fishing and aquaculture, while the remaining portion is left for both users of the ocean. Four management regimes are presented in section three: pure open access, open access in the area outside the MPA, and economic optimum with and without an MPA. Due to the complexity of the interactions, analytical results do not provide any insight, hence numerical simulation is conducted in order to investigate the behavior of the models, and is presented in section four. Section five summarizes the major conclusions and suggests some additional future research.

2 | MODELS

2.1 | Basic fishery model

We start with the original situation where the entire area is open for fisheries. Note that for simplicity all time subscripts of time dependent variables are omitted. Suppose the fish stock obeys the logistic law of growth described by the net growth equation

$$\dot{S} = F(S) - y_f = rS\left(1 - \frac{S}{K}\right) - y_f, \tag{1}$$

With S being the stock size, K is the carrying capacity, r is the intrinsic growth rate, $F(S)$ is the natural growth function, and y_f is the harvest rate. Normalizing stock by dividing stock level with carrying capacity gives the net growth function

$$\dot{X} = F(X) - Y_f = rX(1 - X) - Y_f, \quad X = \frac{S}{K}, \quad Y_f = \frac{y_f}{K}. \tag{2}$$

The net benefit function is described as

$$V\left(X, Y_f\right) = \left(Kp_f - \frac{c}{qX}\right) Y_f, \tag{3}$$

where c is a constant unit cost of fishing effort, q denotes catchability, and p_f is constant unit price of fish. All parameters are positive.

2.2 | External effect from CBA on the wild fishery

Now let us introduce CBA that coexists with fisheries in the ocean space. The presence of aquaculture that uses wild juveniles as input is assumed to decrease the intrinsic growth rate of the wild fish stock. In our model we also assume that aquaculture does not affect the distribution of fish. This assumption is reasonable if aquaculture activities are allocated in a minor part of the total ocean area, and operate in a section of limited importance to the wild fish stock (Mikkelsen, 2007).

Let N be produced volume of farmed fish, and the intrinsic growth rate of wild fish can be now redefined as[4]

$$\tilde{r} = \tilde{r}(N) = r\left(1 - \exp\left(-\frac{1}{\alpha N}\right)\right), \qquad \tilde{r} > 0 \text{ for all } 0 < N < \infty, \tag{4}$$

where $\alpha > 0$ is the aquaculture sensitivity coefficient influencing the intrinsic growth rate of wild fish. The expression implies that the intrinsic growth rate of wild fish declines with an increase in aquaculture production in a nonlinear way, with $\tilde{r} = \tilde{r}(0) = r$ and $\tilde{r} = \tilde{r}(\infty) = 0$. Incorporation of a CBA external effect into the intrinsic growth rate of wild fish results in a change in the natural growth function to $F(X, N)$. Particularly, more CBA production means lower natural growth of wild fish, that is, $\frac{\partial F(X,N)}{\partial N} = F_N(X,N) < 0$ and $\frac{\partial^2 F(X,N)}{\partial N^2} = F_{NN}(X,N) \geq 0$.

The net growth equation will now be

$$\dot{X} = F(X,N) - Y_f = \tilde{r}(N)X(1-X) - Y_f. \tag{5}$$

The economic dimensions of the aquaculture operation will be characterized following Hoagland et al. (2003) with adjustments involving different prices for aquaculture and wild fish, nonlinear cost of aquaculture and a nonlinear effect of aquaculture on wild fish intrinsic growth. The aquaculture operating cost (e.g., feed, seed, labor, boats, interest, and other capital items), $C_a(N)$, is a nonlinear and increasing function of total aquaculture production: $C_a(N) = vN^2$, with $\frac{\partial C_a}{\partial N} > 0$ and $\frac{\partial^2 C_a}{\partial N^2} \geq 0$. The investment cost in new aquaculture facilities (e.g., the components and devices of aquaculture), $I(z)$, is a linear function of the increment, z, to the total aquaculture production: $I(z) = bz$.[5] Parameters v and b are positive and constant, denoting the unit operating cost of aquaculture and the cost of investment in new aquaculture facilities for a unit of produced volume of farmed fish, respectively.

For simplicity of exposition while at the same time allowing focus on trade-offs, we assume the market price of fish, p_i ($i = f$ denotes price of wild fish and $i = a$ price of cultured fish), are constant, and the price for wild fish is higher compared to farmed fish. A constant fish price may be a strong assumption as the contribution of aquaculture production to the total fish production may result in lower fish prices for several species as shown by Asche et al. (2001) and Asche (2008). However, for most CBA which is low-intensity and small-scale, the scope for development in the farming process resulting in aquaculture production increase is limited (Asche, 2008). Hence, the local production provided by CBA species is small compared to total worldwide production. For example, the 2013 size of CBA of Atlantic cod in Norway, yellowfin tuna in Japan, and lobster in Vietnam was a relatively insignificant share of about 0.3%, 0.01%, and 0.4%, respectively, of the total worldwide production.[6] We therefore assume that there is no aggregate quantity effect associated with such CBA industries on the global fish price, and set price as an exogenous variable taking a constant value. The total net benefit is defined as the sum of net benefits from the wild fishery and aquaculture taking the form:

$$V(X,N) = \left(Kp_f - \frac{c}{qX}\right) Y_f + \left[p_a N - C_a(N) - I(z)\right]. \tag{6}$$

2.3 | MPA creation

Now let us implement an MPA in the system: subarea 1 includes both the wild fishery and aquaculture, and subarea 2 is an MPA where fishing and aquaculture are forbidden. The total normalized population, hence, consists of two normalized subpopulations, X_1 and X_2, which have the same homogeneous characteristics and $X = X_1 + X_2 = \frac{S_1}{K} + \frac{S_2}{K}$. The total distribution area is set equal to one with subareas 1 and 2 equaling $(1 - m)$ and m, respectively, $0 < m < 1$. Each subpopulation has an individual carrying capacity which is proportional to the size of the subarea. The creation of an MPA raises the possibility of migration or diffusion if there is a difference between the densities of the subpopulation, that is, $\frac{X_1}{1-m}$ and $\frac{X_2}{m}$. We assume that there is net migration from the MPA, where there is higher population density compared to the fishing ground, and the two subpopulations are distributed homogeneously throughout their respective subareas. Therefore, net emigration from the MPA equals net immigration to the fishing ground at a rate $M(X_1, X_2) = \gamma\left(\frac{X_2}{m} - \frac{X_1}{1-m}\right)$, where $\gamma > 0$ is the migration coefficient.[7]

To simplify, we assume that $r_1 = r_2 = r$ in the case without an external effect of CBA. The intrinsic growth rate of the fish stock in the fishable area follows (4) when there are externalities of CBA on wild fisheries. This implies that the presence of CBA reduces the intrinsic growth rate of the fish stock in the fishing ground, with the natural growth rate $F_1(X_1, N)$, implying $\partial F_1(X_1, N)/\partial N = F_{1N}\ (X_1, N) < 0$.

The subpopulation net growths in the two areas are

$$\dot{X}_1 = F_1\ (X_1, N) + M(X_1, X_2) - Y_{1f}\ = r\left(1 - \exp\left(-\frac{1}{\alpha N}\right)\right) X_1\left(1 - \frac{X_1}{(1-m)}\right) + \gamma\left(\frac{X_2}{m} - \frac{X_1}{(1-m)}\right) - Y_{1f}, \quad (7)$$

$$\dot{X}_2 = F_2\ (X_2) - M(X_1, X_2) = rX_2\left(1 - \frac{X_2}{m}\right) - \gamma\left(\frac{X_2}{m} - \frac{X_1}{(1-m)}\right), \quad (8)$$

where, Y_{1f} is the normalized harvest in subarea 1, assumed to be proportional to the population density on the fishing ground. This implies the adjusted Schaefer harvest function $Y_{1f} = \frac{qEX_1}{(1-m)}$ (Flaaten & Mjølhus, 2005, 2010).

The total net benefit from the wild fishery and CBA takes the form

$$V\ (X_1, N) = \left(Kp_f - \frac{c\,(1-m)}{qX_1}\right) Y_{1f} + [p_a N - C_a\,(N) - I\,(z)]. \quad (9)$$

3 | MANAGEMENT REGIMES

3.1 | An open access regime

3.1.1 | External effect from CBA on the wild fishery

The external effect from CBA on the wild fishery depends on produced volume of farmed fish, *N*. We assume that CBA is a competitive industry in the open access fishery case. In open access equilibrium, the steady-state normalized wild stock is $X^{\infty} = \frac{S^{\infty}}{K} = \frac{c}{Kqp_f}$ and the steady-state normalized harvest rate is derived from equation (5) giving $Y_f^{\infty} = F\ (X^{\infty}, N) = r(1 - \exp(-\frac{1}{\alpha N}))X^{\infty}(1 - X^{\infty})$.[8] The steady-state equilibrium wild stock does not depend on CBA production while the steady-state equilibrium wild catch does. The wild catch declines when the CBA production increases.

3.1.2 | MPA creation

When we introduce an MPA in the system, the grounds available for fishing and CBA become smaller than without an MPA. The assumption of the distribution of fish being unaffected by marine farming is kept. In equilibrium, the steady-state normalized stock in the MPA can be calculated as

$$X_2 = -\frac{1}{2}\left(\frac{\gamma}{r} - m\right) + \sqrt{\frac{1}{4}\left(\frac{\gamma}{r} - m\right)^2 + \frac{\gamma m X_1}{r\,(1-m)}}. \quad (10)$$

For open access in subarea 1, the steady-state normalized stock is $X_1^{\infty} = X^{\infty}\quad(1-m) = \frac{c}{Kqp_f}\ (1-m)$. Substituting X_1 with this in equation (10), we can find X_2. The steady-state wild stock size in the entire area will be $S\ = S_1^{\infty}\ +\ S_2 = KX_1^{\infty}\ + KX_2$. From equations (7) and (8), we can find the steady-state harvest rate $Y_{1f}^{\infty} = F_1\ (X_1^{\infty}, N) + F_2(X_2)$. The catch in tons is $y_{1f}^{\infty} = (1-m)\ K[F_1(X_1^{\infty}, N) + F_2(X_2)]$.

3.2 | An optimal management regime

3.2.1 | External effect from CBA on the wild fishery

A manager chooses the level of harvest, Y_f, and the level of investment in CBA, z, in order to maximize the net present value (NPV) of fish production from both CBA and the wild fishery:[9]

$$\max_{Y_f,\, z} \int_0^{\infty} \left\{ \left[\left(Kp_f - \frac{c}{qX} \right) Y_f \right] + \left[p_a N - C_a(N) - I(z) \right] \right\} e^{-\delta t} dt \tag{11}$$

subject to

$$\dot{X} = F(X, N) - Y_f, \quad X\ (0) = X_o\ ,\ X(t) \geq 0, \quad 0 \leq Y_f \leq Y_{f\,max}, \tag{12}$$

$$\dot{N} = z, \quad N\ (0) = 0, \quad N(t) \geq 0, \quad z \geq 0. \tag{13}$$

The current-value Hamiltonian for this problem may be described as

$$H^c = \left[\left(Kp_f - \frac{c}{qX} \right) Y_f \right] + \left[p_a N - C_a(N) - I(z) \right] + \lambda \left[F(X, N) - Y_f \right] + \beta z. \tag{14}$$

With λ and β being the adjoint variables measuring the shadow prices of the associated state variables X and N, the first-order conditions for an optimal solution include

$$\lambda = \left(Kp_f - \frac{c}{qX} \right), \tag{15}$$

$$\beta = b\,, \tag{16}$$

$$\dot{\lambda} = \delta\lambda - \frac{cY_f}{qX^2} - \lambda F_X(X, N), \tag{17}$$

$$\dot{\beta} = \delta\beta - p_a + 2\nu N + \lambda F_N(X, N). \tag{18}$$

The current-value Hamiltonian is linear in the controls, strictly concave in the state variable X, while the growth function $F(X, N)$ is convex in N. This implies that the need to check for the signs of the second derivative of H^c for the current-value Hamiltonian to be concave in X and N, giving the problem a unique solution.[10] The same expression is the case for the optimal management with an MPA.

If the signs of the second derivative of H^c hold, the current-value Hamiltonian is concave, and equations (15)–(18), together with equation (12) can be used to solve for optimal solutions of $X^*,\ N^*,\ Y_f^*$, and the two shadow prices. Equation (16) shows that β, the shadow price of N, is constant and positive, being a traditional “bang–bang” equilibrium, suggesting aquaculture investment should be made as quickly as possible to optimize the size of aquaculture (N^*). The NPV of the wild fishery and CBA at bioeconomic optimum becomes

$$V\ \left(X^*, N^*\right) = \frac{\left[\left(Kp_f - \frac{c}{qX^*} \right) F(X^*, N^*) + \left(p_a N^* - \nu(N^*)^2 \right) \right]}{\delta} - bN^*. \tag{19}$$

3.2.2 | MPA creation

When the original ground is divided into two subareas, one is set aside as an MPA and the other is used for fisheries and aquaculture. We consider the problem where a manager seeks to maximize the net profits from both fishery and aquaculture by choosing the levels of wild harvest, Y_{1f}, and aquaculture investment, z.

$$\max_{Y_{1f},\, z} \int_0^\infty \left\{ \left(Kp_f - \frac{c\,(1-m)}{qX_1} \right) Y_{1f} + \left[p_a N - C_a\,(N) - I\,(z) \right] \right\} e^{-\delta t} dt \tag{20}$$

subject to

$$\dot{X}_1 = F_1\,(X_1, N) + M(X_1, X_2) - Y_{1f}, \quad X_1\ (0) = X_{o1}, \quad X_1\,(t) \geq 0,\ 0 \leq Y_{1f} \leq Y_{1fmax}, \tag{21}$$

$$\dot{X}_2 = F_2\,(X_2) - M(X_1, X_2), \quad X_2\ (0) = X_{o2}\ , \quad X_2\,(t) \geq 0, \tag{22}$$

$$\dot{N} = z, \quad N\ (0) = 0, \quad N\,(t) \geq 0, \quad z\,(t) \geq 0. \tag{23}$$

The current-value Hamiltonian for this problem may be expressed as follows:

$$H^c = \left(Kp_f - \frac{c\,(1-m)}{qX_1} \right) Y_{1f} + \left[p_a N - C_a\,(N) - I\,(z) \right] + \lambda_1 \left[F_1(X_1, N) + M(X_1, X_2) - Y_{1f} \right]$$
$$+ \lambda_2 [F_2(X_2) - M(X_1, X_2)] + \beta z. \tag{24}$$

With λ_1, λ_2, and β being the adjoint variables measuring the shadow prices of the associated state variables X_1, X_2, and N, the first-order conditions for an optimal solution become

$$\lambda_1 = \left(Kp_f - \frac{c\,(1-m)}{qX_1} \right), \tag{25}$$

$$\beta = b, \tag{26}$$

$$\dot{\lambda}_1 = \lambda_1 \left(\delta - F_{1X_1}(X_1, N) + \frac{\gamma}{1-m} \right) - \frac{c\,(1-m)\,Y_{1f}}{qX_1^2} - \frac{\lambda_2 \gamma}{1-m}, \tag{27}$$

$$\dot{\lambda}_2 = \lambda_2 \left(\delta - r \left(1 - \frac{2X_2}{m} \right) + \frac{\gamma}{m} \right) - \frac{\lambda_1 \gamma}{m}, \tag{28}$$

$$\dot{\beta} = \delta\beta - p_a + 2\nu N + \lambda_1 F_{1N}(X_1, N). \tag{29}$$

This is similar to the case of the CBA–fisheries interaction, as equation (26) suggests that aquaculture investment should be made as quickly as possible in order to reach the optimal aquaculture size (N^*). Assuming that a steady-state equilibrium is feasible (i.e., the current-value Hamiltonian is a concave function), equations (25)–(29), together with equations (21) and (22) can be used to solve for optimal solutions of X_1^*, X_2^*, N^*, and Y_{1f}^* and three shadow prices. The NPV of the wild fishery and aquaculture at bioeconomic optimum becomes

$$V\left(X_1^*, N^*\right) = \frac{\left[\left(Kp_f - \frac{c(1-m)}{qX_1^*}\right)\left[F_1\left(X_1^*, N^*\right) + F_2\left(X_2^*\right)\right] + \left(p_aN^* - v(N^*)^2\right)\right]}{\delta} - bN^*. \quad (30)$$

The analytical solutions of the optimal management models with and without an MPA involve multiple roots, are cumbersome, and give no further insight. Hence, we apply a numerical procedure in order to find optimal solutions and to test whether equilibrium states are robust with respect to changes in parameter values.

4 | NUMERICAL SIMULATIONS

In this part, we set values for the parameters $r, K, p_f, p_a, q, c, \alpha, \gamma, v, b, \delta$ (see Table 1) and numerically solve for the cases of open access and optimal management in equilibrium.[11]

4.1 | Open access regime

As can be seen in equations (3) and (5), the equilibrium wild stock level is independent of the presence of CBA, while the equilibrium wild catch is affected by CBA operations. Given the parameter values and a level of CBA production (here we assume $N = 1221 \times 10^{-3}$, which is the optimal size of CBA in the case without an MPA effect), we observe that the total stock size is greater when an MPA is introduced and increases with increasing MPA size (see Figure 1), while the wild catch varies with increasing MPA size. The impact of MPA creation on harvests, as shown in Figure 2, yields a higher wild catch compared to pure open access, except for a relatively large MPA (at least 80% of the entire area).

4.2 | Optimal management regime

We use the software package Mathematica to determine the optimal output variables for a given reserve size. Given the parameter values and model specification, the economic optimal values of normalized

TABLE 1 Parameters for wild fishery and CBA

Variable	Description	Value
α	CBA sensitivity coefficient	3
v	CBA production operating cost	0.4
c	Unit cost of fishing effort	0.3
q	Catchability coefficient	1
p_f	Unit price of wild fish	1.2
p_a	Unit price of farmed fish	1
r	Intrinsic growth rate	0.4
b	Investment cost	0.2
γ	Migration coefficient	0.5
δ	Discount rate	0.07
K	Carrying capacity	1

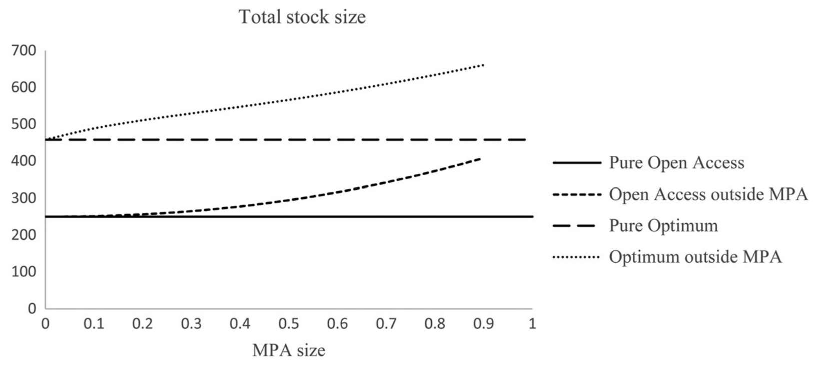

FIGURE 1 Effect of varying size of MPA (m) on wild fish stock size under both open access and optimal management regimes (values in 10^{-3})

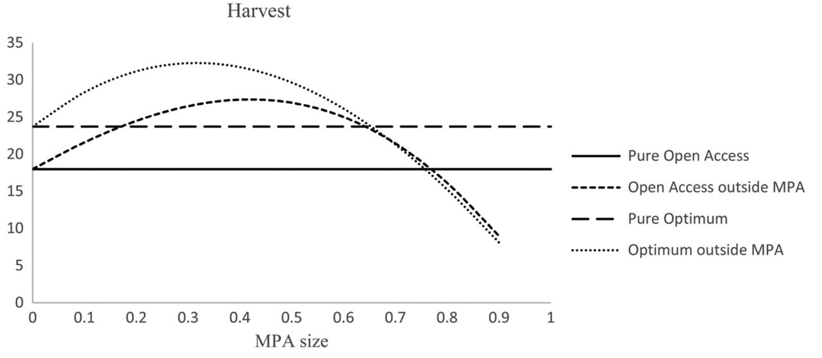

FIGURE 2 Effect of varying size of MPA (m) on wild harvest under both open access (given the optimal level of CBA production: $N = 1221$) and optimal management regimes (values in 10^{-3})

TABLE 2 Simulation results varying m for steady-state equilibria in bioeconomic optimum, and comparing to without an MPA (all values in 10^{-3})

Output variable	Description	Without MPA	With MPA (m) 0.1	0.15	0.2	0.3	0.4	0.5
			438	422	402	358	310	260
S_2^*	Stock size in MPA		51	79	109	171	238	307
S^*	Total stock size	458	489	501	511	530	548	566
N^*	Amount of aquaculture activity	1221	1221	1222	1222	1223	1225	1226
y_f^*, y_{1f}^*	Harvest in tons	24	28	30	31	32	32	30
V^*	Total NPV	8864	8888	8889	8882	8847	8785	8699
V_f^*	NPV from wild fishery	185	208	209	202	167	105	19
V_N^*	NPV from aquaculture	8680	8680	8680	8680	8680	8680	8680

wild fish stock and harvest, aquaculture production level, total net profits and profit for each user in the case with and without an MPA are calculated.[12] We can then estimate stock sizes and harvest rate in volume as in Table 2.[13] Testing for stability of the steady-state equilibrium solutions shows that they are locally asymptotically stable.

The results from the numerical simulation show that the wild fishery (i.e., total stock size and wild harvest) exhibits the same tendencies as that of open access. The optimal stock size increases with an increasing MPA size and is larger than without an MPA, while the optimal wild catch with an MPA varies depending on the MPA size. The optimal wild catch is higher than without an MPA, except for an MPA size greater than 65% of the entire area (see Figure 2). Because the optimal wild catch increases with an increase in MPA size up to about 0.35 and then decreases for a larger MPA size, the wild fishery NPV and total NPV with an MPA show the same general tendencies as that of wild catch. The total NPV can be maximized for a certain MPA size (i.e., $m = 0.15$ in this simulation). The CBA production (i.e., produced volume and NPV) increases slightly for an increasing MPA size and is larger than without an MPA.

For an MPA larger than 55% of the entire area, the wild fishery NPV is negative. This is due to the fact that despite a larger MPA increasing spillover to the fishery, it also reduces the share of the population that becomes available for harvest (Grafton, Kompas, & Lindenmayer, 2005). When increasing MPA size such that $m > 0.55$, the gain from the increase in spillover as a result of the larger biomass, is smaller than the cost of the loss in harvest from the area prior to reserve creation. Hence, there would be no incentive to expand the MPA any further. In this simulation, it should be noted that an MPA is beneficial to fishers (i.e., fishery NPV for a certain size of MPA is greater than without an MPA) only if the negative impact of CBA on the intrinsic growth rate of wild fish stock is large enough (e.g., presence of CBA reduces the intrinsic growth rate of wild fish stock by at least 65%, or the CBA sensitivity coefficient α must be greater than 2).

These numerical simulation results are an illustration of the model, and are a function of the chosen parameters. We therefore study the effect of minor changes (10% increase) for each parameter value on the optimal solutions for situations both without an MPA ($X^*, Y_f^*, N^*, V^*, V_f^*, V_N^*$) and with an MPA ($X_1^*, X_2^*, Y_{1f}^*, N^*, V^*, V_f^*, V_N^*$). The model exposes different sensitivity to change in the parameter values. To enable comparison of this sensitivity, these changes are presented as elasticities, or the ratio of percentage change in the values of output variables to percentage change in the parameter values in the neighborhood of the initial values. This is presented in Table 3.

Table 3 shows that the optimal values in both cases with and without an MPA are robust with regards to the chosen parameter values, except for the intrinsic growth rate, r, price of wild and farmed fish, p_f and p_a, CBA operating costs, v, and CBA sensitivity coefficient, α, each of which suggests that a 10% change in parameter causes more than a 10% change in optimal value. Interestingly, the introduction of an MPA increases the robustness of the model, that is, the elasticities of optimal variables decline when an MPA effect is included in the model. Furthermore, when introducing an MPA the model becomes robust to the perhaps most uncertain parameter, the CBA sensitivity coefficient α.

TABLE 3 Sensitivity analysis

	% change in optimal value												
	Without an MPA						**With an MPA ($m = 0.1$)**						
10% increase in parameter	X^*	Y_f^*	N^*	V^*	V_f^*	V_N^*	X_1^*	X_2^*	Y_{1f}^*	N^*	V^*	V_f^*	V_N^*
α	−2.2	**−10.1**	0.1	−0.2	**−13.0**	0.0	−1.4	−1.3	−6.5	0.1	−0.2	−8.0	0.0
v	2.3	9.2	**−11.2**	**−10.4**	**11.6**	**−11.0**	1.4	1.3	6.2	**−11.2**	**−10.4**	7.5	**−11.0**
c	5.1	0.5	0.1	−0.1	−6.3	0.0	4.4	4.3	−0.1	0.1	−0.2	−7.0	0.0
q	−5.2	−0.9	−0.1	0.1	5.1	−0.0	−4.5	−4.3	−0.2	−0.1	0.2	5.7	−0.0
p_f	−5.2	−0.8	−0.2	0.4	**14.7**	−0.0	−4.5	−4.3	−0.2	−0.2	0.5	**15.2**	−0.0
p_a	−2.5	**−10.4**	**10.4**	**18.8**	**−13.8**	**19.3**	−1.4	−1.4	−6.7	**10.4**	**18.7**	−8.3	**19.3**
r	2.6	**10.4**	−0.1	0.3	**13.2**	−0.0	2.1	2.4	**10.1**	−0.1	0.4	**12.1**	−0.0
b	0.0	0.1	−0.2	−0.3	0.2	−0.3	0.0	0.0	0.1	−0.2	−0.3	0.1	−0.3
γ							0.0	−0.4	0.0	0.0	0.0	0.1	0.0

Note: Numbers in bold show sensitive results.

There is a change in the signs of the elasticities of harvest with regard to the cost of effort, c, from positive to negative in the case without and with an MPA, respectively. An increase in this parameter leads to an increase in harvest in the case without an MPA, because when c increases the total fishing cost increases, resulting in a decrease in fishing effort, E, at maximum economic yield (MEY), and hence an increase in the stock size, X, at MEY. If the absolute value of a decline in E at MEY is smaller than the value of an increase in X at MEY, then an increase in c results in an increase in harvest, Y_f. Introducing an MPA may induce a stronger effect of c on the decline of E compared to the effect on the increase in X_1 at MEY, resulting in a decline in harvest, Y_{1f}. The negative sign of the harvest elasticities for a change in price of wild fish, can be explained in a similar way.

5 | DISCUSSION AND CONCLUSION

Competition between ocean user groups is getting increased attention from scientists (Bishop & Samples, 1980; Hoagland et al., 2003; Jin et al., 2007; Laukkanen, 2001; Lee & Iwasa, 2011). Fishermen and marine farmers are competitive ocean users when aquaculture uses seeds (i.e., juveniles) extracted from the wild, causing an increase in fishing mortality of the wild stock, resulting in the reduction of commercial wild fisheries targeting adult individuals. In this study, we develop a model to analyze the interaction between CBA and wild adult fisheries assuming that the impact of CBA on wild fisheries is via reduction of the intrinsic growth rate of the wild fish stock and hence wild stock and harvest. An MPA can be considered a potential management tool to ensure that sufficient numbers of young fish reproduce for the persistence of the population. We therefore include an MPA effect in this interaction model to test whether or not an MPA can enhance economic benefits of both fishers and farmers. The model is applied for both open access and economically optimal management regimes.

As indicated by the sensitivity analysis, the results, with and without an MPA, are most sensitive to change in price of wild and farmed fish, operating costs of CBA, and the intrinsic growth rate of wild fish. While the last term is a biological parameter, the three first parameters, however, can relatively easily be identified in the market.

Given the assumption of the negative effect of CBA on the intrinsic growth rate of wild fish, our model results in a decline in the wild fish growth rate and hence wild harvest. Implementing an MPA of a certain size may reduce the conflicts of interests between fishers and farmers. Under open access, the steady-state equilibrium wild stock level is independent of the presence of CBA, it increases with increasing MPA size and is larger than without an MPA.[14] The wild catch, however, is negatively affected by CBA production and varies depending on the MPA size. With a certain size of MPA, it is greater than pure open access. This result is somewhat similar to existing studies (Hannesson, 1998; Sanchirico & Wilen, 2001), despite our underlying model including a CBA effect on wild fisheries outside the reserve.

However, in the literature a closed area combined with an optimal management policy is not considered to be as beneficial as pure optimal fisheries management policies in a deterministic model, as aggregate biomass can be increased, but fishing yield decreases for increasing marine reserve sizes (Anderson, 2002; Conrad, 1999). Contrary to this, our results show that the implementation of MPAs of certain sizes combined with optimal management outside the reserve can give better results (e.g., wild catch, CBA production, NPVs) in comparison to optimal management without an MPA. These results are mainly driven by the assumption that CBA reduces the intrinsic growth rate of the fish stock in the fishing ground, resulting in a higher intrinsic growth rate within the MPA compared to the fishable area. The migration of fish into the fishable area will increase due to the higher natural reproductive rate within the MPA, allowing for higher values for both fishery and aquaculture compared to optimal management without an MPA.

Our result is somewhat in agreement with Punt et al. (2010, 2013) and Schnier (2005a,b) who suggest that establishing an MPA can benefit a fishery if the resource has heterogeneous growth. Their models require that an MPA increases the average intrinsic growth rate in the unfished area, while our model places the growth moderation on the other side: CBA reduces the intrinsic growth rate of fish in the fished area. However, our study does not only indicate the positive impacts of MPAs on wild fisheries by definition, it also points to another important impact of MPAs, namely that they may mitigate economic conflicts between ocean resource users.

Our findings are of interest because they show that an MPA of a certain size combined with economically optimal harvest and CBA investment can increase the economic benefits for both users. It should however be noted that there are various issues that need to be considered when using MPAs as a management tool for sustainable fisheries and CBA. For instance, the knowledge of the biology of the target species (i.e., spawning sites and times, critical habitats for juveniles, migration routes) is essential in planning MPAs in order to sustainably manage wild populations and continue to supply economic and societal benefits. We, however, simplify the issues for the MPA implementation in order to discuss the use of MPAs as a management tool for solving conflicts between diverse ocean resource users.

MPAs are a well-known, if also criticized, tool for fishery management and may provide a useful approach to avoid and solve conflicts between ocean users, such as fishers and farmers. However, MPAs also provide other potential benefits such as tourism, research, and educational activities. In future work, considering these activities together with the benefits from fisheries and aquaculture should allow the definition of a broader social value of MPAs.

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[1] In this study, the term MPAs are defined as the no-take zones.

[2] Our model is similar to models presented by Schnier (2005a,b) and Punt et al. (2010, 2013) as they present MPAs that increase the intrinsic growth rate of fish stock. Our model works in the opposite direction: the CBA reduces the intrinsic growth rate of fish stock outside the MPA. We do not, however, only consider the CBA operation as the environmental effect upon the commercial fisheries, but we also include economic conflicts between ocean resource users.

[3] Mikkelsen (2007) modeled a linear external effect of aquaculture on the growth rate of the fish stock.

[4] Simon et al. (2012) indicate that the relationship between the intrinsic growth rate and the natural mortality of the fish stock in early life stages can be described by a decreasing convex relationship. In this article, we assume that CBA development increases the natural mortality of the wild fish stock in early life stages, thus the functional form of the intrinsic growth rate is described as in equation (4).

[5] For simplicity, we assume the investment costs of CBA take a linear functional form so that the optimization problem is linear in the control variable, z, the level of investment in CBA.

[6] The estimates of CBA and worldwide production are based on the Food and Agricultural Organization's (FAOs) FishStat database.

[7] The MPA model of this article is based on the basic MPA model suggested by Conrad (1999) and Hannesson (1998), and presented as Model B in Flaaten and Mjolhus (2010).

[8] The net profit of the fishery is $\pi(S, y_f) = (p_f - \frac{c}{qS})y_f$, where y_f is harvest rate in tons. In open access the net profit becomes zero, and the stock size will be $S^{\infty} = \frac{c}{qp_f}$, while the normalized stock is $X^{\infty} = \frac{S^{\infty}}{K} = \frac{c}{Kqp_f}$.

[9] Note that the fish stock ($S(t)$, $X(t)$, $X_1(t)$, $X_2(t)$), harvest ($y_f(t)$, $Y_f(t)$, $Y_{1f}(t)$), aquaculture acreage $N(t)$, aquaculture investment $z(t)$, adjoint variables $\lambda(t)$ and $\beta(t)$ are all time dependent variables.

[10] H^c is concave in X and $N \Leftrightarrow H^c_{XX} \le 0$, $H^c_{NN} \le 0$, and $H^c_{XX} \cdot H^c_{NN} - H^{c2}_{XN} \ge 0$.

[11] The purpose of this study is to examine whether MPA creation can alleviate the conflicts of interests between wild fisheries and CBA. We therefore consider a case where under economic optimality, both users coexist, and ignore two other potential optimal outcomes (corner

solutions) resulting in only one of the two uses. Given our baseline parameter values, the model produces an internal stable steady-state equilibrium. However, the corner solutions may result if there are significant changes in any parameter or parameter combination.

[12] Given the parameter values, the resulting values of the optimal solutions for fish stock, harvest, and aquaculture production are inserted into the second derivative of the current-value Hamiltonian function, H^c, subject to X and N. The results show that the signs of the second derivative of H^c satisfy the demands for a concave current-value Hamiltonian function, implying the problem has a unique solution.

[13] The stock sizes, S_1^*, S_2^*, S^*, and catch in weight, y_f^*, y_{1f}^*, can be calculated as follows:

$$S_1^* = KX_1^*; \quad S_2^* = KX_2^*; \quad S^* = S_1^* + S_2^*; \qquad y_f^* = KY_f^* = KF(X^*, N^*);$$

$$y_{1f}^* = (1-m)KY_{1f}^* = (1-m)K[F_1(X_1^*, N^*) + F_2(X_2^*)].$$

[14] The fact that CBA in our analysis does not affect the equilibrium stock under open access is basically a consequence of applying the Gordon–Schaefer model combined with the assumption that CBA reduces the intrinsic growth rate of the fish stock. Applying a cohort model, or a two-stock model, with a mature and an immature substock could have allowed the CBA to impact the equilibrium stock in the fishery. However, in principle a two-stock model with stock interactions may have similar characteristics to that of a one-stock model with an effect via the intrinsic growth rate.

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PAPER II

Bui Bich Xuan, Erlend Dancke Sandorf and Margrethe Aanesen. Informing Management Strategies for a Reserve: Results from a Discrete Choice Experiment Survey.

Informing Management Strategies for a Reserve: Results from a Discrete Choice Experiment Survey

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Abstract

It is well-known that operating within the boundaries of a national park provides commercial actors with the opportunity to charge a price premium, though this has to a lesser degree been demonstrated for marine protected areas. We estimate national tourists' willingness-to-pay a price premium for boat trips in the Nha Trang Bay Marine Protected Are, Vietnam, using a discrete choice experiment. Our results show that tourists are willing to pay an average price premium of 18 USD per trip for a large improvement in environmental quality, and that avoiding the loss of jobs for local fishermen is of minor importance. Furthermore, the economic benefits generated from management scenarios that combine biodiversity restoration and environmental quality improvement within the reserve sufficient to cover additional costs of such improvements.

KEYWORDS: Marine Protected Areas, Discrete Choice Experiment, Ecosystem Service Valuation.

1. Introduction

Marine protected areas (MPAs) may be established to protect biodiversity, sustainably manage fisheries, and develop non-extractive uses of the area, e.g. in the form of "eco-tourism" (Alban et al. 2008). The first two objectives are broadly studied in the literature, and MPAs have been shown to be an appropriate management tool for biodiversity conservation (Halpern 2003; Halpern et al. 2009) and sustainable fisheries (Sanchirico et al. 2006; Schnier 2005b; Schnier 2005a), the latter is, however, still debated. Development of non-extractive activities is often regarded as less important and has therefore received less attention (Alban et al. 2008).

Although the number of MPAs worldwide have increased substantially, from 0.9% to 8.4% of areas under national jurisdiction in the period 1990 – 2014 (Juffe-Bignoli et al. 2014), data suggests that only 20 – 30% of MPAs are effectively managed, with the remaining being regarded as "paper parks" (Depondt & Green 2006). The most important obstacle to the success of MPAs is the lack of funding for management (Depondt & Green 2006). Running an MPA is costly and funding often comes from limited public budgets. Maintaining biological diversity and environmental quality, not to mention making improvements, is challenging. Consequently, how to get a sustainable financial source to cover maintenance of an MPA is a highly relevant question, and especially in developing countries.

It is well known that the use of terrestrial protected areas (national parks) for non-extractive commercial activities, such as eco-tourism, yields a price-premium due to the status of the area as especially serene (Jacobsen & Thorsen 2010). Some studies underline the attractiveness of MPAs for tourists such as coral reefs, biodiversity, sea mammals, and water quality (Bosetti & Pearce 2003; Can & Alp 2012; Madani et al. 2013; Schuhmann et al. 2013; Parsons & Thur 2008; Wallmo & Edwards 2008). Authors show that tourists are willing to pay more than the current fees for improved biodiversity and environmental quality within MPAs, and it has been demonstrated that "eco-tourism" can serve as a source of funding for the management of MPAs (Depondt & Green 2006; Madani et al. 2013).

While the development of MPA-based tourism may increase revenues in the local economy, it may at the same time lead to potential conflicts of interest between tourists and local fishers (Bosetti & Pearce 2003; Milazzo et al. 2002; Ngoc & Flaaten 2010; Lee & Iwasa 2011). In the short-term, fishermen may oppose expanding the MPA for tourism development for fear of losing their jobs due to unavailable fishing grounds, reduced harvest due to smaller fishing grounds, and increased harvesting costs due to having to go further for available fishing grounds. On the other hand, in the long-term, local fishers may benefit from MPA creation or expansion due to positive spillover effects from the MPA to nearby fishing areas, as suggested in the literature (Sanchirico et al. 2005; Sanchirico & Wilen 2001).

In this paper, we use the Nha Trang Bay marine protected area (NTB MPA) in Vietnam as the empirical background. The objective of the NTB MPA is "to enable local island communities to improve their livelihoods and, in partnership with other stakeholders, effectively protect and sustainably manage the marine biodiversity at NTB as a model for collaborative MPA management in Vietnam" (Dung, 2009). However, after one decade of protection, the recovery of biodiversity within the NTB MPA was very low, including both improvements and deteriorations (Tuan 2011).

Insufficient funds for monitoring and enforcement of the protection regulations are presumed to be contributing factors (Van 2013). Currently, the annual management cost of the NTB MPA is 150,000 USD. User fees levied on tourists visiting Mun Island, that is located in the MPA, cover about 80 percent of the cost and the remaining 20 percent comes from government subsidies.[1] Tourists visiting the MPA purchase their boat tickets through tourism companies.

[1] Source of numbers: Nha Trang Bay Marine Protected Area Authority.

Included in the ticket price is an MPA user fee for swimming or diving, which the tourism company transfers to the government for funding the MPA.

The aim of the paper is to elicit national tourists' willingness to pay (WTP) a price premium for boat trips within the MPA using a discrete choice experiment (DCE) survey of Vietnamese tourists visiting the NTB MPA. The motivation given for the price premium is improvements in the environmental quality and increased biodiversity within the MPA, which result from an expansion of the MPA. So far, the application of DCEs to MPAs have been concerned with estimating benefits of environmental goods such as biodiversity, coral cover, endangered species, environmental quality and habitat values (Boxall et al. 2012; Can & Alp 2012; Madani et al. 2013; Stefanski & Shimshack 2015; Schuhmann et al. 2013). To our knowledge, so far no valuation study has included social factors such as unemployment of local fishermen that are affected by environmental improvements in an MPA.

DCE studies of environmental issues on land have considered this factor and show that respondents reaction to local unemployment is somewhat ambiguous.[2] Some studies show a positive WTP to maintain rural employment (Birol & Cox 2007; Morrison 2002; Morrison et al. 1999; Othman et al. 2004), while others show that people do not care about employment effects of a policy change (Adamowicz et al. 1998). As the NTB MPA provides jobs for a significant number of local fishers who live on the islands within the MPA, a loss of their livelihoods may be a consequence of restoring the environmental quality of the MPA. It is therefore relevant to include both social and environmental variables in the survey.

Our results show that tourists are willing to pay, on average, a substantial price premium on the current ticket price for a large increase in environmental quality. Environmental quality is by far the most important factor to national tourists and is larger by an order of magnitude compared to, for example, coral cover. Furthermore, we find that people are almost indifferent

[2] Local unemployment is also denoted non-use value of employment (Bennett and Blamey, 1999)

towards employment effects and that WTP to avoid job-loss is very small. Looking at the welfare effects of simulated management scenarios we find that the benefits generated for improved biodiversity restoration and environmental quality are sufficient to cover the management costs.

The rest of the paper is organized as follows: section two presents a description of the NTB MPA, section three presents survey design, sampling, and model specification. Section four presents the results of the study and discussion of those results, section five presents the management implications and the last section contains conclusions.

2. Study Area

Nha Trang city is located on the coast in the central part of Vietnam. NTB covers approximately 507 km^2 and is a hub of marine biodiversity, marine aquaculture, commercial fishing, tourism, and shipping. The biodiversity in NTB is the highest in Vietnamese coastal waters (Tuan et al. 2002) and it is relatively high for the overall Pacific Ocean (Van Nguyen & Phan 2008) with 350 species of hard coral (accounting for over 40% of all reef-building coral species in the world), 220 species of demersal fish, 160 species of mollusks, 18 species of echinoderms, and 62 species of algae and seagrass. This marine area is considered a major nursery ground supplying fish larvae to other Vietnamese waters and possibly also to Cambodian waters (Dung 2009).

An assessment in 2002 indicated that marine biodiversity in NTB had declined substantially (Tuan et al. 2002). Coral reefs and some commercial fish stocks were in poor condition, and many species had become locally extinct due to human activities such as overfishing, aquaculture, tourism and urban run-off (Dung 2009). Recognizing the importance of biodiversity in NTB and the increasing pressures on marine resources, the government established the first MPA in Vietnam here in 2002 (initially named Hon Mun MPA and later changed to NTB MPA) with a total protected area of 160 km^2 consisting of nine islands and their surrounding waters (Figure 1). The MPA is regulated into three zones with different levels of use and protection (Tuan et al. 2002). First are the core zones (red color) with an area of 16 km^2, stretching from the water's edge out to 300 meters, including four islands with the highest biodiversity, and allowing tourism only. Second are the buffer zones (yellow color) from the core zones' border out to 300 meters and/or 300 meters from the water's edge of the remaining islands. Traditional fishing gears, marine aquaculture and tourism are allowed in these areas, but no trawling. Third are the transition zones (light blue color), open to all activities but including limitations on bottom trawl with regard to mesh size and engine power.

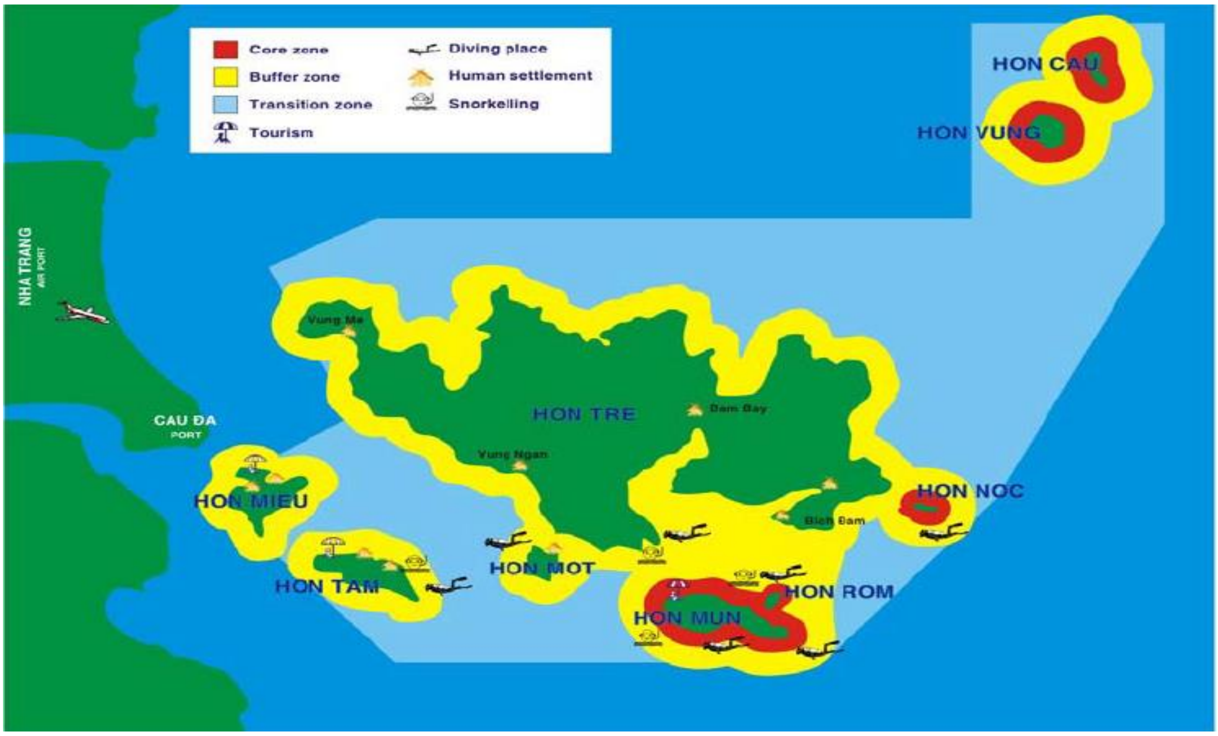


Figure 1. The location of NTB MPA. Source: Tuan et al. 2002

Although the NTB MPA was established with the main purpose of biodiversity conservation, it failed to achieve this goal (Tuan 2011). Two main reasons have been suggested are: i) unplanned and unregulated human activities within the MPA have increased the pressure on local resources, and ii) the regulated core zones are too narrow to ensure biodiversity restoration and prevent marine environmental pollution (Dung 2009). Expanding the core zone and changing management policy may improve marine biodiversity, coral reef cover and coastal environmental quality. However, the financing for sustaining and running the NTB MPA is indicated to be one of the greatest challenges (Dung 2009) and expanding the core zone of the MPA is expected to increase management costs.

Initial funding for establishing and running the NTB MPA was provided for four years by the Global Environment Fund through the World Bank, the Danish International Development Agency, the International Union for Conservation of Nature (IUCN), and the Government of Vietnam. At the end of 2005, external funding by donations came to a halt (Thu et al. 2005). Since then income from user fees levied on tourists when visiting the Mun islands, the core zone with the richest biodiversity, has provided the main funding for running the NTB MPA. One possible way to cover the additional management costs is to increase the boat trip ticket price for tourists taking the sea/islands tour within the MPA. The managers' challenge is to explore whether there is WTP increased fees among the visitors and determine the size of this potential increase.

3. Survey Design and Model Specification

3.1. Survey Design

The good to be valued in this DCE is a hypothetical core zone expansion of the NTB MPA. Following a literature review and focus group discussions, four attributes and their levels were determined. These were coral cover, environmental quality, fishermen's job losses and ticket price.

The attribute live hard coral cover represents one type of biodiversity in the NTB MPA. Expanding the core zones in the MPA is a strategy to increase biodiversity in general as well as the abundance and species of coral in particular. Halpern (2003) showed that on average, creating a reserve may over a period of time double density, triple biomass, and raise organism size and diversity by 20 - 30% relative to an unprotected area.

The change in coral cover is indicated to have positive impact on tourists' utility (Parsons & Thur 2008; Schuhmann et al. 2013). The coral cover attribute has three levels; an average of 13% live hard coral cover was recorded in the last assessment (Tuan et al., 2005) and this is taken as the status quo (SQ) attribute level. 20% and 30% are two alternative levels for this attribute after 10 years of core zone expansion.[3]

Phu et al. (2013) show that tourists are usually very concerned about the environmental quality within the NTB MPA, and in particular waste and seascape disturbance. Tourists can see the solid waste everywhere from the dock where they await their tour boats, along the edge of the islands, on the islands, and/or on the seabed where they scuba dive and snorkel.[4] The presence of floating plastic bottles belonging to seed lobster traps and floating aquaculture cages which tourists can see when they are on the boat trips, are believed to reduce the scenic view. Hence, the coastal environmental quality in terms of visible waste and seascape disturbance is included as an attribute. This attribute takes three levels: low, medium, and high quality with low quality also considered as the SQ level.

Job loss of the fishermen is the last non-cost attribute in the design. The re-zoning scheme of the NTB MPA, as discussed above, may have short-run unemployment effects for the local fishermen. Vietnamese tourists visiting the MPA may be concerned about these effects when

[3] The assumed levels of coral cover are based on NTB biological indicators as follows. Firstly, before 1994, the average coral cover in the NTB was recorded to be 30% (Ben et al. 2015). Secondly, coral cover in the core zone - Mun island has increased by 50% after 4 years of protection (Tuan et al. 2005). Thirdly, coral reefs are distributed mostly along the coast and around the islands within the NTB MPA (Tuan et al. 2005), of which, some places not located in the current MPA core zone, have high quality reef habitats. If the expansion of the core zone takes place in these sites, this may promise a quick increase in coral cover.

[4] About ten tons of solid waste are discharged into the NTB MPA daily (Anon, 2015). Although solid waste collection efforts are made, they are inefficient because the waste is distributed throughout the MPA by sea currents and pushed again by tides onto the beaches.

they make choices regarding the alternatives of the MPA management plan. About 1,100 households live on the islands within the MPA, of which 80% are fishermen, and 75% of them engage in fishing near shore around these islands (Van 2013). Assuming that a maximum of 30% of existing near shore fishermen would lose their job, the levels of the job loss attribute are set to be 0, 50, 100, and 200, where 0 is the SQ-level.

Finally, the payment vehicle is a price premium for boat trip tickets within the MPA, which takes four levels; 20, 50, 100, and 200 thousand Vietnamese Dong (VND).[5] Respondents were told that the added revenue from the increased ticket price will accrue to a marine environmental fund that will be used to cover management costs for increased biodiversity and improved environmental quality in the NTB MPA. Table 1 presents the attributes and their levels.

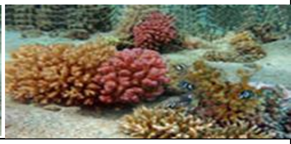
Table 1. Attributes and levels used for the DCE.

Attribute	Coral cover	Environmental quality	Job loss	Cost
Variable name	Coral	Environment	Job loss	Cost
Description	The average cover of live hard coral within the MPA	Visible waste and floating traps/cages within the MPA	The number of lost jobs for fishermen	Increase in ticket price of sea/islands tour (1000 VND)
Status quo (SQ)	13%	Low	0	0
Level 1	20%	Low	0	20
Level 2	30%	Medium	50	50
Level 3		High	100	100
Level 4			200	200

[5] The exchange currency in 2015 is 1USD = 22,547 VND (The State Bank of Vietnam, www.sbv.gov.vn/)

The four attributes and their levels constitute 96 possible combinations. Based on parameters obtained from two pilots, an efficient design for a multinomial logit model was developed. The design assumed attribute level balance and was constructed using the D-efficiency criterion, aiming at generating parameter estimates with as small standard errors as possible (Scarpa & Rose 2008). The D-efficiency design produced 12 choice sets, which were in turn randomly blocked into two different questionnaire versions (6 choice sets per block). An example of a choice card is presented in Table 2.

Table 2. Example of the choice card.

Attribute	Status quo	Plan A	Plan B
Coral cover	13% 	30% 	20% 
Environmental quality	Low quality 	Low quality 	High quality 
Fishermen job loss	0	50	200
Increased ticket price (VND)	0	20.000	50.000
I prefer	☐	☐	☐

3.2. Model specification

3.2.1. Mixed logit model

The analysis of discrete choice data is based on the random utility model. This model assumes that a person's utility from choosing a specific alternative is described by one systematic and one random component. The former is a function of observed variables while the latter includes all unobserved variables, which impact on the utility of choosing a specific alternative. Hence, utility can be expressed as;

$$U_{nit} = \beta_n x_{nit} + \varepsilon_{nit} \quad (1)$$

where, U_{nit} is the utility of individual *n* obtained from choosing alternative *i* in choice situation *t*, x_{nit} is a vector of observed attributes of alternative i, β_n is a vector of random parameters, and ε_{nit} is a random component of the utility of alternative *i* and is assumed to be independent and identically distributed (iid) following a type 1 extreme value distribution.

Because we assume heterogeneous preferences, the model parameters are not fixed across the population and are weighted following a pre-specified distribution. Let the vector of random parameters be denoted as:

$$\beta_n = \beta + \Gamma\eta_n \quad (2)$$

where β is a vector of population means of the parameters, Γ is the lower Cholesky matrix with standard deviations on the diagonal and η_n is a draw from a specified distribution, often the normal or log-normal. We allow for preference correlation by letting the off-diagonal elements of Γ be non-zero.

We use the mixed logit model (MXL) to estimate national tourists' preferences for increasing biodiversity, environmental quality and loss of fishers' jobs within the NTB MPA. The benefit of the MXL is that it allows the modeling of heterogeneous preferences across respondents and correlated parameters. Tourists who like seeing coral might also prefer high coastal environmental quality, thus the coefficients of coral cover and environmental quality attributes may be correlated. Similarly, tourists who have preferences for good environmental conditions might prefer a reduction of the number of fishermen within the MPA.

The mixed logit probability of the sequence of choices made by a respondent is the integral over the product of the logit formula for all possible values of β

$$\Pr(i_n|x_n) = \int \prod_{t=1}^{T} \frac{\exp(\mu\beta_n' x_{nit})}{\sum_{j\in C} \exp(\mu\beta_n' x_{njt})} f(\beta)d\beta \quad (3)$$

where μ is a scale parameter that is typically set equal to one and is inversely proportional to the standard deviation of the error terms, and $f(\beta)$ is a density function. The integral does not have a closed-form solution and is approximated through simulation (Hensher et al. 2005). Aggregating over all respondents yields the likelihood function, and for ease of estimation one normally estimates the parameters by maximizing the log of the likelihood function (Train 2009).

3.2.2. Measurements of unconditional WTP and Welfare

The results from a DCE model can be used for deriving the marginal attribute WTPs to determine the amount of money individuals are willing to pay in order to get some benefit from the implementation of a project or new policy, and the change in consumer surplus (ΔCS) resulting from this project or change of policy (Train, 2009).

Unconditional mean WTPs are calculated as the ratio of the parameter estimates of a non-cost attribute and the cost attribute, *ceteris paribus*.

$$WTP_k = -\frac{\beta_k}{\beta_c} \tag{4}$$

where β_k and β_c are the estimates for X_k (non-cost attribute) and X_c (cost attribute) respectively. The change in consumer surplus is calculated as the change in utility (in monetary terms) that a person receives in the new management scenario compared to the current situation.

$$\Delta CS = -\frac{1}{\beta_c}\left[\ln\left(\sum\nolimits_{j\in C} \exp\left(V_{njt}^{1}\right)\right) - \ln\left(\sum\nolimits_{j\in C} \exp\left(V_{njt}^{0}\right)\right)\right] \tag{5}$$

where the superscripts 0 and 1 refer to the current situation and a new management scenario, respectively.

3.3 Sampling

The DCE survey was administered in April 2015 during the weekends. Face-to-face interviews were conducted with a convenience sample of 150 Vietnamese visitors (not including two pilot groups) participating in boat island tours within the NTB MPA. Each respondent answered six choice cards. They were asked to choose their most preferred alternative, and were told that there were no right or wrong answers. The respondents were also told that reasons for increasing the ticket price was, amongst others, to raise funds for the environmental improvement of the MPA. In order to reduce hypothetical bias the questionnaire contained a cheap-talk script to remind respondents of their budget constraint (Cummings & Taylor, 1999; Carlsson et al., 2005). It also included questions regarding socio-demographic variables and attitudes towards environmental protection in general.[6]

Sample characteristics are given in Table 3. The sample has a higher male share (61%) compared to the national average (49.5%). This is because when we requested a family to answer the questionnaire in most cases a man was the representative. Only 5% of respondents are above 50 years of age while the official national recorded number is 17%. The reason may be that very few older tourists wanted to answer the questionnaire because they cannot read or had forgotten their glasses, or they allowed their accompanying children to answer the questionnaire. The official average education at undergraduate level was 7.3% in 2014, while it is 73% of sample in this study.[7] The average income of the respondents is about 300 USD/month, but we do not have data on average income levels of the population for a comparison. The surveyed sample in this study is Vietnamese tourists visiting the NTB MPA, which may have different characteristics to the Vietnamese population in general.

Table 3. Descriptive statistics for respondents, n = 150.

Characteristics	**Frequency**	**Percent (%)**
Gender		
Male	92	61
Female	58	39
Age		
19 – 30 years	87	58
31 – 50 years	55	37
51 years and above	8	5
Marital status		
Single	62	41
Married	88	59

[6] The questionnaire is available from the corresponding author upon request.
[7] Reported by General Statistic Office of Vietnam (GSO). https://gso.gov.vn/Default_en.aspx?tabid=491

Education		
High school or below	23	15
Undergraduate	109	73
Graduate	17	11
Occupation		
Labourer	92	61
Others	58	39
Individual income per month (USD)		
< 178	30	20
178 – 356	76	51
> 356	38	25
Residence		
Ha Noi	21	14
Da Nang	7	5
Ho Chi Minh	52	35
Others	70	47

4. Results and Discussion

4.1. Model Estimates

We use two models to estimate Vietnamese tourists' willingness to pay for restoring the environmental quality of NTB MPA. The first model is a standard mixed logit model and the second is a mixed logit model with correlated parameters. Both models are estimated in R using the "gmnl"-package (Sarrias & Daziano 2015) and 1000 standard Halton draws.

The job loss attribute is a measure of social cost, so it is reasonable to argue that people have negative marginal utility for this attribute and thus assume this attribute to be log-normally distributed. However, it is also possible to argue that tourists have positive preferences for the job loss attribute for two reasons. First, if tourists believe that floating juvenile lobster traps destroying the MPA seascape are the consequences of fishermen's activities, they may like to reduce the number of fishermen in order to have a more attractive seascape. Second, if tourists believe that resources are overexploited and fishermen are putting pressure on fish stocks and destroying coral due to their extraction of dead coral for making lobster traps, they may like to reduce the number of fishers in order to conserve coral and limit the fishing pressure. We therefore choose a normal distribution for the *job loss* random parameter. We let estimated parameters of coral cover and environmental attributes follow a normal distribution, and keep the cost attribute parameter fixed. In this study, having a fixed cost parameter makes marginal WTP follow a normal distribution, and we avoid extreme WTP values associated with e.g. a log-normally distributed cost coefficient. We let the environmental attribute enter the model using a dummy specification.

The estimated results of the two models are reported in Table 4. The model accounting for correlation across random parameters gives the better fit. The two MXL models give statistically significant standard deviations for all attributes, indicating that the data supports preference heterogeneity across respondents for these attributes. The signs and statistical significance of all attribute coefficients in the two models are consistent, except for a change in the sign of the mean estimate of the *job loss* attribute in the MXL model with correlation. It is however still insignificant.

Table 4. Model estimate results (standard errors in parentheses).

	MXL	
	without correlation	with correlation
Fixed parameters		
cost	-0.004(0.002)**	-0.005(0.002)**
Random parameters		
coral	0.065(0.015)***	0.111(0.018)***
coral_SD	0.080(0.012)***	0.093(0.015)***
med.env	0.939(0.143)***	1.134(0.188)***
med.env_SD	0.950(0.202)***	1.587(0.209)***
high.env	1.648(0.228)***	2.052(0.307)***
high.env_SD	1.090(0.238)***	2.258(0.334)***
job loss	-0.00001(0.0008)	0.0007(0.0009)
job loss_SD	0.006(0.001)***	0.006(0.001)***
Likelihood ratio index	0.126	0.171
Log-likelihood at convergence	-855.22	-804.64
Number of observations	900	900
Number of parameters estimated	9	15
AIC	1728.45	1639.28
Likelihood ratio test		101.17***

Note: 1) med.env = medium environmental quality, high.env = high environmental quality; 2) The numbers are the mean values of normally distributed parameters of the attributes; 3) SD: standard deviations of the same distribution; 4) * p<0.05; ** p<0.01; *** p<0.001.

Table 5. Correlations matrix (upper triangular) and Cholesky matrix (lower triangular and diagonal) – standard error in parentheses.

	coral	med.env	high.env	job loss
Coral	0.0929(0.015)	0.8383	0.8360	0.0369
med.env	1.3305(0.265)	-0.8653(0.300)	0.9979	-0.0610
high.env	1.8877(0.389)	-1.2307(0.370)	-0.1449(0.314)	-0.1242
job loss	0.0002(0.001)	0.0009(0.002)	0.0058(0.001)	-0.0003(0.004)

Because the output of the MXL model with correlation gives a better fit, we will report the results from this model. Given the assumption of normally distributed variables, the means and the standard deviations of *coral* and *high.env* imply that 88% and 82% of the sampled population, respectively, have a positive value for the coral cover and high environmental quality. There are also indications of strong positive correlations among taste intensities for coral cover and environmental quality (see Table 5, upper triangular), implying that an individual, who cares about the corals and would like to see an increase in coral cover, also cares about improvement in environmental quality.

The insignificant estimated mean value of the *job loss* random parameter indicates that on average tourists do not care about fishermen losing their jobs, but the standard deviation is statistically significant, suggesting that tourists have heterogeneous preferences with regard to this attribute. In particular, 45% of the respondents want to avoid unemployment for fishermen with the remaining having a positive utility of job loss. This can be better understood in light of survey results of national tourists visiting the NTB MPA carried out by Phu et al. (2013). They show that tourists believe that catching lobster juveniles has negative environmental effects, such as creating waste (93% of the sample), destroying coral reefs and seascape (92%

and 89% of the sample) and depleting the lobster stock (69% of the sample). Given these results, it is likely that some tourists would prefer to reduce the number of fishermen within the MPA. Looking at the lower Cholesky matrix (Table 5, lower triangular and diagonal) we see that the significant standard deviation for the job loss parameter is caused by cross-product correlations between *job loss* and *high.env*. This implies that the respondents who are more sensitive to the change in environmental quality, from low to high, are likely to have higher (positive) marginal utility of fishermen losing their jobs.

The overall fit of the MXL model with correlation is low by conventional standards used to describe probabilistic discrete choice models. However, there are still many studies in the environmental valuation literature having Pseudo-R^2 less than 0.2 (Birol et al. 2006; Cerda et al. 2013; Othman et al. 2004), so this result is not uncommon. In addition, the chosen attributes in this study, except for job loss, are highly significant factors in the choice of an MPA management scenario.

4.2 Unconditional WTP and Welfare calculation

The unconditional marginal WTPs and the changes in CS are calculated using the results of the MXL model with correlation.[8] Table 6 shows that, the simulated mean marginal WTP is highest for the environmental attribute, while it is smaller for increases in coral cover and insignificant for fishermen's job loss. These estimates indicate that visitors are willing to pay a small amount for an increase in live hard coral cover, and much more for an improvement in the environmental quality. In other words, Vietnamese tourists are far more concerned about environmental quality within the MPA than about the coral cover and job loss for fishermen. Our findings are in concert with other valuation studies of protected areas on land, indicating that respondents from outside a region usually have a positive value for biodiversity and

[8] The unconditional marginal WTPs and the changes in CS are the simulated estimates using 150,000 draws, which corresponds to 1000 draws per individual.

environmental quality attributes (Cerda et al. 2013) but care less about social attributes, such as local employment effects (Adamowicz et al., 1998).

Table 6. WTP estimates generated by the MXL model with correlation (USD/trip/person)

Attribute	2.5% quantile	Median	Mean	97.5% quantile
Coral	0.667	0.973	0.988	1.975
med.env	5.731	9.956	10.272	24.107
high.env	11.807	17.956	18.418	39.021
job loss	-0.008	0.006	0.007	0.034

There are reasonable explanations for the dominance of the environmental attribute. From the survey design we are looking at relatively small changes in coral cover (from 13% today to a maximum of 30%), but large changes in environmental quality (low quality with lots of visible waste and seascape disturbance to a situation with no visible waste and nice sea-view). In addition, tourists can see the solid waste everywhere within the MPA together with the floating plastic bottles of the seed lobster traps and floating aquaculture cages over the sea surface. The beauty of coral reefs is, however, only visible to the tourists who are actually diving or snorkeling, and these account for only 2% of the total number of national tourists visiting the MPA.[9] Hence, the visual presentation of the environmental attribute in the choice cards as well as the visible waste presented many places in the NTB probably have a strong influence on tourists' choices in comparison to the coral attribute.

In Table 7, we present the simulated estimates of the change in consumer surplus (ΔCS) for twelve possible management scenarios relative to the current situation. As expected, ΔCS increases as we move to management scenarios with improved environmental conditions. All scenarios with low environmental quality have low ΔCS (between 0.4 and 1.6 USD), whereas

[9] Nha Trang Bay Marine Protected Area Authority.

the ΔCS for all scenarios with high environmental quality is substantial (between 18 and 20 USD). Hence, the single most important attribute that affects the tourists' welfare is environmental quality, while the change in coral cover and fishermen's job loss are both of minor importance. This matches the fact that despite current waste collection efforts, solid waste is still among the most serious issues in the NTB MPA (Anon 2011; Anon 2014; Anon 2015; Anon 2013; Anon 2009). In addition, Vietnamese tourists visiting the NTB express that they are very concerned with regard to the environmental quality at the NTB MPA (Phu et al. 2013).

Table 7. New management scenarios and their changes in welfare, using the MXL model with correlation

Scenario	Attributes			ΔCS (USD/trip/person)			
	Coral cover	Environmental quality	Job loss	2.5% quantile	Median	Mean	97.5% quantile
SQ	13%	Low	0	0	0	0	0
Scenario 1	20%	Low	50	-0.336	0.373	0.415	1.809
Scenario 2	20%	Low	100	-0.734	0.679	0.761	3.505
Scenario 3	20%	Low	200	-1.532	1.292	1.453	6.879
Scenario 4	20%	High	50	11.992	18.321	18.833	40.600
Scenario 5	20%	High	100	12.086	18.627	19.179	42.049
Scenario 6	20%	High	200	12.134	19.228	19.871	44.221
Scenario 7	30%	Low	50	-0.244	0.472	0.514	1.999
Scenario 8	30%	Low	100	-0.644	0.777	0.860	3.658
Scenario 9	30%	Low	200	-1.442	1.386	1.551	7.073
Scenario 10	30%	High	50	12.063	18.417	18.932	40.809
Scenario 11	30%	High	100	12.158	18.722	19.278	42.216

Scenario 12	30%	High	200	12.204	19.322	19.970	44.396

5. Management Implications

Ten years after the establishment of the NTB MPA, it has failed to achieve the main goal of biodiversity restoration (Tuan 2011). While many factors contribute to this failure, it is believed that lack of funding for management is the most important (Dung 2009; Van 2013). In this paper, we use a valuation survey to test whether national tourists are willing to pay a price premium for visiting the MPA, when the premium should contribute to cover costs of biodiversity restoration and improved environmental quality. The results show a positive willingness to pay to contribute to restoration of the environmental quality of the MPA. The question remains whether this WTP is sufficient to cover restoration costs?

The total price premium for national tourists can be calculated by multiplying the per unit change in consumer surplus (ΔCS) (see Table 7) with the number of national tourists visiting the NTB MPA. The number of national tourists visiting NTB MPA was about 500,000 annually in the period of 2010 – 2014.[10] One must, however, take into consideration reduced demand from national tourists if a new management strategy that implies higher boat trip ticket price is implemented. On the other hand, better environmental conditions may attract new tourists. As we do not have any data on price-elasticities for this type of service, we are not able to estimate changes in demand due to a change in boat ticket price within the MPA.

In recent years, the annual management costs for the NTB MPA have been approximately USD 0.15 million. In addition, the new management scenarios suggested in our survey will increase the costs of managing the NTB MPA, but we do not have any estimate of the cost-increase required for the improvements. However, one reference of relevance is the total costs for establishing and operating the NTB MPA in the initial four years of the project (from 2002 to

[10] Nha Trang Bay Marine Protected Area Authority.

2005), which was about 2 million dollars (Nam & Herman 2005), which means on average USD 0.5 million per year. We use Scenario 12 in Table 7 as an example to calculate the total ΔCS and to have a comparison to these management costs. If we assume that that an increase in ticket price does not affect demand, with 500,000 annual visitors, the expected ΔCS is USD 9.985 million. Using the 2.5 and 97.5 percentiles of the distribution yields the lower- and upper-bound estimates, which are USD 6.1 million and USD 22.198 million, respectively. These estimated values are off the same magnitude as the above-mentioned costs.

A sensitivity analysis, assuming that the number of tourists drop by 90 percent to 50,000 national tourists, shows that the average ΔCS is USD 0.999 million, with lower- and upper-bound values of USD 0.61 million and USD 2.22 million. Even if we use the lower-bound mean estimate of USD 0.61 million, the ΔCS is four times as high as the current management costs of USD 0.15 million and also higher than the initial funding of the project of USD 0.5 million per year. This result indicates that it is economically viable to implement alternative management scenarios, which secure high environmental quality in the NTB MPA.

These results do have policy implications. First, we find that respondents have strong preferences for environmental quality. This suggests that policy makers should, at the outset, focus on policies to address issues related to environmental quality. While we are careful not to make policy prescriptions, a few potential programs could be explored. Effective means of collecting and disposing of waste could prove beneficial. In addition, the presence of juvenile lobster fishery and aquaculture is indicated to have negative effects on both coral reef habitats and environmental quality, through extracting dead corals for making lobster traps, destroying scenery with floating aquaculture cages and juvenile lobster traps, and directly discharging into the sea bed the waste from aquaculture activities (Phu et al. 2013). Therefore, policies aimed at regulating these activities may not only increase tourists' utility through reducing solid waste and give more attractive sea views, but also reduce pressure on coral beds.

We also find evidence that there is a strong correlation between preferences for better environmental quality and increased coral cover. Hence, expanding the NTB MPA core zone in combination with some regulations and policies on environmental protection within the MPA would not only increase tourists' utilities but also contribute to biodiversity conservation and sustainable tourism development. It goes without saying, that any new policy implemented within the MPA must be accompanied by proper systems for monitoring and enforcement. This study indicates that there may well be possibilities to fund such increases in management cost via a price premium on the ticket price.

Beside unemployment effects there are alternative negative short-term effects of the MPA core zone expansion on local fishermen such as reduced catches and increased fishing costs. Therefore, to soften the negative short-term impacts of this management policy on local fishermen, the increased income from tourism related to the MPA may be used partly to secure livelihoods for those who are most negatively affected. For example, local fishermen could be compensated for losing their jobs or avoiding specific areas under a 'payment for environmental services' scheme as suggested by Schuhmann et al. (2013). Moreover, they could also receive support for alternative income generation as was carried out in the initial the NTB MPA project, i.e. organize relevant courses to provide knowledge and skills for alternative livelihoods such as handicrafts, animal husbandry, tourism service and trading, and give financial support programs connected with these activities (Thu et al. 2005).

6. Conclusions

The aim of this study is to use a DCE to derive national tourists' WTP for seeing biodiversity restored and environmental quality improved in the NTB MPA in Vietnam. Based on marginal WTP for selected attributes, the change in consumer surplus for national tourists visiting the MPA can be calculated. Applying estimates on change in consumer surplus and costs of management scenarios, managers can assess whether it is economically viable to implement new management scenarios. The results of this paper show that although there are heterogeneous preferences across respondents for environmental attributes connected to the MPA, on average there is positive and significant WTP for improving coral cover and environmental quality. Estimation of change in consumer surplus and restoration costs suggests that management scenarios focusing on improving environmental quality within the NTB MPA are economically profitable.

Our findings add to the empirical evidence suggesting that environmental protection and biodiversity conservation is not only good ecological policy, but viable economic policy as well. In addition, our findings also show that the tourism values of an MPA can be provided by both the ecological attributes (i.e. coral cover and environmental quality) and the social attribute (i.e. employment opportunities or losses), where the former is normally presented in the literature of MPA valuation while the latter is not.

The survey focused solely on national tourists, while international tourists constitute 23%.[11] Future research should include both groups of tourists. Changes in the NTB MPA management policies could increase benefits not only to tourists, but to other stakeholders as well (e.g. fishermen, fisheries-based aquaculture, tourism operators, research and education, etc.). This study, however, discusses only economic benefits from tourism and the operating costs of the MPA. Future research should also include various stakeholders' preferences with regards to the changes in MPA management policies to provide a more comprehensive picture in order to efficiently achieve common goals of fishery management, conservation, research, education and tourism development. Also, a full benefit-cost analysis is necessary for more fully informed decision-making regarding new management strategies.

Acknowledgments

[11] The share of international tourists is calculated from the tourism statistical data of Khanh Hoa Province (http://www.nhatrang-travel.com/index.php?cat=3003).

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PAPER III

Bui Bich Xuan. Extractive and Non-extractive Values of a Marine Protected Area: A Bio-economic Model Application.

Extractive and Non-extractive Values of a Marine Protected Area: A Bio-economic Model Application

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Abstract

MPAs are often established for multiple goals, and hence provide multiple benefits involving different stakeholders. These benefits, however, are most often studied separately. The inclusion of multiple benefits of an MPA in the model may give more complex economic implications than predicted in studies solely focusing on one benefit. This study presents a bioeconomic model that allows for the incorporation of fisheries and tourism values related to the MPA in order to capture the complexities of managing the multiple uses of a natural resource. Estimating a tourism value from a discrete choice experiment, and including data from the anchovy fishery, we find that the optimal reserve is substantially larger than the current one. The expansion of MPA size for an optimum may result in the short-term negative impacts on local fishers. A part of the tourism revenue could however be used to compensate for the losses of the local fishers to soften these negative impacts.

Keywords: *Marine Protected Area, Fishery, Tourism, Bioeconomic Model, Management*

1. Introduction

Marine protected areas (MPAs) have often been established for the purpose of protecting and recovering biodiversity and habitats (Balmford et al. 2004), and have also been seen as an alternative fisheries management tool (Rodwell et al. 2003). Restoration of marine biodiversity and seascapes of sub-marine areas due to such protection is attractive for tourism and other recreational activities (Alban et al. 2008). In this way, MPAs are not only a management tool for fisheries, but they may also provide resources for tourism development.

In the economic literature, these two benefits provided by MPAs are often evaluated separately with the help of various methodologies. Researchers often prefer using bioeconomic models to analyze the benefits of MPAs for fisheries (Hannesson 1998; Conrad 1999; Flaaten & Mjølhus 2010). Whereas, to evaluate tourism or recreational benefits provided by MPAs, non-market valuation methods such as revealed preference techniques (i.e. travel cost, hedonic pricing, production function methods) and stated preference techniques (i.e. contingent valuation and discrete choice experiments) are often preferred (Bhat 2003; Schuhmann et al. 2013; Boncoeur et al. 2002).

Some studies have taken into account the combination of either non-extractive and extractive values or non-use and use values in bioeconomic models to maximize social welfare in connection to natural resources (Alexander 2000; Bulte et al. 1998; Clark et al. 2010; Moyle & Evans 2008; Rondeau 2001). Creating an MPA can be considered as an investment in natural capital, and this natural asset is often a provider of multiple services, and hence multiple benefits (Alban et al. 2008). In the MPA literature, however, there are a few studies that take into account the combination of multiple benefits generated by MPAs. Boncoeur et al. (2002) used a bioeconomic model to analyze the impacts of an MPA creation on both fishing and ecotourism. Their results show that implementing an MPA does not only increase the benefits for the fisheries, but it also generates additional income through tourism activities (i.e. seal watching). In this vein, Merino et al. (2009) presented a bioeconomic model that permits the partial evaluation of a three-zone MPA system as regards to coexistence of fisheries and tourism activities. Empirical data of fishing and tourism in the Medes Islands MPA was used for an application of the model. They show that the revenues from tourism are substantially greater than fisheries revenues. Lee & Iwasa (2011) also use a bioeconomic model to determine the optimal size of the MPA for maximizing the total benefits generated by fisheries and tourism. To our knowledge, no published study integrates non-extractive values estimated from a discrete choice experiment (DCE) into a bioeconomic MPA model in order to maximize benefit from multi-use of natural resources (see however Armstrong et al. (2015) who insert non-use values estimated from a DCE into a bioeconomic model of optimal management of interactions between renewable and non-renewable resources).

This paper presents the basic bioeconomic model of MPAs as a fisheries management tool (Conrad 1999; Hannesson 1998) including non-extractive (i.e. tourism) value generated by MPAs, in order to investigate the optimal management of multiple services provided by the MPA. This study therefore provides two main contributions to the literature: Firstly, it presents a framework for combining both bioeconomic modelling and non-market valuation approaches. This framework allows estimation of multi-benefits (i.e. extractive and non-extractive value) generated by MPAs. Secondly, it applies empirical data from both the anchovy purse seine fishery in the south-central of Vietnam and the Nha Trang Bay (NTB) MPA tourism values, which derived from a discrete choice experiment survey, for a maximization of total welfare from multi-use of natural resources. Different to earlier studies on the multi-activities context (see Boncoeur et al. 2002; Merino et al. 2009; Bulte et al. 1998; Horan & Shortle 1999), where non-market value of the resource (i.e. non-consumptive and non-use value) is often a function of the natural resource (e.g. seal stock, whale stock, fish stock), in this paper tourism value is formulated as a function of the size of the MPA. Moreover, MPA size is considered as a control variable in the maximization problem of this study. The next part of the paper presents the research site and research subjects. Models and data are presented in the third and fourth sections. The fifth section is the numerical simulation and discussion, the last section presents the conclusion.

2. Study Site and Subjects

Khanh Hoa is a coastal province in south-central Vietnam. Fisheries and tourism are among the most important economic industries of Khanh Hoa province. Fisheries in Khanh Hoa are open access and multispecies, using various types of gears such as gill net, longline, trawl net, purse seine, and lift net. A total of almost 10,000 vessels fish in Khanh Hoa waters, of which less than 8% have an engine power greater than 90 horsepower (HP), allowing offshore fishing.[1] Hence, the majority of fishing boats in Khanh Hoa are small scale and operate in the coastal zone. The annual average revenue of the fisheries is about 273 million USD, contributing 13.5% to the gross domestic product (GDP) of Khanh Hoa province during the period of 2011 – 2015.[2]

Tourist activities in Khanh Hoa are mostly characterized as island tourism. Khanh Hoa has a long coastline of 520 km, about 200 islands, and six bays and lagoons. The number of tourists visiting Khanh Hoa has been increasing at an average rate of 18% annually from 2011 to 2015. The annual revenues of the tourism industry are about 243 million USD, contributing 12% of the province's GDP. In 2015, about 4.1 million tourists, of which one fourth were foreigners, visited Khanh Hoa.[3]

[1] Khanh Hoa Department of Capture Fisheries and Resources Protection (DECAFIREP), 2015.
[2] Khanh Hoa Deparment of Statistics.
[3] Khanh Hoa Department of Culture, Sports and Tourism.

Nha Trang Bay is one of the most famous bays in Vietnam, where the first Vietnamese MPA was established in 2002. The MPA had a multi-purpose, focusing on biodiversity conservation, livelihood improvement for local residents (i.e. fishermen) in partnership with other stakeholders (Thu et al. 2005). NTB MPA has a total area of 160 km^2, encompassing nine islands and surrounding waters. It has been shown to have the highest marine biodiversity in Vietnamese coastal waters and also relatively high for the Pacific Ocean overall, including a multitude of different habitats (i.e. coral reefs, seagrass beds, mangroves, sand-muddy areas, and rocky shores) (Tuan et al. 2002). Moreover, this marine area is considered a major spawning and nursery ground supporting fish larvae to other Vietnamese and possibly Cambodian waters (Dung 2009). NTB MPA is a three-zone MPA, comprising a 16 km^2 core zone (red color in Figure 1, including five islands: Mun, Rom, Noc, Vung and Cau), protected from fishing and other activities, except for tourism; a buffer zone (yellow color) allowing traditional fishing gear, marine aquaculture and tourism, but no trawling; and a transition zone (light blue color) open to all activities, though limiting bottom trawl with regards to mesh size and engine power (see Figure 1).[4]

[4] On 9/12/2014, the Khanh Hoa government issued some new regulations for the NTB MPA. Firstly, The NTB MPA core zone now includes a part of eastern and the southeastern of Hon Tre, which has been named a strictly protected area; the buffer zone is also renamed as the ecological rehabilitation zone. Secondly, fishing is not allowed in either area. However, in this paper we use the collected data before 2011 so we still keep the former NTB MPA scheme for our analysis.

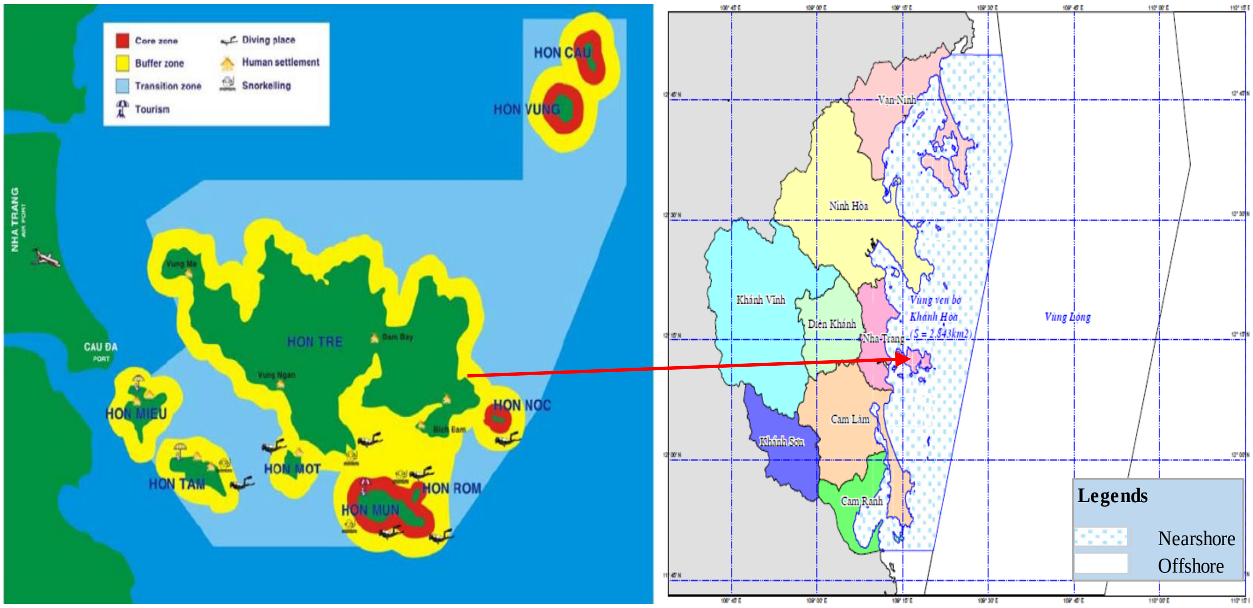

Figure 1. The zoning scheme of NTB MPA in 2001 (Tuan et al., 2005) and the Map of Khanh Hoa, Vietnam.

The NTB MPA is not only a place for protecting marine biodiversity in general and the exploited stocks from fishing in particular, it is also one of the most popular destinations for tourists visiting Khanh Hoa province, where they can enjoy tourism activities such as diving, snorkeling, swimming, water sports, etc. The number of tourists visiting the NTB MPA has been increasing from 30 thousand people in 1995 to more than 600 thousand people in 2015.[5] The aim of this study is to evaluate the benefits provided by the NTB MPA for the both fisheries and tourism industries in Khanh Hoa province.

Because of the complexity of multi-species fisheries and limited data availability, we concentrate on the anchovy purse seine fishery which is open access outside the MPA core zone. Anchovies are among the most traded fish species in the world. They are a small, schooling pelagic fish belonging to the *Engraulidae* (Mediterranean and European) and *Anchoa* (North America) family (FAO 2012). Two of five commercial anchovy species (*Encrasicolus* and *Stolephorus*) are found

[5] Source: NTB MPA Authority, 2015

in Vietnam (Thi et al. 2007). In Khanh Hoa province, anchovy is one of the most important inshore fisheries and is mainly fished by anchovy purse seiners (Thuy & Flaaten 2013). The annual operating time is about 8-10 months, divided into two seasons: high season is from February to October and low season is the remaining months (Thuy & Flaaten 2013). Anchovies are distributed mainly in shallow coastal water and near islands (Thi et al. 2007). The popular fishing grounds of anchovy purse seiners are around Nha Trang Bay and Cam Ranh Bay (Kim Anh et al. 2007).

Despite the fact that anchovies are highly migratory they have preferred grounds for spawning and nursing situated in coastal areas, around islands and in bays. The NTB MPA is suggested to be one of the most important spawning and nursery grounds of Khanh Hoa waters. A few studies on egg and larvae distribution in the coastal waters of Khanh Hoa province show that fish eggs and larvae are present at all sample stations, which are located around islands within the bays with favorable habitats for fish (i.e. coral reefs, seagrass beds, etc.) (Viet et al. 2014; Quang 2008; Phung et al. 2002). These studies also show that the sample stations within the NTB MPA have high relative density of fish eggs and larvae, of which anchovies make up the major component.

3. Model specification

3.1. Bioeconomic model

Though the NTB MPA is a three-zone MPA, the buffer zone and transition zone allow open access fisheries with the exception of trawling. Hence, in the case of the purse seine anchovy fishery we can model the area of Khanh Hoa waters as a two-patch system, in which one patch (i.e., the core zone) is protected from fishing, denoted the reserve zone (RZ). The remaining MPA areas (i.e. the buffer and transition zones) and the adjacent waters belonging to Khanh Hoa province that are open to fishing, are denoted as a harvest zone (HZ). Tourism activities are allowed in the entire area.

The model is based on Conrad's (1999) bioeconomic marine reserve model. Fish population is assumed uniformly distributed over the whole area. K is carrying capacity of the entire area. A fraction m, $0<m<1$, of the entire area denotes the share of RZ, and $(1-m)$ is the fraction of HZ, making the carrying capacities in the RZ and HZ equal to mK and $(1-m)K$, respectively. The dispersal parameter, which is fish movement between the two areas, is denoted γ ($\gamma > 0$). In this study, it is assumed that expansion of the RZ increases the carrying capacity of the fish stock within the RZ due to increased nutrient supply, spawning and nursery grounds, as well as refuge from predators (Foley et al. 2012).

We assume that a resource manager wants to include both extractive and non-extractive activities for maximizing the total benefits of both activities. Therefore, the basic net present value function of the fisheries is extended by adding tourism value, $V(m)$, which is dependent on the size of the RZ and is an increasing concave function in m, i.e. $V'(m) > 0$ and $V''(m) < 0$. The net present value function is described as follow: [6]

$$\max_{\langle Y,m \rangle} NPV = \int_0^{\infty} \left\{ \left(p - \frac{c}{qX_1} \right) Y + V(m) \right\} e^{-\delta t} dt \qquad (1)$$

subject to:

$$\dot{X}_1 = rX_1 \left(1 - \frac{X_1}{(1-m)K} \right) + \gamma \left(\frac{X_2}{mK} - \frac{X_1}{(1-m)K} \right) - Y$$

$$= F_1(X_1, m) + M(X_1, X_2, m) - Y \qquad (2)$$

$$\dot{X}_2 = rX_2 \left(1 - \frac{X_2}{mK} \right) - \gamma \left(\frac{X_2}{mK} - \frac{X_1}{(1-m)K} \right)$$

[6] Tourism development in the MPA contributes to the increase in use value of the MPA. This, however, can have direct or indirect negative impacts on marine species (e.g. sessile invertebrates) and habitats (i.e. seagrass beds, macroflora), when there is intensive and unregulated tourism development in MPAs (see Milazzo et al. 2002 for more discussion). If this is the case, a function of environmental damage could be included in the model to give a broader picture. However, in this study, the empirical data applied in the model comes from the NTB MPA, which we will assume has necessary solutions and regulations to limit the negative effects caused by tourism within the MPA, as indicated by Van (2013). We therefore do not include the environmental damage function in the model.

$$= F_2(X_2, m) - M(X_1, X_2, m) \quad (3)$$

where, $F_1(X_1, m)$ and $F_2(X_2, m)$ are the natural growth of the fish stock within the HZ and RZ respectively; $M(X_1, X_2, m)$ is the net migration term; X_1 is the fish stock size within the HZ; X_2 is the fish stock in the RZ; Y denotes harvest, being the standard Schaefer harvest function: $Y = qEX_1$; c is the cost per unit effort; q is catchability; and p is the price of fish, assumed to be constant.[7]

The current-value Hamiltonian for this problem may be expressed as follows:

$$H^c = \left(p - \frac{c}{qX_1}\right)Y + V(m) + \lambda[F_1(X_1, m) + M(X_1, X_2, m) - Y]$$
$$+\beta[F_2(X_2, m) - M(X_1, X_2, m)] \quad (4)$$

With λ and β being the adjoint variables measuring the shadow prices of the associated state variables X_1 and X_2. The current-value Hamiltonian is linear in the control variable, harvest, and strictly concave in the control variable, MPA size, as well as the state variables, fish stock in the HZ and RZ. This ensures that the necessary and sufficient conditions are satisfied making the solution unique.

3.2. Non-extractive value V(m)

The estimation of tourism value of the NTB MPA is based on data from a DCE survey in 2015 that was conducted using a convenience sample of 150 national tourists visiting the MPA. The survey aimed at valuing the Vietnamese tourists' willingness to pay (WTP) for an expansion of the NTB MPA core zone. As of today, the core zone of the NTB MPA is 16 km^2 and the question raised is whether a larger core zone should be implemented.

3.2.1. Survey design

[7] The constant price of fish may be a strong assumption. However, it is a reasonable assumption in this case since anchovy catch is part of a large world market, where Vietnam's share is small, therefore not impacting price, but where Vietnamese fishers instead have to accept exogenous market prices.

Based on focus group discussions and the scientific literature (Dung 2009; Phu et al. 2013; Tuan et al. 2005), four attributes were chosen to describe the good (core zone expansion) to be valued, including: 1) live hard coral cover, 2) environmental quality represented by visible waste and seascape disturbance, 3) fishermen's job losses, 4) the costs of further protection expressed as the increased boat trip ticket price. Coral and cost are continuous variables, while environment and job loss are categorical variables using dummy code (see Table 1). Though the good valued is a hypothetical core zone expansion of the NTB MPA, we chose not to include the MPA core zone size as an attribute, because of causality. That is, the increase in coral cover, environmental quality, and fishermen's job losses can be seen as a result of an increase in the MPA core zone size. Hence, inclusion of the core zone size attribute may encourage respondents to try to understand the causal relations among attributes and potentially to simplify their decision making process, resulting in a reduction in marginal WTP for the other attributes (Bennett & Blamey 2001).

Each choice situation consists of a status quo of keeping the current state (SQ) and two alternatives with increased MPA core zones. An example choice card is presented in Figure 2. The combination of attribute levels on the choice cards was done by applying a D-efficiency design (Scarpa & Rose 2008). Twelve choice cards were produced and blocked randomly into two versions for the survey, and hence each respondent faces 6 choice cards.

Table 1. Attributes and levels used for the DCE.

Attribute	Coral cover	Environmental quality	Job loss	Cost
Variable name	Coral	Environment	Job loss	Cost

Description	The average coral cover within the MPA (%)	Visible waste and floating bottles/cages within the MPA	The number of lost jobs for fishermen	Increase in ticket price of sea/islands tour (1000 VND)[8]
Status quo (SQ)	13	Low	0	0
Level 1	20	Low (base level)	0 (base level)	20
Level 2	30	Medium (med.env)	50 (small.loss)	50
Level 3		High (high.env)	100 (med.loss)	100
Level 4			200 (large.loss)	200

Attribute	**Status quo**	**Plan A**	**Plan B**
Coral cover	13% 	30% 	20% 
Environmental quality	Low 	Low 	High 
Fishermen job loss	0	50	200
Cost (1000 VND)	0	20	50
I prefer	☐	☐	☐

Figure 2. An example of choice card.

3.2.2. Model specification

[8] VND is Vietnamese Dong. In 2015, the exchange currency was 1USD = 22,547 VND (The State Bank of Vietnam, www.sbv.gov.vn/).

The analysis of discrete choice data is based on the random utility maximization theory, where we assume that a person's utility from choosing a specific alternative is described by one systematic and one random component. The systematic component is a function of observed variables while the random component includes all unobserved variables. A random utility model for the chosen alternative can be expressed as;

$$U_{nit} = \beta_n x_{nit} + \varepsilon_{nit} \qquad (5)$$

where, U_{nit} is the utility of individual *n* obtained from choosing alternative *i* in choice situation *t*, x_{nit} is a vector of observed attributes of alternative i, β_n is a vector of attributes parameters, and ε_{nit} is a random component of the utility of alternative *i* and is assumed to be independent and identically distributed (iid) following a type 1 extreme value distribution.

Because of the random component, the preferences cannot be predicted with certainty, leading to the probability of a chosen alternative *i* being expressed in terms of a logistic distribution. In this study, we use the mixed logit model (MXL), which accounts for random taste variation among the individuals, to estimate equation (5). The mixed logit probability function of the sequence of choices made by a respondent is the integral over the product of the logit formula for all possible values of β.

$$\Pr(i_n|x_n) = \int \prod_{t=1}^{T} \frac{\exp(\mu\beta_n' x_{nit})}{\sum_{j\in C} \exp\left(\mu\beta_n' x_{njt}\right)} f(\beta) d\beta \qquad (6)$$

where μ is a scale parameter that is typically set equal to one and is inversely proportional to the standard deviation of the error terms. The integral does not have a closed-form solution and is approximated through simulation. Aggregating over all respondents yields the likelihood function, and for ease of estimation the parameters are estimated by maximizing the log of the likelihood function (Train 2009).

The results from the MXL model estimation can be used to derive the amount of money individuals are willing to pay (or consumer surplus) in order to get some benefits from an implemented project or new policy (Train 2009).

$$CS_n = -\frac{1}{\beta_c} \ln \left(\sum_{j \in C} \exp(V_{njt}) \right) \tag{7}$$

where CS_n is the consumer surplus of individual n, and β_c is the estimate for the cost attribute.

3.2.3. Results

It is possible that tourists' WTP for coral cover may be influenced by their preferences for environmental quality and fishermen job loss, we therefore include the interaction effects between coral cover and environmental and job loss variables in the model to estimate the tourism value of different MPA management policies. We let the parameters of non-cost variables follow normal distributions and the parameter of cost variable be fixed. The MXL model is estimated in R using the "gmnl"-package (Sarrias & Daziano 2016) and 1000 standard Halton draws. The estimated results are shown in Table 2.

Table 2 Model estimate results

Attributes	Mean (standard error)	Standard deviation (standard error)
Cost	-0.00624(0.00182)***	
Coral	0.05709(0.01815)**	0.11103(0.01557)***
Coral*med.env	0.05355(0.00754)***	0.03408(0.00907)***
Coral*high.env	0.11205(0.01708)***	0.06082(0.01456)***
Coral*small.loss	-0.00616(0.01205)	0.05734(0.02422)*
Coral*med.loss	0.00728(0.01115)	0.07828(0.01502)***

Coral*large.loss	-0.00414(0.00966)	0.05208(0.01367)***
Model characteristics		
Likelihood ratio index		0.142
Log-likelihood at convergence		-841.25
Number of observations		900
Number of parameters estimated		7
AIC		1 708.49

***, **, * indicate estimates significant at 0.1%, 1% and 5%, respectively

From the data in Table 2, we want to determine the consumer surplus (CS) per individual for different management scenarios. Because the estimated parameters of the interactive variables between coral and job loss are statistically insignificant, they are excluded when calculating CS for different management scenarios.

Because we do not have an evaluation of the expanded-core zone size for an increasing level of coral cover and environmental quality, we have to make assumptions regarding these relationships (denoted as management scenarios in Table 3). They are: (1) small RZ scenario, which is the current MPA core zone size of 16 km^2, corresponding to the current status with 13% coral cover and low environmental quality; (2) medium RZ scenario, where the MPA core zone size is almost doubled (e.g. 30 km^2) by including a part of the buffer zone, corresponding to 20% coral cover and medium environmental quality; and (3) large RZ scenario, in which the core zone is expanded to encompass the equivalent of a half of the NTB MPA (e.g., 80 km^2) by including the whole buffer zone, corresponding to 30% coral cover and high environmental quality.

The assumed relationship between MPA core zone size and coral cover are based on several NTB biological indicators. Before 1994, the average coral cover in the NTB was recorded to be 30%

(Ben et al. 2015), and was reduced to 13% on average by 2002, due to human activities (Tuan et al. 2002). The coral cover around Mun island, an important core zone in the NTB MPA, has increased by 50% after 4 years of protection (Tuan et al., 2005). Moreover, the distribution of coral reefs are mostly along the coast and around the islands within the NTB MPA (i.e. the core zone and buffer zone of NTB MPA) (Tuan et al. 2005). Hence, we may assume that if a part or all of the buffer zone of the NTB MPA is added to the core zone, corresponding to the second and the third management scenarios, and these areas are properly monitored and regulated, then the coral cover within the NTB MPA could be expected to increase after some years of protection.

Table 3. Consumer surplus in USD per individual resulting from the MXL model [9]

Management scenarios	Attributes		Mean *CS* (95% confident interval)
	Coral cover	Environment quality	
Small RZ (SQ) (16 km^2)	13%	Low	4.8 (2.7-7.4)
Medium RZ (30 km^2)	20%	Medium	14.9 (10.5-24.0)
Large RZ (80 km^2)	30%	High	31.6 (24.4-55.3)

The tourism value of the MPA can then be calculated as the consumer surplus for each assumed management scenario as shown in Table 3. Based on three point estimates of the *CS* per individual

[9] The exchange rate is 1USD equaling 20,828 VND and 22,547 VND in 2011 and in 2015, respectively (State Bank of Vietnam, 2016). The mean CS values are inflated back to 2011 using the consumer price index (CPI) reported by World Bank (2016).

for the three scenarios (i.e., small, medium and large RZs), we specify a non-linear tourism value function, which depends on the size of the RZ following the natural logarithmic functional form:

$$v(m) = b\ log(mA) + \theta \qquad (8)$$

where, b and θ are 16.7 and – 41.6, respectively (R^2=0.9998).[10] A is the total study area (inshore area of Khanh Hoa waters).[11] Taking the total number of national tourists visiting the NTB MPA in 2011, N=401,300, and multiplying by $v(m)$, we can derive the total tourism value $V(m)$ as shown in equation 1.[12]

Due to the uncertainty regarding the assumptions of the expanded RZ size we carry out sensitivity analysis of the estimated parameter, b. It can be shown that parameter b is insensitive to the change in RZ size.

4. The anchovy fishery data

The biological and economic parameter values of the anchovy fishery in the model mainly come from the results of the study carried out by Thuy & Flaaten (2013). They use the inshore anchovy purse seine fishery data in Khanh Hoa province to estimate the backward-bending harvest function. s based on four different models; a general supply function (i.e. harvest is a function of price), the Gordon-Schaefer, Ricker, and Gompertz-Fox models. They also determined the price (P_{MSY}) and yield (H_{MSY}) corresponding to the maximum sustainable yield level.

To find the carrying capacity, the estimated results based on the Gordon-Schaefer model in Thuy & Flaaten (2013) are applied, giving $H_{MSY} = rK/4$ = 142,000 tons. Inserting r = 0.53, the lowest intrinsic growth rate value reported by Thi et al (2007) for different anchovy species in southwest

[10] The value of the estimated parameter, b, indicates that when the size of protected area increases 1%, the average consumer surplus per individual increases by 0.167 USD.

[11] The study site is Khanh Hoa waters which includes the NTB MPA where the DCE survey took place. Hence the tourism value function will depend on the RZ size which is proportional to the total area of study, i.e. the Khanh Hoa inshore waters.

[12] Source of number of national tourists: NTB MPA Authority, 2015.

Vietnam[13], into this equation gives K = 1,071,698 tons. To determine the values of catchability, the equation $p_0=c/qK$ shown by Thuy & Flaaten (2013) is applied. Here p_0 = 80 USD/ton is the minimum price that fishermen will accept, derived from the emperical Gordon-Schaefer model estimation, and c = 59,134 USD/vessel/year is the average cost per vessel over the period. Inserting these two values and the K value into the function gives catchability q = 0.00069. The parameters, including their sources, are given in Table 4.

Table 4. Biological and economic parameters of the anchovy purse seine fishery and tourism sector.

Parameter	Unit	Measure	Source/explanation
δ		0.07	Average interest rate during last 5 years (State Bank of VN)
γ		100,000	Guesstimated
r	Year^{-1}	0.53	Thi et al. (2007)
K	Tons	1,071,698	Calculated from Thuy & Flaaten (2013)
p_0	USD/Ton	80	Thuy & Flaaten (2013)
p	USD/Ton	288	Thuy & Flaaten (2013)
q	Boat^{-1}	0.00069	Calculated from Thuy & Flaaten (2013)
c	USD/vessel/year	59,134	Thuy & Flaaten (2013)
b		16.7	Estimated from own valuation study data
A	Km^2	2843	Map of Khanh Hoa province
N	Person	401,300	NTB Authority, 2011

[13] Thi et al. (2007) report that the intrinsic growth rates of anchovy species in southeast Vietnamese are relatively high, ranging from 0.53 to 0.90 per year. We choose the most conservative measure.

5. Numerical Simulation and Discussions

We apply the software package Mathematica to determine an optimal solution for coexistence of both fishery and tourism. The results are found to be locally asymptotically stable and are reported in Table 5.

Table 5. The optimal estimated variables from equation (1).

X_1^*	X_2^*	X^*	Y^*	m^*	NPV^*	NPV_f^*	NPV_v^*
545,085 [a]	175,653 [a]	720,738 [a]	122,295 [a]	0.20	598 [b]	228 [b]	370 [b]

Note: [a, b] measured in tons and million USD, respectively.

NPV_f^* and NPV_v^* denote the net present values from the anchovy fishery and tourism, respectively, and NPV^* is the total net present value.

The results suggest that to maximize the net present value of both fishery and tourism activities, the optimal size of RZ needed is 20% (i.e. approximately 568 km^2) of the nearshore waters of Khanh Hoa province. This is much larger than the present NTB MPA core zone (16 km^2), and indeed larger than the whole NTB MPA (160 km^2). It should however be noted that the current NTB MPA was established based on the characteristics of the NTB with its' 507 km^2 water area. In other words, the current MPA core zone is 3.2% of the NTB area and 0.56% of Khanh Hoa waters, which is believed to be too narrow in order to ensure biodiversity restoration and prevent marine environmental pollution, even within the NTB solely (Dung 2009). Therefore, it is possible to imagine the development of a network of no-take zones within Khanh Hoa waters for the purpose of biodiversity conservation, sustainable resource use and tourism development.

Particularly, the expansion of the RZ within the NTB area can be a small proportion of the suggested optimal RZ size, e.g. 20% of NTB area or approximately an increased area of 100 km^2. The remainder of RZ increase can be located in different parts of Khanh Hoa waters, i.e. Van

Phong and Cam Ranh Bays, where there exist similar characteristics regarding ecosystems, biodiversity, and tourism attractiveness as in NTB (Latypov & Selin 2012; Son et al. 2008; Long et al. 2014; Phung et al. 2002; Quang 2008).

Thuy & Flaaten (2013)indicate that biological overfishing has taken place in the open access anchovy fishery in Khanh Hoa province. In this study, we find that the open access anchovy harvest in Khanh Hoa province is beyond the optimal harvest level, which here has been estimated to be 122,295 tons, implying that the fishing effort should be reduced by 44% compared to the level in year 2011, in order to reach the optimal yield.[14]

The results also show that the net present value of the tourism sector is greater than that of the anchovy fishery. This result does however require some caveats. Firstly, the tourism value is shown to highly depend on both number of tourists visiting the NTB MPA, *N*, and the slope parameter, *b*, of the tourism value function (see Table 6). The tourism demand may vary due to the impacts of the increase in boat trip ticket price and the improvement in marine environmental quality within the MPA. As we do not have any data on price- and environment-elasticities for this type of service, we are not able to estimate changes in tourism demand due to a change in MPA management policies. Moreover, the DCE survey for tourists is conducted only in the NTB MPA, while the study site is the whole of Khanh Hoa waters where there exists alternative tourism sites, which may need to be protected as suggested in this study. Hence, if the DCE survey had been conducted in these areas, the number of tourists willing to pay for the potential MPAs may have increased. The value of the estimated slope parameter, *b*, is negatively affected by the assumed RZ sizes and positively by the estimated WTPs values. The latter is likely to be upwards biased due to the hypothetical nature of the payment mechanism used in the DCE method (Birol et al., 2006),

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[14] The fishing effort in 2011 reported by Thuy & Flaaten (2013) was 581 vessels, while the optimal fishing effort calculated from the estimations of this study is $E^*=Y^*/(qX_1^*)$=122,259/(0.00069*545,085)=325 vessels.

and thus it is often suggested as a reference for upper bound values of the tourism benefit generated by the MPA.

Secondly, it should be noted that only the anchovy fishery is taken into account in this study, instead of the whole multi-species complex which is representative of Khanh Hoa fisheries. Therefore, if these expressed elements are taken into account, the magnitude of the optimal variables may well be changed and hence the net present values of both fisheries and tourism sectors.

Some studies indicate that the non-extractive values (i.e. tourism) or non-use value of MPAs may be much larger than the extractive values (i.e. fisheries) (Bulte et al. 1998; Merino et al. 2009). If this is the case in this study, there may be a corner solution requiring that the whole studied area should be protected and dedicated to tourism development. In this simulation, the corner solution occurs when either the value of the *b* parameter is greater than 55 or the number of tourists, *N*, is greater than 1,320,000, *ceteris paribus*. Concerning the case of no reserve ($m = 0$), then the ocean area is open access for both fisheries and tourism activities, presenting potential conflicts between the groups regarding resource use (see Lee & Iwasa 2011 for more discussion).

A sensitivity analysis for all parameters with direct impacts on the outputs is carried out in order to study the effects of small changes in value of each parameter on the values of the optimal variables in the model (see Table 6). These changes are presented as elasticities, or the ratio of percentage change in the values of output variables to 10% change in the parameter values in the neighborhood of the initial values.

Table 6. Sensitivity analysis. Sensitive results are marked in bold and negative numbers in parentheses.

10% increase in	% change in optimal values						
	X_1^*	X_2^*	Y^*	m^*	NPV^*	NPV_f^*	NPV_v^*
δ	(0.46)	(0.17)	0.38	0.04	**(11.07)**	**(11.19)**	**(10.99)**
r	2.03	(9.74)	**10.70**	(9.79)	3.91	**12.85**	(2.48)
K	8.40	(0.22)	**13.23**	(9.49)	8.04	**21.37**	(2.41)
γ	(0.13)	(0.49)	0.48	0.72	0.24	0.32	0.19
c	3.41	2.14	(3.51)	0.47	(4.64)	**(13.28)**	0.12
p	(1.93)	**(11.59)**	3.96	(8.44)	8.27	**21.57**	(2.15)
q	(3.46)	(1.36)	2.62	0.16	4.12	**10.11**	0.04
b	(1.64)	9.58	(1.85)	8.32	**10.23**	(3.89)	**17.32**
A	-	-	-	-	1.65	-	2.65
N	(1.64)	9.58	(1.85)	8.32	6.48	(3.89)	**12.00**

The sensitivity analysis shows that the optimal RZ size and stock size in HZ are robust to the changes in all parameter values, while the stock size in RZ is sensitive to change in the fish price. The harvest is highly dependent on the intrinsic growth rate and carrying capacity. As could be expected, the change in discount factor has a significant effect on the magnitude of all kinds of discounted profits. A change in most fisheries biological and economic parameter values has significant effects on the net present value of the fishery, while the net present value of tourism is sensitive to the changes in tourism parameter values. However, the total net present value is robust to a change in all parameter values, except for *b*. Interestingly, the model is robust to the most uncertain parameter: the dispersal of fish.

6. Conclusions

This paper models the coexistence of two activities, fisheries and tourism, generated by an MPA, by integrating a fishery bioeconomic model with an estimation of tourism values of the MPA. Data from the anchovy purse seine fishery in Khanh Hoa province and a DCE survey of national tourists visiting the NTB MPA are applied in order to illustrate the optimal management for multiple services provided by the MPA.

The results suggest an expansion of RZ size for an optimization of the net benefits from the sum of fishing and tourism activities. The RZ size expansion, however, may increase economic conflicts between fishers and tourists due to short-term impacts of RZ expansion on local fishers. For instance, some fishers may lose their jobs due to unavailable fishing grounds, their catches may decline due to smaller areas for fishing, and their fishing costs may increase as a result of having to travel further for fishing. On the other hand, they can gain more benefits in the long-term due to positive spillover effects from the MPA to nearby fishing areas, as suggested in the literature (Sanchirico et al., 2005; Sanchirico & Wilen, 2001).

To soften the negative short-term impacts of the RZ expansion policies on local fishers, the increase in consumer surplus of tourists could be used partly to compensate for the losses of fishers, using a “payment for environmental services” scheme as suggested by Schuhmann et al. (2013). Local fishers can also receive direct and indirect financial support for alternative income generation, i.e. attending for free different courses that provide knowledge and skills for new occupations such as handicrafts, animal husbandry, tourism service and trading; they could also potentially borrow money from financial support programs connected to alternative income generation (Thu et al. 2005).

The fisheries bioeconomic model used in this study has been shown to disfavor MPAs in economic terms, i.e. MPA implementation may be good for conservation policies but gives less benefits to fishers than conventional management tools (Conrad 1999; Flaaten & Mjølhus 2010; Hannesson 1998). In this work, another positive effect of MPAs (i.e. non-extractive values) is added to the model, giving the possibility to illustrate multiple benefits of MPAs. The non-use and alternative values of MPAs are beyond the scope of this study and hence not included in the model. Despite this, the framework presented in this study allows for these values to be integrated in the model for an even broader estimation of multiple benefits provided by MPAs, and hence the optimal size of MPA for multiple values could be determined.

This study does however have some limitations; the DCE survey was only carried out in the NTB MPA while the fishery and the total area analysed encompasses the whole of Khanh Hoa waters. Although benefits transfer of the NTB MPA values can be applied for other locations within Khanh Hoa waters, it still may not reflect fully the tourism values generated by potential protected areas in the total area. This indicates the need to conduct more surveys for tourists visiting outside the NTB to obtain a comprehensive overview and provide more complete information for policy makers. Furthermore, the fisheries of Khanh Hoa are multi-species, multi-fleet and multi-gear, future research should take into account data from other important species than solely anchovies for a proper assessment of extractive values impacted by the MPA.

Though conflicts amongst multiple goals of MPA implementation, such as between recreation and fisheries or conservation, are real, they are not explicitly taken into account in this study. Milazzo et al. (2002) indicate that the intensive and unregulated tourism development in MPAs is causing severe threats to marine organisms and habitats at the local scale. This affects directly the effectiveness of MPAs as regards biodiversity conservation and hence indirectly fisheries. In addition, under open access fisheries as is currently the case in NTB, tourism development in general may put more pressure on harvest both directly (e.g. tourists as anglers) and indirectly (e.g.

through fish consumption), and hence reduce the fish stock at the local scale. Still, these effects of tourism are most probably much smaller than the effects fisheries have on the fish stocks.

The negative impacts of tourism on conservation and fisheries can however be solved as follows. Controlled harvest policies outside the MPA can help secure a sustainable level of the resource. Some tourism regulations (e.g. entrance fees and quotas) may limit the number of tourists visiting the place, so it does not exceed a threshold of environmental damage. Other strategies, such as education and training, can also be implemented tomanage tourism impacts on marine environments. Nevertheless, incorporating into the model the broader economic conflicts amongst stakeholders is of interest for future research.

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